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Toxicological Review of Benzo[a]pyrene

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

November 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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CONTENTS

TABLES

FIGURES

ABBREVIATIONS

1-OH-Py	1-hydroxypyrene
AchE	acetylcholine esterase
ADAF	age-dependent adjustment factor
Ah	aryl hydrocarbon
AHH	aryl hydrocarbon hydroxylase
AhR	aryl hydrocarbon receptor
AIC	Akaike's Information Criterion
AKR	aldo-keto reductase
AMI	acute myocardial infarction
ANOVA	analysis of variance
ARNT	Ah receptor nuclear translocator
AST	aspartate transaminase
ATSDR	Agency for Toxic Substances and Disease Registry
BMC	benchmark concentration
BMCL	benchmark concentration lower confidence limit
BMD	benchmark dose
BMDL	benchmark dose, 95% lower bound
BMDS	Benchmark Dose Software
BMR	benchmark response
BPDE	benzo[a]pyrene-7,8-diol-9,10-epoxide
BPQ	benzo[a]pyrene semiquinone
BrdU	bromodeoxyuridine
BSM	benzene-soluble matter
BUN	blood urea nitrogen
BW	body weight
CA	chromosomal aberration
CalEPA	California Environmental Protection Agency
CASRN	Chemical Abstracts Service Registry Number
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CHO	Chinese hamster ovary
CI	confidence interval
CYP	cytochrome
CYP450	cytochrome P450
DAF	dosimetric adjustment factor
dbcAMP	dibutyl cyclic adenosine monophosphate
DMSO	dimethyl sulfoxide
DNA	deoxyribonucleic acid
EC	European Commission
EH	epoxide hydrolase
ELISA	enzyme-linked immunosorbent assay
EPA	Environmental Protection Agency
EROD	7-ethoxyresorufin-O-deethylase
ETS	environmental tobacco smoke
EU	European Union
Fe_2O_3	ferrous oxide
FSH	follicle stimulating hormone
GABA	gamma-aminobutyric acid
GD	gestational day
GI	gastrointestinal
GJIC	gap junctional intercellular communication
GSH	reduced glutathione
GST	glutathione-S-transferase
GSTM1	glutathione-S-transferase M1
hCG	human chorionic gonadotropin
HEC	human equivalent concentration
HED	human equivalent dose
HERO	Health and Environmental Research Online
HFC	high-frequency cell
HPLC	high-performance liquid chromatography
hprt	hypoxanthine guanine phosphoribosyl transferase
HR	hazard ratio
Hsp90	heat shock protein 90
i.p.	intraperitoneal
i.v.	intravenous
Ig	immunoglobulin
IHD	ischemic heart disease
IRIS	Integrated Risk Information System
LDH	lactate dehydrogenase
LH	luteinizing hormone
LOAEL	lowest-observed-adverse-effect level
MAP	mitogen-activated protein
MCL	Maximum Contaminant Level
MCLG	Maximum Contaminant Level Goal
MIAME	Minimum Information About a Microarray Experiment
MLE	maximum likelihood estimate
MMAD	mass median aerodynamic diameter
MN	micronucleus
MPPD	Multi-Path Particle Deposition
mRNA	messenger ribonucleic acid
MS	mass spectrometry
NCE	normochromatic erythrocyte
NCEA	National Center for Environmental Assessment
NIOSH	National Institute for Occupational Safety and Health
NK	natural-killer

NMDA	N-methyl-D-aspartate
NOAEL	no-observed-adverse-effect level
NPL	National Priorities List
NQO	NADPH:quinone oxidoreductase
NRC	National Research Council
NTP	National Toxicology Program
OECD	Organisation for Economic Co-operation and Development
OR	odds ratio
ORD	Office of Research and Development
PAH	polycyclic aromatic hydrocarbon
PBMC	peripheral blood mononuclear cell
PBPK	physiologically based pharmacokinetic
PCA	Principal Components Analysis
PCE	polychromatic erythrocyte
PCNA	proliferating cell nuclear antigen
PND	postnatal day
POD	point of departure
PUVA	psoralen plus ultraviolet-A
RBC	red blood cell
$RDDR_{ER}$	regional deposited dose ratio for extrarespiratory effects
RfC	inhalation reference concentration
RfD	oral reference dose
RNA	ribonucleic acid
ROS	reactive oxygen species
RR	relative risk
s.c.	subcutaneous
SCC	squamous cell carcinoma
SCE	sister chromatid exchange
SCSA	sperm chromatin structure assay
SD	standard deviation
SE	standard error
SEM	standard error of the mean
SHE	Syrian hamster embryo
SIR	standardized incidence ratio
SMR	standardized mortality ratio
SOAR	Systematic Omics Analysis Review
SOD	superoxide dismutase
SRBC	sheep red blood cells
SSB	single-strand break
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TK	thymidine kinase
ToxR	Toxicological Reliability Assessment
TPA	12-O-tetradecanoylphorbol-13-acetate
TUNEL	terminal deoxynucleotidyl transferase dUTP nick end labeling
TWA	time-weighted average
UCL	upper confidence limit
UDP-UGT	uridine diphosphate-glucuronosyltransferase
UDS	unscheduled DNA synthesis
UF	uncertainty factor
UF_A	interspecies uncertainty factor
UF_D	database deficiencies uncertainty factor
UF_H	intraspecies uncertainty factor
UF_L	LOAEL-to-NOAEL uncertainty factor
UF_S	subchronic-to-chronic uncertainty factor
UVA	ultraviolet-A
UVB	ultraviolet-B
WBC	white blood cell
WESPOC	water escape pole climbing
WT	wild type
WTC	World Trade Center
XPA	xeroderma pigmentosum group A

AUTHORS | CONTRIBUTORS | REVIEWERS

Assessment Team

Christine Cai, MS Glinda Cooper, Ph.D. (formerly with EPA) Louis D'Amico, Ph.D. Jason Fritz, Ph.D. Martin Gehlhaus, MHS Catherine Gibbons, Ph.D. Karen Hogan, MS Andrew Kraft, Ph.D. Kathleen Newhouse, MS (Assessment Manager) Linda Phillips, Ph.D. Margaret Pratt, Ph.D. Keith Salazar, Ph.D. John Schaum, MS (retired) Suryanarayana Vulimiri, DVM Gene Ching-Hung Hsu, Ph.D. (formerly with EPA)	U.S. EPA Office of Research and Development National Center for Environmental Assessment Washington, DC
Lyle Burgoon, Ph.D. John Cowden, Ph.D. Amanda Persad, Ph.D. John Stanek, Ph.D.	U.S. EPA Office of Research and Development National Center for Environmental Assessment Research Triangle Park, NC
Chris Brinkerhoff, Ph.D. Emma McConnell, MS	Oak Ridge Institute for Science and Education Fellow
Scott Glaberman, Ph.D.	American Association for the Advancement of Science Fellow

Contributors

Lynn Flowers, Ph.D. John Fox, Ph.D. Paul White, Ph.D. Maria Spassova, Ph.D.	U.S. EPA Office of Research and Development National Center for Environmental Assessment Washington, DC

Production Team

Taukecha Cunningham Maureen Johnson Terri Konoza Vicki Soto	U.S. EPA Office of Research and Development National Center for Environmental Assessment Washington, DC

Contractor Support

Heather Carlson-Lynch, S.M. Peter McClure, Ph.D. Megan Riccardi Kelly Salinas Joe Santodonato Julie Melia, Ph.D.	SRC, Inc., Syracuse, NY
George Holdsworth, Ph.D. Lutz W. Weber, Ph.D.	Oak Ridge Institute for Science and Education, Oak Ridge, TN
Janusz Z. Byczkowski, Ph.D., D.Sc.	JZB Consulting, Fairborn, OH

Executive Direction

Kenneth Olden, Ph.D., Sc.D., L.H.D. (Center Director-Retired) Michael Slimak, Ph.D., (Acting Center Director) Lynn Flowers, Ph.D. (Associate Director for Health) Vincent Cogliano, Ph.D. (IRIS Program Director) Samantha Jones, Ph.D. (IRIS Associate Director for Science) Jamie B. Strong, Ph.D. (Toxic Effects Branch Chief-now at Office of Water) Ravi Subramaniam (Acting Toxic Effects Branch Chief)	U.S. EPA/ORD/NCEA Washington, DC

Internal Reviewers

Stephen Nesnow, Ph.D. (retired)	U.S. EPA National Health and Environmental Effects Research Laboratory Research Triangle Park, NC
Rita Schoeny, Ph.D. (retired)	U.S. EPA Office of Water Washington, DC

Reviewers

This assessment was provided for review to scientists in EPA's Program and Region Offices. Comments were submitted by:

Office of Children's Health Protection, Washington DC
Office of Policy, Washington, DC
Office of Solid Waste and Emergency Response, Washington DC
Office of Water, Washington DC
Region 2, New York, NY
Region 3, Philadelphia, PA
Region 8, Denver, CO

This assessment was provided for review to other federal agencies and the Executive Office of the President (EOP). A summary and EPA's disposition of major comments from the other federal agencies and EOP is available on the IRIS website. Comments were submitted by:

Agency for Toxic Substances and Disease Registry, Centers for Disease Control and Prevention, Department of Health and Human Services
Department of Defense
National Aeronautics and Space Administration
National Institute for Occupational Safety and Health, Centers for Disease Control, Department of Health and Human Services
Office of Management and Budget, Executive Office of the President
White House Council on Environmental Quality, Executive Office of the President

This assessment was released for public comment on August 21, 2013 and comments were due on November 21, 2013. A summary and EPA's disposition of the comments received from the public is included in Appendix G of the Supplemental Information to the Revised External Review draft of the Toxicological Review. Comments were received from the following entities:

Utility Solid Waste Activities Group
CDM Smith Inc, Pavement Coating Technology Council
Electric Power Research Institute
Duke Energy
CH2M Hill
Gradient
American Chemistry Council
Arcadis on behalf of: American Coke and Coal Chemicals Institute, American Fuels and Petrochem American Petroleum Institute, Asphalt Institute, Association of American Railroads, Beazer East, Inc. and Pavement Coatings Technology Council
Agnes Francisco
Melanie Nembhard

This assessment was peer reviewed by independent, expert scientists external to EPA convened by EPA's Science Advisory Board (SAB), the Chemical Assessment Advisory Committee Augmented for the IRIS Benzo[a]pyrene Assessment. A peer review meeting was held on April 15 to 17, 2015. The report of the SAB's review of EPA's Draft Toxicological Review of Benzo[a]pyrene, dated April 5, 2016, is available on the IRIS website. A summary and EPA's disposition of the comments received from the SAB is included in Appendix G.

Dr. Elaine M. Faustman (chair), Professor and Director, Institute for Risk Analysis and Risk Communication, School of Public Health, University of Washington, Seattle, WA
Dr. Scott Bartell, Associate Professor, Program in Public Health, University of California - Irvine, Irvine, CA
Dr. Ronald Baynes, Professor, Population Health & Pathobiology, College of Veterinary Medicine, North Carolina State University, Raleigh, NC
Dr. Annette Bunge, Professor Emeritus, Chemical & Biological Engineering, Colorado School of Mines, Golden, CO
Dr. Scott Burchiel, Distinguished Professor, Pharmaceutical Sciences, College of Pharmacy, University of New Mexico, Albuquerque, NM
Dr. Anna Choi, Research Scientist, Environmental Health, Harvard School of Public Health, Boston, MA
Dr. John DiGiovanni, Professor and Coulter R. Sublett Chair in Pharmacy, Division of Pharmacology and Toxicology and Department of Nutritional Sciences, Dell Pediatric Research Institute, The University of Texas at Austin, Austin, TX

Dr. Joanne English, Senior Toxicologist, Toxicology Services, NSF International, Ann Arbor, MI
Dr. William Michael Foster, Independent Consultant, Durham, NC
Dr. Chris Gennings, Professor, Department of Biostatistics, Icahn School of Medicine at Mount Sinai, New York, NY
Dr. Helen Goeden, Senior Toxicologist, Minnesota Department of Health, St. Paul, MN
Dr. Sean Hays, President, Summit Toxicology, Allenspark, CO
Dr. John Kissel, Ph.D., Department of Environmental and Occupational Health Sciences, Public Health, University of Washington, Seattle, WA
Dr. Edward Levin, Professor, Psychiatry, Duke University Medical Center, Durham, NC
Dr. Maureen Lichtveld, Professor and Chair, Global Environmental Health Sciences, School of Public Health and Tropical Medicine, Tulane University, New Orleans, LA
Dr. Abby A. Li, Senior Managing Scientist, Health Science Practice, Exponent Incorporated, San Francisco, CA,
Dr. Barry McIntyre, Senior Toxicologist, Toxicology Branch, National Toxicology Program, National Institute of Environmental Health Sciences, Research Triangle Park, NC
Dr. Bhagavatula Moorthy, Professor, Pediatrics, Baylor College of Medicine, Houston, TX
Dr. Miriam Poirier, Head, Carcinogen-DNA Interactions Section, National Cancer Institute, National Institutes of Health, NIH-NCI, Bethesda, MD
Dr. Kenneth M. Portier, Director of Statistics, Department of Statistics and Evaluation, American Cancer Society, Atlanta, GA
Dr. Kenneth Ramos, Associate Vice-President of Precision Health Sciences and Professor of Medicine, Arizona Health Sciences Center, University of Arizona, Tuscon, AZ
Dr. Stephen M. Roberts, Professor, Center for Environmental and Human Toxicology, University of Florida, Gainesville, FL
Dr. Richard Schlesinger, Associate Dean, Dyson College of Arts and Sciences, Pace University, New York, NY
Dr. Leslie T. Stayner, Director, Epidemiology & Biostatistics, Epidemiology & Biostatistics, School of Public Health, University of Illinois, Chicago, IL
Dr. Alan Stern, Chief, Bureau for Risk Analysis, Division of Science, Research and Environmental Health, New Jersey Department of Environmental Protection, Trenton, NJ
Dr. Charles Vorhees, Professor, Pediatrics, Division of Neurology, Cincinnati Children's Research Foundation/University of Cincinnati, Cincinnati, OH
Dr. Christi Walter, Professor and Chair, Cellular & Structural Biology, School of Medicine, University of Texas Health Science Center at San Antonio, San Antonio, TX
Designated Federal Officer: Dr. Diana Wong

PREFACE

This Toxicological Review, prepared under the auspices of the U.S. Environmental Protection Agency's (EPA's) Integrated Risk Information System (IRIS) program, critically reviews the publicly available studies on benzo[a]pyrene in order to identify potential adverse health effects and to characterize exposure-response relationships. Benzo[a]pyrene is found in the environment and in food. Benzo[a]pyrene occurs in conjunction with other structurally related chemical compounds known as polycyclic aromatic hydrocarbons (PAHs).[1] Benzo[a]pyrene is universally present in these mixtures and is routinely analyzed and detected in environmental media contaminated with PAH mixtures: thus it is often used as an indicator chemical to measure exposure to PAH mixtures (Boström et al., 2002). It also serves Boström et al. (2002) as an index chemical for deriving potency factors for PAH mixtures, such as in EPA's Relative Potency Factor Approach (U.S. EPA, 1993).

Benzo[a]pyrene is listed as a hazardous substance under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), is found at 524 hazardous waste sites on the National Priorities List (NPL) and is ranked number 8 out of 275 chemicals on the Priority List of Hazardous Substances for CERCLA (ATSDR, 2011). This ranking is based on a combination of factors that include the frequency of occurrence at NPL sites, the potential for human exposure, and the potential health hazard. Benzo[a]pyrene is also listed as a drinking water contaminant under the Safe Drinking Water Act and a Maximum Contaminant Level Goal (MCLG) and enforceable Maximum Contaminant Level (MCL) have been established[2]. It is also one of the chemicals included in EPA's Persistent Bioaccumulative and Toxic Chemical Program (http://www.epa.gov/pbt/pubs/benzo.htm). In air, benzo[a]pyrene is regulated as a component in a class of chemicals referred to as Polycyclic Organic Matter, defined as a Hazardous Air Pollutant by the 1990 amendments to the Clean Air Act.

This assessment updates IRIS assessment of benzo[a]pyrene that was developed in 1987. The previous assessment included a cancer descriptor and oral slope factor. New information has become available, and this assessment reviews information on all health effects by all exposure routes. Organ/system-specific reference values are calculated based on developmental (including developmental neurotoxicity), reproductive, and immune system 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.

[1]PAHs are a large class of chemical compounds formed during the incomplete combustion of organic matter. They consist of only carbon and hydrogen arranged in two or more fused rings.
[2]MCLG = 0, MCL = 0.0002 mg/L.

This assessment was conducted in accordance with EPA guidance, which is cited and summarized in the Preamble to Toxicological Reviews. Appendices for chemical and physical properties, toxicokinetic information, and summaries of toxicity studies are provided as *Supplemental Information* to this assessment.

In April 2011, the National Research Council made recommendations for improving the development of IRIS assessments. In May 2014, they reviewed the IRIS Program again and found that EPA had made substantial improvements to the IRIS Program in a short amount of time. As part of this review, the NRC evaluated the August 2013 public comment draft of benzo[a]pyrene to gauge EPA's progress in implementing the 2011 NRC recommendations. The NRC stated that "the new document structure, which is reflected in the toxicological review of benzo[a]pyrene, leads to better organized and streamlined assessments and reduces redundancies" and that "the draft assessment shows that the IRIS program has taken several additional steps toward addressing the recommendations in the 2011 NRC formaldehyde report."

This streamlined assessment uses tables, figures, and appendices to increase transparency and clarity. It has distinct sections for literature search and study selection, hazard identification, and dose-response assessment. A comprehensive, systematic literature search and screening approach is documented in a table (databases, keywords) and flow diagram (inclusion and exclusion of studies). All references were added to the Health and Environmental Research Online (HERO) database. Studies were evaluated uniformly for aspects of design, conduct, or reporting that could affect the interpretation of results and contribution to the synthesis of evidence. A summary of the evaluation is included in the section on methods for identifying and selecting studies. The evidence is presented in standardized summary and evidence tables and in exposure-response arrays.

In the hazard identification and dose-response sections, there are subsections for each target organ or system. The evidence is synthesized for each dataset, integrated for each target organ/system, and then integrated across different target organs/systems. The IRIS Program used existing guidelines to systematically approach the integration of human, animal, and mechanistic evidence. For each health outcome, the IRIS Program evaluated the consistency of a possible association, the strength of the association, the presence of a dose-response relationship, whether the exposure preceded the effect, and the biological plausibility of the response and its relevance to humans. Thus, this assessment provides a streamlined presentation of information, integrated hazard identification of all toxic effects, and reference values for each target organ or system. Additionally, this assessment contains an expanded discussion of the rationale for study selection and evaluation, as well as other key assessment decisions.

The IRIS program released this assessment for peer review in September 2014, prior to the program implementing systematic review. The approach to implementation of NRC recommendations is to use procedures and tools available at the time, without holding assessments until new methods become available. Accordingly, the IRIS program conducted literature searches

using tools and documentation standards then available. Problem formulation materials and protocol development began with assessments starting draft development in 2015, after this assessment was well into peer review. Implementation of systematic review is a process of continuous improvement subject to periodic review by the Chemical Assessment Advisory Committee of the EPA's Science Advisory Board. This assessment represents a step in the evolution of the IRIS program.

For additional information about this assessment or for general questions regarding IRIS, please contact EPA's IRIS Hotline at 202-566-1676 (phone), 202-566-1749 (fax), or hotline.iris@epa.gov.

Chemical Properties

Benzo[a]pyrene is a five-ring PAH. It is a pale yellow crystalline solid with a faint aromatic odor. It is relatively insoluble in water and has low volatility. Benzo[a]pyrene is released to the air from both natural and anthropogenic sources and removed from the atmosphere by photochemical oxidation; reaction with nitrogen oxides, hydroxy and hydroperoxy radicals, ozone, sulfur oxides, and peroxyacetyl nitrate; and wet and dry deposition to land or water. In air, benzo[a]pyrene is predominantly adsorbed to particulates but may also exist as a vapor at high temperatures (ATSDR, 1995).

Uses and Pathways of Exposure

There is no known commercial use for benzo[a]pyrene; it is only produced as a research chemical. Benzo[a]pyrene is ubiquitous in the environment primarily as a result of incomplete combustion emissions. It is found in fossil fuels, crude oils, shale oils, and coal tars (HSDB, 2012). It is released to the environment via both natural sources (such as forest fires) and anthropogenic sources including stoves/furnaces burning fossil fuels (especially wood and coal), motor vehicle exhaust, cigarettes, and various industrial combustion processes (ATSDR, 1995). Benzo[a]pyrene is also found in soot and coal tars. Several studies have reported that urban run-off from asphalt-paved car parks treated with coats of coal-tar emulsion seal could account for a large proportion of PAHs in many watersheds (Rowe and O'Connor, 2011; Van Metre and Mahler, 2010; Mahler et al., 2005). Benzo[a]pyrene exposure can also occur to workers involved in the production of aluminum, coke, graphite, and silicon carbide, and in coal tar distillation. The major sources of non-occupational exposure are tobacco products, inhalation of polluted air, ingestion of contaminated food and water, and through cooking processes that involve smoke (HSDB, 2012). Dermal exposure can occur through contact with materials containing soot, tar, or crude petroleum, including pharmaceutical products containing coal tar, such as coal tar-based shampoos and treatments for eczema and psoriasis (Cal/EPA, 2010; IARC, 2010).

Benzo[a]pyrene persists for a long period of time in the atmosphere in the particulate phase and is thus efficiently transported over long distances. It is lipophilic with low water solubility; therefore, once deposited in water or sediments, it adsorbs strongly to sediments and particulate

matter and degrades slowly over several years (Cal/EPA, 2010; GLC, 2007). Because of its presence in high concentrations in the waters and sediments of the Great Lakes and St. Lawrence river ecosystem, it is 1 of the 12 level I substances identified and targeted for reduction in the Great Lakes Region (GLC, 2007).

Most aquatic organisms metabolize benzo[a]pyrene, eliminating it in days, and thus, it is not expected to bioconcentrate in these organisms; however, several aquatic organisms such as plankton, oysters, and some fish cannot metabolize benzo[a]pyrene (U.S. EPA, 2010a). Thus, the data on benzo[a]pyrene bioconcentration in aquatic organisms varies from low to very high (HSDB, 2012). Biomagnification of benzo[a]pyrene in the food chain has not been reported (ATSDR, 1995). Additional information on benzo[a]pyrene exposure and chemical properties can be found in Appendix A.

Assessments by Other National and International Health Agencies

Toxicity information on benzo[a]pyrene has been evaluated by the World Health Organization, Health Canada, the International Agency for Research on Cancer, and the European Union. The results of these assessments are presented in Appendix B. It is important to recognize that these assessments were prepared at different times, for different purposes, using different guidelines and methods, and that newer studies have been included in the IRIS assessment.

PREAMBLE TO IRIS TOXICOLOGICAL REVIEWS

The Preamble summarizes the objectives and scope of the IRIS program, general principles and systematic review procedures used in developing IRIS assessments, and the overall development process and document structure.

1. Objectives and Scope of the IRIS Program

Soon after EPA was established in 1970, it was at the forefront of developing risk assessment as a science and applying it in support of actions to protect human health and the environment. EPA's IRIS program[2] contributes to this endeavor by reviewing epidemiologic and experimental studies of chemicals in the environment to identify adverse health effects and characterize exposure–response relationships. Health agencies worldwide use IRIS assessments, which are also a scientific resource for researchers and the public.

IRIS assessments cover the hazard identification and dose–response steps of risk assessment. Exposure assessment and risk characterization are outside the scope of IRIS assessments, as are political, economic, and technical aspects of risk management. An IRIS assessment may cover one chemical, a group of structurally or toxicologically related chemicals, or a chemical mixture. Exceptions outside the scope of the IRIS program are radionuclides, chemicals used only as pesticides, and the "criteria air pollutants" (particulate matter, ground-level ozone, carbon monoxide, sulfur oxides, nitrogen oxides, and lead).

Enhancements to the IRIS program are improving its science, transparency, and productivity. To improve the science, the IRIS program is adapting and implementing principles of systematic review (i.e., using explicit methods to identify, evaluate, and synthesize study findings). To increase transparency, the IRIS program discusses key science issues with the scientific community and the public as it begins an assessment. External peer review, independently managed and in public, improves both science and transparency. Increased productivity requires that assessments be concise, focused on EPA's needs, and completed without undue delay.

IRIS assessments follow EPA guidance[3] and standardized practices of systematic review. This Preamble summarizes and does not change IRIS operating procedures or EPA guidance.

Periodically, the IRIS program asks for nomination of agents for future assessment or reassessment. Selection depends on EPA's priorities, relevance to public health, and availability of pertinent studies. The IRIS multiyear agenda[4] lists upcoming assessments. The IRIS program may also assess other agents in anticipation of public health needs.

2. Planning an Assessment: Scoping, Problem Formulation, and Protocols

Early attention to planning ensures that IRIS assessments meet their objectives and properly frame science issues.

Scoping refers to the first step of planning, where the IRIS program consults with EPA's program and regional offices to ascertain their needs. Scoping specifies the agents an

[2]IRIS program website: http://www.epa.gov/iris/.
[3]EPA guidance documents: http://www.epa.gov/iris/basic-information-about-integrated-risk-information-system#guidance/.
[4]IRIS multiyear agenda: https://www.epa.gov/iris/iris-agenda.

assessment will address, routes and durations of exposure, susceptible populations and lifestages, and other topics of interest.

Problem formulation refers to the science issues an assessment will address and includes input from the scientific community and the public. A preliminary literature survey, beginning with secondary sources (e.g., assessments by national and international health agencies and comprehensive review articles), identifies potential health outcomes and science issues. It also identifies related chemicals (e.g., toxicologically active metabolites and compounds that metabolize to the chemical of interest).

Each IRIS assessment comprises multiple systematic reviews for multiple health outcomes. It also evaluates hypothesized mechanistic pathways and characterizes exposure–response relationships. An assessment may focus on important health outcomes and analyses rather than expand beyond what is necessary to meet its objectives.

Protocols refer to the systematic review procedures planned for use in an assessment. They include strategies for literature searches, criteria for study inclusion or exclusion, considerations for evaluating study methods and quality, and approaches to extracting data. Protocols may evolve as an assessment progresses and new agent-specific insights and issues emerge.

3. Identifying and Selecting Pertinent Studies

IRIS assessments conduct systematic literature searches with criteria for inclusion and exclusion. The objective is to retrieve the pertinent primary studies (i.e., studies with original data on health outcomes or their mechanisms). *PECO statements* (Populations, Exposures, Comparisons, Outcomes) govern the literature searches and screening criteria. "Populations" and animal species generally have no restrictions. "Exposures" refers to the agent and related chemicals identified during scoping and problem formulation and may consider route, duration, or timing of exposure. "Comparisons" means studies that allow comparison of effects across different levels of exposure. "Outcomes" may become more specific (e.g., from "toxicity" to "developmental toxicity" to "hypospadias") as an assessment progresses.

For studies of absorption, distribution, metabolism, and elimination, the first objective is to create an inventory of pertinent studies. Subsequent sorting and analysis facilitates characterization and quantification of these processes.

Studies on mechanistic events can be numerous and diverse. Here, too, the objective is to create an inventory of studies for later sorting to support analyses of related data. The inventory also facilitates generation and evaluation of hypothesized mechanistic pathways.

The IRIS program posts initial protocols for literature searches on its website and adds search results to EPA's HERO database.[5] Then the IRIS program takes extra steps to ensure identification of pertinent studies: by encouraging the scientific community and the public to identify additional studies and ongoing research; by searching for data submitted under the Toxic Substances Control Act or the Federal Insecticide, Fungicide, and Rodenticide Act; and by considering late-breaking studies that would impact the credibility of the conclusions, even during the review process.[6]

4. Evaluating Study Methods and Quality

IRIS assessments evaluate study methods and quality, using uniform approaches for each group of similar studies. The objective is that subsequent syntheses can weigh study results on their merits. Key concerns are potential *bias* (factors that affect the magnitude or direction of an effect) and *insensitivity* (factors that limit the ability of a study to detect a true effect).

[5]Health and Environmental Research Online: https://hero.epa.gov/hero/.
[6]IRIS "stopping rules": https://www.epa.gov/sites/production/files/2014-06/documents/iris_stoppingrules.pdf.

For human and animal studies, the evaluation of study methods and quality considers study design, exposure measures, outcome measures, data analysis, selective reporting, and study sensitivity. For human studies, this evaluation also considers selection of participant and referent groups and potential confounding. Emphasis is on discerning bias that could substantively change an effect estimate, considering also the expected direction of the bias. Low sensitivity is a bias towards the null.

Study-evaluation considerations are specific to each study design, health effect, and agent. Subject-matter experts evaluate each group of studies to identify characteristics that bear on the informativeness of the results. For carcinogenicity, neurotoxicity, reproductive toxicity, and developmental toxicity, there is EPA guidance for study evaluation (U.S. EPA, 2005a, 1998b, 1996, 1991c). As subject-matter experts examine a group of studies, additional agent-specific knowledge or methodologic concerns may emerge and a second pass become necessary.

Assessments use evidence tables to summarize the design and results of pertinent studies. If tables become too numerous or unwieldy, they may focus on effects that are more important or studies that are more informative.

The IRIS program posts initial protocols for study evaluation on its website, then considers public input as it completes this step.

5. Integrating the Evidence of Causation for Each Health Outcome

Synthesis within lines of evidence. For each health outcome, IRIS assessments synthesize the human evidence and the animal evidence, augmenting each with informative subsets of mechanistic data. Each synthesis considers aspects of an association that may suggest causation: consistency, exposure–response relationship, strength of association, temporal relationship, biological plausibility, coherence, and "natural experiments" in humans (U.S. EPA, 1994a) (U.S. EPA, 2005a).

Each synthesis seeks to reconcile ostensible inconsistencies between studies, taking into account differences in study methods and quality. This leads to a distinction between *conflicting evidence* (unexplained positive and negative results in similarly exposed human populations or in similar animal models) and *differing results* (mixed results attributable to differences between human populations, animal models, or exposure conditions) (U.S. EPA, 2005a).

Each synthesis of human evidence explores alternative explanations (e.g., chance, bias, or confounding) and determines whether they may satisfactorily explain the results. Each synthesis of animal evidence explores the potential for analogous results in humans. Coherent results across multiple species increase confidence that the animal results are relevant to humans.

Mechanistic data are useful to augment the human or animal evidence with information on precursor events, to evaluate the human relevance of animal results, or to identify susceptible populations and lifestages. An agent may operate through multiple mechanistic pathways, even if one hypothesis dominates the literature (U.S. EPA, 2005a).

Integration across lines of evidence. For each health outcome, IRIS assessments integrate the human, animal, and mechanistic evidence to answer the question: *What is the nature of the association between exposure to the agent and the health outcome?*

For cancer, EPA includes a standardized hazard descriptor in characterizing the strength of the evidence of causation. The objective is to promote clarity and consistency of conclusions across assessments (U.S. EPA, 2005a).

Carcinogenic to humans: convincing epidemiologic evidence of a causal association; or strong human evidence of cancer or its key precursors, extensive animal evidence, identification of mode-of-action and its key precursors in animals, and strong evidence that they are anticipated in humans.

Likely to be carcinogenic to humans: evidence that demonstrates a potential hazard to humans. Examples include a plausible association in humans with supporting experimental evidence, multiple positive

results in animals, a rare animal response, or a positive study strengthened by other lines of evidence.

Suggestive evidence of carcinogenic potential: evidence that raises a concern for humans. Examples include a positive result in the only study, or a single positive result in an extensive database.

Inadequate information to assess carcinogenic potential: no other descriptors apply. Examples include little or no pertinent information, *conflicting evidence*, or negative results not sufficiently robust for *not likely*.

Not likely to be carcinogenic to humans: robust evidence to conclude that there is no basis for concern. Examples include no effects in well-conducted studies in both sexes of multiple animal species, extensive evidence showing that effects in animals arise through modes-of-action that do not operate in humans, or convincing evidence that effects are not likely by a particular exposure route or below a defined dose.

If there is credible evidence of carcinogenicity, there is an evaluation of mutagenicity, because this influences the approach to dose–response assessment and subsequent application of adjustment factors for exposures early in life (U.S. EPA, 2005a), (U.S. EPA, 2005b).

6. Selecting Studies for Derivation of Toxicity Values

The purpose of toxicity values (slope factors, unit risks, reference doses, reference concentrations; see section 7) is to estimate exposure levels likely to be without appreciable risk of adverse health effects. EPA uses these values to support its actions to protect human health.

The health outcomes considered for derivation of toxicity values may depend on the hazard descriptors. For example, IRIS assessments generally derive cancer values for agents that are *carcinogenic* or *likely to be carcinogenic*, and sometimes for agents with *suggestive evidence* (U.S. EPA, 2005a).

Derivation of toxicity values begins with a new evaluation of studies, as some studies used qualitatively for hazard identification may not be useful quantitatively for exposure–response assessment. Quantitative analyses require quantitative measures of exposure and response. An assessment weighs the merits of the human and animal studies, of various animal models, and of different routes and durations of exposure (U.S. EPA, 1994a). Study selection is not reducible to a formula, and each assessment explains its approach.

Other biological determinants of study quality include appropriate measures of exposure and response, investigation of early effects that precede overt toxicity, and appropriate reporting of related effects (e.g., combining effects that comprise a syndrome, or benign and malignant tumors in a specific tissue).

Statistical determinants of study quality include multiple levels of exposure (to characterize the shape of the exposure–response curve) and adequate exposure range and sample sizes (to minimize extrapolation and maximize precision) (U.S. EPA, 2012b).

Studies of low sensitivity may be less useful if they fail to detect a true effect or yield toxicity values with wide confidence limits.

7. Deriving Toxicity Values

General approach. EPA guidance describes a two-step approach to dose–response assessment: analysis in the range of observation, then extrapolation to lower levels. Each toxicity value pertains to a route (e.g., oral, inhalation, dermal) and duration or timing of exposure (e.g., chronic, subchronic, gestational) (U.S. EPA, 2002).

IRIS assessments derive a candidate value from each suitable data set. Consideration of candidate values yields a toxicity value for each organ or system. Consideration of the organ/system-specific values results in the selection of an overall toxicity value to cover all health outcomes. The organ/system-specific values are useful for subsequent cumulative risk assessments that consider the combined effect of

multiple agents acting at a common anatomical site.

Analysis in the range of observation. Within the observed range, the preferred approach is modeling to incorporate a wide range of data. Toxicokinetic modeling has become increasingly common for its ability to support target-dose estimation, cross-species adjustment, or exposure-route conversion. If data are too limited to support toxicokinetic modeling, there are standardized approaches to estimate daily exposures and scale them from animals to humans (U.S. EPA, 1994a), (U.S. EPA, 2005a), (U.S. EPA, 2011, 2006).

For human studies, an assessment may develop exposure–response models that reflect the structure of the available data (U.S. EPA, 2005a). For animal studies, EPA has developed a set of empirical ("curve-fitting") models[7] that can fit typical data sets (U.S. EPA, 2005a). Such modeling yields a *point of departure*, defined as a dose near the lower end of the observed range, without significant extrapolation to lower levels (e.g., the estimated dose associated with an extra risk of 10% for animal data or 1% for human data, or their 95% lower confidence limits)(U.S. EPA, 2005a), (U.S. EPA, 2012b).

When justified by the scope of the assessment, toxicodynamic ("biologically based") modeling is possible if data are sufficient to ascertain the key events of a mode-of-action and to estimate their parameters. Analysis of model uncertainty can determine the range of lower doses where data support further use of the model (U.S. EPA, 2005a).

For a group of agents that act at a common site or through common mechanisms, an assessment may derive relative potency factors based on relative toxicity, rates of absorption or metabolism, quantitative structure–activity relationships, or receptor-binding characteristics (U.S. EPA, 2005a).

Extrapolation: slope factors and unit risks. An *oral slope factor* or an *inhalation unit risk* facilitates subsequent estimation of human cancer risks. Extrapolation proceeds linearly (i.e., risk proportional to dose) from the point of departure to the levels of interest. This is appropriate for agents with direct mutagenic activity. It is also the default if there is no established mode-of-action (U.S. EPA, 2005a).

Differences in susceptibility may warrant derivation of multiple slope factors or unit risks. For early-life exposure to carcinogens with a mutagenic mode-of-action, EPA has developed default *age-dependent adjustment factors* for agents without chemical-specific susceptibility data (U.S. EPA, 2005a), (U.S. EPA, 2005b).

If data are sufficient to ascertain the mode-of-action and to conclude that it is not linear at low levels, extrapolation may use the reference-value approach (U.S. EPA, 2005a).

Extrapolation: reference values. An *oral reference dose* or an *inhalation reference concentration* is an estimate of human exposure (including in susceptible populations) likely to be without appreciable risk of adverse health effects over a lifetime (U.S. EPA, 2002). Reference values generally cover effects other than cancer. They are also appropriate for carcinogens with a nonlinear mode-of-action.

Calculation of reference values involves dividing the point of departure by a set of *uncertainty factors* (each typically 1, 3, or 10, unless there are adequate chemical-specific data) to account for different sources of uncertainty and variability (U.S. EPA, 2002), (U.S. EPA, 2014).

Human variation: An uncertainty factor covers susceptible populations and lifestages that may respond at lower levels, unless the data originate from a susceptible study population.

Animal-to-human extrapolation: For reference values based on animal results, an uncertainty factor reflects cross-species differences, which may cause humans to respond at lower levels.

Subchronic-to-chronic exposure: For chronic reference values based on subchronic studies, an uncertainty factor reflects the likelihood that a lower level over a longer duration may induce a similar response. This

[7]Benchmark Dose Software: http://www.epa.gov/bmds/.

factor may not be necessary for reference values of shorter duration.

Adverse-effect level to no-observed-adverse-effect level: For reference values based on a lowest-observed-adverse-effect level, an uncertainty factor reflects a level judged to have no observable adverse effects.

Database deficiencies: If there is concern that future studies may identify a more sensitive effect, target organ, population, or lifestage, a *database uncertainty factor* reflects the nature of the database deficiency.

8. Process for Developing and Peer-Reviewing IRIS Assessments

The IRIS process (revised in 2009 and enhanced in 2013) involves extensive public engagement and multiple levels of scientific review and comment. IRIS program scientists consider all comments. Materials released, comments received from outside EPA, and disposition of major comments (steps 3, 4, and 6 below) become part of the public record.

Step 1: Draft development. As outlined in section 2 of this Preamble, IRIS program scientists specify the scope of an assessment and formulate science issues for discussion with the scientific community and the public. Next, they release initial protocols for the systematic review procedures planned for use in the assessment. IRIS program scientists then develop a first draft, using structured approaches to identify pertinent studies, evaluate study methods and quality, integrate the evidence of causation for each health outcome, select studies for derivation of toxicity values, and derive toxicity values, as outlined in Preamble sections 3–7.

Step 2: Agency review. Health scientists across EPA review the draft assessment.

Step 3: Interagency science consultation. Other federal agencies and the Executive Office of the President review the draft assessment.

Step 4: Public comment, followed by external peer review. The public reviews the draft assessment. IRIS program scientists release a revised draft for independent external peer review. The peer reviewers consider whether the draft assessment assembled and evaluated the evidence according to EPA guidance and whether the evidence justifies the conclusions.

Step 5: Revise assessment. IRIS program scientists revise the assessment to address the comments from the peer review.

Step 6: Final agency review and interagency science discussion. The IRIS program discusses the revised assessment with EPA's program and regional offices and with other federal agencies and the Executive Office of the President.

Step 7: Post final assessment. The IRIS program posts the completed assessment and a summary on its website.

9. General Structure of IRIS Assessments

Main text. IRIS assessments generally comprise two major sections: (1) Hazard Identification and (2) Dose–Response Assessment. Section 1.1 briefly reviews chemical properties and toxicokinetics to describe the disposition of the agent in the body. This section identifies related chemicals and summarizes their health outcomes, citing authoritative reviews. If an assessment covers a chemical mixture, this section discusses environmental processes that alter the mixtures humans encounter and compares them to mixtures studied experimentally.

Section 1.2 includes a subsection for each major health outcome. Each subsection discusses the respective literature searches and study considerations, as outlined in Preamble sections 3 and 4, unless covered in the front matter. Each subsection concludes with evidence synthesis and integration, as outlined in Preamble section 5.

Section 1.3 links health hazard information to dose–response analyses for each health outcome. One subsection identifies susceptible populations and lifestages, as observed in human

or animal studies or inferred from mechanistic data. These may warrant further analysis to quantify differences in susceptibility. Another subsection identifies biological considerations for selecting health outcomes, studies, or data sets, as outlined in Preamble section 6.

Section 2 includes a subsection for each toxicity value. Each subsection discusses study selection, methods of analysis, and derivation of a toxicity value, as outlined in Preamble sections 6 and 7.

Front matter. The Executive Summary provides information historically included in IRIS summaries on the IRIS program website. Its structure reflects the needs and expectations of EPA's program and regional offices.

A section on systematic review methods summarizes key elements of the protocols, including methods to identify and evaluate pertinent studies. The final protocols appear as an appendix.

The Preface specifies the scope of an assessment and its relation to prior assessments. It discusses issues that arose during assessment development and emerging areas of concern.

This Preamble summarizes general procedures for assessments begun after the date below. The Preface identifies assessment-specific approaches that differ from these general procedures.

August 2016

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EXECUTIVE SUMMARY

Summary of Occurrence and Health Effects

Benzo[a]pyrene is a five-ring polycyclic aromatic hydrocarbon (PAH). Benzo[a]pyrene (along with other PAHs) is released into the atmosphere as a component of smoke from forest fires, industrial processes, vehicle exhaust, cigarettes, and through the burning of fuel (such as wood, coal, and petroleum products). Oral exposure to benzo[a]pyrene can occur by eating certain food products, such as charred meats, where benzo[a]pyrene is formed during the cooking process or by eating foods grown in areas contaminated with benzo[a]pyrene (from the air and soil). Dermal exposure may occur from contact with soils or materials that contain soot, tar, or crude petroleum products or by using certain pharmaceutical products containing coal tars, such as those used to treat the skin conditions, eczema and psoriasis. The magnitude of human exposure to benzo[a]pyrene and other PAHs depends on factors such as lifestyle (e.g., diet, tobacco smoking), occupation, and living conditions (e.g., urban versus rural setting, domestic heating, and cooking methods).

Animal studies demonstrate that exposure to benzo[a]pyrene is associated with developmental (including developmental neurotoxicity), reproductive, and immunological effects. In addition, epidemiology studies involving exposure to PAH mixtures have reported associations between internal biomarkers of exposure to benzo[a]pyrene (benzo[a]pyrene diol epoxide-DNA adducts) and adverse birth outcomes (including reduced birth weight, postnatal body weight, and head circumference), neurobehavioral effects, and decreased fertility.

Studies in multiple animal species demonstrate that benzo[a]pyrene is carcinogenic at multiple tumor sites (alimentary tract, liver, kidney, respiratory tract, pharynx, and skin) by all routes of exposure. In addition, there is strong evidence of carcinogenicity in occupations involving exposure to PAH mixtures containing benzo[a]pyrene, such as aluminum production, chimney sweeping, coal gasification, coal-tar distillation, coke production, iron and steel founding, and paving and roofing with coal tar pitch. An increasing number of occupational studies demonstrate a positive exposure-response relationship with cumulative benzo[a]pyrene exposure and lung cancer.

Effects Other Than Cancer Observed Following Oral Exposure

In animals, oral exposure to benzo[a]pyrene has been shown to result in developmental toxicity (including developmental neurotoxicity), reproductive toxicity, and immunotoxicity. Developmental effects in rats and mice include neurobehavioral changes and cardiovascular effects following gestational exposures. Reproductive and immune effects include decreased sperm counts, ovary weight, and follicle numbers, and decreased immunoglobulin and B cell numbers and thymus weight following oral exposures in adult animals. In humans, benzo[a]pyrene exposure

occurs in conjunction with other PAHs and, as such, attributing the observed effects to benzo[a]pyrene is complicated. However, some human studies report associations between particular health endpoints and internal measures of exposure, such as benzo[a]pyrene-deoxyribonucleic acid (DNA) adducts, or external measures of benzo[a]pyrene exposure. Overall, the human studies report developmental, neurobehavioral, reproductive, and immune effects that are generally analogous to those observed in animals, and provide qualitative, supportive evidence for hazards associated with benzo[a]pyrene exposure.

Oral Reference Dose (RfD) for Effects Other Than Cancer

Organ- or system-specific RfDs were derived for hazards associated with benzo[a]pyrene exposure where data were amenable (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.

Developmental toxicity, represented by neurobehavioral changes persisting into adulthood, was chosen as the basis for the overall oral RfD as the available data indicate that developmental neurotoxicity represents the most sensitive hazard of benzo[a]pyrene exposure. The neurodevelopmental study by Chen et al. (2012) was used to derive the RfD. Altered responses in three behavioral tests (i.e., Morris water maze, elevated plus maze, and open field tests) were selected to represent the critical effect of abnormal behavior, due to the consistency (i.e., each of these responses were affected in two separate cohorts of rats, including testing as juveniles and as adults; similar effects in these behavioral tests were observed across studies) and sensitivity of these responses, and the observed dose-response relationship of effects across dose groups. Benchmark dose (BMD) modeling for each of the three endpoints resulted in $BMDL_{1SD}$ values that clustered in the range 0.086–0.16 mg/kg-day. The lower end of this range of BMDLs, 0.086 mg/kg-day, was selected to represent the point of departure (POD) from these three endpoints for RfD derivation.

The overall RfD was calculated by dividing the POD for altered behavior in three tests of nervous system function by a composite uncertainty factor (UF) of 300 to account for the extrapolation from animals to humans (10), for interindividual differences in human susceptibility (10), and for deficiencies in the toxicity database (3).

Table ES-1. Organ/system-specific RfDs and overall RfD for benzo[a]pyrene

Effect	Basis	RfD (mg/kg-d)	Confidence
Developmental	Neurobehavioral changes Gavage neurodevelopmental study in rats (postnatal days [PNDs] 5–11) Chen et al. (2012)	3×10^{-4}	Medium
Reproductive	Decreased ovarian follicles and ovary weight Gavage subchronic (60 d) reproductive toxicity study in rats Xu et al. (2010)	4×10^{-4}	Medium
Immunological	Decreased thymus weight and serum IgM Gavage subchronic (35 d) study in rats De Jong et al. (1999)	2×10^{-3}	Low
Overall RfD	**Developmental toxicity (including developmental neurotoxicity)**	3×10^{-4}	**Medium**

Confidence in the Overall Oral RfD

The overall confidence in the RfD is medium. Confidence in the principal study (Chen et al., 2012) is medium. The design, conduct, and reporting of this neurodevelopmental study was good and a wide variety of neurotoxicity endpoints were measured across 40 litters of rats. However, some uncertainty exists regarding the authors' use of dam rotation across litters (an attempt to reduce potential nurturing bias) and a within-litter dosing design, by potentially introducing maternal stress or other unanticipated consequences in the pups, and some informative experimental details were omitted, including the sensitivity of some assays at the indicated developmental ages and lack of reporting of individual animal- or gender-specific data for all outcomes. Several subchronic and developmental studies covering a wide variety of endpoints are also available; however, a multigeneration toxicity study with exposure throughout development and across generations is not available, and the available neurotoxicity studies did not comprehensively evaluate all potentially vulnerable lifestages of nervous system development. Therefore, confidence in the database is medium.

Effects Other Than Cancer Observed Following Inhalation Exposure

In animals, inhalation exposure to benzo[a]pyrene has been shown to result in developmental and reproductive toxicity. Studies in rats following inhalation exposure show decreased embryo/fetal survival and nervous system effects in offspring, and decreased testes weight and sperm counts in adult animals. Overall, the available human PAH mixtures studies report developmental and reproductive effects that are generally analogous to those observed in animals, and provide qualitative, supportive evidence for the hazards associated with benzo[a]pyrene exposure.

Inhalation Reference Concentration (RfC) for Effects Other Than Cancer

An attempt was made to derive organ- or system-specific RfCs for hazards associated with benzo[a]pyrene exposure where data were amenable (see Table ES-2). 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.

Developmental toxicity, represented by decreased embryo/fetal survival, was chosen as the basis for the proposed inhalation RfC as the available data indicate that developmental effects represent a sensitive hazard of benzo[a]pyrene exposure. The developmental inhalation study in rats by Archibong et al. (2002) and the observed decreased embryo/fetal survival (i.e. increased resorptions) following exposure to benzo[a]pyrene on gestation days (GDs) 11–20 were used to derive the overall RfC. The lowest-observed-adverse-effect level (LOAEL) of 25 μg/m^3 based on decreased embryo/fetal survival was selected as the POD. The LOAEL was adjusted to account for the discontinuous daily exposure to derive the POD_{ADJ} and the human equivalent concentration (HEC) was calculated from the POD_{ADJ} by multiplying by the regional deposited dose ratio ($RDDR_{ER}$) for extrarespiratory (i.e., systemic) effects, as described in *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994b). These adjustments resulted in a POD_{HEC} of 4.6 μg/m^3, which was used as the POD for RfC derivation.

The RfC was calculated by dividing the POD by a composite UF of 3,000 to account for toxicodynamic differences between animals and humans (3), interindividual differences in human susceptibility (10), LOAEL-to-no-observed-adverse-effect level (NOAEL) extrapolation (10), and deficiencies in the toxicity database (10).

Table ES-2. Organ/system-specific RfCs and overall RfC for benzo[a]pyrene

Effect	Basis	RfC (mg/m^3)	Confidence
Developmental	Decreased embryo/fetal survival Developmental toxicity study in rats (GDs 11–20) Archibong et al. (2002)	2×10^{-6}	Low-medium
Reproductive	Reduced ovulation rate and ovary weight Premating study in rats (14 d) Archibong et al. (2012)	3×10^{-6}	Low-medium
Overall RfC	**Developmental toxicity**	2×10^{-6}	Low-medium

[a]Not calculated due to UF >3,000.

Confidence in the Overall Inhalation RfC

The overall confidence in the RfC is low-to-medium. Confidence in the principal study (Archibong et al., 2002) is medium. The conduct and reporting of this developmental inhalation

study were adequate; however, a NOAEL was not identified. Confidence in the database is low due to the lack of a multigeneration toxicity study and the lack of information on varied toxicity endpoints following subchronic and chronic inhalation exposure. However, confidence in the RfC is bolstered by consistent systemic effects observed by the oral route (including reproductive and developmental effects) and similar effects observed in human populations exposed to PAH mixtures.

Evidence for Human Carcinogenicity

Under EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), benzo[a]pyrene is "carcinogenic to humans" based on strong and consistent evidence in animals and humans. The evidence includes an extensive number of studies demonstrating carcinogenicity in multiple animal species exposed via all routes of administration and increased cancer risks, particularly in the lung and skin, in humans exposed to different PAH mixtures containing benzo[a]pyrene. Mechanistic studies provide strong supporting evidence that links the metabolism of benzo[a]pyrene to DNA-reactive agents with key mutational events in genes that can lead to tumor development. These events include formation of specific DNA adducts and characteristic mutations in oncogenes and tumor suppressor genes that have been observed in humans exposed to PAH mixtures. This combination of human, animal, and mechanistic evidence provides the basis for characterizing benzo[a]pyrene as "carcinogenic to humans."

Quantitative Estimate of Carcinogenic Risk From Oral Exposure

Lifetime oral exposure to benzo[a]pyrene has been associated with forestomach, liver, oral cavity, jejunum or duodenum, and auditory canal tumors in male and female Wistar rats, forestomach tumors in male and female Sprague-Dawley rats, and forestomach, esophagus, tongue, and larynx tumors in female $B6C3F_1$ mice (male mice were not tested). Less-than-lifetime oral exposure to benzo[a]pyrene has also been associated with forestomach tumors in more than 10 additional bioassays with several strains of mice. The Kroese et al. (2001) and Beland and Culp (1998) studies were selected as the best available studies for dose-response analysis and extrapolation to lifetime cancer risk following oral exposure to benzo[a]pyrene. These studies included histological examinations for tumors in many different tissues, contained three exposure levels and controls, contained adequate numbers of animals per dose group (~50/sex/group), treated animals for up to 2 years, and included detailed reporting methods and results (including individual animal data).

Time-weighted, average daily doses were converted to human equivalent doses (HEDs) on the basis of $(\text{body weight})^{3/4}$ scaling (U.S. EPA, 1992). EPA then used the multistage-Weibull model for the derivation of the oral slope factor. This model was used because it incorporates the time at which death-with-tumor occurred and can account for differences in mortality observed between the exposure groups. Using linear extrapolation from the $BMDL_{10}$, human equivalent oral slope factors were derived for each gender/tumor site combination (slope factor = $0.1/BMDL_{10}$) reported

by Kroese et al. (2001) and Beland and Culp (1998). The oral slope factor of **1 per mg/kg-day** based on the tumor response in the alimentary tract (forestomach, esophagus, tongue, and larynx) of female $B6C3F_1$ mice (Beland and Culp, 1998) was selected as the factor with the highest value (most sensitive) among a range of slope factors derived.

Quantitative Estimate of Carcinogenic Risk From Inhalation Exposure

Inhalation exposure to benzo[a]pyrene has been associated with squamous cell neoplasia in the larynx, pharynx, trachea, nasal cavity, esophagus, and forestomach of male Syrian golden hamsters exposed for up to 130 weeks to benzo[a]pyrene condensed onto NaCl particles (Thyssen et al., 1981). Supportive evidence for the carcinogenicity of inhaled benzo[a]pyrene comes from additional studies with hamsters exposed to benzo[a]pyrene via intratracheal instillation. The Thyssen et al. (1981) bioassay represents the only study of lifetime exposure to inhaled benzo[a]pyrene.

A time-to-tumor dose-response model was fit to the time-weighted average (TWA) continuous exposure concentrations and the individual animal incidence data for the overall incidence of tumors in the upper respiratory tract or pharynx. The inhalation unit risk of **6×10^{-4} per µg/m³** was calculated by linear extrapolation (slope factor = $0.1/BMCL_{10}$) from a $BMCL_{10}$ of 0.16 mg/m³ for the occurrence of upper respiratory and upper digestive tract (forestomach) tumors in male hamsters chronically exposed by inhalation to benzo[a]pyrene (Thyssen et al., 1981).

Susceptible Populations and Lifestages

Benzo[a]pyrene has been determined to be carcinogenic by a mutagenic mode of action in this assessment. According to the *Supplemental Guidance for Assessing Susceptibility from Early Life Exposure to Carcinogens* (U.S. EPA, 2005b), individuals exposed during early life to carcinogens with a mutagenic mode of action are assumed to have an increased risk for cancer. The oral slope factor of 1 per mg/kg-day and inhalation unit risk of 0.0006 per µg/m³, calculated from data applicable to adult exposures, do not reflect presumed early life susceptibility to this chemical. Although some chemical-specific data exist for benzo[a]pyrene that demonstrate increased early life susceptibility to cancer, these data were not considered sufficient to develop separate risk estimates for childhood exposure. In the absence of adequate chemical-specific data to evaluate differences in age-specific susceptibility, the *Supplemental Guidance* (U.S. EPA, 2005b) recommends that age-dependent adjustment factors (ADAFs) be applied in estimating cancer risk. The ADAFs are 10- and 3-fold adjustments that are combined with age specific exposure estimates when estimating cancer risks from early life (<16 years of age) exposures to benzo[a]pyrene.

Regarding effects other than cancer, there are epidemiological studies that report associations between developmental effects (decreased postnatal growth, decreased head circumference, and neurodevelopmental delays), reproductive effects and internal biomarkers of exposure to benzo[a]pyrene. Studies in animals also indicate alterations in neurological

development and heightened susceptibility to reproductive effects following gestational or early postnatal exposure to benzo[a]pyrene. More preliminary data suggests effects on cardiovascular, kidney, pulmonary and immune system development may result from early life exposures, although few in vivo developmental studies exist to confirm these findings.

Key Issues Addressed in Assessment

The overall RfD and RfC were developed based on effects observed following exposure to benzo[a]pyrene during a critical window of development. The derivation of a general population toxicity value based on exposure during development has implications regarding the evaluation of populations exposed outside of the developmental period and the averaging of exposure to durations outside of the critical window of susceptibility. Discussion of these considerations is provided in Sections 2.1.5 and 2.2.5.

LITERATURE SEARCH STRATEGY|STUDY SELECTION

The literature search strategy used to identify primary, peer-reviewed literature pertaining to benzo[a]pyrene was conducted using the databases listed in Table LS-1 (see Appendix C for the complete list of keywords). References from previous assessments by the U.S. Environmental Protection Agency (EPA) and other national and international health agencies were also examined to ensure that critical studies were not missed by the literature search. EPA conducted a comprehensive, systematic literature search for benzo[a]pyrene through February, 2012. The literature search results were shared on the EPA Docket (https://www.gpo.gov/fdsys/pkg/FR-2012-07-16/html/2012-17145.htm), and the public was invited to review the literature search results and submit additional information to EPA (e.g. unpublished studies or other primary technical sources that are not available through the open literature). The inclusion/exclusion criteria that were applied to the 2012 and 2014 literature searches conducted prior to external peer review are presented in Table LS-1.

Following external peer review, the literature search was updated (August 2016). Consistent with the IRIS Stopping Rules (http://www.epa.gov/sites/production/files/2014-06/documents/iris_stoppingrules.pdf), manual screening of the literature search update focused on identifying new studies which might change a major conclusion of the assessment. Upon review, the potentially pertinent references identified in the post-peer review literature search did not impact the assessment's conclusions; thus, these studies were not added to the assessment. The references identified with the latest update, including bibliographic information and abstracts, can be found on the Health and Environmental Research Online (HERO) website (http://hero.epa.gov/benzoapyrene) and are tagged as "August 2016 Update".

Table LS-1. Summary of the search strategy employed for benzo[a]pyrene

Database	Keywords
Pubmed Toxcenter Toxline	Chemical name (CASRN): benzo[a]pyrene (50-32-8) Synonyms: benzo[d,e,f]chrysene, benzo[def]chrysene, 3,4-benzopyrene, 1,2-benzpyrene, 3,4-bp, benz(a)pyrene, 3,4-benzpyren, 3,4-benzpyrene, 4,5-benzpyrene, 6,7-benzopyrene, benzopirene, benzo(alpha)pyrene Standard toxicology search keywords Toxicity (including duration, effects to children and occupational exposure); development; reproduction; teratogenicity; exposure routes; pharmacokinetics; toxicokinetics; metabolism; body fluids; endocrinology; carcinogenicity; genotoxicity; antagonists; inhibitors

Database	Keywords
TSCATS ChemID Chemfinder CCRIS HSDB GENETOX RTECS	Searched by CASRNs and chemical names (including synonyms)

[a]Primary and secondary keywords used for the Pubmed, Toxcenter, and Toxline databases can be found in the Supplemental Information.

Figure LS-1 depicts the 2012 literature search, study selection strategy, and number of references associated with each stage of literature screening. Approximately 20,700 references were identified with the initial keyword search. Based on a secondary keyword search followed by a preliminary manual screen of titles or abstracts by a toxicologist, approximately 1,190 references were identified that provided information potentially relevant to characterizing the health effects or physical and chemical properties of benzo[a]pyrene. A more detailed manual review of titles, abstracts, and/or papers was then conducted. Notable exclusions from the Toxicological Review are large numbers of animal in vivo or in vitro studies designed to identify potential therapeutic agents that would prevent the carcinogenicity or genotoxicity of benzo[a]pyrene and toxicity studies of benzo[a]pyrene in nonmammalian species (e.g., aquatic species, plants).

For the updated literature search conducted for the timeframe January 2012 through August 2014, the search terms included benzo(a)pyrene AND (rat OR mouse OR mice) and results were screened manually by title, abstract, and/or full text using the exclusion criteria outlined in Figure LS-1. Relevant studies that could potentially impact the hazard characterization and dose-response assessment were identified and considered. Several pertinent studies, published since the last comprehensive literature search (i.e. 2012), were identified and incorporated into the text where relevant.

In addition to the comprehensive literature search, more iterative literature searches were conducted throughout the draft development process. For example, specialized searches were conducted during draft development to provide additional context for potential mechanisms of hazards identified from in vivo subchronic, chronic, or developmental studies. Additional literature may be sought to fill in data gaps and to help inform a responses to peer review comments. Any references from the iterative searches were captured and documented in the updated literature searches of 2014 and 2016.

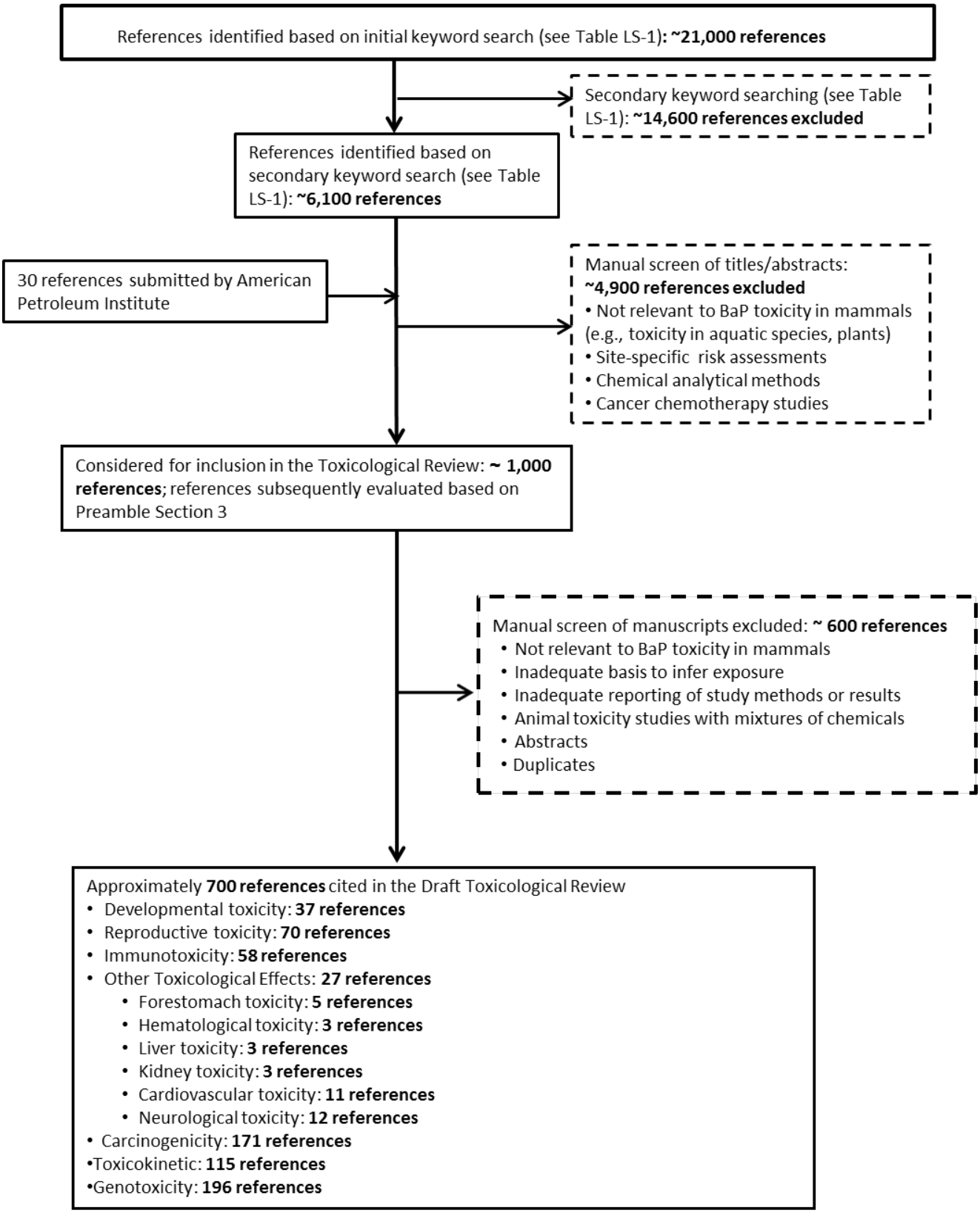

Figure LS-1. Study selection strategy.

