IPCS

INTERNATIONAL PROGRAMME ON CHEMICAL SAFETY

UNEP

Environmental Health Criteria 202

Selected Non-heterocyclic Polycyclic Aromatic Hydrocarbons



IOMC INTER-ORGANIZATION PROGRAMME FOR THE SOUND MANAGEMENT OF CHEMICALS
A cooperative agreement among UNEP, ILO, FAO, WHO, UNIDO, UNITAR and OECD

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Environmental Health Criteria 202

SELECTED NON-HETEROCYCLIC POLYCYCLIC AROMATIC HYDROCARBONS

First and second drafts prepared by staff members at the Fraunhofer Institute of Toxicology and Aerosol Research, Hanover, Germany, under the coordination of Dr R.F. Hertel, Dr G. Rosner, and Dr J. Kielhorn, in cooperation with Dr E. Menichini, Italy, Dr P.L. Grover, United Kingdom, and Dr J. Blok, Netherlands. Dr P. Muller, Canada, and Dr R. Schoeny and Dr T.L. Mumford, USA, prepared and revised the drafts of Appendix I.

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Geneva, 1998

The **International Programme on Chemical Safety** (IPCS), established in 1980, is a joint venture of the United Nations Environment Programme (UNEP), the International Labour Organisation (ILO), and the World Health Organization (WHO). The overall objectives of the IPCS are to establish the scientific basis for assessment of the risk to human health and the environment from exposure to chemicals, through international peer review processes, as a prerequisite for the promotion of chemical safety, and to provide technical assistance in strengthening national capacities for the sound management of chemicals.

The **Inter-Organization Programme for the Sound Management of Chemicals** (IOMC) was established in 1995 by UNEP, ILO, the Food and Agriculture Organization of the United Nations, WHO, the United Nations Industrial Development Organization, the United Nations Institute for Training and Research, and the Organisation for Economic Co-operation and Development (Participating Organizations), following recommendations made by the 1992 UN Conference on Environment and Development to strengthen cooperation and increase coordination in the field of chemical safety. The purpose of the IOMC is to promote coordination of the policies and activities pursued by the Participating Organizations, jointly or separately, to achieve the sound management of chemicals in relation to human health and the environment.

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NOTE TO READERS OF THE CRITERIA MONOGRAPHS

Every effort has been made to present information in the Criteria monographs as accurately as possible without unduly delaying their publication. In the interest of all users of the Environmental Health Criteria monographs, readers are requested to communicate any errors that may have occurred to the Director of the International Programme on Chemical Safety, World Health Organization, Geneva, Switzerland, in order that they may be included in corrigenda.

* * *

A detailed data profile and a legal file can be obtained from the International Register of Potentially Toxic Chemicals, Case postale 356, 1219 Châtelaine, Geneva, Switzerland (Telephone No. 979 9111).

* * *

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Environmental Health Criteria

PREAMBLE

Objectives

The WHO Environmental Health Criteria Programme was initiated in 1973, with the following objectives:

(i) to assess information on the relationship between exposure to environmental pollutants and human health and to provide guidelines for setting exposure limits;
(ii) to identify new or potential pollutants;
(iii) to identify gaps in knowledge concerning the health effects of pollutants;
(iv) to promote the harmonization of toxicological and epidemiological methods in order to have internationally comparable results.

The first Environmental Health Criteria (EHC) monograph, on mercury, was published in 1976; numerous assessments of chemicals and of physical effects have since been produced. Many EHC monographs have been devoted to toxicological methods, e.g. for genetic, neurotoxic, teratogenic, and nephrotoxic effects. Other publications have been concerned with e.g. epidemiological guidelines, evaluation of short-term tests for carcinogens, biomarkers, and effects on the elderly.

Since the time of its inauguration, the EHC Programme has widened its scope, and the importance of environmental effects has been increasingly emphasized in the total evaluation of chemicals, in addition to their health effects.

The original impetus for the Programme came from resolutions of the World Health Assembly and the recommendations of the 1972 United Nations Conference on the Human Environment. Subsequently, the work became an integral part of the International Programme on Chemical Safety (IPCS), a cooperative programme of UNEP, ILO, and WHO. In this manner, with the strong support of the new partners, the importance of occupational health and environmental effects was fully recognized. The EHC monographs have become widely established, used, and recognized throughout the world.

The recommendations of the 1992 United Nations Conference on Environment and Development and the subsequent establishment of the Intergovernmental Forum on Chemical Safety, with priorities for action in the six programme areas of Chapter 19, Agenda 21, lend further weight to the need for EHC assessments of the risks of chemicals.

Scope

The Criteria monographs are intended to provide critical reviews of the effect on human health and the environment of chemicals, combinations of chemicals, and physical and biological agents. They include reviews of studies that are of direct relevance for the evaluation and do not describe every study that has been carried out. Data obtained worldwide are used, and results are quoted from original studies, not from abstracts or reviews. Both published and unpublished reports are considered, and the authors are responsible for assessing all of the articles cited; however, preference is always given to published data, and unpublished data are used only when relevant published data are absent or when the unpublished data are pivotal to the risk assessment. A detailed policy statement is available that describes the procedures used for citing unpublished proprietary data, so that this information can be used in the evaluation without compromising its confidential nature (WHO, 1990).

In the evaluation of human health risks, sound data on humans, whenever available, are preferred to data on experimental animals. Studies of animals and in-vitro systems provide support and are used mainly to supply evidence missing from human studies. It is mandatory that research on human subjects be conducted in full accord with ethical principles, including the provisions of the Helsinki Declaration.

The EHC monographs are intended to assist national and international authorities in making risk assessments and subsequent risk management decisions. They represent a thorough evaluation of risks and are not in any sense recommendations for regulation or setting standards. The latter are the exclusive purview of national and regional governments.

Content

The layout of EHC monographs for chemicals is outlined below.

- Summary: a review of the salient facts and the risk evaluation of the chemical
- Identity: physical and chemical properties, analytical methods
- Sources of exposure
- Environmental transport, distribution, and transformation
- Environmental levels and human exposure
- Kinetics and metabolism in laboratory animals and humans
- Effects on laboratory mammals and in-vitro test systems
- Effects on humans
- Effects on other organisms in the laboratory and the field
- Evaluation of human health risks and effects on the environment
- Conclusions and recommendations for protection of human health and the environment

- Further research
- Previous evaluations by international bodies, e.g. the International Agency for Research on Cancer, the Joint FAO/WHO Expert Committee on Food Additives, and the Joint FAO/WHO Meeting on Pesticide Residues

Selection of chemicals

Since the inception of the EHC Programme, the IPCS has organized meetings of scientists to establish lists of chemicals that are of priority for subsequent evaluation. Such meetings have been held in Ispra, Italy (1980); Oxford, United Kingdom (1984); Berlin, Germany (1987); and North Carolina, United States of America (1995). The selection of chemicals is based on the following criteria: the existence of scientific evidence that the substance presents a hazard to human health and/or the environment; the existence of evidence that the possible use, persistence, accumulation, or degradation of the substance involves significant human or environmental exposure; the existence of evidence that the populations at risk (both human and other species) and the risks for the environment are of a significant size and nature; there is international concern, i.e. the substance is of major interest to several countries; adequate data are available on the hazards.

If it is proposed to write an EHC monograph on a chemical that is not on the list of priorities, the IPCS Secretariat first consults with the cooperating organizations and the participating institutions.

Procedures

The order of procedures that result in the publication of an EHC monograph is shown in the following flow chart. A designated staff member of IPCS, responsible for the scientific quality of the document, serves as Responsible Officer (RO). The IPCS Editor is responsible for the layout and language. The first draft, prepared by consultants or, more usually, staff at an IPCS participating institution is based initially on data provided from the International Register of Potentially Toxic Chemicals and reference data bases such as Medline and Toxline.

The draft document, when received by the RO, may require an initial review by a small panel of experts to determine its scientific quality and objectivity. Once the RO finds the first draft acceptable, it is distributed in its unedited form to over 150 EHC contact points throughout the world for comment on its completeness and accuracy and, where necessary, to provide additional material. The contact points, usually designated by governments, may be participating institutions, IPCS focal points, or individual scientists known for their particular expertise. Generally, about four months are allowed before the comments are considered by the RO and author(s). A second draft

incorporating the comments received and approved by the Director, IPCS, is then distributed to Task Group members, who carry out a peer review at least six weeks before their meeting.

The Task Group members serve as individual scientists, not as representatives of any organization, government, or industry. Their function is to evaluate the accuracy, significance, and relevance of the information in the document and to assess the risks to health and the environment from exposure to the chemical. A summary and recommendations for further research and improved safety are also drawn up. The composition of the Task Group is dictated by the range of expertise required for the subject of the meeting and by the need for a balanced geographical distribution.

The three cooperating organizations of the IPCS recognize the important role played by nongovernmental organizations, so that representatives from relevant national and international associations may be invited to join the Task Group as observers. While observers may provide valuable contributions to the process, they can speak only at the invitation of the Chairperson. Observers do not participate in the final evaluation of the chemical, which is the sole responsibility of the Task Group members. The Task Group may meet in camera when it considers that to be appropriate.

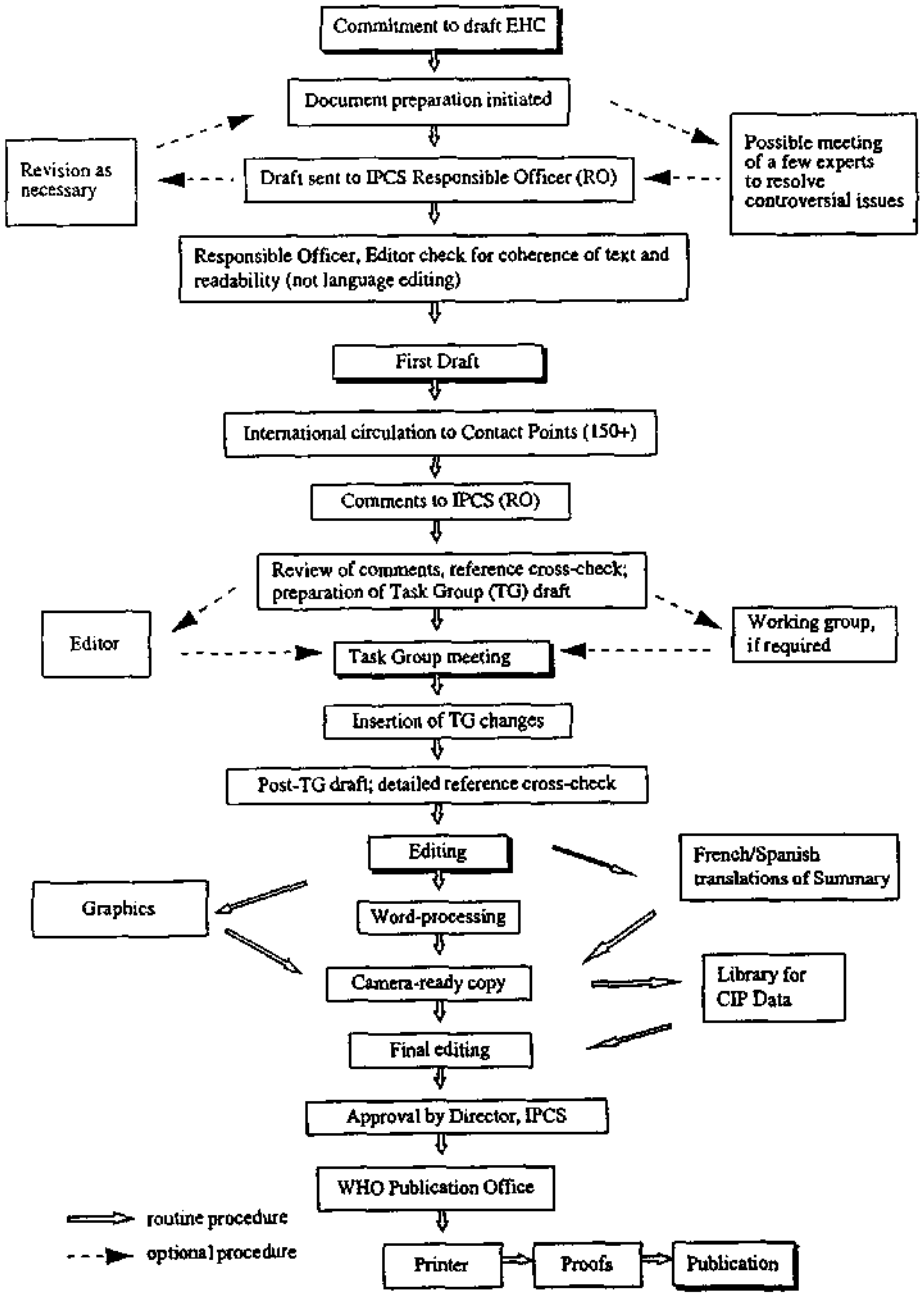
All individuals who participate in the preparation of an EHC monograph as authors, consultants, or advisers must, in addition to serving in their personal capacity as scientists, inform the RO if at any time a conflict of interest, whether actual or potential, could be perceived in their work. They are required to sign a statement to that effect. This procedure ensures the transparency and probity of the process.

When the Task Group has completed its review and the RO is satisfied as to the scientific correctness and completeness of the document, it is edited for language, the references are checked, and camera-ready copy is prepared. After approval by the Director, IPCS, the monograph is submitted to the WHO Office of Publications for printing. At this time, a copy of the final draft is also sent to the Chairperson and Rapporteur of the Task Group to check for any errors.

It is accepted that the following criteria should initiate the updating of an EHC monograph: new data are available that would substantially change the evaluation; there is public concern about health or environmental effects of the agent because of greater exposure; an appreciable time has elapsed since the last evaluation.

All participating institutions are informed, through the EHC progress report, of the authors and institutions proposed for the drafting of the documents. A comprehensive file of all comments received on drafts of each EHC monograph is maintained and is available on request. The chairpersons of task groups are briefed before each meeting on their role and responsibility in ensuring that these rules are followed.

EHC PREPARATION FLOW CHART



WHO TASK GROUP ON ENVIRONMENTAL HEALTH CRITERIA FOR SELECTED NON-HETEROCYCLIC POLYCYCLIC AROMATIC HYDROCARBONS
Hanover, Germany, 25–29 September 1995

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ENVIRONMENTAL HEALTH CRITERIA FOR SELECTED NON-HETEROCYCLIC POLYCYCLIC AROMATIC HYDROCARBONS

A WHO Task Group on Environmental Health Criteria for selected Non-Heterocyclic Polycyclic Aromatic Hydrocarbons met at the Fraunhofer Institute for Toxicology and Aerosol Research, Hanover, Germany, from 25 to 29 September 1995. Dr E. Smith, IPCS, welcomed the participants on behalf of Dr M. Mercier, Director of the IPCS, and on behalf of the three IPCS cooperating organizations (UNEP, ILO, and WHO). The Group reviewed and revised the draft and made an evaluation of the risks for human health and the environment from exposure to nonheterocyclic polycyclic aromatic hydrocarbons (PAH).

The first and second drafts of the EHC on non-heterocyclic PAH were prepared under the coordination of Dr R.F. Hertel, Dr G. Rosner, and Dr J. Kielhorn of the Fraunhofer Institute of Toxicology and Aerosol Research, Hanover, Germany, by the authors Dr S. Arteli, Dr A. Boehncke, Dr O. Creutzenberg, Dr I. Mangelsdorf of the same Institute, in cooperation with Dr E. Menichini, Italy and Dr P.L. Grover, United Kingdom, and Dr J. Blok, Netherlands. The Appendix on risk assessment methods for PAHs was prepared by Dr P. Muller, Canada, and Dr R. Schoeny and Dr T.L. Mumford, USA.

Dr E.M. Smith of the IPCS Central Unit was responsible for the scientific aspects of the monograph and Mrs E. Heseltine, Lajarthe, France, for the editing.

The efforts of all who helped in the preparation and finalization of the monograph are gratefully acknowledged.

1. SUMMARY

1.1 Selection of compounds for this monograph

Polycyclic aromatic hydrocarbons (PAH) constitute a large class of compounds, and hundreds of individual substances may be released during incomplete combustion or pyrolysis of organic matter, an important source of human exposure. Studies of various environmentally relevant matrices, such as coal combustion effluents, motor vehicle exhaust, used motor lubricating oil, and tobacco smoke, have shown that the PAH in these mixtures are mainly responsible for their carcinogenic potential.

PAH occur almost always in mixtures. Because the composition of such mixtures is complex and varies with the generating process, all mixtures containing PAH could not possible be covered in detail in this monograph. Thus, 33 individual compounds (31 parent PAH and two alkyl derivatives) were selected for evaluation on the basis of the availability of relevant data on toxicological end-points and/or exposure (Table 1). Since epidemiological studies, which are essential for risk assessment, were available only for mixtures, however, Sections 8 and 10 present the results of studies of mixtures of PAH, in contrast to the rest of the monograph.

Numerous papers and reviews have been published on the occurrence, distribution, and transformation of PAH in the environment and on their ecotoxicological and toxicological effects. Only references from the last 10–15 years are cited in this monograph, unless no other information was available; reviews are cited for older studies and for further information.

1.2 Identity, physical and chemical properties, and analytical methods

The term 'polycyclic aromatic hydrocarbons' commonly refers to a large class of organic compounds containing two or more fused aromatic rings made up of carbon and hydrogen atoms. At ambient temperatures, PAH are solids. The general characteristics common to the class are high melting- and boiling-points, low vapour pressure, and very low water solubility which tends to decrease with increasing molecular mass. PAH are soluble in many organic solvents and are highly lipophilic. They are chemically rather inert. Reactions that are of interest with respect to their environmental fate and possible sources of loss during atmospheric sampling are photodecomposition and reactions with nitrogen oxides, nitric acid, sulfur oxides, sulfuric acid, ozone, and hydroxyl radicals.