Selection of studies for inclusion in the Toxicological Review was based on consideration of the extent to which the study was informative and relevant to the assessment and general study quality considerations. In general, the relevance of health effect studies was evaluated as outlined in the Preamble and EPA guidance (*A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002) and *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhaled Dosimetry* (U.S. EPA, 1994a). The reasons for excluding epidemiological and animal studies from the references identified by the keyword search are provided in Figure LS-1.

The available studies examining the health effects of benzo[a]pyrene exposure in humans are discussed and evaluated in the hazard identification sections of the assessment (Section 1), with specific limitations of individual studies and of the collection of studies noted. The common major limitation of the human epidemiological studies (with respect to identifying potential adverse health outcomes specifically from benzo[a]pyrene) is that they all involve exposures to complex mixtures containing other PAHs and other compounds. The evaluation of the epidemiological literature focuses on studies in which possible associations between external measures of exposure to benzo[a]pyrene or biomarkers of exposure to benzo[a]pyrene (e.g., benzo[a]pyrene-DNA adducts or urinary biomarkers) and potential adverse health outcomes were evaluated. Pertinent mechanistic studies in humans (e.g., identification of benzo[a]pyrene-DNA adducts and characteristics of mutations in human tumors) were also considered in assessing the weight of evidence for the carcinogenicity of benzo[a]pyrene.

The health effects literature for benzo[a]pyrene is extensive. All animal studies of benzo[a]pyrene involving repeated oral, inhalation, or dermal exposure that were considered to be of acceptable quality, whether yielding positive, negative, or null results, were considered in assessing the evidence for health effects associated with chronic exposure to benzo[a]pyrene. These studies were evaluated for aspects of design, conduct, or reporting that could affect the interpretation of results and the overall contribution to the synthesis of evidence for determination of hazard potential using the study quality considerations outlined in the Preamble. Discussion of study strengths and limitations (that ultimately supported preferences for the studies and data relied upon) were included in the text where relevant.

Animal toxicity studies involving short-term duration and other routes of exposure were also evaluated to inform conclusions about health hazards, especially regarding mode of action. The references considered and cited in this document, including bibliographic information and abstracts, can be found on the Health and Environmental Research Online (HERO) website[8] (http://hero.epa.gov/benzoapyrene).

[8]HERO (Health and Environmental Research On-line) is a database of scientific studies and other references used to develop EPA's risk assessments aimed at understanding the health and environmental effects of pollutants and chemicals. It is developed and managed in EPA's Office of Research and Development (ORD) by the National Center for Environmental Assessment (NCEA). New studies are added continuously to HERO.

1. HAZARD IDENTIFICATION

1.1. PRESENTATION AND SYNTHESIS OF EVIDENCE BY ORGAN/SYSTEM

NOTE: In the environment, benzo[a]pyrene occurs in conjunction with other structurally related chemical compounds known as polycyclic aromatic hydrocarbons (PAHs).[9] Accordingly, there are few epidemiologic studies designed to solely investigate the effects of benzo[a]pyrene. There are, however, many epidemiologic studies that have investigated the effects of exposure to PAH mixtures. Benzo[a]pyrene is universally present in these mixtures and is routinely analyzed and detected in environmental media contaminated with PAH mixtures; thus, it is often used an indicator chemical to measure exposure to PAH mixtures (Boström et al., 2002).

1.1.1. Developmental Toxicity

Human and animal studies provide evidence for PAH- and benzo[a]pyrene-induced developmental effects (including developmental neurotoxicity). Effects on embryo/fetal survival, postnatal growth, neurobehavioral function, and development have been demonstrated in human populations exposed to PAH mixtures during gestation. Animal studies demonstrate various effects including changes in embryo/fetal survival, pup weight, blood pressure, fertility, reproductive organ weight and histology, and nervous system function in gestationally and/or early postnatally treated animals.

Studies in humans and animal models indicate that benzo[a]pyrene and metabolites are widely distributed in maternal and fetal tissues, supporting placental transfer (Madhavan and Naidu, 1995; Withey et al., 1993; Neubert and Tapken, 1988; Shendrikova and Aleksandrov, 1974). In addition, benzo[a]pyrene can be readily detected in human milk, especially in smokers (Yu et al., 2011; Lapole et al., 2007; Zanieri et al., 2007). However, benzo[a]pyrene is also readily metabolized, widely distributed, and eliminated in mammals (see Section D.1 in the Supplemental Information). Evaluations of benzo[a]pyrene distribution and elimination in lactating animals do not suggest that neonatal lactational doses to benzo[a]pyrene are magnified over maternal doses (Lapole et al., 2007; Lavoie et al., 1987b; West and Horton, 1976).

Altered Birth Outcomes

Human and animal studies provide evidence that benzo[a]pyrene exposure may lead to altered outcomes reflecting growth and development in utero or in early childhood. Two cohort studies in pregnant women in China and the United States examined cord blood levels of benzo[a]pyrene-7,8-diol-9,10 epoxide (BPDE)-deoxyribonucleic acid (DNA) adducts in relation to

[9]PAHs are a large class of chemical compounds formed during the incomplete combustion of organic matter.

measures of child growth following exposure to PAH mixtures (Tang et al., 2006; Perera et al., 2005b; Perera et al., 2004) (Table 1-1). In the Chinese cohort, high benzo[a]pyrene-adduct levels were associated with reduced weight at 18, 24, and 30 months of age, but not at birth (Tang et al., 2006). In the U.S. cohort, an independent effect on birth weight was not observed with either benzo[a]pyrene-adducts or environmental tobacco smoke (ETS) exposure; however, a doubling of cord blood adducts in combination with ETS exposure in utero was seen, corresponding to an 8% reduction in birth weight (Perera et al., 2005b). ETS, also called secondhand smoke, is the smoke given off by a burning tobacco product and the smoke exhaled by a smoker contains over 7,000 chemicals, including benzo[a]pyrene. No associations were seen with birth length (or height at later ages) in either of these cohort studies.

Two cohort studies in pregnant women in Spain and Norway evaluated the relationship between the dietary intake of benzo[a]pyrene estimated by food questionnaire and birth weight and length of offspring (Duarte-Salles et al., 2013; Duarte-Salles et al., 2012) (Table 1-1). In the Spanish cohort, benzo[a]pyrene intake was associated with decreased birth weight, reduced birth length, and small for gestational age (SGA) among infants born to women with low, but not high vitamin C intake (stratified as < or > mean vitamin C in the diet: 189.41 mg/day) (Duarte-Salles et al., 2012). These findings were confirmed by the larger Norwegian cohort, which demonstrated a relationship between dietary benzo[a]pyrene intake and reduced birth weight and length in offspring of all women including nonsmokers (Duarte-Salles et al., 2013). The magnitude of the association was higher in women consuming less than the recommended vitamin C intake of 85 mg/day, suggesting that vitamin C may exert a protective influence against the reduced birth weight associated with benzo[a]pyrene in the diet.

A Chinese case-control study indicated that PAH exposure may be associated with increased risk of fetal death (Wu et al., 2010). A strong association was seen between maternal blood benzo[a]pyrene-DNA adduct levels and risk of delayed miscarriage (fetal death before 14 weeks of gestation), with a fourfold increased risk for levels above compared with below the median. However, no significant difference in adduct levels was detected between fetal tissue from cases compared to controls.

Decreased embryo or fetal survival, as evidenced by decreased litter size, has also been noted in pregnant animals treated following implantation by the oral and inhalation routes. An approximate 40% decrease in litter size was noted in mouse dams treated by gavage on GDs 7–16 at doses of 160 mg/kg-day, but no decreases were observed at 10 or 40 mg/kg-day (Mackenzie and Angevine, 1981). Several lower dose studies of rats treated on GDs 14–17 with doses of up to 1.2 mg/kg-day benzo[a]pyrene did not observe any difference in litter size (Jules et al., 2012; McCallister et al., 2008; Brown et al., 2007). By the inhalation route, a dose-related increase in embryo/fetal resorptions was observed , with a 19% increase following the lowest tested exposure of 25 μg/m^3 on GDs 11–20 in F344 rats, and increasing rates of resorptions seen at the two higher doses (Archibong et al., 2002). Another publication from the same group of collaborators Wu et al.

(2003a) graphically reported embryo/fetal survival as part of a study analyzing metabolites of benzo[a]pyrene and activation of the aryl hydrocarbon receptor (AhR) and cytochrome P450 (CYP450) 1A1 (Wu et al., 2003a). This study did not report the number of dams or litters, and no numerical data were reported. The study authors reported greater decreases in embryo/fetal survival at 75 and 100 μg/m^3 benzo[a]pyrene on GDs 11–20, approximately 39 and 33%, as compared to the 25 μg/m^3 group,(approximately 73% survival). The paired carbon black control groups (for each exposure level) showed 78-93% survival.

In animals (Table 1-2 and Figure 1-1), reduced body weight in offspring has also been noted in some developmental studies. Decreases in body weight (up to 13%) were observed in mice following prenatal gavage exposure (gestation days [GDs] 7–16), and as time from exposure increased (postnatal days [PNDs] 20–42), the dose at which effects were observed decreased (from 40 to 10 mg/kg-day, respectively) (Mackenzie and Angevine, 1981). In addition, decreases in body weight (approximately 10–20%) were observed in rats on PNDs 36 and 71 following gavage exposure at only 2 mg/kg-day on PNDs 5–11 (Chen et al., 2012), and on PND 8 following gavage exposure to 10 or 25 mg/kg-d on PNDs 1–7 (Liang et al., 2012). At doses up to 1.2 mg/kg-day and follow-up to PND 30, two developmental studies in rats did not observe decrements in pup body weight following treatment from GD 14 to 17 (Jules et al., 2012; McCallister et al., 2008). Maternal toxicity was not observed in mouse or rat dams exposed to up to 160 mg/kg-day benzo[a]pyrene (Jules et al., 2012; McCallister et al., 2008; Brown et al., 2007; Kristensen et al., 1995; Mackenzie and Angevine, 1981).

Fertility in Offspring

Several studies suggest that gestational exposure to maternal tobacco smoke decreases the future fertility of female offspring (Ye et al., 2010; Jensen et al., 1998; Weinberg et al., 1989) (Table 1-1). In animal models, marked effects on the development of male and female reproductive organs and the fertility of animals exposed gestationally has also been demonstrated (Kristensen et al., 1995; Mackenzie and Angevine, 1981) (Table 1-2 and Figure 1-1). In two studies examining reproductive effects in mice, decreased fertility and fecundity in F1 animals was observed following exposure to doses ≥10 mg/kg-day during gestation (Kristensen et al., 1995; Mackenzie and Angevine, 1981). When F1 females were mated with untreated males, a dose-related decrease in fertility of >30% was observed, in addition to a 20% decrease in litter size starting at the lowest dose tested of 10 mg/kg-day (Mackenzie and Angevine, 1981). A dose-related decrease in fertility was also observed in male mice treated gestationally with benzo[a]pyrene. At the lowest dose tested (10 mg/kg-day), a 35% decrease in fertility was observed when gestationally exposed animals were mated with untreated females (Mackenzie and Angevine, 1981). Similar effects on fertility were observed in another developmental study in mice (Kristensen et al., 1995). F1 females (bred continuously for 6 months) in this study had 63% fewer litters, and litters were 30% smaller as compared to control animals. The fertility of male offspring was not assessed in this study.

Reproductive Organ Effects in Offspring

The above-mentioned studies also demonstrated dose-related effects on male and female reproductive organs in animals exposed gestationally (or during the early postnatal period) to benzo[a]pyrene (Table 1-2 and Figure 1-1). Testicular weight was decreased and atrophic seminiferous tubules and vacuolization were increased at ≥10 mg/kg-day in male mice exposed to benzo[a]pyrene gestationally from GD 7 to 16; severe atrophic seminiferous tubules were observed at 40 mg/kg-day (Mackenzie and Angevine, 1981). Testicular weight on PND 8, serum testosterone on PNDs 8 and 35, testicular daily sperm production on PND 90, and epididymal sperm count on PND 90 were also statistically significantly decreased in Sprague-Dawley rats treated on PNDs 1–7 at doses ≥10 mg/kg-day (Liang et al., 2012). Increased seminiferous tubule vacuolation occurred in rats exposed to 25 mg/kg-d on PNDs 1–7 (examined on PNDs 8 and 35) (Liang et al., 2012).

In female mice treated with doses ≥10 mg/kg-day during gestation, ovarian effects were observed including decreases in ovary weight, numbers of follicles, and corpora lutea (Kristensen et al., 1995; Mackenzie and Angevine, 1981). Specifically, ovary weight in F1 offspring was reduced 31% following exposure to 10 mg/kg-day benzo[a]pyrene (Kristensen et al., 1995), while in another gestational study at the same dose level, ovaries were so drastically reduced in size (or absent) that they were not weighed (Mackenzie and Angevine, 1981). Hypoplastic ovaries with few or no follicles and corpora lutea (numerical data not reported), and ovaries with few or no small, medium, or large follicles and corpora lutea (numerical data not reported) have also been observed in mouse offspring exposed gestationally to benzo[a]pyrene (Kristensen et al., 1995; Mackenzie and Angevine, 1981).

Cardiovascular Effects in Offspring

Increased systolic and diastolic blood pressure was observed in adult animals following gestational treatment with benzo[a]pyrene (Jules et al., 2012) (Table 1-2 and Figure 1-1). Approximate elevations in systolic and diastolic blood pressure of 20–30 and 50–80% were noted in the 0.6 and 1.2 mg/kg-day dose groups, respectively. Heart rate was decreased at 0.6 mg/kg-day, but was increased at 1.2 mg/kg-day.

Immune Effects in Offspring

Several injection studies in laboratory animals suggest that immune effects may occur following gestational or early postnatal exposure to benzo[a]pyrene. These studies are discussed in Section 1.1.3.

Table 1-1. Evidence pertaining to developmental effects of benzo[a]pyrene in humans

Study design and reference	Results
Tang et al. (2006) (Tongliang, China) Birth cohort 150 nonsmoking women who delivered babies between March 2002 and June 2002 Exposure: Mean hours per day exposed to ETS 0.42 (SD 1.19); lived within 2.5 km of power plant that operated from December 2001 to May 2002; benzo[a]pyrene-DNA adducts from maternal and cord blood samples; cord blood mean 0.33 (SD 0.14) (median 0.36) adducts/10^{-8} nucleotides; maternal blood mean 0.29 (SD 0.13) adducts/10^{-8} nucleotides	Relation between cord blood benzo[a]pyrene-DNA adducts and log-transformed weight and height Weight Beta (*p*-value) / Length (height) Beta (*p*-value) Birth: −0.007 (0.73) / −0.001 (0.89) 18 mo: −0.048 (0.03) / −0.005 (0.48) 24 mo: −0.041 (0.027) / −0.007 (0.28) 30 mo: −0.040 (0.049) / −0.006 (0.44) Adjusted for ETS, sex of child, maternal height, maternal weight, and gestational age (for measures at birth)
Perera et al. (2005b); Perera et al. (2004) (New York, United States) Birth cohort 265 pregnant African-American and Dominican nonsmoking women who delivered babies between April 1998 and October 2002 (214 and 208 for weight and length analysis, respectively); approximately 40% with a smoker in the home Exposure: Benzo[a]pyrene-DNA adducts in cord blood samples; mean 0.22 (SD 0.14) adducts/10^{-8} nucleotides; median of detectable values 0.36 adducts/ 10^{-8} nucleotides	Relation between cord blood benzo[a]pyrene-DNA adducts and log-transformed weight and length Weight Beta (*p*-value) / Length Beta (*p*-value) Interaction term: −0.088 (0.05) / −0.014 (0.39) Benzo[a]-pyrene-DNA adducts: −0.020 (0.49) / −0.005 (0.64) ETS in home: −0.003 (0.90) / −0.007 (0.32) Adjusted for ethnicity, sex of newborns, maternal body mass index, dietary PAHs, and gestational age
Wu et al. (2010) (Tianjin, China) Case control study: 81 cases (96% participation rate)—fetal death confirmed by ultrasound before 14 wks of gestation; 81 controls (91% participation rate)—elective abortions; matched by age, gestational age, and gravidity; excluded smokers and occupational PAH exposure Exposure: Benzo[a]pyrene in aborted tissue and maternal blood samples (51 cases and controls; two of four hospitals)	*Benzo[a]pyrene adduct levels (/10^8 nucleotides), mean (± SD)* Cases / Controls / (*p*-value) Maternal blood: 6.0 (± 4.7) / 2.7 (± 2.2) / (<0.001) Aborted tissue: 4.8 (± 6.0) / 6.0 (± 7.4) / (0.29) Low correlation between blood and tissue levels (r = −0.02 in cases, r = −0.21 in controls) Association between benzo[a]pyrene adducts and miscarriage[a] OR / (95% CI) Per unit increase in adducts: 1.37 / (1.12, 1.67) Dichotomized at median: 4.56 / (1.46, 14.3)

Study design and reference	Results
	[a]Conditional logistic regression, adjusted for maternal education, household income, and gestational age; age also considered as potential confounder
Duarte-Salles et al. (2012) (Catalonia, Spain) Birth cohort 657 pregnant women recruited during the first trimester between July 2004 and July 2006 (614 and 604 available for weight and length analysis, respectively); 37% passive smokers, 16% active smokers Exposure: Dietary benzo[a]pyrene intake during first trimester from food questionnaire (µg/d, mean ± SD) of 0.16 ± 0.04 for all women with < mean vitamin C intake and 0.22 ± 0.06 for all women with > mean vitamin C intake. Small for gestational age (SGA) defined as birth weight below the 10th percentile of a Spanish reference population	Relation between dietary benzo[a]pyrene intake during the first trimester and birth weight and length (by vitamin C intake and smoking status; Beta coefficients are for a 1-SD increase in dietary benzo[a]pyrene) Weight Beta (*p*-value) / Length Beta (*p*-value) < Mean vitamin C, all women: −101.63 (0.004) / −0.38 (0.017) < Mean vitamin C, nonsmokers: −88.26 (0.034) / −0.37 (0.048) >≥ Mean vitamin C, all women: 2.05 (0.945) / 0.10 (0.439) >≥ Mean vitamin C, nonsmokers: −3.71 (0.909) / 0.04 (0.784) Adjusted for sex of the child; gestational age; nulliparity; tobacco smoke exposure during pregnancy; maternal region of origin (European or non-European); and maternal education level, height, prepregnancy weight, and energy intake. Relation between SGA births and dietary benzo[a]pyrene intake during the first trimester (all women, by vitamin C intake, benzo[a]pyrene tertile 3 compared to tertiles 1 and 2) OR / (95% CI) < Mean vitamin C: 3.51 / (1.16, 10.59) > Mean vitamin C: 0.81 / (0.23, 2.75) Adjusted for sex of the child; gestational age; nulliparity; tobacco smoke exposure during pregnancy; maternal region of origin (European or non-European); and maternal education level, height, prepregnancy weight, and energy intake.
Duarte-Salles et al. (2013) (Norway) Birth cohort 50,651 pregnant women recruited between 1998 and 2008; 92% nonsmokers, 4% occasional smokers, and 4% daily smokers Exposure: Dietary benzo[a]pyrene intake from food questionnaire (µg/d, mean ± SD) of 0.149 ± 0.048 for all women	Relation between dietary benzo[a]pyrene intake during pregnancy and birth weight and length (Beta coefficients are for a 1-SD increase in dietary benzo[a]pyrene) Weight Beta (*p*-value) / Length Beta (*p*-value) All women: −10.2 (<0.001) / −0.05 (<0.001) All women, vitamin C <85 mg/d: −17.7 (0.003) / Not evaluated All women, vitamin C ≥85 mg/d: −9.1 (<0.001) / Not evaluated Nonsmokers: −10.3 (<0.001) / −0.05 (0.001)

Study design and reference	Results		
	Nonsmokers, vitamin C <85 mg/d	−15.7 (0.016)	Not evaluated
	Nonsmokers, vitamin C ≥85 mg/d	−9.6 (<0.001)	Not evaluated
	Adjusted for sex of the child; gestational age, maternal age, weight gain, parity, and pre-pregnancy body mass index; smoking during pregnancy; and plausibility of reported energy intake and vitamin C intake.		

CI = confidence interval; OR = odds ratio; SD = standard deviation.

Table 1-2. Evidence pertaining to developmental effects of benzo[a]pyrene in animals after oral or inhalation exposure

Study design and reference	Results
Birth outcomes and postnatal growth	

Study design and reference	Results
Mackenzie and Angevine (1981) CD-1 mice, 30 or 60 F0 females/ dose 0, 10, 40, or 160 mg/kg-d by gavage GDs 7–16 (developmental study with continuous breeding protocol) F1 pups (4/sex/litter) were allowed to remain with their mothers until weaning on PND 20. At 6 wks of age, F1 female mice (n = 20–55/group), were paired with an untreated male for a period of 6 mo. At 7 wks of age, F1 male mice (n = 20–45/group), were mated with two untreated females for 5-d periods for 25 d (for a total exposure of 10 untreated females/F1 male).	↓ number of F0 females with viable litters: 46/60, 21/30, 44/60, and 13/30* ↓ F1 body weight at PND 20 % change from control: 0, 4, −7*, and −13* ↓ F1 body weight at PND 42 % change from control: 0, −6*, −6*, and −10* (no difference in pup weight at PND 4) (Gross abnormalities not observed in F1 or F2 animals)
Kristensen et al. (1995) NMRI mice, 9 F0 females/dose 0 or 10 mg/kg-d by gavage GDs 7–16 (developmental study with continuous breeding protocol) At 6 wks of age, one F1 female from each litter (n = 9) was continuously bred with an untreated male for 6 mo. F2 offspring were inspected for gross deformities at birth, weight and sex were recorded 2 d after birth.	Exposed F0 females showed no gross signs of toxicity and no effects on fertility (data not reported) F1 females had statistically significant: ↓ number of F2 offspring ↓ number of pups/litter ↓ number of F2 litters ↑ number of days between F2 litters (Gross abnormalities not observed in F1 or F2 animals. Alteration in pup weight not reported for any generation)
Jules et al. (2012) Long-Evans rats, 6–17 F0 females/dose 0, 0.15, 0.3, 0.6, or 1.2 mg/kg-d by gavage GDs 14–17	No overt signs of toxicity in dams or offspring, differences in pup body weight, or number of pups/litter

Study design and reference	Results
McCallister et al. (2008) Long-Evans Hooded rats, 5–6/group 0 or 0.3 mg/kg-d by gavage GDs 14–17	No difference in number of pups/litter No overt maternal or pup toxicity No difference in liver:body weight Increased brain:body weight ratio at PNDs 15 and 30 (data not shown)
Brown et al. (2007) Long-Evans Hooded rats, 6/group 0, 0.025, or 0.15 mg/kg-d by gavage GDs 14–17	No difference in number of pups/litter or overt maternal or pup toxicity
Chen et al. (2012) Sprague-Dawley rats, 20 pups (10 male and 10 female)/group 0, 0.02, 0.2, or 2 mg/kg-d by gavage PNDs 5–11	Statistically significant decrease in pup body weight (approximate 10–15% decrease) at 2 mg/kg-d measured on PNDs 36 and 71 (no significant alteration of pup weight during treatment period) No differences among treatment groups in developmental milestones: incisor eruption, eye opening, development of fur, testis decent, or vaginal opening
Liang et al. (2012) Sprague-Dawley rats, 5–6 litters/dose (12 pups per litter) 0, 5, 10, or 25 mg/kg-d by gavage PND 1–7	↓ body weight at PND 8 % change from control: 0, −0.6, −16*, and −17* (no difference at PND 35 and PND 90)
Archibong et al. (2002) F344 rats, 10 females/group 0, 25, 75, or 100 µg/m^3 nose-only inhalation for 4 hrs/d GDs 11–20	↓ embryo/fetal survival ([pups/litter]/[implantation sites/litter] × 100) % embryo/fetal survival: 97, 78*, 38*, and 34*%
Wu et al. (2003a) F344 rats, n not reported 0, 25, 75, 100 µg/m^3 nose-only inhalation for 4 hrs/d on GDs 11–20	↓ birth index (%), estimated from graph: 78–93[b], 73, 39*, 33%*[c] Birth index calculated in same manner as embryo/fetal survival in (related publication: Archibong et al., 2002). Decreased birth index estimated based on concurrent carbon black controls reported for each exposure concentration. SD or SEM not reported for most groups.

Study design and reference	Results
Reproductive effects in offspring	
Mackenzie and Angevine (1981) CD-1 mice, 30 or 60 F0 females/ dose 0, 10, 40, or 160 mg/kg-d by gavage GDs 7–16 (developmental study with continuous breeding protocol) At 6 wks of age, F1 female mice (n = 20–55/group), were paired with an untreated male for a period of 6 mo. At 7 wks of age, F1 male mice (n = 20–45/group), were mated with two untreated females for 5-d periods for 25 d (for a total exposure of 10 untreated females/F1 male).	↓ number of F1 females with viable litters: 35/35, 23/35*, 0/55*, and 0/20* ↓F1 female fertility index (females pregnant/females mated with males × 100): 100, 66*, 0*, and 0* ↓ F1 male fertility index (females pregnant/females mated with males × 100): 80, 52*, 5*, and 0* ↓ F2 litter size from F1 dams (20%) at 10 mg/kg-d (no litters were produced at high doses) ↓ size or absence of F1 ovaries (weights not collected) hypoplastic ovaries with few or no follicles and corpora lutea (numerical data not reported) ↓ testicular weight in F1 offspring % change from control: 0, −42, −82, and ND (statistical significance not reported) ↑ atrophic seminiferous tubules and vacuolization at ≥10 mg/kg-d; severe atrophic seminiferous tubules at 40 mg/kg-d (numerical data not reported)
Kristensen et al. (1995) NMRI mice, 9 F0 females/dose 0 or 10 mg/kg-d by gavage GDs 7–16 (developmental study with continuous breeding protocol) At 6 weeks of age, one F1 female from each litter (n = 9) was continuously bred with an untreated male for 6 months.	↓ number of F2 litters (−63%) ↓ F2 litter size (−30%) ↓ ovary weight (−31%) in F1 females Few or no small, medium, or large follicles and corpora lutea

Study design and reference	Results
Liang et al. (2012) Sprague-Dawley rats, 5–6 litters/dose (12 pups per litter) 0, 5, 10, or 25 mg/kg-d by gavage PND 1–7	↓ testes weight at PNDs 8 and 35 % change from control at PND 8: 0, 4, −14*, and −10* % change from control at PND 35: 0, 1, −6, and −17* (no difference at PND 90) ↓ serum testosterone at PNDs 8, 35, and 90 % change from control at PND 8: 0, −31, −62*, and −62* % change from control at PND 35: 0, −38, −42*, and −42* % change from control at PND 90: 0, −29, −52, and −65* ↑ seminiferous tubule vacuolization on PND 8 and PND 35 at 25 mg/kg-d (numerical data not reported). No difference at PND 90, or at any time point at 5 mg/kg-d. Report does not clearly indicate whether this was observed at 10 mg/kg-d. ↓ testicular daily sperm production on PND 90 Approximate % change from control (data reported graphically): 0, −20, −30*, and −30* ↓ epididymal sperm count on PND 90 Approximate % change from control (data reported graphically): 0, −20, −30*, and −20*
Cardiovascular effects in offspring	
Jules et al. (2012) Long-Evans rats, 6–17 F0 females/dose 0, 0.15, 0.3, 0.6, or 1.2 mg/kg-d by gavage GDs 14–17	↑ systolic blood pressure (measured at PND 53) 15%* increase at 0.6 mg/kg-d 52%* increase at 1.2 mg/kg-d (other dose groups not reported) ↑ diastolic blood pressure (measured at PND 53) 33%* increase at 0.6 mg/kg-d 83% *increase at 1.2 mg/kg-d (other dose groups not reported) Altered heart rate 10%* increase at 0.6 mg/kg-d 8%* decrease at 1.2 mg/kg-d

*Statistically significantly different from the control ($p < 0.05$).
[a]% change from control calculated as: (treated value − control value)/control value × 100.
[b]Carbon black controls were used for each dose group.
[c]Under the assumption of 10 dams per group, as used by Archibong et al. (2002), the Cochran-Armitage trend test yields a statistically significant trend in decreased birth index across the pooled carbon black control groups and the exposure levels tested (p=0.0004).

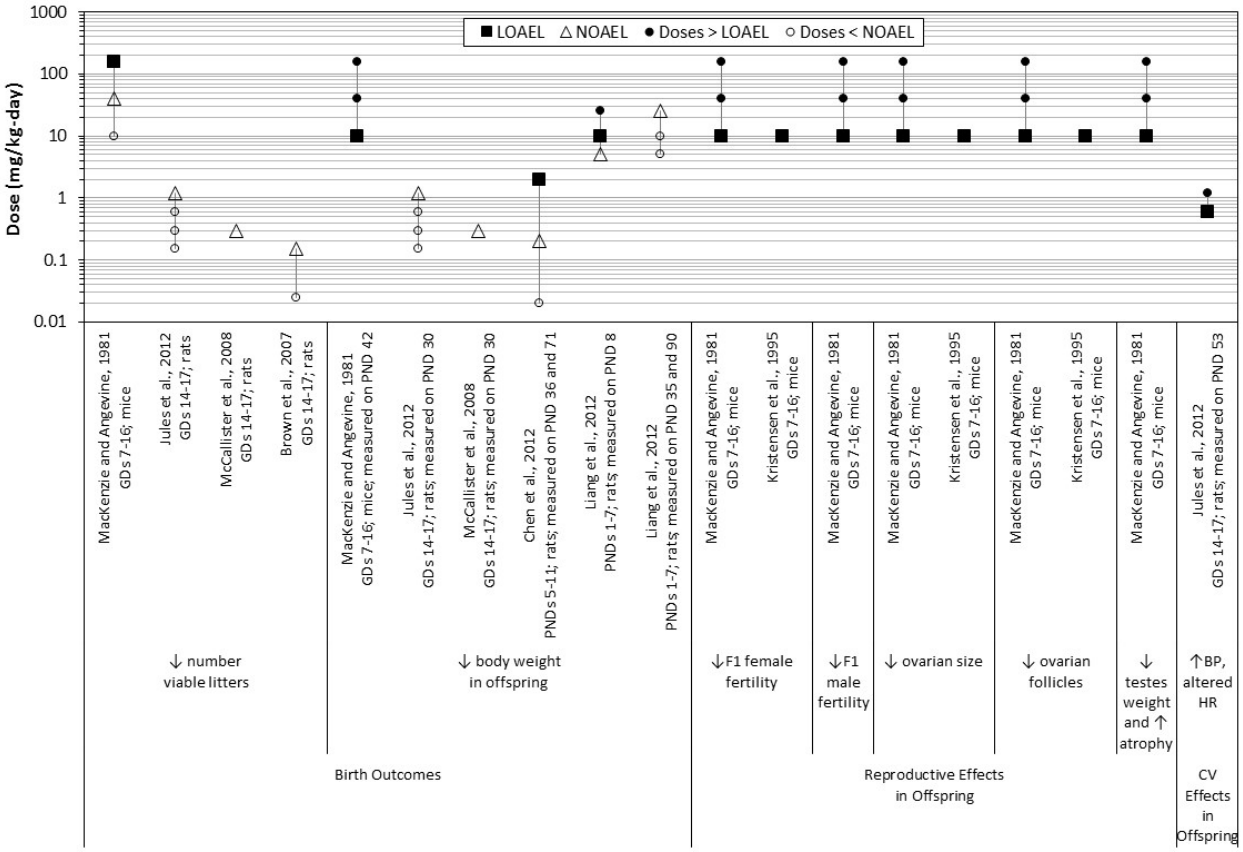

Figure 1-1. Exposure-response array for developmental effects following oral exposure to benzo[a]pyrene.

Neurodevelopmental Effects

Development of the human (and rodent) nervous system is a prolonged process that begins around mid-gestation and continues in one form or another into early adulthood. Thus, it is difficult to pinpoint an exact age at which exposure is no longer interpreted as having the potential to cause neurodevelopmental effects. The majority of oral and inhalation studies examining the potential for benzo[a]pyrene to cause nervous system effects assessed children or rodents after gestational or early neonatal exposure; thus, these data are discussed in the context of developmental effects. One subchronic intraperitoneal (i.p.) exposure study (Tang et al., 2011) beginning at weaning but with most of the exposure in adulthood is also discussed in the context of developmental effects, as pronounced changes in brain development (e.g., myelination and synaptic refinement/ strengthening) occur during adolescence.

Additional data on potential nervous system effects involved benzo[a]pyrene exposure in sexually mature, adult humans and rodents, or in animals exposed beginning around puberty (i.e., approximately 5 weeks of age). Although these latter studies included a brief period of pubertal exposure which may be interpreted as relevant to neurodevelopment, nearly all of the exposure occurred after the animals reached sexual maturity. Thus, studies in human adults or in rodents

beginning after 5 weeks of age are discussed in Section 1.1.4 in the context of nervous system effects interpreted to result from "adult" exposure. A summary of the exposure paradigms used in the database of studies evaluating the potential nervous system effects of benzo[a]pyrene exposure is illustrated in Figure 1-2.

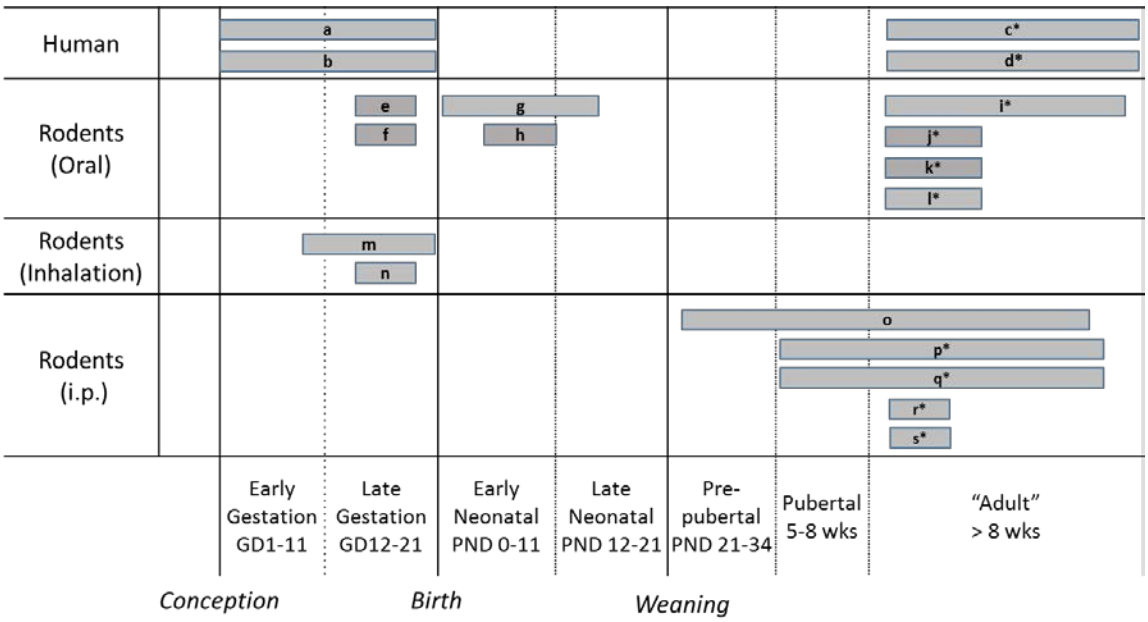

Figure 1-2. Exposure timing in benzo[a]pyrene studies examining nervous system effects.

The available human and animal studies are presented (Note: exposure durations are not to scale): (a) Perera et al. (2012a); Tang et al. (2006); Tang et al. (2008); (b) Perera et al. (2004); Perera et al. (2005b); Perera et al. (2012b); (c) Qiu et al. (2013); (d) Niu et al. (2010); (e) Sheng et al. (2010); (f) McCallister et al. (2008); (g) Bouayed et al. (2009a); (h) Chen et al. (2012); (i) Chengzhi et al. (2011); (j) Bouayed et al. (2009b); (k) Bouayed et al. (2012); (l) Maciel et al. (2014); (m) Wormley et al. (2004); (n) Li et al. (2012); (o) Tang et al. (2011); (p) Xia et al. (2011); (q) Qiu et al. (2011); (r) Grova et al. (2007); (s) Grova et al. (2008). * = studies discussed separately in Section 1.1.4.

There is evidence in humans and animals that benzo[a]pyrene induces developmental neurotoxicity. Two epidemiology studies that examined benzo[a]pyrene-specific measures observed effects on neurodevelopment and behavior in young children. Altered learning and memory, motor activity, anxiety-like behavior, and electrophysiological changes have also been observed in animals following developmental oral and inhalation exposure to benzo[a]pyrene.

Important developmental processes such as neurogenesis and migration are largely completed during mid-late gestation in humans and rodents, the disruption of which can have serious adverse consequences. Next, the mammalian brain undergoes periods of rapid brain growth, particularly during the last 3 months of pregnancy in humans, which has been compared to the first 1–2 weeks of life in the rat and mouse neonate (Dobbing and Sands, 1979, 1973). This period is characterized by axonal and dendritic outgrowth and the establishment of mature

neuronal connections. Also during this critical period, animals acquire many new motor and sensory abilities (Kolb and Whishaw, 1989). Complementing the behavioral effects reported in observational studies of exposed humans, there is a growing literature of animal studies that shows changes in motor and cognitive function following acute or repeated perinatal or early neonatal exposure to benzo[a]pyrene (Bouayed et al., 2009a; McCallister et al., 2008; Wormley et al., 2004). These effects are described below.

Cognitive function

Head circumference at birth is associated with measures of intelligence in children, even among term infants (Broekman et al., 2009; Gale et al., 2006). The two birth cohort studies that examined maternal or cord blood levels of benzo[a]pyrene-specific DNA adducts in relation to head circumference provide some evidence of an association, most strongly within the context of an interaction with ETS (Tang et al., 2006; Perera et al., 2005b; Perera et al., 2004) (Table 1-3). In these studies, the internal benzo[a]pyrene-specific dose metrics were in general agreement with personal air sampling monitors worn by the mothers (Perera et al., 2012b). The cohort in Tongliang, China also examined intelligence quotient scores at age 5 years (Perera et al., 2012a). An interaction with ETS was seen in this analysis, with larger decrements seen on the full scale and verbal scales with increased benzo[a]pyrene-DNA adduct levels in the presence of prenatal exposure to ETS compared to the effects seen in the absence of prenatal exposure.

Animal studies have also provided evidence of altered behaviors in learning and memory tests following lactational or direct postnatal oral exposure to benzo[a]pyrene (Chen et al., 2012; Bouayed et al., 2009a) (Table 1-4). In mice, spatial working memory was measured using the Y-maze spontaneous alternation test (Bouayed et al., 2009a). This test records alternations between arm entries in a Y-shaped maze as a measure of memory, as rodents typically prefer to investigate a new arm of the maze. To a lesser extent, this test can also reflect changes in sensory processing, novelty preference, and anxiety-related responses in rodents. An improvement in performance was evident in mice, as exhibited by significant increases in spontaneous alternations in the Y-maze test in mice on PND 40 following lactational exposure to 2 mg/kg-day benzo[a]pyrene (but not 20 mg/kg-day) from PND 0 to 14 (Bouayed et al., 2009a). The total number of arm entries in the Y-maze was unaffected by lactational exposure, suggesting that changes in motor function were not driving this response. In another test of short-term memory, wild type $CPR^{lox/lox}$ transgenic mice (a conditional knock-out mouse model designed to allow for tissue-specific deletion of CYP450 reductase) exhibited deficiencies in novel object recognition tests months after in utero oral or inhalation exposure to benzo[a]pyrene during late gestation (Li et al., 2012; Sheng et al., 2010). These findings suggest impairment in short-term memory, although these tests also reflect locomotor exploratory behavior, response to novelty, and attention.

In rats, spatial learning and memory was investigated using the Morris water maze, which measures the ability of a rat to navigate to a target platform using external spatial cues (Chen et al., 2012; Tang et al., 2011). Increased escape latency (time to find the hidden platform), as well as

decreased time in the target quadrant and decreased number of platform crossings during a probe trial with the platform removed were observed in PND 39–40 rats following postnatal oral exposure to 2 mg/kg-day benzo[a]pyrene (Chen et al., 2012). These effects were more pronounced in animals tested at PNDs 74–75, with effects observable at ≥0.2 mg/kg-day. No difference in swim speed was observed during the probe trial tests (swim speed did not appear to be analyzed during the hidden platform trials) between treatment groups, suggesting that the observed changes are not attributable to general motor impairment. Similar findings in Morris water maze performance were observed in rats exposed to 2.5 and 6.25 mg/kg (but not at 1 mg/kg) benzo[a]pyrene by daily i.p. exposure beginning at weaning and continuing for 14 weeks, although swim speed or distance traveled were not measured (Tang et al., 2011). In both rat studies, these decrements in Morris water maze performance were not attributable to effects of benzo[a]pyrene exposure on learning and memory processes alone. Specifically, visual examinations of the improvements in escape latency (slopes) over the four-to-five learning trial days were not noticeably affected by treatment dose, suggesting that all groups learned at a similar rate, despite treated animals displaying a baseline impairment in performance from testing day 1. Typically, these non-specific differences in latency on trial day 1 reflect noncognitive influences on performance (Cory-Slechta et al., 2001). In Chen et al. (2012), four trials were conducted per day and averaged for each animal at each trial day (data for these multiple daily trials were not provided). Thus, it is unclear whether the dose-related increases in escape latency already observable at trial day 1 reflect effects on learning across those same day trials or other effects that were already present prior to testing (e.g., altered motor function, anxiety, vision, etc.); however, the study by Tang et al., also noted a baseline impairment in water maze performance and this study only tested animals once per day suggesting that the decrements present on testing day 1 in Chen et al. (2012) did not reflect impaired learning during the first trial day.

Performance in the Morris water maze probe trials was also affected by benzo[a]pyrene exposure (Chen et al., 2011; Tang et al., 2011); however, as it is not clear that the groups learned to a comparable extent in hidden platform tests prior to testing memory retention, it is unclear how to interpret these results.

Overall, the dose-dependent decreases in the performance of the benzo[a]pyrene-exposed rats in the Morris water maze hidden platform trials were shown to persist for weeks to months after post-natal exposure in two separate cohorts of male and female rats and, although the behavioral effect cannot clearly be attributed to learning or memory, these data represent a persistent and adverse neurobehavioral change.

Neuromuscular function, coordination, and sensorimotor development

Motor behavior and coordination, assessed by locomotion, reaching, balance, comprehension, drawing, and hand control was one of the specific domains assessed in the Chinese birth cohort evaluated by Tang et al. (2008). In children aged 2 years, decreased scores were seen in relation to increasing benzo[a]pyrene- DNA adducts measured in cord blood, with a Beta per unit

increase in adducts of −16 (p = 0.04), and an approximate twofold increased risk of development delay per unit increase in adducts (Table 1-3).

In laboratory animals (Table 1-4 and Figure 1-3), impaired performance in neuromuscular and sensorimotor tests have been consistently observed in mice lactationally exposed to ≥2 mg/kg-day benzo[a]pyrene from PND 0 to 14 (Bouayed et al., 2009a) and in rat pups postnatally exposed to ≥0.02 mg/kg-day benzo[a]pyrene from PND 5 to 11 (Chen et al., 2012). In the righting reflex test, significant increases in righting time were observed in PND 3−5 (but not PND 7−9) mice and in PND 12−16 (but not PND 18) rats. These decrements did not show a monotonic dose response. In another test of sensorimotor function and coordination, dose-dependent increases in latency in the negative geotaxis test were observed in PND 5−9 (but not PND 11) mice and in PND 12−14 (but not PND 16−18) rats. The forelimb grip strength test of neuromuscular strength was also evaluated in both mice and rats, but alterations were only observed in mice. In mice, a dose-dependent increase in duration of forelimb grip was observed on PNDs 9 and 11 during lactational exposure to benzo[a]pyrene. The Water Escape Pole Climbing test was also used to evaluate neuromuscular function and coordination in mice (Bouayed et al., 2009a). No effect on climbing time was observed, suggesting no change in muscle strength. However, increased latency in pole grasping and pole escape in PND 20 male pups was observed, highlighting potential decrements in visuomotor integration and/or coordination, although anxiety or fear-related responses cannot be ruled out. Treatment-dependent increases in pup body weight around the testing period complicate the interpretation of these results.

Chen et al. (2012) observed statistically significant delays on the order of ~0.2−0.3 seconds in the surface righting test and ~3−4 seconds in the negative geotaxis test. The authors found no effect of gender, therefore the data for male and female rats were pooled for these measures. However, it should be noted that differences in the maturation of these developmental landmarks following challenge have been shown to exist between males and females. In the surface righting test, differences between groups were generally 0.5 seconds or less; as these measures were manually recorded, it is unclear how reliably these small differences could be detected. An additional uncertainty in interpreting these results involves the lack of consideration of litter effect (i.e., the tendency of littermates to respond more like each other than like offspring in other litters), such as through identifying the litter for each individual animal. Consistent with the results of Bouayed et al. (2009a) in mice, Chen et al. (2012) reported effects in these tests at earlier postnatal ages that did not persist when tested at later postnatal ages. Negative geotaxis and surface righting are discrete endpoints routinely used as part of a neurobehavioral test battery to assess acquisition of behavioral reflexes. Chen et al. (2012) used the surface righting and negative geotaxis tests as quantitative measures of sensorimotor function at PND 12 and beyond. Typically in these tests, animals are observed on consecutive days (e.g., PNDs 3−12) and time to acquisition of these phenotypes is measured. Notably, in rats, the functional phenotypes assessed in these assays are largely acquired by the postnatal ages tested by Chen et al. (2012). Likely due to the exposure

window tested by Chen et al. (2012), PNDs 5–11, the authors did not measure performance over consecutive days earlier during development (e.g., PNDs 3–11) in order to identify baseline acquisition of these behaviors. Although these tests as conducted by Chen et al. (2012) cannot discern a developmental delay, the data support a transient impairment of sensorimotor function in animals that have already developed this reflex (e.g., able to orient 180 degrees and able to right within 2 seconds). Thus, taken together with the results of Bouayed et al. (2009a), these data indicate that benzo[a]pyrene may affect sensorimotor function at a particular developmental lifestage(s).

Anxiety and activity

Anxiety/depression and attention/hyperactivity symptoms in children ages 6–7 years were examined via questionnaire in relation to prenatal air monitoring of benzo[a]pyrene and other PAHs, and in relation to benzo[a]pyrene-specific DNA adducts measured at birth in a follow-up of a birth cohort study conducted in New York City (Perera et al., 2012b). PAH exposure levels (based on personal air monitoring, n = 253) and benzo[a]pyrene-specific DNA adducts measured in cord blood samples (n = 138) were both positively associated with symptoms of anxiety/depression and attention problems (see Table 1-3). Given the limited sample size, however, the cord blood results are based on relatively sparse data (<5 in the borderline or clinical range in the low exposure referent group). Associations with maternal blood adducts were similar to or slightly smaller than those seen with cord blood adducts. Exposure was treated as a dichotomy (i.e., for adducts, detectable compared with non-detectable levels) in these analyses.

Decreased anxiety-like behavior was reported in both rats and mice weeks to months following postnatal oral exposure to benzo[a]pyrene (Chen et al., 2012; Bouayed et al., 2009a) (Table 1-4). A decrease in anxiety, indicative of a change in nervous system function, can impair an organism's ability to react to a potentially harmful situation. This decreased ability of an organism to adapt to the environment is considered to be an adverse effect according to EPA's Neurotoxicity guidelines (U.S. EPA, 1998a). Anxiety-like behaviors were measured in both species using an elevated plus maze, where an increase in the time spent in the closed arms of the maze is considered evidence of anxious behavior while an increased time spent in the open arms reflects increased risk taking and/or reduced anxiety. Following lactational exposure to ≥2 mg/kg-day benzo[a]pyrene, mice exhibited significant increases in the percent open arm entries and percent time spent in open arms of the maze, as well as significantly decreased entries into closed arms of the maze (the latter was observed at 2, but not 20, mg/kg-day), on PND 32 (Bouayed et al., 2009a). To rule out potential differences in total activity or general motivation and exploration, the authors expressed the open arm data as percentages, and they also demonstrated that there were no exposure-related effects on the total number of arm entries. The mice also exhibited decreased latency of the first entry into an open arm following lactational exposure to 20 mg/kg-day benzo[a]pyrene. Similar results were reported for rats, with decreased anxiety-like behavior following oral benzo[a]pyrene exposure from PND 5 to 11, although sex-specific differences were

observed (Chen et al., 2012). In females, postnatal exposure to ≥0.2 mg/kg-day benzo[a]pyrene was associated with a significant increase in the number of open arm entries and a significant decrease in the number of closed arm entries on PND 70. Significantly increased time in open arms of the maze was reported in PND 70 female rats following postnatal exposure to ≥0.02 mg/kg-day. Less sensitive effects (i.e., observable at 2 mg/kg-day) were observed in a second cohort of exposed rats tested at PND 35. Male rats also showed decreased anxiety-like behavior on PND 70, although the doses needed to detect these responses were higher than females (i.e., increases at ≥2 mg/kg-day for open arm entries and ≥0.2 mg/kg-day for time spent in open arms). A significant decrease in latency to enter an open arm of the maze was observed in both male and female rat pups exposed to 2 mg/kg-day benzo[a]pyrene. Similar to the observations in mice, exposure did not appear to have an effect on total activity or general motivation of the rats, as total arm entries were unchanged by treatment.

Increased spontaneous locomotor activity in the open field on PNDs 34 and 69 has been reported in two cohorts of rats postnatally exposed to 2 and ≥0.2 mg/kg-day, respectively (Chen et al., 2012), but not in mice exposed lactationally to doses up to 20 mg/kg-day and tested on PND 15 (Bouayed et al., 2009a). Interestingly, no differences in the open field test were observed in rats that were postnatally exposed and tested on PNDs 18 and 20, suggesting either that longer latencies between exposure and testing may be required, or that these developmental effects may only manifest in more mature rats (Chen et al., 2012). An apparent increased sensitivity of older animals was also present in the elevated plus maze and Morris water maze tests performed by Chen et al. (2012). Both Chen et al. (2012) and Bouayed et al. (2009a) measured total horizontal and vertical activity, increases in which could indicate increased motor activity or decreased anxiety (less fear of the open spaces/bright lights). However, the relative contributions of these two behavioral components to the increased activity observed by Chen et al. (2012) could not be separated, as the authors did not evaluate activity in central versus peripheral regions of the field (i.e., anxious rodents will spend less time in the center of the field).

Electrophysiological changes

Electrophysiological effects of gestational exposure to benzo[a]pyrene have been examined in two studies (by the same research group) through implanted electrodes in the rat cortex and hippocampus, and using in vitro preparations after in vivo exposure (Li et al., 2012) (Table 1-4). Maternal inhalation exposure to 0.1 mg/m^3 resulted in reduced long-term potentiation in the dentate gyrus of male offspring between PND 60 and 70 (Wormley et al., 2004), and decreased inward currents in cortical neurons isolated on PND 1 and cultured for 7 days (Li et al., 2012); however, significant embryo/fetal resorptions at this exposure level and uncertainties in extrapolating from the ex vivo preparations complicates the interpretation of these results. Oral exposure of dams to 0.3 mg/kg-day for 4 days during late gestation resulted in decreased evoked neuronal activity in male offspring following mechanical whisker stimulation between PND 90 and 120 (McCallister et al., 2008). Specifically, the authors noted reduced spike numbers in both short

and long latency responses following whisker stimulation. These effects were observed several months post-exposure, suggesting that gestational benzo[a]pyrene exposure may have long-lasting functional effects on neuronal activity elicited by sensory stimuli.

Table 1-3. Evidence pertaining to the neurodevelopmental effects of benzo[a]pyrene from PAH mixtures

Reference and study design	Results
Tang et al. (2008); Tang et al. (2006) (Tongliang, China) Birth cohort 150 nonsmoking women, delivered March 2002–June 2002; lived within 2.5 km of power plant that operated from December 2001 to May 2002 Outcomes: Head circumference at birth; Gesell Developmental Schedule, administered by physicians at 2 yrs of age (four domains: motor, adaptive, language, and social); standardized mean score = 100 ± SD 15 (score <85 = developmental delay) Exposure: Mean hrs/d exposed to ETS 0.42 (SD 1.19); lived within 2.5 km of power plant that operated from December 2001 to May 2002; benzo[a]pyrene-DNA adducts from maternal and cord blood samples; cord blood mean 0.33 (SD 0.14) (median 0.36) adducts/10^{-8} nucleotides; maternal blood mean 0.29 (SD 0.13) adducts/10^{-8} nucleotides	Relation between cord blood benzo[a]pyrene-DNA adducts and log-transformed head circumference Beta (*p*-value) Birth: −0.011 (0.057) 18 mo: −0.012 (0.085) 24 mo: −0.006 (0.19) 30 mo: −0.005 (0.31) High versus low, dichotomized at median, adjusted for ETS, sex of child, maternal height, maternal weight, Cesarean section delivery, maternal head circumference, and gestational age (for measures at birth)
Tang et al. (2008); Tang et al. (2006) (see above for population and exposure details) n = 110 for Developmental Quotient analysis; no differences between the 110 participants in this analysis and the nonparticipants with respect to maternal age, gestational age, birth weight, birth length, or birth head circumference; higher maternal education was suggested (direction of suggestive association not reported, *p* = 0.056)	Association between benzo[a]pyrene adducts and development Motor: −16.0 (−31.3, −0.72)* / 1.91 (1.22, 2.97)* Adaptive: −15.5 (−35.6, 4.61) / 1.16 (0.76, 1.76) Language: −16.6 (−33.7, 0.46) / 1.31 (0.84, 2.05) Social: −9.29 (−25.3, 6.70) / 1.52 (0.93, 2.50) Average: −14.6 (−28.8, −0.37)* / 1.67 (0.93, 3.00) [a]Linear regression of change in Developmental Quotient per unit increase in benzo[a]pyrene adducts
	Beta (95% CI)[a] / OR (95% CI)[b]

Reference and study design	Results
Outcomes: Gesell Developmental Schedule, administered by physicians at 2 yrs of age (four domains: motor, adaptive, language, and social); standardized mean score = 100 ± SD 15 (score <85 = developmental delay)	[b]Logistic regression of risk of developmental delay (defined as normalized score <85) per 1 unit (0.1 adducts/10^{-8} nucleotides) increase in adducts Both analyses adjusted for sex, gestational age, maternal education, ETS, and cord lead levels
Perera et al. (2012a); Tang et al. (2008); Tang et al. (2006) (see above for population and exposure details) 132 (83%) followed through age 5; 100 of these had complete data for analysis; no differences between the 100 participants in this analysis and the nonparticipants with respect to adduct levels, ETS exposure, IQ measures, maternal age, gestational age, or infant gender; higher maternal education (60 and 35% with ≥ high school, respectively, in participants and nonparticipants, $p < 0.05$) Outcomes: Wechsler Preschool and Primary Intelligence Quotient scale (Shanghai version)	ETS measure not correlated with benzo[a]pyrene adduct measures (i.e., absolute value of Spearman r < 0.10) Relation between cord blood benzo[a]pyrene-DNA adducts, ETS exposure, and IQ measures Beta (95% CI) Main effect / With ETS interaction term Full scale: −2.42 (−7.96, 3.13) / −10.10 (−18.90, −1.29) Verbal: −1.79 (−7.61, 4.03) / −10.35 (−19.61, −1.10) Performance: −2.57 (−8.92, 3.79) / −7.78 (−18.03,2.48) Beta per 1 unit increase in log-transformed cord adducts, adjusted for ETS exposure, gestational age, maternal education, cord lead, maternal age, and gender
Perera et al. (2012b); Perera et al. (2005b); Perera et al. (2004) (United States, New York) Birth cohort 265 pregnant women: African-American and Dominican nonsmoking women who delivered babies between April 1998 and October 2002 (253 and 207 for behavior and head circumference analysis, respectively); approximately 40% with a smoker in the home Outcomes: Head circumference at birth Exposure: Benzo[a]pyrene-DNA adducts from maternal and cord blood samples; mean 0.22 (SD 0.14) adducts/10^{-8} nucleotides; median of detectable values 0.36 adducts/10^{-8} nucleotides	Relation between cord blood benzo[a]pyrene-DNA adducts, ETS exposure, and log-transformed head circumference Beta (*p*-value) Interaction term: −0.032 (0.01) benzo[a]pyrene-DNA adducts: −0.007 (0.39) ETS in home: −0.005 (0.43) High versus low, dichotomized at 0.36 adducts/10^{-8} nucleotides, adjusted for ethnicity, sex of newborns, maternal body mass index, dietary PAHs, and gestational age

Reference and study design	Results
Perera et al. (2012b) n = 215 with outcome data and no missing covariate data); no differences between the participants in this analysis and the nonparticipants with respect to adduct levels, ETS exposure, maternal age, gestational age, and socioeconomic variables; participants more likely to be female and African-American Outcomes: Child Behavior Checklist (118 items), completed by mothers for children ages 6–7 yrs. Two domains: anxious/depression, attention problems (normalized T-score ≤65 = borderline or clinical syndrome); also used for scales of anxiety problems and attention deficit hyperactivity problems based on DSM classification	Logistic regression of risk of borderline or clinical status in relation PAH levels and to detectable levels of benzo[a]pyrene adducts PAH: Anxious/depressed — Prevalence 6.3 %; OR (95% CI) 8.9 (1.7, 46.5); Cord blood OR (95% CI) 2.6 (0.69, 9.4) Attention problems — 6.7%; 3.8 (1.1, 12.7); 4.1 (0.99, 16.6) Anxiety (DSM) — 9.5%; 4.6 (1.5, 14.3); 2.5 (0.84, 7.7) Attention deficit – hyperactivity (DSM) — 7.9%; 2.3 (0.79, 6.7); 2.6 (0.68, 10.3) Exposure dichotomized for PAH as above and below median (2.273 ng/m^3) for parent population and for cord blood benzo[a]pyrene adducts as detectable (n = 56 cord blood samples) versus non-detectable (n = 92); adjusted for sex, gestational age, maternal education, maternal IQ, prenatal ETS, ethnicity, age, heating season, prenatal demoralization, and HOME inventory

*Statistically significantly different from the control ($p < 0.05$).

DSM = Diagnostic and Statistical Manual of Mental Disorders; HOME = Home Observation for Measurement of the Environment; IQ = intelligence quotient.