Table 1. Polycyclic aromatic hydrocarbons evaluated in this monograph

Common name	CAS name	Synonym[a]	CAS Registry No.
Acenaphthylene	Acenaphthylene		91-20-3
Acenaphthene	Acenaphthylene, 1,2-dihydro-		208-96-8
Anthanthrene	Dibenzo[*def,mno*]chrysene		191-26-4
Anthracene	Anthracene		120-12-7
Benz[*a*]anthracene	Benz[*a*]anthracene	1,2-Benzanthracene, tetraphene	56-55-3
Benzo[*a*]fluorene	11*H*-Benzo[*a*]fluorene	1,2-Benzofluorene	238-84-6
Benzo[*b*]fluorene	11*H*-Benzo[*b*]fluorene	2,3-Benzofluorene	243-17-4
Benzo[*b*]fluoranthene	Benz[*e*]acephenanthrylene	3,4-Benzofluoranthene	205-99-2
Benzo[*ghi*]fluoranthene	Benzo[*ghi*]fluoranthene	2,13-Benzofluoranthene	203-12-3
Benzo[*j*]fluoranthene	Benzo[*j*]fluoranthene	10,11-Benzofluoranthene	205-82-3
Benzo[*k*]fluoranthene	Benzo[*k*]fluoranthene	11,12-Benzofluoranthene	207-08-9
Benzo[*ghi*]perylene	Benzo[*ghi*]perylene	1,12-Benzoperylene	191-24-2
Benzo[*c*]phenanthrene	Benzo[*c*]phenanthrene	3,4-Benzophenanthrene	195-19-7
Benzo[*a*]pyrene	Benzo[*a*]pyrene	3,4-Benzopyrene[b]	50-32-8
Benzo[*e*]pyrene	Benzo[*e*]pyrene	1,2-Benzopyrene	192-97-2
Chrysene	Chrysene	1,2-Benzophenanthrene	218-01-9
Coronene	Coronene	Hexabenzobenzene	191-07-1
Cyclopenta[*cd*]pyrene	Cyclopenta[*cd*]pyrene	Cyclopenteno[*cd*]pyrene	27208-37-3
Dibenz[*a,h*]anthracene	Dibenz[*a,h*]anthracene	1,2:5,6-Dibenzanthracene	53-70-3
Dibenzo[*a,e*]pyrene	Naphtho[1,2,3,4-*def*]chrysene	1,2:4,5-Dibenzopyrene	192-65-4
Dibenzo[*a,h*]pyrene	Dibenzo[*b,def*]chrysene	3,4:8,9-Dibenzopyrene	189-64-0

Table 1 (contd)

Common name	CAS name	Synonym[a]	CAS Registry No.
Dibenzo[*a,i*]pyrene	Benzo[*rst*]pentaphene	3,4:9,10-Dibenzopyrene	189-55-9
Dibenzo[*a,l*]pyrene	Dibenzo[*def,p*]chrysene	1,2:3,4-Dibenzopyrene	191-30-0
Fluoranthene	Fluoranthene		206-44-0
Fluorene	9*H*-Fluorene		86-73-7
Indeno[1,2,3-*cd*]pyrene	Indeno[1,2,3-*cd*]-pyrene	2,3-*o*-Phenylenpyrene	193-39-5
5-Methylchrysene	Chrysene, 5-methyl-		3697-24-3
1-Methylphenanthrene	Phenanthrene, 1-methyl-		832-69-9
Naphthalene	Naphthalene		91-20-3
Perylene	Perylene	*peri*-Dinaphthalene	198-55-0
Phenanthrene	Phenanthrene		85-01-8
Pyrene	Pyrene	Benzo[*def*]phenanthrene	129-00-0
Triphenylene	Triphenylene	9,10-Benzophenanthrene	217-59-4

Extensive lists of synonyms have been reported by the IARC (1983) and Loening & Merritt (1990).

[a]Common synonym appearing in the literature

[b]Also reported as benzo[*def*]chrysene

Ambient air is sampled by collecting suspended particulate matter on glass-fibre, polytetrafluoroethylene, or quartz-fibre filters by means of high-volume or passive samplers. Vapour-phase PAH, which might volatilize from filters during sampling, are commonly trapped by adsorption on polyurethane foam. The sampling step is by far the most important source of variability in results.

Air is sampled at the workplace at low flow rates; particles are collected on glass-fibre or polytetrafluoroethylene filters and vapours on Amberlite XAD-2 resin. Devices for sampling stack gases are composed of a glass-fibre or quartz-fibre filter in front of a cooler to collect condensable matter and an adsorbent (generally XAD-2) cartridge. Vehicle exhausts are sampled under laboratory conditions, with standard driving cycles simulating on-road conditions. Emissions are collected either undiluted or after dilution with filtered cold air.

Many extraction and purification techniques have been described. Depending on the matrix, PAH are extracted from samples with a Soxhlet apparatus, ultrasonically, by liquid-liquid partition, or, after sample dissolution or alkaline digestion, with a selective solvent. Supercritical fluid extraction from various environmental solids has also been used. The efficiency of extraction depends heavily on the solvent used, and many of the solvents commonly used in the past were not appropriate. Extracted samples are usually purified by column chromatography, particularly on alumina, silica gel, or Sephadex LH-20 but also by thin-layer chromatography.

Identification and quantification are routinely performed by gas chromatography with flame ionization detection or by high-performance liquid chromatography (HPLC) with ultraviolet and fluorescence detection, generally in series. In gas chromatography, fused silica capillary columns are used, with polysiloxanes (SE-54 and SE-52) as stationary phases; silica-C18 columns are commonly used in HPLC. A mass spectrometric detector is often coupled to a gas chromatograph in order to confirm the identity of peaks.

The choice of PAH to be determined depends on the purpose of the measurement, e.g. for health-orientated or ecotoxicological studies or to investigate sources. Testing for different sets of compounds may be required or recommended at national and international levels.

1.3 Sources of human and environmental exposure

Little information is available on the production and processing of PAH, but it is probable that only small amounts of PAH are released as a direct result of these activities. The PAH found principally are used as intermediates in the production of polyvinylchloride and plasticizers (naphthalene), pigments (acenaphthene, pyrene), dyes (anthracene, fluoranthene), and pesticides (phenanthrene).

The largest emissions of PAH result from incomplete combustion of organic materials during industrial processes and other human activities, including:

- processing of coal, crude oil, and natural gas, including coal coking, coal conversion, petroleum refining, and production of carbon blacks, creosote, coal-tar, and bitumen;
- aluminium, iron and steel production in plants and foundries;
- heating in power plants and residences and cooking;
- combustion of refuse;
- motor vehicle traffic; and
- environmental tobacco smoke.

PAH, especially these of higher molecular mass, entering the environment via the atmosphere are adsorbed onto particulate matter. The hydrosphere and geosphere are affected secondarily by wet and dry deposition. Creosote-preserved wood is another source of release of PAH into the hydrosphere, and deposition of contaminated refuse, like sewage sludge and fly ash, contributes to emissions of PAH into the geosphere. Little information is available about the passage of PAH into the biosphere. PAH occur naturally in peat, lignite, coal, and crude oil. Most of the PAH in hard coals are tightly bound within the coal structure and cannot be leached out.

The release of PAH into the environment has been determined by identification of a characteristic PAH concentration profile, but this has been possible in only a few cases. Benzo[*a*]pyrene has frequently been used as an indicator of PAH, especially in older studies. Generally, emissions of PAH are only estimates based on more or less reliable data and give only a rough idea of exposure.

The most important sources of PAH are as follows:

Coal coking: Airborne emissions of PAH from coal coking in Germany have decreased significantly over the last 10 years as a result of technical improvements to existing plants, closure of old plants, and reduced coke production. Similar situations are assumed to exist in western Europe, Japan, and the USA, but no data were available.

Production of aluminium (mainly special coal anodes), *iron, and steel* and the binding agents used in moulding sand *in foundries*: Little information is available.

Domestic and residential heating: Phenanthrene, fluoranthene, pyrene, and chrysene are emitted as major components. The emissions from wood stoves are 25–1000 times higher than those from charcoal-fired stoves, and in areas where wood burning predominates for domestic heating the major portion of airborne PAH may be derived from this source, especially in winter. The release of PAH during residential heating is thus assumed to be an important source in developing countries where biomass is often burnt in relatively simple stoves.

Cooking: PAH may be emitted during incomplete combustion of fuels, from cooking oil, and from food being cooked.

Motor vehicle traffic: The main compounds released from petrol-fuelled vehicles are fluoranthene and pyrene, while naphthalene and acenaphthene are abundant in the exhaust of diesel-fuelled vehicles. Although cyclopenta[*cd*]-pyrene is emitted at a high rate from petrol-fuelled engines, its concentration in diesel exhaust is only just above the limit of detection. The emission rates, which depend on the substance, the type of vehicle, its engine conditions, and the test conditions, range from a few nanograms per kilometre to > 1000 mg/km. PAH emissions from vehicle engines are dramatically reduced by fitting catalytic converter devices.

Forest fires: In countries with large forest areas, fires can make an imprtant contribution to PAH emissions.

Coal-fired power plants: PAH released into the atmosphere from such plants consist mainly of two- and three-ring compounds. In contaminated areas, the PAH levels in ambient air may be higher than those in the stack gases.

Incineration of refuse: The PAH emissions in stack gases from this souce in a number of countries were < 10 mg/m^3.

1.4 Environmental transport, distribution, and transformation

Several distribution and transformation processes determine the fate of both individual PAH and mixtures. Partitioning between water and air, between water and sediment, and between water and biota are the most important of the distribution processes.

As PAH are hydrophobic with low solubility in water, their affinity for the aquatic phase is very low; however, in spite of the fact that most PAH are released into the environment via the atmosphere, considerable concentrations are also found in the hydrosphere because of their low Henry's law constants. As the affinity of PAH for organic phases is greater than that for water, their partition coefficients between organic solvents, such as octanol, and water are high. Their affinity for organic fractions in sediment, soil, and biota is also high, and PAH thus accumulate in organisms in water and sediments and in their food. The relative importance of uptake from food and from water is not clear. In *Daphnia* and molluscs, accumulation of PAH from water is positively correlated with the octanol:water partition coefficient (K_{ow}). In fish and algae that can metabolize PAH, however, the internal concentrations of different PAH are not correlated with the K_{ow}.

Biomagnification—the increase in the concentration of a substance in animals in successive trophic levels of food chains—of PAH has not been observed in aquatic systems and would not be expected to occur, because most organisms have a high biotransformation potential for PAH. Organisms at higher trophic levels in food chains show the highest potential biotransformation.

PAH are degraded by photodegradation, biodegradation by microorganisms, and metabolism in higher biota. Although the last route of transformation is of minor importance for the overall fate of PAH in the environment, it is an important pathway for the biota, since carcinogenic metabolites may be formed. As PAH are chemically stable, with no reactive groups, hydrolysis plays no role in their degradation. Few standard tests for the biodegradation of PAH are available. In general, they are biodegraded under aerobic conditions, the biodegradation rate decreasing drastically with the number of aromatic rings. Under anaerobic conditions, degradation is much slower.

PAH are photooxidized in air and water in the presence of sensitizing radicals like OH, NO_3, and O_3. Under laboratory conditions, the half-life of the reaction with airborne OH radicals is about one day, whereas reactions with NO_3 and O_3 usually have much lower velocity constants. The adsorption of high-molecular-mass PAH onto carbonaceous particles in the environment should stabilize the reaction with OH radicals. The reaction of two- to four-ring PAH, which occur mainly in the vapour phase, with NO_3 leads to nitro-PAH, which are known mutagens. The photooxidation of some PAH in water seems to be more rapid than in air. Calculations based on physicochemical and degradation parameters indicate that PAH with four or more aromatic rings persist in the environment.

1.5 Environmental levels and human exposure

PAH are ubiquitous in the environment, and various individual PAH have been detected in different compartments in numerous studies.

1.5.1 Air

The levels of individual PAH tend to be higher in winter than in summer by at least one order of magnitude. The predominant source during winter is residential heating, while that during summer is urban motor vehicle traffic. Average concentrations of 1–30 ng/m^3 of individual PAH were detected in the ambient air of various urban areas. In large cities with heavy motor vehicle traffic and extensive use of biomass fuel, such as Calcutta, levels of up to 200 ng/m^3 of individual PAH were found. Concentrations of 1–50 ng/m^3 were detected in road tunnels. Cyclopenta[*cd*]pyrene and pyrene were present at concentrations up to 100 ng/m^3. In a subway station, PAH concentrations of up to 20 ng/m^3 were measured. Near industrial sources, the average concentrations of individual PAH ranged from 1 to 10 ng/m^3. Phenanthrene was present at up to a maximum of about 310 ng/m^3.

The background values of PAH are at least one or two orders of magnitude lower than those near sources like motor vehicle traffic. For example, the levels at 1100 m ranged from 0.004 to 0.03 ng/m^3.

1.5.2 *Surface water and precipitation*

Most of the PAH in water are believed to result from urban runoff, from atmospheric fallout (smaller particles), and from asphalt abrasion (larger particles). The major source of PAH varies, however, in a given body of water. In general, most samples of surface water contain individual PAH at levels of up to 50 ng/litre, but highly polluted rivers had concentrations of up to 6000 ng/litre. The PAH levels in groundwater are within the range 0.02–1.8 ng/litre, and drinking-water samples contain concentrations of the same order of magnitude. Major sources of PAH in drinking-water are asphalt-lined storage tanks and delivery pipes.

The levels of individual PAH in rainwater ranged from 10 to 200 ng/litre, whereas levels of up to 1000 ng/litre have been detected in snow and fog.

1.5.3 *Sediment*

The concentrations of individual PAH in sediment were generally one order of magnitude higher than those in precipitation.

1.5.4 Soil

The main sources of PAH in soil are atmospheric deposition, carbonization of plant material, and deposition from sewage and particulate waste. The extent of pollution of soil depends on factors such as its cultivation, its porosity, and its content of humic substances.

Near industrial sources, individual PAH levels of up to 1 g/kg soil have been found. The concentrations in soil from other sources, such as automobile exhaust, are in the range 2–5 mg/kg. In unpolluted areas, the PAH levels were 5–100 µg/kg soil.

1.5.5 *Food*

Raw food does not normally contain high levels of PAH, but they are formed by processing, roasting, baking, or frying. Vegetables may be contaminated by the deposition of airborne particles or by growth in contaminated soil. The levels of individual PAH in meat, fish, dairy products, vegetables and fruits, cereals and their products, sweets, beverages, and animal and vegetable fats and oils were within the range 0.01–10 µg/kg. Concentrations of over 100 µg/kg have been detected in smoked meat and up to 86 µg/kg in smoked fish; smoked cereals contained up to 160 µg/kg. Coconut oil contained up to 460 µg/kg. The levels in human breast milk were 0.003–0.03 µg/kg.

1.5.6 *Aquatic organisms*

Marine organisms are known to adsorb and accumulate PAH from water. The degree of contamination is related to the extent of industrial and urban

development and shipping movements. PAH concentrations of up to 7 mg/kg have been detected in aquatic organisms living near industrial effluents, and the average levels of PAH in aquatic animals sampled at contaminated sites were 10–500 µg/kg, although levels of up to 5 mg/kg were also detected.

The average levels of PAH in aquatic animals sampled at various sites with unspecified sources of PAH were 1–100 µg/kg, but concentrations of up to 1 mg/kg were found, for example, in lobsters in Canada.

1.5.7 *Terrestrial organisms*

The concentrations of PAH in insects ranged from 730 to 5500 µg/kg. The PAH content of earthworm faeces depends significantly on the location: those in a highly industrialized region in eastern Germany contained benzo[*a*]pyrene at concentrations up to 2 mg/kg.

1.5.8 *General population*

The main sources of nonoccupational exposure are: polluted ambient air, smoke from open fireplaces and cooking, environmental tobacco smoke, contaminated food and drinking-water, and the use of PAH-contaminated products. PAH can be found in indoor air as a result of residential heating and environmental tobacco smoke at average concentrations of 1–100 ng/m^3, with a maximum of 2300 ng/m^3.

The intake of individual PAH from food has been estimated to be 0.10–10 µg/day per person. The total daily intake of benzo[*a*]pyrene from drinking-water was estimated to be 0.0002 µg/person. Cereals and cereal products are the main contributors to the intake of PAH from food because they are a major component of the total diet.

1.5.9 *Occupational exposure*

Near a coke-oven battery, the levels of benzo[*a*]pyrene ranged from < 0.1 to 100–200 µg/m^3, with a maximum of about 400 µg/m^3. In modern coal gasification systems, the concentration of PAH is usually < 1 µg/m^3 with a maximum of 30 µg/m^3. Personal samples taken from operators of petroleum refinery equipment showed exposure to 2.6–470 µg/m^3. In samples of air taken near bitumen processing plants at refineries, the total PAH levels were 0.004–50 µg/m^3. Near road paving operations, the total PAH concentrations in personal air samples were up to 190 µg/m^3, and the mean value in area air samples was 0.13 µg/m^3. The PAH levels in personal air samples taken at an aluminium smelter were 0.05–9.6 µg/m^3, but urine samples of workers at an aluminium plant contained very low levels. Area air samples contained PAH concentrations of up to 5 µg/m^3 in one German foundry, 3–40 µg/m^3 at iron mines and 4–530 µg/m^3 at copper mines. The concentrations of PAH in cooking fumes in a food factory ranged from 0.07 to 26 µg/m^3.

1.6 Kinetics and metabolism

PAH are absorbed through the pulmonary tract, the gastrointestinal tract, and the skin. The rate of absorption from the lungs depends on the type of PAH, the size of the particles on which they are absorbed, and the composition of the adsorbent. PAH adsorbed onto particulate matter are cleared from the lungs more slowly than free hydrocarbons. Absorption from the gastrointestinal tract occurs rapidly in rodents, but metabolites return to the intestine via biliary excretion. Studies with ^{32}P-postlabelling of percutaneous absorption of mixtures of PAH in rodents showed that components of the mixtures reach the lungs, where they become bound to DNA. The rate of percutaneous absorption in mice according to the compound.

PAH are widely distributed throughout the organism after administration by any route and are found in almost all internal organs, but particularly those rich in lipids. Intravenously injected PAH are cleared rapidly from the bloodstream of rodents but can cross the placental barrier and have been detected in fetal tissues.

The metabolism of PAH is complex. In general, parent compounds are converted via intermediate epoxides to phenols, diols, and tetrols, which can themselves be conjugated with sulfuric or glucuronic acids or with glutathione. Most metabolism results in detoxification, but some PAH are activated to DNA-binding species, principally diol epoxides, which can initiate tumours. PAH metabolites and their conjugates are excreted via the urine and faeces, but conjugates excreted in the bile can be hydrolysed by enzymes of the gut flora and reabsorbed. It can be inferred from the available information on the total human body burden that PAH do not persist in the body and that turnover is rapid. This inference excludes those PAH moieties that become covalently bound to tissue constituents, in particular nucleic acids, and are not removed by repair.

1.7 Effects on laboratory mammals and *in vitro*

The acute toxicity of PAH appears to be moderate to low. The well-characterized PAH, naphthalene, showed oral and intravenous LD_{50} values of 100–500 mg/kg body weight (bw) in mice and a mean oral LD_{50} of 2700 mg/kg bw in rats. The values for other PAH are similar. Single high doses of naphthalene induced bronchiolar necrosis in mice, rats, and hamsters.

Short-term studies showed adverse haematological effects, expressed as myelotoxicity with benzo[*a*]pyrene, haemolymphatic changes with dibenz[*a*,*h*]-anthracene, and anaemia with naphthalene; however, in a seven-day study by oral and intraperitoneal administration in mice, tolerance to the effect of naphthalene was observed.

Systemic effects caused by long-term treatment with PAH have been described only rarely, because the end-point of most studies has been

carcinogenicity. Significant toxic effects are manifested at doses at which carcinogenic responses are also triggered.

In studies of adverse effects on the skin after dermal application, non- or weakly carcinogenic PAH such as perylene, benzo[*e*]pyrene, phenanthrene, pyrene, anthracene, acenaphthalene, fluorene, and fluoranthene were inactive, whereas carcinogenic compounds such as benz[*a*]anthracene, dibenz[*a*,*h*]-anthracene, and benzo[*a*]pyrene caused hyperkeratosis. Anthracene and naphthalene vapours caused mild eye irritation. Benzo[*a*]pyrene induced contact hypersensitivity in guinea-pigs and mice.

Benz[*a*]anthracene, benzo[*a*]pyrene, dibenz[*a*,*h*]anthracene, and naphthalene were embrotoxic to mice and rats. Benzo[*a*]pyrene also had teratogenic and reproductive effects. Intensive efforts have been made to elucidate the genetic basis of the embryotoxic effect of benzo[*a*]pyrene. Fetal death and malformations are observed only if the cytochrome P450 monooxygenase system is inducible, either in the mother (with placental permigration) or in the embryo. Not all of the effects observed can be explained by genetic predisposition, however: in mice and rabbits, benzo[*a*]pyrene had transplacental carcinogenic activity, resulting in pulmonary adenomas and skin papillomas in the progeny. Reduced fertility and oocyte destruction were also observed.

PAH have also been studied extensively in assays for genotoxicity and cell transformation; most of the 33 PAH covered in this monograph are genotoxic or probably genotoxic. The only compounds for which negative results were found in all assays were anthracene, fluorene, and naphthalene. Owing to inconsistent results, phenanthrene and pyrene could not be reliably classified for genotoxicity.

Comprehensive work on the carcinogenicity of PAH shows that 26 of the 33 studied are, or are suspected of being, carcinogenic (Table 2). The best-characterized PAH is benzo[*a*]pyrene, which has been studied by all current methods in seven species. PAH that have been the subject of 12 or more studies are anthanthrene, anthracene, benz[*a*]anthracene, chrysene, dibenz[*a*,*h*]-anthracene, dibenzo[*a*,*i*]pyrene, 5-methylchrysene, phenanthrene, and pyrene. Special studies of the phototoxicity, immunotoxicity, and hepatotoxicity of PAH are supplemented by reports on the ocular toxicity of naphthalene. Anthracene, benzo[*a*]pyrene, and some other PAH were phototoxic to mammalian skin and in cell cultures *in vitro* when applied with ultraviolet radiation. PAH have generally been reported to have immunosuppressive effects. After intraperitoneal treatment of mice with benzo[*a*]pyrene, immunological parameters were strongly suppressed in the progeny for up to 18 months. Increased liver regeneration and an increase in liver weight have also been observed. The effect of naphthalene in inducing formation of cataracts in the rodent eye has been attributed to the inducibility of the cytochrome P450 system in studies in which genetically different mouse strains were used.

Theoretical models to predict the carcinogenic potency of PAH from their structures, based on a large amount of experimental work, were presented as early as the 1930s. The first model was based on the high chemical reactivity of certain double bonds (the K-region theory). A later systematic approach was based on the chemical synthesis of possible metabolites and their mutagenic activity. This 'bay region' theory proposes that epoxides adjacent to a bay region yield highly stabilized carbonium ions. Other theoretical approaches are the 'di-region theory' and the 'radical cation potential theory'.

Many individual PAH are carcinogenic to animals and may be carcinogenic to humans, and exposure to several PAH-containing mixtures has been shown to increase the incidence of cancer in human populations. There is concern that those PAH found to be carcinogenic in experimental animals are likely to be carcinogenic in humans. PAH produce tumours both at the site of contact and at distant sites. The carcinogenic potency of PAH may vary with the route of exposure. Various approaches to assessing the risk associated with exposure to PAH, singly and in mixtures, have been proposed. No one approach is endorsed in this monograph; however, the data requirements, assumptions, applicability, and other features of three quantitative risk assessment processes that have been validated to some degree are described.

1.8 Effects on humans

Because of the complex profile of PAH in the environment and in workplaces, human exposure to pure, individual PAH has been limited to scientific experiments with volunteers, except in the case of naphthalene which is used as a moth-repellant for clothing.

After dermal application, anthracene, fluoranthene, and phenanthrene induced specific skin reactions, and benzo[*a*]pyrene induced reversible, regressive verrucae which were classified as neoplastic proliferations. The systemic effects of naphthalene are known from numerous cases of accidental intake, particularly by children. The lethal oral dose is 5000–15 000 mg for adults and 2000 mg taken over two days for a child. The typical effect after dermal or oral exposure is acute haemolytic anaemia, which can also affect fetuses transplacentally.

Tobacco smoking is the most important single factor in the induction of lung tumours and also for increased incidences of tumours of the urinary bladder, renal pelvis, mouth, pharynx, larynx, and oesophagus. The contribution of PAH in the diet to the development of human cancer is not considered to be high. In highly industrialized areas, increased body burdens of PAH due to polluted ambient air were detected. Psoriasis patients treated with coal-tar are also exposed to PAH.

Occupational exposure to soot as a cause of scrotal cancer was noted for the first time in 1775. Later, occupational exposure to tars and paraffins was

Table 2. Summary of results of tests for genotoxicity and carcinogenicity for the 33 polycyclic aromatic hydrocarbons studied

Compound	Genotoxicity	Carcinogenicity
Acenaphthene	(?)	(?)
Acenaphthylene	(?)	No studies
Anthracene	-	-
Benz[a]anthracene	+	+
Benzo[*a*]fluorene	(?)	(?)
Benzo[*a*]pyrene	+	+
Benzo[*b*]fluoranthene	+	+
Benzo[*b*]fluorene	(?)	(?)
Benzo[*c*]phenanthrene	(+)	+
Benzo[*e*]pyrene	+	?
Benzo[*ghi*]fluoranthene	(+)	(-)
Benzo[*ghi*]perylene	+	-
Benzo[*j*]fluoranthene	+	+
Benzo[*k*]fluoranthene	+	+
Chrysene	+	+
Coronene	(+)	(?)
Cyclopenta[*cd*]pyrene	+	+
Dibenzo[*a,e*]pyrene	+	+
Dibenz[*a,h*]anthracene	+	+
Dibenzo[*a,h*]pyrene	(+)	+
Dibenzo[*a,i*]pyrene	+	+
Dibenzo[*a,l*]pyrene	(+)	+
Fluoranthene	+	(+)
Fluorene	-	-
Indeno[1,2,3-*cd*]pyrene	+	+
1-Methylphenanthrene	+	-
5-Methylchrysene	+	+
Naphthalene	-	?
Perylene	+	-
Phenanthrene	(?)	(?)
Pyrene	(?)	-
Triphenylene	+	-

+, positive; -, negative; ?, questionable
Parentheses, result derived from small database

reported to induce skin cancer. The lung is now the main site of PAH-induced cancer, whereas skin tumours have become more rare because of better personal hygiene.

Epidemiological studies have been conducted of workers exposed at coke ovens during coal coking and coal gasification, at asphalt works, foundries, and aluminium smelters, and to diesel exhaust. Increased lung tumour rates due to exposure to PAH have been found in coke-oven workers, asphalt workers, and workers in Söderberg potrooms of aluminium reduction plants. The highest risk was found for coke-oven workers, with a standardized mortality ratio of 195. Dose–response relationships were found in several studies. In aluminium plants, not only urinary bladder cancer but also asthma-like symptoms, lung function abnormalities, and chronic bronchitis have been observed. Coke-oven workers were found to have decreased serum immunoglobulin levels and decreased immune function. Occupational exposure to naphthalene for five years was reported to have caused cataract.

Several methods have been developed to assess internal exposure to PAH. In most of the studies, PAH metabolites such as urinary thioethers, 1-naphthol, β-naphthylamine, hydroxyphenanthrenes, and 1-hydroxypyrene were measured in urine. The latter has been used widely as a biological index of exposure.

The genotoxic effects of PAH have been determined by testing for mutagenicity in urine and faeces and for the presence of micronuclei, chromosomal aberrations, and sister chromatid exchange in peripheral blood lymphocytes. In addition, adducts of benzo[*a*]pyrene with DNA in peripheral lymphocytes and other tissues and with proteins like albumin as well as antibodies to DNA adducts have been measured.

1-Hydroxypyrene in urine and DNA adducts in lymphocytes have been investigated as markers in several studies. 1-Hydroxpyrene can be measured more easily than DNA adducts, there is less variation between individuals, and lower levels of exposure can be detected. Both markers have been used to assess human exposure in various environments. Increased 1-hydroxpyrene excretion or DNA adducts were found at various workplaces in coke plants, aluminum manufacturing, wood impregnation plants, foundries, and asphalt works. The highest exposures were those of coke-oven workers and workers impregnating wood with creosote, who took up 95% of total of PAH through the skin, in contrast to the general population in whom uptake via food and tobacco smoking predominate.

Estimates of the risk associated with exposure to PAH and PAH mixtures are based on estimates of exposure and the results of epidemiological studies. Data for coke-oven workers resulted in a relative risk for lung cancer of 15.7. On this basis, the risk of the general population for developing lung cancer over a lifetime has been calculated to be 10^{-4} to 10^{-5} per ng of benzo[*a*]pyrene per m^3 air. In other words, about one person in 10 000 or 100 000 would be expected to develop lung cancer in his or her lifetime as a result of exposure to benzo[*a*]pyrene in air.

1.9 Effects on other organisms in the laboratory and the field

PAH are acutely toxic to fish and *Daphnia magna* in combination with absorption of ultraviolet radiation and visible light. Metabolism and degradation alter the toxicity of PAH. At low concentrations, PAH can stimulate the growth of microorganisms and algae. The most toxic PAH for algae are benz[*a*]anthracene (four-ring), the concentration at which given life parameters are reduced by 50% (EC_{50}) being 1–29 µg/litre, and benzo[*a*]pyrene (five-ring), with an EC_{50} of 5–15 µg/litre. The EC_{50} values for algae for most three-ring PAH are 240–940 µg/litre. Naphthalene (two-ring) is the least toxic, with EC_{50} values of 2800–34 000 µg/litre.

No clear difference in sensitivity was found between different taxonomic groups of invertebrates like crustaceans, insects, molluscs, polychaetes, and echinoderms. Naphthalene is the least toxic, with 96-h LC_{50} values of 100–2300 µg/litre. The 96-h LC_{50} values for three-ring PAH range between < 1 and 3000 µg/litre. Anthracene may be more toxic than the other three-ring PAH, with 24-h LC_{50} values between < 1 and 260 µg/litre. The 96-h LC_{50} values for four-, five-, and six-ring PAH are 0.2–1200 µg/litre. Acute toxicity (LC_{50}) in fish was seen at concentrations of 110 to > 10 000 µg/litre of naphthalene, 30–4000 µg/litre of three-ring PAH (anthracene, 2.8–360 µg/litre), and 0.7–26 µg/litre for four- or five-ring PAH.

Contamination of sediments with PAH at concentrations of 250 mg/kg was associated with hepatic tumours in free-living fish. Tumours have also been induced in fish exposed in the laboratory. Exposure of fish to certain PAH can also cause physiological changes and affect their growth, reproduction, swimming performance, and respiration.

2. IDENTITY, PHYSICAL AND CHEMICAL PROPERTIES, AND ANALYTICAL METHODS

2.1 Identity

The name 'polycyclic aromatic hydrocarbons' (PAH) commonly refers to a large class of organic compounds containing two or more fused aromatic rings, even though in a broad sense non-fused ring systems should be included. In particular, the term 'PAH' refers to compounds containing only carbon and hydrogen atoms (i.e. unsubstituted parent PAH and their alkyl-substituted derivatives), whereas the more general term 'polycyclic aromatic compounds' also includes the functional derivatives (e.g. nitro- and hydroxy-PAH) and the heterocyclic analogues, which contain one or more hetero atoms in the aromatic structure (aza-, oxa-, and thia-arenes). Some authors refer to polycyclic aromatic compounds as 'polycyclic organic matter', and the term 'polynuclear' is frequently used for 'polycyclic', as in 'polynuclear aromatic compounds'. More than 100 PAH have been identified in atmospheric particulate matter (Lao et al., 1973; Lee et al., 1976a) and in emissions from coal-fired residential furnaces (Grimmer et al., 1985), and about 200 have been found in tobacco smoke (Lee et al., 1976b, 1981).

The selection of PAH evaluated in this monograph is discussed in Section 1. The nomenclature, common names, synonyms, and abbreviations used are given in Table 1 in that section. The structural formulae are shown in Figure 1. Molecular formulae, relative molecular masses, and CAS Registry numbers are given in Table 3.

2.1.1 Technical products

Technical-grade naphthalene, also known as naphthalin and tar camphor, has a minimum purity of 95%. The impurities reported are benzo[*b*]thiophene (thianaphthene) when naphthalene is obtained from coal-tar and methylindenes when it is derived from petroleum (Society of German Chemists, 1989).

Commercially available anthracene, also known by the trade name Tetra Olive N2G (IARC, 1983), has a purity of 90–95% (Hawley, 1987). The impurities reported are phenanthrene, chrysene, carbazole (Hawley, 1987), tetracene, naphthacene (Budavari et al., 1989), and pyridine at a maximum of 0.2% (IARC, 1983). The following purities were reported for other technical-grade products: acenaphthene, 95–99%; fluoranthene, > 95% (Griesbaum et al., 1989); fluorene, about 95%; phenanthrene, 90%; and pyrene, about 95% (Franck & Stadelhofer, 1987).

The other compounds are generally produced as chemical intermediates and for research purposes (see also sections 3.2.2 and 3.2.3). Reference materials certified to be of geater than 99% purity are available for 22 of the

Table 3. Identity of polycyclic aromatic hydrocarbons covered in this volume ranked according to molecular mass

Compound	Molecular formula	Relative molecular mass	CAS Registry No.
Naphthalene	$C_{10}H_8$	128.2	91-20-3
Acenaphthylene	$C_{12}H_8$	152.2	208-96-8
Acenaphthene	$C_{12}H_{10}$	154.2	83-32-9
Fluorene	$C_{13}H_{10}$	166.2	86-73-7
Anthracene	$C_{14}H_{10}$	178.2	120-12-7
Phenanthrene	$C_{14}H_{10}$	178.2	85-01-8
1-Methylphenanthrene	$C_{15}H_{12}$	192.3	832-69-9
Fluoranthene	$C_{16}H_{10}$	202.3	206-44-0
Pyrene	$C_{16}H_{10}$	202.3	129-00-0
Benzo[*a*]fluorene	$C_{17}H_{12}$	216.3	238-84-6
Benzo[*b*]fluorene	$C_{17}H_{12}$	216.3	243-17-4
Benzo[*ghi*]fluoranthene	$C_{18}H_{10}$	226.3	203-12-3
Cyclopenta[*cd*]pyrene	$C_{18}H_{10}$	226.3	2720837-3
Benz[*a*]anthracene	$C_{18}H_{12}$	228.3	56-55-3
Benzo[*c*]phenanthrene	$C_{18}H_{12}$	228.3	195-19-7
Chrysene	$C_{18}H_{12}$	228.3	218-01-9
Triphenylene	$C_{18}H_{12}$	228.3	217-59-4
5-Methylchrysene	$C_{19}H_{14}$	242.3	3697-24-3
Benzo[*b*]fluoranthene	$C_{20}H_{12}$	252.3	205-99-2
Benzo[*j*]fluoranthene	$C_{20}H_{12}$	252.3	205-82-3
Benzo[*k*]fluoranthene	$C_{20}H_{12}$	252.3	207-08-9
Benzo[*a*]pyrene	$C_{20}H_{12}$	252.3	50-32-8
Benzo[*e*]pyrene	$C_{20}H_{12}$	252.3	192-97-2
Perylene	$C_{20}H_{12}$	252.3	198-55-0
Anthanthrene	$C_{22}H_{12}$	276.3	191-26-4
Benzo[*ghi*]perylene	$C_{22}H_{12}$	276.3	191-24-2
Indeno[1,2,3-*cd*]pyrene	$C_{22}H_{12}$	276.3	193-39-5
Dibenz[*a,h*]anthracene	$C_{22}H_{14}$	278.4	53-70-3
Coronene	$C_{24}H_{12}$	300.4	191-07-1
Dibenzo[*a,e*]pyrene	$C_{24}H_{14}$	302.4	192-65-4
Dibenzo[*a,h*]pyrene	$C_{24}H_{14}$	302.4	189-64-0
Dibenzo[*a,i*]pyrene	$C_{24}H_{14}$	302.4	189-55-9
Dibenzo[*a,l*]pyrene	$C_{24}H_{14}$	302.4	191-30-0

PAH considered (Community Bureau of Reference, 1992); the remaining compounds are commercially available as chemical standards, with a purity of 99% or more.

Figure 1. Structural formulae of polycyclic aromatic hydrocarbons covered in this monograph



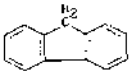
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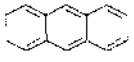
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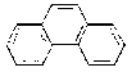
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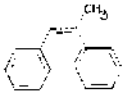
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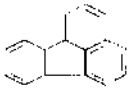
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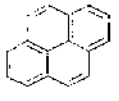
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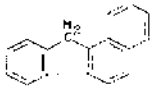
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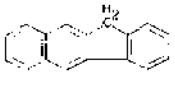
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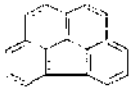
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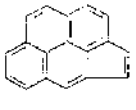
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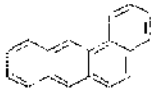
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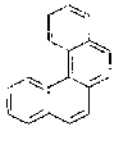
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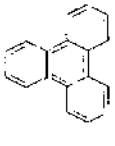
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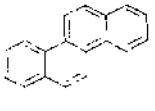
Benz[*a*]anthracene



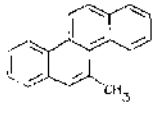
Benzo[*c*]phenanthrene



Triphenylene

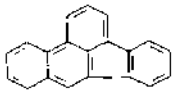


Chrysene

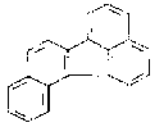


5-Methylcholanthrene

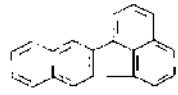
Figure 1 (contd)



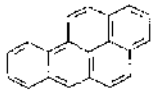
Benzo[*b*]fluoranthene



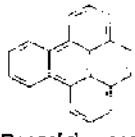
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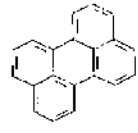
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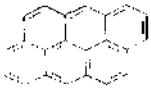
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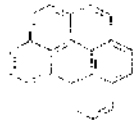
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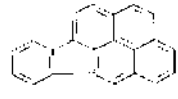
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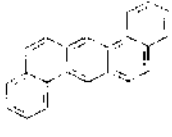
Anthanthrene



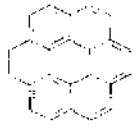
Benzo[*ghi*]perylene



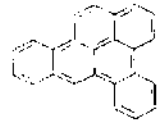
Indeno[1,2,3-*cd*]pyrene



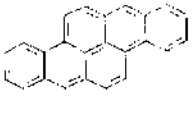
Dibenz[*a,h*]anthracene



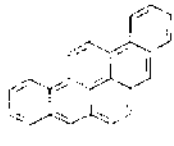
Coronene



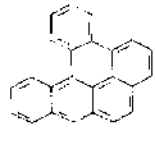
Dibenzo[*a,e*]pyrene



Dibenzo[*a,h*]pyrene



Dibenzo[*a,i*]pyrene



Dibenzo[*a,l*]pyrene

2.2 Physical and chemical properties

Physical and chemical properties relevant to the toxicological and ecotoxicological evaluation of the PAH are summarized in Table 4. It should be kept in mind that the values for any one parameter may be derived from different sources, with different methods of measurement or calculation, so that individual values cannot be compared directly unless the original sources are consulted. In particular, the vapour pressures reported in the literature for the same PAH vary by up to several orders of magnitude (Mackay & Shiu, 1981; Lane, 1989). Variations are also seen in the reported solubility in water of various PAH, although the values are generally within one order of magnitude (National Research Council Canada, 1983). Flash-points were available only for three compounds with high molecular mass (for napthalene, 78.9 °C by the open-cup method and 87.8 °C by the closed cup method; anathrene, 121 °C by the closed-cup method; and phenanthrene, 171 °C by the open cup method). Explosion limitswere avaialable only for naphthalene (0.9–5.9 vol %) and ananthrene (0.6 vol %) (Lewis, 1992). Vapour density (air = 1) was 4.42 for naphathalene (IARC, 1973), 5.32 for acenaphthene, 6.15 for anathrene (Lewis, 1992), 6.15 for phenanthrene, and 8.7 for benzo[*a*]pyrene (National Institute for Occupational Safety and Health and Occupational Safety and Health Administration, 1981).