Table 1-4. Evidence pertaining to the neurodevelopmental effects of benzo[a]pyrene in animals

Reference and study design	Results[a]
Cognitive function	
Chen et al. (2012)[b] Sprague-Dawley rats, 20 pups (10 male and 10 female)/group 0, 0.02, 0.2, or 2 mg/kg-d by gavage PNDs 5–11	Hidden Platform test in Morris water maze, escape latency: Adolescent test period (PNDs 36–39): significant increase at 2 mg/kg-d only Adult test period (PNDs 71–74): significant increase in at ≥0.2 mg/kg-d Increases in latency were already ~30% greater than controls at 2 mg/kg-d on the first trial day (i.e., on PND 36 or 71) All experimental groups exhibited similar improvements in escape latency, as slopes were parallel across the 4 trial days Probe test in the Morris water maze (trial day 5): Time spent in the target quadrant: PND 40: significant decrease at 2 mg/kg-d only PND 75: significant decrease at ≥0.2 mg/kg-d Number of platform crossings: PND 40: significant decrease at 2 mg/kg-d only

Reference and study design	Results[a]
	PND 75: significant decrease at ≥0.2 mg/kg-d (in females) and 2 mg/kg-d (in males)
Bouayed et al. (2009a) Female Swiss albino mice, 5/group 0, 2, or 20 mg/kg-d maternal gavage PNDs 0−14 (lactational exposure)	Significant increase in the percent of spontaneous alternations in the Y-maze alternation test at 2 mg/kg-d but not at 20 mg/kg-d No effect on the total number of arm entries in the Y-maze alternation test
Neuromuscular function, coordination, and sensorimotor development	
Chen et al. (2012)[b] Sprague-Dawley rats, 20 pups (10 male and 10 female)/group 0, 0.02, 0.2, or 2 mg/kg-d by gavage PNDs 5−11	Latency in the surface righting reflex test PND 12: significant increase at 0.2 mg/kg-d only PND 14: significant increase at 0.02 and 2 mg/kg-d only PND 16: significant difference at 2 mg/kg-d only PND 18: no significant difference Latency in the negative geotaxis test PND 12: significant increase at all doses PND 14: significant increase at 2 mg/kg-d only PNDs 16 and 18: no significant difference No effect on duration of forelimb grip in forelimb grip strength test No effect on the latency to retract from the edge in cliff aversion test Note: Males and females were pooled for all analyses
Bouayed et al. (2009a) Female Swiss albino mice, 5/group 0, 2, or 20 mg/kg-d by maternal gavage PNDs 0−14 (lactational exposure)	Significant increase in righting time in the surface righting reflex test at both doses on PNDs 3 and 5 (but not PNDs 7 and 9) Significant increase in latency in the negative geotaxis time for 20 mg/kg-d dose group at PNDs 5, 7, and 9 (no significant difference at PND 11) Significant increase in duration of forelimb grip in forelimb grip strength test at both dose groups on PND 9 (statistically significant at PND 11 only at high dose) Significant increase in pole grasping latency in male pups in the water escape pole climbing test at 20 mg/kg-d No effect on climbing time in the water escape pole climbing test Significant increase in pole escape latency in the water escape pole climbing test in male rats at 20 mg/kg-d
Anxiety and/or motor activity	
Chen et al. (2012)[b] Sprague-Dawley rats, 20 pups (10 male and 10 female)/group 0, 0.02, 0.2, or 2 mg/kg-d by gavage PNDs 5−11	Elevated plus maze: Number of entries into open arms: PND 35: No difference PND 70: >26% increase at ≥0.2 mg/kg-d (females) and 38% 2 mg/kg-d (males)

Reference and study design	Results[a]
	Significant decrease in the number of entries into closed arms at PND 70 at ≥0.2 mg/kg-d (in females) and 2 mg/kg-d (in males) (no difference at PND 35) Significant increase in the time spent in open arms at PND 35 at 2 mg/kg-d in females and at PND 70 at doses ≥0.02 mg/kg-d in females and ≥0.2 mg/kg-d in males Significant decrease in latency time to first enter an open arm on PND 70 at ≥0.2 mg/kg-d (no difference at PND 35) No effect on the total number of arm entries between treatment groups (calculated by EPA from graphically reported open and closed arm entries) Open field test: Significant increase in the number of squares: PND 34, significant increase at 2 mg/kg-d; PND 69, significant increase at ≥0.02 mg/kg-d (no difference at PNDs 18 and 20) Significant increase in rearing activity at 0.2 mg/kg-d on PND 69 (no difference at PNDs 18, 20, and 34)
Bouayed et al. (2009a) Female Swiss albino mice, 5/group 0, 2, or 20 mg/kg-d by maternal gavage PNDs 0−14 (lactational exposure)	Elevated plus maze: Significantly increased time in open arms at ≥2 mg/kg-d Significantly increased percentage of entries into open arms at ≥2 mg/kg-d Significantly decreased entries into closed at 2 mg/kg-d, but not at 20 mg/kg-d Significantly decreased latency time to enter an open arm at 20 mg/kg-d No effect on the total number of arm entries No significant effect of gender on performance was detected, so males and females were pooled for analyses Open field test: No significant change in activity on PND 15, but data not provided
Electrophysiological changes	
McCallister et al. (2008) Long-Evans Hooded rats, 5−6/group 0 or 0.3 mg/kg-d by gavage GDs 14−17	Statistically significant decreases in stimulus-evoked cortical neuronal activity on PNDs 90−120 Reduction in the number of spikes in both the short and long latency periods on PNDs 90−120 (numerical data not presented)

Reference and study design	Results[a]
Wormley et al. (2004) F344 rats, 10 females/group 0 or 100 µg/m^3 by nose-only inhalation for 4 hrs/d GDs 11–21	Electrophysiological changes in the hippocampus (PNDs 60–70): Consistently lower long term potentiation following gestational exposure (statistical analysis not reported) % change relative to control: −26% Note: significant decrease in embryo/fetal survival observed (99% in controls versus 34% in treated group)

[a]% change from control calculated as: (treated value − control value)/control value × 100.
Significant denotes statistical significance; instances of statistical significance ($p < 0.05$) as reported by study authors.
[b]Authors used the Least Significant Difference (LSD) test which can inflate statistical significance. Magnitudes of effect and overall biological significance were also considerations in developing the weight of evidence across outcomes and studies.

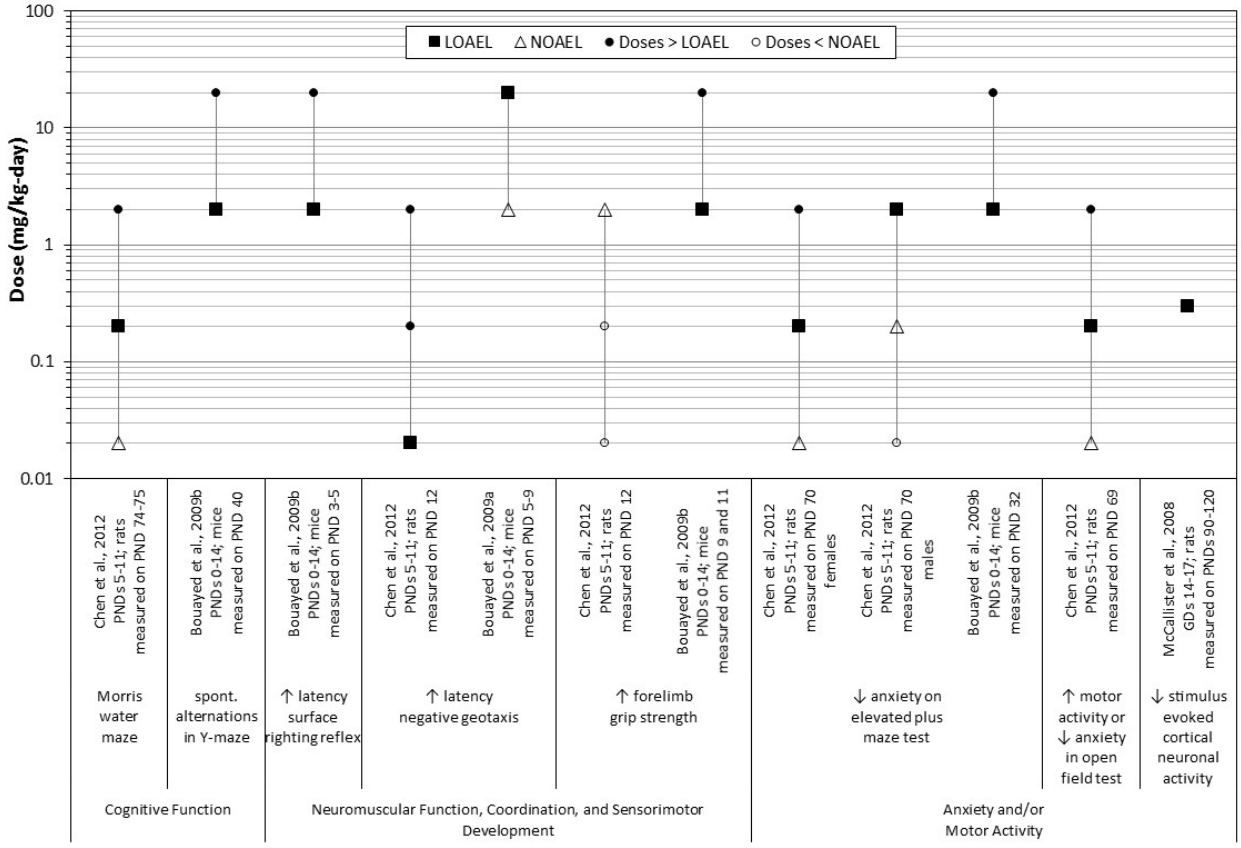

Figure 1-3. Exposure-response array for neurodevelopmental effects following oral exposure.

Mode of Action Analysis—Developmental Toxicity and Developmental Neurotoxicity

Data regarding the potential mode of action for the various manifestations of developmental toxicity associated with benzo[a]pyrene exposure are limited, and the mode of action for

developmental toxicity is not known. General hypothesized modes of action for the various observed developmental effects include, but are not limited to, genotoxicity and mutagenicity, altered cell signaling (e.g. through the Ah receptor), cytotoxicity, and oxidative stress.

Benzo[a]pyrene is well established as a mutagen (see Section 1.1.5). It is therefore plausible that exposure could result in mutations in male and female germ cells, as well as fetal tissues leading to decreased viability, birth defects, and altered development in offspring. An increasing body of information suggests that genotoxicity (including fragmentation and strand breaks) in male germ cells can lead to decreased embryo viability post-fertilization (Borini et al., 2006; Seli et al., 2004).

It is plausible that developmental effects of benzo[a]pyrene may be mediated by altered cell signaling through the AhR. Benzo[a]pyrene is a ligand for the AhR, and activation of this receptor regulates downstream gene expression including the induction of CYP enzymes important in the conversion of benzo[a]pyrene into reactive metabolites. Studies in AhR knock-out mice indicate that AhR signaling during embryogenesis is essential for normal liver, kidney, vascular, hematopoietic, and immune development (Schmidt et al., 1996; Fernandez-Salguero et al., 1995). In experiments in AhR responsive and less-responsive mice, the mice with the less-responsive AhR were protected from renal injury as adults following gavage treatment with 0.1 or 0.5 mg/kg-day benzo[a]pyrene from GD 10 to 13. Renal injury was indicated by an increase in urinary albumin and a decrease in glomerular number (Nanez et al., 2011). Another study at much higher doses (200 mg/kg-day by i.p. on GD 7, 10, or 12) found increased developmental effects in AhR responsive C57BL/6 mice as compared to nonresponsive AhR AKR mice (Shum et al., 1979). Specifically, resorptions, malformations, and congenital abnormalities as well as decreased fetal bodyweight were observed more commonly in AhR responsive mice. Similar findings were observed in a developmental toxicity study of two PAHs (3MC and DMBA) in mice, with increases in stillborns, resorptions, and malformations in AH responsive strains, indicating mechanisms of developmental toxicity may be related to AhR signaling (Nebert et al., 1977).

Low birth weight has been associated with prenatal exposure to PAHs in human populations (Duarte-Salles et al., 2013; Duarte-Salles et al., 2012; Perera et al., 2005b). Several epidemiology studies have revealed an inverse association between low birth weight and increased blood pressure, hypertension, and measures of decreased renal function as adults (Zandi-Nejad et al., 2006). It has been hypothesized that this may be attributable to a congenital nephron deficit associated with intrauterine growth restriction (Nanez et al., 2011; Zandi-Nejad et al., 2006).

Oxidative stress, through oxidative DNA and protein damage induced through reactive metabolites of benzo[a]pyrene, has also been proposed as a contributing mechanism for various developmental effects (as reviewed by Wells et al., 1997). Studies in GSH deficient mice suggest that oxidative stress may mediate male reproductive effects observed after developmental exposure (Nakamura et al., 2012). Some evidence also indicates increased susceptibility to oxidative lung damage in rats exposed to relatively high intraperitoneal doses of

benzo[a]pyrene during gestation (Thakur et al., 2014). Several studies investigating mechanisms of developmental neurotoxicity (Li et al., 2012; Sheng et al., 2010) also indicate a role for increases in brain oxidative stress as a contributor to behavioral changes elicited by developmental exposure to benzo[a]pyrene (discussed further below).

No clear mode(s) of action for the observed neurodevelopmental changes following benzo[a]pyrene exposure have been demonstrated. General hypothesized mechanisms with limited support are related to altered central nervous system neurotransmission. These mechanisms involve altered neurotransmitter gene expression, neurotransmitter levels, and neurotransmitter receptor signaling in regions associated with spatial learning, anxiety, and aggression, such as the hippocampus, striatum, amygdala, and hypothalamus (Li et al., 2012; Qiu et al., 2011; Tang et al., 2011; Xia et al., 2011; Bouayed et al., 2009a; Grova et al., 2008; Brown et al., 2007; Grova et al., 2007; Stephanou et al., 1998).

Mechanistic studies in rodents exposed as adults, which exhibit some of the same behavioral changes as animals exposed during development (see Section 1.1.4), may also inform potential mode(s) of action for the observed neurodevelopmental changes. Specifically regarding potential changes in spatial learning and memory processes, multiple studies in developing (Li et al., 2012; Tang et al., 2011; McCallister et al., 2008; Wormley et al., 2004); and adult (Maciel et al., 2014; Qiu et al., 2013; Tang et al., 2011; Grova et al., 2008; Grova et al., 2007) rodents suggest that changes in N-methyl-D-aspartate (NMDA) receptor signaling seen with benzo[a]pyrene exposure (e.g., changes in expression patterns of NR2A and NR2B subunits) and possibly related effects on synapse strength and long-term potentiation, may be responsible for behavioral effects in tests of learning and memory.

Alternatively, or perhaps in concert with changes in NMDAR signaling, a series of experiments by the same lab, using mice exposed to benzo[a]pyrene by inhalation (Li et al., 2012) or gavage (Sheng et al., 2010) during late gestation (i.e., GDs 14–17), indicate a role for increases in brain oxidative stress (possibly due to oxidative metabolites of benzo[a]pyrene) as a contributor to the persistent behavioral changes elicited by developmental exposure. In the wild type offspring, benzo[a]pyrene exposure induced changes in brain markers of glutamate-associated neurotransmission (i.e. levels of Sp4 transcription factor and the Sp4 target, NR2A in neonates and levels of glutamate at PND 100) and oxidative stress (i.e. neonatal F_2-isoprostane levels) (Li et al., 2012), which might be related to decrements in short term memory in the novel object test observed at either PND 40 (Sheng et al., 2010) or PND 100 (Li et al., 2012). These changes, including the decrements in short term memory, were largely reversed by knocking out brain NADPH cytochrome P450 reductase (Li et al., 2012), an enzyme believed to be involved in the oxidative metabolism of benzo[a]pyrene. Interestingly, the changes induced by benzo[a]pyrene exposure during late gestation, including increases in oxidative metabolites and associated molecular markers, were generally highest during the period of active synaptogenesis (e.g., PNDs 3–15).

In relation to potential changes in anxiety-like behaviors (and also relevant to effects on learning and memory processes), many commonly used anti-anxiety medications work by increasing brain serotonin levels (e.g., selective serotonin reuptake inhibitors), increasing brain dopamine levels (e.g., dopamine reuptake inhibitors), or by targeting gamma-aminobutyric acid (GABA) receptors (e.g., benzodiazepines). Although $GABA_A$ receptor messenger ribonucleic acid (mRNA) in whole-brain homogenates was unchanged following lactational benzo[a]pyrene exposure, exposure at ≥2 mg/kg-day from PND 1 to 14 caused dose-dependent decreases in serotonin receptor ($5HT_{1A}$) expression (Bouayed et al., 2009a; Stephanou et al., 1998); however with short term oral exposure in adult mice, $5HT_{1A}$ expression was increased at 2 mg/kg-d (although it was unchanged at 20 mg/kg-d; Bouayed et al., 2012). Additional support for identifying alterations in monoamine neurotransmitter signaling (serotonin and dopamine signaling, in particular) as a potential mechanism(s) in the altered anxiety-like behaviors observed following benzo[a]pyrene exposure is provided in multiple studies of rodents exposed as adults (Bouayed et al., 2012; Qiu et al., 2011; Xia et al., 2011; Stephanou et al., 1998; Jayasekara et al., 1992) and in a single study of blood neurotransmitter levels in occupationally-exposed men (Niu et al., 2010). Overall, these data suggest possible effects of benzo[a]pyrene exposure on NMDA receptor expression and regulation of monoamine neurotransmitters including serotonin and dopamine, but these findings require additional studies to clarify and extend understanding of these events.

Summary of Developmental Effects

Developmental effects following in utero exposure to PAH mixtures or benzo[a]pyrene alone have been reported in humans and in animal models. In human populations, decreased head circumference, decreased birth weight, decreased postnatal weight, as well as increased frequency of miscarriage have been reported. Analogous effects in laboratory animals, including decreased pup weight and increased embryo/fetal resorptions, have been noted following gestational or early postnatal exposure to benzo[a]pyrene by the oral or inhalation route (Chen et al., 2012; Archibong et al., 2002; Mackenzie and Angevine, 1981). Reproductive function is also altered in mice treated gestationally with benzo[a]pyrene (Kristensen et al., 1995; Mackenzie and Angevine, 1981). These effects include impaired reproductive performance in F1 offspring (male and female) and alterations of the weight and histology of reproductive organs (ovaries and testes).

The available human and animal data also support the conclusion that benzo[a]pyrene is a developmental neurotoxicant. Human studies of environmental PAH exposure in two cohorts have observed neurotoxic effects, including suggestions of reduced head circumference (Tang et al., 2006; Perera et al., 2005b; Perera et al., 2004), impaired cognitive ability (Perera et al., 2009; Tang et al., 2008), impaired neuromuscular function (Tang et al., 2008), and increased attention problems and anxious/depressed behavior following prenatal exposure (Perera et al., 2012b). These effects were seen in birth cohort studies in different populations (New York City and China), in studies using specific benzo[a]pyrene measures (i.e., adduct levels measured in cord blood

samples) (Perera et al., 2012b; Tang et al., 2008; Tang et al., 2006; Perera et al., 2005b; Perera et al., 2004). The available evidence from mice and rats also demonstrates significant and persistent developmental impairments following exposure to benzo[a]pyrene. Impaired learning and memory behaviors, decreased anxiety-like behaviors, and impaired neuromuscular function were consistently observed in multiple neurobehavioral tests in two separate species at comparable oral doses, and in the absence of maternal or neonatal toxicity (Chen et al., 2012; Bouayed et al., 2009a).

The available evidence from mice and rats also demonstrates significant and persistent developmental impairments following exposure to benzo[a]pyrene. The most compelling evidence derives from postnatal oral exposure studies in rats and mice testing a battery of behavioral tests and observing consistent deficits in tests of learning, memory, and anxiety-related behaviors, sensorimotor development, and neuromuscular function, often with effects observed across multiple parameters tested for each of the individual behavioral tests (Chen et al., 2012; Bouayed et al., 2009a). Despite the differences in design of these two studies (e.g., different species; lactational exposure versus direct gavage; etc.), the results were largely in agreement, with effects observed at comparable oral doses generally at ≥0.2 mg/kg-day (Chen et al., 2012) and at ≥2 mg/kg-day (Bouayed et al., 2009a) and in the absence of maternal or neonatal toxicity. In these studies, behavioral alterations were detectable from early neonatal ages through adulthood, including treatment-related changes in weanlings, pre-pubertal juveniles, and sexually mature animals, which suggests that the neurotoxic effects of developmental benzo[a]pyrene exposure may be irreversible. The findings of Chen et al. (2012) and Bouayed et al. (2009a) following early postnatal exposure (i.e., PNDs 5−11 and 0−14, respectively) are supported by a single i.p. exposure study in weanling rats demonstrating Morris water maze decrements (similar to Chen et al., 2012; Tang et al., 2011), and by a number of more limited studies (e.g., typically single dose) in rats and mice which observed persistent decrements in short term memory and electrophysiological changes after oral or inhalation exposure during late gestation (Li et al., 2012; Sheng et al., 2010; McCallister et al., 2008; Wormley et al., 2004).

While both Chen et al. (2012) and Bouayed et al. (2009a) provide evidence of neurotoxicity across a variety of traditional behavioral assays, there were notable strengths and weaknesses of each study. Chen et al. (2012) was a fairly large study, with 40 litters of rats evaluated in multiple behavioral tests at various ages. The study was designed with the intended goal of reducing bias, including consideration of litter effects, blinding of manually observed behaviors, and a randomized order of testing. However, the authors did not report methods for achieving matched pup ages across the 40 litters and they rotated dams across litters to prevent potential nurturing bias. Dam rotation is an unproven approach that may have introduced unexpected effects in the pups, including effects on maternal caretaking (e.g., neglect of high-dose pups) and dosing accuracy (e.g., cross-contamination from littermates). Bouayed et al. (2009a) was a smaller and less robust study evaluating multiple measures across several ages using 5 litters of mice per dose group, and which used appropriate statistical analyses and included a detailed evaluation of maternal caretaking

behaviors. However, the study design did not account for potential litter effects (i.e., each litter was assigned a dose group), treatment-related changes in body weight complicate some interpretations, and protocols for observer blinding and randomized testing were not reported.

Of the behaviors tested by Chen et al. (2012), the endpoints providing the most convincing evidence of behavioral toxicity were alterations in open field, Morris water maze, and elevated plus maze tests, as multiple parameters were affected in each of these tests, the effects were observed across two cohorts of rats, and altered behavior was demonstrated to persist in juvenile and adult animals weeks to months after exposure. Bouayed et al. (2009a) did not evaluate effects in adult animals, but of the behaviors tested after exposure, effects on water escape pole climbing in male weanlings and on elevated plus maze performance in juveniles of both sexes were the most sensitive treatment-related changes.

In summary, it has been consistently demonstrated that developmental exposure to benzo[a]pyrene, particularly during late gestation or early postnatal development, causes persistent neurobehavioral effects that have been observed across two species, multiple strains, and both sexes of experimental animals, and across several behavioral domains. These data are supported by observations suggesting developmental neurotoxicity in children exposed to PAH mixtures, including-altered head circumference and neurobehavioral changes, as well as molecular changes in experimental animals which are consistent with altered CNS function. While not every endpoint tested was affected to the same extent, or in the same manner, across studies and species, all of the identified studies reported at least one nervous system effect of developmental exposure, demonstrating a high level of consistency within the available database.

In conclusion, although significant exposure gaps remain to be tested (most notably, benzo[a]pyrene exposures spanning gestation and lactation), EPA identified developmental toxicity (including developmental neurotoxicity) as a human hazard of benzo[a]pyrene exposure.

Susceptible Populations and Lifestages

Childhood susceptibility to benzo[a]pyrene toxicity is indicated by epidemiological studies reporting associations between adverse birth outcomes and developmental effects and internal biomarkers of exposure to benzo[a]pyrene, presumably via exposure to complex PAH mixtures (Perera et al., 2012b; Perera et al., 2009; Tang et al., 2008; Tang et al., 2006; Perera et al., 2005b; Perera et al., 2005a; Perera et al., 2004). The occurrence of benzo[a]pyrene-specific DNA adducts in maternal and umbilical cord blood in conjunction with exposure to ETS was associated with reduced birth weight and head circumference in offspring of pregnant women living in New York City (Perera et al., 2005b). In other studies, elevated levels of BPDE-DNA adducts in umbilical cord blood were associated with: (1) reduced birth weights or reduced head circumference (Perera et al., 2005a; Perera et al., 2004); and (2) decreased body weight at 18, 24, and 30 months (Tang et al., 2008; Tang et al., 2006).

Studies in animals exposed during development also support effects on pup growth (Chen et al., 2012; Liang et al., 2012; Kristensen et al., 1995; Mackenzie and Angevine, 1981), development of reproductive organs (Kristensen et al., 1995; Mackenzie and Angevine, 1981) and fertility (Wu et al., 2003a; Archibong et al., 2002; Kristensen et al., 1995; Mackenzie and Angevine, 1981). Studies evaluating developmental immunotoxicity following environmentally relevant exposures to benzo[a]pryene are not available in the database, however, studies utilizing intraperitoneal exposure paradigms provide a strong indication of potential developmental Immunotoxicity (see *Developmental Immunotoxicty* in Section 1.1.3).

Studies in humans and experimental animals indicate that exposure to PAHs in general, and benzo[a]pyrene in particular, may impact neurological development. Observational studies in humans have suggested associations between gestational exposure to PAHs and later measures of neurodevelopment (Perera et al., 2009; Tang et al., 2008). In the Perera et al. (2009) study, the exposure measures are based on a composite of eight PAHs measured in air. In Tang et al. (2008)., increased levels of benzo[a]pyrene-DNA adducts in cord blood were associated with decreased developmental quotients in offspring (Tang et al., 2008).

Evidence in animals of the effects of benzo[a]pyrene on neurological development includes: (1) decrements in reflex-related behaviors associated with neuromuscular coordination and sensorimotor function (Chen et al., 2012; Bouayed et al., 2009a); (2) disrupted learning and/or short-term memory processes (Chen et al., 2012; Li et al., 2012; Sheng et al., 2010; Bouayed et al., 2009a); and (3) decreased anxiety-related responses (Chen et al., 2012; Bouayed et al., 2009a). Mechanistic studies also support findings of benzo[a]pyrene induced alterations in electrophysiological response to stimulation of the dentate gyrus of the hippocampus (Wormley et al., 2004; Wu et al., 2003a) and decreased evoked response in the field cortex (McCallister et al., 2008).

1.1.2. Reproductive Toxicity

Human and animal studies provide evidence for benzo[a]pyrene-induced male and female reproductive toxicity. Effects on sperm quality and male fertility have been demonstrated in human populations highly exposed to PAH mixtures (Soares and Melo, 2008; Hsu et al., 2006). The use of internal biomarkers of exposure in humans (e.g., BPDE-DNA adducts) support associations between benzo[a]pyrene exposure and these effects. In females, numerous epidemiological studies indicate that cigarette smoking reduces fertility; however, few studies have specifically examined levels of benzo[a]pyrene exposure and female reproductive outcomes. Animal studies demonstrate decrements in sperm quality, changes in testicular histology, and hormone alterations following benzo[a]pyrene exposure in adult male animals, and decreased fertility and ovotoxic effects in adult females following exposure to benzo[a]pyrene.

Male Reproductive Effects

Fertility

Effects on male fertility have been demonstrated in populations exposed to mixtures of PAHs. Spermatozoa from smokers have reduced fertilizing capacity, and embryos display lower implantation rates (Soares and Melo, 2008). Occupational PAH exposure has been associated with higher levels of PAH-DNA adducts in sperm and male infertility (Gaspari et al., 2003). In addition, men with higher urinary levels of PAH metabolites have been shown to be more likely to be infertile (Xia et al., 2009). Studies were not identified that directly examined the reproductive capacity of adult animals following benzo[a]pyrene exposure. However, a dose-related decrease in fertility was observed in male mice treated in utero with benzo[a]pyrene, as discussed in Section 1.1.1.

Sperm parameters

Effects on semen quality have been demonstrated in populations exposed to mixtures of PAHs including coke oven workers and smokers (Soares and Melo, 2008; Hsu et al., 2006). Coke oven workers had higher frequency of oligospermia (19 versus 0% in controls) and twice the number of morphologically abnormal sperm (Hsu et al., 2006). Elevated levels of BPDE-DNA adducts have been measured in the sperm of populations exposed to PAHs occupationally (Gaspari et al., 2003) and through cigarette smoke (Phillips, 2002; Zenzes et al., 1999). A higher concentration of BPDE-DNA adducts was observed in sperm not selected for intrauterine insemination or in vitro fertilization based on motility and morphology in patients of fertility clinics (Perrin et al., 2011b; Perrin et al., 2011a). An association between benzo[a]pyrene exposure levels and increased sperm DNA fragmentation using the sperm chromatin structure assay was observed by Rubes et al. (2010). However, it is currently unclear whether the sperm chromatin structure assay, which measures sperm fragmentation following denaturation, is predictive of fertility (Sakkas and Alvarez, 2010; ASRM, 2008).

In several studies in rats and mice, a decrease in sperm count, motility, and production and an increase in morphologically abnormal sperm have been reported (Table 1-5 and Figure 1-4). Alterations in these sperm parameters have been observed in different strains of rats and mice and across different study designs and routes of exposure.

Decreases in epididymal sperm counts (25–50% compared to controls) have been reported in Sprague-Dawley rats and C57BL6 mice treated with 1–5 mg/kg-day benzo[a]pyrene by oral exposure for 42 or 84 days (Chen et al., 2011; Mohamed et al., 2010). Another subchronic study noted a 44% decrease in vas deferens sperm concentration at doses ≥50 mg/kg-day in Hsd:ICR (CD1) mice (Jeng et al., 2013). Additionally, a 15% decrease in epididymal sperm count was observed at a much lower dose in Sprague-Dawley rats exposed to benzo[a]pyrene for 90 days (Chung et al., 2011). However, confidence in this study is limited because the authors dosed the animals with 0.001, 0.01, and 0.1 mg/kg-day benzo[a]pyrene, but only reported on sperm parameters at the mid-dose, and no other available studies demonstrated findings in the range of the mid- and high-dose. In rats, an oral short-term study and a subchronic inhalation study lend

support for the endpoint of decreased sperm count (Arafa et al., 2009; Archibong et al., 2008; Ramesh et al., 2008). Significantly decreased sperm count and daily sperm production (20–40% decrease from control in each parameter) were observed in rats following 10 days of gavage exposure to 50 mg/kg-day (Arafa et al., 2009) and following gavage dosing with 10 mg/kg-day on PNDs 1–7 (Liang et al., 2012). In addition, a 69% decrease from controls in sperm count was observed in rats following inhalation exposure to 75 μg/m^3 benzo[a]pyrene for 60 days (Archibong et al., 2008; Ramesh et al., 2008).

Both oral and inhalation exposure of rodents to benzo[a]pyrene have been shown to lead to decreased epididymal sperm motility and altered morphology. Decreased motility of 20–30% compared to controls was observed in Hsd:ICR (CD1) mice (≥100 mg/kg-day), C57BL6 mice (≥1 mg/kg-day), and Sprague-Dawley rats (0.01 mg/kg-day) following subchronic oral exposure (Jeng et al., 2013; Chung et al., 2011; Mohamed et al., 2010). The effective doses spanned several orders of magnitude; however, as noted above, reporting is limited in the study that observed effects at 0.01 mg/kg-day benzo[a]pyrene (Chung et al., 2011). A short-term oral study in rats also reported a significantly decreased number of motile sperm (~40% decrease) following 10 days of gavage exposure to 50 mg/kg-day benzo[a]pyrene (Arafa et al., 2009). In addition, decreased sperm motility was observed following inhalation exposure to 75 μg/m^3 benzo[a]pyrene in rats for 60 days (Archibong et al., 2008; Ramesh et al., 2008) and to ≥75 μg/m^3 for 10 days (Inyang et al., 2003). Abnormal sperm morphology was observed in Sprague-Dawley rats treated with 5 mg/kg day benzo[a]pyrene by gavage for 84 days (Chen et al., 2011), Hsd:ICR (CD1) mice exposed to >50 mg/kg-day benzo[a]pyrene by gavage for 60 days (Jeng et al., 2013), and in F344 rats exposed to 75 μg/m^3 benzo[a]pyrene by inhalation for 60 days (Archibong et al., 2008; Ramesh et al., 2008).

Testicular changes

Several studies have demonstrated dose-related effects on male reproductive organs in adult animals exposed subchronically to benzo[a]pyrene (Table 1-5 and Figure 1-4). Decreases in testicular weight of approximately 35% have been observed in a 60-day gavage study in Hsd:ICR (CD1) mice at 100 mg/kg-day (Jeng et al., 2013), in a 10-day gavage study in adult Swiss albino rats at 50 mg/kg-day (Arafa et al., 2009), and following subchronic inhalational exposure of adult F344 rats to 75 μg/m^3 (Archibong et al., 2008; Ramesh et al., 2008). No effects on testes weight were observed in Wistar rats exposed for 35 days to gavage doses up to 50 mg/kg-day (Kroese et al., 2001), F344 rats exposed for 90 days to dietary doses up to 100 mg/kg-day (Knuckles et al., 2001), or Sprague-Dawley rats exposed for 90 days to gavage doses up to 0.1 mg/kg-day (Chung et al., 2011). Strain differences may have contributed to differences in response; however, F344 rats exposed to benzo[a]pyrene via inhalation showed effects on testicular weight (Archibong et al., 2008; Ramesh et al., 2008). In addition, decreased testicular weight has also been observed in offspring following in utero and early postnatal exposure to benzo[a]pyrene as discussed in Section 1.1.1.

Histological changes in the testis have often been reported to accompany decreases in testicular weight. Apoptosis, as evident by increases in terminal deoxynucleotidyl transferase dUTP nick end labeling (TUNEL) positive germ cells and increases in caspase-3 staining, was evident in seminiferous tubules of Sprague-Dawley rats following 90 days of exposure to ≥0.001 and 0.01 mg/kg-day, respectively, benzo[a]pyrene by gavage (Chung et al., 2011). However, the study authors did not observe testicular atrophy or azospermia in any dose group. Hsd:ICR (CD1) mice exposed to 50 or 100 mg/kg-day by gavage for 30 days showed loss of seminiferous tubule integrity and Sertoli cell fidelity (Jeng et al., 2013). Seminiferous tubules were reported to look qualitatively similar between controls and animals exposed to benzo[a]pyrene by inhalation doses of 75 μg/m^3 for 60 days (Archibong et al., 2008; Ramesh et al., 2008). However, when histologically examined, statistically significantly reduced tubular lumen size and length were observed in treated animals. Seminiferous tubule diameters also appeared to be reduced in exposed animals, although this difference did not reach statistical significance (Archibong et al., 2008; Ramesh et al., 2008). In addition, histological changes in the seminiferous tubules have also been observed in offspring following in utero exposure to benzo[a]pyrene as discussed in Section 1.1.1.

Epididymal changes

In addition to testicular effects, histological effects in the epididymis have been observed following 90-day gavage exposure to benzo[a]pyrene (Chung et al., 2011) (Table 1-5 and Figure 1-4). Specifically, statistically significant decreased epididymal tubule diameter (for caput and cauda) was observed at doses ≥0.001 mg/kg-day. At the highest dose tested (0.1 mg/kg-day), diameters were reduced approximately 25%. A 60-day gavage study in Hsd:ICR(CD1) mice observed a 27% decrease in cauda epididymis weight at 100 mg/kg-day (Jeng et al., 2013); however, no change in epididymis weight was observed following an 84-day treatment in Sprague-Dawley rats of 5 mg/kg-day benzo[a]pyrene (Chen et al., 2011).

Hormone changes

Several animal models have reported decreases in testosterone following both oral and inhalation exposure to benzo[a]pyrene (Table 1-5 and Figure 1-4). In male Sprague-Dawley rats, decreases in testosterone have been observed following 90-day oral exposures (Chung et al., 2011; Zheng et al., 2010). Statistically significant decreases of 15% in intratesticular testosterone were observed at 5 mg/kg-day in one study (Zheng et al., 2010), while a second study in the same strain of rats reported statistically significant decreases of approximately 40% in intratesticular testosterone and 70% in serum testosterone at 0.1 mg/kg-day (Chung et al., 2011). In addition, Sprague-Dawley rats treated with 10 mg/kg-day by gavage on PNDs 1–7 exhibited statistically significantly decreased serum testosterone (≥40%) when examined at PND 8 and PND 35 (Liang et al., 2012). Statistically significant decreases in intratesticular testosterone (80%) and serum testosterone (60%) were also observed following inhalation exposure to 75 μg/m^3 benzo[a]pyrene in F344 rats for 60 days (Archibong et al., 2008; Ramesh et al., 2008). In contrast to these findings,

exposure of Sprague-Dawley rats to 5 mg/kg-day benzo[a]pyrene by gavage produced a transient increase in serum testosterone at 4 and 8 weeks, which returned to controls levels at 12 weeks of exposure (Chen et al., 2011). Statistically significant increases in serum luteinizing hormone (LH) have also been observed in Sprague-Dawley rats following gavage exposure to benzo[a]pyrene at doses of ≥0.01 mg/kg-day (Chung et al., 2011) and in F344 rats following inhalation exposure to 75 µg/m^3 benzo[a]pyrene for 60 days (Archibong et al., 2008; Ramesh et al., 2008).

Table 1-5. Evidence pertaining to the male reproductive toxicity of benzo[a]pyrene in adult animals after oral or inhalation exposure

Reference and study design	Results[a,b]
Sperm quality	
Mohamed et al. (2010) C57BL/6 mice (6 wks old), 10 males/dose (treated before mating with unexposed females) 0, 1, or 10 mg/kg-d by gavage (F0 males only) 42 d Sperm parameters assessed in F0 males 2 wks after cessation of exposure (14 wks of age). Sperm parameters assessed in untreated F1, F2, and F3 males at 14 wks of age.	↓ epididymal sperm count in F0 mice Approximate % change from control (data reported graphically): 0, −50*, and −70* ↓ epididymal sperm motility in F0 mice Approximate % change from control (data reported graphically): 0, −20*, and −50* ↓ epididymal sperm count in untreated F1 and F2 generations (data reported graphically) No effects were observed in the F3 generation
Jeng et al. (2013) Hsd:ICR (CD1) mice (10 wks old), 8 males/dose 0, 1, 10, 50, 100 mg/kg-d by gavage 30 or 60 d Spermatozoa were obtained from a consistent length of vas deferens	↓ Sperm concentration at 30 d (% change from control): 0, −33, −31, −26, −18 ↓ Sperm concentration at 60 d (% change from control): 0, −19, 7, −44, −42 ↓ Sperm motility at 30 d (% change from control): 0, 1, −16, −39, −16 ↓ Sperm motility at 60 d (% change from control): 0, −26, −19, −12, −28* ↓ Sperm vitality at 30 d (% change from control): 0, 9, −15, −33, −11 ↓ Sperm vitality at 60 d (% change from control): 0, −3, −2, −4, −6* ↑ % abnormal sperm (abnormal head) at 30 d: 10, 11, 12, 14, 12 ↑ % abnormal sperm (abnormal head) at 60 d: 17, 27, 23, 29*, 34*

Reference and study design	Results[a,b]
Chen et al. (2011) Sprague-Dawley rats (5–6 wks old), 10 males/dose 0 or 5 mg/kg-d by gavage 28, 56, or 84 d	↓ epididymal sperm count at 84 d (% change from control; no change at 28 or 56 d) 0 and −29* ↑ % abnormal epididymal sperm at 84 d (no change at 28 or 56 d) 5 and 8*
Chung et al. (2011) Sprague-Dawley rats (8 wks old), 20–25 males/dose 0, 0.001, 0.01, or 0.1 mg/kg-d by gavage 90 d	↓ epididymal sperm motility (% change relative to control; reported only for 0.01 mg/kg-d) 0 and −30* No statistically significant decrease in epididymal sperm count
Ramesh et al. (2008); Archibong et al. (2008) F344 rats (12–13 wks old), 10 males/group 0 or 75 µg/m^3, 4 hrs/d by inhalation 60 d (sperm parameters were assessed 72 hrs after final exposure)	↓ epididymal sperm motility (% change from control) 0 and −73* ↓ epididymal sperm count (% change from control) 0 and −69* ↑ % abnormal epididymal sperm 33 and 87* ↓ spermatids/g testis (approximate % change from control; numerical data not reported) 0 and −45*
Testicular changes (weight, histology)	
Mohamed et al. (2010) C57BL/6 mice (6 wks old), 10 males/dose (treated before mating with unexposed females) 0, 1, or 10 mg/kg-d by gavage (F0 males only) 42 d F0 males sacrificed 2 wks after cessation of exposure (14 wks of age). Unexposed F1, F2, and F3 males sacrificed at 14 wks of age.	↓ seminiferous tubules with elongated spermatids in F0 males (approximate % change from control; numerical data not reported) 0, −20*, and −35* No statistically significant change in area of seminiferous epithelium of testis in F0 males (approximate % change from control; numerical data not reported) 0, 5, and 20 Testicular findings non-significant in F1 and F3 generations, but significant at the high dose in F2 males.

Reference and study design	Results[a,b]
Jeng et al. (2013) Hsd:ICR (CD1) mice (10 wks old), 8 males/dose 0, 1, 10, 50, or 100 mg/kg-d by gavage 30 or 60 d	↑ testicular lesions after 30 d of exposure were characterized as decreased seminiferous tubule integrity and loss of Sertoli cell fidelity at 50 mg/kg-d; variation in the seminiferous tubule diameter and reduced organization and integrity, decreased luminal volume of mature sperm, and uneven Sertoli cell maintenance of the seminiferous epithelium at 100 mg/kg-d. Histopathology findings were not reported for the 60 d exposure. ↓ decreased testis weight at 30 d (% change from control): 0, −4, −7, 1, −10 ↓ decreased testis weight at 60 d (% change from control): 0, 3, −15, −23, −35*
Chung et al. (2011) Sprague-Dawley rats (8 wks old), 20−25 males/dose 0, 0.001, 0.01, or 0.1 mg/kg-d by gavage 90 d	↑ number of apoptotic germ cells per tubule (TUNEL or caspase 3 positive) No change in testis weight or histology
Chen et al. (2011) Sprague-Dawley rats (5−6 wks old), 10 males/dose 0 or 5 mg/kg-d by gavage 84 d	↑ testicular lesions characterized as irregular arrangement of germ cells and absence of spermatocytes (numerical data not reported) No change in testis weight
Archibong et al. (2008); Ramesh et al. (2008) F344 rats (12−13 wks old), 10 males/group 0 or 75 µg/m^3, 4 hrs/d by inhalation 60 d Animals sacrificed 72 hrs after final exposure.	↓ decreased testis weight (% change from control) 0 and 34* ↓ size of seminiferous tubule lumens and reduced tubular length No change in % of tubules with elongated spermatids
Kroese et al. (2001) Wistar rats (6 wks old), 10/sex/dose 0, 1.5, 5, 15, or 50 mg/kg-d by gavage, 5 d/wk 35 d	No change in testis weight (data not shown)
Knuckles et al. (2001) F344 rats (8 wks old), (number of animals examined unclear) 0, 5, 50, or 100 mg/kg-d in diet 90 d	No change in testis weight (data not shown)

Reference and study design	Results[a,b]
Epididymal changes (weight, histology)	
Chung et al. (2011) Sprague-Dawley rats (8 wks old), 20–25 males/dose 0, 0.001, 0.01, or 0.1 mg/kg-d by gavage 90 d	↓ diameter of caput epididymal tubule (n = 5; numerical data not reported) ↓ diameter of cauda epididymal tubule (n = 5; numerical data not reported)
Chen et al. (2011) Sprague-Dawley rats (5–6 wks old), 10/dose 0 or 5 mg/kg-d by gavage 84 d	No change in epididymis weight
Jeng et al. (2013) Hsd:ICR (CD1) mice (10 wks old), 8 males/dose 0, 1, 10, 50, 100 mg/kg-d by gavage 30 or 60 d	↓ decreased cauda epididymis weight at 30 d (% change from control): 0, −16, −26, −26, −20 ↓ decreased cauda epididymis weight at 60 d (% change from control): 0, −13, −13, −17, −27*
Hormone changes	
Chung et al. (2011) Sprague-Dawley rats (8 wks old), 20–25 males/dose 0, 0.001, 0.01, 0.1 mg/kg-d by gavage 90 d	↓ Intratesticular testosterone (approximate % change from control; data reported graphically) 0, −12, −25, and −40* ↓ Serum testosterone (approximate % change from control; numerical data not reported) 0, 0, −35, and −70* ↑ serum LH (approximate % change from control; numerical data not reported) 0, 33, 67*, and 87* ↓ Human chorionic gonadotropin (hCG) or dibutyl cyclic adenosine monophosphate (dbcAMP)-stimulated testosterone production in Leydig cells
Chen et al. (2011) Sprague-Dawley rats (5–6 wks old), 10 males/dose 0 or 5 mg/kg-d by gavage 28, 56, or 84 d	↑ serum testosterone at 28 and 56 d only (approximate % change from control; data reported graphically): 28 d 0, 160* 56 d 0, 100* 84 d 0, −10
Zheng et al. (2010) Sprague-Dawley rats (6 wks old), 8 males/dose 0, 1, or 5 mg/kg-d by gavage 90 d	↓ Intratesticular testosterone (approximate % change from control; numerical data not reported) 0, −15, and −15*

Reference and study design	Results[a,b]
Archibong et al. (2008); Ramesh et al. (2008) F344 rats (12−13 wks old), 10 adult males/group 0 or 75 µg/m^3, 4 hrs/d by inhalation 60 d	↓ intratesticular testosterone, 72 hours post exposure (approximate % change from control; numerical data not reported) 0 and −80* ↓ serum testosterone (approximate % change from control) 0 and −70* ↑ serum LH (approximate % change from control) 0 and 50*

*Statistically significantly different from the control ($p < 0.05$).
[a]% change from control calculated as: (treated value − control value)/control value × 100.
[b]Endpoints accessed directly following the exposure period unless otherwise indicated.

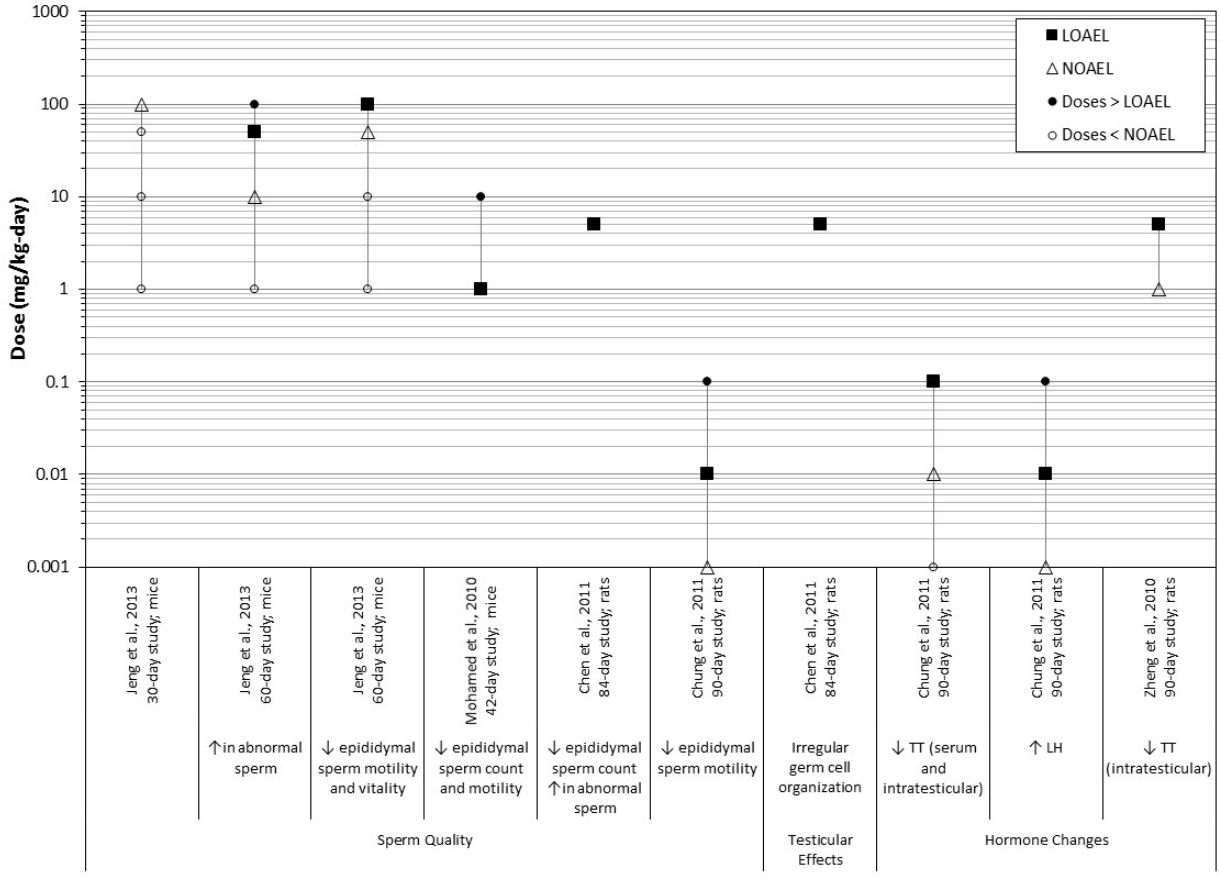

Figure 1-4. Exposure-response array for male reproductive effects following oral exposure in adult animals.

Mode-of-action analysis—male reproductive effects

Exposure to benzo[a]pyrene in laboratory animals induces male reproductive effects including decreased sperm quality, decreased levels of testosterone, increased levels of LH, and histological changes in the testis. Hypothesized modes of action of include benzo[a]pyrene-mediated DNA damage to male germ cells leading to cytotoxicity, apoptosis and decreased embryo viability post-fertilization, compromised function of Sertoli and Leydig cells, oxidative stress, and altered regulation of the stAR promoter.

Decrements in sperm quality has been demonstrated in populations highly exposed to PAH mixtures (Soares and Melo, 2008; Hsu et al., 2006) as well as in male animals exposed to benzo[a]pyrene (see Table 1-5). Genotoxicity in male germ cells has been hypothesized to lead to cytotoxicity and apoptosis (Chung et al., 2011; Perrin et al., 2011b; Perrin et al., 2011a; Olsen et al., 2010; Revel et al., 2001) as well as mutations (Xu et al., 2014) and decreased embryo viability post-fertilization associated with sperm DNA damage (Borini et al., 2006; Seli et al., 2004).

A study in tobacco smokers suggests that direct DNA damage from the reactive metabolite BPDE may decrease sperm motility (Perrin et al., 2011a). In this study, motile sperm were separated from non-motile sperm using a "swim-up" self-migration technique. The investigators found that the motile sperm selected by this method had significantly fewer BPDE-adducts than non-selected sperm.

Numerous studies have indicated that benzo[a]pyrene reduces testosterone levels in adult animals following oral or inhalation exposure (Chung et al., 2011; Zheng et al., 2010; Archibong et al., 2008; Ramesh et al., 2008). It is plausible that the effects on sperm quality and histological changes of the reproductive organs are secondary to an insufficiency of testosterone (Inyang et al., 2003). One study has hypothesized that benzo[a]pyrene perturbs the production of testosterone by Leydig cells (Chung et al., 2011). This study found a statistically significant reduction in testicular testosterone in rats treated with 0.1 mg/kg-day benzo[a]pyrene for 90 days and found that testosterone production in isolated Leydig cells was also inhibited approximately 50%, even in cultures stimulated with human chorionic gonadotropin and dibutryl cyclic adenosine monophosphate.

Leydig cell function is thought to be regulated by testicular macrophages (Hales, 2002). When testicular macrophages are activated and produce inflammatory mediators, Leydig cell testosterone production is inhibited (Hales, 2002). Zheng et al. (2010) treated rats with 5 mg/kg-day benzo[a]pyrene for 90 days and reported a statistically significant increase in ED-1 type testicular macrophages and a statistically significant decrease in intratesticular testosterone.

Some studies suggest that male reproductive effects may be secondary to increased oxidative stress. Arafa et al. (2009) reported that male reproductive effects observed following benzo[a]pyrene exposure could be ameliorated by antioxidant pre-treatment. This study reported decreased sperm count, motility, and production, in addition to decreased testis weight following a 10 day oral administration in rats of 50 mg/kg-day benzo[a]pyrene. Pretreatment with the citris

flavonoid hesperidin protected rats from all of these effects except the decrease in sperm motility. In addition, studies in GSH deficient mice suggest that oxidative stress may mediate male reproductive effects observed after developmental exposure (Nakamura 2012).

Female Reproductive Effects

Fertility

In women, exposure to cigarette smoke has been shown to affect fertility, including effects related to pregnancy, ovulatory disorders, and spontaneous abortion (as reviewed in Waylen et al., 2009; Cooper and Moley, 2008; Soares and Melo, 2008). In addition, several studies suggest that in utero exposure to maternal tobacco smoke also decreases the future fertility of female offspring (Ye et al., 2010; Jensen et al., 1998; Weinberg et al., 1989). Benzo[a]pyrene levels in follicular fluid and benzo[a]pyrene-DNA adducts in granulosa-lutein cells and oocytes and in human cervical cells have been associated with smoking status and with amount smoked (Neal et al., 2008; Mancini et al., 1999; Melikian et al., 1999; Zenzes et al., 1998; Shamsuddin and Gan, 1988).

Few epidemiological studies have examined the specific influence of components of PAH mixtures on fertility or other reproductive outcomes; EPA identified only two studies with specific data on benzo[a]pyrene (Table 1-6). One of these studies addressed the probability of conception among women undergoing in vitro fertilization (Neal et al., 2008). Follicular fluid benzo[a]pyrene levels were significantly higher among the women who did not conceive compared with women who did get pregnant. No association was seen between conception and serum levels of benzo[a]pyrene. The other study examined risk of delayed miscarriage (fetal death before 14 weeks of gestation), using a case-control design with controls selected from women undergoing elective abortion (Wu et al., 2010). A strong association was seen between maternal blood benzo[a]pyrene-DNA adduct levels and risk of miscarriage, with a fourfold increased risk for levels above compared with below the median. Benzo[a]pyrene-DNA adduct levels were similar in the aborted tissue of cases compared with controls.

Experimental studies in mice also provide evidence that benzo[a]pyrene exposure affects fertility (see Table 1-7 and Figure 1-5). Decreased fertility and fecundity (decreased number of F0 females producing viable litters at parturition) was statistically significantly reduced by about 35% in adult females exposed to 160 mg/kg-day of benzo[a]pyrene (Mackenzie and Angevine, 1981). In another study, F0 females showed no signs of general toxicity or effects on fertility following gavage exposure to 10 mg/kg-day on GDs 7–16 (Kristensen et al., 1995). Decrements in fertility were more striking in the offspring exposed during development, as described in μSection 1.1.1 ("Developmental Toxicity"), as exposure in utero appears to impair the development of reproductive organs.

One study suggests that exposure to benzo[a]pyrene prior to mating decreases the ovulation rate in female rats treated by inhalation to 50, 75 or 100 $\mu g/m^3$ for 4 hours per day for 14 days prior to mating (Archibong et al., 2012). A dose related decrease in ovulation rate and a

corresponding decrease in the number of pups born per litter was seen starting at the lowest dose. At the highest concentration tested, a decrease in litter size was seen which was only partially explained by the decreased ovulation rate.

Ovarian effects

Human epidemiological studies that directly relate ovotoxicity and benzo[a]pyrene exposure are not available; however, smoking, especially during the time of the peri-menopausal transition, has been shown to accelerate ovarian senescence (Midgette and Baron, 1990). Benzo[a]pyrene-induced ovarian toxicity has been demonstrated in animal studies. In adult female rats treated by gavage, statistically significant, dose-related decreases in ovary weight have been observed in female rats treated for 60 days at doses ≥5 mg/kg (2.5 mg/kg-day adjusted) (Xu et al., 2010). At 10 mg/kg in adult rats (5 mg/kg-day adjusted), ovary weight was decreased 15% (Xu et al., 2010). Apparent decreases in ovary weight of approximately 10−13% were also observed in a 14-day inhalation study in F344 rats treated with 50, 75, or 100 μg/m^3 benzo[a]pyrene (Archibong et al., 2012).

Two additional studies in adult rats, not designed to investigate reproductive endpoints, did not detect changed in ovary weight. Specifically, ovary weight was investigated, but significant findings were not reported (data not shown) in Wistar rats (n=10) exposed for 35 days to gavage doses up to 50 mg/kg-day (Kroese et al., 2001). In a study in F344 rats exposed for 90 days to dietary doses up to 100 mg/kg-day, ovarian weight was reportedly measured, but was not among the organs with significant weight changes. However, data for this endpoint, as well as the number of animals examined, was not reported for this exposure duration (Knuckles et al., 2001).

As discussed in Section 1.1.1 (Developmental Toxicity), severe reductions in ovarian weight of offspring gestationally treated with benzo[a]pyrene were reported in mice (Kristensen et al., 1995; Mackenzie and Angevine, 1981). Specifically, ovary weight in F1 offspring was reduced 31% following exposure to 10 mg/kg-day benzo[a]pyrene (Kristensen et al., 1995), while in another gestational study at the same dose level, ovaries were so drastically reduced in size (or absent) that they were not weighed (Mackenzie and Angevine, 1981).

In adult female rats treated by gavage, dose-related decreases in the number of primordial follicles have been observed in female rats treated for 60 days at doses ≥2.5 mg/kg-day, with a statistically significant decrease of approximately 20% at the high dose (Xu et al., 2010) (Table 1-7 and Figure 1-5). No notable differences in other follicle populations and corpora lutea were observed. However, in utero studies exposing dams to the same doses produced offspring with few or no follicles or corpora lutea (Kristensen et al., 1995; Mackenzie and Angevine, 1981). Additional support for the alteration of female reproductive endpoints comes from i.p. experiments in animals and in vitro experiments. Several studies have observed ovarian effects (decreased numbers of ovarian follicles and corpora lutea, absence of folliculogenesis, oocyte degeneration, and decreased fertility) in rats and mice exposed via i.p. injection (Borman et al., 2000; Miller et al., 1992; Swartz

and Mattison, 1985; Mattison et al., 1980). Further evidence is available from in vitro studies showing inhibition of antral follicle development and survival, as well as decreased production of estradiol, in mouse ovarian follicles cultured with benzo[a]pyrene for 13 days (Sadeu and Foster, 2011). Likewise, follicle stimulating hormone (FSH)-stimulated growth of cultured rat ovarian follicles was inhibited by exposure to benzo[a]pyrene (Neal et al., 2007).

Hormone levels

Alteration of estrus cyclicity and hormone levels have been observed in female rats following oral or inhalation exposure to benzo[a]pyrene (Table 1-7 and Figure 1-5). Inhalation exposure to benzo[a]pyrene:carbon black particles during gestation resulted in decreases in plasma progesterone, estradiol, and prolactin in pregnant rats (Archibong et al., 2002). A 14 day inhalation study in adult F344 rats noted estrous cycle length was significantly increased in rats in the high dose group (100 μg/m^3). The investigators measured hormone levels in this dose group during the four different stages of the estrus cycle and noted decreased estrodial during proestrus, decreased progesterone during diestrus I, increased FSH during all stages, and decreased LH during proestrus (Archibong et al., 2012). Similar alterations have been noted following oral exposures with statistically significant, dose-related decreases in estradiol and altered estrus cyclicity observed in female rats treated for 60 days at doses ≥2.5 mg/kg-day by gavage (Xu et al., 2010). Mechanistic experiments have also noted decreased estradiol output in murine ovarian follicles cultured with benzo[a]pyrene in vitro for 13 days, but did not find any decrease in progesterone (Sadeu and Foster, 2011).

Uterine effects

One subchronic animal study is available that investigated effects in the uterine cervix following oral exposure to benzo[a]pyrene (Table 1-7 and Figure 1-5). Statistically-significant dose-related increases in the incidence of cervical inflammatory cells were observed in mice exposed twice a week for 98 days to benzo[a]pyrene via gavage at doses ≥2.5 mg/kg (Gao et al., 2011). Uterine cervical effects of increasing severity, including epithelial hyperplasia, atypical hyperplasia, apoptosis, and necrosis, were observed at higher doses. This study also observed a depression of bodyweight (10, 15, and 30%) and elevated mortality in the two higher dose groups (4 and 8%), suggesting potential treatment related toxicity. Gao et al. (2011) also evaluated effects on the uterine cervix in separate groups of mice exposed via i.p. injection, and observed similar responses in these groups of mice. Gao et al. (2011) considered the hyperplasia responses to be preneoplastic lesions. Cervical neoplasia was not reported in the available chronic bioassays, but uterine tissue was not subjected to histopathology examination in either bioassay (Kroese et al., 2001; Beland and Culp, 1998). It is plausible, that an increase in uterine inflammation during pregnancy could impact parturition, and pre-term birth (as reviewed by Gomez-Lopez et al., 2014), however, the relationship of the observed uterine inflammation and hyperplasia in Gao et al. (2011) to reproductive function is uncertain.

Table 1-6. Evidence pertaining to the female reproductive effects of benzo[a]pyrene in humans

Reference and study design	Results
Probability of conception	
Neal et al. (2008) 36 women undergoing in vitro fertilization (19 smokers, 7 passive smokers, and 10 nonsmokers) Exposure: benzo[a]pyrene in serum and follicular fluid	Benzo[a]pyrene levels (ng/mL) Follicular fluid: Conceived 0.1; Did not conceive 1.7; *p*-value <0.001 Serum: Conceived 0.01; Did not conceive 0.05; *p*-value Not reported
Fetal death	
Wu et al. (2010) (Tianjin, China) Case control study: 81 cases (96% participation rate)—fetal death confirmed by ultrasound before 14 wks gestation; 81 controls (91% participation rate)—elective abortions; matched by age, gestational age, and gravidity; excluded smokers and occupational PAH exposure Exposure: benzo[a]pyrene in aborted tissue and maternal blood samples (51 cases and controls, 2 of 4 hospitals)	Benzo[a]pyrene adduct levels (/10^8 nucleotides), mean (±SD) Maternal blood: Cases 6.0 (± 4.7); Controls 2.7 (± 2.2); *p*-value <0.001 Aborted tissue: Cases 4.8 (± 6.0); Controls 6.0 (± 7.4); *p*-value 0.29 Low correlation between blood and tissue levels (r = −0.02 in cases, r = −0.21 in controls) Association between benzo[a]pyrene adducts and miscarriage[a] Per unit increase in adducts: OR 1.37; 95% CI 1.12, 1.67 Dichotomized at median: OR 4.56; 95% CI 1.46, 14.3 [a]Conditional logistic regression, adjusted for maternal education, household income, and gestational age; age also considered as potential confounder

Table 1-7. Evidence pertaining to the female reproductive effects of benzo[a]pyrene in adult animals after oral or inhalation exposure

Reference and study design	Results[a]
Fertility	
Mackenzie and Angevine (1981) CD-1 mice, 30 or 60 F0 females/dose 0, 10, 40, or 160 mg/kg-d by gavage GDs 7–16	↓ number of F0 females with viable litters 46/60, 21/30, 44/60, and 13/30*
Kristensen et al. (1995) NMRI mice, 9 females/dose 0 or 10 mg/kg-d by gavage GDs 7–16	No changes in fertility of F0 females
Archibong et al. (2012) Fisher 344 rats, 20 females/dose 0, 50, 75, or 100 µg benzo[a]pyrene/m^3 nose only inhalation for 4 hrs/d 14 d prior to mating (carbon black used as carrier particle; no carrier particle control used)	↓ number of pups born per litter 15, 13.4, 12.3, 4.3** ↓ embryo/fetal survival (%) 98, 96, 96, and 52** [% Embryo/fetal survival = number of pups/(number of ovulated eggs from females mated with vasectomized males) × 100]
Ovarian effects (weight, histology, follicle numbers)	
Xu et al. (2010) Sprague-Dawley rats, 6 females/dose 0, 5, or 10 mg/kg by gavage every other day (2.5 and 5 mg/kg-d, adjusted) 60 d	↓ ovary weight (% change from control) 0, −11*, and −15* ↓ number of primordial follicles (% change from control; data presented graphically) 0, −6, −22 ↑ increased apoptosis of ovarian granulosa cells (approximate % apoptosis) 2, 24*, and 14*
Knuckles et al. (2001) F344 rats, (number of animals examined not reported) 0, 5, 50, or 100 mg/kg-d in diet 90 d	No changes in ovary weight (data not reported)
Kroese et al. (2001) Wistar rats, 10/sex/dose 0, 1.5, 5, 15, or 50 mg/kg-d by gavage 5 d/wk 35 d	No changes in ovary weight (data not reported)

Reference and study design	Results[a]
Archibong et al. (2012) Fisher 344 rats, 20 females/dose 0, 50, 75, or 100 µg benzo[a]pyrene/m^3 nose only inhalation for 4 hrs/d 14 d (carbon black used as carrier particle; no carrier particle control used)	↓ ovary weight (% change from control) 0, −10, −13, and −12 ↓ ovulation rate (total number of ovulated eggs recovered from both oviducts and confirmed by the total number of corpora lutea; (% change from control) 0, −9, −16, and −46*
Hormone levels	
Xu et al. (2010) Sprague-Dawley rats, 6 females/ dose 0, 5, or 10 mg/kg by gavage every other day (2.5 and 5 mg/kg-d, adjusted) 60 d	↓ serum estradiol (approximate % change from control) 0, −16, and −25* Prolonged estrous cycle
Archibong et al. (2002) F344 rats, 10 females/group 0, 25, 75, or 100 µg/m^3 by inhalation 4 hrs/d GDs 11−20 (serum hormones tested at GD 15 and 17 in 0, 25, and 75 µg/m^3 dose groups)	↓ F0 estradiol, approximately 50% decrease at 75 µg/m^3 at GD 17 ↓ F0 prolactin, approximately 70% decrease at 75 µg/m^3 at GD 17 ↑ F0 plasma progesterone approximately 17% decrease at 75 µg/m^3 at GD 17
Archibong et al. (2012) Fisher 344 rats, 20 females/dose 0, 50, 75, or 100 µg benzo[a]pyrene/m^3 nose only inhalation for 4 hrs/d 14 d (carbon black used as carrier particle; no carrier particle control used)	Serum hormone concentrations measured in highest dose group and reported by stage of estrous cycle and compared to control: ↓ serum estradiol in proestrus ↓ serum progesterone in diestrus I ↑ serum FSH at all stages of estrous cycle ↓ serum LH in proestrus Prolonged estrous cycle in the high dose group (results presented graphically)
Uterine effects	
Gao et al. (2011) ICR mice, 26 females/dose 0, 2.5, 5, or 10 mg/kg by gavage 2 d/wk 98 d	↑ cervical epithelial hyperplasia: 0/26, 4/26, 6/25*, and 7/24* ↑ cervical atypical hyperplasia: 0/26, 0/26, 2/25, and 4/24* ↑ inflammatory cells in cervical epithelium: 3/26, 10/26, 12/25*, and 18/24*

*Statistically significantly different from the control ($p < 0.05$).

[a]% change from control calculated as: (treated value − control value)/control value × 100.

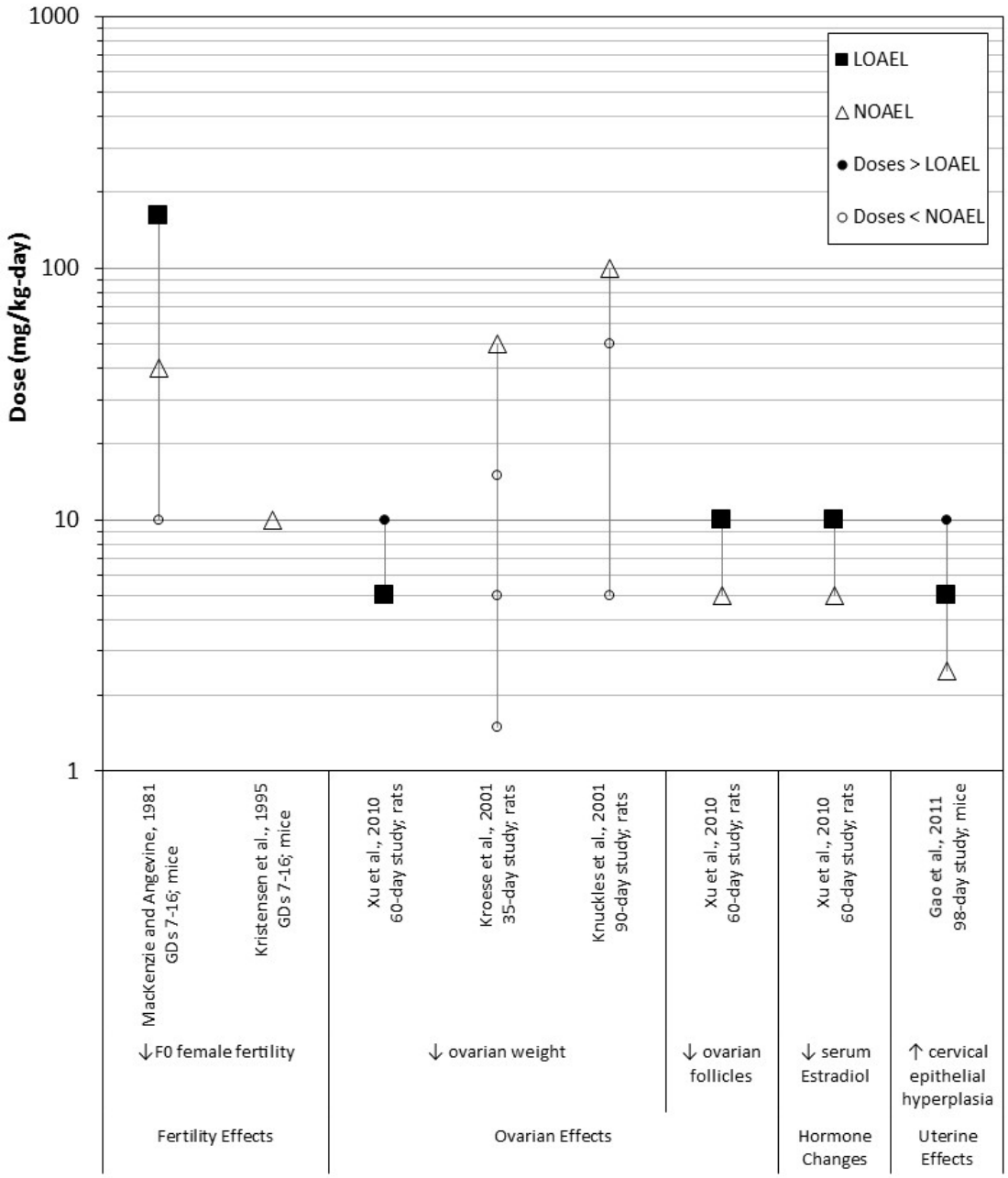

Figure 1-5. Exposure-response array for female reproductive effects following oral exposure in adult animals.

Mode-of-action analysis—female reproductive effects

Although the mechanisms underlying female reproductive effects following benzo[a]pyrene exposure are not fully established, associations with stimulation of apoptosis, impairment of steroidogenesis, alterations in hormone balance, genotoxicity and cytotoxicity have been made. Benzo[a]pyrene is well established as a genotoxic agent (see Section 1.1.5) and benzo[a]pyrene-DNA adducts have been detected in ovarian tissue (Ramesh et al., 2010). It is therefore plausible

that exposure to female germ cells, could result in cellular damage, reducing oocyte and embryo viability, and potentially transmit mutations to surviving offspring. In female rats exposed to a single oral dose (5 mg/kg), benzo[a]pyrene-DNA adducts were noted to be higher in ovarian tissue than the liver (Ramesh et al). In addition, increased BPDE-DNA adducts were found in oocytes and DNA strand breaks in cumulous cells in female mice exposed to a single oral dose of 13 mg/kg benzo[a]pyrene (Einaudi et al., 2014).

Ovarian lesions in benzo[a]pyrene-exposed rats have been associated with increased apoptosis in ovarian granulosa cells and alteration in hormone-mediated regulation of folliculogenesis (Xu et al., 2010), and results from in vitro and interperitoneal experiments provide support for an association between benzo[a]pyrene exposure and impaired folliculogenesis, steroidogenesis, and oocyte maturation (Kummer et al., 2013; Sadeu and Foster, 2011; Neal et al., 2007; Mattison, 1980). A growing body of research suggests that benzo[a]pyrene triggers the induction of apoptosis in oocytes through AhR-driven expression of pro-apoptotic genes, including Bax (Sadeu and Foster, 2013; Kee et al., 2010; Neal et al., 2010; Pru et al., 2009; Matikainen et al., 2002; Matikainen et al., 2001; Robles et al., 2000). Other proposed mechanisms include the impairment of folliculogenesis from reactive metabolites (Takizawa et al., 1984; Mattison and Thorgeirsson, 1979, 1977) or by a decreased sensitivity to FSH-stimulated follicle growth (Neal et al., 2007). Based on findings that an ERα antagonist counteracted effects of subcutaneously administered benzo[a]pyrene on uterine weight (decreased in neonatal rats and increased in immature rats), interactions with ERα have been proposed, possibly via occupation of ERα binding sites or via AhR-ER-crosstalk (Kummer et al., 2008; Kummer et al., 2007). However, several in vitro studies have demonstrated low affinity binding of benzo[a]pyrene to the estrogen receptor and alteration of estrogen-dependent gene expression (Liu et al., 2006; van Lipzig et al., 2005; Vondrácek et al., 2002; Fertuck et al., 2001; Charles et al., 2000), so the role of the ER in benzo[a]pyrene-induced reproductive toxicity is unclear.

Some evidence in rodents suggests decrements in female fertility from benzo[a]pyrene exposure may be related to disruption in the estrous cycle and the balance of reproductive hormones (Zhao et al., 2014; Archibong et al., 2012). One study in rats noted inhalation exposure prior to mating resulted in alterations in reproductive hormones, estrous cycle length, decreased ovulation rate, and the production of smaller litters (Archibong et al., 2012). Another study dosed mice following mating and prior to implantation (GDs 1–5) and noted that i.p. doses of benzo[a]pyrene as low as 0.2 mg/kg increased plasma estrogen and progesterone, altered the morphology of the endometrium, and decreased the number of implantation sites and the overall pregnancy rate in mice (Zhao et al., 2014).

Summary of Reproductive Effects

Male reproductive effects

Exposure to benzo[a]pyrene in laboratory animals induces male reproductive effects including decreased levels of testosterone and increased levels of LH, decreased sperm count and motility, histological changes in the testis, and decreased reproductive success. These findings in animals are supported by decrements in sperm quality and decreased fertility in human populations exposed to PAH mixtures (Soares and Melo, 2008; Hsu et al., 2006). In laboratory animals, male reproductive toxicity has been observed after oral and inhalation exposure to rats or mice. Effects seen after oral exposures include impaired fertility, effects on sperm parameters, decreased reproductive organ weight, testicular lesions, and hormone alterations (Chen et al., 2011; Chung et al., 2011; Mohamed et al., 2010; Zheng et al., 2010; Mackenzie and Angevine, 1981). In addition to oral exposure, male reproductive effects of benzo[a]pyrene have also been observed following inhalation exposure in rats (Archibong et al., 2008; Ramesh et al., 2008; Inyang et al., 2003). The male reproductive effects associated with benzo[a]pyrene exposure are considered to be biologically plausible and adverse.

In conclusion, EPA identified male reproductive system effects as a human hazard of benzo[a]pyrene exposure.

Female reproductive effects

A large body of mechanistic data, both in vivo and in vitro, suggests that benzo[a]pyrene impacts fertility through the disruption of folliculogenensis. This finding is supported, albeit indirectly, by observations of premature ovarian senescence in women exposed to cigarette smoke (Midgette and Baron, 1990). Evidence for female reproductive toxicity of benzo[a]pyrene comes from studies of human populations exposed to PAH mixtures as well as laboratory animal and in vitro studies. In addition, two human studies observed associations specifically between benzo[a]pyrene measures and two fertility-related endpoints: decreased ability to conceive (Neal et al., 2008; Neal et al., 2007) and increased risk of early fetal death (i.e., before 14 weeks of gestation) (Wu et al., 2010). Studies in multiple strains of rats and mice indicate fertility-related effects including decreases in ovarian follicle populations and decreased fecundity. Decreased serum estradiol has also been noted in two different strains of rats exposed by oral or inhalation exposure. The reproductive effects associated with benzo[a]pyrene exposure are biologically supported and relevant to humans.

In conclusion, EPA identified female reproductive effects as a human hazard of benzo[a]pyrene exposure.

Susceptible Populations and Lifestages

Epidemiological studies indicate that exposure to complex mixtures of PAHs, such as through cigarette smoke, is associated with measures of decreased fertility in humans (Neal et al., 2008; El-Nemr et al., 1998) and that prenatal exposure to cigarette smoking is associated with reduced fertility of women later in life (Weinberg et al., 1989). A case-control study in a Chinese

population has also indicated that women with elevated levels of benzo[a]pyrene-DNA adducts in maternal blood were 4 times more likely to have experienced a miscarriage (Wu et al., 2010).

Evidence from animal models indicate that benzo[a]pyrene exposure can decrease the ability of females to maintain pregnancy. Inhalation exposure of pregnant female rats to benzo[a]pyrene:carbon black aerosols on GDs 11–20 caused an increase in embryo/fetal resorptions, as evidenced by decreased litter size in pregnant animals treated after implantation (Archibong et al., 2002). Evidence for increased embryo/fetal resorptions was also demonstrated by decreased litter size following high dose oral exposure of pregnant mice on GDs 7-16 (Mackenzie and Angevine, 1981). In addition to effects observed following in utero exposure, decreased production of offspring has also been observed with benzo[a]pyrene administration prior to mating (Archibong et al., 2012; Mattison et al., 1980). Reduced litter sizes at birth, was observed following a 14-day pre-mating inhalation exposure period, and was associated with decreases in ovulation rate (Archibong et al., 2012). A continuous breeding study also noted a decreased production of offspring in mice following a single i.p. exposure to to benzo[a]pyrene two weeks prior to mating (Mattison et al., 1980).

Oral multigenerational studies of benzo[a]pyrene exposure in mice demonstrated effects on fertility and the development of reproductive organs (decreased ovary and testes weight) in both male and female offspring of pregnant mice exposed to 10–160 mg/kg-day on GDs 7–16 (Kristensen et al., 1995; Mackenzie and Angevine, 1981). Persistent reductions in sperm parameters have also been observed in SD rats following early gestational exposure (GDs 1–7) to 5–10 mg/kg-day (Liang et al., 2012).

Reductions in female fertility associated with decreased ovary weight and follicle number following gestational exposure (as discussed in Section 1.1.1) are supported by observations of: (1) destruction of primordial follicles (Borman et al., 2000; Mattison et al., 1980) and decreased corpora lutea (Miller et al., 1992; Swartz and Mattison, 1985) in adult female mice following i.p. exposure; (2) decreased ovary weight in adult female rats following oral exposure (Xu et al., 2010); and (3) stimulation of oocyte apoptosis (Matikainen et al., 2002; Matikainen et al., 2001) or by a decreased sensitivity to FSH-stimulated follicle growth (Neal et al., 2007).

Reductions in male fertility associated with decreased testes weight following gestational exposure (as discussed in Section 1.1.1) are supported by observations of: (1) decreased sperm count, altered serum testosterone levels, testicular lesions, and/or increased numbers of apoptotic germ cells in adult rats following repeated oral exposure to benzo[a]pyrene (Chung et al., 2011; Chen et al., 2010; Zheng et al., 2010; Arafa et al., 2009); (2) decreased epididymal sperm counts in adult F0 and F1 generations of male mice following 6 weeks of oral exposure of the F0 animals to benzo[a]pyrene (Mohamed et al., 2010); and (3) decreased testis weight, decreased testicular or plasma testosterone levels, and/or decreased sperm production, motility, and density in adult male rats following repeated inhalation exposure to aerosols of benzo[a]pyrene:carbon black (Archibong et al., 2008; Ramesh et al., 2008; Inyang et al., 2003).

1.1.3. Immunotoxicity

No evidence of an association between benzo[a]pyrene exposure in utero and allergic sensitization was found in the one available human study of developmental immunotoxicity that provided data specific to benzo[a]pyrene (Jedrychowski et al., 2011) (Table 1-8). No other human studies evaluating immune effects following exposure to benzo[a]pyrene alone are available for any route of exposure. However, a limited number of occupational human studies, particularly in coke oven workers (Zhang et al., 2012; Wu et al., 2003b; Winker et al., 1997; Szczeklik et al., 1994), show effects on immune parameters associated with exposure to PAH mixtures. These studies are of limited utility because effects associated specifically with benzo[a]pyrene cannot be distinguished from other constituents of the PAH mixture. Subchronic and short-term animal studies have reported immunotoxic effects of benzo[a]pyrene by multiple routes of exposure (Table 1-9 and Figure 1-6). Effects include changes in thymus weight and histology, decreased B cell percentages and other alterations in the spleen, and immune suppression. Data obtained from subchronic oral gavage studies are supported by short-term, i.p., intratracheal, and subcutaneous (s.c.) studies. Additionally, there is evidence in animals for effects of benzo[a]pyrene on the developing immune system. No studies were located that examined immune system endpoints following inhalation exposure of animals to benzo[a]pyrene.

Thymus Effects

Decreased thymus weights (up to 62% compared to controls) were observed in male and female Wistar rats exposed by gavage to 10–90 mg/kg-day benzo[a]pyrene for 35 or 90 days (Kroese et al., 2001; De Jong et al., 1999). This effect may be due to thymic atrophy. The incidence of slight thymic atrophy was increased in males (6/10) and females (3/10) at a dose of 30 mg/kg-day in a 90-day study, although there was no evidence of atrophy at any lower dose (Kroese et al., 2001). Additionally, at the highest dose tested (90 mg/kg-day) in one of the 35-day studies, the relative cortex surface area of the thymus and thymic medullar weight were significantly reduced (De Jong et al., 1999). Other histopathological changes in the thymus (increased incidence of brown pigmentation of red pulp; hemosiderin) were observed in Wistar rats of both sexes at 50 mg/kg-day in a 35-day study; however, this tissue was not examined in intermediate-dose groups (Kroese et al., 2001). Consistent with the effects observed in these studies, decreased thymus weights and reduced thymic cellularity were observed in i.p. injection studies that exposed mice to doses ranging from 50 to 150 mg/kg in utero (Holladay and Smith, 1995, 1994; Urso and Johnson, 1988).

Spleen Effects

Reduced splenic cellularity, indicated by decreased relative and absolute number of B cells in the spleen (decreased up to 41 and 61% compared to controls, respectively) and decreased absolute number of splenic cells (31% decrease at the highest dose), was observed in a subchronic study in male Wistar rats administered 3–90 mg/kg-day benzo[a]pyrene by gavage for 35 days (De

Jong et al., 1999). While the effect on the relative number of B cells was dose-related, the lower doses did not affect the number of B cells or the absolute splenic cell number. The reduced splenic cell count at the highest dose was attributed by the study authors to the decreased B cells, and suggests a possible selective toxicity of benzo[a]pyrene to B cell precursors in the bone marrow. The spleen effects observed in De Jong et al. (1999) are supported by observations of reduced spleen cellularity and decreased spleen weights following i.p. injection or in utero benzo[a]pyrene exposure to doses ranging from 50 to 150 mg/kg (Holladay and Smith, 1995; Urso et al., 1988).

In addition to physical effects on the spleen, several studies have demonstrated functional suppression of the spleen following benzo[a]pyrene exposure. Dose-related decreases in sheep red blood cell (SRBC) specific serum IgM levels after SRBC challenge were reported in rats (10 or 40 mg/kg-day) and mice (5, 20,or 40 mg/kg-day) following s.c. injection of benzo[a]pyrene for 14 days (Temple et al., 1993). Similarly, reduced spleen cell responses, including decreased numbers of plaque forming cells and reduced splenic phagocytosis to SRBC and lipopolysaccharide challenge, were observed in $B6C3F_1$ mice exposed to doses ≥40 mg/kg-day benzo[a]pyrene by i.p. or s.c. injection for 4–14 days (Lyte and Bick, 1985; Dean et al., 1983; Munson and White, 1983) or by intratracheal instillation for 7 days (Schnizlein et al., 1987).

Immunoglobulin Alterations

Alterations in immunoglobulin levels have been associated with exposure to PAH mixtures in a limited number of human studies. Some occupational studies have reported evidence of immunosuppression following PAH exposure. For example, reductions in serum IgM and/or IgA titers were reported in coke oven workers (Wu et al., 2003b; Szczeklik et al., 1994). Conversely, immunostimulation of immunoglobulin levels has also been observed in humans, specifically elevated IgG (Karakaya et al., 1999) and elevated IgE (Wu et al., 2003b) following occupational PAH exposure.

Decreases in serum IgM (13–33% compared to controls) and IgA levels (22–61% compared to controls) were observed in male Wistar rats exposed to 3–90 mg/kg-day benzo[a]pyrene by gavage for 35 days (De Jong et al., 1999); however, these reductions were consistent, but not dose-dependent. Similarly, reductions in IgA (9–38% compared to controls) were also observed in male and female $B6C3F_1$ mice exposed to doses of 5–40 mg/kg benzo[a]pyrene by s.c. injection for 14 days (Munson and White, 1983). Reductions in serum IgG levels of 18–24%, although not statistically significant, were observed in female $B6C3F_1$ mice exposed to doses ≥50 mg/kg benzo[a]pyrene by i.p. injection for 14 days (Dean et al., 1983).

Hematological Alterations

Altered hematological parameters, including decreases in red blood cell (RBC) count, hemoglobin, and hematocrit have been observed in laboratory animals following benzo[a]pyrene exposure (Table 1-9). Statistically significant decreases in RBC count, hemoglobin, and hematocrit were observed in male Wistar rats at doses ≥10 mg/kg-day for 35 days (De Jong et al., 1999). A

minimal, but statistically significant increase in mean cell volume and a decrease in mean cell hemoglobin were observed at the highest dose (90 mg/kg-day), which may indicate dose-related toxicity for the RBCs and/or RBC precursors in the bone marrow (De Jong et al., 1999). Similarly, male and female F344 rats also showed maximal decreases in RBC counts, hematocrit, and hemoglobin levels between 10 and 12% in a 90-day dietary study (Knuckles et al., 2001). Findings were significant for RBC counts and hematocrit in males at ≥50 mg/kg-day, while decreased RBC counts and hematocrit in females and hemoglobin levels in both sexes were only significant in the 100 mg/kg-day group (Knuckles et al., 2001). Small, but not statistically significant, decreases in RBC counts and hemoglobin were observed in both 35- and 90-day studies in Wistar rats (Kroese et al., 2001). It should be noted that when observed, the magnitudes of the decreases in RBCs, hemoglobin, and hematocrit were generally small; about 18% at 90 mg/kg-day and <10% at lower doses (De Jong et al., 1999) and about 10% in F344 rats (Knuckles et al., 2001). A decrease in white blood cells (WBCs), attributed to reduced numbers of lymphocytes and eosinophils, was also observed at 90 mg/kg-day following gavage exposure for 35 days (De Jong et al., 1999).

Immune Suppression and Sensitization

Some occupational studies of coke oven emissions have reported evidence of immunosuppression following PAH exposure. Reduced mitogenic responses in T cells (Winker et al., 1997) and reduced T-lymphocyte proliferative responses (Karakaya et al., 2004) have been observed following occupational exposure to PAH. Increased levels of apoptosis were observed in the peripheral blood mononuclear cells (a population of lymphocytes and monocytes) of occupationally exposed coke oven workers, which is a response that may contribute to immunodeficiency in this population (Zhang et al., 2012). However, a limitation of this study is that it does not attribute the proportion of apoptotic activity to a specific class of cells and does not include assessment of other potential markers of immunotoxicity in peripheral blood.

Results of functional immune assays in laboratory animals following short-term i.p. and s.c. exposures add to the evidence for benzo[a]pyrene immunotoxicity. Resistance to Streptococcus pneumonia or Herpes simplex type 2 was dose dependently reduced in $B6C3F_1$ mice following s.c. injection of ≥5 mg/kg-day benzo[a]pyrene for 14 days (Munson et al., 1985). Reduced cell proliferation, IFN-γ release, and IL-4 release were observed in male and female C56BL/6 mice following short-term exposure to a gavage dose of 13 mg/kg benzo[a]pyrene as measured in a modified local lymph node assay (van den Berg et al., 2005). A statistically significant decrease in natural killer cell activity was observed in male Wistar rats (Effector:Target cell ratio was 40.9 ± 28.4% that of controls) exposed to 90 mg/kg-day by gavage for 35 days (De Jong et al., 1999); however, splenic natural killer cell activity was not affected in $B6C3F_1$ mice after s.c. injection of 40 mg/kg-day benzo[a]pyrene for 14 days (Munson et al., 1985). The magnitude of the dose and duration of the exposure may account for the discrepancy between these two studies. Single i.p. injections of 50 mg/kg benzo[a]pyrene decreased pro- and/or pre-B-lymphocytes and

neutrophils in the bone marrow of C57BL/6J mice without affecting the numbers of immature and mature B-lymphocytes or GR-1+ myeloid cells (Galván et al., 2006).

In contrast to studies that have shown immunosuppression, benzo[a]pyrene may also induce sensitization responses. Epicutaneous abdominal application of 100 µg benzo[a]pyrene to C3H/HeN mice, followed by ear challenge with 20 µg benzo[a]pyrene 5 days later, produced a contact hypersensitivity (a significant ear swelling) response (Klemme et al., 1987).

Developmental Immunotoxicity

No evidence of an association between benzo[a]pyrene exposure in utero and allergic sensitization was found in the one available human study of developmental immunotoxicity that provided data specific to benzo[a]pyrene (Jedrychowski et al., 2011). This birth cohort study found no statistically significant difference in maternal cord blood levels of benzo[a]pyrene-DNA adducts between 5-year-old children with or without dermal atopy to one of four common allergens, and no statistically significant association when the data were analyzed using a logistic regression model that adjusted for children's gender, maternal age, maternal education, maternal atopy, and ETS (Jedrychowski et al., 2011) (Table 1-8). In a New York City birth cohort study, statistically significant associations were found between serum levels of cockroach IgE (an indicator of hypersensitivity) in 5- and 9-year-old children and high urinary levels of metabolites of three PAHs (naphthalene, phenanthrene, and pyrene), but this study did not examine exposure metrics specific to benzo[a]pyrene (Jung et al., 2015). Another birth cohort study found an association between increased methylation of promoter regions of the interferon gene (IFNγ, an important gene in the etiology of allergic asthma) in DNA from cord WBCs and personal air measures of maternal exposure to several carcinogenic PAHs including benzo[a]pyrene, but possible associations with benzo[a]pyrene exposure measures alone were not evaluated (Tang et al., 2012).

As noted above, several i.p. injection studies suggest that cell-mediated and humoral immunity may be altered by exposure to high doses of benzo[a]pyrene during gestation. Suppression of the mixed lymphocyte response, the graft-versus-host response, and suppression of the plaque-forming cell response to SRBCs was observed in mice exposed in utero to 150 mg/kg during mid (GDs 11–13), late (GDs 16–18), or both (GDs 11–17) stages of gestation; these effects persisted until 18 months of age (Urso and Gengozian, 1984, 1982, 1980). Fetal thymic atrophy, as assessed by reductions in cellularity (74–95%, compared to controls), was observed in mice exposed to 50–150 mg/kg benzo[a]pyrene from GD 13 to 17, when examined on GD 18 (Holladay and Smith, 1994). Analysis of cell surface markers (e.g., CD4, CD8) from the same study indicate that benzo[a]pyrene may inhibit and/or delay thymocyte maturation, possibly contributing to the observed thymic atrophy (Holladay and Smith, 1994). Consistent with these findings, several other studies have noted decreased thymocyte numbers and disrupted T cell maturation after in utero exposure to benzo[a]pyrene (Rodriguez et al., 1999; Holladay and Smith, 1995; Lummus and Henningsen, 1995; Urso et al., 1992; Urso and Johnson, 1987).

The fetal liver is the primary hematopoietic organ during gestation and a major source of thymocyte precursors beginning around GD 10 or 11 in mice (Landreth and Dodson, 2005; Penit and Vasseur, 1989). Statistically significant reductions in total cellularity in the fetal liver of 54 and 67% were reported in offspring after i.p. exposures of 50 or 100 mg/kg benzo[a]pyrene, respectively, to the dams on GDs 13–17 (Holladay and Smith, 1994). The decreased fetal liver cellularity was accompanied by decreased expression of terminal deoxynucleotidyl transferase and CD45R cellular markers, which are known to be present in cortical thymocyte progenitors in the fetal liver (Holladay and Smith, 1994; Fine et al., 1990; Silverstone et al., 1976). These data also suggest that benzo[a]pyrene disrupts liver hematopoiesis during gestation and may interfere with prolymphoid seeding of the thymus, possibly contributing to thymic atrophy and cell-mediated immunosuppression. Decreased numbers of $CD4^+$ T-cells have been reported in the spleen of 1-week-old mice following in utero benzo[a]pyrene exposure by i.p. injection to the dams, demonstrating the potential for downstream effects on T-cell development (Rodriguez et al., 1999). The decreased numbers of CD4+ T-cells correspond with observations of decreased proliferation in the presence of Concanavalin A and a weak response compared to controls in an allogeneic mixed lymphocyte reaction assay (Urso and Kramer, 2008).

Postnatal exposure to benzo[a]pyrene has also been suggested to cause immune effects. Dose-dependent decreases in erythrocytes (attributed to reduced bone marrow erythropoiesis), as well as reduced expression of IL-4 and IFN-γ were observed in the pups of Wistar rats exposed to 0.1–10 mg/kg-day benzo[a]pyrene by subcutaneous injection for 14 days (Matiasovic et al., 2008). This finding suggests that benzo[a]pyrene may alter the immune response to infection or vaccination in developing animals.

Table 1-8. Evidence pertaining to immune effects of benzo[a]pyrene in humans

Study design and reference	Results		
Jedrychowski et al. (2011) (Krakow, Poland) Birth cohort (5-year follow-up) 224 women who delivered full-term babies between January 2001 and February 2004 [Skin prick testing for four aeroallergens (Dermatophagiodes pteronyssinus, Dematophagoides farina, dog hair, cat hair) in children at 5 yrs of age]	Benzo[a]pyrene adduct levels (/10^8 nucleotides) in cord blood from mothers of children with and without dermal atopy at 5 years of age		
		Geometric mean	(95% CI)
	Atopic (N = 37)	0.23	(0.21–0.24)
	Non-atopic (N = 187)	0.21	(0.18–0.25)
	Association between benzo[a]pyrene-DNA adducts and atopic status[a]		
		RR	(95% CI)
	Relative risk for atopy	0.12	(0.01–1.85)

Study design and reference	Results
Exposure: Benzo[a]pyrene-DNA adducts in cord blood samples; geometric mean 0.22 (95% CI 0.21−0.24) adducts/ 10^{-8} nucleotides; DNA adducts with benzo[a]pyrene tetraols were determined with an HPLC fluorometric assay	[a]RR calculated from a binary outcome logistic regression model, adjusted for child's gender, parity, maternal age, maternal education, maternal atopy, and ETS; positive atopic status defined as positive skin prick test to at least one tested aeroallergen

1

Table 1-9. Evidence pertaining to the immune effects of benzo[a]pyrene in animals after oral or inhalation exposure

Reference and study design	Results[a]
Thymus effects	
Kroese et al. (2001) Wistar rats, 10/sex/dose 0, 3, 10, or 30 mg/kg-d by gavage 5 d/wk 90 d	↓ thymus weight Females (% change from control): 0, −3, −6, and −28* Males (% change from control): 0, 0, −13, and −29* ↑ slight thymic atrophy Females (incidence): 0/10, 0/10, 0/10, and 3/10 Males (incidence): 0/10, 2/10, 1/10, and 6/10*
De Jong et al. (1999) Wistar rats, 8 males/dose 0, 3, 10, 30, or 90 mg/kg-d by gavage 5 d/wk 35 d	↓ thymus weight % change from control: 0, −9, −15*, −25*, and −62*
Kroese et al. (2001) Wistar rats, 10/sex/dose 0, 1.5, 5, 15, or 50 mg/kg-d by gavage 5 d/wk 35 d	↓ thymus weight Females (% change from control): 0, 13, 8, −3, and −17* Males (% change from control): 0, −8, −11, −27*, and −33*
Spleen effects	
De Jong et al. (1999) Wistar rats, 8 males/dose 0, 3, 10, 30, or 90 mg/kg-d by gavage 5 d/wk 35 d	↓ relative number (%) of B cells in spleen % change from control: 0, −8, −13*, −18*, and −41* ↓ total number of B cells in spleen % change from control: 0, 13, −13, −13, and −61* Change in total cell number in the spleen % change from control: 0, 20, 0, +7, and −31*
Immunoglobulin alterations	
De Jong et al. (1999) Wistar rats, 8 males/dose 0, 3, 10, 30, or 90 mg/kg-d by gavage 5 d/wk 35 d	↓ serum IgM % change from control: 0, −13, −14, −33*, and −19 ↓ serum IgA % change from control: 0, −27, −22, −28, and −61*
Hematological alterations	
Kroese et al. (2001) Wistar rats, 10/sex/dose 0, 3, 10, or 30 mg/kg-d by gavage 5 d/wk 90 d Wistar rats, 10/sex/dose 0, 1.5, 5, 15, or 50 mg/kg-d by gavage 5 d/wk for 35 d	RBC count and hemoglobin changes not statistically significant in males or females at any dose (numerical data not reported) RBC count: changes not statistically significant (numerical data not reported) Hemoglobin: changes not statistically significant (numerical data not reported)

Reference and study design	Results[a]
Knuckles et al. (2001) F344 rats, 20/sex/dose 0, 5, 50, or 100 mg/kg-d by diet 90 d	↓ RBC count Females (% change from control): statistically significant at 100 mg/kg-d (numerical data not reported) Males (% change from control): statistically significant at 50 and 100 mg/kg-d (numerical data not reported) ↓ hematocrit Females (% change from control): statistically significant at 100 mg/kg-d (numerical data not reported) Males (% change from control): statistically significant at 50 and 100 mg/kg-d (numerical data not reported) ↓ hemoglobin Females: statistically significant at 100 mg/kg-d (numerical data not reported) Males: statistically significant at 100 mg/kg-d (numerical data not reported)
De Jong et al. (1999) Wistar rats, 8 males/dose 0, 3, 10, 30, or 90 mg/kg-d by gavage 5 d/wk 35 d	↓ RBC count % change from control: 0, −1, −5*, −10*, and −18* ↓ hemoglobin % change from control: 0, −1, −7*, −10*, and −18* ↓ hematocrit % change from control: 0, 0, −6*, −8*, and −14* ↓ WBC count % change from control: 0, −8, −9, −9, and −43* ↑ mean cell volume % change from control: 0, 0, −3, 0, and 3* ↓ mean corpuscular hemoglobin concentration % change from control: 0, −1, −1, −1, and −3*

*Statistically significantly different from the control ($p < 0.05$).

[a]% change from control calculated as: (treated value − control value)/control value × 100.

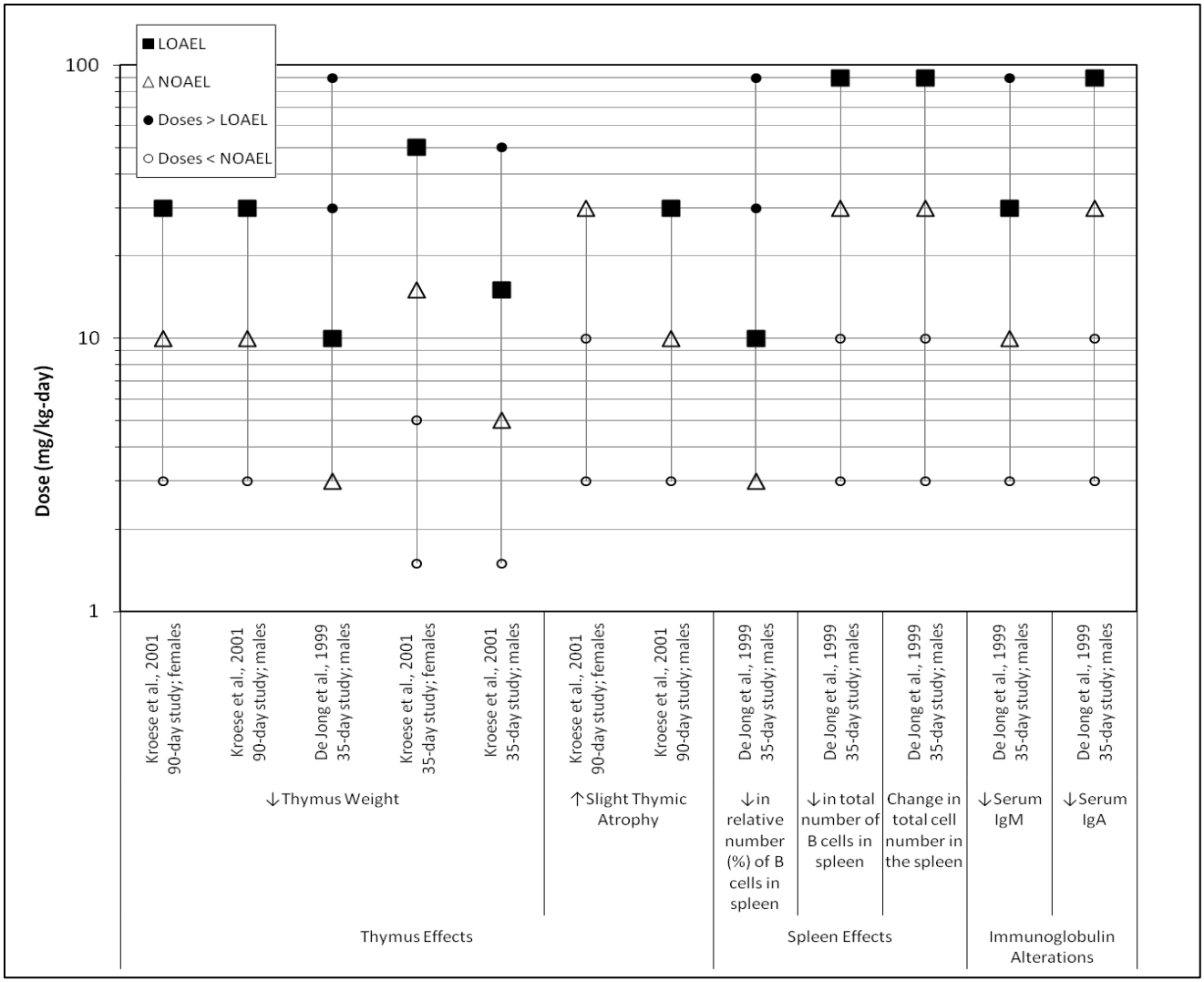

Figure 1-6. Exposure-response array for immune effects following oral exposure.

Mode-of-Action Analysis—Immune Effects

Exposure to benzo[a]pyrene induces immunosuppressive effects such as decreased numbers of B cells in the spleen and decreased thymus weight and cellularity following oral, i.p., s.c., or intratracheal exposure in experimental animals. The key events underlying benzo[a]pyrene immunotoxicity have not been determined definitively, but likely involve AhR activation, metabolism, as well as genotoxicity, mutagenicity, cytotoxicity and apoptosis.

Benzo[a]pyrene is well established as a genotoxic agent (see Section 1.1.5) and benzo[a]pyrene-DNA adducts are routinely detected in WBCs of humans occupationally exposed to PAH mixtures (see Table 1-20). Increased apoptosis has been detected peripheral blood mononuclear cells of coke oven workers highly exposed to PAH mixtures (Zhang et al., 2012). It is

therefore likely that exposure can result in cellular damage, cell death, and mutations in immune cell populations, potentially altering immune function.

Benzo[a]pyrene is also a well-known ligand for the AhR (Okey et al., 1994; Nebert et al., 1993; Postlind et al., 1993). Ligands of the AhR have been shown to have a role in regulating hematopoietic stem cells in the bone marrow, a major site of B cell proliferation and antibody production (Esser, 2009). Benzo[a]pyrene reduced B cell lymphopoiesis at concentrations as low as 10nM (Hardin et al., 1992). Furthermore, Ah-responsive (C57BL/6) mice showed greater dose-dependent reductions in B cell lymphopoiesis than those observed in Ah-nonresponsive (DBA/2) mice (Hardin et al., 1992). Addition of the AhR antagonist and CYP450 inhibitor, α-naphthaflavone, inhibited the benzo[a]pyrene-induced suppression of B cell lymphopoiesis in a concentration-dependent fashion. Similarly, the CYP1A1 inhibitor, 1-(1-propynyl) pyrene, blocked benzo[a]pyrene-induced B cell growth inhibition but not growth inhibition caused by the benzo[a]pyrene metabolite, BPDE; these data suggest that a CYP1A1-dependent metabolite of benzo[a]pyrene is responsible for the B cell growth suppressive effects observed after benzo[a]pyrene exposure (Allan et al., 2006).

T-cells also appear to be similarly sensitive to benzo[a]pyrene. In vitro assays in human peripheral blood T cell indicate that low concentrations of benzo[a]pyrene (10−100 nM) suppress T cell mitogenesis (using phytohemagglutinin or concanavalin A) and that the mechanism involves AhR and P450 related processes (Davila et al., 1996; Mudzinski, 1993).

Altogether, these data suggest that benzo[a]pyrene may depress B and Tcell proliferation via the AhR and metabolism of benzo[a]pyrene to reactive metabolites.

Summary of Immune Effects

Evidence for immunotoxic effects of benzo[a]pyrene exposure comes from animal studies that vary in route and duration of exposure. There are no human epidemiological studies that provide specific support for benzo[a]pyrene immunotoxicity; however, immunosuppression has been observed in studies following occupational exposure to PAH mixtures. However, these findings are limited by co-exposures to other constituents of PAH mixtures.

Effects such as altered thymus weight and histology, spleen effects, and altered immunoglobulin levels observed by the oral route reported in animal bioassays provide some evidence of immunotoxicity following benzo[a]pyrene exposure; however, in vivo functional assays provide stronger support for immunotoxicity (WHO, 2012). The immunological changes observed in the available subchronic gavage studies are supported by a larger database of in vivo studies of benzo[a]pyrene (by parenteral exposure) indicating functional immunosuppression such as decreased proliferative responses to antigens and decreased resistance to pathogens or tumor cells (Kong et al., 1994; Blanton et al., 1986; Munson et al., 1985; White et al., 1985; Dean et al., 1983; Munson and White, 1983).

Few studies are available to inform the effects of benzo[a]pyrene on the developing immune system. In particular, there is a lack of studies evaluating functional changes in the immune system

following developmental exposure, especially following oral or inhalation exposures. An additional datagap includes the lack of studies which treated offspring postnatally, up to approximately PND 45, when the immune system continues to develop (Burns-Naas 2008). However, the available i.p. exposure studies of gestationally and early postnatally treated animals provide a strong indication of potential development immunotoxicity and suggest the need for further study.

Although the key events underlying the mode of action of benzo[a]pyrene immunotoxicity are not firmly established, there is evidence of physical alterations to tissues/organs of the immune system, as well as decreases in immune function. Evidence of benzo[a]pyrene-associated immunotoxicity is supported by consistent thymic effects observed in two oral studies, as well as splenic effects, and varying immunosuppressive responses observed in short-term or in vitro tests.

EPA concluded there was suggestive evidence that immunotoxicity is a potential human hazard of benzo[a]pyrene exposure.

Susceptible Populations and Lifestages

The severity and persistence of immune effects observed during in utero studies suggests that immunotoxicity may be greater during early life than adulthood (Dietert and Piepenbrink, 2006; Holladay and Smialowicz, 2000; Urso and Gengozian, 1982). Urso and Gengozian (1982) provide experimental support demonstrating that immunosuppression from benzo[a]pyrene exposure during gestation was greater than for mice exposed after birth to a 25-fold higher dose. There is also substantial literature indicating that disruption of the immune system during certain critical periods of development (e.g., initiation of hematopoiesis, migration of stem cells, expansion of progenitor cells) may have significant and lasting impacts on lifetime immune function (e.g., Burns-Naas et al., 2008; Dietert, 2008; Landreth, 2002; Dietert et al., 2000). In addition, chemical-specific studies show increased dose sensitivity and disease persistence from developmental versus adult chemical exposure (reviewed in Luebke et al., 2006).

Thymus toxicity is a sensitive and specific effect of benzo[a]pyrene and has been observed in both prenatal and adult exposure studies. The thymus serves as a major site of thymocyte proliferation and selection for maturation, and impairment can lead to cell-mediated immune suppression (Kuper et al., 2002; De Waal et al., 1997; Kuper et al., 1992). The thymus is believed to be critical for T lymphocyte production during early life and not in adulthood (Hakim et al., 2005; Schönland et al., 2003; Petrie, 2002; Mackall et al., 1995). Therefore, the decreases in thymus weight observed in studies of adult animals exposed to benzo[a]pyrene suggest that immunosuppression may be a heightened concern for individuals developmentally exposed to benzo[a]pyrene.

The available studies evaluating immune effects provide limited comparisons for sensitivity of immune effects of benzo[a]pyrene across species and sexes. Two subchronic oral studies evaluated immune endpoints in rats (Kroese et al., 2001; De Jong et al., 1999), however, no subchronic or chronic studies were identified in mice by the oral or inhalation route. One study in rats evaluated immune endpoints in both males and female rats (Kroese et al., 2001). In this study,

similar effects were seen in male and female rats, with some indication of effects in the thymus occurring in males at a slightly lower dose (Kroese et al., 2001).

In addition, few studies are available which directly compare immunotoxicity of benzo[a]pyrene across species or strains. A study in female F344 rats and B6C3F1 mice noted a similar degree of immune suppression in rats and mice using the SRBC functional assay after 14-day i.p. exposure to benzo[a]pyrene (Temple et al., 1993). Another study evaluated benzo[a]pyrene in several in vitro assays for immunotoxicity in human and rodent cells. The endpoints assessed included cytoxicity, cytokine release, myelotoxicity, and mitogen responsiveness. The authors reported that benzo[a]pyrene immunotoxicity was generally similar in mice, rats, and human cells with IC50 values in the same range (Carfi' et al., 2007).

1.1.4. Other Toxicity

There is some evidence that benzo[a]pyrene can produce non-cancer effects in the liver, kidney, cardiovascular system and nervous system in animals exposed as adults (effects on these organ systems following developmental exposure are discussed as part of Section 1.1.1 "Developmental Toxicity"). However, there is less robust and consistent evidence for these effects as compared to organ systems described earlier in Sections 1.1.1–1.1.3. Therefore, at this time, no conclusions are drawn regarding these effects as human hazards of benzo[a]pyrene exposure. A brief overview of the evidence pertaining to these organ systems, with particular focus on findings from subchronic or chronic oral and inhalation exposures, is included below.

Liver Effects

Liver effects other than cancer associated with benzo[a]pyrene exposure primarily include changes in liver weight (Table 1-10). Increased liver weight was reported in a 90-day study in both male and female Wistar rats given benzo[a]pyrene by gavage (Kroese et al., 2001). Both females (17% increase) and males (29% increase) demonstrated statistically significant increased liver weights at the highest dose tested (30 mg/kg-day); a statistically significant increase (15%) was also reported in males at 10 mg/kg-day. Similar to the findings in the 90-day study by Kroese et al. (2001), increased liver:body weight ratios were observed at the highest dose in a 90-day dietary study in male F344 rats, although there was no change observed in female liver weights (Knuckles et al., 2001). Increased liver:body weight ratios were also observed in both sexes at high doses (600 and 1,000 mg/kg) in an accompanying acute study (Knuckles et al., 2001). A statistically significant increase in liver weight was also observed in male Wistar rats given 90 mg/kg-day benzo[a]pyrene by gavage for 35 days (De Jong et al., 1999). Consistent with the findings by De Jong et al. (1999), a statistically significant increased liver weight (about 18%) was also observed in both male and female Wistar rats at the highest dose (50 mg/kg-day) given by gavage in a 35-day study (Kroese et al., 2001).

Limited exposure-related differences in clinical chemistry parameters associated with liver toxicity were observed; no differences in alanine aminotransferase or serum aspartate

transaminase levels were observed, and a small dose-related decrease in γ-glutamyl transferase was observed in males only exposed to benzo[a]pyrene for 90 days (Kroese et al., 2001).

Treatment-related lesions in the liver (oval cell hyperplasia) were identified as statistically significantly increased following exposure to 90 mg/kg-day benzo[a]pyrene for 35 days; however, incidence data were not reported (De Jong et al., 1999). A 2-year carcinogenicity study (Kroese et al., 2001) observed some histopathological changes in the liver; however, organs with tumors were not evaluated. Since many of the animals in the highest two doses developed liver tumors, the dose responsiveness of the histological changes is unclear.

A dose-dependent increase in liver microsomal ethoxyresorufin-o-deethylase (EROD) activity, indicative of CYP1A1 induction, was observed in both sexes at doses ≥1.5 mg/kg-day in a 35-day study (Kroese et al., 2001). However, at the highest dose tested, with the greatest fold induction in EROD activity, there was no evidence of associated adverse histopathologic findings. Overall, increased liver weight was reported across a few studies of varying exposure durations providing some evidence of the liver as a target of benzo[a]pyrene exposure, however, these changes in liver weight do not appear to be substantially supported by histological findings or other indicators of hepatoxicity. Therefore, at this time, no conclusion is drawn regarding liver toxicity as a human hazard of benzo[a]pyrene exposure.

Kidney Effects

There is minimal evidence of kidney toxicity following exposure to benzo[a]pyrene (Table 1-10). Statistically significant decreases in kidney weight were observed at doses of 3, 30, and 90 mg/kg-day, but not at 10 mg/kg-day, in a 35-day gavage study in male Wistar rats (De Jong et al., 1999). In a 35-day gavage study with a similar dose range in male and female Wistar rats, no statistically significant changes in kidney weights were observed at any dose (Kroese et al., 2001). Histopathological analysis of kidney lesions revealed an apparent dose-responsive increase in the incidence of abnormal tubular casts in the kidney in male F344 rats exposed by diet for 90 days (Knuckles et al., 2001). The casts were described as molds of distal nephrons lumen and were considered by the study authors to be indicative of renal dysfunction. However, the statistical significance of the kidney lesions is unclear. Several gaps and inconsistencies in the reporting make interpretation of the kidney effects difficult, including: (1) no reporting of numerical data; (2) no indication of statistical significance in the accompanying figure for kidney lesions; (3) discrepancies between the apparent incidences and sample sizes per dose group; and (4) uncertainty in how statistical analysis of histopathological data was applied. As such, the significance of the abnormal tubular casts is unclear. One study indicated that gestational exposure may result in changes in kidney histology and function in mice with a highly inducible Ah receptor (C57BL/6), but not in another strain of mice (D2N) with a less inducible Ah receptor (Nanez et al., 2011). At 52 weeks, offspring of C57BL/6 dams treated with 0.1 or 0.5 mg/kg benzo[a]pyrene on GDs 10–13 were found to have a decreased number of podocytes per glomeruli (at both doses) and an increase in urine albumin in the high-dose group. Another study treated adult female rats with 10 mg/kg

benzo[a]pyrene by i.p. exposure once a week for 16 weeks demonstrated changes in clinical indicators consistent with kidney toxicity including elevations in urinary protein, protein/creatinine ratios, and microalbumin (Nanez et al., 2005).

Overall, few studies by environmentally relevant routes of exposure are available to inform the potential of kidney effects after subchronic or chronic exposure to benzo[a]pyrene, thus at this time, no conclusion is drawn regarding kidney toxicity as a human hazard of benzo[a]pyrene exposure.

Cardiovascular Effects

Several studies of cardiovascular effects in populations highly exposed to benzo[a]pyrene as a component of a complex PAH mixtures are available, however, it is difficult to attribute effects of these exposures to any one component of the mixture. Very limited information is available which evaluates the potential cardiovascular toxicity from subchronic or chronic exposure to benzo[a]pyrene in animal models. Numerous short term exposure studies, studies by less environmentally relevant routes of exposure (e.g., injection, instillation), and in vitro studies were identified in the literature, and while these studies may be useful in understanding potential mechanisms of toxicity, the ability of these data to predict chronic health effects is uncertain.

Atherosclerotic vascular disease and increased risk of cardiovascular mortality have been associated with cigarette smoking (Ramos and Moorthy, 2005; Miller and Ramos, 2001; Thirman et al., 1994) and, to a more limited degree, occupational exposure to PAH mixtures (Friesen et al., 2010; Friesen et al., 2009; Burstyn et al., 2005; Chau et al., 1993). Elevated mortality due to cardiovascular disease was observed in a PAH-exposed occupational population (coke oven plant workers), but elevated cardiovascular mortality was also observed in the non-exposed or slightly exposed populations (Chau et al., 1993). Elevated risks of ischemic heart disease (IHD) were associated with past cumulative benzo[a]pyrene exposure among aluminum smelter workers (with a 5-year lag), although the trend was not statistically significant; there was no observed association with more recent benzo[a]pyrene exposure (Friesen et al., 2010). Elevated risk of mortality from IHD was also associated with cumulative benzo[a]pyrene exposure in a cohort of male asphalt workers (although not statistically significant); the trend in average benzo[a]pyrene exposure and association with IHD was statistically significant, with an approximately 60% increase in risk between the lowest and highest exposure groups (Burstyn et al., 2005). The two studies that associate benzo[a]pyrene exposure with cardiovascular effects (Friesen et al., 2010; Burstyn et al., 2005) rely on statistical models to create exposure groups rather than direct measurement of the cohort under examination. Additionally, while these studies used benzo[a]pyrene exposure groupings for analysis, they cannot address co-exposures that may have occurred in the occupational setting (asphalt or aluminum smelters) or exposures that occurred outside the workplace.

Data on cardiovascular effects of chronic or subchronic benzo[a]pyrene exposure in adult wild-type laboratory animals exposed by environmentally-relevant exposure routes (oral,

inhalation, or dermal) are limited. In C57BL/6J mice fed benzo[a]pyrene at an approximate dose of 12.5 mg/kg-day for 15 weeks in conjunction with an atherogenic diet (high in fat and cholesterol), "modest" atherosclerotic lesions (not further quantified or described, but apparently increased relative to controls) were seen in the aorta (Uno et al., 2014). In a shorter duration study, rats treated by gavage once per week for 2 weeks with 0.05 mg/kg benzo[a]pyrene exhibited significantly higher serum LDL-cholesterol than controls; serum HDL-cholesterol and phospholipids were not affected (Chung and Jung, 2003).

In a developmental study, increased systolic and diastolic blood pressure was observed in the offspring of dams exposed to increasing concentrations of benzo[a]pyrene (Jules et al., 2012) (See "Developmental Toxicity" Section 1.1.1 and Table 1-1). At the highest dose tested (1.2 mg/kg body weight by gavage to the dams), systolic pressures were elevated approximately 50% and diastolic pressures were elevated approximately 80% above controls. An intranasal exposure of 0.01 mg/kg-day benzo[a]pyrene in adult male rats also produced an increase in blood pressure following a 7-day exposure (Gentner and Weber, 2011).

Few in vivo evaluations of the effect of subchronic or chronic benzo[a]pyrene exposure on the development of atherosclerosis have been conducted in wild type animals, by environmentally relevant routes of exposure. However, several studies by other routes of exposure and in genetically predisposed animal models (e.g., Apo E−/− mice) are available. ApoE−/− mice develop spontaneous atherosclerosis, which is thought to be due to enhanced oxidative stress from the lack of ApoE, and appears to be enhanced by benzo[a]pyrene exposure and greatly impacted by Ah receptor binding affinity. Oral exposure to benzo[a]pyrene lead to a 2.5-fold increase over controls in ethenoDNA adducts (stable biomarkers of oxidative stress) in the aortas of ApoE null mice (Godschalk et al., 2003). In a study comparing mice of different Ah receptor inducibility, ApoE−/−mice expressing the high affinity AhR gene (C57BL/6J) had more aortic segments with plaque and more extensive plaque area compared with mice expressing low affinity AhR (B6.D2N-*Ahr*^{d}/J) after 10 weeks of exposure to 10 mg/kg-day benzo[a]pyrene by gavage (Kerley-Hamilton et al., 2012). Evidence for the important role of oxidative stress in benzo[a]pyrene-induced atherosclerosis comes from a study (Yang et al., 2012) comparing cardiovascular effects in ApoE knock-out mice with normal expression of scavengers (SOD and catalase) with those overexpressing human Cu/Zn SOD or catalase. Overexpression of either scavenger significantly inhibited benzo[a]pyrene-induced increases in the area and morphology of atherosclerotic lesions, aortic lipid peroxidation, measures of ROS, and endothelial expression of adhesion molecules (VCAM-1 and ICAM-1). The available studies suggest that benzo[a]pyrene exposure in ApoE−/− mice enhances the progression of atherosclerosis through a general local inflammatory process.

In an 8 week i.p. study, reduced endothelial integrity and increased smooth muscle cell mass, both related to atherosclerosis, have been observed in Sprague-Dawley rats exposed to 10 mg/kg benzo[a]pyrene by i.p. injection (once/week) (Zhang and Ramos, 1997).

A large body of mechanistic data exists for various cardiovascular endpoints in vitro and in mammalian and non mammalian animal models. Available mechanistic evidence suggests that benzo[a]pyrene may influence cardiovascular endpoints through several parallel or interrelated mechanisms related to AhR activation. These candidate mechanisms include: stimulation of oxidative stress/DNA adducts (Yang et al., 2009; Curfs et al., 2005; Curfs et al., 2004; Godschalk et al., 2003; Izzotti et al., 1994); induction of pro-inflammatory genes and genes linked to cardiac hypertrophy (Huang et al., 2014; Gan et al., 2012; Ichihara et al., 2009; Knaapen et al., 2007; Curfs et al., 2005; Curfs et al., 2004; Das et al., 1985), alterations in the metabolism of arachidonic acid to vasoactive eicosanoids (Aboutabl et al., 2011; Aboutabl et al., 2009; Bugiak and Weber, 2009), and modulation of myocyte or endothelial cell intracellular calcium via interaction with β-adrenergic receptors (Mayati et al., 2012a; Mayati et al., 2012b; Irigaray et al., 2006).

While some evidence suggests cardiovascular effects in highly PAH exposed populations and a large body of animal studies suggest potential mechanisms for cardiovascular effects, issues of co-exposure in human studies as well as the lack of experimental animal studies examining cardiovascular endpoints in wild-type laboratory animals exposed by environmentally relevant routes for subchronic or chronic durations make it difficult to characterize the human hazard from exposure to benzo[a]pyrene. Overall, the short duration studies and studies by other routes of exposure (e.g., i.p. and installation), as well as studies in genetically modified, highly susceptible animal strains (e.g., APOE−/− mice) provide suggestive evidence of cardiovascular toxicity associated with benzo[a]pyrene exposure.

Nervous System Effects following Adult Exposure

Note: The evidence for nervous system effects following exposure during development are evaluated under Section 1.1.1. "Developmental Toxicity".

Two studies of nervous system effects in occupational populations highly exposed to benzo[a]pyrene as a component of a complex PAH mixtures are available (Qiu et al., 2013; Niu et al., 2010), however, it is difficult to attribute effects of these exposures to any one component of the mixture. No chronic exposure studies in animal models are available which evaluate nervous system effects following benzo[a]exposure. However, several shorter duration oral studies are available in rats and mice (see Table 1-11). Supplemental information is provided by several i.p. studies. While these studies may be somewhat informative for hazard, the ability of these data to predict chronic health effects by environmentally relevant routes of exposure is uncertain.

Neurobehavioral function and mood state were evaluated in two studies of men occupationally exposed to PAH mixtures (Qiu et al., 2013; Niu et al., 2010). Alterations in neurobehavioral function were evaluated in coke oven workers using the Neurobehavioral Core Test Battery (self-reported symptoms by questionnaire). Air concentrations of benzo[a]pyrene and urinary levels of the PAH metabolite, 1-hydroxypyrene, were used as markers of PAH exposure. In both studies, high occupational exposure in coke oven workers compared to controls was associated with decrements in short-term memory and/or attention in digit span tests. In addition,

Qiu et al. (2013) reported an association between PAH exposure, based on comparisons of coke oven workers versus controls as well as urinary 1- hydroxypyrene measures, and decrements in tests related to sensorimotor coordination (i.e., reaction time, digit symbol, and pursuit aiming tests), as well as lower self-reported health ratings on the profile of mood states questionnaire. However, Niu et al. (2010) did not detect these associations using the same test battery. When comparisons were specific to the air benzo[a]pyrene concentrations defined by work location, Qiu et al. (2013) found an association between increasing benzo[a]pyrene levels and decrements in reaction time and pursuit aiming, while Niu et al. (2010) observed that increasing levels were associated with decreased plasma levels of several neurotransmitters involved in mood, learning, and/or memory (e.g., norepinephrine, dopamine, 5-hydroxytryptamine, aspartic acid, γ-aminobutyric acid). The extent to which plasma concentrations of neurotransmitters predict those in the central nervous system is uncertain.

Alterations in neuromuscular, autonomic, sensorimotor, and electrophysiological endpoints have been reported in adult rats and mice following acute or short-term exposure to benzo[a]pyrene (Bouayed et al., 2009b; Grova et al., 2008; Grova et al., 2007; Saunders et al., 2006; Liu et al., 2002; Saunders et al., 2002; Saunders et al., 2001). Impaired performance in tests of learning and memory (i.e., Morris water maze or novel object recognition) was observed following subchronic gavage exposure to 2 mg/kg-day benzo[a]pyrene in adult rats (Maciel et al., 2014; Chen et al., 2011; Chengzhi et al., 2011) and following short-term i.p. exposure in adult mice (Qiu et al., 2011; Xia et al., 2011; Grova et al., 2007). These findings are somewhat limited, as the oral exposure studies in rats were conducted with only a single dose group. Further, the results of the novel object recognition test were complicated by a lack of automated recording and blinding, no evaluation of total activity, and an apparent preference for the novel object when testing long-term memory 24 hours after training (recognition index was similar to controls) that was not present at 1.5 hours after training (Maciel et al., 2014).

Tests of anxiety- and activity-related behaviors were difficult to interpret. Decreased anxiety-like behavior in the elevated plus maze was observed following short-term i.p. exposure to the high dose of 200 mg/kg-day (a dose at which there was general toxicity), while overall activity appeared to be increased in hole board testing at ≥20 mg/kg-day(Grova et al., 2008). Relatedly, while spontaneous locomotor activity in a 15-minute trial was increased after oral exposure to 2 mg/kg-day in rats (Maciel et al., 2014), no changes were observed during a 5-minute activity trial in mice orally exposed to 0.02–20 mg/kg-day benzo[a]pyrene (Bouayed et al., 2012).

Effects on depressive-like activity were also mixed: animals orally exposed to 0.02 or 0.2 mg/kg-day for 17 days showed decreased immobility time in the tail suspension test, but no effect was observed in the two the higher doses (2 or 20 mg/kg-day) and changes were not observed at any dose in the forced swim test (Bouayed et al., 2012). In addition, a 28-day gavage study in male mice observed an increase in consummatory sexual behavior in mice treated with 0.02 and 0.2 mg/kg-day, whereas aggressive behavior (as measured by the resident intruder test)

was only increased in the low dose group (0.02 mg/kg-day) (Bouayed et al., 2009b). Interpretation of the resident intruder data is further complicated because authors did not clearly define attack behavior and failed to indicate whether scorers were blinded to the treatment. Effects of oral benzo[a]pyrene exposure on motor- and sensory-related behaviors in adult rats are supported by a series of gavage studies testing single, high doses (i.e., ≥12.5 mg/kg) of benzo[a]pyrene (Saunders et al., 2006; Saunders et al., 2002; Saunders et al., 2001).