The physical and chemical properties are largely determined by the conjugated α-electron systems, which vary fairly regularly with the number of rings and molecular mass, giving rise to a more or less wide range of values for each parameter within the whole class. At room temperature, all PAH are solids. The general characteristics common to the class are high melting- and boiling-points, low vapour pressure, and very low solubility in water. PAH are soluble in many organic solvents (IARC, 1983; Agency for Toxic Substances and Disease Registry, 1990; Lide, 1991) and are highly lipophilic.

Vapour pressure tends to decrease with increasing molecular mass, varying by more than 10 orders of magnitude. This characteristic affects the adsorption of individual PAH onto particulate matter in the atmosphere and their retention on particulate matter during sampling on filters (Thrane & Mikalsen, 1981). Vapour pressure increases markedly with ambient temperature (Murray et al., 1974), which additionally affects the distribution coefficients between gaseous and particulate phases (Lane, 1989). Solubility in water tends to decreases with increasing molecular mass. For additional information, refer to section 4.1.

PAH are chemically inert compounds (see also section 4.4). When they react, they undergo two types of reaction: electrophilic substitution and addition. As the latter destroys the aromatic character of the benzene ring that is affected, PAH tend to form derivatives by the former reaction; addition is often followed by elimination, resulting in net substitution. The chemical and photochemical reactions of PAH in the atmosphere have been reviewed

Table 4. Physical and chemical properties of polycyclic aromatic compounds covered in this monograph, ranked by molecular mass

Compound	Colour	Melting-point[a] (°C)	Boiling-point (°C)	Vapour pressure (Pa at 25 °C)	Density[c]	*n*-Octanol: water partition coefficient (log K_{ow})	Solubility in water at 25 °C (µg/litre)[d]	Henry's law constant at 25 °C (kPa)
Naphthalene	White[b]	81	217.9[e]	10.4[g]	1.154[25/4 h]	3.4[i]	3.17×10^4	4.89×10^{-2}[k]
Acenaphthylene		92–93		8.9×10^{-1}[g]	0.899[16/2 h]	4.07[l]		1.14×10^{-3}[l]
Acenaphthene	White[b]	95	279[b]	2.9×10^{-1}[g]	1.024[90/4 h]	3.92[l]	3.93×10^3	1.48×10^{-2}[k]
Fluorene	White[e]	115–116	295[e]	8.0×10^{-2}[g]	1.203[20/4 h]	4.18[m]	1.98×10^3	1.01×10^{-2}[n]
Anthracene	Colourless[o]	216.4	342[o]	8.0×10^{-4}[g]	1.283[25/4 h]	4.5[i]	73	7.3×10^{-2}[n]
Phenanthrene	Colourless[p]	100.5	340[b]	1.6×10^{-2}[g]	0.980[4 h]	4.6[i]	1.29×10^3	3.98×10^{-3}[k]
1-Methylphenanthrene		123	354–355[r]			5.07[o]	255 (24 °C)[t]	
Fluoranthene	Pale yellow[b]	108.8	375[b]	1.2×10^{-3}[g]	1.252[0/4 h]	5.22[b]	260	6.5×10^{-4} (20 °C)[w]
Pyrene	Colourless[e]	150.4	393[b]	6.0×10^{-4}[g]	1.271[23/4 h]	5.18[i]	135	1.1×10^{-3}[k]
Benzo[*a*]fluorene	Colourless[x]	189–190[b]	398–400[y]			5.32[z]	45	
Benzo[*b*]fluorene	Colourless[x]	213.5	401–402[z]		1.226[aa]	5.75[c]	2.0	
Benzo[*ghi*]fluoranthene	Yellow[bb]	128.4	432[cc]		1.345[23/4 dd]			
Cyclopenta[*cd*]pyrene	Orange[x]	170	439[ee]					
Benz[*a*]anthracene	Colourless[b]	160.7	400[b]	2.8×10^{-5}[g]	1.226[aa]	5.61[f]	14	
Benzo[*c*]phenanthrene	Colourless[x]	66.1			1.265[ff]			
Chrysene	Colourless with blue fluoresence[b]	253.8	448[b]	8.4×10^{-5} (20 °C)[gg]	1.274[20/4 e]	5.91[e]	2.0	
Triphenylene	Colourless[x]	199	425[hh]		1.3[p]	5.45[hh]	43	

Table 4 (contd)

Compound	Colour	Melting-point[a] (°C)	Boiling-point (°C)	Vapour pressure (Pa at 25 °C)	Density[c]	*n*-Octanol: water partition coefficient (log K_{ow})	Solubility in water at 25 °C (µg/litre)[d]	Henry's law constant at 25 °C (kPa)
5-Methylchrysene	Colourless[a]	117.1	458[ii]				62 (27 °C)[jj]	
Benzo[*b*]fluoranthene	Colourless[c]	168.3	481[kk]	6.7×10^{-5} (20 °C)[ll]		6.12[f]	1.2[jj] (20 °C)[w]	5.1×10^{-5}
Benzo[*j*]fluoranthene	Yellow[h]	165.4	480[ee]	2.0×10^{-6}[ff]		6.12[mm]	2.5[nn]	
Benzo[*k*]fluoranthene	Pale yellow[h]	215.7	480[h]	1.3×10^{-8} (20 °C)[oo]		6.84[m]	0.76[f]	4.4×10^{-5} (20 °C)[w]
Benzo[*a*]pyrene	Yellowish[c]	178.1	496[kk]	7.3×10^{-7}[mm]	1.351[pp]	6.50[n]	3.8	3.4×10^{-5} (20 °C)[w]
Benzo[*e*]pyrene	Pale yellow[c]	178.7	493[kk]	7.4×10^{-7}[qq]		6.44[n]	5.07 (23 °C)[rr]	
Perylene	Yellow to colourless[c]	277.5	503[ss]		1.35[tt]	5.3[uu]	0.4	
Anthanthrene	Golden yellow[vv]	264	547[ww]		1.39[tt]			
Benzo[*ghi*]perylene	Pale yellow-green[xx]	278.3	545[ll]	1.4×10^{-8}[ww]	1.329[20/4][xx]	7.10[q]	0.26	2.7×10^{-5} (20 °C)[w]
Indeno[1,2,3-*cd*]pyrene	Yellow[i]	163.6	536[yy]	1.3×10^{-6} (20 °C)[gg]		6.58[f]	62[f]	2.9×10^{-5} (20 °C)[w]
Dibenz[*a,h*]anthracene	Colourless[c]	266.6	524[yy]	1.3×10^{-8} (20 °C)[yy]	1.282[f]	6.50[zz]	0.5 (27 °C)[jj]	7×10^{-6}[f]
Coronene	Yellow[h]	439	525[aaa]	2.0×10^{-10}[qq]	1.371[h]		5.4[bbb]	0.14

Table 4 (contd)

Compound	Colour	Melting-point[a] (°C)	Boiling-point (°C)	Vapour pressure (Pa at 25 °C)	Density[c]	*n*-Octanol: water partition coefficient (log K_{ow})	Solubility in water at 25 °C (µg/litre)[d]	Henry's law constant at 25 °C (kPa)
Dibenzo[*a*,*e*]pyrene	Pale yellow[b]	244.4	592[vv]					
Dibenzo[*a*,*h*]pyrene	Golden yellow[j]	317	596[vv]					
Dibenzo[*a*,*i*]pyrene	Greenish-yellow[j]	282	594[vv]	3.2×10^{-10} [mm]		7.30[kk]	0.17[l]	4.31×10^{-6} [l]
Dibenzo[*a*,*l*]pyrene	Pale yellow[j]	162.4	595[vv]					

[a] From Karcheret al. (1985); Karcher (1988)
[b] From Lewis (1992)
[c] When two temperatures are given as superscripts, they indicate the specific gravity, i.e. the density of the substance at the first reported temperature relative to the density of water at the second reported temperature. When there is no value, or only one, for temperature, the datum is in grams per millilitre, at the indicated temperature, if any.
[d] From Mackay & Shiu (1977), except where noted
[e] From Budavari (1989)
[f] From National Toxicology Program (1993)
[g] From Sonnefeld et al. (1983)
[h] From Lide (1991)
[i] From IARC (1977)
[j] From Karickhoff et al. (1979)
[k] From Mackay et al. (1979)

Table 4 (contd)

[l] Calculated by Syracuse Research Center; from National Toxicology Program (1993)
[m] Calculated as per Leo et al. (1971); from US Environmental Protection Agency (1980)
[n] From Mackay & Shiu (1981)
[o] When pure, colourless with violet fluorescence; from Budavari (1989)
[p] From Hawley (1987)
[q] From National Institute for Occupational Safety and Health and Occupational Safety and Health Administration (1981)
[r] From Kruber & Marx (1938)
[s] Calculated by Karcher et al. (1991)
[t] From May et al. (1978)
[u] From Bruggeman et al. (1982)
[v] At ambient temperature; from Inokuchi & Nakagaki (1959)
[w] From Ten Hulscher et al. (1992)
[x] Personal observation by J. Jacob, Germany, on high-purity, certified reference materials
[y] From Kruber (1937)
[z] Calculated by Miller et al. (1985)
[aa] From Schuyer et al. (1953)
[bb] From IARC (1983)
[cc] From Kruber & Grigoleit (1954)
[dd] From Ehrlich & Beevers (1956)
[ee] Reported by Grimmer (1983a)
[ff] From Beilstein Institute for Organic Chemistry (1993)
[gg] Reported by Sims & Overcash (1983)
[hh] Calculated by Yalkowsky & Valvani (1979)
[ii] Calculated by White (1986)
[jj] From Davis et al. (1942)

Table 4 (contd)

[kk] From review by Bjørseth (1983); original references cited by White (1986)
[ll] Temperature not given; reported by Sims & Overcash (1983)
[mm] Calculated by National Toxicology Program (1993)
[nn] Temperature not given; unpublished result cited by Wise et al. (1981)
[oo] From US Environmental Protection Agency (1980)
[pp] From Kronberger & Weiss (1944)
[qq] From review of Santodonato et al. (1981)
[rr] Calculated by Ruepert et al. (1985)
[ss] From Verschueren (1983)
[tt] From Schwarz (1977)
[uu] From Brooke et al. (1986)
[vv] From Agency for Toxic Substances and Disease Registry (1990)
[xx] From White (1948)
[yy] Estimated from gas chromatographis retention time; from Grimmer (1983a)
[zz] From Means et al. (1980)
[aaa] From Von Boente (1955)

(Valerio et al., 1984; Lane, 1989). After photodecomposition in the presence of air and sunlight, a number of oxidative products are formed, including quinones and endoperoxides. PAH have been shown experimentally to react with nitrogen oxides and nitric acid to form the nitro derivatives of PAH, and to react with sulfur oxides and sulfuric acid (in solution) to form sulfinic and sulfonic acids. PAH may also be attacked by ozone and hydroxyl radicals present in the atmosphere. The formation of nitro-PAH is particularly important owing to their biological impact and mutagenic activity (IARC, 1984a, 1989a). In general, the above reactions are of interest with regard to the environmental fate of PAH, but the results of experimental studies are difficult to interpret because of the complexity of interactions occurring in environmental mixtures and the difficulty in eliminating artefacts during analytical determinations. These reactions are also considered to be responsible for possible losses of PAH during ambient atmospheric sampling (see section 2.4.1.1).

2.3 Conversion factors

Atmospheric concentrations of PAH are usually expressed as micrograms or nanograms per cubic meter. At 25 °C and 101.3 kPa, the conversion factors for a compound of given relative molecular mass are obtained as follows:

$$\text{ppb} = \mu g/m^3 \times 24.45/\text{relative molecular mass}$$

$$\mu g/m^3 = \text{ppb} \times \text{relative molecular mass}/24.45.$$

For example, for benzo[*a*]pyrene, 1 ppb = 10.3 $\mu g/m^3$ and 1 $\mu g/m^3$ = 0.0969 ppb.

2.4 Analytical methods

Tables 5 and 6 present as examples a limited number of methods that are applied to 'real' samples of different matrices. The methods and sources were selected, as far as possible, according to the following criteria: accessibility of the bibliographic source, completeness of the description of the procedure, practicability with common equipment for this type of analysis (even if experienced personnel are required), recency, and whether it is an official, validated, or recommended method.

2.4.1 Sampling

2.4.1.1 Ambient air

The physical state of PAH in the atmosphere must be considered when selecting the sampling apparatus. Compounds with five or more rings are almost exclusively adsorbed on suspended particulate matter, whereas

Table 5. Analytical methods for polycyclic aromatic hydrocarbons in air

Matrix	Sampling, extraction	Clean-up	Analysis	Limit of detection	Reference
Ambient air	Sampling on GF+PUF, at 45 m^3/h; Soxhlet extraction with cyclohexane	Liquid–liquid partition with cyclohexane: H_2O:DMSO, then CC with SiO_2	GC/MS		Yamasaki et al. (1982)
	Sampling on GF+PUF, at 30 m^3/h; Soxhlet extraction with petroleum ether (GF) and DCM (PUF)	CC with Al_2O_3 + SiO_2	HPLC/FL	0.01–0.7 ng/m^3	Keller & Bidleman (1984)
	Sampling on GF (particle diameter < 15 μm), at 68 m^3/h; Soxhlet extraction with cyclohexane, DCM, and acetone	TLC with SiO_2	HPLC/UV + FL	0.01–0.3 ng/m^3	Greenberg et al. (1985)
	Sampling on GF at 83 m^3/h; sonication (cyclohexane)	TLC with SiO_2	GC/FID		Valerio et al. (1992)
Emissions (municipal incinerator)	Sampling by glass wool, condenser, and XAD-2; extraction with acetone (glass-wool and XAD-2, by Soxhlet)	Liquid–liquid partition with DMF	GC/FID	10 ng/m^3	Colmsjö et al. (1986a)
Vehicle exhaust	Sampling by GF and condenser; liquid-liquid partition with acetone:H_2O:cyclohexane and DMF:H_2O:cyclohexane	CC with SiO_2 and Sephadex LH-20	GC/FID	2.5–20 ng per test	Grimmer et al. (1979)
	Sampling in dilution tunnel by PTFE-coated GF and condenser; Soxhlet extraction of filter (DCM) and condensate (acetone); remaining aqueous phase extracted with DCM	Liquid–liquid partition with cyclohexane: H_2O:DMF	GC/FID or GC/MS		Westerholm et al. (1988)

Table 5 (contd)

Matrix	Sampling, extraction	Clean-up	Analysis	Limit of detection[a]	Reference
Indoor air	Sampling on GF (particle diameter < 10 μm) at 10 l/min; sonication (cyclohexane)	TLC with acetyloxylated cellulose	Spectrofluorescence (benzo[*a*]pyrene only)		Lioy et al. (1988)
	Sampling on quartz-fibre filtre and XAD-4 at 226 l/min; Soxhlet extraction with DCM		GC/MS		Chuang et al. (1991)
	Sampling on PTFE-coated GF at 20 l/minfor 24 h; Soxhlet extraction with DCM	filtration; then CC SiO_2 cartridge), optional	HPLC/FL	0.02–0.12 ng/m^3[b]	Daisey & Gundel (1993)
	Sampling on GF and PUF, at 20 litres/min for 24 h; Soxhlet extraction (10% ether: *n*-hexane)		GC/FID, GC/MS or HPLC/UV + FL		US Environmental Protection Agency (1990)
Workplace air	Sampling on PTFE filter and XAD-2 at 2 l/min; sonication or Soxhlet extraction of filter[c], extraction of XAD-2 with toluene (for GC) or acetonitrile (for HPLC)		GC/FID	0.3–0.5 μg per sample	NIOSH (1994a,b)
			HPLC/UV + FL	0.05–0.8 μg per sample	

Table 5 (contd)

Matrix	Sampling, extraction	Clean-up	Analysis	Limit of detection[a]	Reference
Workplace air	Sampling on filter (GF, quartz fibre, PTFE or silver membrane) at 2 litres/min; sonication or Soxhlet extraction with cyclohexane or toluene	CC (XAD-2)	GC/FID	~ 0.5 µg/m³	German Research Commission (1991)
Tobacco smoke	Sampling by acetone trap; solvent partition scheme (acids/bases/neutral compounds/PAH)	CC (SiO_2 + Sephadex LH-20); then HPLC/UV	GC/MS + NMR	ng/cigarette	Lee et al. (1976b)

GF, glass fibre; PUF, polyurethane foam; DMSO, dimethyl sulfoxide; CC, column chromatography; GC, gas chromatography; MS, mass spectrometry; DCM, dichloromethane; HPLC, high-performance liquid chromatography; FL, fluorescence detection; TLC, thin-layer chromatography; UV, ultraviolet detection; FID, flame-ionization detection; DMF, *N*-dimethylformamide; PTFE, polytetrafluoroethylene; NMR, nuclear magnetic resonance

[a] Various PAH

[b] The following PAH can be determined: fluoranthene, pyrene, chrysene, benzo[*e*]pyrene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene.