Two studies from the same laboratory reported learning and memory deficits in male Sprague-Dawley rats exposed to daily i.p. doses of 2.5 mg/kg-day for 13 (Xia et al., 2011) or 14 weeks (Qiu et al., 2011; Xia et al., 2011; Grova et al., 2007) starting when animals were 5 weeks old. Notably, the exposure windows for these studies overlapped briefly with pubertal brain development and, although the bulk of exposure occurred during adulthood, it is difficult to discern whether the observed effects are developmental or adult in origin. Interestingly, results in the Morris water maze after subchronic oral (Chengzhi et al., 2011) or i.p. (Qiu et al., 2011; Xia et al., 2011) exposure in "adult" rats identified an effect of exposure across all trial days during hidden platform testing, indicating lack of an effect on learning and supporting an effect on some other, unknown behavior(s); short-term i.p. exposure in mice only resulted in increased latencies on trial day 5 alongside unexplained decreases in latency on day 1 (Grova et al., 2007). Notably, the observations from adult rat studies are consistent with apparent effects on noncognitive behavior(s) in the Morris water maze after developmental exposure (see Section 1.1.1).

Overall, few studies are available to inform the neurotoxic potential of oral or inhalation exposure to benzo[a]pyrene in adults (Table 1-11). Notably, one subchronic, oral exposure study in rats (Chengzhi et al., 2011) indicated effects on Morris water maze performance at 2 mg/kg-day that were similar to results that have been observed after postnatal oral exposure and i.p. exposure in weanlings or adult rats. Further, this behavioral alteration is supported by behavioral effects at 2 mg/kg-day in a second oral exposure study in rats (Maciel et al., 2014). However, two oral exposure studies in adult mice (Bouayed et al., 2012; Bouayed et al., 2009b) were difficult to interpret as supportive evidence, as effects of benzo[a]pyrene exposure on behavior were not observed at the highest doses tested, namely 0.2 mg/kg-day (Bouayed et al., 2009b) and 2–20 mg/kg-day (Bouayed et al., 2012). Similarly, evidence consistent with benzo[a]pyrene exposure-induced behavioral effects in two studies of male coke oven workers is complicated by co-exposures to other constituents of PAH mixtures, and the relevance of behavioral effects observed in several studies of adult rodents following i.p. exposure to oral or inhalation exposure paradigmsis difficult to infer. Thus, while suggestive evidence of neurobehavioral effects following adult exposure exists, due to the limitations of the oral and inhalation database, a conclusion could not be drawn regarding the potential for nervous system toxicity to be a human hazard of benzo[a]pyrene exposure in adults, and additional studies are warranted. Overall, however, a human neurotoxicity hazard is identified based on exposure to benzo[a]pyrene during development (see Section 1.1.1).

Table 1-10. Evidence pertaining to liver, kidney, and cardiovascular effects of benzo[a]pyrene in animals after oral or inhalation exposure

Reference and study design	Results[a]
Liver effects	
Kroese et al. (2001) Wistar rats, 10/sex/dose 0, 3, 10, or 30 mg/kg-d by gavage 5 d/wk 90 d Wistar rats, 10/sex/dose 0, 1.5, 5, 15, or 50 mg/kg-d by gavage 5 d/wk for 35 d	↑ liver weight Females (% change from control): 0, −2, 4, and 17* Males (% change from control): 0, 7, 15*, and 29* Liver histopathology: no effects reported ↑ liver weight Females (% change from control): 0, 3, 2, 9, and 18* Males (% change from control): 0, 2, 1, 3, and 18* Liver histopathology: no effects reported
Knuckles et al. (2001) F344 rats, 20/sex/dose 0, 5, 50, or 100 mg/kg-d by diet 90 d	↑ liver:body weight ratio Females: no change (numerical data not reported) Males (% change from control): 23% change reported at 100 mg/kg-d (numerical data not reported)
De Jong et al. (1999) Wistar rats, 8 males/dose 0, 3, 10, 30, or 90 mg/kg-d by gavage 5 d/wk 35 d	↑ liver weight % change from control: 0, −9, 7, 5, and 15* ↑ liver oval cell hyperplasia (numerical data not reported) reported as significant at 90 mg/kg-d;
Kidney effects	
Knuckles et al. (2001) F344 rats, 6−8/sex/dose 0, 5, 50, or 100 mg/kg-d by diet 90 d	↑ abnormal tubular casts Females: not statistically significant (numerical data not reported) Males: apparent dose-dependent increase (numerical data not reported)
De Jong et al. (1999) Wistar rats, 8 males/dose 0, 3, 10, 30, or 90 mg/kg-d by gavage 5 d/wk 35 d	↓ kidney weight % change from control: 0, −11*, −4, −10*, and −18*
Kroese et al. (2001) Wistar rats, 10/sex/dose 0, 1.5, 5, 15, or 50 mg/kg-d by gavage 5 d/wk 35 d	Kidney weight: no change (data not reported)

Reference and study design	Results[a]
Cardiovascular effects	
Uno et al. (2014) C57BL/6J mice, male (n not reported) 0 or 12.5 mg/kg-d in diet (all animals fed an atherogenic diet) 15 wks	"modest" increase in atherosclerotic lesions were seen in the aorta (numerical data not reported)

*Statistically significantly different from the control ($p < 0.05$).
[a]% change from control calculated as: (treated value − control value)/control value × 100.
[b]Reported incidences may not fully account for the occurrence of hyperplasias due to the scoring of only the highest-level lesion in an individual animal (e.g., animals with forestomach tumors that also showed hyperplasia would not have the observation of hyperplasia recorded).
[c]Based on the assumption that daily benzo[a]pyrene intake at 5 ppm was one-fifth of the 25-ppm intake (about 21 µg/d) and using TWA body weights of 0.032 kg for the control, 5- and 25-ppm groups and 0.026 kg for the 100-ppm group.

Table 1-11. Evidence pertaining to neurotoxicity following repeated oral or inhalation exposure to benzo[a]pyrene in adult humans and animals

Reference and study design	Results
Human Studies	
Qiu et al. (2013) n = 100 male coke oven workers from Chongqing, China employed for ≥1 yr, mean age 41 yrs; 100 controls from oxygen plant in same company, mean age 39 yrs, comparable age, education level, smoking status, alcohol consumption, socioeconomic status, and general physical condition as exposed. Three air samples (one for controls) of at least 4-hr duration on successive days, analyzed for benzo[a]pyrene; control levels: 0.003 µg/m^3; exposed workplace means: 0.028, 0.781, or 2.82 µg/m^3 for coke oven bottom (n = 17), side (n = 34), and top (n = 49) sites, respectively. Post-shift urine samples analyzed for creatinine-corrected 1-OHP; control: 1.89 µmol/mol Cr; exposed: 3.61 µmol/mol Cr. *Outcomes: Neurobehavioral core test battery (Anger, 2003) including profile of mood states (POMS) questionnaire, simple reaction time, digit span, pursuit aiming, digit symbol, visual retention, and Santa Ana dexterity.*	Statistically significant changes ($p < 0.05$), with direction and magnitude of change, in exposed compared to control: POMS scores: tension-anxiety (↑26%), fatigue-inertia (↑30%) Simple reaction time: mean reaction time (↑12%) Digit span: n forward span (↓9%); n total span (↓6%) Digit symbol: n correct (↓20%) Pursuit aiming: n correct (↓10%); n total (↓8%) *Measures reported as not statistically significant:* POMS scores: depression-dejection; anger-hostility (note: $p = 0.087$ and ↑27%); vigor-activity; confusion-bewilderment Digit span: n backward span Santa Ana dexterity test: n preferred or non-preferred hand Pursuit aiming: n errors *Note: Analysis of covariance on neurobehavioral data stratified by exposure level (coke oven bottom, side, and top) showed significantly lower performance on simple reaction time, correct pursuit aiming, and error pursuit aiming tests with higher levels of airborne benzo[a]pyrene. Adverse changes in POMS (tension-anxiety, fatigue-inertia, and confusion-bewilderment), simple reaction time, and digit span, but not in digit symbol or pursuit aiming, were*

	significantly associated with higher post-shift urinary 1-OHP measures in simple linear regression analyses.
Niu et al. (2010) n = 176 male coke oven workers from Taiyuan, China employed for ≥1 yr, mean age 38 yrs; 48 controls from warehouse workers in same company, mean age 40 yrs; matched by socioeconomic status, age, education, lifestyle (smoking and drinking), and health (depression, family history, and medication) Air samples of 6-hr duration on three successive days, analyzed for benzo[a]pyrene; control levels: 0.0102 µg/m^3; exposed workplace means: 0.0195, 0.1859, or 1.624 µg/m^3 for coke oven bottom, side, and top sites, respectively. Post-shift urine samples analyzed for creatinine-corrected 1-OHP; control: 2.77 µmol/mol Cr; exposed: 3.66 µmol/mol Cr. *Outcomes: Neurobehavioral core test battery (Anger, 2003) including profile of mood states (POMS) questionnaire, simple reaction time, digit span, pursuit aiming, digit symbol, visual retention, and number of dots; plasma levels of neurotransmitters (norepinephrine, dopamine, 5-hydroxytryptamine, 5-hydroxyindoleacetic acid, homovanillic acid, γ-aminobutyric acid, aspartic acid, glutamic acid, glycine, and acetylcholine) and erythrocyte acetylcholinesterase activity.*	*Statistically significant changes ($p < 0.05$), with direction and magnitude of change, in exposed compared to control:* Digit span: forward span (↓13%); total span (↓11%) *Measures reported as not statistically significant:* POMS questionnaire (anger-hostility, confusion-bewilderment, depression-dejection, fatigue-inertia, tension-anxiety, and vigor-activity); simple reaction time; digit span: backward span; Santa Ana manual dexterity; digit symbol; Benton visual retention; dotting *Note: Statistical analysis methods were not reported; statistically significant changes ($p < 0.05$). Stratification of neurobehavioral data by tertile of creatinine-corrected urinary 1-OHP showed significantly lower performance on total and forward digit span tests (but no other tests) with higher levels of urinary 1-OHP. Plasma levels of norepinephrine, dopamine, aspartic acid, and γ-aminobutyric acid were significantly lower in exposed workers versus controls. Plasma acetylcholine was significantly increased over controls, and erythrocyte acetylcholinesterase activity was significantly decreased, in exposed workers. Stratification by tertile of creatinine-corrected urinary 1-OHP showed significant negative associations between urinary 1-OHP and plasma norepinephrine, plasma aspartic acid, and erythrocyte acetylcholinesterase, and a significant positive association with plasma acetylcholine.*
Rodent Studies	
Chengzhi et al. (2011) Sprague-Dawley rats, male, 32/dose 0 or 2 mg/kg-d by gavage 90 d	↑ time required for treated rats to locate platform in water maze (data reported graphically) on trial days 1–4 No information regarding swim speed or path length
Bouayed et al. (2009b) Swiss albino mice, male, 9/dose 0, 0.02, or 0.2 mg/kg-d by gavage 28 d	Significant decrease in latency to attack and increase in the number of attacks in the resident-intruder test at 0.02 mg/kg-d (but not at high dose) Significant increase in mount number in the copulatory behavior test at 0.02 and 0.2 mg/kg-d
Bouayed et al. (2009b) Swiss albino mice, male, 10/dose 0, 0.02, 0.2, 2, or 20 mg/kg-d by gavage 17 d	Open field locomotor activity unaffected Immobility time increased in the tail suspension test reduced at 0.02 and 0.2 mg/kg-d (but not at 2 or 20 mg/kg-d) Immobility time in the forced swimming test unaffected

Maciel et al. (2014) Wistar rats, male, 12/dose 0 or 2 mg/kg-d by gavage 28 d	↑ in total distance traveled in the locomotor activity test (numerical data not reported) ↓in short-term memory (tested 1.5 hr after training), but no change in long-term memory (tested 24 hr after training) in the novel object recognition test (numerical data not reported)

*Statistically significantly different from the control ($p < 0.05$).

1.1.5. Carcinogenicity

Evidence in Humans

Numerous epidemiologic studies indicate an association between PAH related occupations and lung, bladder, and skin cancer (Table 1-12). This discussion primarily focuses on epidemiologic studies that included a direct measure of benzo[a]pyrene exposure. All identified studies have co-exposures to other PAHs. The identified studies were separated into tiers according to the extent and quality of the exposure analysis and other study design features:

Tier 1: Detailed exposure assessment conducted (using a measure of benzo(a)pyrene exposure), large sample size (>~50 exposed cases), and adequate follow-up period to account for expected latency (e.g., >20 years for lung cancer).

Tier 2: Exposure assessment, sample size, or follow-up period did not meet the criteria for Tier 1, or only a single-estimate exposure analysis was conducted.

For lung cancer, each of the Tier 1 studies observed increasing risks of lung cancer with increasing cumulative exposure to benzo[a]pyrene (measured in μg/m^3-years), and each of these studies addressed in the analysis the potential for confounding by smoking (Armstrong and Gibbs, 2009; Spinelli et al., 2006; Xu et al., 1996) (Table 1-13). These three studies represent different geographic locations and two different industries. The pattern of results in the Tier 2 studies was mixed, as would be expected for studies with less precise exposure assessments or smaller sample sizes: one of the standardized mortality ratio (SMR) estimates was <1.0, with the other eight estimates ranging from 1.2 to 2.9 (Table 1-14). In considering all of the available studies, particularly those with the strongest methodology, there is considerable support for an association between benzo[a]pyrene exposure and lung cancer, although the relative contributions of benzo[a]pyrene and of other PAHs cannot be established.

For bladder cancer, the cohort and nested case-control studies observed a much smaller number of cases compared with lung cancer; this limits their ability to examine exposure-response relationships. Three cohort studies with detailed exposure data, however, identified 48–90 cases (Burstyn et al., 2007; Gibbs and Sevigny, 2007a; Gibbs et al., 2007; Gibbs and Sevigny, 2007b) (Spinelli et al., 2006) (Tier 1 studies, Table 1-15). Although cumulative exposure (up to

approximately 2 μg/m^3-years) was not related to increasing risk in the study of asphalt workers by Burstyn et al. (2007), an exposure-response was seen with the wider exposure range (i.e., >80 μg/m^3-years) examined in two studies of aluminum smelter workers by Gibbs and Sevigny (2007a); Gibbs et al. (2007); Gibbs and Sevigny (2007b); and (Spinelli et al., 2006). This difference in response is not surprising, given that the highest exposure group in the asphalt worker studies corresponded to the exposures seen in the lowest exposure categories in the studies of aluminum smelter workers. The five studies with more limited exposure information or analyses each included between 2 and 16 bladder cancer cases, with relative risk (RR) estimates ranging from 0.6 to 2.9. None of these individual effect estimates was statistically significant (Tier 2 studies, Table 1-15).

Two of the identified occupational studies contained information on risk of mortality from melanoma. Neither of these studies observed increased risks of this type of cancer, with an SMR of 0.91 (95% confidence interval [CI] 0.26, 2.48) (22 cases) in Spinelli et al. (2006) and 0.58 (95% CI 0.12, 1.7) in Gibbs et al. (2007)(3 cases). These studies did not include information on non-melanoma skin cancers.

Non-melanoma skin cancer, specifically squamous cell carcinoma, is of particular interest with respect to dermal PAH exposures. The literature pertaining to this kind of cancer and PAH exposure goes back to the 18th century work of Sir Percivall Pott describing scrotal cancer, a squamous cell skin cancer, in English chimney sweeps (Brown and Thornton, 1957). Recent studies of chimney sweeps in several Nordic countries have not found increases in non-melanoma skin cancer incidence (Hogstedt et al., 2013; Pukkala et al., 2009; Evanoff et al., 1993), likely due to greatly reduced exposure associated with better occupational hygiene (IARC, 2012). A study among asphalt workers (roofers) reported an increased risk of mortality from non-melanoma skin cancer among asphalt workers (roofers), with an SMR of 4.0 (95% CI: 1.0, 10.9) among workers employed ≥20 years (Hammond et al., 1976). In addition to this study, two studies in Scandinavian countries examined non-melanoma skin cancer risk in relation to occupations with likely dermal exposure to creosote (i.e., timber workers and brick makers) using incidence data from population registries (Karlehagen et al., 1992; Törnqvist et al., 1986). The standardized incidence ratio (SIR) estimates were 1.5 (95% CI: 0.7, 2.6) based on five exposed cases and 2.37 (95% CI: 1.08, 4.50) based on nine cases in Törnqvist et al. (1986) and Karlehagen et al. (1992). Because non-melanoma skin cancers are rarely fatal if caught early, and the preventative excision of precancerous lesions is common, the available occupational studies and cancer registries likely underestimate the risk of squamous cell carcinoma (Carøe et al., 2013; Voelter-Mahlknecht et al., 2007; ONS, 2003; Letzel and Drexler, 1998).

In addition to cohorts of workers occupationally exposed to PAH mixtures, populations exposed to benzo[a]pyrene through topical coal tar formulations for the treatment of psoriasis, eczema, and dermatitis have also been studied (Roelofzen et al., 2010; Mitropoulos and Norman, 2005; Hannuksela-Svahn et al., 2000; Stern et al., 1998; Maier et al., 1996; Jemec and Østerlind,

1994; Stern and Laird, 1994; Bhate et al., 1993; Lindelöf and Sigurgeirsson, 1993; Torinuki and Tagami, 1988; Jones et al., 1985; Pittelkow et al., 1981; Maughan et al., 1980; Stern et al., 1980). Epidemiological studies examining skin cancer risk in relation to various types of topical tar exposure are summarized in the Supplemental Information, Table D-6. Case reports, reviews, and studies that did not include a measure of coal tar use are not included. The available studies examining therapeutic topical coal tar use and risk of skin cancer were limited by low quality exposure data with high potential of exposure misclassification (e.g., Roelofzen et al., 2010; Mitropoulos and Norman, 2005; Hannuksela-Svahn et al., 2000; Maier et al., 1996; Jemec and Østerlind, 1994; Lindelöf and Sigurgeirsson, 1993); potential outcome misclassification (e.g., Jemec and Østerlind, 1994); small size (e.g., Jemec and Østerlind, 1994; Torinuki and Tagami, 1988); short duration of follow-up (e.g., Torinuki and Tagami, 1988); choice of referent group (i.e., there was no referent group or the referent group did not consist of psoriasis patients) (e.g., Jemec and Østerlind, 1994; Bhate et al., 1993; Jones et al., 1985; Pittelkow et al., 1981; Maughan et al., 1980); and/or differences in disease ascertainment between cases and the reference population (e.g., Pittelkow et al., 1981; Maughan et al., 1980). Although some clinic-based studies appear to indicate increased risk with coal tar exposure, these studies used a regimen of coal tar in conjunction with ultraviolet-B (UVB) therapy or among patients also treated with PUVA, and thus cannot distinguish the effects of coal tar from the effects of UVB or PUVA (e.g., Stern et al., 1998; Maier et al., 1996; Stern and Laird, 1994; Lindelöf and Sigurgeirsson, 1993; Stern et al., 1980).

There is some uncertainty regarding whether the anatomical properties of psoriatic skin limit the utility of the available epidemiological studies in coal tar treated patients for predicting whether benzo[a]pyrene induces skin cancer in the general population. Psoriatic skin is characterized by hyperkeratosis caused by abnormally rapid cell proliferation and greatly increased rates of desquamation (shedding of skin cells). Both hyperkeratosis and desquamation could be protective with respect to skin cancer risk from dermal PAH exposure. Desquamation can reduce penetration of compounds past the stratum corneum, so lipophilic chemicals such as benzo[a]pyrene may not reach the metabolically active layers of the skin (Reddy et al., 2000). Reduced absorption of PAHs into the dermally active layers of the skin is consistent with the findings of Roelofzen et al. (2012) which found reduced PAH-DNA adducts in biopsied skin of psoriasis patients as well as reduced 1-hydroxypyrene levels (a PAH metabolite) in urine as compared to healthy volunteers following exposure to coal tar ointments.

Therefore, because of the anatomical differences between psoriatic skin and normal skin, as well as limitations of the above studies with respect to study design and analysis, EPA did not consider these studies further in the evaluation of the risk of skin cancer from exposure to benzo[a]pyrene. Although EPA does not consider the available studies sufficient to evaluate the risk of skin cancer, acute studies of coal tar treated patients provide in vivo evidence of benzo[a]pyrene-specific genotoxicity (increased BPDE-DNA adducts) in human skin (Godschalk et

al., 2001; Rojas et al., 2001; Zhang et al., 1990), an early key event in the carcinogenic mode of action of benzo[a]pyrene (see Figure 1-7 of Section 1.1.5).

Lung, bladder, and skin cancers are the cancers that have been observed in occupational studies of PAH mixtures (Benbrahim-Tallaa et al., 2012; Baan et al., 2009; Secretan et al., 2009). The reproducibility of lung, bladder, and skin cancers in different populations and exposure settings after occupational exposure to PAH mixtures (see Table 1-12) adds plausibility to the hypothesis that common etiologic factors may be operating. The potential role that benzo[a]pyrene may play as a causal agent is further supported by the observation that cancers at these same sites are also increased in the studies that included a direct measure of benzo[a]pyrene.

Table 1-12. Cancer sites for PAH-related agents reviewed by IARC

PAH-related mixture or occupation	Sites with *sufficient evidence* in humans	Sites with *limited evidence* in humans	Reference
Aluminum production	Lung, urinary bladder		Baan et al. (2009)
Carbon electrode manufacture		Lung	IARC (2010)
Coal gasification	Lung		Baan et al. (2009)
Coal tar distillation	Skin		Baan et al. (2009)
Coal tar pitch (paving and roofing)	Lung	Urinary bladder	Baan et al. (2009)
Coke production	Lung		Baan et al. (2009)
Creosotes		Skin	IARC (2010)
Diesel exhaust	Lung	Urinary bladder	Benbrahim-Tallaa et al. (2012)
Indoor emissions from household combustion of biomass fuel (primarily wood)		Lung	Secretan et al. (2009)
Indoor emissions from household combustion of coal	Lung		Secretan et al. (2009)
Mineral oils, untreated or mildly treated	Skin		Baan et al. (2009)
Shale oils	Skin		Baan et al. (2009)
Soot (chimney sweeping)	Lung, skin	Urinary bladder	Baan et al. (2009)

Source: Adapted from IARC (2010).

Table 1-13. Summary of epidemiologic studies of benzo[a]pyrene (direct measures) in relation to lung cancer risk: Tier 1 studies

Reference and study design	Results
Armstrong and Gibbs (2009) (Quebec, Canada) Cohort, aluminum smelter workers, seven plants 16,431 (15,703 men; 728 women); duration minimum 1 yr, began work 1966–1990; follow-up through 1999 (mean ~30 yrs); smoking information collected from medical records Exposure: Job exposure matrix ~5,000 personal benzo[a]pyrene measures from the 1970s to 1999 Related references: Lavoué et al. (2007) (exposure data); Armstrong et al. (1994); Gibbs et al. (2007); Gibbs and Sevigny (2007a); Gibbs and Sevigny (2007b)	SMR 1.32 (1.22, 1.42) [677 cases] Lung cancer risk by cumulative benzo[a]pyrene exposure Median benzo[a]-pyrene µg/m^3-yrs / n cases / SMR (95% CI) / RR (95% CI) 0 / 35 / 0.62 (0.44, 0.87) / 1.0 (referent) 10 / 266 / 1.09 (0.96, 1.23) / 1.75 (1.23, 2.48) 30 / 70 / 1.88 (1.47, 2.38) / 3.02 (2.01, 4.52) 60 / 53 / 1.21 (0.91, 1.59) / 1.94 (1.27, 2.97) 120 / 114 / 1.93 (1.59, 2.32) / 3.09 (2.12, 4.51) 240 / 116 / 1.79 (1.48, 2.15) / 2.86 (1.96, 4.18) 480 / 23 / 2.36 (1.49, 3.54) / 3.77 (2.23, 6.38) No evidence of confounding by smoking Additional modeling as continuous variable: RR 1.35 (95% CI 1.22, 1.51) at 100 µg/m^3-yrs (0.0035 per µg/m^3-yrs increase); other shapes of exposure-response curve examined.
Spinelli et al. (2006) (British Columbia, Canada) Cohort, aluminum smelter workers; 6,423 (all men); duration minimum ≥3 yrs; began work 1954–1997; follow-up through 1999 (14% loss to follow-up; mean ~24 yrs); smoking information from self-administered questionnaire Exposure: Job exposure matrix using 1,275 personal benzo[a]pyrene measures from 1977 to 2000 (69% for compliance monitoring) Related references: Friesen et al. (2006) (exposure data); Spinelli et al. (1991)	SMR: 1.07 (0.89, 1.28) [120 cases] SIR: 1.10 (0.93, 1.30) [147 cases] Lung cancer risk by cumulative benzo[a]pyrene exposure Benzo[a]pyrene µg/m^3-yrs / n cases / RR (95% CI)[a] 0–0.5 / 25 / 1.0 (referent) 0.5–20 / 42 / 1.23 (0.74, 2.03) 20–40 / 23 / 1.35 (0.76, 2.40) 40–80 / 25 / 1.36 (0.78, 2.39) ≥80 / 32 / 1.79 (1.04, 3.01) [a]Adjusting for smoking category; trend $p < 0.001$.

Xu et al. (1996) (China) Nested case-control in iron-steel worker cohort 610 incident cases (96% participation); 959 controls (94% participation) (all men); duration data not reported; smoking information collected from interviews; next-of-kin interviews with 30% of lung cancer cases and 5% of controls Exposure: Job exposure matrix 82,867 historical monitoring records, 1956–1992	Lung cancer risk by cumulative benzo[a]pyrene exposure Benzo[a]-pyrene ($\mu g/m^3$-yrs) / n cases / RR (95% CI)[a] <0.84 / 72 / 1.1 (0.8, 1.7) 0.85–1.96 / 117 / 1.6 (1.2, 2.3) 1.97–3.2 / 96 / 1.6 (1.1, 2.3) ≥3.2[b] / 105 / 1.8 (1.2, 2.5) [a]Adjusting for birth year and smoking category; trend $p < 0.004$. Referent group is "nonexposed" (employed in administrative or low-exposure occupations). [b]Study table IV unclear; could be ≥3.0 for this category.

Table 1-14. Summary of epidemiologic studies of benzo[a]pyrene (direct measures) in relation to lung cancer risk: Tier 2 studies

Reference and study design	Results
Limited follow-up period (≤20 yrs)	
Friesen et al. (2009) (Australia) Cohort, aluminum smelter workers; 4,316 (all men); duration minimum 90 d; began work after 1962; follow-up through 2002, mean 16 yrs (maximum 20 yrs); Smoking information from company records if employed before 1995 and study interviews if employed after 1994 Exposure: Job/task exposure matrix using TWA benzo[a]pyrene measures (n = 655), 1977–2004 (79% from 1990 to 2004)	RR 1.2 (0.7, 2.3) [19 cases in exposed; 20 in unexposed] Lung cancer risk by cumulative benzo[a]pyrene exposure Benzo[a]-pyrene $\mu g/m^3$-yrs / n cases / RR (95% CI)[a] 0 / 20 / 1.0 (referent) >0–0.41 / 6 / 0.7 (0.3, 1.8) 0.41–10.9 / 6 / 1.4 (0.6, 3.5) >10.9 / 7 / 1.7 (0.7, 4.2) [a]Poisson regression, adjusting for smoking; trend $p = 0.22$.
Proxy measure	
Olsson et al. (2010) (Denmark, Norway, Finland, Israel) Nested case-control, asphalt workers; 433 lung cancer cases (65% participation); 1,253 controls (58% participation), matched by year of birth, country (all men); duration: minimum ≥2 seasons, median 8 seasons; began work 1913–1999; follow-up: from 1980 to 2002–2005 (varied by country); smoking information from interviews Exposure: Compilation of coal tar exposure measures, production characteristics, and repeat measures in asphalt industry in each country used to develop exposure matrix	Lung cancer risk by cumulative coal tar exposure[a] Coal tar unit-yrs[a] / n cases / RR / (95% CI) 0.39–4.29 / 43 / 1.31 / (0.87, 2.0) 4.30–9.42 / 32 / 0.98 / (0.62, 1.6) 9.43–16.88 / 30 / 0.97 / (0.61, 1.6) 16.89–196.48 / 54 / 1.60 / (1.09, 2.4) (trend p-value) / / / (0.07) [a]Adjusting for set, age, country, tobacco pack-years.

<table>
<tr><th>Reference and study design</th><th>Results</th></tr>
<tr><td>Related references: Boffetta et al. (2003); Burstyn et al. (2000)</td><td></td></tr>
<tr><td>Costantino et al. (1995) (United States, Pennsylvania)

Cohort, coke oven workers; 5,321 and 10,497 unexposed controls (non-oven steel workers; matched by age, race, date of first employment) (all men); duration data not reported; worked in 1953; follow-up through 1982 (length data not reported)

Exposure: Average daily exposure coal tar pitch volatiles: 3.15 mg/m^3 top-side full-time jobs, 0.88 mg/m^3 side jobs; used to calculate weighted cumulative exposure index

Related reference: Dong et al. (1988) (exposure data)</td><td>SMR 1.95 (1.59, 2.33) [255 cases]

Lung cancer risk by cumulative exposure

Coal tar pitch volatiles (mg/m^3-mo) | n cases | RR (95% CI)[a]
0 | 203 | 1.0 (referent)
1–49 | 34 | 1.2 (0.85, 1.8)
50–199 | 43 | 1.6 (1.1, 2.3)
200–349 | 59 | 2.0 (1.5, 2.8)
350–499 | 39 | 2.0 (1.6, 3.2)
500–649 | 27 | 2.7 (2.0, 4.6)
≥650 | 56 | 3.1 (2.4, 4.6)

[a]Adjusting for age, race, coke plant, period of follow-up; trend $p < 0.001$.</td></tr>
<tr><td colspan="2">Limited exposure information</td></tr>
<tr><td>Liu et al. (1997)(China)

Cohort, various carbon plants and aluminum smelter workers; 6,635 (all men); duration minimum 15 yrs; began work before 1971; follow-up: through 1985 (mean ~14 yrs); smoking information from questionnaire

Exposure: Area samples from one carbon plant, 1986–1987</td><td>SMR 2.2 (1.1, 2.8) [50 cases]

Lung cancer risk by exposure category

Exposure category | Mean benzo[a]-pyrene µg/m^3 | n cases | SMR (95% CI)[a]
None | – | 13 | 1.49 (0.83, 2.5)
Low | – | 6 | 1.19 (0.48, 2.5)
Moderate | 0.30 | 5 | 1.52 (0.55, 3.4)
High | 1.19 | 26 | 4.30 (2.9, 6.2)

[a]Calculated by EPA from data in paper.</td></tr>
<tr><td>Berger and Manz (1992) (Germany)

Cohort, coke oven workers; 789 (all men); duration minimum 10 yrs (mean 27 yrs); began work 1900–1989; follow-up through 1989 (length data not reported); smoking information from plant records and interviews

Exposure: Mean benzo[a]pyrene: 28 µg/m^3 (range 0.9–89 µg/m^3)</td><td>SMR 2.88 (2.28, 3.59) [78 cases]</td></tr>
</table>

Reference and study design	Results
(Hansen, 1989); Hansen (1991) (Denmark) Cohort, asphalt workers; 679 workers (applicators) (all men); duration data not reported; employed 1959 to 1980; follow-up to 1986 (mean ~11 yrs); smoking information from 1982 surveys of industry and general population Exposure: Asphalt fume condensate, 35 personal samples during flooring: median 19.7 mg/m^3 (range 0.5–260 mg/m^3)	SMR 2.90 (1.88, 4.3) [25 cases] (ages 40–89) SMR 2.46 (1.59, 3.6) [25 cases] (with smoking adjustment)
Gustavsson et al. (1990) (Sweden) Cohort, gas production (coke oven) workers; 295 (all men); duration minimum 1 yr, median 15 yrs; employed 1965–1972; follow-up: 1966–1986 (mortality); 1966–1983 (incidence; mean ~15 yrs); smoking information from interviews with older workers Exposure: Area sampling - top of ovens; benzo[a]pyrene, 1,964 mean 4.3 µg/m^3 (range 0.007–33 µg/m^3); 1,965 mean 0.52 µg/m^3 (0.021–1.29 µg/m^3)	SMR 0.82 (0.22, 2.1) [4 cases] (referent group = employed men) SIR 1.35 (0.36, 3.5) [4 cases]
Moulin et al. (1989) (France) Cohort and nested case-control, two carbon electrode plants; 1,302 in Plant A (all men), employed in 1975; follow-up 1975–1985 (incidence); smoking information from plant records; 1,115 in Plant B (all men); employed in 1957; follow-up 1957–1984 (mortality); duration of employment and follow-up data not reported Exposure: Benzo[a]pyrene, 19 area samples and 16 personal samples in Plant A (personal sample mean 2.7 µg/m^3; range 0.59–6.2 µg/m^3); 10 area samples and 7 personal samples in Plant B; personal sample mean 0.17 µg/m^3, range 0.02–0.57 µg/m^3	Plant A: SMR 0.79 (0.32, 1.6) [7 cases] Plant B: SMR 1.18 (0.63, 2.0) [13 cases] Internal comparison (case-control), ≥1 yr duration: Plant A: OR 3.42 (0.35, 33.7) [7 cases, 21 controls] Plant B: OR 0.49 (0.12, 2.0) [13 cases, 33 controls]
Hammond et al. (1976) (United States) Cohort, asphalt roofers; 5,939 (all men); duration minimum 9 yrs, began before 1960; follow-up through 1971 Exposure: 52 personal samples (masks with filters) during specific jobs and tasks; mean benzo[a]pyrene 16.7 µg per 7-hr d	SMR 1.6 (1.3, 1.9) [99 cases] (≥20 yrs since joining union) (CIs calculated by EPA from data in paper)

Table 1-15. Summary of epidemiologic studies of benzo[a]pyrene (direct measures) in relation to bladder cancer risk

Reference and study design	Results
Tier 1 studies	
Burstyn et al. (2007) (Denmark, Norway, Finland, Israel) Cohort, asphalt workers; 7,298 (all men); duration minimum ≥2 seasons, median 8 seasons; began work 1913–1999; follow-up began around 1960, ended around 2000 (years varied by country); median 21 yrs; smoking information not collected Exposure: Compilation of benzo[a]pyrene measures, production characteristics, and repeat measures in asphalt industry in each country used to develop exposure matrix Related references: Boffetta et al. (2003); Burstyn et al. (2000)	48 incident bladder cancer cases (39 cases in analyses with 15-yr lag) Bladder cancer risk by cumulative benzo[a]pyrene exposure[a] Benzo[a]-pyrene µg/m^3-yrs[a] / n cases / RR (95% CI) (no lag)[b] / RR (95% CI) (15-yr lag)[c] 0–0.253 / 12 / 1.0 (referent) / 1.0 (referent) 0.253–0.895 / 12 / 0.69 (0.29, 1.6) / 1.1 (0.44, 2.9) 0.895–1.665 / 12 / 1.21 (0.45, 3.3) / 1.7 (0.62, 4.5) ≥1.665 / 12 / 0.84 (0.24, 2.9) / 1.1 (0.30, 4.0) [a]Adjusting for age, calendar period, total duration of employment, country. [b]Trend p = 0.9. [c]Trend p = 0.63. Stronger pattern seen with average exposure in 15-yr lag (RR 1.5, 2.7, 1.9 in second through fourth quartile; trend p = 0.15)
Gibbs et al. (2007); Gibbs and Sevigny (2007a); Gibbs and Sevigny (2007b) (Quebec, Canada) Cohort, aluminum smelter workers, seven plants 16,431 (15,703 men; 728 women); duration minimum 1 yr, began work 1966–1990; follow-up: through 1999 (mean ~30 yrs); smoking information collected from medical records Exposure: Job exposure matrix using ~5,000 personal benzo[a]pyrene measures from the 1970s to 1999 Related references: Lavoué et al. (2007) (exposure data); Armstrong et al. (1994); Gibbs (1985); Gibbs and Horowitz (1979)	Hired before 1950: SMR 2.24 (1.77, 2.79) [78 cases] Bladder cancer risk by cumulative benzo[a]pyrene exposure Benzo[a]-pyrene µg/m^3-yrs[a] / n cases / SMR (95% CI) / Smoking-adjusted RR[b] 0 / 3 / 0.73 (0.15, 2.1) / 1.0 (referent) 10 / 14 / 0.93 (0.45, 1.4) / 1.11 30 / 3 / 1.37 (0.28, 4.0) / 1.97 60 / 1 / 0.35 (0.9, 1.9) / 0.49 120 / 15 / 4.2 (2.4, 6.9) / 8.49 240 / 30 / 6.4 (4.3, 9.2) 480 / 12 / 23.9 (12.2, 41.7) [a]Category midpoint. [b]CIs not reported; highest category is ≥80 µg/m^3-yrs (n observed = 57). Mortality risk reduced in cohort hired in 1950–1959, SMR = 1.23. Similar patterns seen in analysis of bladder cancer incidence.

Reference and study design	Results
Spinelli et al. (2006) (British Columbia, Canada) See Table 1-13 for study details; this study is considered a "Tier 2") study for bladder cancer because of the smaller number of bladder cancer cases (n = 12) compared with lung cancer cases (n = 120)	SMR 1.39 (0.72, 2.43) [12 cases] SIR 1.80; CI 1.45–2.21 [90 cases, including in situ] Bladder cancer risk by cumulative benzo[a]pyrene exposure Benzo[a]-pyrene µg/m^3-years / n cases / RR (95% CI)[a] 0–0.5 / 17 / 1.0 (referent) 0.5–20 / 20 / 0.83 (0.43, 1.59) 20–40 / 13 / 1.16 (0.56, 2.39) 40–80 / 18 / 1.50 (0.77, 2.94) ≥80 / 22 / 1.92 (1.02, 3.65) [a]Adjusting for smoking category; trend $p < 0.001$.
Tier 2 studies	
Friesen et al. (2009) (Australia) See Table 1-14 for study details	RR 0.6 (0.2, 2.0) [five cases in exposed; eight in unexposed] Bladder cancer risk by cumulative benzo[a]pyrene exposure Benzo[a]-pyrene µg/m^3-yrs / n cases / RR (95% CI)[a] 0 / 8 / 1.0 (referent) >0–0.41 / 1 / 0.2 (0.03, 1.9) 0.41–10.9 / 2 / 0.7 (0.2, 3.7) >10.9 / 2 / 1.2 (0.2, 5.6) [a]Poisson regression, adjusting for smoking category; trend $p = 0.22$.
Costantino et al. (1995) (United States, Pennsylvania) See Table 1-14 for study details	SMR 1.14 (0.61, 2.12) (16 cases)
Hammond et al. (1976) (United States) See Table 1-14 for study details	SMR 1.7 (0.94, 2.8) (13 cases) (≥20 yrs since joining union) (CIs calculated by EPA from data in paper)
Moulin et al. (1989) (France) See Table 1-14 for study details	Plant A: 0 observed cases; expected <1.0 Plant B: SMR 1.94 (0.40, 5.0) (3 cases)
Gustavsson et al. (1990) (Sweden) See Table 1-14 for study details	SMR 2.85 (0.30, 10.3) (2 cases) (referent group = employed men)

Evidence in Animals

Oral exposure

Evidence of tumorigenicity following oral exposure to benzo[a]pyrene has been demonstrated in rats and mice. As summarized in Table 1-16, oral exposure to benzo[a]pyrene has resulted in an increased incidence of tumors in the alimentary tract in male and female rats (Kroese et al., 2001; Brune et al., 1981) and female mice (Beland and Culp, 1998; Culp et al., 1998), liver carcinomas in male and female rats, kidney adenomas in male rats (Kroese et al., 2001), and auditory canal tumors in both sexes (Kroese et al., 2001).

Forestomach tumors have been observed in several lifetime cancer bioassays in rats and mice following both gavage and dietary exposure to benzo[a]pyrene at doses ranging from 0.016 mg/kg-day in Sprague-Dawley rats to 3.3 and 10 mg/kg-day in B6C3F_1 mice and Wistar rats, respectively (Kroese et al., 2001; Beland and Culp, 1998; Culp et al., 1998; Brune et al., 1981). In addition, multiple less-than-lifetime oral exposure cancer bioassays in mice provide supporting evidence that oral exposure to benzo[a]pyrene is associated with an increased incidence of forestomach tumors (Weyand et al., 1995; Benjamin et al., 1988; Robinson et al., 1987; El-Bayoumy, 1985; Triolo et al., 1977; Wattenberg, 1974; Roe et al., 1970; Biancifiori et al., 1967; Chouroulinkov et al., 1967; Fedorenko and Yansheva, 1967; Neal and Rigdon, 1967; Berenblum and Haran, 1955).

Increases in the incidence of forestomach hyperplasia have also been observed in Wistar rats following shorter-term, subchronic, and chronic gavage exposure (Kroese et al., 2001; De Jong et al., 1999) and in B6C3F1 mice following chronic dietary exposure (Beland and Culp, 1998; Culp et al., 1998). Forestomach hyperplasia occurred at shorter durations and at lower doses than tumors in rats and mice exposed to benzo[a]pyrene for up to 2 years (Kroese et al., 2001; Beland and Culp, 1998). Kroese et al. (2001) reported that the forestomach lesions demonstrated a progression over the course of intercurrent sacrifices; the authors described early lesions as focal or confluent basal hyperplasia, followed by more advanced hyperplasia with squamous cell papilloma, culminating in squamous cell carcinoma, indicating that forestomach hyperplasia may be a histological precursor to neoplasia observed in the forestomach after chronic exposure to benzo[a]pyrene.

Although humans do not have a forestomach, similar squamous epithelial tissue is present in the oral cavity (IARC, 2003; Wester and Kroes, 1988); therefore, forestomach tumors observed in rodents following benzo[a]pyrene exposure are considered relevant for the assessment of cancer hazard in humans (Beland and Culp, 1998). For further discussion, see Sections 1.2 and 2.3.4.

Elsewhere in the alimentary tract, dose-related increases of benign and malignant tumors were observed. In rats, oral cavity tumors were induced in both sexes and adenocarcinomas of the jejunum were induced in males (Kroese et al., 2001). In mice, tumors were induced in the tongue, esophagus, and larynx of females (males were not tested) (Beland and Culp, 1998; Culp et al., 1998).

Chronic oral exposure to benzo[a]pyrene resulted in a dose-dependent increased incidence of liver carcinomas in both sexes of Wistar rats, with the first liver tumors detected in week 35 in high-dose male rats; liver tumors were described as complex, with a considerable proportion

(59/150 tumors) metastasizing to the lungs (Kroese et al., 2001). Treatment-related hepatocellular tumors were not observed in mice (Beland and Culp, 1998; Culp et al., 1998).

An increased incidence of kidney tumors (cortical adenomas) was observed in male Wistar rats following chronic gavage exposure (Kroese et al., 2001) (Table 1-16). The kidney tumors were observed at the mid- and high-dose groups. Treatment-related kidney tumors were not observed in two other chronic studies (Beland and Culp, 1998; Brune et al., 1981).

Lung tumors were also observed following almost nine months of dietary exposure to approximately 10 mg/kg-day in female AJ mice (Weyand et al., 1995). Other lifetime exposure studies did not report treatment-related increases in lung tumors (Kroese et al., 2001; Beland and Culp, 1998; Culp et al., 1998).

Table 1-16. Tumors observed in chronic oral animal bioassays

Study design and reference	Results
Kroese et al. (2001) Wistar (Riv:TOX) rats (52/sex/dose group) 0, 3, 10, or 30 mg/kg-d by gavage 5 d/wk 2 yrs	Forestomach incidences: M: 0/52; 7/52*; 18/52*; and 17/52* (papilloma) M: 0/52; 1/52; 25/52*; and 35/52* (squamous cell carcinoma) F: 1/52; 3/51; 20/51*; and 25/52* (papilloma) F: 0/52; 3/51; 10/51*; and 25/52* (squamous cell carcinoma) Oral cavity incidences: M: 0/24; 0/24; 2/37; and 10/38* (papilloma) M: 1/24; 0/24; 5/37; and 11/38* (squamous cell carcinoma) F: 0/19; 0/21; 0/9; and 9/31*(papilloma) F: 1/19; 0/21; 0/9; and 9/31* (squamous cell carcinoma) Jejunum (adenocarcinomas) incidences: M: 0/51; 0/50; 1/51; and 8/49* F: 0/50; 0/48; 0/50; and 2/51 Duodenum (adenocarcinomas) incidences: M: 0/51; 0/50; 0/51; and 1/49 F: 0/49; 0/48; 0/50; and 2/51 Liver (adenomas and carcinomas) incidences: M: 0/52; 3/52; 15/52*; and 4/52 (adenoma) M: 0/52; 1/52; 23/52*; and 45/52* (carcinoma) F: 0/52; 2/52; 7/52*; and 1/52 (adenoma) F: 0/52; 0/52; 32/52*; and 50/52* (carcinoma) Kidney (cortical adenoma) incidences: M: 0/52; 0/52; 7/52*; and 8/52* F: increase not observed Auditory canal[b] (Zymbal gland) (carcinomas) incidences: M: 0/1; 0/0; 2/7; and 19/33* F: 0/0; 0/1; 0/0; and 13/20*

Study design and reference	Results
Beland and Culp (1998); Culp et al. (1998) B6C3F$_1$ mice: female (48/dose group) 0, 5, 25, or 100 ppm (average daily doses[a]: 0, 0.7, 3.3, and 16.5 mg/kg-d) in the diet 2 yrs	Forestomach (papillomas and squamous cell carcinomas) incidences: 1/48; 3/47; 36/46*; and 46/47* Esophagus (papillomas and carcinomas) incidences: 0/48; 0/48; 2/45; and 27/46* Tongue (papillomas and carcinomas) incidences: 0/49; 0/48; 2/46; and 23/48* Larynx (papillomas and carcinomas) incidences: 0/35; 0/35; 3/34; and 5/38
Brune et al. (1981) Sprague-Dawley rats: male and female (32/sex/dose) Gavage: 0, 6, 18, 39 mg/kg-yr (0, 0.016, 0.049, 0.107 mg/kg-d) Diet: 0, 6, 39 mg/kg-yr (0, 0.016, 0.107 mg/kg-d) Treated until moribund or dead 2 yrs	Forestomach (papillomas and carcinomas[c]); gavage incidences: 3/64; 12/64*; 26/64*; and 14/64* Forestomach (papillomas); diet incidences: 2/64; 1/64; and 9/64* Larynx and esophagus (papillomas); gavage incidences: 3/64; 1/64; 0/64; and 0/64 Larynx and esophagus (papillomas); diet incidences: 1/64; 2/64; and 1/64

*Indicates statistical significance as identified in study.
[a]Based on the assumption that daily benzo[a]pyrene intake at 5 ppm was one-fifth of the 25-ppm intake (about 21 µg/day) and using TWA body weights of 0.032 kg for the control, 5- and 25-ppm groups and 0.026 kg for the 100-ppm group.
[b]Incidences are for number of rats with tumors compared with number of tissues examined histologically. Auditory canal tissue was examined histologically when abnormalities were observed on macroscopic examination.
[c]Two malignant forestomach tumors were observed (one each in the mid- and high-dose groups).

Inhalation exposure

The inhalation database of benzo[a]pyrene carcinogenicity studies consists of one lifetime inhalation bioassay in male hamsters (Thyssen et al., 1981). Intratracheal instillation studies in hamsters are also available (Feron and Kruysse, 1978; Ketkar et al., 1978; Feron et al., 1973; Henry et al., 1973; Saffiotti et al., 1972).

Several long term intratracheal installation studies in hamsters evaluated the carcinogenicity of benzo[a]pyrene (Feron and Kruysse, 1978; Feron et al., 1973; Henry et al., 1973; Saffiotti et al., 1972). These studies treated animals with benzo[a]pyrene once a week in a saline solution (0.5–0.9%) for ≥8 months and observed animals for 1–2 years following cessation of exposure. Tumors in the larynx, trachea, bronchi, bronchioles, and alveoli were observed. Individual studies also reported tumors in the nasal cavity and forestomach. These intratracheal instillation studies support the carcinogenicity of benzo[a]pyrene in the respiratory tract; however,

conversion of a dose delivered by intratracheal instillation to an inhalation concentration is problematic due to different patterns of deposition and retention.

Lifetime inhalation exposure to benzo[a]pyrene resulted in the development of tumors in the respiratory tract and pharynx in Syrian golden hamsters (Thyssen et al., 1981). The authors stated that the rates of tumors of other organs generally corresponded to the rates in controls. U.S. EPA (1990) obtained individual animal data from the study authors (including individual animal pathology reports for the respiratory and upper digestive tracts, time-to-death data, and exposure chamber monitoring data) (Clement Associates, 1990); this information is summarized in Table 1-17. Concentration-dependent increased incidences of tumors in the upper respiratory tract, including the larynx and trachea, were seen at exposure concentrations of ≥9.5 mg/m^3. In addition, a decrease in mean tumor latency was observed in the larynx and trachea. Nasal cavity tumors were observed at the mid- and high-concentration, but the incidences were not dose-dependent. A concentration-related increase in tumors in the upper digestive tract (pharynx and esophagus) was also reported. In addition, a single forestomach tumor was observed in each of the mid- and high-concentration groups. The study authors suggested that the upper digestive tract tumors were a consequence of mucociliary particle clearance. All nasal, forestomach, esophageal, and tracheal tumors occurred in hamsters that also had tumors in the larynx or pharynx, except in the mid-concentration group, where two animals with nasal tumors had no tumors in the pharynx or larynx.

A re-analysis of the individual animal pathology reports and the exposure chamber monitoring data provided by the study authors yielded estimates of average continuous lifetime exposures for each individual hamster. Group averages of individual average continuous lifetime exposure concentrations were 0, 0.25, 1.01, and 4.29 mg/m^3 for the control through high-exposure groups (U.S. EPA, 1990).

Table 1-17. Tumors observed in chronic inhalation animal bioassays

Reference and study design	Results[b]
Thyssen et al. (1981) Syrian golden hamsters: male (26–34 animals/group placed on study) 0, 2.2, 9.5, or 46.5 mg/m^3 on NaCl particles by nose only inhalation for 3–4.5 hrs, 5–7 d/wk (TWA exposure concentrations[a]: 0, 0.25, 1.01, and 4.29 mg/m^3) Treated until moribund or dead (up to 130 wks) MMAD: not reported	Larynx incidences: 0/26; 0/21; 11/26; and 11/25 mean tumor latency[c]: 107 and 68 wks Pharynx incidences: 0/23; 0/19; 9/22; and 18/23 mean tumor latency: 97 and 68 wks Trachea incidences: 0/27; 0/21; 2/26; and 3/25 mean tumor latency: 115 and 63 wks Nasal cavity incidences: 0/26; 0/22; 4/26; and 1/34 mean tumor latency: 116 and 79 wks Esophagus incidences: 0/27; 0/22; 0/26; and 2/34 mean tumor latency: 71 wks Forestomach incidences: 0/27; 0/22; 1/26; and 2/34 mean tumor latency: 119 and 72 wks

[a]Duration-adjusted inhalation concentrations calculated from exposure chamber monitoring data and exposure treatment times. Daily exposure times: 4.5 hours/day, 5 days/week on weeks 1–12; 3 hours/day, 5 days/week on weeks 13–29; 3.7 hours/day, 5 days/week on week 30; 3 hours/day, 5 days/week on weeks 31–41; and 3 hours/day, 7 days/week for reminder of the experiment.

[b]Thyssen et al. (1981) reported only the incidences of malignant tumors, confirmed by comparison with the original study pathology data (Clement Associates, 1990). The incidences summarized here include relevant benign tumors (papillomas, polyps, and papillary polyps). The malignant tumors were squamous cell carcinomas, with the exception of one in situ carcinoma of the larynx and one adenocarcinoma of the nasal cavity, both in the 9.5 mg/m^3 group. Denominators reflect the number of animals examined for histopathology for each tissue. See Section D.4.2 and Table E-27 in the Supplemental Material for study details and a complete listing of individual data, respectively.

[c]Mean time of observation of tumor, 9.5 and 46.5 mg/m^3 concentration groups.

[d]Thyssen et al. (1981) did not report statistical significance testing. See Section D.4.2.

Dermal exposure

Repeated application of benzo[a]pyrene to skin (in the absence of exogenous promoters) has been demonstrated to induce skin tumors in mice, rats, rabbits, and guinea pigs. These studies have been reviewed by multiple national and international health agencies (IARC, 2010; IPCS, 1998; ATSDR, 1995; IARC, 1983, 1973). Mice have been the most extensively studied species in dermal carcinogenesis studies of benzo[a]pyrene because of evidence that they may be more sensitive than other animal species; however, comprehensive comparisons of species differences in sensitivity to

lifetime dermal exposure are not available. Systemic tumors in benzo[a]pyrene-treated mice were not increased compared to controls in lifetime dermal bioassays in which macroscopic examination of internal organs was included (Higginbotham et al., 1993; Habs et al., 1980; Schmähl et al., 1977; Schmidt et al., 1973; Roe et al., 1970; Poel, 1959).

The analysis in this document focuses on lifetime carcinogenicity bioassays in several strains of mice following repeated dermal exposure to benzo[a]pyrene (Table 1-18). These studies involved 2- or 3-times/week exposure protocols, at least two exposure levels plus controls, and histopathological examinations of the skin and other tissues (Sivak et al., 1997; Grimmer et al., 1984; Habs et al., 1984; Grimmer et al., 1983; Habs et al., 1980; Schmähl et al., 1977; Schmidt et al., 1973; Roe et al., 1970; Poel, 1963, 1959).

Additional studies in mice observed skin tumors following benzo[a]pyrene exposure and were considered supportive for hazard identification, but were not considered further in this assessment because of the availability of the lifetime studies identified above. These studies included several "skin painting" studies in mouse skin that did not report the doses applied (e.g., Wynder and Hoffmann, 1959; Wynder et al., 1957); several less than lifetime studies (Albert et al., 1991; Nesnow et al., 1983; Emmett et al., 1981; Levin et al., 1977); initiation-promotion studies utilizing acute dosing of benzo[a]pyrene followed by repeated exposure to a potent tumor promoter; and studies involving vehicles expected to interact with or enhance benzo[a]pyrene carcinogenicity (e.g., Bingham and Falk, 1969).

One study applied benzo[a]pyrene (topically once a week for 6 months) to immuno-compromised mice with human skin grafts (n = 10) and did not observe tumors, whereas all three control mice (mice with no skin grafts) developed skin tumors (Urano et al., 1995). The authors concluded this result indicates that human skin is much less susceptible to benzo[a]pyrene than mouse skin. Although some studies indicate that the skin grafts maintain some metabolic function (Das et al., 1986), it is unclear whether the human skin grafts maintain the same viability, vascularization, and full metabolic capacity as human skin in vivo (Kappes et al., 2004). In addition, no control was used to account for the trauma of the surgery and potential loss of viability in the transplanted skin (i.e., mice with grafted mouse skin were not used as a control). Another concern is the short amount of time allowed for tumor development. All of the mice with human skin grafts treated with benzo[a]pyrene died within 6 months of the start of treatment (Urano et al., 1995). While 6 months is generally sufficient for the development of tumors in mouse skin, it is unclear that this much smaller fraction of a human lifetime would be sufficient time for the development of human skin cancer if the human skin grafts retain human properties (e.g., better DNA repair, slower rate of cell turnover, and generally slower toxicokinetics) as human latency for squamous cell carcinoma in PAH-exposed occupational cohorts is thought to be >20 years (Young et al., 2012; Voelter-Mahlknecht et al., 2007; Everall and Dowd, 1978). Potent mutagenic carcinogens such as 7,12-dimethylbenz[a]anthracene, methylcholanthrene, and methylnitronitrosoguanidine also fail to produce skin tumors in this model system (Soballe et al., 1996; Urano et al., 1995; Graem, 1986).

Therefore, the ability of this model system to predict hazard for human skin cancer risk (particularly from metabolically active carcinogens) is unclear.

Table 1-18. Tumors observed in chronic dermal animal bioassays

Reference and study design	Results[a]
Poel (1959) C57L mice: male (13–56/dose) 0, 0.15, 0.38, 0.75, 3.8, 19, 94, 188, 376, or 752 µg Dermal; 3 times/wk for up to 103 wks or until the appearance of a tumor by gross examination	Skin tumors (gross skin tumors and epidermoid carcinoma); dose-dependent decreased time of tumor appearance incidences: Gross skin tumors: 0/33; 5/55; 11/55; 7/56; 41/49; 38/38; 35/35; 12/14; 14/14; and 13/13 Epidermoid carcinoma: 0/33; 0/55; 2/55; 4/56; 32/49; 37/38; 35/35; 10/14; 12/14; and 13/13 Cytotoxicity: information not provided
Poel (1963) SWR, C3HeB, or A/He mice: male (14–25/dose) 0, 0.15, 0.38, 0.75, 3.8, 19.0, 94.0, or 470 µg Dermal; 3 times/wk until mice died or a skin tumor was observed	Skin tumors and dose-dependent decreased time of first tumor appearance incidences: SWR: 0/20; 0/25; 2/22; 15/18; 12/17; 16/16; 16/17; and 14/14 C3HeB: 0/17; 0/19; 3/17; 4/17; 11/18; 17/17; 18/18; and 17/17 A/He mice: 0/17; 0/18; 0/19; 0/17; 0/17; 21/23; 11/16; and 17/17 Cytotoxicity: information not provided
Roe et al. (1970) Swiss mice: female (50/dose) 0, vehicle, 0.1, 0.3, 1, 3, or 9 µg Dermal; 3 times/wk for up to 93 wks	Skin tumors; malignant skin tumors were observed in 4/41 and 31/40 mice in the two high-dose groups, respectively incidences: 0/43; 0/47; 1/42; 0/42; 1/43; 8/41; and 34/46 Cytotoxicity: information not provided
Schmidt et al. (1973) NMRI mice: female (100/group) Swiss mice: female (100/group) 0, 0.05, 0.2, 0.8, or 2 µg Dermal; 2 times/wk until spontaneous death occurred or until an advanced carcinoma was observed	Skin tumors (carcinomas) incidences: NMRI: 2/100 at 2 µg (papillomas); 2/100 at 0.8 µg and 30/100 at 2 µg (carcinomas) Swiss: 3/80 at 2 µg (papillomas); 5/80 at 0.8 µg and 45/80 at 2 µg (carcinomas) Cytotoxicity: information not provided
Schmähl et al. (1977) NMRI mice: female (100/group) 0, 1, 1.7, or 3 µg Dermal; 2 times/wk until natural death or until they developed a carcinoma at the site of application	Skin tumors (papillomas and carcinomas) incidences: 0/81; 1/77; 0/88; and 2/81 (papillomas) 0/81; 10/77; 25/88; and 43/81 (carcinomas) Cytotoxicity: information not provided
Habs et al. (1980) NMRI mice: female (40/group) 0, 1.7, 2.8, or 4.6 µg Dermal; 2 times/wk until natural death or gross observation of infiltrative tumor growth	Skin tumors and dose-dependent increase in age-standardized tumor incidence incidences: 0/35; 8/34; 24/35; and 22/36 age-standardized tumor incidence:

Reference and study design	Results[a]
	0, 24.8, 89.3, and 91.7% Cytotoxicity: information not provided
Grimmer et al. (1984); Grimmer et al. (1983) CFLP mice: female (65–80/group) 0, 3.9, 7.7, or 15.4 µg (1983 study) 0, 3.4, 6.7, or 13.5 µg (1984 study) Dermal; 2 times/wk for 104 wks	Skin tumors (papillomas and carcinomas) with a decrease in tumor latency incidences: 1983: 0/80; 7/65; 5/64; and 2/64 (papillomas) 0/80; 15/65; 34/64; and 54/64 (carcinomas) 1984: 0/65; 6/64; 8/65; and 4/65 (papillomas) 0/65; 37/64; 45/65; and 53/65 (carcinomas) Cytotoxicity: information not provided
Habs et al. (1984) NMRI mice: female (20/group) 0, 2, or 4 µg Dermal; 2 times/wk for life	Skin tumors (papillomas and carcinomas) with a decrease in mean survival time incidences: 0/20; 2/20; and 0/20 (papillomas) 0/20; 7/20; and 17/20 (carcinomas) Cytotoxicity: information not provided
Sivak et al. (1997); NIOSH (1989) C3H/HeJ mice: male (30/group) 0, 0.05, 0.5, or 5 µg Dermal; 2 times/wk for up to 104 wks	Skin tumors (papillomas and carcinomas) incidences: 0/30; 0/30; 5/30 (1 papilloma, 1 keratoacanthoma, 3 carcinomas); and 27/30 (1 papilloma, 28 carcinomas) Cytotoxicity: 80% incidence of scabs and sores in highest dose group; no cytotoxicity noted at lower doses

[a]Statistical significance not reported by study authors.

Mode-of-Action Analysis—Carcinogenicity

The carcinogenicity of benzo[a]pyrene, the most studied PAH, is well documented in animal models (IARC, 2010; Xu et al., 2009; Jiang et al., 2007; Jiang et al., 2005; Xue and Warshawsky, 2005; Ramesh et al., 2004; Boström et al., 2002; Penning et al., 1999; IPCS, 1998; Harvey, 1996; ATSDR, 1995; Cavalieri and Rogan, 1995; U.S. EPA, 1991b). The primary mode of action by which benzo[a]pyrene induces carcinogenicity is via a mutagenic mode of action. This mode of action is presumed to apply to all tumor types and is relevant for all routes of exposure. The general sequence of key events associated with a mutagenic mode of action for benzo[a]pyrene is: (1) bioactivation of benzo[a]pyrene to DNA-reactive metabolites via three possible metabolic activation pathways: a diol epoxide pathway, a radical cation pathway, and an *o*-quinone and ROS pathway; (2) direct DNA damage by reactive metabolites, including the formation of DNA adducts and ROS-mediated damage; (3) formation and fixation of DNA mutations, particularly in tumor suppressor genes or oncogenes associated with tumor initiation; and (4) clonal expansion of mutated cells during the promotion and progression phases of cancer development. These events are depicted as stages of benzo[a]pyrene-induced carcinogenesis in Figure 1-7.

Benzo[a]pyrene is a complete carcinogen, in that it can act as both an initiator and a promoter of carcinogenesis. Initiation via direct DNA damage (key event 2) can occur via all three

metabolites of benzo[a]pyrene. DNA damage that is not adequately repaired leads to mutation (key event 3), and these mutations can undergo clonal expansion (key event 4) enabled by multiple mechanisms also induced by benzo[a]pyrene, including AhR binding leading to an upregulation of genes related to biotransformation, growth, and differentiation, and regenerative cell proliferation resulting from cytotoxicity and a sustained inflammatory response. However, there is not sufficient evidence that these mechanisms, which contribute to the promotion and progression phases of cancer development, act independently of DNA damage and mutation to produce benzo[a]pyrene-induced tumors (see *Other possible modes of action*, below). The available human, animal, and in vitro evidence supports a mutagenic mode of action as the primary mode by which benzo[a]pyrene induces carcinogenesis.

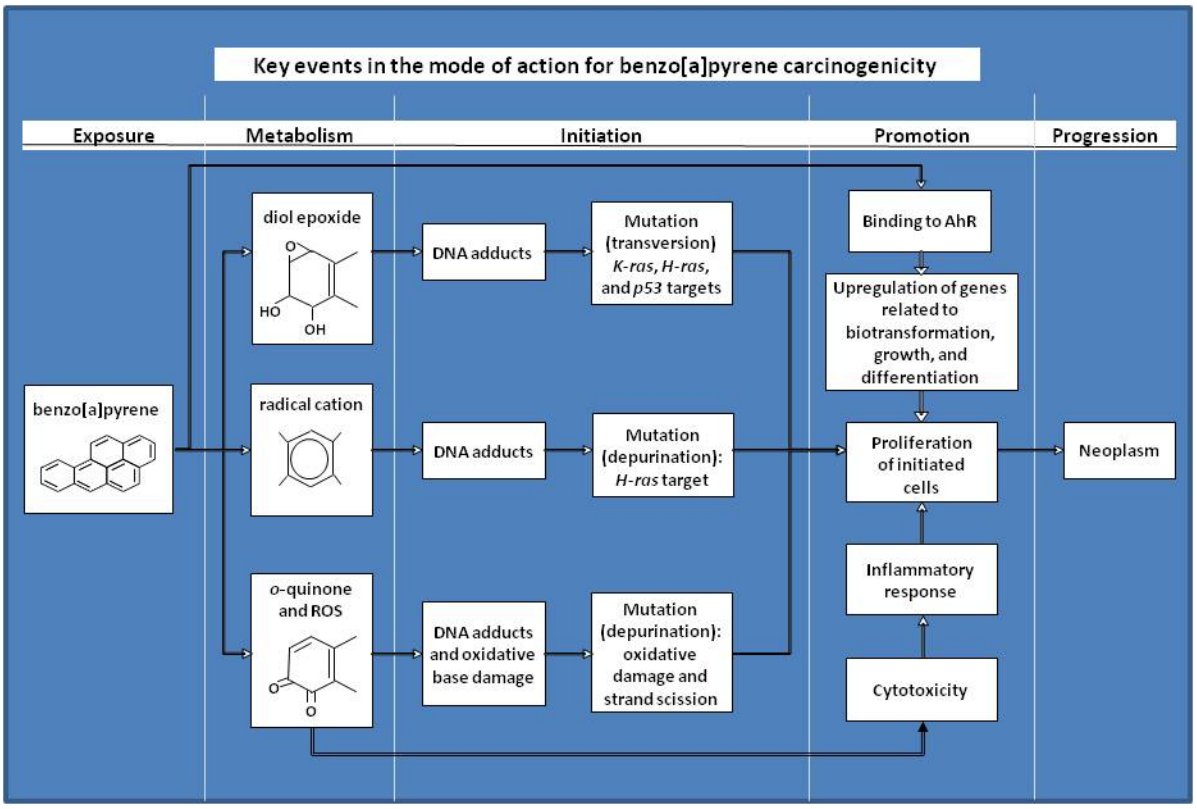

Figure 1-7. Proposed metabolic activation pathways and key events in the carcinogenic mode of action for benzo[a]pyrene.

Data in Support of the Mode of Action

Summary of metabolic activation pathways

Diol epoxide pathway. Benzo[a]pyrene diol epoxide metabolites, believed to be the most potent DNA-binding metabolites of benzo[a]pyrene, are formed through a series of Phase I metabolic reactions (see Section D.1.4 of the Supplemental Information for a more detailed review). The initial metabolism is carried out primarily by the inducible activities of CYP enzymes including

CYP1A1, CYP1B1, and CYP1A2, producing four benzo[a]pyrene epoxides. Further metabolism by epoxide hydrolase and the mixed function oxidase system yields trans-dihydrodiols, one of which, benzo[a]pyrene-7,8-diol (formed from benzo[a]pyrene-7,8-oxide), is the metabolic precursor to the potent DNA-binding metabolite benzo[a]pyrene-7,8-diol-9,10-epoxide (BPDE) (Grover, 1986). The stereochemical nature of the diol epoxide metabolite (i.e., anti- versus syn-diol epoxides) affects the number and type of adducts and mutation that occurs; the enantiomer (+)-benzo[a]pyrene-7R,8S-diol-9S,10R-epoxide [(+)-anti-BPDE] is the most potent DNA-binding metabolite of benzo[a]pyrene (Geacintov et al., 1997). Benzo[a]pyrene diol epoxide metabolites interact preferentially with the exocyclic amino groups of deoxyguanine and deoxyadenine (Geacintov et al., 1997; Jerina et al., 1991). Adducts may give rise to mutations unless these adducts are removed by DNA repair processes prior to replication. Transversion mutations (e.g., GC→TA or AT→TA) are the most common type of mutation found in mammalian cells following diol epoxide exposure (Boström et al., 2002).

Radical cation pathway. Radical cation formation involves a one-electron oxidation by CYP or peroxidase enzymes (i.e., horseradish peroxidase, prostaglandin H synthetase) that produce electrophilic radical cation intermediates (Cavalieri and Rogan, 1995, 1992). Radical cations can be further metabolized to phenols and quinones (Cavalieri et al., 1988e; Cavalieri et al., 1988d), or they can form unstable adducts with DNA that ultimately result in depurination. The predominant depurinating adducts occur at the N-3 and N-7 positions of adenine and the C-8 and N-7 positions of guanine (Cavalieri and Rogan, 1995).

o-Quinone/ROS pathway. The *o*-quinone metabolites of PAHs are formed by enzymatic dehydrogenation of dihydrodiols (Bolton et al., 2000; Penning et al., 1999; Harvey, 1996; ATSDR, 1995) (see Appendix D of the Supplemental Information). Dihydrodiol dehydrogenase enzymes are members of the α-keto reductase gene superfamily. o-Quinone metabolites are potent cytotoxins, are weakly mutagenic, and are capable of producing a broad spectrum of DNA damage. These metabolites can interact directly with DNA as well as result in the production of ROS (i.e., hydroxyl and superoxide radicals) that may produce further cytotoxicity and DNA damage. The *o*-quinone/ROS pathway also can produce depurinated DNA adducts from benzo[a]pyrene metabolites. In this pathway, and in the presence of $NAD(P)^+$, aldo-keto reductase oxidizes benzo[a]pyrene-7,8-diol to a ketol, which subsequently forms benzo[a]pyrene-7,8-dione. This and other PAH *o*-quinones react with DNA to form unstable, depurinating DNA adducts. In the presence of cellular reducing equivalents, *o*-quinones can also activate redox cycles, which produce ROS (Penning et al., 1996). DNA damage in in vitro systems following exposure to benzo[a]pyrene-7,8-dione or other *o*-quinone PAH derivatives occurs through the aldo-keto reductase (AKR) pathway and can involve the formation of stable DNA adducts (Balu et al., 2004), N-7 depurinated DNA adducts (Mccoull et al., 1999), DNA damage from ROS (8-oxo-7,8-dihydro-2'-deoxyguanosine adducts) (Park et al., 2006), and strand scission (Flowers et al., 1997; Flowers et al., 1996).

Summary of genotoxicity and mutagenicity

The ability of metabolites of benzo[a]pyrene to cause mutations and other forms of DNA damage in both in vivo and in vitro studies is well documented (see genotoxicity tables in Appendix D in Supplemental Information). With metabolic activation (e.g., the inclusion of S9), benzo[a]pyrene is consistently mutagenic in the prokaryotic *Salmonella*/Ames and *Escherichia coli* assays. In mammalian in vitro studies, benzo[a]pyrene is consistently mutagenic and clastogenic, and induces cell transformation both with and without metabolic activation. Cytogenetic damage in the form of chromosomal aberrations (CAs), micronuclei (MN), sister chromatid exchanges (SCEs), and aneuploidy are commonplace following benzo[a]pyrene exposure as are DNA adduct formation, single-strand breaks (SSB), and induction of DNA repair and unscheduled DNA synthesis (UDS). In vitro mammalian cell assays have been conducted in various test systems, including human cell lines.

In the majority of in vivo studies, benzo[a]pyrene has tested positive in multiple species and strains and under various test conditions for cell transformation, CAs, DNA adducts, DNA strand breaks, MN formation, germline mutations, somatic mutations (*H-ras*, *K-ras*, *p53*, *lacZ*, *hprt*), and SCEs. Human studies are available following exposures to PAH mixtures through cigarette smoke or occupational exposure in which benzo[a]pyrene-specific DNA adducts have been detected, and it has been demonstrated qualitatively that benzo[a]pyrene metabolites damage DNA in exposed humans.

Experimental support for the hypothesized mode of action

EPA's *Guidelines for Carcinogen Risk Assessment* [Section 2.4; (U.S. EPA, 2005a)] describe a procedure for evaluating mode-of-action data for cancer. A framework for analysis of mode of action information is provided below, providing context for the key events depicted in Figure 1-7.

Strength, consistency, and specificity of association. An extensive database of in vitro and in vivo studies demonstrating the genotoxicity and mutagenicity of benzo[a]pyrene following metabolic activation provides supporting evidence of a mutagenic mode of action for benzo[a]pyrene carcinogenicity (see Table 1-19 and Section D.5.1 of the Supplemental Information). In vitro studies overwhelmingly support the formation of DNA adducts, mutagenesis in bacteria, yeast, and mammalian cells, several measures of cytogenetic damage (CA, SCE, MN), and DNA damage. In vivo systems in animal models are predominantly positive for somatic mutations following benzo[a]pyrene exposure.

Strong evidence links the benzo[a]pyrene diol epoxide metabolic activation pathway with key mutational events in genes that are associated with tumor initiation (i.e., mutations in the *p53* tumor suppressor gene and *H-ras* or *K-ras* oncogenes) (Table 1-19). The mutagenic potency of the benzo[a]pyrene diol epoxide BPDE (and specifically (+)-anti-BPDE) has been confirmed in mutagenicity assays in bacterial and in vitro mammalian systems (Malaveille et al., 1977; Newbold

and Brookes, 1976). The major BPDE-DNA adducts observed in vivo are formed through reaction of BPDE with the exocyclic amino groups of deoxyguanosine and deoxyadenosine, with a preference for forming the stable benzo[a]pyrene-7,8-diol-9,10-epoxide-N2-deoxyguanosine (BPdG) adducts (Xue and Warshawsky, 2005; Jeffrey et al., 1976; Daudel et al., 1975). BPdG adducts form in human cells and mouse skin (Grover et al., 1976; Osborne et al., 1976), and were first chemically confirmed in humans in placental tissues (Manchester et al., 1988).

G→T transversions, displaying strand bias, are the predominant type of mutations caused by benzo[a]pyrene in several biological systems (Liu et al., 2005; Hainaut and Pfeifer, 2001; Marshall et al., 1984) and sites of DNA adduction at guanine positions in cultured human HeLa cells or plasmids containing the human *p53* gene exposed to benzo[a]pyrene diol epoxide correspond to *p53* mutational hotspots observed in human lung cancers (Denissenko et al., 1996; Puisieux et al., 1991). Mutational hotspots have been linked to regions of inefficient nucleotide excision repair of BPDE-DNA adducts, both in vitro in *hprt* in human fibroblasts (Wei et al., 1995) and *p53* in human bronchial epithelial BEAS-2B cells, and in vivo in nontumorous lung tissue from smokers with lung cancer (Hussain et al., 2001). Results in support of a mutagenic mode of action via benzo[a]pyrene diol epoxide include observations of frequent G→T transversion mutations in *p53* and ras genes in lung tumors of human cancer patients exposed to coal smoke (Keohavong et al., 2003; Demarini et al., 2001). In addition, mice exposed to benzo[a]pyrene in the diet (Culp et al., 2000) or by i.p. injection (Nesnow et al., 1998a; Nesnow et al., 1998b; Nesnow et al., 1996, 1995; Mass et al., 1993) had forestomach or lung tumors, respectively, showing BPdG adduct formation and frequent G→T or C transversions in the *K-ras* gene.

An experimental challenge to the hypothesized mode of action further strengthens the association between the diol epoxide pathway, DNA adduct formation, and tumorigenesis. Isothiocyanates, reported to inhibit lung tumorigenesis in mice treated with benzo[a]pyrene, were also observed to significantly reduce conversion of benzo[a]pyrene by cytochrome P450 enzymes to the benzo[a]pyrene-7,8-diol, and significantly reduce formation of BPDE-DNA adducts in the lung and liver of female A/J mice exposed via gavage (Sticha et al., 2000). ther supporting evidence includes observations of elevated BPDE-DNA adduct levels in WBCs of groups of coke oven workers and chimney sweeps, occupations with known elevated risks of cancer (Vineis et al., 2007; Pavanello et al., 2006; Pavanello et al., 2005; Pavanello et al., 2004; Pavanello et al., 1999; Rojas et al., 1998; Rojas et al., 1995), and in lung tissue from tobacco smokers with lung cancer (Rojas et al., 2004; Godschalk et al., 2002; Rojas et al., 1998; Andreassen et al., 1996; Alexandrov et al., 1992). Several epidemiological studies have indicated that PAH-exposed individuals who are homozygous for a CYP1A1 polymorphism, which increases the inducibility of this enzyme (thus increasing the capacity to produce benzo[a]pyrene diol epoxide), have increased levels of PAH or BPDE-DNA adducts (Aklillu et al., 2005; Alexandrov et al., 2002; Bartsch et al., 2000; Perera and Weinstein, 2000).

Support for the *o*-quinone/ROS pathway contributing to tumor initiation via mutagenic events includes in vitro demonstration that several types of DNA damage can occur from *o*-quinones and ROS (Park et al., 2006; Balu et al., 2004; Mccoull et al., 1999; Flowers et al., 1997; Flowers et al., 1996). In addition, benzo[a]pyrene-7,8-dione can induce mutations in the *p53* tumor suppressor gene using an in vitro yeast reporter gene assay (Park et al., 2008; Shen et al., 2006; Yu et al., 2002), and dominant *p53* mutations induced by benzo[a]pyrene-7,8-dione in this system corresponded to *p53* mutational hotspots observed in human lung cancer tissue (Park et al., 2008).

Dose-response concordance and temporal relationship. Studies in humans demonstrating that benzo[a]pyrene-induced mutational events in *p53* or *ras* oncogenes precede tumor formation are not available, but there is evidence linking benzo[a]pyrene exposure to signature mutational events in humans. In vitro exposure of human *p53* knock-in murine fibroblasts to 1 µM benzo[a]pyrene for 4–6 days induced *p53* mutations with similar features to those identified in *p53* mutations in human lung cancer; i.e., predominance of G→T transversions with strand bias and mutational hotspots at codons 157–158 (Liu et al., 2005). *Anti*-BPDE exposure in vitro activated the cloned human c-Ha-*ras*-1 proto-oncogene and covalently bound to DNA, preferentially forming adducts at the N2 position of guanine (Marshall et al., 1984).

Bennett et al. (1999) demonstrated a dose-response relationship between smoking history/intensity and the types of *p53* mutations associated with benzo[a]pyrene (G→T transversions) in human lung cancer patients (Table 1-19). In lung tumors of nonsmokers, 10% of *p53* mutations were G→T transversions, versus 40% in lung tumors from smokers with >60 pack-years of exposure.

Skin and forestomach tumors in mice showed a dose-response relationship, including temporal variation, with levels of BPDE-DNA adducts (Table 1-19). In a study using mice treated dermally with benzo[a]pyrene once or twice per week for up to 15 weeks (10, 25, or 50 nmol benzo[a]pyrene per application), levels of benzo[a]pyrene-DNA adducts in the skin, lung, and liver increased with increasing time of exposure and increasing dose levels (Talaska et al., 2006). Levels at the end of the exposure period were highest in the skin; levels in the lung and liver at the same time were 10- and 20-fold lower, respectively. The dose-response for benzo[a]pyrene-DNA adducts in skin and lung increased in an apparent biphasic, curvilinear, manner with increasing steepness at the mid to high exposure concentrations.

Mice exposed dermally to benzo[a]pyrene followed by the promoter 12-*O*-tetradecanoyl-phorbol 13-acetate (TPA) for 26 weeks were found to have c-Ha-*ras* mutations in normal-looking and hyperplastic lesions as well as in tumors (Wei et al., 1999). These mutations increased in frequency from normal to hyperplastic to neoplastic skin samples, indicating activation of the c-Ha-*ras* proto-oncogene may precede tumorigenesis. Although the c-Ha-*ras* mutation did not appear in all skin tumors tested (21% were negative), the vast majority of these mutations detected in tumors (74 and 61% for the low and high dose groups, respectively) were G→T transversions.

Another study examined the dose-response relationship and the time course of benzo[a]pyrene-induced skin damage (Table 1-19), DNA adduct formation, and tumor formation in female mice. Mice were treated dermally with 0, 16, 32, or 64 µg of benzo[a]pyrene once per week for 29 weeks (Albert et al., 1991). Indices of skin damage and levels of BPDE-DNA adducts in skin reached plateau levels in exposed groups by 2–4 weeks of exposure. With increasing dose level, levels of BPDE-DNA adducts (fmol/µg DNA) initially increased in a linear manner and began to plateau at doses ≥32 µg/week. Tumors began appearing after 12–14 weeks of exposure for the mid- and high-dose groups and at 18 weeks for the low-dose group. At study termination (35 weeks after start of exposure), the mean number of tumors per mouse was approximately one per mouse in the low- and mid-dose groups and eight per mouse in the high-dose group. The time-course data indicate that benzo[a]pyrene-induced increases in BPDE-DNA adducts preceded the appearance of skin tumors, consistent with the formation of DNA adducts as a precursor event in benzo[a]pyrene-induced skin tumors. A follow-up to this study by the same authors (Albert et al., 1996) measured DNA adducts, necrosis, and inflammation (marked by an increase in leukocytes) in the skin of treated mice after 5 weeks of dermal exposure. In the 64 µg/week dose group, statistically elevated levels of DNA adducts, inflammation, and necrosis were reported; however, in the lower dose group (16 µg/week), DNA adducts were statistically significantly elevated without increases in inflammation and necrosis.

Culp and Beland (1994) demonstrated a dose-dependent increase in DNA adducts in the livers, lungs, and forestomachs of male B6C3F$_1$ mice fed coal tar for 28 days or benzo[a]pyrene for 21 days. Culp and Beland (1994) then compared dose-response relationships for BPDE-DNA adducts and tumors in female B6C3F$_1$ mice exposed to benzo[a]pyrene in the diet at 0, 18.5, 90, or 350 µg/day for 28 days (to examine adducts) or 2 years (to examine tumors) (Table 1-19). The benzo[a]pyrene dose-tumor response data showed a sharp increase in forestomach tumor incidence between the 18.5 µg/day group (6% incidence) and the 90 µg/day group (78% incidence). The BPDE-DNA adduct levels in forestomach showed a relatively linear dose-response throughout the benzo[a]pyrene dose range tested. The appearance of increased levels of BPDE-DNA adducts in the target tissue at 28 days is temporally consistent with the contribution of these adducts to the initiation of forestomach tumors. Furthermore, about 60% of the examined tumors had mutations in the *K-ras* oncogene at codons 12 and 13, which were G→T or G→C transversions indicative of BPDE reactions with DNA (Culp et al., 1996). In addition to forestomach tumors, the 2-year oral exposures to benzo[a]pyrene also resulted in an increased incidence of tumors of the esophagus, tongue, and larynx in female mice (Beland and Culp, 1998).

Biological plausibility and coherence. The evidence for a mutagenic mode of action for benzo[a]pyrene is consistent with the current understanding that mutations in *p53* and *ras* oncogenes are associated with increased risk of tumor initiation (Table 1-19). The benzo[a]pyrene database is internally consistent in providing evidence for the formation of BPDE-DNA adducts and BPDE-induced mutations associated with tumor initiation in cancer tissue from humans exposed to

complex mixtures containing benzo[a]pyrene(Keohavong et al., 2003; Pfeifer and Hainaut, 2003; Pfeifer et al., 2002; Demarini et al., 2001; Hainaut and Pfeifer, 2001; Bennett et al., 1999), in animals exposed to benzo[a]pyrene (Culp et al., 2000; Nesnow et al., 1998a; Nesnow et al., 1998b; Nesnow et al., 1996, 1995; Mass et al., 1993), and in in vitro systems (Denissenko et al., 1996; Puisieux et al., 1991). Consistent supporting evidence includes: (1) elevated BPDE-DNA adduct levels in tobacco smokers with lung cancer (Rojas et al., 2004; Godschalk et al., 2002; Rojas et al., 1998; Andreassen et al., 1996; Alexandrov et al., 1992); (2) demonstration of dose-response relationships between G→T transversions in *p53* mutations in lung tumors and smoking intensity (Bennett et al., 1999); (3) the extensive database of in vitro and in vivo studies demonstrating the genotoxicity and mutagenicity of benzo[a]pyrene following metabolic activation; and (4) general consistency between temporal and dose-response relationships for BPDE-DNA adduct levels and tumor incidence in studies of animals exposed to benzo[a]pyrene (Culp et al., 1996; Albert et al., 1991). There is also supporting evidence that contributions to tumor initiation through mutagenic events can be made by the radical cation (Chakravarti et al., 1995; Rogan et al., 1993) and *o*-quinone/ROS metabolic activation pathways (Park et al., 2008; Park et al., 2006; Shen et al., 2006; Balu et al., 2004; Yu et al., 2002; Mccoull et al., 1999; Flowers et al., 1997; Flowers et al., 1996).

Table 1-19. Experimental support for the postulated key events for mutagenic mode of action

1. Bioactivation of benzo[a]pyrene to DNA-reactive metabolites via three possible metabolic activation pathways: a diol epoxide pathway, a radical cation pathway, and an *o*-quinone and ROS pathway
Evidence that benzo[a]pyrene metabolites induce key events: • Metabolism of benzo[a]pyrene via all three pathways has been demonstrated in multiple in vitro studies, and the diol epoxide and radical cation metabolic activation pathways have been demonstrated in in vivo studies in humans and animals (see *Metabolic Activation Pathways* section) • Multiple in vivo studies in humans and animals have demonstrated distribution of reactive metabolites to target tissues *Human evidence that key events are necessary for carcinogenesis:* • Humans with CYP polymorphisms or lacking a functional GSTM1 gene form higher levels of benzo[a]pyrene diol epoxides, leading to increased BPDE-DNA adduct formation and increased risk of cancer (Vineis et al., 2007; Pavanello et al., 2005; Pavanello et al., 2004; Alexandrov et al., 2002; Perera and Weinstein, 2000)
2. Direct DNA damage by the reactive metabolites, including the formation of DNA adducts and ROS-mediated damage
Evidence that benzo[a]pyrene metabolites induce key events: • Reactive benzo[a]pyrene metabolites have demonstrated genotoxicity in most in vivo and in vitro systems in which they have been tested, including the bacterial mutation assay, transgenic mouse assay, dominant lethal mutations in mice, BPDE-DNA adduct detection in humans and animals, and DNA damage, CAs, MN formation, and SCE in animals (Appendix D in Supplemental Information)

- Multiple in vivo benzo[a]pyrene animal exposure studies have demonstrated DNA adduct formation in target tissues that precede tumor formation and increase in frequency with dose (Culp et al., 1996; Talaska et al., 1996; Culp and Beland, 1994; Albert et al., 1991)
- Treatment with isothiocyanates, which inhibit the biotransformation of benzo[a]pyrene to the 7,8-diol and BPDE-DNA adduct formation, also inhibits lung tumorigenesis in mice exposed to benzo[a]pyrene (Sticha et al., 2000)
- Benzo[a]pyrene diol epoxide metabolites interact preferentially with the exocyclic amino groups of deoxyguanine and deoxyadenine in DNA (Geacintov et al., 1997; Jerina et al., 1991; Koreeda et al., 1978; Jeffrey et al., 1976)
- Benzo[a]pyrene *o*-quinone metabolites are capable of activating redox cycles and producing ROS that cause oxidative base damage (Park et al., 2006; Balu et al., 2004; Mccoull et al., 1999; Flowers et al., 1997; Flowers et al., 1996)

Human evidence that key events are necessary for carcinogenesis:

- Detection of benzo[a]pyrene diol epoxide-specific DNA adducts is associated with increased cancer risk in humans that are occupationally exposed (see *Evidence in Humans* section)
- These benzo[a]pyrene diol epoxides formed BPDE-DNA adducts preferentially at guanine residues that have been detected in tissues of humans with cancer who were exposed to PAHs (Vineis and Perera, 2007; Rojas et al., 2004; Godschalk et al., 2002; Li et al., 2001; Pavanello et al., 1999; Rojas et al., 1998; Andreassen et al., 1996; Alexandrov et al., 1992)

3. Formation and fixation of DNA mutations, particularly in tumor suppressor genes or oncogenes associated with tumor initiation

Evidence that benzo[a]pyrene metabolites induce key events:

- Several in vivo exposure studies have observed benzo[a]pyrene diol epoxide-specific mutational spectra (e.g., G→T transversion mutations) in *K-ras*, *H-ras*, and *p53* in forestomach or lung tumors (Culp et al., 2000; Nesnow et al., 1998a; Nesnow et al., 1998b; Nesnow et al., 1996, 1995; Mass et al., 1993)
- Multiple studies in vivo and in vitro have identified benzo[a]pyrene-specific mutations in *H-ras*, *K-ras*, and *p53* in target tissues preceding tumor formation (Liu et al., 2005; Wei et al., 1999; Culp et al., 1996) (Chakravarti et al., 1995; Ruggeri et al., 1993)

Human evidence that key events are necessary for carcinogenesis:

- DNA adducts formed by the benzo[a]pyrene diol epoxide reacting with guanine bases lead predominantly to G→T transversion mutations; these specific mutational spectra have been identified in PAH-associated tumors in humans at mutational hotspots, including oncogenes (*K-ras*) and tumor suppressor genes (*p53*) (Liu et al., 2005; Keohavong et al., 2003; Pfeifer and Hainaut, 2003; Pfeifer et al., 2002; Demarini et al., 2001; Hainaut and Pfeifer, 2001; Bennett et al., 1999; Denissenko et al., 1996; Puisieux et al., 1991; Marshall et al., 1984; Koreeda et al., 1978; Jeffrey et al., 1976)

4. Clonal expansion of mutated cells during the promotion and progression phases of cancer development
Evidence that benzo[a]pyrene metabolites induce key events: • Benzo[a]pyrene has been shown to be a complete carcinogen, in that skin tumors in mice, rats, rabbits, and guinea pigs have been associated with repeated application of benzo[a]pyrene to skin in the absence of exogenous promoters (IPCS, 1998; Sivak et al., 1997; ATSDR, 1995; Grimmer et al., 1984; Habs et al., 1984; Grimmer et al., 1983; IARC, 1983; Habs et al., 1980; Schmähl et al., 1977; IARC, 1973; Schmidt et al., 1973; Roe et al., 1970; Poel, 1963, 1959) • Mice exposed dermally to benzo[a]pyrene for 26 weeks were found to have increased frequencies of *H-ras* mutations in exposure-induced hyperplastic lesions that were further increased in tumors (Wei et al., 1999) • AhR activation by PAHs (including benzo[a]pyrene) upregulates genes responsible for tumor promotion and increases tumor incidence in mice (Ma and Lu, 2007; Talaska et al., 2006; Shimizu et al., 2000)

Other possible modes of action

The carcinogenic process for benzo[a]pyrene is likely to be related to some combination of molecular events resulting from the formation of several reactive metabolites that interact with DNA to form adducts and produce DNA damage resulting in mutations in cancer-related genes, such as tumor suppressor genes or oncogenes. These events may reflect the initiation potency of benzo[a]pyrene. However, benzo[a]pyrene possesses promotional capabilities that may be related to AhR affinity, immune suppression, cytotoxicity and inflammation (including the formation of ROS), as well as the inhibition of gap junctional intercellular communication (GJIC).

The ability of certain PAHs to act as initiators and promoters may increase their carcinogenic potency. The promotional effects of PAHs appear to be related to AhR affinity and the upregulation of genes related to growth and differentiation (Boström et al., 2002). The genes regulated by this receptor belong to two major functional groups (i.e., induction of metabolism or regulation cell differentiation and proliferation). PAHs bind to the cytosolic AhR in complex with heat shock protein 90. The ligand-bound receptor is then transported to the nucleus in complex with the AhR nuclear translocator protein. The AhR complex interacts with AhR elements of DNA to increase the transcription of proteins associated with induction of metabolism and regulation of cell differentiation and proliferation. Following benzo[a]pyrene exposure, Ah-responsive mice were more susceptible to tumorigenicity in target tissues such as liver, lung, and skin as compared to Ah-unresponsive mice (Ma and Lu, 2007; Talaska et al., 2006; Shimizu et al., 2000).

Benzo[a]pyrene has both inflammatory and immunosuppressive effects that may function to promote tumorigenesis. Inflammatory responses to cytotoxicity may contribute to the tumor promotion process; for example, benzo[a]pyrene quinones (1,6-, 3,6-, and 6,12-benzo[a]pyrene-quinone) generated ROS and increased cell proliferation by enhancing the epidermal growth factor receptor pathway in cultured breast epithelial cells (Burdick et al., 2003). In addition, several studies have demonstrated that exposure to benzo[a]pyrene increases the production of inflammatory cytokines, which may contribute to cancer progression (N'Diaye et al., 2006; Tamaki

et al., 2004; Garçon et al., 2001b; Garçon et al., 2001a; Albert et al., 1996; Albert et al., 1991). One of these studies, Albert et al. (1996), measured DNA adducts, necrosis, and inflammation (marked by an increase in leukocytes) in the skin of benzo[a]pyrene-treated mice after 5 weeks of dermal exposure. In the highest dose group, statistically elevated levels of DNA adducts, inflammation, and necrosis were reported; however, in the lower dose group, DNA adducts were statistically significantly elevated without increases in inflammation and necrosis. It is likely that inflammation promotes the formation of tumors at high doses of benzo[a]pyrene.

In addition to inflammation, immunosuppressive effects of benzo[a]pyrene have been noted (as reviewed in, Zaccaria and Mcclure, 2013). Immune effects of benzo[a]pyrene exposure (see Section 1.1.3) may provide an environment where tumor cells can evade detection by immune surveillance mechanisms normally responsible for recognizing and eliminating nascent cancer cells (Hanahan and Weinberg, 2011). In addition, the developing fetus may be even more sensitive to these effects; Urso and Gengozian (1980) found that mice exposed to benzo[a]pyrene in utero not only had a significantly increased tumor incidence as adults but also a persistently suppressed immune system.

Gap junctions are channels between cells that are crucial for differentiation, proliferation, apoptosis, and cell death. Interruption of GJIC is associated with a loss of cellular control of growth and differentiation, and consequently with the two epigenetic steps of tumor formation, promotion and progression. Thus, the inhibition of gap junctional intercellular communication by benzo[a]pyrene, observed in vitro (Sharovskaya et al., 2006; Bláha et al., 2002), provides another mechanism of tumor promotion.

In summary, there are tumor-promoting effects of PAH exposures that are not mutagenic. Although these effects are observed following benzo[a]pyrene-specific exposures, the occurrence of BPDE-DNA adducts and associated mutations that precede both cytotoxicity and tumor formation and increase with dose provides evidence that mutagenicity is the primary event that initiates tumorigenesis following benzo[a]pyrene exposures. A biologically plausible mode of action may involve a combination of effects induced by benzo[a]pyrene, with mutagenicity as the initiating tumorigenic event. Subsequent AhR activation and cytotoxicity could then lead to increased ROS formation, regenerative cell proliferation, and inflammatory responses, which, along with evasion of immune surveillance and GJIC, would provide an environment where the selection for mutated cells increases the rate of mutation, allowing clonal expansion and progression of these tumor cells to occur. However, it was determined that, in comparison to the large database on the mutagenicity of benzo[a]pyrene, there were insufficient data to develop a separate mode of action analysis for these promotional effects.

Conclusions about the hypothesized mode of action

There is sufficient evidence to conclude that the major mode of action for benzo[a]pyrene carcinogenicity involves mutagenicity mediated by DNA reactive metabolites. The evidence for a mutagenic mode of action for benzo[a]pyrene is consistent with the current understanding that

mutations in *p53* and *ras* oncogenes are associated with increased risk of tumor initiation. The benzo[a]pyrene database provides strong and consistent evidence for BPDE-induced mutations associated with tumor initiation in cancer tissue from humans exposed to complex mixtures containing benzo[a]pyrene, in animals exposed to benzo[a]pyrene, and in in vitro systems. Supporting evidence suggests that contributions to tumor initiation through potential mutagenic events can be made by the radical cation and *o*-quinone/ROS metabolic activation pathways. Other processes may contribute to the carcinogenicity of benzo[a]pyrene via the promotion and progression phases of cancer development (e.g., inflammation, cytotoxicity, sustained regenerative cell proliferation).

Support for the Hypothesized Mode of Action in Test Animals

Benzo[a]pyrene induces gene mutations in a variety of in vivo and in vitro systems and produces tumors in all animal species tested and by all routes of exposure (see Appendix D in Supplemental Information). Strong, consistent evidence in animal models supports the postulated key events: the metabolism of benzo[a]pyrene to DNA-reactive intermediates, the formation of DNA adducts, the subsequent occurrence of mutations in oncogenes and tumor suppressor genes, and the clonal expansion of mutated cells.

Relevance of the Hypothesized Mode of Action to Humans

A substantial database indicates that the postulated key events for a mutagenic mode of action all occur in human tissues. Evidence is available from studies of humans exposed to PAH mixtures (including coal smoke and tobacco smoke) indicating a contributing role for benzo[a]pyrene diol epoxide in inducing key mutational events in genes that are associated with tumor initiation (mutations in the *p53* tumor suppressor gene and *H-ras* or *K-ras* oncogenes). The evidence includes observations of a spectrum of mutations in *ras* oncogenes and the *p53* gene in lung tumors of human patients exposed to coal smoke or tobacco smoke) that are similar to the spectrum of mutations caused by benzo[a]pyrene diol epoxide in several biological systems, including tumors from mice exposed to benzo[a]pyrene. Additional supporting evidence includes correspondence between hotspots of *p53* mutations in human lung cancers and sites of DNA adduction by benzo[a]pyrene diol epoxide in experimental systems, and elevated BPDE-DNA adduct levels in respiratory tissue of lung cancer patients or tobacco smokers with lung cancer.

Populations or Lifestages Particularly Susceptible to the Hypothesized Mode of Action

A mutagenic mode of action for benzo[a]pyrene-induced carcinogenicity is considered relevant to all populations and lifestages. The current understanding of biology of cancer indicates that mutagenic chemicals, such as benzo[a]pyrene, are expected to exhibit a greater effect in early life exposure versus later life exposure (U.S. EPA, 2005b; Vesselinovitch et al., 1979). The EPA's *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (U.S. EPA, 2005b) recommends the application of age-dependent adjustment factors (ADAFs) for carcinogens

that act through a mutagenic mode of action. Since a determination that benzo[a]pyrene acts through a mutagenic mode of carcinogenic action has been made, ADAFs should be applied along with exposure information to estimate cancer risks for early-life exposure.

Toxicokinetic information suggest early lifestages may have lower levels of some CYP enzymes than adults (Ginsberg et al., 2004; Cresteil, 1998); thus, lower levels of mutagenic metabolites may be formed in early lifestages. Though expression of bioactivating enzymes is believed to be lower in the developing fetus and children, metabolism of benzo[a]pyrene still occurs, as indicated by the detection of benzo[a]pyrene-DNA or protein adducts or urinary metabolites (Naufal et al., 2010; Ruchirawat et al., 2010; Suter et al., 2010; Mielzyńska et al., 2006; Perera et al., 2005a; Tang et al., 1999; Whyatt et al., 1998). While expression of CYP enzymes is lower in fetuses and infants, the greater liver to body mass ratio and increased blood flow to liver in fetuses and infants may compensate for the decreased expression of CYP enzymes (Ginsberg et al., 2004). Activity of Phase II detoxifying enzymes in neonates and children is adequate for sulfation but decreased for glucuronidation and glutathione conjugation (Ginsberg et al., 2004). The conjugation of benzo[a]pyrene-4,5-oxide with glutathione was approximately one-third less in human fetal liver cytosol compared to adult liver cytosol (Pacifici et al., 1988).

In addition, newborn or infant mice develop liver and lung tumors more readily than young adult mice following acute i.p. exposures to benzo[a]pyrene (Vesselinovitch et al., 1975). These results indicate that exposure to benzo[a]pyrene during early life stages presents additional risk for cancer, compared with exposure during adulthood, despite lower metabolic activity in early lifestages. Population variability in metabolism and detoxification of benzo[a]pyrene, in addition to DNA repair capability, may affect cancer risk. Polymorphic variations in the human population in CYP1A1, CYP1B1, and other CYP enzymes have been implicated as determinants of increased individual cancer risk in some studies (Ickstadt et al., 2008; Aklillu et al., 2005; Alexandrov et al., 2002; Perera and Weinstein, 2000). Some evidence suggests that humans lacking a functional GSTM1 gene have higher BPDE-DNA adduct levels and are thus at greater risk for cancer (Binkova et al., 2007; Vineis et al., 2007; Pavanello et al., 2005; Pavanello et al., 2004; Alexandrov et al., 2002; Perera and Weinstein, 2000). In addition, acquired deficiencies or inherited gene polymorphisms that affect the efficiency or fidelity of DNA repair may also influence individual susceptibility to cancer from environmental mutagens (Wang et al., 2010; Ickstadt et al., 2008; Binkova et al., 2007; Matullo et al., 2003; Shen et al., 2003; Cheng et al., 2000; Perera and Weinstein, 2000; Wei et al., 2000; Amos et al., 1999). In general, however, available support for the role of single polymorphisms in significantly modulating human PAH cancer risk from benzo[a]pyrene or other PAHs is relatively weak or inconsistent. Combinations of polymorphisms, on the other hand, may be critical determinants of a cumulative DNA-damaging dose, and thus indicate greater susceptibility to cancer from benzo[a]pyrene exposure (Vineis et al., 2007).

Analysis of Toxicogenomics Data

An analysis of pathway-based transcriptomic data was conducted to help inform the cancer mode of action for benzo[a]pyrene (see the Supplemental Information for details of this analysis). These data support a mutagenic and cellular proliferation mode of action that follows three candidate pathways: aryl hydrocarbon signaling; DNA damage regulation of the G1/S phase transition; and/or Nrf2 regulation of oxidative stress. Specifically, the analysis showed that benzo[a]pyrene may activate the AhR, leading to the formation of oxidative metabolites and radicals which may lead to oxidative damage and DNA damage. Subsequently, DNA damage can occur and activate p53 and p53 target genes, including p21 and MDM2. In addition, the data indicate that p53 signaling may be decreased under these conditions, as ubiquitin and MDM2 are both upregulated, and work together to degrade p53. Furthermore, the transcriptional upregulation of cyclin D may result in enough cyclin D protein to overcome the p21 inhibitory competition for CDK4, allowing for G1/S phase transition to occur. The data also support the hypothesis that an upregulation of proliferating cell nuclear antigen (PCNA) in combination with the upregulation of ubiquitin indicates that cells are moving towards the G1/S phase transition. Although the alterations to the Nrf2 pathway suggest cells are preparing for a pro-apoptotic environment, there is no transcriptional evidence that the apoptotic pathways are being activated.

There are uncertainties associated with the available transcriptomics data. For instance, the available studies only evaluate gene expression following benzo[a]pyrene exposure and do not monitor changes in protein or metabolite expression, which would be more indicative of an actual cellular state change. Further research is required at the molecular level to demonstrate that the cellular signaling events being inferred from such data are actually operative and result in phenotypic changes. In addition, this analysis relied upon two short term studies that evaluated mRNA expression levels in a single tissue (liver) and species (mouse) and were conducted at relatively high doses.

1.2. SUMMARY AND EVALUATION

1.2.1. Weight of Evidence for Effects Other than Cancer

The weight of the evidence from human and animal studies indicates that the strongest evidence for human hazards following benzo[a]pyrene exposure is for developmental toxicity (including neurodevelopmental toxicity), reproductive toxicity, and, to a lesser extent, immunotoxicity. Some supporting data in humans exposed to PAH mixtures is available which utilizes benzo[a]pyrene air monitoring data or report associations between particular health endpoints and concentrations of benzo[a]pyrene-DNA adducts in blood or tissue. In general, the available human studies report effects that are analogous to the effects observed in animal toxicological studies (especially those regarding developmental and reproductive effects), and provide qualitative, supplemental evidence for the effect-specific hazards identified in Sections 1.1.1–1.1.3. The endpoint categories included in Section 1.1.4 “Other Toxicities”, i.e., liver,

kidney, cardiovascular and nervous system toxicity (in adult animals) had less robust evidence of hazard from the available chronic or subchronic oral and inhalation exposure studies in the benzo[a]pyrene database.

In animals, evidence of developmental toxicity (including developmental neurotoxicity) has been observed across species and dosing regimens (noting that the potential for developmental toxicity following combined gestational and neonatal exposure has not been tested). The available evidence from mice and rats treated by gavage during gestation or in the early postnatal period demonstrate developmental effects including decreased body weight, decreased fetal survival, decreased fertility, atrophy of reproductive organs, abnormal neurophysiological responses, and altered neurobehavioral outcomes (Chen et al., 2012; Jules et al., 2012; Sheng et al., 2010; Bouayed et al., 2009a; McCallister et al., 2008; Kristensen et al., 1995; Mackenzie and Angevine, 1981). Evidence of developmental toxicity has also been observed following inhalation exposure in animals. Decreased fetal survival has been observed in rats exposed to benzo[a]pyrene via inhalation during gestation (Wormley et al., 2004; Archibong et al., 2002).

Several studies in animals have indicated that oral exposure to benzo[a]pyrene in early life may result in altered neurobehavioral outcomes and sensorimotor development (Chen et al., 2012; Bouayed et al., 2009a). Following inhalation exposure during gestation, testing at adult ages also suggested possible changes in neurophysiological measures in rats and mice (Li et al., 2012; Wormley et al., 2004) and behavioral responses in mice (Li et al., 2012). Long-lasting behavioral effects may be irreversible and have been consistently observed after exposure to benzo[a]pyrene across species, and across exposure and testing paradigms. Although a mode-of action has not been defined, these alterations are supported by mechanistic changes in levels of brain monoamine neurotransmitters and NMDA receptors, as well as increases in oxidative stress. Overall, the oral and inhalation data support the conclusion that developmental toxicity (including developmental neurotoxicity) is a human hazard following exposure to benzo[a]pyrene.

When considering the data on potential nervous system effects across lifestages, findings of behavioral changes were largely coherent across studies of varied design. For example, similar decrements were observed in Morris water maze tests after oral exposure to neonatal or adult rats (Chen et al., 2012; Chengzhi et al., 2011), i.p. exposure to weanling or pubertal rats (Qiu et al., 2011; Tang et al., 2011; Xia et al., 2011), and i.p. exposure to adult mice (Grova et al., 2007). Likewise, consistent changes in elevated plus maze performance were observed after oral exposure to neonatal rats or mice (Chen et al., 2012; Bouayed et al., 2009a), and i.p. exposure to adult mice (Grova et al., 2007). While the rodent behavioral alterations may relate to effects observed in humans after prenatal or adult exposure to PAH mixtures, including changes in mood, short term memory, and sensorimotor-related responses, it is recognized that animals and humans may not necessarily experience the same effects or functional changes (Francis et al., 1990). Overall, while the adult neurotoxicity data are somewhat consistent with developmental neurotoxicity endpoints, and considered suggestive of a hazard, these data were comparably less robust than the data

supporting developmental neurotoxicity as a hazard, and additional studies are needed to identify adult neurotoxicity as a human hazard of benzo[a]pyrene exposure (see Section 1.1.4). However, as stated above, the data support neurotoxicity, based on developmental neurotoxicity, as a human hazard of benzo[a]pyrene exposure.

In animals, evidence of reproductive toxicity has been observed across species and dosing regimens. Male and female reproductive toxicity, as evidenced by effects on sperm parameters, decreased reproductive organ weights, histological changes, and hormone alterations, have been observed after oral exposure in rats and mice (Chen et al., 2011; Chung et al., 2011; Mohamed et al., 2010; Zheng et al., 2010; Mackenzie and Angevine, 1981). Evidence of reproductive toxicity has also been observed following inhalation exposure in animals. Male reproductive toxicity, as evidenced by effects on sperm parameters, decreased testes weight, and hormone alterations, has also been observed in rats following subchronic inhalation exposure to benzo[a]pyrene (Archibong et al., 2008; Ramesh et al., 2008). Female reproductive toxicity, as evidenced by modified hormone levels (Archibong et al., 2002), as well as decreased ovulation and estrous cycle length (Archibong et al., 2012) has been observed following inhalation exposure. Overall, the oral and inhalation data support the conclusion that reproductive toxicity is a human hazard following exposure to benzo[a]pyrene.

Benzo[a]pyrene exposure has also been shown to lead to altered immune cell populations and histopathological changes in immune system organs (Kroese et al., 2001; De Jong et al., 1999), as well as thymic and splenic effects following subchronic oral exposure. Varying immunosuppressive responses are also observed in short-term oral and injection studies. Overall, the available animal data support the conclusion that immunotoxicity is a potential human hazard of benzo[a]pyrene exposure.

Effects in other organ systems were observed following benzo[a]pyrene exposure including liver, kidney, and cardiovascular effects (see Section 1.1.4) but had less robust evidence of hazard from the available chronic or subchronic exposure studies in the benzo[a]pyrene database, and are discussed below.

Short duration animal studies and studies by other routes of exposure (e.g., i.p. and installation), as well as studies in genetically modified, highly susceptible animal strains (e.g., APOE−/− mice) provide suggestive evidence of cardiovascular toxicity associated with benzo[a]pyrene exposure. However, the interpretation of hazard is complicated by the paucity of studies examining cardiovascular endpoints in humans and wild-type laboratory animals exposed by environmentally relevant routes for subchronic or chronic durations. In addition, interpretation of evidence of cardiovascular effects from human studies is complicated by issues of co-exposure in in populations highly exposed to benzo[a]pyrene as a component of a complex PAH mixtures. Thus, considering the lower confidence in this hazard, the cardiovascular endpoints were not considered further for dose-response.

Evidence of liver effects following subchronic to chronic exposure to benzo[a]pyrene was generally limited to increases in liver weight with little evidence for histological findings or other indicators of hepatoxicity. Therefore, at this time, no conclusion is drawn regarding liver toxicity as a human hazard of benzo[a]pyrene exposure induced toxicity.

Few studies are available to inform the potential of kidney effects after subchronic or chronic exposure to benzo[a]pyrene. Confidence in the single subchronic study which observed an apparent increase in kidney lesions in one sex of rats (Knuckles et al., 2001) was decreased by incomplete reporting of study methods and results (see Section 1.1.4). Therefore, at this time, no conclusion is drawn regarding kidney toxicity as a human hazard of benzo[a]pyrene exposure.

1.2.2. Weight of Evidence for Carcinogenicity

Under EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b), benzo[a]pyrene is "carcinogenic to humans." This guidance emphasizes the importance of weighing all of the evidence in reaching conclusions about human carcinogenic potential. The descriptor of "carcinogenic to humans" can be used when the following conditions are met: (a) there is strong evidence of an association between human exposure and either cancer or the key precursor events of the agent's mode of action but not enough for a causal association; (b) there is extensive evidence of carcinogenicity in animals; (c) the mode or modes of carcinogenic action and associated key precursor events have been identified in animals; and (d) there is strong evidence that the key precursor events that precede the cancer response in animals are anticipated to occur in humans and progress to tumors, based on available biological information. The data supporting these four conditions for benzo[a]pyrene are presented below and in Table 1-20.

a) Strong Human Evidence of Cancer or its Precursors

There is a large body of evidence for human carcinogenicity for complex PAH mixtures containing benzo[a]pyrene, including soot, coal tars, coal-tar pitch, mineral oils, shale oils, and smoke from domestic coal burning (IARC, 2010; Baan et al., 2009). There is also evidence of carcinogenicity, primarily of the lung and skin, in occupations involving exposure to PAH mixtures containing benzo[a]pyrene, such as chimney sweeping, coal gasification, coal-tar distillation, coke production, iron and steel founding, aluminum production, and paving and roofing with coal tar pitch (IARC, 2010; Baan et al., 2009; Straif et al., 2005). Increased cancer risks have been reported among other occupations involving exposure to PAH mixtures such as carbon black and diesel exhaust (Benbrahim-Tallaa et al., 2012; Bosetti et al., 2007). There is extensive evidence of the carcinogenicity of tobacco smoke, of which benzo[a]pyrene is a notable constituent. The methodologically strongest epidemiology studies (in terms of exposure assessment, sample size, and follow-up period) provide consistent evidence of a strong association between benzo[a]pyrene exposure and lung cancer. Three large epidemiology studies in different geographic areas, representing two different industries, observed increasing risks of lung cancer with increasing cumulative exposure to benzo[a]pyrene (measured in $\mu g/m^3$-years), with approximately a twofold

increased risk at the higher exposures; each of these studies addressed potential confounding by smoking (Armstrong and Gibbs, 2009; Spinelli et al., 2006; Xu et al., 1996) (Table 1-13). Although the relative contributions of benzo[a]pyrene and of other PAHs cannot be established, the exposure-response patterns seen with the benzo[a]pyrene measures make it unlikely that these results represent confounding by other exposures. Similarly, for bladder cancer, two of the three cohort studies with detailed exposure data observed an increasing risk with exposures >80 μg/m^3-years(Gibbs and Sevigny, 2007a; Gibbs et al., 2007; Gibbs and Sevigny, 2007b; Spinelli et al., 2006) (Table 1-15). The exposure range was much lower in the third study (Burstyn et al., 2007; Gibbs and Sevigny, 2007a; Gibbs et al., 2007; Gibbs and Sevigny, 2007b), such that the highest exposure group only reached the level of exposure seen in the lowest exposure categories in the other studies. Data pertaining to non-melanoma skin cancer is limited to studies with more indirect exposure measures, e.g., based on occupations with likely dermal exposure to creosote (i.e., timber workers, brick makers, and power linesmen); the RR estimates seen in the four available studies that provide risk estimates for this type of cancer ranged from 1.5 to 4.6, with three of these four estimates >2.5 and statistically significant (Pukkala, 1995; Karlehagen et al., 1992; Törnqvist et al., 1986; Hammond et al., 1976). These four studies provide support for the association between dermal PAH exposure, including benzo[a]pyrene exposure, and skin cancer. Although it is likely that multiple carcinogens present in PAH mixtures contribute to the carcinogenic responses, strong evidence is available from several studies of humans exposed to PAH mixtures supporting a contributing role for benzo[a]pyrene diol epoxide in inducing key mutagenic precursor cancer events in target tissues. Elevated BPDE-DNA adducts have been reported in smokers compared to nonsmokers, and the increased adduct levels in smokers are typically increased twofold compared with nonsmokers (Phillips, 2002). Elevated BPDE-DNA adduct levels have been observed in WBCs of groups of coke oven workers and chimney sweeps, occupations with known elevated risks of cancer (Rojas et al., 2000; Bartsch et al., 1999; Pavanello et al., 1999; Bartsch et al., 1998; Rojas et al., 1998), and in lung tissue from tobacco smokers with lung cancer (Rojas et al., 2004; Godschalk et al., 2002; Bartsch et al., 1999; Rojas et al., 1998; Andreassen et al., 1996; Alexandrov et al., 1992).

Mutation spectra distinctive to diol epoxides have been observed in the tumor suppressor gene *p53* and the *K-ras* oncogene in tumor tissues taken from lung cancer patients who were chronically exposed to two significant sources of PAH mixtures: coal smoke and tobacco smoke. Hackman et al. (2000) reported an increase of GC→TA transversions and a decrease of GC→AT transitions at the hprt locus in T-lymphocytes of humans with lung cancer who were smokers compared to non-smokers. Lung tumors from cancer patients exposed to emissions from burning smoky coal showed mutations in *p53* and *K-ras* that were primarily G→T transversions (76 and 86%, respectively) (Demarini et al., 2001). Keohavong et al. (2003) investigated the *K-ras* mutational spectra from nonsmoking women and smoking men chronically exposed to emissions from burning smoky coal, and smoking men who resided in homes using natural gas; among those with *K-ras* mutations, 67, 86, and 67%, respectively, were G→T transversions. Lung tumors from

tobacco smokers showed a higher frequency of *p53* mutations that were G→T transversions compared with lung tumors in non-smokers (Pfeifer and Hainaut, 2003; Pfeifer et al., 2002; Hainaut and Pfeifer, 2001), and the frequency of these types of *p53* mutations in lung tumors from smokers increased with increasing smoking intensity (Bennett et al., 1999).

Similarly, investigations of mutagenesis following specific exposures to benzo[a]pyrene (as opposed to PAH mixtures) have consistently observed that the benzo[a]pyrene diol epoxide is very reactive with guanine bases in DNA, and that G→T transversions are the predominant type of mutations caused by benzo[a]pyrene diol epoxide in several biological test (Pfeifer and Hainaut, 2003; Hainaut and Pfeifer, 2001). Following treatment of human HeLa cells with benzo[a]pyrene diol epoxide, Denissenko et al. (1996) reported that the distribution of BPDE-DNA adducts within *p53* corresponded to mutational hotspots observed in *p53* in human lung cancers. Benzo[a]pyrene exposure induced mutations in embryonic fibroblasts from human *p53* "knock-in" mice that were similar to those found in smoking related human cancers, with a predominance of G→T transversions that displayed strand bias and were also located in the same mutational hotspots found in *p53* in human lung tumors (Liu et al., 2005). These results, combined with a mechanistic understanding that mutations in *p53* (which encodes a key transcription factor in DNA repair and regulation of cell cycle and apoptosis) may be involved in the initiation phase of many types of cancer, are consistent with a common mechanism for mutagenesis following exposures to PAH mixtures and provide evidence of a contributing role of benzo[a]pyrene diol epoxide in the carcinogenic response of humans to coal smoke and tobacco smoke.

Therefore, while the epidemiological evidence alone does not establish a causal association between human exposure and cancer, there is strong evidence that the key precursor events of benzo[a]pyrene's mode of action are likely to be associated with tumor formation in humans.

b) Extensive Animal Evidence

In laboratory animals (rats, mice, and hamsters), exposures to benzo[a]pyrene via the oral, inhalation, and dermal routes have been associated with carcinogenic responses both systemically and at the site of administration. Three 2-year oral bioassays are available that associate lifetime benzo[a]pyrene exposure with carcinogenicity at multiple sites. These bioassays observed forestomach, liver, oral cavity, jejunum, kidney, auditory canal (Zymbal gland), and skin or mammary gland tumors in male and female Wistar rats (Kroese et al., 2001); forestomach tumors in male and female Sprague-Dawley rats (Brune et al., 1981); and forestomach, esophagus, tongue, and larynx tumors in female B6C3F$_1$ mice (Beland and Culp, 1998; Culp et al., 1998). Repeated or short-term oral exposure to benzo[a]pyrene was associated with forestomach tumors in additional bioassays with several strains of mice (Weyand et al., 1995; Benjamin et al., 1988; Robinson et al., 1987; El-Bayoumy, 1985; Triolo et al., 1977; Wattenberg, 1974; Roe et al., 1970; Biancifiori et al., 1967; Chouroulinkov et al., 1967; Fedorenko and Yansheva, 1967; Neal and Rigdon, 1967; Berenblum and Haran, 1955). EPA has considered the uncertainty associated with the relevance of forestomach tumors for estimating human risk from benzo[a]pyrene exposure. While humans do not have a forestomach, squamous epithelial tissue similar to that seen in the rodent forestomach exists in the oral cavity and upper two-thirds of the esophagus in humans (IARC, 2003). Human studies, specifically associating exposure to benzo[a]pyrene with the alimentary tract tumors are not currently available. However, benzo[a]pyrene-DNA adducts have been detected in oral and esophageal tissue obtained from smokers (reviewed by Phillips, 2002) and several epidemiological studies have identified increased exposure to PAHs as an independent risk factor for esophageal cancer (Abedi-Ardekani et al., 2010; Szymańska et al., 2010; Gustavsson et al., 1998; Liu et al., 1997). Thus, EPA concluded that forestomach tumors in rodents are relevant for assessing the carcinogenic risk to humans.

Lifetime inhalation exposure to benzo[a]pyrene was associated primarily with tumors in the larynx and pharynx of male Syrian golden hamsters exposed to benzo[a]pyrene:NaCl aerosols (Thyssen et al., 1981). Additionally, less-than-lifetime oral exposure cancer bioassays in mice provide supporting evidence that exposure to benzo[a]pyrene is associated with an increased incidence of lung tumors in mice (Weyand et al., 1995; Robinson et al., 1987; Wattenberg, 1974). In additional studies with hamsters, intratracheal instillation of benzo[a]pyrene was associated with upper and lower respiratory tract tumors (Feron and Kruysse, 1978; Ketkar et al., 1978; Feron et al., 1973; Henry et al., 1973; Saffiotti et al., 1972). Dermal application of benzo[a]pyrene (2–3 times/week) has been associated with mouse skin tumors in numerous lifetime bioassays (Sivak et al., 1997; Grimmer et al., 1984; Habs et al., 1984; Grimmer et al., 1983; Habs et al., 1980; Schmähl et al., 1977; Schmidt et al., 1973; Roe et al., 1970; Poel, 1963, 1959). Skin tumors in rats, rabbits, and guinea pigs have also been associated with repeated application of benzo[a]pyrene to skin in the absence of exogenous promoters (IPCS, 1998; ATSDR, 1995; IARC, 1983, 1973). When followed by repeated exposure to a potent tumor promoter, acute dermal exposure to

benzo[a]pyrene induced skin tumors in numerous studies of mice, indicating that benzo[a]pyrene is a strong tumor-initiating agent in the mouse skin model (Weyand et al., 1992; Cavalieri et al., 1991; Rice et al., 1985; El-Bayoumy et al., 1982; LaVoie et al., 1982; Raveh et al., 1982; Cavalieri et al., 1981; Slaga et al., 1980; Wood et al., 1980; Slaga et al., 1978; Hoffmann et al., 1972).

Carcinogenic responses in animals exposed to benzo[a]pyrene by other routes of administration include: (1) liver or lung tumors in newborn mice given acute postnatal i.p. injections (LaVoie et al., 1994; Busby et al., 1989; Weyand and Lavoie, 1988; LaVoie et al., 1987a; Wislocki et al., 1986; Busby et al., 1984; Buening et al., 1978; Kapitulnik et al., 1978); (2) increased lung tumor multiplicity in A/J adult mice given single i.p. injections (Mass et al., 1993); (3) injection site tumors in mice following s.c. injection (Nikonova, 1977; Pfeiffer, 1977; Homburger et al., 1972; Roe and Waters, 1967; Grant and Roe, 1963; Steiner, 1955; Rask-Nielsen, 1950; Pfeiffer and Allen, 1948; Bryan and Shimkin, 1943; Barry et al., 1935); (4) injection site sarcomas in mice following intramuscular injection(Sugiyama, 1973); (5) mammary tumors in rats with intramammilary administration (Cavalieri et al., 1991; Cavalieri et al., 1988c; Cavalieri et al., 1988b; Cavalieri et al., 1988a); (6) cervical tumors in mice with intravaginal application (Näslund et al., 1987); and (7) tracheal tumors in rats with intratracheal implantation (Topping et al., 1981; Nettesheim et al., 1977).

Therefore, the animal database provides extensive evidence of carcinogenicity in animals.

c) Key Precursor Events have been Identified in Animals

There is sufficient evidence to conclude that benzo[a]pyrene carcinogenicity involves a mutagenic mode of action mediated by DNA-reactive metabolites. The benzo[a]pyrene database provides strong and consistent evidence for BPDE-induced mutations associated with tumor initiation in cancer tissue from humans exposed to complex mixtures containing benzo[a]pyrene, in animals exposed to benzo[a]pyrene, and in in vitro systems. Other processes may contribute to the carcinogenicity of benzo[a]pyrene via the promotion and progression phases of cancer development (e.g., inflammation, cytotoxicity, sustained regenerative cell proliferation, anti-apoptotic signaling), but the available evidence best supports a mutagenic mode of action as the primary mode by which benzo[a]pyrene acts.

d) Strong Evidence that the Key Precursor Events are Anticipated to Occur in Humans

Mutations in *p53* and *ras* oncogenes have been observed in tumors from mice exposed to benzo[a]pyrene in the diet (Culp et al., 2000) or by i.p. injection (Nesnow et al., 1998a; Nesnow et al., 1998b; Nesnow et al., 1996, 1995; Mass et al., 1993). Mutations in these same genes have also been reported in lung tumors of human cancer patients, bearing distinctive mutation spectra (G→T transversions) that correlate with exposures to coal smoke (Keohavong et al., 2003; Demarini et al., 2001) or tobacco smoke (Pfeifer and Hainaut, 2003; Pfeifer et al., 2002; Hainaut and Pfeifer, 2001; Bennett et al., 1999).