[c] Appropriate solvent must be determined by recovery tests on specific samples.

Table 6. Analytical methods for polycyclic aromatic hydrocarbons in matrices other than air

Matrix	Extraction	Clean-up	Analysis	Limit of detection[a]	Reference
Tap-water	Preconcentration on PUF; extraction (with acetone and cyclohexane)	Liquid–liquid partition with cyclohexane: H_2O:methanol and cyclohexane: H_2O: DMSO; then CC (Florisil)	GC/FID or TLC (Al_2O_3; acetyl cellulose) with FL detector	0.1 ng/litre	Basu & Saxena (1978a)
Groundwater	Liquid–liquid partition with DCM	CC (SiO_2), if needed	GC/FID GC/MS HPLC/UV + FL	µg/litre level 10 µg/litre 0.01–2 µg/litre	US Environmental Protection Agency (1986a)
Wastewater	Liquid–liquid partition with DCM	CC (SiO_2), if needed	GC/FID or HPLC/UV + FL	0.01–0.2 µg/litre (by HPLC)	US Environmental Protection Agency (1984a)
Seawater	Liquid–liquid partition with *n*-hexane or CCl_4	CC (SiO_2 + Al_2O_3)	GC/FID or HPLC/UV		Desideri et al. (1984)
Soil	Sonication with DCM	CC (Al_2O_3); then liquid–liquid partition (*n*-hexane:H_2O:DMSO)	GC/MS	1 µg/kg	Vogt et al. (1987)
	Soxhlet extraction with DCM	CC (Florisil cartridge)	HPLC/UV + FL	1 µg/kg	Jones et al. (1989a)

Table 6 (contd)

Matrix	Extraction	Clean-up	Analysis	Limit of detection[a]	Reference
Sediment	Soxhlet extraction with DCM	CC (SiO_2 + Sephadex LH20)	HPLC/DAD/MS		Quilliam & Sim (1988)
	Sonication with acetone: *n*-hexane	CC (Florisil)	HPLC/UV + FL	1–160 µg/kg	Marcus et al. (1988)
Meat and fish products (I), vegetable oils (II),and sewage sludge (III)	(I) digestion (alcoholic KOH), then liquid–liquid partition (methanol: H_2O:cyclohexane) (II) dissolution in cyclohexane (III) refluxing with acetone	Liquid–liquid partition with cyclohexane: H_2O:DMF); then CC (SiO_2 + Sephadex LH20)	GC/FID	2.5–20 ng/ sample	Grimmer & Böhnke (1979b)
Food (total diet)	Refluxing with alcoholic KOH, extraction with isooctane	Liquid–liquid partition (isooctane:H_2O:DMF); then CC (SiO_2 cartridge)	HPLC/FL	0.002–0.7 µg/kg	Dennis et al. (1983)
	Saponification with alcoholic KOH, extraction with cyclohexane	CC (SiO_2)	HPLC/FL	0.03–2 µg/kg	de Vos et al. (1990)
	Saponification with alcoholic KOH, extraction with isooctane	CC (Florisil); then liquid–liquid partition isooctane:H_2O:DMSO)	TLC/UV+FL	0.02 µg/kg (benzo[*a*]pyrene)	Howard (1979); Fazio (1990)
Seafood	Digestion with alcoholic KOH, extraction with TCTFE	CC (Al_2O_3 + SiO_2 + C18 cartridge)	HPLC/FL	0.01–0.6 µg/kg	Perfetti et al. (1992)

Table 6 (contd)

Matrix	Extraction	Clean-up	Analysis	Limit of detection[a]	Reference
Smoked food	Digestion with alcoholic KOH, extraction with TCTFE	CC (Al_2O_3 + SiO_2); liquid–liquid partition (cyclohexane:H_2O:DMSO)	HPLC/UV+FL	0.03–0.4 µg/kg	Joe et al. (1984)
	Refluxing with cyclohexane or TCTFE, extraction with methanol:H_2O	Liquid-liquid partition with cyclohexane:H_2O: DMF); then CC (SiO_2)	TLC/FL[b] (only benzo[*a*]pyrene)	0 0.5 ng/kg	IUPAC (1987)
Solid waste	Soxhlet extraction with DCM or sonication with DCM:acetone	CC (SiO_2), if needed	GC/FID GC/MS HPLC/UV + FL	µg/kg level 1–200 mg/kg µg/kg level	US Environmental Protection Agency (1986b)
Mineral oil and fuel	Liquid–liquid partition with cyclohexane:H_2O:DMF)	CC (SiO_2 + Sephadex LH20)	GC/FID	100 ng/kg	Grimmer & Böhnke (1979a)
Medicinal oil (cyclohexane: H_2O:DMF)	Liquid–liquid partition	CC (SiO_2 + Sephadex LH20)	HPLC/FL + GC/FID	0.2–200 ng/kg	Geahchan et al. (1991)
Plants	Sonication (acetonitrile), extraction with pentane	CC (SiO_2)	GC/FID		Coates et al. (1986)

Table 6 (contd)

Matrix	Extraction	Clean-up	Analysis	Limit of detection[a]	Reference
Urine	Adjusted to pH3, extraction in C_{18} cartridge, metabolites reduced with hydriodic acid	CC (SiO_2 cartridge)	HPLC/FL[c]		Becher & Bjørseth (1983)
Urine and faeces	Addition of HCl, refluxing with toluene, addition of methanol and diazomethanol in ether (faeces saponified before acidification)	CC (SiO_2) + Sephadex LH20	GC/MS[d]		Jacob et al. (1989)
Tissue	Homogenization (benzene: *n*-hexane)	CC (Florisil)	GC/MS	5-50 µg/kg	Liao et al. (1988)
Skin[e]	Sonication of exposure pads with DCM, centrifugation		HPLC/FL	6 ng/cm²	Jongeneelen et al. (1988a)

PUF, polyurethane foam; DMSO, dimethyl sulfoxide; CC, column chromatography; GC, gas chromatography; FID, flame ionization detection; FL, fluorescence detection; DCM, dichloromethane; MS, mass spectrometry; UV, ultraviolet detection; DAD, diode-array detector; DMF, *N*-dimethylformamide; TLC, thin-layer chromatography; TCTFE, 1,1,2-trichlorotrifluoroethane

[a] Various PAH

[b] Benzo[*a*]pyrene content estimated to be > 0.6 µg/kg (screening method)

[c] Determination of unmetabolized and metabolized PAH

[d] Determination of pyrene and 1-hydroxypyrene

[e] Measurement of skin contamination with soft polypropylene exposure pads mounted on skin sites

lower-molecular-mass PAH are partially or totally present in the vapour phase (Coutant et al., 1988). When ambient air is monitored, it is common practice to monitor only particle-bound PAH (Menichini, 1992a), probably because of the increased work involved in trapping volatile compounds, both in assembling the sampling unit and in analysing samples, and also because lighter compounds are of lesser toxicological interest. Of the PAH that are classified as 'probably' and 'possibly' carcinogenic to humans (IARC, 1987), only benz[*a*]anthracene is found at significant levels in the vapour phase (Van Vaeck et al., 1984; Coutant et al., 1988; Back et al., 1992).

Sampling is generally performed by collecting total suspended particulate matter for 24 h on glass-fibre filters by means of high-volume samplers. Other filters that have been used are quartz fibres (Hawthorne et al., 1992), polytetrafluoroethylene (PTFE) membranes (Benner et al., 1989; Back et al., 1992), and, in comparisons, PTFE-coated glass fibres (Lindskog et al., 1987; De Raat et al., 1990). The effects of these materials on the decomposition of PAH during sampling have been compared (see section 2.2). Some studies indicated that higher recoveries are obtained with PTFE and PTFE-coated filters (Lee et al., 1980a; Grosjean, 1983); however, more recent investigations did not confirm this finding (Lindskog et al., 1987; Ligocki & Pankow, 1989; De Raat et al., 1990). Moreover, when cellulose acetate membrane filters were compared with glass-fibre filters, they had similar efficiency for collecting heavier PAH, but the former had greater efficiency for collecting three- and four-ring compounds (Spitzer & Dannecker, 1983).

The most widely used method for trapping vapour-phase PAH is adsorption on plugs of polyurethane foam located behind the filter (Keller & Bidleman, 1984; Chuang et al., 1987; De Raat et al., 1987a; Benner et al., 1989; Hawthorne et al., 1992). This method is widely accepted, probably because of the low pressure drop, the low blanks, the low cost, and ease of handling. Among the other sorbents tested (see also reviews by Leinster & Evans, 1986; Davis et al., 1987), further polymeric materials have received particular attention, including Amberlite XAD-2 resin, which is a valid alternative to polyurethane foam (Chuang et al., 1987), Porapak PS, which has been successfully tested in combination with a silanized glass-fibre filter at a flow rate of 2 m^3/h (Jacob et al., 1990a), and Tenax® (Back et al., 1992).

The trapped vapours contain both the PAH that were initially present in the vapour phase and those already collected on the filter and volatilized during sampling (the 'blowing-off' effect) (Van Vaeck et al., 1984; Coutant et al., 1988). The amount of PAH found in the vapour phase increases with ambient temperature (Yamasaki et al., 1982). Samplers incorporating an annular denuder, as well as a filter and back-up trap, have been used to investigate phase distribution and artefact formation (Coutant et al., 1988, 1992).

Sampling times are restricted to 24 h in order to avoid sample degradation and losses. Grimmer et al. (1982) proposed a useful method for controlling

losses due to chemical degradation and volatilization from filters which is based on the invariability of PAH profiles (i.e. the ratio of all PAH to one another) at different collection times. The adsorption of gas-phase PAH onto a quartz-fibre filter has been investigated as a possible sampling artefact (Hart & Pankow, 1994); the results suggested that overestimation of particle-associated PAH can be avoided by replacing quartz-fibre filters with a PTFE membrane filters, or can be corrected by using back-up quartz-fibre filters.

Elutriators and cascade impactors have been used to achieve particle size-selective sampling of PAH (Menichini, 1992a). Instruments designed as additions to high-volume samplers are available, including 'PM10' inlets, which allow collection of airborne particles with a 50% cutoff at the aerodynamic diameter of 10 μm (US Environmental Protection Agency, 1987a; Lioy et al., 1988; Hawthorne et al., 1992), and cascade impactors (Van Vaeck et al., 1984; Catoggio et al., 1989).

When PAH are collected in indoor air, samplers operating at 20 or 200 litre/min are commonly used. The filter and sorbent materials are those used for outdoor air (Wilson et al., 1991; see also Table 5).

The sampling step is by far the most important source of variability in the results of atmospheric PAH determination. Most investigations are difficult to compare because of differences in factors such as season, meteorological conditions, time of day, number and characteristics of sampling sites, and sampling parameters (Menichini, 1992a). Passive biological sampling has been investigated as an approach to long-term sampling of atmospheric PAH (Jacob & Grimmer, 1992), and preliminary correlation factors have been determined by comparing the PAH profiles in biological (plants, particularly) and air samples. Of the matrices tested, spruce sprouts were found to be the most suitable.

2.4.1.2 *Workplace air*

The general considerations described for ambient air are also valid for the working environment. Less volatile PAH may be retained than in ambient air because of the high temperatures that are often found at the workplace. In the potroom of an aluminium plant where Søderberg electrodes were used, 42% of benz[*a*]anthracene was found in the vapour phase (Andersson et al., 1983), and in an iron foundry at a site where the temperature of the PAH source was 600–700 °C, four- to seven-ring PAH represented about 70% of the total in the vapour phase (Knecht et al., 1986).

Glass-fibre or PTFE filters are usually used to collect particle-bound PAH. A number of back-up systems can be used to efficiently trap volatile PAH, including liquid impingers and solid sorbents such as Tenax®-GC, Chromosorb, and XAD-2 (Bjørseth & Becher, 1986; Davis et al., 1987). The latter seems to be the most practical. The US National Institute for Occupational Safety and

Health (1994a,b) recommended use of a PTFE-laminated membrane followed by a tube containing two sections of XAD-2. For sampling in bright sunlight, opaque or foil-wrapped filter cassettes can be used to prevent degradation.

The exposure of workers is estimated by taking air samples at various locations in the workplace or by personal sampling, in which workplace air is pumped through a filter attached to clothing close to the breathing zone for a specified time. Both procedures provide an estimate and not a precise measurement of an individual's exposure.

2.4.1.3 *Combustion effluents*

The validity of a collected sample, i.e. the degree to which it reflects the 'true' composition of the emission, is a crucial factor in the determination of PAH in emissions. The problems associated with efficient collection of volatile PAH are enhanced when sampling combustion effluents, such as stack gases and vehicle exhausts, because of the elevated temperatures at sampling positions.

A sampling device for stack gases is constituted by a glass- or quartz-fibre filter, followed by a special unit which generally consists in a cooler for collecting condensable matter and an adsorbent cartridge (Colmsjö et al., 1986a; Funcke et al., 1988). Tenax® has been used as an adsorbent (Jones et al., 1976), but XAD-2 seems to be more suitable (Warman, 1985) and is generally preferred. Two sampling procedures have been described in detail by the US Environmental Protection Agency (1986c). In the first ('Modified method 5 sampling train'), the unit basically includes a glass- or quartz-fibre filter kept at around 120 °C, a condenser coil that conditions the gas at a maximum of 20 °C, and a bed of XAD-2 jacketed to maintain the internal gas temperature at about 17 °C. The second ('Source assessment sampling system') is often used for stationary investigations (Warman, 1985). The apparatus consists of a stainless-steel probe, which enters an oven containing the filter, preceded by three cyclone separators in series, with cutoff diameters of 10, 3, and 1 μm; the volatile organic compounds are cooled and trapped on XAD-2. The sorbent is followed by a condensate collection trap and an impinger train.

Motor vehicle exhausts are sampled under laboratory conditions, by chassis or engine dynamometer testing. Standard driving cycles are employed to simulate on-road conditions (Stenberg, 1985; see also section 3.2.7.2).

Two basic techniques have been used to collect, sample, and analyse exhaust (Levsen, 1988; IARC, 1989a). In the first—raw gas sampling—the exhaust pipe is connected directly to the sampling apparatus; undiluted emissions are cooled in a condenser and then allowed to pass through a filter for collection of particulates (Grimmer et al., 1979, 1988a; Society of German Engineers, 1989). A second technique—dilution tube sampling—is now often used, in which hot exhaust is diluted with filtered cold air in a tunnel, from

which samples are collected isokinetically. This technique simulates the process of dilution that occurs under real conditions on the road (US Environmental Protection Agency, 1992a).

Particles are almost always collected on glass-fibre, glass-fibre with PTFE binder, quartz-fibre filters, or PTFE membranes; the latter have been reported to be particularly efficient and chemical inert (Lee & Schuetzle, 1983). Glass-fibre filters impregnated with liquid paraffin are also used (Grimmer et al., 1979; Society of German Engineers, 1989). Vapour-phase PAH (Stenberg, 1985) may be collected by cryo-condensation (Stenberg et al., 1983) or on an adsorbent trap with a polymeric material such as XAD-2 (Lee & Schuetzle, 1983).

Artefacts may be introduced during collection on filters as a result of chemical conversion of PAH, particularly into nitro-PAH and oxidation products (Lee & Schuetzle, 1983; Schuetzle, 1983; IARC, 1989a). These effects have not been fully evaluated.

2.4.1.4 Water

The concentrations of PAH in uncontaminated groundwater supplies and in drinking-water are generally very low, at 0.1 and 1 ng/litre (see sections 5.1.2.1 and 5.1.2.2). This implies that serious errors arising from adsorption losses and contamination occur during collection and storage of samples or that a preconcentration step may be needed to enrich the sample. It is recommended that sampling be performed on-site, directly in the extraction vessel (Smith et al., 1981).

Various solid sorbents have been successfully used for preconcentration (Smith et al., 1981), including Tenax®-GC, prefiltered if necessary (Leoni et al., 1975); XAD resins (Griest & Caton, 1983); open-pore polyurethane foam (Basu et al., 1987); and prepacked disposable cartridges of bonded-phase silica gel (Chladek & Marano, 1984; Van Noort & Wondergem, 1985a). Solid sorbents have limitations when the sample contains suspended material, since adsorbed PAH may be lost by filtration (Van Noort & Wondergem, 1985a).

2.4.1.5 Solid samples

Some foodstuffs (Liem et al., 1992), soil, sediment, tissues, and plants usually require homogenization before a sample is extracted.

2.4.2 Preparation

As most environmental samples contain only small amounts of PAH, sophisticated techniques are required for their detection and quantification. Therefore, efficient extraction from the sample matrix is usually followed by one or more purification steps, so that the sample to be analysed is as free as

possible from impurities and interference. Many extraction and purification techniques and combinations ('isolation schemes') have been described, validated, and recommended, but no single scheme is commonly recognized as 'the best' for a given matrix. The isolation schemes have been classified according to groups of matrices (Jacob & Grimmer, 1979; Grimmer, 1983a), as summarized briefly below.

PAH are extracted from a sample (Lee et al., 1981; Santodonato et al., 1981; Grimmer, 1983a; Griest & Caton, 1983) with:

— a Soxhlet apparatus, from filters loaded with particulate matter, vehicle exhausts, or sediments;
— directly by liquid-liquid partition, for water samples; or
— after complete dissolution (e.g. fats and vegetable and mineral oils) or alkaline digestion of samples (e.g. meat products) by a selective solvent such as *N,N*-dimethylformamide (Natusch & Tomkins, 1978) or dimethyl sulfoxide. Complete extraction of PAH from samples such as soot emitted by diesel engines, carbon blacks, and other carbonaceous materials is particularly difficult.

Extraction of PAH from soil, sediment, sewage sludge, and vehicle exhaust particulates by refluxing with various solvents has been investigated. In all cases, toluene was found to be the most efficient solvent, especially for vehicle exhaust (Jacob et al., 1994).

As an alternative to Soxhlet extraction, ultrasonic extraction (Griest & Caton, 1983) has advantages in terms of reduced time of extraction (minutes versus hours) and superior recovery efficiency and reproducibility, particularly for solid samples and filters loaded with particulate matter. Comparisons of techniques depend, however, on the matrix, solvent, and experimental conditions. Supercritical fluid extraction (Langenfeld et al., 1993) has gained attention as a rapid alternative to conventional liquid extraction from polyurethane foam sorbents (Hawthorne et al., 1989a), soil (Wenclawiak et al., 1992), and other environmental solids such as urban dust, fly ash, and sediment (Hawthorne & Miller, 1987). This technique can also be directly coupled with on-column gas chromatography (see section 2.4.3.1); the extract is quantitatively transferred onto the gas chromatographic column for a rapid (< 1 h) analysis with maximal sensitivity. This technique has been used for urban dust samples (Hawthorne et al., 1989b).

Extracted samples are usually purified from interfering substances by adsorption column chromatography. The classical sorbents, alumina and silica gel, are widely used. In addition, the hydrophobic Sephadex LH-20 has been found to be suitable for isolating PAH from nonaromatic, nonpolar compounds, which is important if the sample is analysed by gas chromatography (Grimmer & Böhnke, 1979a); it has also been used in partition chromatography as a carrier of the stationary phase, to separate PAH from alkyl derivatives (Grimmer & Böhnke, 1979b). Chromatography on silica gel and Sephadex is often combined (Jacob & Grimmer, 1979; Grimmer, 1983a).

Clean-up has also been achieved by eluting extracted samples through XAD-2 (soil samples: Spitzer & Kuwatsuka, 1986), XAD-2 and Sephadex LH-20 in series (foods: Vaessen et al., 1988), or Florisil (food, water, and sediment samples: references given in Table 6).

Conventional chromatographic columns may be substituted by prepacked commercial cartridges, which have advantages in terms of time and solvents consumed and reproducibility. For example, silica cartridges have been used to purify foodstuffs (Dennis et al., 1983), urine (Becher & Bjørseth, 1983), vehicle emissions (Benner et al., 1989), mineral oil mist (Menichini et al., 1990), and atmospheric samples (Baek et al., 1992); soil samples have been cleaned up on Florisil cartridges (Jones et al., 1989a).

Preparative thin-layer chromatography is also used for, e.g. air particulates (see Table 5) and vegetable oils (Menichini et al., 1991a).

Handling of samples in the absence of ultraviolet radiation is recommended at all stages in order to avoid photodecomposition of PAH (Society of German Engineers, 1989; US Environmental Protection Agency, 1990; US National Institute for Occupational Safety and Health, 1994a,b). It is also generally recommended that possible sources of interference and contamination be controlled, particularly from solvents (US Environmental Protection Agency, 1984a, 1986b, 1990), and that samples be refrigerated until extraction (US Environmental Protection Agency, 1984a; US National Institute for Occupational Safety and Health, 1994a,b).

2.4.3 Analysis

PAH are now routinely identified and quantified by gas chromatography or high-performance liquid chromatography (HPLC). Each technique has a number of relative advantages. Both are rather expensive, particularly HPLC, and require qualified operating personnel; nevertheless, they are considered necessary in order to analyse 'real' samples for a large number of PAH with accuracy and precision.

2.4.3.1 Gas chromatography

Excellent separation (≥ 3000 plates per meter) is obtained by the use of commercially available fused silica capillary columns, making it possible to analyse very complex mixtures containing more than 100 PAH.

The most widely used stationary phases are the methylpolysiloxanes: especially SE-54 (5% phenyl-, 1% vinyl-substituted) and SE-52 (5% phenyl-substituted), but SE-30 and OV-101 (unsubstituted), OV-17 (50% phenyl-substituted), Dexsil 300 (carborane-substituted) and their equivalent phases are also used. Chemically bonded phases are used increasingly because they can be rinsed to restore column performance and undergo little 'bleeding' at the

high temperatures of analysis (about 300 °C) that are required for determining high-boiling-point compounds.

Nematic liquid crystal phases (Bartle, 1985) have also been used to separate some isomeric compounds that are poorly resolved by siloxane phases, such as chrysene and triphenylene on *N,N'*-bis(*para*-methoxybenzylidene)-α,α'-bi-*para*-toluidine (Janini et al., 1975) and *N,N'*-bis(*para*-phenylbenzylidene)-α,α'-bi-*para*-toluidine (Janini et al., 1976).

Splitless or on-column injection is necessary to gain sensitivity in trace analysis, the latter being preferred as it allows better reproducibility. Flame ionization detectors are almost always used because of the excellent linearity, sensitivity, and reliability of their response. Since the signal is related linearly to the carbon mass of the compound, PAH are recorded in proportion to their quantities, and the chromatogram shows the quantitative composition of the sample directly. Because flame ionization detectors are non-selective, samples for gas chromatography must be highly purified. Peak identification, which is done routinely from data on retention, must be confirmed by analysing samples on a different gas chromatographic column, by an independent technique, such as HPLC, or by directly coupling a mass spectrometric detector to the gas chromatograph (Lee et al., 1981; Olufsen & Bjørseth, 1983; Bartle, 1985; Hites, 1989).

Mass spectrometers have gained wide acceptance. They are powerful tools for identifying compounds, especially when commercially available libraries of reference spectra are used to match the spectra obtained and to control the purity of a compound. As isomeric compounds often have indistinguishable spectra, however, the final assignment must also be based on retention.

On-line coupling of liquid chromatography, capillary gas chromatography, and quadrupole mass spectrometry has been used to determine PAH in vegetable oils (Vreuls et al., 1991).

2.4.3.2 *High-performance liquid chromatography*

The packing material considered most suitable for separating PAH consists of silica particles chemically bonded to linear C18 hydrocarbon chains; selection of the appropriate phase has been discussed in detail by Wise et al. (1993). Typically, 25-cm columns packed with 5-μm particles are used in the gradient elution technique, and the mobile phase consists of mixtures of acetonitrile and water or methanol and water ('reversed-phase HPLC'). As the efficiency of separation that can be achieved with HPLC columns is much lower than that with capillary gas chromatography, HPLC is generally less suitable for separating samples containing complex PAH mixtures.

The advantages of HPLC derive from the capabilities of the detectors with which it is used. Those most widely used for PAH are ultraviolet and fluorescence detectors, generally arranged in series, with flow-cell photometers

or spectrophotometers. Both, but especially the latter, are highly specific and sensitive: the detection limits with fluorescence are at least one order of magnitude lower than those with ultraviolet detection. The specificity of fluorescence detectors allows the determination of individual PAH in the presence of other nonfluorescing substances. In addition, since different PAH have different absorptivity or different fluorescence spectral characteristics at given wavelengths, the detectors can be optimized for maximal response to specific compounds. This may prove advantageous in the identification of unresolved components. In particular, wavelength-programmed fluorescence detection, to measure changes in excitation and emission wavelengths during a chromatographic run (Hansen et al., 1991a), is being used for the analysis of environmental samples (Wise et al., 1993). HPLC is suitable to a limited degree for lower-molecular-mass compounds like naphthalene, acenaphthene, and acenaphthylene, for which the detection limits are relatively high (US Environmental Protection Agency, 1984a).

Owing to the selectivity of packing materials, various isomers that cannot be separated efficiently on the usual capillary gas chromatographic columns can be resolved at baseline and identified by HPLC. Such isomers include the pairs chrysene–triphenylene and benzo[*b*]fluoranthene–benzo[*k*]fluorathene (Wise et al., 1980). Coupling of a mass spectrometer to HPLC has also been used in detecting PAH (e.g., Quilliam & Sim, 1988).

As much information on isomeric structure can be obtained from spectra seen during the elution of chromatographic peaks, an ultraviolet diode-array detector has been used to confirm peaks (Dong & Greenberg, 1988; Kicinski et al., 1989). For applications of HPLC to determination of PAH, reference should be made to published reviews (Lee et al., 1981; Wise, 1983, 1985).

2.4.3.3 *Thin-layer chromatography*

Thin-layer chromatography is commonly used only for identifying individual compounds, such as benzo[*a*]pyrene, during screening (IUPAC, 1987) or for identifying selected PAH, such as the six PAH that WHO (1971) recommended be determined in drinking-water (Borneff & Kunte, 1979). It is an inexpensive, quick analytical technique but has low separation efficiency. The last parameter is improved by two-dimensional processes (see, e.g. Borneff & Kunte, 1979). Quantification may be done by spectrophotometric or spectrofluorimetric methods in solution after the scrubbed substance spot has been extracted (Howard, 1979; Fazio, 1990) or *in situ* by scanning spectrofluorimetry (Borneff & Kunte, 1979).

Acetylated cellulose is the adsorbent that has been used most widely for one-step separation of PAH fractions, and mixed aluminium oxide and acetylated cellulose have been used for two-dimensional development (Daisey, 1983).

2.4.3.4 *Other techniques*

A number of unconventional instruments and techniques based on spectroscopic principles have been developed as possible alternatives to the chromatographic methods for PAH. Most of them are, however, expensive, require skilled personnel, and are not yet considered useful for the practising analyst (Wehry, 1983; Vo-Dinh, 1989).

Low-temperature luminescence in frozen solutions ('Shpol'skii effect') has been used for various environmental samples, particularly to identify methylated PAH isomers (Garrigues & Ewald, 1987; Saber et al., 1987). This technique was used widely in the countries of former Soviet Union (Dikun, 1967). Synchronous luminescence and room temperature phosphorimetry have been reported to be simple, cost-effective techniques for screening PAH (Vo-Dinh et al., 1984; Abbott et al., 1986).

Infrared analysis, particularly Fourier transform infrared spectroscopy coupled to gas chromatography (Stout & Mamantov, 1989), and capillary supercritical fluid chromatography (Wright & Smith, 1989) have also been used. Various environmental samples have been analysed by packed column supercritical fluid chromatography, with rapid separation of PAH (Heaton et al., 1994).

2.4.4 *Choice of PAH to be quantified*

The choice of PAH depends on the purpose of the measurement. For example, carcinogenic PAH are of interest in studies of human health, but other, more abundant PAH may be of interest in ecotoxicological studies. Quantification of a number of PAH is advantageous when the profiles are to be correlated with sources and/or effects.

Table 7 lists the PAH that are required or recommended to be determined at national or international levels. According to an EEC (1980) Directive, which followed a WHO (1971) recommendation, the concentrations of six reference compounds (also known as 'Borneff PAH') must be measured in drinking-water in order to check its compliance with the cumulative limit value for the PAH class. The choice of these six PAH by WHO was not based on toxicological considerations but on the fact that analytical investigations were then largely confined to these relatively easily detected compounds (WHO, 1984).

The method required by the US Environmental Protection Agency (1984a) for the analysis of municipal and industrial wastewater covers the determination of 16 'priority pollutant PAH' considered to be representative of the class. Outside the USA, this list of compounds is often taken as a reference list for the analysis of various environmental matrices.

Table 7. Some polycyclic aromatic hydrocarbons required or recommended for determination by various authorities

Compound	WHO/EEC[a] (drinking-water)	US EPA[b] (waste water)	European Aluminium Association[c]	Italy[d] (air)	Norway[e]	
					Health	Environment
Acenaphthene		X				
Acenaphthylene		X				
Anthracene		X	X			X
Anthanthrene					X	X
Benz[*a*]anthracene		X	X	X	X	X
Benzo[*a*]fluorene			X			
Benzo[*a*]pyrene	X		X	X	X	X
Benzo[*b*]fluoranthene	X	X	X	X	X	X
Benzo[*b*]fluorene			X			
Benzo[*c*]phenanthrene					X	X
Benzo[*e*]pyrene			X			
Benzo[*ghi*]perylene	X	X	X			X
Benzo[*j*]fluoranthene		X		X	X	X
Benzo[*k*]fluoranthene	X	X	X	X	X	X
Chrysene		X	X		X	X
Cyclopenta[*cd*]pyrene					X	X
Dibenzo[*a,e*]pyrene			X		X	X
Dibenz[*a,h*]anthracene		X	X	X	X	X
Dibenzo[*a,h*]pyrene			X		X	X
Dibenzo[*a,i*]pyrene			X		X	X
Dibenzo[*a,l*]pyrene					X	X
Fluoranthene	X	X	X			X
Fluorene		X				
Indeno[1,2,3-*cd*]pyrene	X	X	X	X	X	X
Naphthalene		X				X
Phenanthrene		X	X			X
Pyrene		X	X			X
Triphenylene			X			

[a] Recommended by WHO (1971) and required by an EEC (1980) Directive

[b] Required by the US Environmental Protection Agency (1984a) for the analysis of municipal and industrial wastewater

[c] Recommended by the European Aluminium Association, Environmental Health and Safety Secretariat (1990)

[d] Recommended by the Italian National Advisory Toxicological Committee for health-related studies (Menichini, 1992b)

[e] Recommended at the International Workshop on polycyclic aromatic hydrocarbons (State Pollution Control Authority and Norwegian Food Control Authority, 1992) for studies of health and on the environment

The European Aluminium Association (1990) recommended that 19 PAH be determined in all samples from operations in the aluminium industry. The list is based on the PAH composition of emissions from aluminium smelters, the IARC classification of carcinogenicity, and existing official lists. The Italian National Advisory Toxicological Committee (Menichini, 1992b) recommended inclusion of the determination of seven PAH in health-related investigations on the basis of their 'probable' or 'possible' carcinogenicity to humans (IARC, 1987) and their occurrence in the environment. An International Workshop on PAH (State Pollution Control Authority and Norwegian Food Control Authority, 1992) recommended that 15 carcinogenic PAH be determined in studies of health and another six abundant PAH in studies of the environment.

3. SOURCES OF HUMAN AND ENVIRONMENTAL EXPOSURE

Appraisal

Coal and crude oils contain polycyclic aromatic hydrocarbons (PAH) in considerable concentrations owing to diagenetic formation in fossil fuels. The main PAH produced commercially are naphthalene, acenaphthene, anthracene, phenanthrene, fluoranthene, and pyrene. The release of PAH during production and processing, predominantly of plasticizers, dyes, and pigments, is of only minor importance. Most PAH enter the environment via the atmosphere from incomplete combustion processes, such as:

- *processing of coal and crude oil: e.g. refining, coal gasification, and coking;*
- *heating: power plants and residential heating with wood, coal, and mineral oil;*
- *fires: e.g. forest, straw, agriculture, and cooking;*
- *vehicle traffic; and*
- *tobacco smoking.*

Industrial processes such as coal coking, aluminium, iron and steel production, and foundries make important contributions to the levels of PAH in the environment. An important indoor source of exposure to airborne PAH, especially in developing countries, is cooking fumes (see section 5.2).

The hydrosphere and the geosphere are affected secondarily by wet and dry deposition. PAH are released directly into the hydrosphere, for example during wood preservation with creosotes. Deposition of contaminated refuse like sewage sludge and fly ash may cause further emissions into the geosphere.

It is very difficult to identify a source on the basis of the ratio of the measured concentrations of different individual PAH, and such studies are in most cases inconclusive.

3.1 Natural occurrence

In some geographical areas, forest fires and volcanoes are the main natural sources of PAH in the environment (Baek et al., 1991). In Canada, about 2000 tonnes of airborne PAH per year are attributed to natural forest fires (Environment Canada, 1994). On the basis of samples from volcanoes, Ilnitsky et al. (1977) estimated that the worldwide release of benzo[*a*]pyrene from this source was 1.2–14 t/year; no estimate was given of total PAH emissions from this source.

Coal is generally considered to be an aromatic material. Most of the PAH in coal are tightly bound in the structure and cannot be leached out, and the total PAH concentrations tend to be higher in hard coal than in soft coals, like lignite

and brown coal. Hydroaromatic structures represent 15–25% of the carbon in coal. The PAH identified include benz[*a*]anthracene, benzo[*a*]pyrene, benzo[*e*]pyrene, perylene, and phenanthrene (Neff, 1979; Anderson et al., 1986). Table 8 shows the typical contents of PAH in different crude oils, such as those derived from coal conversion or from shale.

Two rare PAH minerals have been described: the greenish-yellow, fluorescent curtisite from surface vents of hot springs at Skagg Springs, California, USA, and the bituminous mercury ore idrialite from Idria, Yugoslavia, the two main components of which are chrysene and dibenz[*a*,*h*]-anthracene. These minerals are assumed to have been formed by the pyrolysis of organic material at depths below that at which petroleum id generated (West et al., 1986).

3.2 Anthropogenic sources

3.2.1 PAH in coal and petroleum products

Commercial processing of coal leads first to coal-tars, which are further processed to yield pitch, asphalt, impregnating oils (creosotes for the preservation of wood), and residue oils such as anthracene oil (IARC, 1985). The concentration of PAH in coal-tars is generally ≤ 1%; naphthalene and phenanthrene are by far the most abundant compounds, occurring at concentrations of 5–10%. Comparable levels were detected in high-temperature coal-tar pitches. The PAH content of soots is about one order of magnitude lower, and that of carbon and furnace blacks ranges from about 1 to 500 mg/kg, pyrene being present at the highest concentration (IARC, 1984a; Nishioka et al., 1986). The PAH contents of some impregnating oils, bitumens, asphalts, and roof paints are shown in Table 9. In bitumens, PAH constitute only a minor part of the total content of polyaromatic compounds.

The concentrations of PAH in petrol and diesel fuels for vehicles and in heating oils are several parts per million. Almost all compounds are present at ≤ 1 mg/kg; only phenanthrene, anthracene, and fluoranthene are sometimes found at ≥ 10 mg/kg (Herlan, 1982). The PAH levels in unused engine lubricating oils are of the same order of magnitude. During the use of petrol-fuelled engine oils, the PAH content rises dramatically, by 30–500 times; in comparison, the total PAH levels in used diesel-fuelled engine oils were only 1.4–6.1 times greater than that in an unused sample. The major constituents of used oils are pyrene and fluoranthene, although benzo[*b*]fluoranthene, benzo[*j*]-fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, and dibenz[*a*,*h*]anthracene were also detected at considerable concentrations (IARC, 1984a; Carmichael et al., 1990).

PAH have also been found in machine lubricating and cutting oils, which is of interest for the estimation of exposure in the workplace. The concentrations were < 7 mg/kg, although phenanthrene may have been present at a higher level

Table 8. Polycyclic aromatic hydrocarbon content of crude oils from various sources

Compound	PAH content (mg/kg) in crude oil from		
	Coal[a]	Petroleum	Shale
Acenaphthene	1700/1800	147–348	147–903
Anthracene	4100	204–321	231–986
Anthanthrene	Trace/< 800	NR	0.3
Benz[*a*]anthracene	Trace/< 2200	1–7	1
Benzo[*a*]fluorene	2100/2500	11–22	53
Benzo[*a*]pyrene	< 500/< 1200	0.1–4	3–192
Benzo[*b*]fluorene	< 1500/3400	≤ 13	140
Benzo[*c*]phenanthrene	< 600/< 2200	NR	NR
Benzo[*e*]pyrene	< 1200/1300	0.5–29	1–19
Benzofluorenes[b]	< 500/< 1300	23	NR
Benzo[*ghi*]fluoranthene	3200	NR	NR
Benzo[*ghi*]perylene	4300/6600	ND–8 ND–5	1–5
Chrysene	< 1500/2500	7–26	3–52
Coronene	NR	0.2	NR
Dibenz[*a,h*]anthracene	NR	0.4-0.7	1–5
Fluoranthene	< 1900/< 3700	2–326	6-400
Fluorene	5300/9900	106-220	104–381
1-Methylphenanthrene	< 1200/< 5100	> 21	NR
Naphthalene	100/2800	402–900	203–1390
Perylene	Trace/< 600	6–31	0.3–68
Phenanthrene	12 000/20 400	> 129–322	221–842
Pyrene	14 200/35 000	2–216	18–421
Triphenylene	NR	3/13	0.5

From Guerin et al. (1978), Weaver & Gibson (1979), Grimmer et al. (1983a), Sporstøl et al. (1983), IARC (1985, 1989b)
Ranges represent at least three values; NR, not reported; ND, not detected
[a] Two crude oils from coal conversion; single measurements
[b] Isomers not specified

(Grimmer et al., 1981a; Rimatori et al., 1983; Menichini et al., 1990; Paschke et al., 1992).