Table 1-20. Supporting evidence for the *carcinogenic to humans* cancer descriptor for benzo[a]pyrene

Evidence	Reference
a) Strong human evidence of cancer or its precursors	
• Increased risk of lung, bladder, and skin cancer in humans exposed to complex PAH mixtures containing benzo[a]pyrene	IARC (2004); IARC (2010); Secretan et al. (2009);Baan et al. (2009); Benbrahim-Tallaa et al. (2012)
• Benzo[a]pyrene-specific biomarkers detected in humans exposed to PAH mixtures associated with increased risk of cancer	
– BPDE-DNA adducts in WBCs of coke oven workers and chimney sweeps	Bartsch et al. (1998); Bartsch et al. (1999); Pavanello et al. (1999); Rojas et al. (1998); Rojas et al. (2000)
– BPDE-DNA adducts in smokers	Phillips (2002)
• Benzo[a]pyrene-specific DNA adducts have been detected in target tissues in humans exposed to PAH mixtures	
– BPDE-DNA adducts in non-tumor lung tissues of cigarette smokers with lung cancer and in skin of eczema patients treated with coal tar	Rojas et al. (2004); Alexandrov et al. (1992); Bartsch et al. (1999); Godschalk et al. (2002); Rojas et al. (1998); Andreassen et al. (1996); Godschalk et al. (1998)
– BPDE-DNA adduct formation in *p53* in human cells in vitro corresponds to mutational hotspots at guanine residues in human lung tumors	Denissenko et al. (1996); Puisieux et al. (1991)
• Benzo[a]pyrene-specific mutational spectra identified in PAH-associated tumors in humans	
– GC→TA transversions and GC→AT transitions at hprt locus in T-lymphocytes of humans with lung cancer	Hackman et al. (2000)
– G→T transversions in exposed human-*p53* knock-in mouse fibroblasts at the same mutational hotspot in *p53* from smoking-related lung tumors in humans	Liu et al. (2005)

Evidence	Reference
– G→T transversions at the same mutational hotspot in *p53* and *K-ras* in human lung tumors associated with smoky coal exposures	Demarini et al. (2001); Keohavong et al. (2003)
– Increased percentage of G→T transversions in *p53* in smokers versus nonsmokers	Hainaut and Pfeifer (2001); Pfeifer et al. (2002); Pfeifer and Hainaut (2003); Bennett et al. (1999)
b) Extensive animal evidence	
Oral exposures	
• Forestomach tumors in male and female rats and in female mice following lifetime exposure	Kroese et al. (2001); Brune et al. (1981); Beland and Culp (1998); Culp et al. (1998)
• Forestomach tumors in mice following less-than-lifetime exposures	Benjamin et al. (1988); Berenblum and Haran (1955); Biancifiori et al. (1967); Chouroulinkov et al. (1967); El-Bayoumy (1985); Fedorenko and Yansheva (1967); Neal and Rigdon (1967); Robinson et al. (1987); Roe et al. (1970); Triolo et al. (1977); Wattenberg (1974); Weyand et al. (1995)
• Alimentary tract and liver tumors in male and female rats following lifetime exposure	Kroese et al. (2001)
• Kidney tumors in male rats following lifetime exposure	Kroese et al. (2001)
• Auditory canal tumors in male and female rats following lifetime exposure	Kroese et al. (2001)
• Esophageal, tongue, and laryngeal tumors in female mice following lifetime exposure	Beland and Culp (1998); Culp et al. (1998)
• Lung tumors in mice following less-than-lifetime exposure	Robinson et al. (1987); Wattenberg (1974); Weyand et al. (1995)
Inhalation exposures	
• Upper respiratory tract tumors in male hamsters following chronic exposure	Thyssen et al. (1981)

Evidence	Reference
Dermal exposures	
• Skin tumors in mice following lifetime exposures without a promoter	Grimmer et al. (1984); Grimmer et al. (1983); Habs et al. (1984); Habs et al. (1980); Poel (1959); Poel (1963); Roe et al. (1970); Schmähl et al. (1977); Schmidt et al. (1973); Sivak et al. (1997)
• Skin tumors in rats, rabbits, and guinea pigs following subchronic exposures	IPCS (1998); ATSDR (1995); IARC (1973); IARC (1983)
Other routes of exposure	
• Respiratory tract tumors in hamsters following intratracheal instillation	Feron et al. (1973); Feron and Kruysse (1978); Henry et al. (1973); Ketkar et al. (1978); Saffiotti et al. (1972)
• Liver or lung tumors in newborn mice given acute postnatal i.p. injections	Buening et al. (1978); Busby et al. (1984); Busby et al. (1989); Kapitulnik et al. (1978); LaVoie et al. (1987a); LaVoie et al. (1994); Weyand and Lavoie (1988); Wislocki et al. (1986)
• Lung tumor multiplicity in A/J adult mice given single i.p. injections	Mass et al. (1993)
c) Identification of key precursor events have been identified in animals	
• Bioactivation of benzo[a]pyrene to DNA-reactive metabolites has been shown to occur in multiple species and tissues by all routes of exposure • Direct DNA damage by the reactive metabolites, including the formation of DNA adducts and ROS-mediated damage • Formation and fixation of DNA mutations, particularly in tumor suppressor genes or oncogenes associated with tumor initiation	See 'Experimental Support for Hypothesized Mode of Action' section
d) Strong evidence that the key precursor events are anticipated to occur in humans	
• Mutations in *p53* or *ras* oncogenes have been observed in forestomach or lung tumors from mice exposed to benzo[a]pyrene	Culp et al. (2000); Mass et al. (1993); Nesnow et al. (1998a); Nesnow et al. (1998b); Nesnow et al. (1995); Nesnow et al. (1996)
– G→T transversions in *ras* oncogenes or the *p53* gene have been observed in lung tumors of human cancer patients exposed to coal smoke	Demarini et al. (2001); Keohavong et al. (2003)

Evidence	Reference
– Higher frequency of G→T transversions in lung tumors from smokers versus nonsmokers	Bennett et al. (1999); Hainaut and Pfeifer (2001); Pfeifer et al. (2002); Pfeifer and Hainaut (2003)

1

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 and Effects for Dose-Response Analysis

In Section 1.2.1, developmental, reproductive, and immunological toxicities were highlighted as human hazards or potential human hazards of benzo[a]pyrene exposure by the oral route. Studies within each effect category were evaluated using general study quality characteristics (as discussed in Section 6 of the Preamble) to help inform the selection of studies from which to derive toxicity values. Rationales for selecting the studies and effects to represent each of these hazards are summarized below.

Human studies are preferred over animal studies when quantitative measures of exposure are reported and the reported effects are determined to be associated with exposure. For benzo[a]pyrene, human studies of environmental polycyclic aromatic hydrocarbon (PAH) mixtures across multiple cohorts have observed effects following exposure to complex mixtures of PAHs. The available data suggest that benzo[a]pyrene exposure may pose health hazards other than cancer including reproductive and developmental effects such as infertility, miscarriage, and reduced birth weight (Wu et al., 2010; Neal et al., 2008; Tang et al., 2008; Perera et al., 2005b; Perera et al., 2005a), effects on the developing nervous system (Perera et al., 2012a; Perera et al., 2009), and cardiovascular effects (Friesen et al., 2010; Burstyn et al., 2005). However, the available human studies that utilized benzo[a]pyrene-deoxyribonucleic acid (DNA) adducts as the exposure metric do not provide external exposure levels of benzo[a]pyrene from which to derive a value, and exposure is likely to have occurred by multiple routes. In addition, uncertainty exists due to concurrent exposure to other PAHs and other components of the mixture (such as metals).

Animal studies were evaluated to determine which provided the most relevant routes and durations of exposure; multiple exposure levels to provide information about the shape of the dose-response curve; and power to detect effects at low exposure levels (U.S. EPA, 2002). The oral database for benzo[a]pyrene includes a variety of studies and datasets that are suitable for use in

deriving reference values. Specifically, chronic effects associated with benzo[a]pyrene exposure in animals include observations of organ weight and histological changes and hematological parameters observed in several oral cancer bioassays (Kroese et al., 2001; Beland and Culp, 1998). Multiple subchronic studies are available that characterize a variety of effects other than cancer. In addition, several developmental studies are available that help inform hazards of exposure during sensitive developmental windows.

Developmental Toxicity

Numerous animal studies observed endpoints of developmental toxicity (including developmental neurotoxicity) following oral exposure during gestational or early postnatal development (Chen et al., 2012; Jules et al., 2012; Bouayed et al., 2009a; Kristensen et al., 1995; Mackenzie and Angevine, 1981) and were evaluated for dose-response analysis based on the above considerations. As summarized in Section 1.1.1, two studies demonstrated decreased fertility among gestationally exposed females (Kristensen et al., 1995; Mackenzie and Angevine, 1981). Kristensen et al. (1995), with only one dose group, was not considered further given its concordance with Mackenzie and Angevine (1981), which had multiple groups. Mackenzie and Angevine (1981) demonstrated developmental effects in a multi-dose study with relevant routes and durations of exposure; however, the doses studied (10−160 mg/kg-day) were much higher than doses evaluated in the other available developmental toxicity studies (Chen et al., 2012; Jules et al., 2012) and was not considered further for RfD derivation. Similarly, the developmental study by Bouayed et al. (2009a) used the same tests as Chen et al. (2012), but the doses evaluated were higher (2 and 20 mg/kg-day compared to 0.02, 0.2, and 2 mg/kg-day, respectively). From the available studies demonstrating developmental toxicity, the studies conducted by Chen et al. (2012) and Jules et al. (2012) were identified as the most informative studies for dose-response analysis to characterize effects in the low-dose region.

Jules et al. (2012) reported increases in both systolic (approximately 20−50%) and diastolic (approximately 33−83%) pressure in adult rats that were exposed gestationally to benzo[a]pyrene. Given the magnitude of the response and the appearance of these effects in adulthood following gestational exposure, these endpoints were selected for dose-response analysis, however, some uncertainty exists in the absence of other studies evaluating these outcomes. The neurodevelopmental study by Chen et al. (2012) was a well-designed and well-conducted study that evaluated multiple developmental endpoints and measures of neurotoxicity in neonatal, adolescent, and adult rats after early postnatal exposure (see Section 1.1.1 for more detail). Chen et al. (2012) observed increased locomotion in the open field test, increased latency in negative geotaxis and surface righting tests, decreased anxiety-like behaviors in the elevated plus maze test, and impaired performance in the Morris water maze test at various time points following neonatal benzo[a]pyrene treatment. While an understanding of the underlying biological perturbation(s) causing altered performance in these tests remains incomplete (see discussion in Section 1.1.1), all of the results are interpreted to indicate an effect of benzo[a]pyrene treatment during development

on neurobehavioral function. Since any behavioral effect observed in animals is assumed to pose a potential hazard to humans, and because behavioral effects seen in animal studies may not always be the same as those produced in humans (U.S. EPA, 1998a), all of the observed behavioral changes are interpreted to be relevant for estimating potential neurotoxic risks in humans.

Of the neurobehavioral endpoints observed in Chen et al. (2012), three behavioral tests, the open field activity test, elevated plus maze, and the Morris water maze represent the most compelling evidence of benzo[a]pyrene exposure-induced developmental neurotoxicity (see discussion in Section 1.1.1) and thus were identified as the most informative for neurobehavioral endpoints for dose-response analysis.

Reproductive Toxicity

Male reproductive toxicity was demonstrated in a number of subchronic studies (Jeng et al., 2013; Chen et al., 2011; Chung et al., 2011; Mohamed et al., 2010; Zheng et al., 2010). Chung et al. (2011) as not included in the dose-response analysis because numerical data were not reported or were only reported for the mid-dose of three doses. Chen et al. (2011), a subchronic study that corroborated other available multi-dose studies, is considered supportive, but was not considered for RfD derivation due to the use of a single dose level and limited reporting of numerical data. Jeng et al. (2013) observed effects (decreased testicular weight, decreases in sperm parameters) similar to other studies in the database but at much higher doses. The studies conducted by Mohamed et al. (2010) and Zheng et al. (2010) were identified as the most informative male reproductive toxicity studies for dose-response analysis. Decreased sperm count and motility observed by Mohamed et al. (2010) and decreased intratesticular testosterone levels observed by Zheng et al. (2010) were selected for dose-response analysis as both represent sensitive endpoints of male reproductive toxicity and are indicators of potentially decreased fertility. These effects are also consistent with human studies in PAH exposed populations, as effects on male fertility and semen quality have been demonstrated in epidemiological studies of smokers (reviewed by Soares and Melo, 2008).

Female reproductive toxicity was demonstrated in two subchronic oral studies (Gao et al., 2011; Xu et al., 2010). Specifically, Xu et al. (2010) demonstrated decreased ovary weight and follicle number, and Gao et al. (2011) reported an increase in inflammatory cells and hyperplasia in the cervix following oral exposure to benzo[a]pyrene. These studies were identified as the most informative studies on female reproductive toxicity for dose-response analysis.

Gao et al. (2011) identified statistically-significant, dose-related increases in the incidence of cervical inflammatory cells and cervical hyperplasia in mice exposed to low doses of benzo[a]pyrene for 98 days. Cervical effects of increasing severity (including apoptosis, and necrosis) were also observed at higher doses (Gao et al., 2011). This study also observed a depression of bodyweight (10, 15, and 30%) and elevated mortality in the two higher dose groups (4 and 8%), suggesting potential treatment related toxicity. Effects in the cervix have not been reported in other noncancer or cancer bioassays in the database; it is unclear whether the observed

cervical inflammation and hyperplasia are linked to impaired reproductive function. Although there is some uncertainty in this endpoint, the histological lesion of cervical hyperplasia was the most sensitive endpoint observed and was considered further for dose-response.

Xu et al. (2010) identified biologically and statistically significant decreases in ovary weight and primordial follicles in treated animals. These reductions in female reproductive parameters are supported by a large database of animal studies (including mechanistic studies as well as studies using other routes of exposure or shorter durations)) altogether indicating that benzo[a]pyrene is ovotoxic with effects including decreased ovary weight, decreased primordial follicles, and reduced fertility (Borman et al., 2000; Kristensen et al., 1995; Miller et al., 1992; Swartz and Mattison, 1985; Mackenzie and Angevine, 1981; Mattison et al., 1980). Additionally, epidemiology studies indicate that exposure to complex mixtures of PAHs, such as through cigarette smoke, is associated with measures of decreased fertility in humans (Neal et al., 2008; El-Nemr et al., 1998). Specific associations have also been made between infertility and increased levels of benzo[a]pyrene in follicular fluid in women undergoing in vitro fertilization (Neal et al., 2008).

Immunotoxicity

As described in Section 1.1.3, the immune system was identified as a potential human hazard of benzo[a]pyrene exposure based on findings of decreased thymus weight and immunoglobulin alterations, as well as effects on cellularity and functional changes in the immune system in animals, and supporting data from mechanistic studies and short term assays. The only available oral repeat exposure studies to support development of an immune RfD were conducted by Kroese et al. (2001) and De Jong et al. (1999). These subchronic studies used multiple exposure levels and typical sample sizes (10 or 8 rats/group, respectively). For the endpoint of decreased thymus weight (observed in both studies), , the Kroese et al. (2001) study is preferred due to its longer duration (90 days compared to 35 days with De Jong et al., 1999).

Functional evaluations of immune system response, thought to be more sensitive than observational immune endpoints (WHO, 2012), were not evaluated in the oral, repeat exposure studies available for benzo[a]pyrene. Therefore, the observational immune endpoints of decreased thymus weight, in Kroese et al. (2001), decreased IgM and IgA levels, and decreased relative numbers of B cells, in De Jong et al. (1999), were selected. It is recognized that thymus weight changes on their own have been noted to be less reliable indicators of immunotoxicity (Luster et al., 1992). However, there are converging lines of evidence that support the use of this endpoint, among others as representative of benzo[a]pyrene immunotoxicity. Alterations in immunoglobulin levels have been noted in humans after exposure to PAHs, as well as in animal studies after exposure to benzo[a]pyrene. Changes in B cell populations in the spleen provide additional evidence of immunotoxicity. Finally, functional effects on the immune system, including dose-related decreases in SRBC-specific IgM levels and dose-dependent decreases in resistance to pneumonia or Herpes simplex type 2 following short-term subcutaneous (s.c.) injection have been reported (Temple et al., 1993; Munson et al., 1985). The observed decreases in thymus weight, IgM

and IgA levels, and number of B cells associated with exposure to benzo[a]pyrene were concluded to be representative of immunotoxicity following benzo[a]pyrene exposure and were selected for dose-response analysis.

2.1.2. Methods of Analysis

Among the endpoints representing the hazards of benzo[a]pyrene exposure, the data for neurobehavioral changes (i.e., changes in open field activity, elevated plus maze activity, and Morris water maze performance) (Chen et al., 2012), decreased ovary weight and follicle count (Xu et al., 2010), increased cervical hyperplasia Gao et al. (2011), and decreased thymus weight (Kroese et al., 2001) were judged to support dose-response modeling.

For the neurobehavioral tests (Chen et al., 2012), identified as the most informative for dose-response (i.e., open field activity, elevated plus maze, and Morris water maze), multiple datasets were available (e.g., results for different lifestages, both sexes, and related metrics from the same assay). Data for both sexes were modeled. In addition, for these tests, effects observed in juveniles persisted and were more pronounced in adults, thus, the response measures in adulthood were preferred. For the open field activity test, effects on horizontal locomotion in the open field were more sensitive than rearing measures, thus, these data were preferred. In the elevated plus maze, several related measures were reported including increased number of open arm entries, increased time spent in the open arms, and decreased closed arm entries; ultimately, the number of entries into the open arms was used for dose-response analysis (for further discussion see Appendix E.1.1). For Morris water maze, escape latency in the hidden platform trials was the preferred measure of impaired performance as differences in probe trial performance were not considered reliable since the treated animals never reached the same baseline level of proficiency prior to testing.

As no biologically based dose-response models are available for benzo[a]pyrene, EPA evaluated a range of dose-response models thought to be consistent with underlying biological processes to determine how best to empirically model each dose-response relationship in the range of observed data. In general, the models in EPA's Benchmark Dose Software (BMDS; U.S. EPA, 2012b) relevant to each data type (e.g., continuous or dichotomous) were applied. Additional models were also considered when feasible. For example, the study design for escape latency in the Morris water maze test Chen et al. (2012) involved repeated evaluations for individual animals across several testing days (PNDs 71–74), and could conceivably support repeated measures models that would address intraindividual variation; however, the necessary individual animal data were not available. Consequently, continuous models in BMDS were applied to the reported results from Chen et al. (2012) for each day of testing (PNDs 71–74) and are included in Table 2-1.

Consistent with EPA's *Benchmark Dose Technical Guidance Document* (U.S. EPA, 2012b), the benchmark dose (BMD) and the 95% lower confidence limit on the BMD (BMDL) were estimated using a benchmark response (BMR) of 1 standard deviation (SD) from the control mean for continuous data or a BMR of 10% extra risk for dichotomous data, in the absence of information

regarding what level of change is considered biologically important, and in order to facilitate a consistent basis of comparison across endpoints, studies, and assessments. For outcomes with information available regarding what level of change is considered biologically important, these considerations informed BMR selection, as summarized below.

For reduced follicle counts, a BMR of 10% relative deviation from control levels was judged to represent a minimally important degree of change, by the following reasoning. There is no consensus in the scientific community regarding what degree of change in follicle number is biologically significant; in the absence of this data it has been suggested that a detectable decrease in follicle number be considered adverse (Heindel, 1998). Power analyses by Heindel (1998) provides a basis for inferring a critical level. More specifically, the power analyses focused on identifying follicle counts reduced by 20% or more. Therefore, a 10% change in follicle count was defined as a minimally important degree of change.

For escape latency, a 1 SD BMR based on the observed variability was used, taking into account the repeated measures experimental design, following an evaluation of the reported standard errors across trial days and dose groups. Although there was a slight tendency of the reported mean escape latencies in the control and mid-dose groups to decrease over the four successive days, the low- and high-dose groups showed no particular pattern (see Table E-1); thus all variability estimates were taken to be equally representative. An overall SD of 9 seconds resulted both from an average of the control group standard deviations across trial days and from a grand average of all of the standard deviations across trial days and dose groups.

Further details including the modeling output and graphical results for the selected model for each endpoint can be found in Appendix E of the Supplemental Information.

For outcomes with sufficient data for modeling and that were successfully modeled, the BMDLs were used to define points of departure (PODs). For the overall POD for neurobehavioral effects, the BMDs of several behavioral tests (from the same study) were closely clustered. These neurobehavioral endpoints represent behavioral changes in the same group of rats, following early postnatal exposure to benzo[a]pyrene, which persisted into adulthood. Thus, the BMDLs representing different behavioral manifestations of neurotoxicity were considered together to define the POD for neurobehavioral changes.

Other endpoints identified in Section 2.1.1 had insufficient data to support dose-response modeling, but allowed identification of a NOAEL or LOAEL for use as PODs. Specifically, the data for epididymal sperm counts in the Mohamed et al. (2010) study were reported only as percentages of control responses, without actual control values; the variability in blood pressure measurements (i.e., SE) in Jules et al. (2012) were reported inconsistently; and the observed decreases in IgM and IgA (De Jong et al., 1999), although consistently depressed with exposure, showed highly heterogeneous variability. These datasets were not amenable to dose-response modeling, thus, a NOAEL or LOAEL was used as the POD.

Human equivalent doses (HEDs) for oral exposures were derived from the PODs estimated from the laboratory animal data as described in EPA's *Recommended Use of Body Weight$^{3/4}$ as the Default Method in Derivation of the Oral Reference Dose* (U.S. EPA, 2011). In this guidance, EPA advocates a hierarchy of approaches for deriving HEDs from data in laboratory animals, with the preferred approach being physiologically-based pharmacokinetic (PBPK) modeling. Other approaches can include using chemical-specific information in the absence of a complete pharmacokinetic (i.e., toxicokinetic) model. As discussed in Appendix D of the Supplemental Information, several animal PBPK models for benzo[a]pyrene have been developed and published, but a validated human PBPK model for benzo[a]pyrene for extrapolating doses from animals to humans is not available. In lieu of either 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 applied to extrapolate toxicologically equivalent doses of orally administered agents from adult laboratory animals to adult humans for the purpose of deriving an oral RfD.

Consistent with EPA guidance (U.S. EPA, 2011), the points of departure (PODs) estimated based on effects in adult animals are converted to HEDs employing a standard dosimetric adjustment factor (DAF) derived as follows:

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

where

BW_a = animal body weight; and

BW_h = human body weight.

Using BW_a of 0.25 kg for rats and 0.035 kg for mice and 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 this DAF to the POD identified for effects in adult rats or mice yields a POD_{HED} as follows (see Table 2-1):

$$POD_{HED} = POD_{ADJ} \times DAF.$$

BW$^{3/4}$ scaling was not employed for deriving HEDs from the developmental toxicity study by Chen et al. (2012) in which doses were administered directly to early postnatal animals because of several areas of uncertainty. The first issue was whether allometric (i.e., body weight^3/4) scaling, derived from data in adult animals, holds when extrapolating doses in neonatal animals. This uncertainty arises because of the absence of quantitative information to characterize the toxicokinetic and toxicodynamic differences between animals and humans in early lifestages (U.S. EPA, 2011). In addition, interspecies extrapolation across early life stages is also complicated by differences in temporal patterns of development across species. U.S. EPA (2011) states that when such an extrapolation is considered, key developmental processes need to be matched in a species-dependent manner, because the temporal pattern of development differs across species. In the

study at issue, Chen et al. (2012), neurobehavioral changes were observed in adult rats after dosing on PNDs 5–11. This postnatal period of brain development in rats is believed to be more akin to human brain development occurring in the third trimester of pregnancy (Dobbing and Sands, 1979, 1973), thus challenging the suitability of extrapolating from rats directly exposed on PNDs 5–11 to third trimester humans with transplacental exposure.

Table 2-1 summarizes the sequence of calculations leading to the derivation of a human-equivalent POD for each data set discussed above.

Table 2-1. Summary of derivation of PODs

Endpoint and reference	Species/ sex	Model[a]	BMR	BMD mg/kg-d	BMDL mg/kg-d	POD mg/kg-d	POD_{ADJ}[b] mg/kg-d	POD_{HED}[c] mg/kg-d
Developmental								
Neurobehavioral changes: Open field crossed squares at PND 69 Chen et al. (2012)	Male and Female Sprague-Dawley rats	Exponential 4	1 SD	0.23	0.11	0.092[d]	0.092	0.092
Neurobehavioral changes: Elevated plus maze open arm entries at PND 70 Chen et al. (2012)	Female Sprague-Dawley rats	Exponential 4	1 SD	0.21	0.092			
Neurobehavioral changes: Morris water maze hidden platform trial escape latency at PNDs 71–74 Chen et al. (2012)	Male and Female Sprague-Dawley rats	Hill CV Hill CV Hill CV Hill NCV	1 SD (9 sec)	PND71: 0.49 PND72: 0.33 PND73: 0.27 PND74: 0.23	0.16 0.16 0.12 0.13			
Cardiovascular effects at PND 53 Jules et al. (2012)	Long-Evans rats	LOAEL (0.6 mg/kg-d) (15% ↑ in systolic blood pressure; 33% ↑ in diastolic blood pressure)				0.6	0.6	0.15
Reproductive								
Decreased ovary weight Xu et al. (2010)	Female Sprague-Dawley rats	Linear	1 SD	2.3	1.5	1.5	1.5	0.37
Decreased ovarian follicle count Xu et al. (2010)	Female Sprague-Dawley rats	Linear	10% RD	2.3	1.6	1.6	1.6	0.38
Decreased intratesticular testosterone	Male Sprague-	NOAEL (1 mg/kg-d) (15% ↓ in testosterone)				1	1	0.24

Endpoint and reference	Species/ sex	Model[a]	BMR	BMD mg/kg-d	BMDL mg/kg-d	POD mg/kg-d	POD_{ADJ}[b] mg/kg-d	POD_{HED}[c] mg/kg-d
Zheng et al. (2010)	Dawley rats							
Decreased sperm count and motility Mohamed et al. (2010)	Male C57BL/6 mice	LOAEL (1 mg/kg-d) (50% ↓ in sperm count; 20% ↓ in sperm motility)				1	1	0.15
Cervical epithelial hyperplasia Gao et al. (2011)	Female ICR mice	Log-logistic	10% ER	0.58	0.37	0.37	0.37	0.06
Immunological								
Decreased thymus weight Kroese et al. (2001)	Female Wistar rats	Linear	1SD	10.5	7.6	7.6	7.6	1.9
Decreased thymus weight Kroese et al. (2001)	Male Wistar rats	Linear	1SD	16.4	11.3	11.3	11.3	2.7
Decreased IgM levels De Jong et al. (1999)	Male Wistar rats	NOAEL (10 mg/kg-d) (14% ↓ in IgM)				10	7.1	1.7
Decreased IgA levels De Jong et al. (1999)	Male Wistar rats	NOAEL (30 mg/kg-d) (28% ↓ in IgA)				30	21	5.2
Decreased number of B cells De Jong et al. (1999)	Male Wistar rats	NOAEL (30 mg/kg-d) (7% ↑ in B cells at NOAEL; 31% ↓ at LOAEL)				30	21	5.2

[a]For modeling details, see Appendix E.1 in Supplemental Information.
[b]For studies in which animals were not dosed daily, PODs were adjusted to calculate the TWA daily doses (following BMD modeling), with the exception of Xu et al. (2010) and Gao et al. (2011), for which the TWA daily doses were used for BMD modeling.
[c]HED PODs were calculated using $BW^{3/4}$ scaling (U.S. EPA, 2011) for effects from dosing studies in adult animals (i.e., Gao et al., 2011; Mohamed et al., 2010; Xu et al., 2010; De Jong et al., 1999) or for developmental effects resulting from in utero exposures (Jules et al., 2012). $BW^{3/4}$ scaling was not employed for deriving HEDs from studies in which doses were administered directly to early postnatal animals (i.e., Chen et al., 2012).
[d]The POD for neurobehavioral changes based on the lower end of the BMDLs (i.e., 0.092–0.16 mg/kg-day) for three sensitive behavioral measures.

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), also described in the Preamble, five possible areas of uncertainty and variability were considered. An explanation follows:

A UF for extrapolation from a LOAEL to NOAEL, UF_L, of 1 was applied when the POD was based on a NOAEL (Zheng et al., 2010; De Jong et al., 1999). A value of 1 was applied when a BMR of

a 1 SD (Chen et al., 2012; Kroese et al., 2001) or 10% change (Gao et al., 2011) from the control was selected under an assumption that it represents a minimal biologically significant response level. A NOAEL was not determined for the most sensitive effects observed in Jules et al. (2012) and Mohamed et al. (2010). At the LOAEL, Jules et al. (2012) observed statistically significant increases in systolic (15%) and diastolic (33%) blood pressure when measured in adulthood following gestational exposure. Regarding the study by Mohamed et al. (2010), the authors observed a statistically significant decrease in sperm count (50%) and motility (20%) in treated F0 males at the lowest dose tested. The data in this study was not reported sufficiently to enable dose-response modeling, since the authors did not report a measure of the variability (SD or SEM) for the control group. Therefore, a UF of 10 was applied to approximate a NOAEL for studies (Jules et al., 2012; Mohamed et al., 2010) which observed a high magnitude of response at the LOAEL.

A subchronic to chronic uncertainty factor, UF_S, of 10 was applied when the POD was based on a subchronic study (studies in Table 2-2, other than the two developmental toxicity studies, were 42–90 days in duration) to account for the possibility that longer exposure may induce effects at a lower dose. A 1 was applied when dosing occurred during gestation (Jules et al., 2012) or the early postnatal period (Chen et al., 2012). A UF_S of 1 was applied for PODs from developmental studies. The developmental period is recognized as a susceptible lifestage when exposure during a time window of development is more relevant to the induction of developmental effects than lifetime exposure (U.S. EPA, 1991c); therefore, an adjustment for duration is not warranted.

An interspecies uncertainty factor, UF_A, of 3 ($10^{1/2}$ = 3.16, rounded to 3) was applied, to all PODs in Table 2-2 except Chen et al. (2012), because $BW^{3/4}$ scaling is being used to extrapolate oral doses from laboratory animals to humans. Although $BW^{3/4}$ scaling addresses some aspects of cross-species extrapolation of toxicokinetic and toxicodynamic processes, some residual uncertainty 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 a UF of 3. $BW^{3/4}$ scaling was not employed for deriving HEDs from studies in which doses were administered directly to early postnatal animals (i.e. Chen et al., 2012), therefore a value of 10 was applied to account for the absence of quantitative information to characterize either the toxicokinetic or toxicodynamic differences between animals and humans at this lifestage.

An intraspecies uncertainty factor, UF_H, of 10 was applied to account for variability and uncertainty in toxicokinetic and toxicodynamic susceptibility within the subgroup of the human population most sensitive to the health hazards of benzo[a]pyrene (U.S. EPA, 2002). In the case of benzo[a]pyrene, the PODs were derived from studies in inbred animal strains and are not considered sufficiently representative of the exposure and dose-response of the susceptible human subpopulations (in this case, the developing fetus). In certain cases, the toxicokinetic component of this factor may be replaced when a PBPK model is available that incorporates the best available information on variability in toxicokinetic disposition in the human population (including sensitive subgroups). In the case of benzo[a]pyrene, insufficient information is available to quantitatively

estimate variability in human susceptibility; therefore, the full value for the intraspecies UF was retained.

A database uncertainty factor, UF_D, of 3 was applied to account for database deficiencies, including the lack of a standard multigenerational study or extended 1-generation study that includes exposure from premating through lactation, useful for understanding the potential for benzo[a]pyrene exposure to cause reproductive and neurodevelopmental effects. Benzo[a]pyrene has been shown to affect fertility in adult male and female animals by multiple routes of exposure and decreased fertility in adult male and female mice is observed both following premating exposure and following gestational exposure (see Section 1.1.1. and 1.1.2). Therefore, it is plausible that exposure occurring over a more comprehensive period of development or over multiple generations could result in a more sensitive POD than the POD selected for developmental neurotoxicity.

Some additional uncertainties exist in the benzo[a]pyrene database, including the paucity of sensitive studies evaluating endpoints of immune and cardiovascular toxicity. The lack of developmental immunotoxicity studies, especially those examining functional endpoints, is an uncertainty in the benzo[a]pyrene database. Some consideration was given to cardiovascular effects through the candidate value derived for developmental effects on the cardiovascular system (Jules et al., 2012).

The POD for the overall RfD was based on several sensitive neurobehavioral endpoints observed following treatment during a sensitive period of brain development and were among the lowest effect levels observed in the benzo[a]pyrene database, even among other developmental studies utilizing low doses of benzo[a]pyrene; thus, application of a full database UF of 10 was not judged to be warranted. However, because studies following a more comprehensive period of developmental exposure (i.e., early gestation through lactation, if not through adolescence) were not available, a database UF of 3 was applied to help address residual uncertainty associated with the potential for effects at lower doses.

Table 2-2 is a continuation of Table 2-1 and summarizes the application of UFs to each POD to derive a candidate value for each data set. The candidate values presented in the table below 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 benzo[a]pyrene.

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

Endpoint and reference	POD_{HED} (mg/kg-d)	POD type	UF_L	UF_S	UF_A	UF_H	UF_D	Composite UF	Candidate value (mg/kg-d)
Developmental									

Endpoint and reference	POD_{HED} (mg/kg-d)	POD type	UF_L	UF_S	UF_A	UF_H	UF_D	Composite UF	Candidate value (mg/kg-d)
Neurobehavioral changes in rats Chen et al. (2012)	0.092	$BMDL_{1SD}$	1	1	10	10	3	300	3×10^{-4}
Cardiovascular effects in rats Jules et al. (2012)	0.15	LOAEL	10	1	3	10	3	1,000	2×10^{-4}
Reproductive									
Decreased ovary weight in rats Xu et al. (2010)	0.37	$BMDL_{1SD}$	1	10	3	10	3	1,000	4×10^{-4}
Decreased ovarian follicles in rats Xu et al. (2010)	0.38	$BMDL_{10RD}$	1	10	3	10	3	1,000	4×10^{-4}
Decreased intratesticular testosterone in rats Zheng et al. (2010)	0.24	NOAEL	1	10	3	10	3	1,000	2×10^{-4}
Decreased sperm count and motility in mice Mohamed et al. (2010)	0.15	LOAEL	10	10	3	10	3	10,000	Not calculated due to UF >3,000[a]
Cervical epithelial hyperplasia in mice Gao et al. (2011)	0.06	$BMDL_{10}$	1	10	3	10	3	1,000	6×10^{-5}
Immunological									
Decreased thymus weight in rats Kroese et al. (2001)	1.9	$BMDL_{1SD}$	1	10	3	10	3	1,000	2×10^{-3}
Decreased serum IgM in rats De Jong et al. (1999)	1.7	NOAEL	1	10	3	10	3	1,000	2×10^{-3}
Decreased serum IgA in rats De Jong et al. (1999)	5.2	NOAEL	1	10	3	10	3	1,000	5×10^{-3}
Decreased number of B cells in rats De Jong et al. (1999)	5.2	NOAEL	1	10	3	10	3	1,000	5×10^{-3}

[a]As recommended in EPA's *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002), the derivation of a reference value that involves application of the full 10-fold UF in four or more areas of extrapolation should be avoided.

Figure 2-1 presents graphically the candidate values, UFs, and PODs, with each bar corresponding to one data set described in Tables 2-1 and 2-2.

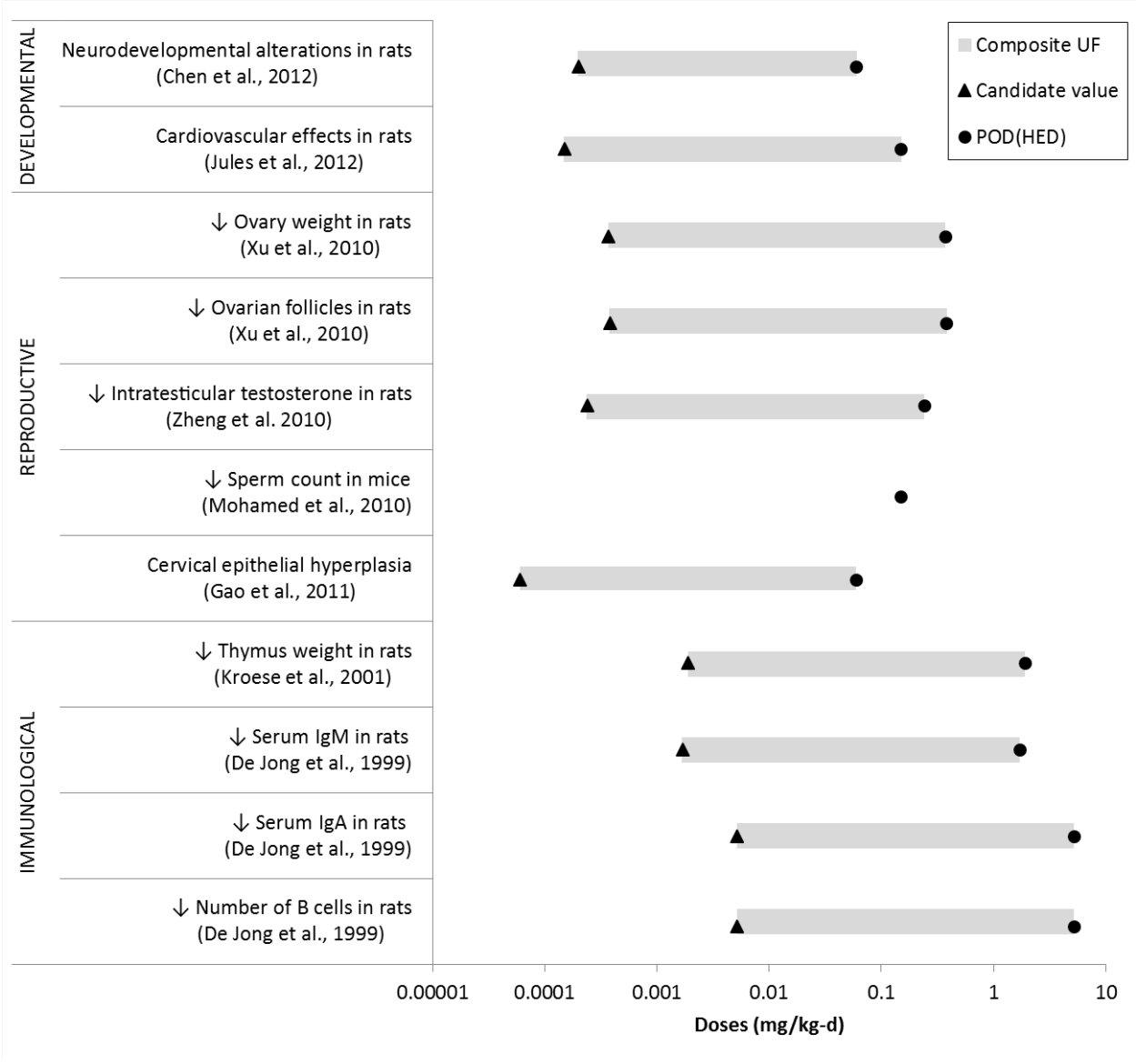

Figure 2-1. Candidate values with corresponding PODs and composite UFs.

2.1.4. Derivation of Organ/System-Specific Reference Doses

Table 2-3 distills the candidate values from Table 2-2 into a single value for each organ or system. 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 2-3. Organ/system-specific RfDs and overall RfD for benzo[a]pyrene

Effect	Basis	RfD (mg/kg-d)	Study exposure description	Confidence
Developmental	Neurobehavioral changes	3×10^{-4}	Critical window of development (postnatal)	Medium
Reproductive	Ovotoxicity (decreased primordial follicles and ovary weight)	4×10^{-4}	Subchronic	Medium
Immunological	Decreased thymus weight and serum IgM	2×10^{-3}	Subchronic	Low
Overall RfD	**Developmental toxicity (including developmental neurotoxicity)**	**3×10^{-4}**	Critical window of development (postnatal)	**Medium**

Developmental Toxicity

Candidate values to represent developmental toxicity were derived based on neurobehavioral changes and cardiovascular effects following developmental exposure. While the candidate value derived for developmental cardiovascular effects is slightly lower than the candidate value based on developmental neurotoxicity, the support across the database for developmental cardiovascular effects is considerably smaller, with one in vivo rodent study evaluating cardiovascular endpoints. Several in vivo studies in mice and rats, by multiple routes of exposure, support the findings of neurobehavioral changes following developmental exposure to benzo[a]pyrene. Therefore, the candidate value based on neurobehavioral changes in rats (Chen et al., 2012) was selected as the organ/system-specific RfD representing developmental toxicity. This candidate value was selected because it is associated with the application of the smaller composite UF, because it represents multiple neurobehavioral endpoints, and because similar effects were replicated across numerous additional studies (see Section 1.1.1).

Reproductive Toxicity

Among the adverse reproductive effects associated with oral benzo[a]pyrene exposure, decrements in sperm parameters, decreases in testosterone, and effects in the ovary were supported by a large body of evidence. The data supporting cervical effects are limited to a single study, and were therefore given less weight compared to the other reproductive effects. The derivation of a candidate value based on decreased sperm count and motility (Mohamed et al., 2010) involved too much uncertainty (see Table 2-2) and the study used to derive a candidate value based on decreased testosterone (Zheng et al., 2010) did not observe a dose-response relationship (a 15% decrease in testosterone was seen at the low and high doses). The study by Xu et al. (2010) observed dose-response relationships for decreased ovary weight and decreases in primordial

follicle counts. The ovarian effects are supported by a large database of animal studies and human studies of exposure to benzo[a]pyrene and PAH mixtures. Therefore, the candidate value based on decreased ovotoxicity in rats from the Xu et al. (2010) study was selected as the organ/system-specific RfD representing reproductive toxicity. While evidence in the benzo[a]pyrene database supports male and female reproductive hazards, there is more confidence in the POD from Xu et al. (2010) as the basis for an organ/system-specific RfD for reproductive effects.

Immunotoxicity

The candidate values based on decreased thymus weight (Kroese et al., 2001) and serum IgM levels in rats (De Jong et al., 1999) were the same and were selected as the organ/system-specific RfD representing immunotoxicity. The observed decreases in thymus weight, IgM and IgA levels, and number of B cells associated with exposure to benzo[a]pyrene were determined to be representative of immunotoxicity. In combination, these effects provide more robust evidence of immunotoxicity. The candidate values for decreased thymus weight (Kroese et al., 2001) and serum IgM levels in rats (De Jong et al., 1999) were comparable and provided the most sensitive POD; thus, these candidate values were selected as the organ/system-specific RfD representing immunotoxicity.

2.1.5. Selection of the Overall Reference Dose

Multiple organ/system-specific reference doses were derived for effects identified as human hazards or potential hazards from benzo[a]pyrene: developmental toxicity (including developmental neurotoxicity), male and female reproductive toxicity, and immunotoxicity. To estimate an exposure level below which effects from benzo[a]pyrene exposure are not expected to occur, the lowest organ/system-specific RfD (3×10^{-4} mg/kg-day) with the highest confidence was selected as the overall RfD for benzo[a]pyrene. This value, based on induction of neurobehavioral changes in rats exposed to benzo[a]pyrene during a susceptible lifestage, is supported by numerous animal and human studies across a variety of exposure paradigms (see Section 1.1.1).

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). This value should be applied in general population risk assessments. However, decisions concerning averaging exposures over time for comparison with the RfD should consider the types of toxicological effects and specific lifestages of concern. For example, fluctuations in exposure levels that result in elevated exposures during development could potentially lead to an appreciable risk, even if average levels over the full exposure duration were less than or equal to the RfD. Alternatively, developmental toxicity may not be a concern for exposure scenarios in which exposure is occurring outside of the critical window of development.

2.1.6. 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, 1994a). Confidence in the principal study (Chen et al., 2012) is medium. The study design included 40 litters of rats tested in multiple behavioral assays at multiple ages across 4 dose levels (including the control group). The study randomly assigned a total of 10 male and 10 female pups per treatment group per behavioral endpoint, with no more than one male and one female from each litter for behavioral testing. Importantly, all tests were conducted by investigators blinded to treatment, and test order was randomized each day. In addition, the pups were cross-fostered with dams being rotated among litters every 2–3 days to distribute any maternal caretaking differences randomly across litters and treatment groups. Some uncertainty exists regarding the potential for dam rotation across litters and the within-litter dosing design to introduce maternal stress and thus unanticipated consequences in the pups (see Section 1.1.1), and some informative experimental details were omitted, such as the sensitivity of some assays at the indicated developmental ages, gender-specific data for all outcomes and information on procedures for matching pup ages across the 40 litters examined. However, the overall methods and reporting for the specific behavioral endpoints supporting the RfD (i.e., open field locomotion, elevated plus maze activity, and Morris water maze performance) are considered sufficient and well-characterized. Confidence in the database is medium, primarily due to the lack of a multigenerational reproductive toxicity study and lack of a developmental neurotoxicity study with exposure that spanned early gestation through weaning (or longer), given the sensitivity to benzo[a]pyrene during development. Reflecting medium confidence in the principal study and medium confidence in the database, confidence in the RfD is medium.

2.1.7. Previous IRIS Assessment: Reference Dose

An RfD was not derived in the previous IRIS assessment.

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 percent lower bound on the benchmark concentration (BMCL), with UFs generally applied to reflect limitations of the data used.

2.2.1. Identification of Studies and Effects for Dose-Response Analysis

In Section 1.2.1, developmental and reproductive toxicities were identified as hazards of benzo[a]pyrene exposure by the inhalation route. Studies within each effect category were evaluated using general study quality characteristics (as discussed in Section 6 of the Preamble) to help inform the selection of studies from which to derive toxicity values. Rationales for selecting the studies and effects to represent each of these hazards are summarized below.

Human studies of environmental PAH mixtures across multiple cohorts have observed developmental and reproductive effects following prenatal exposure. However, these studies are limited by exposure to complex mixtures of PAHs; and, within individual studies, there may have been more than one route of exposure. In addition, the available human studies that utilized benzo[a]pyrene-DNA adducts as the exposure metric do not provide external exposure levels of benzo[a]pyrene from which to derive an RfC. Although preferred for derivation of reference values, human studies were not considered because of the contribution to the observed hazard of multiple PAHs across multiple routes of exposure and uncertainty due to concurrent exposure to other PAHs and other components of the mixtures (such as metals).

Animal studies were evaluated to determine which provided the most relevant routes and durations of exposure, multiple exposure levels to provide information about the shape of the dose response curve, and relative ability to detect effects at low exposure levels. The only chronic animal inhalation study available for benzo[a]pyrene, Thyssen et al. (1981), was designed as a cancer bioassay and did not report other effects; however, the inhalation database for benzo[a]pyrene includes several shorter duration studies that are sufficient for use in deriving reference values (U.S. EPA, 2002), including one subchronic study and several developmental studies that identify hazards of exposure during sensitive developmental windows. In addition, a 4-week inhalation study in rats is available that investigated, but did not detect, lung injury (Wolff et al., 1989). The inhalation database for benzo[a]pyrene is less extensive than the database of studies by the oral route; however, the types of noncancer effects observed are consistent between routes and are supported by studies in human populations (see Sections 1.1.1, 1.1.2, and 1.1.3).

Developmental Toxicity

Developmental toxicity, as represented by decreased fetal survival and limited evidence of developmental neurotoxicity, was observed in several inhalation studies conducted during gestation in rats (Wormley et al., 2004; Wu et al., 2003a; Archibong et al., 2002) and mice (Li et al., 2012). The studies in rodents demonstrated that the developing fetus is a sensitive target following inhalation exposure to benzo[a]pyrene, in terms of fetal survival. Only the study conducted by Archibong et al. (2002) supported dose-response analysis, with decreased fetal survival observed at the lowest dose tested by the inhalation route on GDs 11–20. Wu et al. (2003a) corroborated the responses seen by Archibong et al. (2002), but was not considered for dose-response modeling due

to lack of study details related to number of dams and litters per group and lack of reporting of numerical results; the reported data supported estimation of a LOAEL (25 μg/m^3).

The neurodevelopmental effects observed by Wormley et al. (2004) (reduced longterm potentiation) and Li et al. (2012) (deficits in object discrimination) were not considered as informative as Archibong et al. (2002) for dose-response analysis. Wormley et al. (2004) employed only a single, relatively high exposure (at which a 66% reduction in fetal survival in rats was observed) and Li et al. (2012) offers supplemental evidence of developmental neurotoxicity, but employed the same, single exposure level as Wormley et al. (2004) and tested genetically manipulated mice developed for use in conditional knock out studies, which may or may not respond comparably to unmanipulated, wild type animals.

Reproductive Toxicity

Male reproductive toxicity, as represented by reductions in sperm quality, both count and motility, and testis weights in adults, was observed by Archibong et al. (2008), Ramesh et al. (2008) and Archibong et al. (2002). Archibong et al. (2008) and Ramesh et al. (2008) reported the results of a 60-day inhalation exposure study in male rats. Although of sufficient duration for developing a reference value, the study utilized a single exposure concentration, which is less informative for dose-response analysis than a design using multiple exposure concentrations. However, the effects in this study are consistent with male reproductive effects observed across multiple studies by the oral route and with human studies in PAH exposed populations, as effects on male fertility and semen quality have been demonstrated in epidemiological studies of smokers (see Section 1.1.2). The endpoints of decreased testes weight and sperm count and motility reported in Archibong et al. (2008) were selected for dose-response analysis as both represent sensitive endpoints of male reproductive toxicity and are indicators of potentially decreased fertility.

For female reproductive toxicity, ovotoxicity—as represented by reduced ovulation rate and ovarian weight—and reduced litter sizes at birth, were observed by Archibong et al. (2012) following a 14-day pre-mating exposure period, distinct from the developmental period studied by Archibong et al. (2002) on GDs 11–20. Decreased ovary weights and ovulation rates were selected for dose-response analysis. Reduced litter sizes were not considered for reference concentration derivation because the means reflected ovulation rates at the lower two exposures, and would have yielded very similar PODs.

2.2.2. Methods of Analysis

As there were no biologically based dose-response models for inhalation exposure to benzo(a)pyrene available, the general methods for dose-response analysis for reference value concentration derivation were the same as for the reference dose (see Section 2.1.2).

The data for decreased fetal survival from Archibong et al. (2002), reported as litter mean percent survival at birth and associated SE, did not yield adequate fits when modeled as continuous data. The underlying responses were dichotomous (alive, not alive), and the resulting pattern of

variability in percentages across exposure groups was incompatible with the normal distribution assumption imposed by BMDS continuous models. Ideally, individual offspring data are needed to address the impact of intralitter correlation on effective sample sizes, but attempts to obtain individual offspring data from the study authors were unsuccessful.

In an attempt to use BMD modeling for these data, the Rao-Scott transformation was applied (Fox et al., 2016; Fung et al., 1998; Rao and Scott, 1992). Briefly, in each exposure group the transformation reflects the ratio of two variances--one under the assumption of complete independence of offspring and the other characterizing the hierarchical relationship of offspring within litters--and results in a decreased sample size (effective sample size). Dichotomous models in BMDS were then applied to the transformed fetal survival data. See Appendix E.1.2 for details of the methodology and the transformed data, as well as modeling details.

For decreased embryo/fetal survival, BMRs of 1% and 5% extra risk were considered, recognizing the severity of the outcome. Extrapolation below the lowest administered exposure is involved in estimating these BMDs for the (Archibong et al., 2002) data, given that the difference in response between the control and lowest exposure was approximately 20%. Taking into account the extent of model uncertainty, modeling results are presented for comparative purposes (see Table E-16) and the lowest exposure, considered a LOAEL, was used as the POD for decreased embryo/fetal survival.

For the endpoint of ovulation rate (Archibong et al., 2012), a BMR of 1% was considered due to the severity of the endpoint, which was correlated directly to reduced litter size. However, the small sample size (5 dams per dose group) did not support extrapolation to this BMR (that is, for the selected model, the BMDL was about tenfold lower than the BMD). Therefore, the lowest exposure was considered a LOAEL and was used as the POD.

For ovary weight (Archibong et al., 2012), a BMR of 10% relative deviation was selected, similar to the relative magnitude of the SD for ovary weight in the Xu et al. (2010) study used for developing a reproductiveRfD (see Section 2.1.2). However, adequate model fits were not achieved, in part due to the low reported variability in ovary weights in this study, relative to historical controls (see Section E.2). Therefore, the lowest exposure, showing a 10% response, was considered a NOAEL.

The study by Archibong et al. (2008)/ Ramesh et al. (2008), was judged not to support dose-response modeling due to the use of a single exposure level. Consequently there was insufficient information to characterize the underlying dose-response relationship. This study observed high magnitudes of response at the only dose tested, specifically, a 34% decrease in testicular weight and a 69% decrease in sperm number. Therefore, LOAELs were used as the PODs for dose-response analysis.

By definition, the RfC is intended to apply to continuous lifetime exposures for humans (U.S. EPA, 1994a). EPA recommends that adjusted continuous exposures be used for developmental

toxicity studies by the inhalation route as well as for inhalation studies of longer durations (U.S. EPA, 2002). The PODs were adjusted to account for the discontinuous daily exposure as follows:

$$POD_{ADJ} = POD \times \text{hours exposed per day/24 hours}$$
$$= LOAEL \times (\text{duration of exposure/24 hours})$$
$$= POD_{ADJ}$$

Next, the human equivalent concentration (HEC) was calculated from the POD_{ADJ} by multiplying by a dosimetric adjustment factor (DAF), which, in this case, was the regional deposited dose ratio ($RDDR_{ER}$) for extrarespiratory (i.e., systemic) effects as described in *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994a). The observed developmental effects are considered systemic in nature (i.e., extrarespiratory) and the normalizing factor for extrarespiratory effects of particles is body weight (i.e., the equivalent dose across species is mass deposited in the entire respiratory tract per unit body weight). The $RDDR_{ER}$ was calculated as follows:

$$RDDR_{ER} = \frac{BW_H}{BW_A} \times \frac{(V_E)_A}{(V_E)_H} \times \frac{(F_{TOT})_A}{(F_{TOT})_H}$$

where:

BW = body weight (kg);
V_E = ventilation rate (L/min); and
F_{TOT} = total fractional deposition.

The total fractional deposition includes particle deposition in the nasal-pharyngeal, tracheobronchial, and pulmonary regions. F_{TOT} for both animals and humans was calculated using the Multi-Path Particle Dosimetry (MPPD) model, a computational model used for estimating human and rat airway particle deposition (MPPD; Version 2.0 © 2006, as accessed through the former Hamner Institute; now publicly available through Applied Research Associates). FTOT was based on the average particle size of 1.7 ± 0.085 µm (mass median aerodynamic diameter [MMAD] ± geometric SD) as report ed in Wu et al. (2003a) for the exposure range 25–100 μm^3. For the model runs, the Yeh-Schum 5-lobe model was used for the human and the asymmetric multiple path model was used for the rat (see Appendix E for MPPD model output). Both models were run under nasal breathing scenarios after adjusting for inhalability. A geometric SD of 1 was used as the default by the model because the reported geometric SD of 0.085 was ≤1.05.

The human parameters used in the model for calculating F_{TOT} and in the subsequent calculation of the POD_{HEC} were as follows: human body weight, 70 kg; V_E, 13.8 L/minute; breathing frequency, 16 per minute; tidal volume, 860 mL; functional residual capacity, 3,300 mL; and upper respiratory tract volume, 50 mL. Although the most sensitive population in Archibong et al. (2002) is the developing fetus, the adult rat dams were directly exposed. Thus, adult rat parameters were used in the calculation of the HEC. The parameters used for the rat were body weight, 0.25 kg (a

generic weight for male and female rats); V_E, 0.18 L/minute; breathing frequency, 102 per minute; tidal volume, 1.8 mL; functional residual capacity, 4 mL; and upper respiratory tract volume, 0.42 mL. All other parameters were set to default values (see Appendix E).

Under these conditions, the MPPD model calculated F_{TOT} values of 0.621 for the human and 0.181 for the rat. Using the above equation, the $RDDR_{ER}$ was calculated to be 1.1.

From this, the POD_{HEC} was calculated as follows:

$$POD_{HEC} = POD_{ADJ} \times RDDR_{ER}$$

Table 2-4 summarizes the sequence of calculations leading to the derivation of a human-equivalent POD for each data set discussed above.

Table 2-4. Summary of derivation of PODs

Endpoint and reference	Species/sex	Model	BMR	BMC (µg/m³)	BMCL (µg/m³)	POD_{ADJ}[a] (µg/m³)	POD_{HEC}[b] (µg/m³)
Developmental							
Decreased embryo/fetal survival Archibong et al. (2002)	Pregnant F344 rats	LOAEL (25 µg/m³) 19% ↓				4.2	4.6
Decreased embryo/fetal survival Wu et al. (2003a)	Pregnant F344 rats	LOAEL (25 µg/m³) 10% ↓				4.2	4.6
Reproductive							
Decreased ovulation rate Archibong et al. (2012)	Female F344 rats	NOAEL (50 µg/m³) 9% ↓				8.3	9.1
Decreased ovary weight Archibong et al. (2012)	Female F344 rats	NOAEL (50 µg/m³) 10% ↓				8.3	9.1
Decreased testis weight Archibong et al. (2008)	Male F344 rats	LOAEL (75 µg/m³) 34% ↓				12.5	13.8
Decreased sperm count and motility Archibong et al. (2008)	Male F344 rats	LOAEL (75 µg/m³) 69% ↓sperm count 73% ↓ sperm motility 54% ↑ abnormal sperm				12.5	13.8

[a]PODs were adjusted for continuous daily exposure: POD_{ADJ}= POD × hours exposed per day/24 hours.
[b]POD_{HEC} calculated by adjusting the POD_{ADJ} by the RDDR calculated using particle size reported in Hood et al. (2000) and Wu et al. (2003a) using MPPD software as detailed in Section 2.2.2 and Appendix E in the Supplemental Information.

2.2.3. 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), also described in the Preamble, five areas of uncertainty and variability were considered, as follows:

A UF for extrapolation from a LOAEL to a NOAEL, UF_L, of 10 was applied when a LOAEL was used as the POD (Archibong et al., 2012; Wu et al., 2003a; Archibong et al., 2002). There was a high magnitude of response at these LOAELs (see Table 2-4). For example, the LOAEL used as the POD for the developmental effect observed in Wu et al. (2003a) was based on a 19% decrease in fetal survival and Archibong et al. (2008) observed a 34% decrease in testis weight and a 69% decrease in sperm count. Therefore, a full UF of 10 was applied to approximate a NOAEL for this study. For decreased ovary weight (Archibong et al., 2012), a decrease of 10% was observed at the lowest dose tested, which is also the BMR used (which is thought to approximate a NOAEL) for ovary weight decreases in the oral database in the Xu et al. (2010) study (see Section 2.1.2). Therefore, this dose was considered a NOAEL and a 1-fold UF was applied.

A subchronic to chronic uncertainty factor, UF_S, of 1 was applied when dosing occurred during gestation (Wu et al., 2003a; Archibong et al., 2002) or the early postnatal period that is relevant to developmental effects (U.S. EPA, 1991a). The developmental period is recognized as a susceptible lifestage when exposure during a time window of development is more relevant to the induction of developmental effects than lifetime exposure (U.S. EPA, 1991c); therefore, an adjustment for duration is not warranted. A partial UF of 3 was applied to effects on ovulation following pre-mating exposure during a sensitive reproductive window (Archibong et al., 2012). A value of 10 was applied when the POD is based on a subchronic study to account for the possibility that longer exposure may induce more severe effects or effects at a lower dose.

A dosimetric adjustment (RDDR) is applied to adjust for differences in particle deposition across species. Accordingly, a reduced interspecies uncertainty factor, UF_A, of 3 ($10^{1/2} = 3.16$, rounded to 3) is applied to account for uncertainties in characterizing toxicodynamic as well as residual toxicokinetic differences in the extrapolation from laboratory animals to humans after inhalation exposure to benzo[a]pyrene as described in EPA's *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994a). Residual toxicokinetic uncertainties include the considerations that the methodology adjusts for interspecies differences in the deposition of the inhaled dose in the entire respiratory tract ignoring regional differences (e.g., a greater percentage would be expected to reach the lung in humans as compared to rats) as well as interspecies differences in clearance. It is reasonable to consider the dose to the entire respiratory tract in either species since the mode of action for the endpoint used as the basis of the RfC (decreased fetal survival) is not known. Furthermore, data for modeling species differences in clearance and metabolism of the deposited particles are not available, therefore, given these uncertainties, the relevant dose metric is considered to be the mass of benzo[a]pyrene

deposited per day in the entire respiratory tract. This metric is thought to be more accurate than using exposure concentration as the default.

An intraspecies uncertainty factor, UF_H, of 10 was applied to account for variability and uncertainty in toxicokinetic and toxicodynamic susceptibility within the subgroup of the human population most sensitive to the health hazards of benzo[a]pyrene (U.S. EPA, 2002). In the case of benzo[a]pyrene, the PODs were derived from studies in inbred animal strains and are not considered sufficiently representative of the exposure and dose-response of the most susceptible human subpopulations (in this case, the developing fetus). In certain cases, the toxicokinetic component of this factor may be replaced when a PBPK model is available that incorporates the best available information on variability in toxicokinetic disposition in the human population (including sensitive subgroups). In the case of benzo[a]pyrene, insufficient information is available to quantitatively estimate variability in human susceptibility; therefore, the full value for the intraspecies UF was applied.

Although hazards are similar across oral and inhalation databases, fewer studies exist by the inhalation route to characterize the dose response of inhaled benzo[a]pyrene. A full database uncertainty factor, UF_D, of 10 was applied to account for database deficiencies, including the lack of a standard multigenerational study or extended 1-generation study that includes exposure from premating through lactation, considering that benzo[a]pyrene has been shown to affect fertility in adult male and female animals by multiple routes of exposure and that decrements in fertility are greater following developmental exposure (see Section 1.1.2).

In addition, the general lack of studies examining functional neurological endpoints following inhalation exposure during development is a significant data gap, considering human and animal evidence indicating altered neurological development following exposure to benzo[a]pyrene alone or through PAH mixtures (see Section 1.1.1). An additional uncertainty in the inhalation database for benzo[a]pyrene includes the lack of studies characterizing immune system toxicity, especially following developmental exposure.

The most sensitive POD for the inhalation candidate values in Table 2-5 is based on the endpoint of decreased embryo/fetal survival observed in Archibong et al. (2002), supported by Wu et al. (2003a). Decreased fetal survival was also observed in oral exposure studies, however it was seen at much higher doses than developmental neurotoxicity. A 16% decrease in F1 fetal survival was observed following treatment with 160 mg/kg-day benzo[a]pyrene, but not at lower doses (Mackenzie and Angevine, 1981); however, other oral studies observed significant neurobehavioral effects at doses of benzo[a]pyrene around 0.2–2 mg/kg-day (Chen et al., 2012; Bouayed et al., 2009a). Considering the relative sensitivity of the systemic health effects observed in the oral database, it is possible that developmental neurotoxicity would occur at exposure concentrations below the POD for the RfC based on decreased fetal survival. According to EPA's *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002; Section 4.4.5), the UF_D is intended to account for the potential for deriving an under-protective RfD/RfC as a result of an

incomplete characterization of the chemical's toxicity, but also including a review of existing data that may also suggest that a lower reference value might result if additional data were available. Therefore, a database UF of 10 for the benzo[a]pyrene inhalation database was applied to account for the lack of a multigenerational study and the lack of a developmental neurotoxicity study.

Table 2-5 is a continuation of Table 2-4 and summarizes the application of UFs to each POD to derive a candidate values for each data set. The candidate values presented in the table below are preliminary to the derivation of the organ/system-specific reference values. These candidate values are considered individually in the selection of an RfC for a specific hazard and subsequent overall RfC for benzo[a]pyrene.

Table 2-5. Effects and corresponding derivation of candidate values

Endpoint	POD_{HEC} $\mu g/m^3$	POD type	UF_L	UF_S	UF_A	UF_H	UF_D	Composite UF[b]	Candidate value[a] mg/m^3
Developmental									
Decreased embryo/fetal survival in rats Archibong et al. (2002)	4.6	LOAEL	10	1	3	10	10	3,000	2×10^{-6}
Decreased embryo/fetal survival in rats Wu et al. (2003a)	4.6	LOAEL	10	1	3	10	10	3,000	2×10^{-6}
Reproductive									
Decreased ovulation rate Archibong et al. (2012)	9.1	LOAEL	10	1	3	10	10	3,000	3×10^{-6}
Decreased ovary weight Archibong et al. (2012)	9.1	NOAEL	1	10	3	10	10	3,000	3×10^{-6}
Decreased testis weight in rats Archibong et al. (2008)	13.8	LOAEL	10	10	3	10	10	30,000	Not calculated due to UF >3,000
Decreased sperm count and motility in rats Archibong et al. (2008)	13.8	LOAEL	10	10	3	10	10	30,000	Not calculated due to UF >3,000

[a]Candidate values were converted from $\mu g/m^3$ to mg/m^3.
[b]As recommended in EPA's *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002), the derivation of a reference value that involves application of the full 10-fold UF in four or more areas of extrapolation should be avoided.

Figure 2-2 presents graphically these candidate values UFs and PODs, with each bar corresponding to one data set described in Tables 2-4 and 2-5.

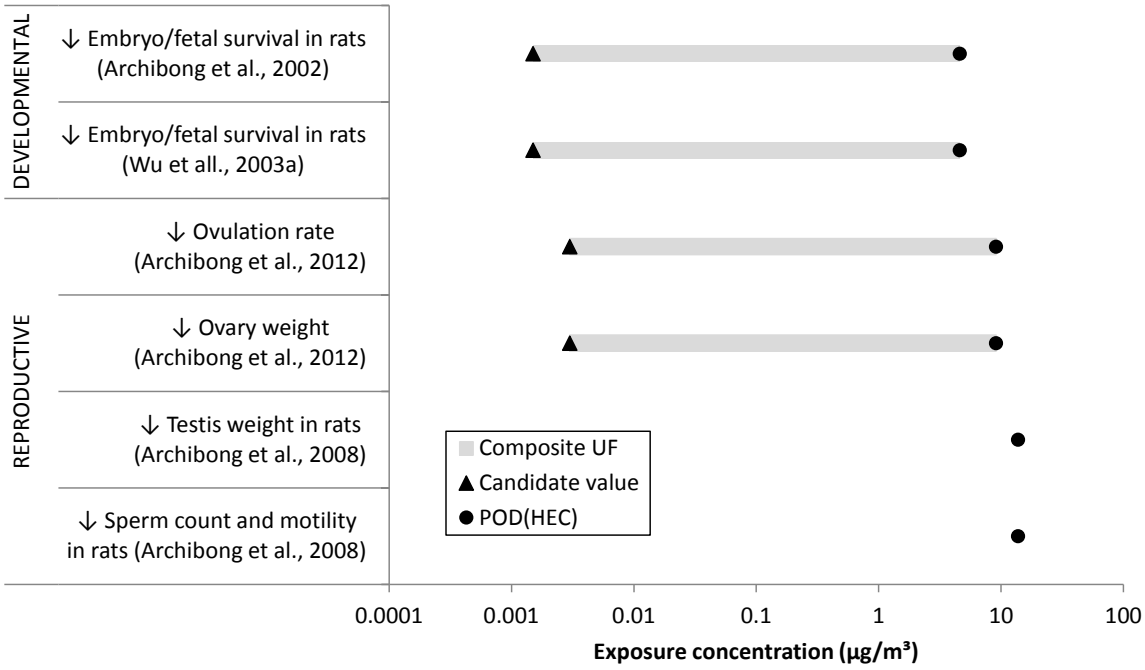

Figure 2-2. Candidate values with corresponding PODs and composite UFs.

2.2.4. Derivation of Organ/System-Specific Reference Concentrations

Table 2-6 distills the candidate values from Table 2-5 into a single value for each organ or system. 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. Candidate values for reproductive toxicity (decreased testis weight and decreased sperm count and motility) from Archibong et al. (2008) were not derived, because as recommended in EPA's *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002), the derivation of a reference value that involves application of the full 10-fold UF in four or more areas of extrapolation should be avoided.

Table 2-6. Organ/system-specific RfCs and overall RfC for benzo[a]pyrene

Effect	Basis	RfC (mg/m^3)	Study exposure description	Confidence
Developmental	Decreased fetal survival	2×10^{-6}	Critical window of development (prenatal)	Low-medium
Reproductive	Decreased ovary weight and ovulation rate	3×10^{-6}	Short term pre-mating exposure	Low-medium
Overall RfC	**Decreased fetal survival**	**2×10^{-6}**	Critical window of development (prenatal)	**Low-medium**

2.2.5. Selection of the Reference Concentration

The derivation of multiple organ/system-specific reference concentrations were considered for effects identified as human hazards of benzo[a]pyrene inhalation exposure, i.e., developmental and reproductive toxicity. Organ/system-specific RfCs to represent reproductive toxicity were calculated based on female reproductive toxicity, specifically decreased ovulation rate. An RfC based on male reproductive toxicity could not be derived due to high uncertainty (i.e., a composite UF of >3,000).

An overall RfC of $\mathbf{2 \times 10^{-6}\ mg/m^3}$ was selected based on the hazard of developmental toxicity. The study by Archibong et al. (2002) was selected for the derivation of the overall RfC, as it observed biologically significant effects at the lowest dose tested by the inhalation route. This study indicates that the developing fetus is a sensitive target following inhalation exposure to benzo[a]pyrene and the observed decreased fetal survival/litter is the most sensitive noncancer effect observed following inhalation exposure to benzo[a]pyrene. Additional support for this endpoint of decreased fetal survival is provided by another developmental/reproductive study conducted via inhalation (Wu et al., 2003a), and one conducted via the oral route (Mackenzie and Angevine, 1981).

This overall RfC is derived to be protective of all 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). This value should be applied in general population risk assessments. However, decisions concerning averaging exposures over time for comparison with the RfC should consider the types of toxicological effects and specific lifestages of concern. For example, 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 RfC. Alternatively, developmental toxicity may not be a concern due to exposure scenarios in which exposure is occurring outside of the critical window of development.

2.2.6. Confidence Statement

A confidence level of high, medium, or low is assigned to the study used to derive the RfC, the overall database, and the RfC 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, 1994a).

The overall confidence in the RfC is low-to-medium. Confidence in the principal study (Archibong et al., 2002) is medium. The conduct and reporting of this developmental study were adequate; however, a NOAEL was not identified. A lower response of 10% extra risk was observed at the same exposure level in a similar study from the same investigators (Wu et al., 2003a), suggesting some uncertainty in characterizing this dose-response relationship. Confidence in the database is low due to the lack of a multigeneration toxicity study, lack of studies on developmental

neurotoxicity and immune endpoints, and lack of information regarding subchronic and chronic inhalation exposure. However, confidence in the RfC is bolstered by consistent systemic effects observed by the oral route (including reproductive and developmental effects) and similar effects observed in human populations exposed to PAH mixtures. Reflecting medium confidence in the principal study and low confidence in the database, confidence in the RfC is low-to-medium.

2.2.7. Previous IRIS Assessment: Reference Concentration

An RfC was not derived in the previous IRIS assessment.

2.2.8. Uncertainties in the Derivation of the RfD and RfC

The following discussion identifies uncertainties associated with the RfD and RfC for benzo[a]pyrene. To derive the RfD, the UF approach (U.S. EPA, 2000, 1994a) was applied to a POD based on neurobehavioral changes in rats treated developmentally. To derive the RfC, this same approach was applied to a POD from a developmental study for the effect of decreased fetal survival. UFs were applied to the POD to account for extrapolating from an animal bioassay to human exposure, the likely existence of a diverse population of varying susceptibilities, and database deficiencies. These extrapolations are carried out with default approaches given the lack of data to inform individual steps.

The database for benzo[a]pyrene contains limited human data. The observation of effects associated with benzo[a]pyrene exposure in humans is complicated by several factors including the existence of benzo[a]pyrene in the environment as one component of complex mixtures of PAHs, exposure to benzo[a]pyrene by multiple routes of exposure within individual studies, and the difficulty in obtaining accurate exposure information. Data on the effects of benzo[a]pyrene alone are derived from a large database of studies in animal models. The database for oral benzo[a]pyrene exposure includes two lifetime bioassays in rats and mice, two developmental studies in mice, and several subchronic studies in rats.

Although the database is adequate for RfD derivation, there is uncertainty associated with the database including that the principal study for the RfD exposed animals during a relatively short period of brain development potentially underestimating the magnitude of resulting neurological effects. Also, the database lacks a comprehensive multi-generation reproductive/developmental toxicity studies and immune system endpoints were not evaluated in the available chronic-duration or developmental studies. Additionally, the only available chronic studies of oral or inhalational exposure to benzo[a]pyrene focused primarily on neoplastic effects leaving non-neoplastic effects mostly uncharacterized.

Additional uncertainty remains that the POD for the overall impact of neurodevelopmental effects might be lower than the selected POD. Specifically, if individual animal data for the Track 4 rats were available, consideration of the changes in any of the three behavioral tests as indicating an abnormal response for each rat would better represent the total behavioral effect, and could result in a lower POD. In addition, as altered performance in these three behavioral tests was more

severe when tested in adult, as compared to juvenile, animals, it is possible that testing animals at even older ages (i.e., after PND 75) would reveal even more sensitive effects of exposure. However, experiments addressing these possibilities were not available and these remain unaddressed uncertainties. Overall, this POD is the best supported value that can be derived using the currently available information, recognizing the multiple effects observed in the same study.

The only chronic inhalation study of benzo[a]pyrene was designed as a lifetime carcinogenicity study and did not examine noncancer endpoints (Thyssen et al., 1981). In addition, subchronic and short-term inhalation studies are available, which examine developmental and reproductive endpoints in rats. Developmental studies by the inhalation route identified biologically significant reductions in the number of pups/litter and percent fetal survival and possible neurodevelopmental effects following gestational exposures. A 14-day premating reproductive study in female rats observed decreased ovulation rate and ovary weight in treated animals. Additionally, a 60-day oral study in male rats reported male reproductive effects (e.g., decreased testes weight and sperm production and motility), but provides limited information to characterize dose-response relationships with chronic exposure scenarios. The study selected as the basis of the RfC provided limited information regarding the inhalation exposures of the animals. Specifically, it is not clear whether the reported concentrations were target values or analytical concentrations and the method used to quantify benzo[a]pyrene in the generated aerosols was not provided. Requests to obtain additional study details from the authors were unsuccessful; therefore, the assumption was made that the reported concentrations were analytical concentrations.

Results from several different studies indicate that the endpoint of decreased number of pups per litter may be impacted during different sensitive windows of exposure, likely by different modes of action. The critical study used for the derivation of the RfC treated dams following implantation and quantification of conceptuses, from GD 11 to 20 and observed a decrease in embryos/fetuses per litter (Archibong et al., 2002). Another study treated female rats for 2 weeks immediately prior to mating and observed a decrease in ovulation rate (e.g., number of oocytes released), and a decreased number of pups born per litter (Archibong et al., 2012). Yet another study treated animals with benzo[a]pyrene by i.p. on GDs 1–5, and observed a decrease in the number of implantation sites and hypothesized that benz[a]pyrene exposure may effect endometrial receptivity (Zhao et al., 2014). These three studies observed similar effects during different exposure windows, indicating that an exposure that included treatment prior to mating and through gestation, would likely result in an even greater reduction in the number of pups produced per litter. Therefore, it is possible that the critical study used by the RfC may have observed a greater reduction of pups born per litter if exposure covered a more comprehensive duration.

Another area of uncertainty in the database pertains to the lack of information regarding fertility in animals exposed gestationally to benzo[a]pyrene, especially in light of developmental

studies by the oral route indicating reduced fertility in the F1 generation and decreased reproductive organ weights. The database also lacks a multigenerational reproductive study via the inhalation route. Areas of uncertainty include the lack of chronic inhalation studies focusing on noncancer effects, limited data on dose-response relationships for impaired male or female fertility with gestational exposure or across several generations, and limited data on immune system endpoints with chronic exposure or developmental exposure to benzo[a]pyrene.

The toxicokinetic and toxicodynamic differences for benzo[a]pyrene between the animal species in which the POD was derived and humans are unknown. PBPK models can be useful for the evaluation of interspecies toxicokinetics; however, the benzo[a]pyrene database lacks an adequate model that would inform potential differences. There is some evidence from the oral toxicity data that mice may be more susceptible than rats to some benzo[a]pyrene effects (such as ovotoxicity) (Borman et al., 2000), although the underlying mechanistic basis of this apparent difference is not understood. Most importantly, it is unknown which animal species may be more comparable to humans.

2.3. ORAL SLOPE FACTOR 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 oral slope factor is a plausible upper bound on the estimate of risk per mg/kg-day of lifetime oral exposure.

2.3.1. Analysis of Carcinogenicity Data

The database for benzo[a]pyrene contains numerous cancer bioassays that identify tumors, primarily of the alimentary tract including the forestomach, following oral exposure in rodents. Three 2-year oral bioassays are available that associate lifetime benzo[a]pyrene exposure with carcinogenicity at multiple sites: forestomach, liver, oral cavity, jejunum, kidney, auditory canal (Zymbal gland) tumors, and skin or mammary gland tumors in male and female Wistar rats (Kroese et al., 2001); forestomach tumors in male and female Sprague-Dawley rats (Brune et al., 1981); and forestomach, esophageal, tongue, and larynx tumors in female B6C3F$_1$ mice (Beland and Culp, 1998; Culp et al., 1998).

In addition to these 2-year cancer bioassays, there are studies available that provide supporting evidence of carcinogenicity but are less suitable for slope factor derivation due to one or more limitations in study design: (1) no vehicle control group; (2) only one benzo[a]pyrene dose group; or (3) a one-time exposure to benzo[a]pyrene (Benjamin et al., 1988; Robinson et al., 1987; El-Bayoumy, 1985; Wattenberg, 1974; Roe et al., 1970; Biancifiori et al., 1967; Chouroulinkov et al., 1967; Berenblum and Haran, 1955). Of the controlled, multiple dose-group, repeat-dosing studies that remain, most treated animals for <1 year (Weyand et al., 1995; Triolo et al., 1977; Fedorenko

and Yansheva, 1967; Neal and Rigdon, 1967). When lifetime studies are not available, shorter studies such as these can support extrapolation to lifetime exposure, but are less optimal for slope factor derivation given the availability of chronic studies.

In another lifetime study, Brune et al. (1981) dosed rats (32/sex/group) with benzo[a]pyrene in the diet or by gavage in a 1.5% caffeine solution, sometimes as infrequently as once every 9th day, for approximately 2 years and observed increased forestomach tumors. This study was not selected for quantitation due to the nonstandard treatment protocol in comparison to the studies conducted by Kroese et al. (2001) and Beland and Culp (1998) and the limited reporting of study methods.

The Kroese et al. (2001) and Beland and Culp (1998) studies were selected as the best available studies for dose-response analysis and extrapolation to lifetime cancer risk following oral exposure to benzo[a]pyrene. The rat bioassay by Kroese et al. (2001) and the mouse bioassay by Beland and Culp (1998) were conducted in accordance with Good Laboratory Practice (GLP) as established by the Organisation for Economic Co-operation and Development (OECD). These studies included histological examinations for tumors in many different tissues, contained three exposure levels and controls, contained adequate numbers of animals per dose group (~50/sex/group), treated animals for up to 2 years, and included detailed reporting of methods and results (including individual animal data).

Details of the rat (Kroese et al., 2001) and female mouse (Beland and Culp, 1998) study designs are provided in Appendix D of the Supplemental Information. Dose-related increasing trends in tumors were noted at the following sites:

- Squamous cell carcinomas (SCCs) or papillomas of the forestomach or oral cavity in male and female rats;
- SCCs or papillomas of the forestomach, tongue, larynx, or esophagus in female mice;
- Auditory canal carcinomas in male and female rats;
- Kidney urothelial carcinomas in male rats;
- Jejunum/duodenum adenocarcinomas in female and male rats;
- Hepatocellular adenomas or carcinomas in male and female rats; and
- SCCs or basal cell tumors of the skin or mammary gland in male rats.

These tumors were generally observed earlier during the study with increasing exposure levels, and showed statistically significantly increasing trends in incidence with increasing exposure level (Cochran-Armitage trend test, $p \leq 0.001$). These data are summarized in Appendix D of the Supplemental Information. As recommended by the National Toxicology Program (NTP) (McConnell et al., 1986) and as outlined in EPA's *Guidelines for Carcinogen Risk Assessment* (U.S.

EPA, 2005a), etiologically similar tumor types (i.e., benign and malignant tumors of the same cell type) were combined for these tabulations when it was judged that the benign tumors could progress to the malignant form. In addition, when one tumor type occurred across several functionally related tissues, as with squamous cell tumors in the tongue, esophagus, larynx, and forestomach, or adenocarcinomas of the jejunum or duodenum, these incidences were also aggregated as counts of tumor-bearing animals.

In the rat study (Kroese et al., 2001), the oral cavity and auditory canal were examined histologically only if a lesion or tumor was observed grossly at necropsy. Consequently, dose-response analysis for these sites was not straightforward. Use of the number of tissues examined histologically as the number at risk would tend to overestimate the proportion with tumors, because the unexamined animals would have been less likely to have tumors if none were observable grossly. On the other hand, use of all animals on study in a group as the number at risk would tend to underestimate if any of the unexamined animals had tumors that could only be detected microscopically. The oral cavity squamous cell tumors were combined with those in the forestomach because both are part of the alimentary tract, recognizing that there was some potential for underestimating this cancer risk.

The auditory canal tumors from the rat study were not considered for dose-response analysis, for several reasons. Unlike the oral cavity tumors, the auditory canal tumors appeared to be independent of the alimentary system tumors, as they were described as a mixture of squamous and sebaceous cells derived from pilosebaceous units, and there was no indication that these tumors were metastases from other sites (in which case, the auditory canal tumors would be repetitions of other tumors, or statistically dependent). As with the oral cavity tumors, the incomplete histological evaluation in the control and lower dose groups did not support dose-response analysis. Alternatively, even if these tumors were similar enough to be combined with the oral cavity and forestomach tumors, with only one exception (one female rat in the high dose group) they were coincident with alimentary tumors, and the joint incidence would be very similar to that of alimentary system tumors. Therefore dose-response analysis was not pursued for this site, either separately or in combination with another tumor type.

The incidence data that were modeled are provided in Appendix E (Kroese et al., 2001; Beland and Culp, 1998).

2.3.2. Dose-Response Analysis—Adjustments and Extrapolation Methods

EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a) recommend that the method used to characterize and quantify cancer risk from a chemical is determined by what is known about the mode of action of the carcinogen and the shape of the cancer dose-response curve. The dose response is assumed to be linear in the low-dose range when the agent is DNA-reactive and has direct mutagenic activity, or if another mode of action that is anticipated to be linear is applicable. EPA concluded that benzo[a]pyrene carcinogenicity involves a mutagenic mode of action (as discussed in Section 1.1.5). Thus, a linear approach to low-dose extrapolation was used.

The high-dose groups of both the rat and mouse studies were dead or moribund by week 76 for male rats, week 72 for female rats, and week 79 for female mice. Due to the occurrence of multiple tumor types, earlier occurrence with increasing exposure and early termination of the high-dose group in each study, methods that can reflect the influence of competing risks and intercurrent mortality on site-specific tumor incidence rates are preferred. In this case, EPA used the multistage-Weibull model, which incorporates the time at which death-with-tumor occurred as well as the dose.

Adjustments for approximating human equivalent slope factors applicable for continuous exposure were applied prior to dose-response modeling. First, continuous daily exposure for the gavage study in rats (Kroese et al., 2001) was estimated by multiplying each administered dose by (5 days)/(7 days) = 0.71, under the assumption of equal cumulative exposure yielding equivalent outcomes. Dosing was continuous in the mouse diet study (Beland and Culp, 1998), so no continuous adjustment was necessary. It was not necessary to adjust the administered doses for lifetime equivalent exposure prior to modeling for the groups terminated early, because the multistage-Weibull model characterizes the tumor incidence as a function of time, from which it provides an extrapolation to lifetime exposure.

Next, consistent with the EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), adjustment for cross-species scaling was considered to address toxicological equivalence across species. Despite extensive research into benzo[a]pyrene toxicokinetics (see Sections 1.1.6, D.1, D.2), very little information directly informs estimates of human-equivalent benzo[a]pyrene doses. It is understood that benzo[a]pyrene carcinogenicity involves a mutagenic mode of action mediated by DNA-reactive metabolites in the tissues where tumors appear. While the metabolites are highly reactive, distribution of benzo[a]pyrene to these tissues may be limited by processes consistent with $BW^{3/4}$ proportionality.

EPA guidance for oral exposures (U.S. EPA, 1992) asserts that, for a portal-of-entry scenario, "the most appropriate dose metric would likely be mass of agent per surface area, e.g., mg/cm^2," but that necessary considerations for implementing this approach have yet to be developed (e.g., surface areas of the GI tract in rodents and humans, including for an as yet unidentified human anatomical equivalent to the rodent forestomach; rates and scenarios of ingestion, including proximal to distal penetration down the GI tract; and diffusion rates). In the absence of this information, the guidance makes a general recommendation to use the $BW^{3/4}$ approach for oral portal-of-entry effects. For the Beland and Culp (1998) study, time-weighted daily average doses were converted to HEDs on the basis of $BW^{3/4}$.

Kroese et al. (2001) administered benzo[a]pyrene via gavage and observed tumors both in the alimentary system and systemically (in the kidney, liver, skin and mammary gland). It is not clear what impact gavage administration has on estimating human-equivalent doses of benzo[a]pyrene for the alimentary system, but the non-alimentary system tumors would be expected to reflect $BW^{3/4}$ proportionality to exposure. In the absence of information to characterize

portal-of-entry dosimetry and given the systemic component of the tumor profile, the time-weighted daily average doses were converted to HEDs on the basis of $BW^{3/4}$. This was accomplished by multiplying administered doses by (animal body weight (kg)/70 kg)$^{1/4}$, where the animal body weights were TWAs from each group, and the U.S. EPA (1988) reference body weight for humans is 70 kg.

PODs for estimating low-dose risk were identified at doses at the lower end of the observed data, corresponding to 10% extra risk. Details of the modeling and the model selection process can be found in Appendix E.2 of the Supplemental Information.

2.3.3. Derivation of the Oral Slope Factor

The PODs estimated for each tumor site are summarized in Table 2-7. The lifetime oral cancer slope factor for humans is defined as the slope of the line from the lower 95% bound on the exposure at the BMD to the control response (slope factor = $0.1/BMDL_{10}$). This slope represents a plausible upper bound on the true risk. Using linear extrapolation from the $BMDL_{10}$, human equivalent oral slope factors were derived for each gender/tumor site combination and are listed in Table 2-7.

Oral slope factors derived from rat bioassay data varied by gender and tumor site (Table 2-7). Values ranged from 0.04 per mg/kg-day, based on kidney tumors in males, to 0.4 per mg/kg-day, based on alimentary tract tumors in males. Slope factors based on liver tumors in male and female rats (0.2 per mg/kg-day) were only slightly lower than slope factors based on alimentary tract tumors (0.2–0.3 per mg/kg-day). The oral slope factor for alimentary tract tumors in female mice was highest at 1 per mg/kg-day (Table 2-7), which was approximately twofold higher than the oral slope factor derived from the alimentary tract tumors in male rats.

Table 2-7. Summary of the oral slope factor derivations

Tumor	Species/ sex	Selected model	BMR	BMD (mg/kg-d)	POD = BMDL (mg/kg-d)	Slope factor[a] $(mg/kg\text{-}d)^{-1}$	
Forestomach, oral cavity: squamous cell tumors Kroese et al. (2001)	Male Wistar rats	Multistage Weibull	10%	0.453	0.281	0.36	
Hepatocellular adenomas or carcinomas Kroese et al. (2001)	Male Wistar rats	Multistage Weibull	10%	0.651	0.449	0.22	
Jejunum/duodenum adenocarcinomas Kroese et al. (2001)	Male Wistar rats	Multistage Weibull	10%	3.03	2.38	0.042	
Kidney: urothelial carcinomas Kroese et al. (2001)	Male Wistar rats	Multistage Weibull	10%	4.65	2.50	0.040	0.5[b]

Tumor	Species/ sex	Selected model	BMR	BMD (mg/kg-d)	POD = BMDL (mg/kg-d)	Slope factor[a] $(mg/kg-d)^{-1}$	
Skin, mammary: Basal cell tumors Squamous cell tumors Kroese et al. (2001)	Male Wistar rats	Multistage Weibull	10%	 2.86 2.64	 2.35 1.77	 0.043 0.056	
Forestomach, oral cavity: squamous cell tumors Kroese et al. (2001)	Female Wistar rats	Multistage Weibull	10%	0.539	0.328	0.3	0.31[b]
Hepatocellular adenomas or carcinomas Kroese et al. (2001)	Female Wistar rats	Multistage Weibull	10%	0.575	0.507	0.2	
Jejunum/duodenum adenocarcinomas Kroese et al. (2001)	Female Wistar rats	Multistage Weibull	10%	3.43	1.95	0.05	
Forestomach, esophagus, tongue, larynx (alimentary tract): squamous cell tumors Beland and Culp (1998)	Female $B6C3F_1$ Mice	Multistage Weibull	10%	0.127	0.071	1.4	1.4

[a]Human equivalent slope factor = $0.1/BMDL_{10HED}$; see Appendix E of the Supplemental Information for details of modeling results.
[b]Slope factor characterizing the risk of incurring at least one of the tumor types listed.

Although the time-to-tumor modeling helps to account for competing risks associated with decreased survival times and other causes of death including other tumors, considering the tumor sites individually does not convey the total amount of risk potentially arising from the sensitivity of multiple sites—that is, the risk of developing any combination of the increased tumor types. A method, for estimating overall risk, involving the assumption that the variability in the slope factors could be characterized by a normal distribution, is detailed in Appendix E.2.1 of the Supplemental Information. The resulting composite slope factor for all tumor types for male rats was 0.5 per mg/kg-day, about 25% higher than the slope factor based on the most sensitive tumor site, oral cavity and forestomach, while for female rats, the composite slope factor did not increase from that for the most sensitive site (Tables 2-, E-27).

The overall risk estimates from male and female rats and female mice spanned about a fivefold range. While EPA's cancer guidelines (U.S. EPA, 2005a) suggest "choosing a single datset if it can be justified as most representative of the overall response in humans, there are no data to support any one result as most relevant for extrapolating to humans. Under the assumption that the three data sets are equally relevant for extrapolating to humans, geometric and harmonic mean of the three slope factors derived here round to 0.60 and 0.50 per mg/kg-day, respectively, about 40% of the highest slope factor. A geometric mean that gives equal weight to rats and mice is 0.74 per mg/kg-day, about 50% of the highest slope factor.

Another consideration in developing a human-equivalent slope factor is that slope factors are intended to provide an upper bound on the cancer risk of a randomly selected individual (U.S. EPA, 2005a), yet EPA's approach to quantifying low-dose cancer risk relies on a 95% upper bound on the cancer risk that typically only addresses experimental variability in homogeneous laboratory animals. The NRC (2009) observed that when cancer risk is expected to be linear at low exposures, as with benzo[a]pyrene, EPA's cancer risk values tend not to address human variability and susceptibility adequately. Concern for sensitive populations (separate from the consideration of increased sensitivity at early lifestages; see Section 2.6, Application of Age-Dependent Adjustment Factors) suggests interpreting the near-continuous range of risk-estimate confidence intervals from the three data sets (see confidence intervals in Tables E-27 and E-28), of 0 to 1.4 per mg/kg-day, to represent a more heterogeneous population and supports use of the high value as a plausible upper bound.

Potential for model uncertainty in the slope factor estimate is also a relevant consideration at this stage. Although EPA's practice has been to rely on multistage models (including the multistage-Weibull model) for carcinogens with a mutagenic mode of action and expected low-dose linear behavior, some model uncertainty was evaluated by applying the range of dichotomous models in BMDS to the B6C3F_1 mice data (Beland and Culp, 1998), after adjustment for intercurrent mortality using the poly-3 approach (Bailer and Portier, 1988). Even including less plausible models that impose non-linear low-dose behavior that is inconsistent with a mutagenic mode of action—i.e., models fit with a slope of 0 risk/dose unit as doses decrease to 0—for each data set the resulting BMD_{10}s and $BMDL_{10}$s were found to encompass the corresponding multistage-Weibull estimate, and to vary overall less than twofold, and less than a factor of 1.5 from the multistage-Weibull estimate (see Appendix E.2.1). Model uncertainty is minimized through the point of departure being near the lowest exposure in each of the data sets.

Given these considerations, EPA selected the most sensitive result to derive the oral slope factor. The slope factor for assessing human cancer risk associated with lifetime oral exposure to benzo[a]pyrene is **1 per mg/kg-day**, based on the alimentary tract tumor response in female B6C3F_1 mice. Note that the oral slope factor should only be used with lifetime human exposures <0.1 mg/kg-day, because above this level, the dose-response relationship is nonlinear and plateaus at 100% at higher exposures. If risk estimates for exposure above 0.1 mg/kg-day would be needed—that is, corresponding to expected overall cancer risks greater than 10%—the full dose-response model as provided in Appendix E.2.1 should be consulted.

The OSF for benzo[a]pyrene is derived with the intention that it will be paired with EPA's relative potency factors (RPFs) for the assessment of the carcinogenicity of PAH mixtures. In addition, regarding the assessment of early life exposures, because cancer risk values calculated for benzo[a]pyrene were derived from adult animal exposures, and because benzo[a]pyrene carcinogenicity occurs via a mutagenic mode of action, exposures which occur during development should include the application of age-dependent-adjustment-factors (see Section 2.6).

EPA's Guidelines for Carcinogen Risk Assessment (U.S. EPA, 2005a) recommend characterizing expected or central estimates of risk, where practicable, and confidence limits on the POD. For the available data, note that a central estimate of risk below the POD is not practicable because the POD determines the low end of the range for which statistical predictions can be supported. The slope of the linear extrapolation from the BMD_{10}, the central tendency estimate at the POD, can be calculated [0.1/(0.127 mg/kg-day) = 0.78 per mg/kg-day], but cannot be considered a central tendency estimate throughout the low dose range (below the POD). For the recommended slope factor, the 95% confidence limits based on the POD are 0.071–0.179 mg/kg-day.

2.3.4. Uncertainties in the Derivation of the Oral Slope Factor

The oral slope factor for benzo[a]pyrene was based on the increased incidence of alimentary tract tumors, including forestomach tumors, observed in a lifetime dietary study in mice (Beland and Culp, 1998). Although humans do not have a forestomach, forestomach effects observed in rodents are believed to be qualitatively supportive of a human hazard, as humans have similar squamous epithelial tissue in their oral cavity (IARC, 2003; Wester and Kroes, 1988). EPA has considered the uncertainty associated with the relevance of forestomach tumors for quantitatively estimating human risk from benzo[a]pyrene exposure. The rodent forestomach serves to store foods and liquids for several hours before contents continue to the stomach for further digestion (Clayson et al., 1990; Grice et al., 1986). Thus, tissue of the forestomach in rodents may be exposed to benzo[a]pyrene for longer durations than analogous human tissues in the oral cavity and esophagus. This suggests that the rodent forestomach may be quantitatively more sensitive to the development of squamous epithelial tumors in the forestomach compared to oral or esophageal tumors in humans.

There appears to be no biological basis for concluding that the mouse study is more representative of human response than the rat study. However, there is likely greater uncertainty in the rat results, owing to administration of benzo[a]pyrene by gavage in soy bean oil as compared with dietary exposure for the mice. Since BaP is a lipophilic compound it will partition differently between the oil and stomach in this case than if it were administered through water. From studies with other lipophilic compounds administered via corn oil gavage in rats, it is known that this increases lymphatic uptake, and consequently, delayed systemic delivery of the compound (Reddy et al., 2005). Furthermore, a bolus gavage dose leads to higher peak concentration compared to ingestion through food or water over the course of a day, which potentially results in nonlinearities due to metabolic saturation. Corn oil gavage is also thought to result in increased tumor promoting ability based on a study in B6C3F1 mice (Klaunig et al., 1990). Because these effects are related to the lipophilicity of the compound, similar results may be expected of soy oil gavage.

Uncertainty in the magnitude of the recommended oral slope factor is reflected to some extent in the range of slope factors among tumors sites and species; the oral slope factor based on the mouse alimentary tract data was about threefold higher than the overall oral slope factor based

on male rat data (Table 2-8). These comparisons show that the selection of target organ, animal species, and interspecies extrapolation can impact the oral cancer risk estimate. However, all of the activation pathways implicated in benzo[a]pyrene carcinogenicity have been observed in human tissues, and associations have been made between the spectra of mutations in tumor tissues from benzo[a]pyrene-exposed animals and humans exposed to complex PAH mixtures containing benzo[a]pyrene (see Section 1.1.5).

Table 2-8. Summary of uncertainties in the derivation of benzo[a]pyrene oral slope factor

Consideration and impact on cancer risk value	Decision	Justification and discussion
Selection of target organ ↓ oral slope factor, up to fivefold, if alimentary tract tumors not selected	Alimentary tract tumors (forestomach, esophagus, tongue, larynx)	Tumor site is concordant across rats and mice, increasing support for its relevance to humans. As there are no data to support any one result as most relevant for extrapolating to humans, the most sensitive result for alimentary tract tumors was used to derive the oral slope factor.
Selection of data set ↓ oral slope factor ~threefold if rat bioassay were selected for oral slope factor derivation	Beland and Culp (1998)	Beland and Culp (1998) was a well-conducted study and had the lowest HEDs of the available cancer bioassays, reducing low-dose extrapolation uncertainty.
Selection of dose metric Alternatives could ↓ or ↑ slope factor	Administered dose	Experimental evidence supports a role for metabolism in toxicity, but actual responsible metabolites have not been identified.
Interspecies extrapolation Alternatives could ↓ or ↑ slope factor (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 area under the curve, $BW^{3/4}$ scaling was used to calculate equivalent cumulative exposures for estimating equivalent human risks. While the true human correspondence is unknown, this overall approach is expected neither to over- nor underestimate human equivalent risks.
Dose-response modeling Alternative models considered, including nonlinear low dose models incompatible with mutagenic mode of action, yielded slope factors which ranged up to 1.5-fold higher than the slope factor	Multistage-Weibull model	No biologically based models for benzo[a]pyrene were available. Because the multistage-Weibull model could address additional available data (time of death with tumor, and whether a tumor caused the death of the animal), this model was superior to other available models.
Low-dose extrapolation ↓ cancer risk estimate would be expected with the application of nonlinear low-dose extrapolation	Linear extrapolation from POD (based on mutagenic mode of action)	Available mode-of-action data support linearity (mutagenicity is a primary mode of action of benzo[a]pyrene).

Consideration and impact on cancer risk value	Decision	Justification and discussion
Statistical uncertainty at POD ↓ oral slope factor 1.8-fold if BMD used as the POD rather than BMDL	BMDL (preferred approach for calculating plausible upper bound slope factor)	Limited size of bioassay results in sampling variability; lower bound is 95% CI on administered exposure at 10% extra risk of alimentary tract tumors.
Sensitive subpopulations ↑ oral slope factor to unknown extent	ADAFs are recommended for early life exposures	No chemical-specific data are available to determine the range of human toxicodynamic variability or sensitivity.

2.3.5. Previous IRIS Assessment: Oral Slope Factor

The previous cancer assessment for benzo[a]pyrene was posted on the IRIS database in 1992. At that time, benzo[a]pyrene was classified as a probable human carcinogen (Group B2) based on inadequate data in humans and sufficient data in animals via several routes of exposure.

Four slope factors were estimated based on studies of dietary benzo[a]pyrene administered either for approximately 2 years in 10-week-old Sprague-Dawley rats (Brune et al., 1981) or for up 7 months in 2-week-old to 5-month-old CFW-Swiss mice, sex unknown (Neal and Rigdon, 1967). Each slope factor reflected extrapolation to humans assuming surface area equivalence ($BW^{2/3}$ scaling), for a twofold increase in estimated risk relative to EPA's subsequent update to $BW^{3/4}$ scaling (U.S. EPA, 1992). A slope factor estimate of 11.7 per mg/kg-day, using a linearized multistage procedure applied to the combined incidence of forestomach, esophageal, and laryngeal tumors, was derived from the Brune et al. (1981) study (see Section 1.1.5 for study details).

Three modeling procedures were used to derive risk estimates from the Neal and Rigdon (1967) bioassay (see Section 1.1.5), resulting in a two-fold range of risk estimates from the same data set:

1) U.S. EPA (1991a) fit a two-stage response model, based on exposure-dependent changes in both transition rates and growth rates of preneoplastic cells, to derive a value of 5.9 per mg/kg-day.
2) U.S. EPA (1991b) derived a value of 9.0 per mg/kg-day by linear extrapolation from the 10% response point to the background response in a re-analysis of the 1990 model.
3) Using a Weibull-type model to reflect less-than-lifetime exposure to benzo[a]pyrene, the U.S. EPA (1991b) assessment derived an upper-bound slope factor estimate of 4.5 per mg/kg-day.

The four slope factor estimates, within threefold of each other, were judged in 1992 to be of equal merit, although based on less-than-optimal datasets. The geometric mean of these four estimates, 7.3 per mg/kg-day, was previously recommended as the oral slope factor.

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 is a plausible upper bound on the estimate of risk per μg/m^3 air breathed for a lifetime.

2.4.1. Analysis of Carcinogenicity Data

The inhalation database demonstrating carcinogenicity of benzo[a]pyrene consists of a lifetime inhalation bioassay in male hamsters (Thyssen et al., 1981) and intratracheal instillation studies, also in hamsters (Feron and Kruysse, 1978; Ketkar et al., 1978; Feron et al., 1973; Henry et al., 1973; Saffiotti et al., 1972). The intratracheal instillation studies provide supporting evidence of carcinogenicity of inhaled benzo[a]pyrene; however, the use of this exposure method alters the deposition, clearance, and retention of substances, and therefore, studies utilizing this exposure technique are not as useful for the quantitative extrapolation of cancer risk from the inhalation of benzo[a]pyrene in the environment (Driscoll et al., 2000).

The bioassay by Thyssen et al. (1981) represents the only lifetime inhalation cancer bioassay available for describing exposure-response relationships for cancer from inhaled benzo[a]pyrene. As summarized in Section 1.1.5, increased incidences of benign and malignant tumors of the pharynx, larynx, trachea, esophagus, nasal cavity, or forestomach were seen with increasing exposure concentration. In addition, survival was decreased relative to control in the high-exposure group; mean survival times in the control, low-, and mid-concentration groups were 96.4, 95.2, and 96.4 weeks, respectively, compared to 59.5 weeks in the high-exposure group animals (Thyssen et al., 1981). Overall, tumors occurred earlier in the highest benzo[a]pyrene exposure group than in the mid-exposure group.

Strengths of the study included exposures until natural death, up to 2.5 years, multiple exposure groups; histological examination of multiple organ systems, and availability of individual animal pathology reports with time of death and tumor incidence data by site in the upper respiratory and digestive tracts. In addition, the availability of weekly chamber air monitoring data and individual times on study allowed the calculation of time-weighted average (TWA) lifetime continuous exposures for each hamster. Group averages of these TWA concentrations were 0, 0.25, 1.01, and 4.29 mg/m^3 (U.S. EPA, 1990).

Several limitations concerning exposure conditions in the Thyssen et al. (1981) study were evaluated for their impact on the derivation of an inhalation unit risk for benzo[a]pyrene. These issues include minimal detail about the particle size distribution of the administered aerosols, variability of chamber concentrations, and the use of a sodium chloride aerosol as a carrier.

First, particle distribution analysis of aerosols, in particular the MMAD and geometric SD, was not reported, although the investigators did report that particles were within the respirable

range for hamsters, with >99% of the particles having diameters 0.2–0.5 μm and >80% having diameters 0.2–0.3 μm.

Second, weekly averages of chamber concentration measurements varied two- to fivefold from the overall average for each group, which exceeds the limit for exposure variability of <20% for aerosols recommended by OECD (2009). For risk assessment purposes, EPA generally assumes that cancer risk is proportional to cumulative exposure, and therefore to lifetime average exposure as estimated here, when there is no information to the contrary. Under this assumption, the variability of the chamber concentrations has little impact on the estimated exposure-response relationship. The impacts of alternative assumptions are considered in Section 2.4.4.

Lastly, exposure occurred through the inhalation of benzo[a]pyrene adsorbed onto sodium chloride aerosols, which might have irritant carrier effects, and will have a different deposition than benzo[a]pyrene adsorbed onto carbonaceous particles (as is more typical in the environment). The above study design and reporting issues concerning the particle size composition, exposure variability, and deposition do not negate the robust tumor response following benzo[a]pyrene inhalation exposure. Consequently, EPA concluded that the strengths of the study supported the use of the data to derive an inhalation unit risk for benzo[a]pyrene. See Section 2.4.4 for a discussion of uncertainties in the unit risk.

2.4.2. Dose-Response Analysis—Adjustments and Extrapolation Methods

Biologically based dose-response models for benzo[a]pyrene are not available. A simplified version of the two-stage carcinogenesis model proposed by Moolgavkar and Venzon (1979) and Moolgavkar and Knudson (1981) has been applied to the Thyssen et al. (1981) individual animal data (U.S. EPA, 1990). However, the simplifications necessary to fit the tumor incidence data reduced that model to an empirical model (i.e., there were no biological data to inform estimates of cell proliferation rates for background or initiated cells, which are generally very sensitive parameters in these cancer models). Sufficient data were available to apply the multistage-Weibull model, as used for the oral slope factor (described in detail in Appendix E of the Supplemental Information), specifically the individual times of death for each animal. Unlike in the oral bioassays, Thyssen et al. (1981) did not determine cause of death for any of the animals. Since the investigators for the oral bioassays considered some of the same tumor types to be fatal at least some of the time, bounding estimates of the POD for these Thyssen et al. (1981) data were developed by treating the tumors alternately as either all incidental to the death of an affected animal or as causing the death of an affected animal.

The tumor incidence data used for dose-response modeling comprised the benign and malignant tumors in the pharynx and respiratory tract (see Table E-27). The tumors in these sites were judged to be sufficiently similar to combine in overall incidences, based on the assumption that the benign tumors could develop into malignancies, as outlined in EPA's *Guidelines for Carcinogen Risk Assessment* (Section 2.2.2.1.2; U.S. EPA, 2005a). Specifically, while the pharynx and larynx are associated with the upper digestive tract and the upper respiratory tract, respectively,

these sites are close anatomically and in some cases where both tissues were affected, the site of origin could not be distinguished (U.S. EPA, 1990). In addition, the benign tumors (e.g., papillomas, polyps, and papillary polyps) were considered early stages of the SCCs in these tissues (U.S. EPA, 1990). Consequently, the overall incidence of SCCs or benign tumors judged to originate from the same cell type (papillomas, polyps, or papillary polyps) were selected for dose-response modeling.

A toxicokinetic model to assist in cross-species scaling of benzo[a]pyrene inhalation exposure was not available. EPA's RfC default dosimetry adjustments (U.S. EPA, 1994a) were utilized in the benzo[a]pyrene RfC calculation (see Section 2.2.2) but could not be applied to the aerosols generated for the inhalation bioassay by Thyssen et al. (1981) as the approaches presented in the RfC methodology guidelines (U.S. EPA, 1994a) were developed for insoluble and nonhygroscopic particles, not the sodium chloride particle used in Thyssen et al. (1981). Further, relative surface areas of the upper respiratory tract in hamsters and humans were not considered relevant because hamsters and humans are not necessarily expected to respond in the same areas of the respiratory tract. Consequently, without data to inform a basis for extrapolation to humans, it was assumed that equal risk for all species would be associated with equal concentrations in air, at least at anticipated environmental concentrations, as would be the case for a soluble gas. This is equivalent to assuming that breathing rate and metabolism of benzo[a]pyrene to DNA-reactive metabolites both scale across species in proportion to $BW^{3/4}$.

The multistage-Weibull model was fit to the TWA exposure concentrations and the individual animal tumor and survival data for tumors in the larynx, pharynx, trachea, or nasal cavity (tumors of the pharynx and upper respiratory tract), using the software program, MultiStage-Weibull (U.S. EPA, 2010b). Modeling results are provided in Appendix E.2 of the Supplemental Information. Because benzo[a]pyrene carcinogenicity involves a mutagenic mode of action, linear low-exposure extrapolation from the $BMCL_{10}$ was used to derive the inhalation unit risk (U.S. EPA, 2005a).

2.4.3. Inhalation Unit Risk Derivation

The results from modeling the inhalation carcinogenicity data from Thyssen et al. (1981) are summarized in Table 2-9. Taking the tumors to have been the cause of death for the experimental animals with tumors, the BMC_{10} and $BMCL_{10}$ values were 0.468 and 0.256 mg/m^3, respectively. Then, taking all of the tumors to have been incidental to the cause of death for each animal with a tumor, the BMC_{10} and $BMCL_{10}$ values were 0.254 and 0.163 mg/m^3, respectively, about twofold lower than the first case. Because the tumors were unlikely to have all been fatal, and cancer risk estimation focuses on cancer incidence rather than death from cancer, the lower $BMCL_{10}$ from the incidental deaths analysis, 0.163 mg/m^3, is recommended for the calculation of the inhalation unit risk.

Using linear extrapolation from the $BMCL_{10}$ (0.163 mg/m^3), an inhalation unit risk of **0.6 per mg/m^3**, or **6×10^{-4} per µg/m^3** (rounding to one significant digit), was calculated. Note that the inhalation unit risk should only be used with lifetime human exposures <0.3 mg/m^3, the

human equivalent POD, because above this level, the dose-response relationship is nonlinear and plateaus at 100% at higher exposures. If risk estimates are needed for exposure above 0.3 mg/m^3—that is, corresponding to overall cancer risks >10%—the full dose-response model as provided in Appendix E.2.2 should be consulted.

The IUR for benzo[a]pyrene is derived with the intention that it will be paired with EPA's relative potency factors (RPFs) for the assessment of the carcinogenicity of PAH mixtures. In addition, regarding the assessment of early life exposures, because cancer risk values calculated for benzo[a]pyrene were derived from adult animal exposures, and because benzo[a]pyrene carcinogenicity occurs via a mutagenic mode of action, exposures which occur during development should include the the application of age-dependent-adjustment-factors (see Section 2.6).

EPA's Guidelines for Carcinogen Risk Assessment (U.S. EPA, 2005a) recommend characterizing expected or central estimates of risk where practicable, and confidence limits on the POD. For the available data, note that a central estimate of risk below the POD is not practicable because the POD determines the low end of the range for which statistical predictions can be supported. The slope of the linear extrapolation from the BMD_{10}, the central tendency estimate at the POD, can be calculated [0.1/(0. 254 mg/m3) = 0.39 per mg/m^3], but cannot be considered a central tendency estimate throughout the low dose range (below the POD). For the recommended slope factor, the 95% confidence limits based on the POD are 0.163–0.324 mg/m^3.

Table 2-9. Summary of the inhalation unit risk derivation

Tumor site and context	Species/ sex	Selected model	BMR	BMC (mg/m^3)	POD = BMCL (mg/m^3)	Unit risk[a] (mg/m^3)$^{-1}$
Upper respiratory tract and pharynx; all treated as cause of death Thyssen et al. (1981)	Male hamsters	Multistage Weibull, 2°	10%	0.468	0.256	0.4
Upper respiratory tract and pharynx; all treated as incidental to death Thyssen et al. (1981)	Male hamsters	Multistage Weibull, 2°	10%	0.254	0.163	0.6

[a]Human equivalent unit risk = 0.10/$BMCL_{10}$; see Appendix E for details of modeling results.

2.4.4. Uncertainties in the Derivation of the Inhalation Unit Risk

Table 2-10 summarizes uncertainties in the derivation of the inhalation unit risk for benzo[a]pyrene; further detail is provided in the following discussion. Only one animal cancer bioassay, in one sex, by the inhalation route is available that describes the exposure-response relationship for respiratory tract tumors with lifetime inhalation exposure to benzo[a]pyrene (Thyssen et al., 1981). Although corroborative information on exposure-response relationships in other animal species is lacking, the findings for upper respiratory tract tumors are consistent with

findings in other hamster studies with intratracheal administration of benzo[a]pyrene (upper and lower respiratory tract tumors), and with some of the portal-of-entry effects in oral exposure studies.

The hamster inhalation bioassay by Thyssen et al. (1981) observed upper respiratory tract tumors, but not lung tumors. The lack of a lung tumor response in hamsters, given the strong association of inhaled PAH mixtures with lung cancer in humans across many studies (see Section 1.1.5) suggests that this study may not be ideal for extrapolating to humans. Hamsters have an apparent lower sensitivity to lung carcinogenesis than rats and mice and a tendency to give false negatives for particles classified as carcinogenic to humans by IARC (Mauderly, 1997). However, hamster laryngeal tumors have been used as an indication of the carcinogenic hazard of cigarette smoke for more than 50 years (IARC, 2002). For example, a large study investigating the inhalation of cigarette smoke in hamsters (n = 4,400) indicated that the larynx was the most responsive tumor site, which the authors indicated was due to a large difference in particle deposition between the larynx and the lung (Dontenwill et al., 1973). EPA's *Guidelines for Carcinogen Assessment* (U.S. EPA, 2005a) stress that site concordance between animals and humans need not always be assumed. Therefore, the robust tumor response in the upper respiratory tract of Syrian golden hamsters was considered to be supportive of the use of the Thyssen et al. (1981) study for the derivation of an inhalation unit risk.

Data from the Thyssen et al. (1981) study were incomplete; histopathology reports were completely missing for four hamsters in the mid-dose group and for single tissues in 21 other hamsters over all four groups. The recommended unit risk (0.6 per mg/m^3) omitted these animals altogether, as if they had never been on study. A reanalysis including the animals with partial histopathology, and assuming no tumors among them, yielded a unit risk of 0.5 per mg/m^3, about 20% lower (see Table E-32).

Additional sensitivity analyses included using other dose-response models and different latency assumptions in the multistage-Weibull model. BMDS dichotomous dose-response models were applied to poly-3 adjusted incidence data to address intercurrent mortality (see Table E-32). These adjusted estimates also considered the length of time on study for the animals with incomplete histopathology. None of these models provided adequate fits. Dropping the high dose led to a better fit to the low exposure region and an adequate overall fit, with a BMD_{10} and $BMDL_{10}$ about 20% higher than for the recommended unit risk.

Alternative assumptions for latency in the time-to-tumor model were more ad hoc, due to lack of information in the scientific literature for respiratory tumors and lack of cause-of-death information in the Thyssen et al (1981) data set. One approach involved judging which tumors were or were not the cause of death. For the purpose of this analysis it was assumed that benign tumors did not cause death and that all malignant tumors were the cause of death; this approach yielded a latency estimate of 14.5 weeks (see Table E-32). However, there were only 5 benign tumors, all in the mid-exposure group; even if this is an accurate accounting of the tumors'

involvement in cause of death, it is not a strong basis for estimating this parameter for the 25 cases observed in the study. Another approach involved fixing latency at a range of values, 2 through 90 weeks; these model versions tended not to fit the data as well as the recommended fit to the incidental tumor data, as shown by larger AIC values (see Table E-32). BMD_{10}s and $BMDL_{10}$s were higher for the lower four latency values assumed, but were very similar to the recommended POD when latency was set at 90 weeks. These results suggest some insensitivity to latency, or possibly that a constant value for latency across exposure levels is not supported.

An additional uncertainty includes the inability to apply U.S. EPA (1994a) dosimetry approaches to extrapolate inhaled concentrations from hamsters to humans, due to the use of a soluble hygroscopic carrier particle (sodium chloride) for the delivery of benzo[a]pyrene. One likely consequence of the use of hygroscopic carrier particles would be the growth of benzo[a]pyrene-sodium chloride particles in the humid environment of the respiratory tract resulting in increased particle diameter and resulting changes in particle deposition, specifically, increased impaction in the upper respiratory tract and less deposition in the lung (Varghese and Gangamma, 2009; Asgharian, 2004; Ferron, 1994; Xu and Yu, 1985). In addition, sodium chloride can be irritating to the respiratory tract, depending on concentration. The Thyssen et al. (1981) study reported that vehicle controls were exposed to 240 $\mu g/m^3$, and it is unclear whether exposure to this concentration of sodium chloride could have potentiated the tumor response seen in the mid- and high-concentration benzo[a]pyrene groups. Exposure to benzo[a]pyrene in the environment predominantly occurs via non-soluble, non-hygroscopic, carbonaceous particles (such as soot and diesel exhaust particles). The potential impact of differences in carrier particle on the magnitude of the inhalation unit risk is unknown.

Extrapolation of risk from hamsters to humans may be informed by considering the equivalent inhaled dose in each species. Using default breathing rates for hamsters and humans of 0.063 and 20 m^3/day (U.S. EPA (1994a), and body weights of 0.1 kg (Thyssen et al., 1981) and 70 kg, the equivalent daily inhaled doses, normalized by body weight, corresponding to the point of departure of 0.61 mg/m^3 are 0.10 and 0.046 mg/kg-day, respectively. This level of agreement is roughly supports assuming equal risk at equal concentrations in air. Alternatives comprised consideration of scaling inhaled doses, in mg/kg-day units, by $bodyweight^{3/4}$ (highlighting allometric differences in metabolism and clearance rates over their lifetimes) and by $bodyweight^{2/3}$ (highlighting species differences proportional to relative surface areas). Both considerations suggest higher risks to humans than to hamsters at the same exposure level, by about fivefold and eightfold, respectively.

Regarding uncertainty associated with exposure characterization, the individual exposure chamber measurements varied from about an order of magnitude less than the target concentration to about twofold higher than the target concentration. Weekly average analytical concentrations were documented to vary by two- to fivefold in all exposed groups, with no particular trends over time. Continuous time-weighted group average concentrations were used for dose-response

modeling under the assumption that equal cumulative exposures are expected to lead to similar outcomes. This assumption is generally expected to lead to an unbiased estimate of risk when there is incomplete information. However, it is possible that peak exposure above some concentration may be more associated with the observed effects, or that deposition of particles may have reached a maximum level or plateau, such as in the high-exposure group. Regarding the role of peak exposures, the higher exposures for each group were distributed evenly throughout the study for the most part, suggesting that any association of risk with peak exposures would also be proportional to cumulative exposure. If particle deposition reached a plateau with the high-exposure group, there is relatively less impact on the unit risk because the derivation relies on the dose-response at lower exposure. But the actual dynamics of particle deposition at these or other exposure levels are not well understood. There is not enough information available to estimate a more quantitative impact on the estimated unit risk due to these uncertainties.

Table 2-10. Summary of uncertainties in the derivation of cancer risk values for benzo[a]pyrene (inhalation unit risk)

Consideration and impact on cancer risk value	Decision	Justification and discussion
Selection of data set and target organ No inhalation unit risk if Thyssen et al. (1981) not used	Respiratory tract tumors from Thyssen et al. (1981)	The Thyssen et al. (1981) bioassay is the only lifetime inhalation cancer bioassay available for describing exposure-response relationships for cancer from inhaled benzo[a]pyrene. Intratracheal installation studies support the association of benzo[a]pyrene exposure with respiratory tract tumors.
Selection of dose metric Alternatives could ↓ or ↑ unit risk	Administered exposure as TWA	Experimental evidence supports a role for metabolism in toxicity, but actual responsible metabolites are not identified. The recommended unit risk is a reasonable estimate if the proportion of the carcinogenic moiety remains the same at lower exposures.
Interspecies extrapolation Alternatives would ↑ ($mg/kg^{2/3}$, $mg/kg^{3/4}$) unit risk 5- to 8-fold	Equal risk per $\mu g/m^3$ (ppm-equivalence) is assumed. The carrier particle used was soluble and hygroscopic, therefore the RfC methodology (U.S. EPA, 1994a) dosimetric adjustments could not be applied.	There are no data to support alternatives. Equal risk per $\mu g/m^3$ is equivalent to assuming that intake scales with $BW^{3/4}$. It does not account for the rate of production of DNA-reactive metabolites in the affected tissues, which are likely to be proportional to $BW^{3/4}$
Dose-response modeling Alternatives could ↓ or ↑ unit risk	Multistage-Weibull model	No biologically based models for benzo[a]pyrene were available. Because the multistage-Weibull model could address additional available data

Consideration and impact on cancer risk value	Decision	Justification and discussion
		(time of death with tumor), this model was superior to other available empirical models.
Low-dose extrapolation ↓ cancer risk estimate would be expected with the application of nonlinear low-dose extrapolation	Linear extrapolation from the POD (based on mutagenic mode of action)	Available mode-of-action data support linearity (mutagenicity is a primary mode of action of benzo[a]pyrene).
Statistical uncertainty at POD ↓ inhalation unit risk 1.4-fold if BMC used as the POD rather than BMCL	BMCL (preferred approach for calculating plausible upper bound unit risk)	Limited size of bioassay results in sampling variability; lower bound is 95% confidence interval (CI) on administered exposure at 10% extra risk of respiratory tract tumors.
Sensitive subpopulations ↑ inhalation unit risk to unknown extent	ADAFs are recommended for early life exposures	No chemical-specific data are available to determine the range of human toxicodynamic variability or sensitivity.

2.4.5. Previous IRIS Assessment: Inhalation Unit Risk

An inhalation unit risk for benzo[a]pyrene was not previously available on IRIS.

2.5. APPLICATION OF AGE-DEPENDENT ADJUSTMENT FACTORS (ADAFs)

Based on sufficient support in laboratory animals and relevance to humans, benzo[a]pyrene is determined to be carcinogenic by a mutagenic mode of action. According to the *Supplemental Guidance for Assessing Susceptibility from Early Life Exposure to Carcinogens* ("*Supplemental Guidance*") (U.S. EPA, 2005b), individuals exposed during early life to carcinogens with a mutagenic mode of action are assumed to have increased risk for cancer. The oral slope factor of 1 per mg/kg-day, and inhalation unit risk of 0.6 per mg/m^3 for benzo[a]pyrene, calculated from data applicable to adult exposures, do not reflect presumed early life susceptibility to this chemical. Although chemical-specific data exist for benzo[a]pyrene that quantitatively demonstrate increased early life susceptibility to cancer (Vesselinovitch et al., 1975), these data were not considered sufficient to develop separate risk estimates for childhood exposure, as they used acute i.p. exposures (U.S. EPA, 2005b). In the absence of adequate chemical-specific data to evaluate differences in age-specific susceptibility, the *Supplemental Guidance* (U.S. EPA, 2005b) recommends that ADAFs be applied in estimating cancer risk.

The *Supplemental Guidance* (U.S. EPA, 2005b) establishes ADAFs for three specific age groups. These ADAFs and their corresponding age groupings are: 10 for individuals exposed at <2 years of age, 3 for exposed individuals at 2–<16 years of age, and 1 for exposed individuals ≥16 years of age. The 10- and 3-fold adjustments are combined with age-specific exposure estimates when estimating cancer risks from early life (<16 years of age) exposures to benzo[a]pyrene. To illustrate the use of the ADAFs established in the *Supplemental Guidance* (U.S.

EPA, 2005b), sample calculations are presented for three exposure duration scenarios, including full lifetime, assuming a constant benzo[a]pyrene exposure of 0.001 mg/kg-day (Table 2-11).

Table 2-11. Sample application of ADAFs for the estimation of benzo[a]pyrene cancer risk following lifetime (70-year) oral exposure

Age group	ADAF	Unit risk (per mg/kg-d)	Sample exposure concentration (mg/kg-d)	Duration adjustment	Cancer risk for age-specific exposure period
0–<2 yrs	10	1	0.001	2 yrs/70 yrs	0.0003
2–<16 yrs	3	1	0.001	14 yrs/70 yrs	0.0006
≥16 yrs	1	1	0.001	54 yrs/70 yrs	0.0008
Total risk					0.002

The example exposure duration scenarios include full lifetime exposure (assuming a 70-year lifespan). Table 2-11 lists the four factors (ADAFs, cancer risk estimate, assumed exposure, and duration adjustment) that are needed to calculate the age-specific cancer risk based on the early age-specific group. The cancer risk for each age group is the product of the four factors in columns 2–5. Therefore, the cancer risk following daily benzo[a]pyrene oral exposure in the age group 0–<2 years is the product of the values in columns 2–5 or $10 \times 1 \times 0.001 \times 2/70 = 3 \times 10^{-4}$. The cancer risk for specific exposure duration scenarios that are listed in the last column are added together to get the total risk. Thus, a 70-year (lifetime) risk estimate for continuous exposure to 0.001 mg/kg-day benzo[a]pyrene is 2×10^{-3}, which is adjusted for early-life susceptibility and assumes a 70-year lifetime and constant exposure across age groups.

In calculating the cancer risk for a 30-year constant exposure to benzo[a]pyrene at an exposure level of 0.001 mg/kg-day for ages 0–30 years, the duration adjustments would be 2/70, 14/70, and 54/70, and the age-specific risks for the three age groups would be 3×10^{-4}, 6×10^{-4}, and 2×10^{-4}, which would result in a total risk estimate of 1×10^{-3}.

In calculating the cancer risk for a 30-year constant exposure to benzo[a]pyrene at an exposure level of 0.001 mg/kg-day for ages 20–50 years, the duration adjustments would be 0/70, 0/70, and 30/70. The age-specific risks for the three groups are 0, 0, and 4×10^{-4}, which would result in a total risk estimate of 4×10^{-4}.

Consistent with the approaches for the oral route of exposure (Table 2-11), the ADAFs should also be applied when assessing cancer risks for subpopulations with early life exposures to benzo[a]pyrene via the inhalation route (presented in Table 2-12).

Table 2-12. Sample application of ADAFs for the estimation of benzo[a]pyrene cancer risk following lifetime (70-year) inhalation exposure

Age group	ADAF	Unit risk (per µg/m^3)	Sample exposure concentration (µg/m^3)	Duration adjustment	Cancer risk for age-specific exposure period
0–<2 yrs	10	6×10^{-4}	0.1	2 yrs/70 yrs	0.00002
2–<16 yrs	3	6×10^{-4}	0.1	14 yrs/70 yrs	0.00004
≥16 yrs	1	6×10^{-4}	0.1	54 yrs/70 yrs	0.00005
Total risk					0.00010

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