PAH were detected in coloured printing oils, the concentrations of individual compounds varying between < 0.0001 and 63 mg/kg (Tetzen, 1989). By far the most abundant compounds were fluoranthene and pyrene (> 1 mg/kg); benzo[*ghi*]fluoranthene, cyclopenta[*cd*]pyrene, benz[*a*]anthracene, benzo[*c*]-phenanthrene, chrysene, triphenylene, benzo[*b+j+k*]fluoranthenes,

Table 9. Polycyclic aromatic hydrocarbon content of impregnating oils, bitumens, asphalts, and roof paints

Compound	Concentration (mg/kg)			
	Impregnating oils	Bitumens (oil-derived)	Road tar (asphalt, coal-derived)	Roof paint
Anthracene	1600–22 500	0.01–0.32	4170–14 400	2380
Anthranthrene	NR	Trace–1.8	NR	NR
Benz[*a*]anthracene	169–11 700	0.14–35	6820–24100	6640
Benzo[*a*]pyrene	45–3490	0.1–27	5110–10 400	5950
Benzo[*b*]fluoranthene	42–3630	5	4490–10 900	5420
Benzo[*e*]pyrene	65–2020	0.03–52	3300–6750	3820
Benzo[*ghi*]perylene	57–570	Trace–15	2390–2730	3270
Benzo[*k*]fluoranthene	24–2610	0.024–0.19	3170–7650	4470
Chrysene	NR	0.04–34		NR
Chrysene + triphenylene	779–12 900	NR	6820–26100	7700
Coronene	NR	0.2–2.8	NR	NR
Fluoranthene	703–85 900	0.15–5	23 500–61 900	12 100
Fluorene	8040–58 400	NR	6310–15 500	2220
Indeno[1,2,3-*cd*]pyrene	57–273	Trace	3100–3530	3320
Perylene	66–744	0.08–39	1550–2300	1730
Phenanthrene	7070–159 300	0.32–7.3	20 300–52 500	8180
Pyrene	604–46 400	0.08–38	15100–42500	8960
Triphenylene	NR	0.3–7.6	NR	NR

From IARC (1985), Lehmann et al. (1986), Knecht & Woitowitz (1990); NR, not reported; ranges represent at least three values

benzo[*a*]pyrene, benzo[*e*]pyrene, anthanthrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, and coronene were found at concentrations of < 0.5 mg/kg.

3.2.2 *Production levels and processes*

Most of the PAH considered in this monograph are formed unintentionally during combustion and other processes. Only a few are produced commercially, including naphthalene, acenaphthene, fluorene, anthracene, phenanthrene, fluoranthene, and pyrene (Franck & Stadelhofer, 1987). The most important industrial product is naphthalene (see section 3.2.3). In 1987, about 220 kt of this compound were produced in western Europe, 190 kt in eastern Europe,

170 kt in Japan, and 110 kt in the USA (Fox et al., 1988); in 1986, > 1 kt was produced in Canada (Environment Canada, 1994). In 1985, about 2.5 kt of acenaphthene and 20 kt of anthracene were produced worldwide (Franck & Stadelhofer, 1987). In 1986, 0.1–1 t anthracene and 1 t fluorene were produced in Canada (Environment Canada, 1994). In 1993, a major producer in Germany produced < 5000 t anthracene, < 1000 t acenaphthene, < 500 t pyrene, < 50 t phenanthrene, and < 50 t fluoranthene (personal communication, Rütgers-VfT AG, 1994).

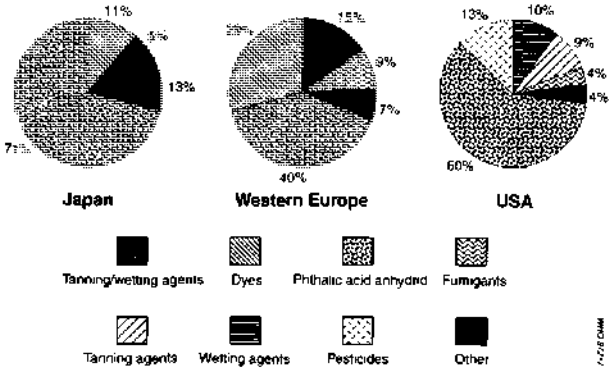
The substances are not synthesized chemically for industrial purposes but are isolated from products of coal processing, mainly hard coal-tar. The raw material is concentrated and the product purified by subsequent distillation and crystallization. Only naphthalene is sometimes isolated from pyrolysis residue oils, olefin fractions, and petroleum-derived fractions; it is also obtained by distillation and crystallization (Collin & Höke, 1985; Franck & Stadelhofer, 1987; Griesbaum et al., 1989; Collin & Höke, 1991). In the USA in 1970, the distribution of capacity was about 60% coal-tar- and 40% petroleum-derived naphthalene (Gaydos, 1981); more detailed data were not available. The purity of the technical-grade products is 90–99% (Collin & Höke, 1985; Franck & Stadelhofer, 1987; Griesbaum et al., 1989; Collin & Höke, 1991; see also Section 2).

3.2.3 *Uses of individual PAH*

The uses of commercially produced PAH are as follows (Collin & Höke, 1985; Franck & Stadelhofer, 1987; Griesbaum et al., 1989; Collin & Höke, 1991):

- *naphthalene*: main use: production of phthalic anhydride (intermediate for polyvinyl chloride plasticizers); also, production of azo dyes, surfactants and dispersants, tanning agents, carbaryl (insecticide), alkylnaphthalene solvents (for carbonless copy paper), and use without processing as a fumigant (moth repellent) (see Figure 2);
- *acenaphthene*: main use, production of naphthalic anhydride (intermediate for pigments); also, for acenaphthylene (intermediate for resins);
- *fluorene*: production of fluorenone (mild oxidizing agent);
- anthracene: main use, production of anthraquinone (intermediate for dyes); also, use without processing as a scintillant (for detection of high-energy radiation);
- *phenanthrene*: main use, production of phenanthrenequinone (intermediate for pesticides); also, for diphenic acid (intermediate for resins)
- *fluoranthene*: production of fluorescent and vat dyes;
- *pyrene*: production of dyes (perinon pigments).

Figure 2. Industrial uses of naphthalene in western Europe, Japan, and the USA, 1985



From Franck & Stadelhofer (1987)

3.2.4 *Emissions during production and processing of PAH*

The emissions of PAH during industrial production and processing in developed countries are not thought to be important in comparison with the release of PAH from incomplete combustion processes, since closed systems and recycling procedures are usually used. Few data were available.

3.2.4.1 *Emissions to the atmosphere*

No data were available.

3.2.4.2 *Emissions to the hydrosphere*

During the refining of aromatic hydrocarbons, and especially hard coal-tar, 80–190 t/year were estimated to be released to the hydrosphere in western Germany until 1987. This quantity was reduced to 8–19 t/year by the installation of new adsorption devices (sand filtration and adsorbent resin) by the two German hard coal-tar refineries in 1989 and 1991 (Klassert, 1993).

3.2.5 *Emissions during the use of individual PAH*

Only naphthalene is used directly (as a moth repellent) without further processing. On the assumption that all naphthalene-containing moth repellent is emitted into the atmosphere, the emissions would have been about 15 000 t/year in western Europe in 1986, about 4400 t/year in Japan in 1987, and about 5500 t/year in the USA in 1987 (Fox et al., 1988).

3.2.6 *Emissions of PAH during processing and use of coal and petroleum products*

Coal coking, coal conversion by gasification and liquefaction, petroleum refining, and the production and use of carbon blacks, creosote, coal-tar, and bitumen from fossil fuels may produce significant quantities of PAH (Anderson et al., 1986). A great deal of information on emissions of PAH is available in the literature; this monograph gives an overview of the most reliable values. The emission profile depends on the source, and specific emission profiles are detectable only in the direct vicinity of the corresponding source. Generally, emissions are estimated on the basis of more or less reliable databases, which are not identified in most publications. The values reported give only a rough idea of the situation.

3.2.6.1 *Emissions to the atmosphere*

(a) Coal coking

During coal coking, PAH are released into the ambient air mainly when an oven is loaded through the charging holes and new coal is suddenly brought into contact with the hot oven, and from leaks around oven doors and battery-top lids (Bjørseth & Ramdahl, 1985; Slooff et al., 1989). The specific emission factor for both benzo[*a*]pyrene and benzo[*e*]pyrene during coal coking was 0.2 mg/kg coal charged (Ahland et al., 1985). The emission factor for total PAH was estimated to about 15 mg/kg coal charged (Bjørseth & Ramdahl, 1985).

Stack gases were measured about 8 m away from the aperture through which coke was discharged at a Belgian coking battery. Although the effluent may have been slightly diluted with ambient air, the following PAH concentrations were detected: benz[*a*]anthracene plus chrysene, 580 ng/m^3; benzo[*k*]fluoranthene, 500 ng/m^3; benzo[*a*]pyrene plus benzo[*e*]pyrene, 470 ng/m^3; fluoranthene, 330 ng/m^3; pyrene, 180 ng/m^3 benzo[*ghi*]perylene, 140 ng/m^3; anthracene plus phenanthrene, 130 ng/m^3; and perylene, 44 ng/m^3 (Broddin et al., 1977).

The release of total PAH in 1985 was estimated to about 630 t/year in the USA, 18 t/year in Sweden, and 5.1 t/year in Norway (Bjørseth & Ramdahl, 1985). The authors emphasized that their data are subject to uncertainty and should be used only as an indication of the order of magnitude. In 1990, the total PAH emission in Canada was estimated to be 13 t/year (Environment Canada, 1994). Further estimates of total annual emissions of individual PAH compounds during the coking of coal are shown in Table 10.

The emission factors for benzo[*a*]pyrene in the coking industry in the North-Rhine Westphalia area of Germany have been assumed to have been reduced to an average of about 60 mg/t coke. The newest plants have emission factors of 40 mg/t coke (Eisenhut et al., 1990). The reduction in PAH discharge

Table 10. Estimated annual emissions of polycyclic aromatic hydrocarbons during coal coking in the Netherlands and western Germany

Compound	Annual emission (t/year)	Year	Reference
Netherlands			
Anthanthrene	0.5	Before 1989	Slooff et al. (1989)
Benz[*a*]anthracene	0.3	1988	Slooff et al. (1989)
Benzo[*a*]pyrene	0.1	Before 1989	Slooff et al. (1989)
Benzo[*ghi*]perylene	0.2	1988	Slooff et al. (1989)
Benzo[*k*]fluoranthene	0.1	1988	Slooff et al. (1989)
Chrysene	0.2	1988	Slooff et al. (1989)
Fluoranthene	1.1	1988	Slooff et al. (1989)
Indeno[1,2,3-*cd*]pyrene	0.1	1988	Slooff et al. (1989)
Naphthalene	1.3	1987	Slooff et al. (1988)
	2.0	Before 1989	Slooff et al. (1989)
Phenanthrene	2.1	1988	Slooff et al. (1989)
Western Germany			
Benzo[*a*]pyrene	1.1	1990	Ministers for the Environment (1992);
	1.7		Zimmermeyer et al. (1991)
Naphthalene	10.0	1987	Society of German Chemists (1989)

was brought about by technical improvements to existing plants, closure of old plants and their partial replacement by new plants, and a reduction in coke production (Zimmermeyer et al., 1991). Decreasing trends in the annual emissions of airborne PAH during coke production are also assumed to have occurred in other industrialized countries (western Europe, Japan, and the USA), but no data were available.

(b) Coal conversion

PAH emission factors measured in the USA during gasification of coal at the end of the 1970s ranged from about 1 µg/g burnt coal for chrysene and 1500 µg/g burnt coal for naphthalene. Three qualities of coal were analysed for naphthalene, acenaphthylene, fluorene, anthracene, phenanthrene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene, and dibenzo[*a,h*]pyrene (Nichols et al., 1981). In 1981, the stack gas of one US pilot

coal gasification plant with an outdoor filter contained 0.2 and 2.1 μg/m³ naphthalene at two sampling times and 6.8 μg/m³ phenanthrene (Osborn et al., 1984). Acenaphthylene was detected at concentrations of 0.11-0.12 μg/m³ in the stack gases of two Canadian pilot coal liquefaction plants (Leach et al., 1987).

(c) Petroleum refining

The average profile of PAH compounds in petroleum refineries indicates that at least 85% of the total concentration is made up of two-ring compounds (naphthalene and its derivatives) and 94% of two- and three-ring compounds. Compounds with five rings or more contributed less than 0.1% at the catalytic cracking unit. In turn-round operations on reaction and fractionation towers, naphthalene and its methyl derivatives accounted for more than 99% of the total PAH (IARC, 1989b).

Little information is available on the concentrations of PAH in stack gases. The levels in one French (Masclet et al., 1984) and two US petroleum refining plants (Karlesky et al., 1987) are available (Table 11); no information was given about the sampling site in the French facility, but sampling in the US plants was at the distillation device and below the cracking tower. The results depended on which fuel was burnt and the positioning and type of sampling device in the stack.

Few data are available on the total release of PAH into the atmosphere during petroleum refining. In western Germany, the emissions of naphthalene during petroleum refining, including hard coal-tar processing, were estimated to be 11 t/year (year not given; Society of German Chemists, 1989). In the Netherlands, the release of total PAH in 1988 was estimated to be about 7 t/year; the burning of pitch contributed 6.6 t/year, regeneration of catalyst, 0.4 t/year, and refining, < 0.01–0.1 t/year (Slooff et al., 1989). In Canada, about 0.1 t total PAH were emitted into the atmosphere in 1990 (Environment Canada, 1994).

(d) Other processes

In a US oil-furnace carbon black plant, the following mean emission factors per kg carbon black produced were found for individual PAH in three runs in the main vent gas: acenaphthylene, 800 μg; pyrene, 500 μg; anthracene plus phenanthrene, 70 μg; fluoranthene, 60 μg; benzo[*ghi*]fluoranthene, 40 μg; benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, 30 μg; benzo[*a*]pyrene plus benzo[*e*]pyrene plus perylene, 30 μg; benzo[*ghi*]perylene plus anthanthrene, 23 μg; chrysene plus benz[*a*]anthracene, 9 μg; indeno[1,2,3-*cd*]pyrene, < 2 μg; and benzo[*c*]phenanthrene, < 2 μg. The release of PAH into ambient air cannot be estimated from these emission factors, however, as an additional combustion device is fitted in most US

Table 11. Polycyclic aromatic hydrocarbon concentrations in the stack gases of petroleum refinery plants in France and the USA

Compound	Concentration (μg/m³)	
	France	USA
Acenaphthene	NR	0.018–0.035
Acenaphthylene	NR	0.013/0.019
Anthracene	3.9	0.003–0.041
Benz[*a*]anthracene	1.6	0.051–0.801
Benzo[*a*]pyrene	0.4	0.261–3.17
Benzo[*b*]fluoranthene	1.3	0.323–0.616[a]
Benzo[*e*]pyrene	2.8	NR
Benzo[*ghi*]perylene	0.7	0.23/0.382
Benzo[*k*]fluoranthene	0.5	NR
Chrysene	1.7	0.021–0.252
Coronene	1.0	NR
Dibenzo[*a,h*]anthracene	NR	0.177
Fluoranthene	2.3	0.030–0.577
Fluorene	2.4	0.041–2.48
Indeno[1,2,3-*cd*]pyrene	1.2	0.25/0.538
Naphthalene	NR	0.052–0.113
Perylene	ND	ND
Phenanthrene	7.9	0.040–9.13
Pyrene	4.3	0.016–3.56

From Masclet et al. (1984) and Karlesky et al. (1987)
NR, not reported; ND, not detected, limit of detection not stated; /, single measurements
[a] Plus benzo[*k*]fluoranthene

carbon-black plants in which the process vent gases are burnt (Serth & Hughes, 1980).

Compounds with five or more rings (e.g. benzo[*a*]pyrene) contributed about 0.3% to the total PAH released from the bitumen processing unit of a refinery (IARC, 1989b). The emissions of PAH from batch asphalt mixers are assumed to be low and to occur mainly in combustion gases (IARC, 1984a), although no experimental data were available.

Few estimates have been made of the annual emissions of PAH from processes in which coal and coal products are used. The total release of PAH to the atmosphere during asphalt production in 1985 was estimated to be about 4 t in the USA, 0.1 t in Norway, and 0.3 t in Sweden (Bjørseth & Ramdahl, 1985). In Canada, the amount emitted in 1990 was estimated to be about 2.5 t

(Environment Canada, 1994). The amount released during carbon-black production and processing in 1985 was estimated to be about 3 t in the USA and < 0.1 t in Sweden (Bjørseth & Ramdahl, 1985). In the Netherlands in 1988, about 3.3 t of total PAH were emitted during the storage and transport of anthracene oil, an intermediate in the processing of hard coal-tar (Slooff et al., 1989).

(e) Use of impregnating oils (creosotes) in wood preservation

Estimates of the total input of PAH into the atmosphere from wood preservation with creosotes were available only for the Netherlands for unspecified years, at about 320 t/year (Slooff et al., 1989) and 840 t/year (Berbee, 1992). In 1988, the PAH input during storage of preserved material was estimated by the same authors to be about 200 t naphthalene, 110 t phenanthrene, 30 t fluoranthene, 5 t anthracene, 1.1 t benz[*a*]anthracene, and 0.02 t benzo[*k*]fluoranthene.

3.2.6.2 *Emissions to the hydrosphere*

(a) Coal coking

The concentrations of PAH reported in wastewater effluents are shown in Table 12. The removal of PAH by biological oxidation in two US coal coking plants was 93 to > 99%. Higher-molecular-mass PAH, benzo[*a*]pyrene, dibenz[*a,h*]anthracene, and benzo[*ghi*]perylene, comprised a greater fraction (about 60%) of the total PAH content in the effluent than in the input stream (Walters & Luthy, 1984). The total concentration of PAH discharged into the aqueous environment from a Norwegian coking plant was estimated to be about 23 kg/d (Berglind, 1982). On the basis of Dutch emission factors, the release in western Europe in 1985 of fluoranthene was calculated to be about 5 t and that of benzo[*a*]pyrene about 0.7 t (Berbee, 1992). The total annual input of PAH into the aqueous environment of the Netherlands was estimated to be about 1.7 t (year not given; Slooff et al., 1989).

(b) Coal conversion

The PAH content of wastewater from coal and shale conversion was < 0.5 mg/litre (Guerin, 1977). In raw, untreated wastewaters from a US pilot coal liquefaction plant, numerous PAH were found to emanate from the liquefaction section, the untreated hydrogenation section, and the still bottoms processing device when two kinds of coal were tested; for example, benzo[*a*]pyrene was found at a concentration of 0.3–52 µg/litre (Robbins et al., 1981). Numerous PAH were found in raw wastewater samples from two US pilot coal gasification plants (Walters & Luthy, 1981; Abbott et al., 1986), the maximum level of benzo[*a*]pyrene being 5.0 µg/litre.

Table 12. Polycyclic aromatic hydrocarbon concentrations (µg/litre) in wastewater effluents from coal coking plants

Compound	[1]	[2]	[3][a]	[4]	[5]
Acenapthene	NR	NR	NR	0.009–2.5	NR
Acenaphthylene	NR	NR	NR	NR	NR
Anthracene	0.31	NR	NR	0.0–2.0	0.1
Anthanthrene	ND	NR	0.040/0.600	NR	NR
Benzo[*j+k*]fluoranthene	NR	NR	NR	NR	NR
Benz[*a*]anthracene	2.0	11.1	0.504/4.9	NR	NR
Benzo[*a*]fluoranthene	0.8	NR	NR	NR	NR
Benzo[*a*]pyrene	NR	3.8	0.622/4.841	4.7–25	NR
Benzo[*b*]fluoranthene	NR	NR	NR	NR	NR
Benzo[*a*]fluorene	0.81	NR	NR	NR	NR
Benzo[*c*]phenanthrene	ND	NR	0.042/0.699	NR	NR
Benzo[*e*]pyrene	NR	NR	0.323/2.928	NR	NR
Benzofluoranthenes[b]	NR	6.9	1.010/8.741	NR	NR
Benzo[*ghi*]fluoranthene	ND	NR	0.042/0.663	NR	NR
Benzo[*ghi*]perylene	2.0	NR	0.445/2.835	0–9.0	NR
Chrysene	NR	7.2	0.732/6.440	1.8–42	NR
Dibenz[*a,h*]anthracene	NR	NR	NR	0.06–3.0	NR
Fluoranthene	2.8	11.2	NR	1.3–10	NR
Fluorene	NR	NR	NR	0.0–1.0	NR
Indeno[1,2,3-*cd*]pyrene	NR	NR	0.371/3.051	NR	NR
1-Methylphenanthrene	ND	NR	NR	NR	NR
Naphthalene	NR	NR	NR	0–4.1	NR
Perylene	ND	NR	0.117/1.348	NR	NR
Phenanthrene	0.4	NR	NR	0.45–2.3	0.5
Pyrene	4.0	12.9	NR	NR	0.38–60

[1] Effluent channel water from one US coking plant (Griest, 1980); **[2]** Effluent channel water from one US coking plant (Griest et al., 1981); **[3]** Raw wastewater from two coking plants in western Germany (Grimmer et al., 1981b); **[4]** Effluents from two US coking plants downstream of company-owned biological oxidation device (Walters & Luthy, 1984); **[5]** Final effluent after biological oxidation; no further information (Jockers et al., 1988)

When the water samples were filtered through solid sorbents, the results may be underestimates of the actual content of polycyclic aromatic hydrocarbons (see section 2.4.1.4)

ND, not detected, limit of detection not given; NR not reported

[a] /, single measurements

[b] Isomers not specified

No information was available about total PAH emissions into the aqueous environment from commercial coal conversion plants. In groundwater near a US in-situ coal gasification site, naphthalene was found at a concentration of 2.7 µg/litre and acenaphthene and fluorene at < 0.1 µg/litre (Pellizzari et al., 1979).

Until 1988, the final effluent from the two hard coal-tar refineries in western Germany contained an average of 50 mg/litre naphthalene, with a maximum of 120 mg/litre. The annual emission of this compound was thus calculated to be about 80 t. By 1991, the estimated release of naphthalene had been reduced to about 8 t/year by the addition of adsorption devices (Klassert, 1993).

(c) Petroleum refining and offshore oil-well drilling

PAH concentrations in wastewater effluents from these sources are summarized in Table 13. A refinery-activated sludge unit with a dual-media filter removed about 95% of the five-ring PAH and 99% of the four-ring PAH from the effluent of a petroleum refinery (Pancirov et al., 1980). A similar elimination efficiency was found for dissolved air flotation treatment of refinery wastewater and subsequent removal by activated sludge. Air stripping of the compounds in the sewage plant seemed to be of minor importance (Snider & Manning, 1982). The concentrations of PAH with more than three rings were found to be < 0.05 µg/litre even in the input to a sewage device and < 0.02 µg/litre in the final effluent (German Society for Mineral-oil and Coal Chemistry, 1984). The authors stated that these levels were of the same order of magnitude as the background concentrations in surface waters.

The discharge of total PAH from a Norwegian petroleum refinery was about 0.26 kg/day (Berglind, 1982). The total concentration of PAH released into the North Sea from offshore oil-well drilling activities was about 2.5 t/year in 1987, comprising 2 t/year from drill rinsing and 0.2 t/year from shipping (Slooff et al., 1989).

(d) Use of impregnating oils (creosotes) in wood preservation

PAH were detected at levels of milligrams per litre in groundwater under a former wood preserving facility in Florida, USA. The concentrations of lower-molecular-mass creosote constituents were smaller in the groundwater than in an unweathered standard, probably because of greater mobility and biodegradability (Mueller & Lantz, 1993; Middaugh et al., 1994).

Model experiments with fresh and seawater were carried out to determine the release of PAH from marine pilings made from southern pine and preserved with creosote (Ingram et al., 1982). The PAH levels per litre fresh water in the leachate at 20 °C after immersion for three days were: naphthalene, 200–350 µg; acenaphthene, 190–230 µg; phenanthrene, 190–230 µg; fluorene,

Table 13. Polycyclic aromatic hydrocarbons in effluents after wastewater treatment in petroleum refineries (µg/litre)

Compound	[1]	[2]	[3]	[4]	[5]
Acenaphthene	NR	4.0	< 0.1–6	NR	NR
Acenaphthylene	NR	1.8	< 0.1–< 1	NR	NR
Anthracene	NR	1.2	< 0.01–< 2	0.26	NR
Benz[*a*]anthracene	NR	0.6	< 0.02–< 1	NR	NR
Benzo[*a*]pyrene	0.57	0.1	0.1–2.9	0.11	NR
Benzo[*b*]fluoranthene	< 0.1	0.2	< 0.06	NR	NR
Benzo[*c*]phenanthrene	NR	0.2	NR	NR	NR
Benzo[*e*]pyrene	0.65	0.3	NR	NR	NR
Benzo[*ghi*]fluoranthene	< 0.4	NR	NR	NR	NR
Benzo[*ghi*]perylene	0.36	NR	< 0.2–< 1	NR	NR
Benzo[*j*]fluoranthene	< 0.2	NR	NR	NR	NR
Benzo[*k*]fluoranthene	< 0.2	0.4[a]	< 0.2	NR	NR
Chrysene	< 0.03	1.4[b]	< 0.02–1.4	NR	NR
Coronene	< 0.01	NR	NR	NR	NR
Dibenz[*a,h*]anthracene	NR	NR	< 0.3–< 1	NR	NR
Fluoranthene	< 0.2	16.0	< 0.1–< 10	0.26	NR
Fluorene	NR	3.4	< 0.1–< 1	1.2	NR
Indeno[1,2,3-*cd*]pyrene	< 0.02	NR	< 1	NR	NR
1-Methylphenanthrene	NR	4.2	NR	NR	NR
Naphthalene	NR	2.4	< 0.1–< 10	15	0.06–9
Perylene	0.14	NR	NR	NR	NR
Phenanthrene	NR	111.0	< 0.2–< 0.5	7.1	0.02–1.2
Pyrene	0.07	16.1	< 0.1–7	NR	NR
Triphenylene	< 0.03	NR	NR	NR	NR

[1] Final effluent from one US petroleum refinery (Pancirov et al., 1980); **[2]** Effluent from one Norwegian petroleum refinery after treatment in oil-separation devices, oil traps, and retention ponds (Berglind, 1982); **[3]** Average results for final effluent from 17 US petroleum refineries (Snider & Manning, 1982); **[4]** Final effluent from one Australian petroleum refinery (Symons & Crick, 1983); **[5]** Average results for the final effluent from six petroleum refineries in western Germany (German Society for Mineral-oil and Coal Chemistry, 1984)
When water samples were filtered through solid sorbents, the results may be underestimates of the actual PAH content (see section 2.4.1.4).
NR, not reported
[a] With benzo[*j*]fluoranthene
[b] With triphenylene

120–150 µg; acenaphthylene, 51–88 µg; anthracene, 48–76 µg; fluoranthene, 27–30 µg; pyrene, 12 µg; and benz[*a*]anthracene, 11–19 µg. The concentrations in seawater were three to four times lower. The amounts of PAH leached

increased with increasing temperature. The concentrations in leachates from pilings that had been in seawater for 12 years were of the same order of magnitude. In contrast, rapidly decreasing PAH concentrations were found three months after the start of the experiment in runoff rainwater from spruce and pine pilings impregnated with hard coal-tar (van Dongen, 1987).

The total PAH emissions into water and soil in the Netherlands from commercial wood preservation were about 28 t/year (year not given). The release of 10 PAH into water during the storage of creosote-preserved wood was about 16 t/year; the PAH measured were naphthalene, anthracene, phenanthrene, fluoranthene, benz[*a*]anthracene, benzo[*a*]pyrene, benzo[*ghi*]-perylene, and indeno[1,2,3-*cd*]pyrene) (Slooff et al., 1989).

In Canada, the maximum release of PAH into water and soil from creosote-treated wood products was estimated to be 2000 t/year, on the basis of the PAH content of creosote, the volume of treated wood, the retention rates of the substances for different wood species, and an estimated 20% loss of PAH during the time the wood was in service, i.e. 40 years for pilings and 50 years for railroad ties (Environment Canada, 1994).

(e) Other sources

PAH may be released into the hydrosphere during leaching of stocks of coal by rain. In model leaching experiments, naphthalene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, chrysene, benz[*a*]anthracene, benzo[*k*]fluoranthene, and benzo[*a*]pyrene were detected at concentrations in the low microgram per litre range, with a maximum of 100 µg/litre; for example, benzo[*a*]pyrene was found at 0.6 µg/litre (Stahl et al., 1984; Fendinger et al., 1989).

PAH were also found in sludge from US coke processing plants in the following concentrations (average of five samples): naphthalene, 430 mg/kg; phenanthrene, 260 mg/kg; acenaphthene, 78 mg/kg; pyrene, 30 mg/kg; chrysene, 28 mg/kg; benzo[*a*]pyrene, 3.8 mg/kg; benzo[*b*]fluoranthene, 3.8 mg/kg; and benzo[*ghi*]perylene, 0.9 mg/kg (Tucci, 1988).

PAH may also leach into drinking-water from coal-tar or asphalt coatings on storage tanks and water distribution pipes. Samples from a five-year-old coal-tar-coated water tank in the USA contained 0.21 µg/litre phenanthrene plus anthracene, 0.081 µg/litre fluoranthene, 0.071 µg/litre pyrene, 0.025 µg/litre naphthalene, and 0.021 µg/litre fluorene (Alben, 1980). Measurements in numerous US drinking-water systems showed that PAH accumulate in the water during transport in these pipes. The total concentration of fluoranthene, benzo[*j*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]-pyrene, and benzo[*ghi*]perylene after transport was in the low nanogram per litre range (Basu et al., 1987). In 1994, a PAH concentration of 2.7 µg/litre was measured in accordance with the German Directive on drinking-water (6.9 µg/litre measured in accordance with US regulations),

which was due to transport through a tar-coated pipe in a central water reservoir; phenanthrene was present at a concentration of 2.8 µg/litre and pyrene at 1.2 µg/litre (State Chemical Analysis Institute, Freiburg, 1995). The release of PAH from this source cannot be estimated from the available data.

During offshore oil and gas production, PAH-containing drilling muds are discharged directly into the sea. The PAH concentrations at some oil and gas platforms in the Gulf of Mexico and the North Sea were found to be 1900 µg/litre for naphthalene and < 0.01 µg/litre each for chrysene, benzo[*b*]fluoranthene, and dibenz[*a*,*h*]anthracene (van Hattum et al., 1993).

The total PAH passing into the oceans from shipping have not been estimated. The worldwide discharge of PAH into the oceans from refineries, marine transportation, and industrial effluents of crude oil was estimated to be about 6 t/year in 1973 and 4.6 t/year in the early 1980s (Suess, 1976), but the basis for these estimates is unknown.

3.2.6.3 *Emissions to the geosphere*

The average PAH concentrations in soil from more than 20 former coking sites in Germany were: naphthalene, 1000 mg/kg; phenanthrene, 500 mg/kg; fluoranthene, 200 mg/kg; pyrene, 200 mg/kg; anthracene, 50 mg/kg; and benzo[*a*]pyrene, 3–5 mg/kg. During vertical leaching, the compounds are distributed according to their mobility. PAH with high-boiling points and low water solubility are present at the highest concentrations at the surface, and more mobile compounds accumulate in deeper soil layers. Naphthalene is usually leached into groundwater, in which it is relatively soluble (Hoffmann, 1993).

The sediment of an effluent channel at one US coking plant contained the following concentrations of PAH (dry weight basis): fluoranthene, 31 mg/kg; pyrene, 23 mg/kg; benzo[*b*+*j*+*k*]fluoranthenes, 23 mg/kg; benzopyrenes, 19 mg/kg; benz[*a*]anthracene, 15 mg/kg; chrysene plus triphenylene, 15 mg/kg; benzo[*ghi*]perylene, 7.3 mg/kg; benzo[*a*]fluorene, 7.2 mg/kg; anthracene, 6.7 mg/kg; perylene, 3.8 mg/kg; phenanthrene, 3.6 mg/kg; benzo[*b*]fluorene, 3.2 mg/kg; benzo[*ghi*]fluoranthene, 2.3 mg/kg; anthanthrene, 2.3 mg/kg; benzo[*c*]phenanthrene, 2.1 mg/kg; and 1-methylphenanthrene, 0.71 mg/kg. In the sediment of an effluent from one US petroleum tank farm, anthracene was detected at 3.4 mg/kg, benz[*a*]anthracene at 0.13 mg/kg, and benzo[*a*]pyrene at < 0.049 mg/kg (Griest, 1980).

Oily sludge originating from a dissolved air flotation unit of the treatment system of a US petrochemical plant effluent was applied to sandy loam samples seven times during a 920-day active disposal period followed by a 360-day inactive 'closure' period, and the decreases in the concentrations of fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, triphenylene, benzo[*ghi*]fluoranthene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, benzo[*k*]fluoranthene, perylene, benzo[*a*]pyrene, benzo[*e*]pyrene, and

benzo[*ghi*]perylene in soil were determined. The initial PAH levels ranged from 0.9 mg/kg benzo[*j*]fluoranthene to 270 mg/kg phenanthrene (dry weight basis). After 1280 days, the three-ring compounds (fluorene, phenanthrene, anthracene) had almost completely disappeared, with 0.2–6.9% remaining, the four-ring substances (fluoranthene, benz[*a*]anthracene, chrysene) had been partly degraded, and the five-ring compounds remained at fairly high concentrations (Bossert et al., 1984).

PAH may be released into soil from polluted industrial sludges and during commercial wood preservation; however, no estimates of the total PAH input into this compartment were available.

3.2.6.4 Emissions into the biosphere

Use of anti-dandruff shampoos containing hard coal-tar may lead to increased body concentrations of PAH, as measured by urinary excretion of the PAH metabolite 1-hydroxypyrene. One shampoo had a total PAH content of 2800 mg/kg, including 290 mg/kg pyrene and 56 mg/kg benzo[*a*]pyrene (no further specification) (van Schooten et al., 1994). Application of a 2% crude coal-tar solution in petrolatum led to significantly increased PAH levels in the blood of five volunteers (Storer et al., 1984; see also Section 8). Measurements of hard coal-tar-containing shampoos in Germany showed concentrations of 7–61 mg/kg benzo[*a*]pyrene. In wood-tar-containing shampoos, benzo[*a*]pyrene was detected at concentrations in the low microgram per kilogram range, but 150 mg benzo[*a*]pyrene were found in one tar bath (State Chemical Analysis Institute, Freiburg, 1995).

3.2.7 Emissions of PAH due to incomplete combustion

PAH not only pre-exist in fossil fuels but more are formed during pyrolysis by a radical mechanism (see Zander, 1980). The domestic activities that may result in significant emissions of PAH emissions are vehicle traffic, tobacco smoking, broiling and smoking of foods, and refuse burning. The industrial activities that result in PAH release are aluminium production with use of Söderberg electrodes, iron and steel production, foundries, tyre production, power plants, incinerators, and stubble burning (Anderson et al., 1986)

3.2.7.1 Industrial point sources

(a) Emissions to the atmosphere

(i) *Power plants fired with coal, oil, and gas fossil fuels*

PAH emitted into the atmosphere from coal-fired power plants consist mainly (69–92%) of two- and three-ring compounds, i.e. naphthalene and phenanthrene and their mono- and dimethyl derivatives. Naphthalene is by far

the major component of PAH fractions (31–35%), although high concentrations of phenanthrene and fluorene are also observed (Bonfanti et al., 1988). Specific emission factors of 0.02 µg emitted per kg combusted were measured for benzo[*a*]pyrene and 0.03 µg/kg for benzo[*e*]pyrene (Ahland et al., 1985).

The concentrations of PAH in stack gases from comparable coal- and oil-fired power plants are shown in Table 14. It is difficult to find a characteristic PAH profile for coal-fired plants. The concentrations were low during undisturbed combustion (Guggenberger et al., 1981; Warman, 1985). Low-molecular-mass PAH are found at higher concentrations than high-molecular-mass compounds in coal combustion effluents (Warman, 1985); the low-molecular-mass PAH phenanthrene, fluoranthene, and pyrene were detected at particularly high concentrations, whereas benzo[*a*]pyrene was found at a level typical of that in ambient air (Kanij, 1987). The specific emission factor for benzo[*a*]pyrene was 3.5–230 µg/t burnt coal (Ahland & Mertens, 1980). As the contribution of benzo[*a*]pyrene to the total release of PAH is small, it was considered not to be a suitable indicator for this source (Guggenberger et al., 1981). In contaminated areas, the PAH concentrations in ambient air may be higher than those in the stack gases, which result from after-burning (Guggenberger et al., 1981).

The inputs of PAH into the atmosphere from power plants were: about 0.001 t benzo[*a*]pyrene in western Germany in 1981 (Ahland et al., 1985) and 0.1 t in 1983 (Grimmer, 1983a); about 1 t/year total PAH in the USA; 0.1 t in Norway and 6.6 t in Sweden in 1985 (Bjørseth & Ramdahl, 1985); about 2 t total PAH in the Netherlands in 1988 (Slooff et al., 1989); and about 11 t total PAH in Canada in 1990 (Environment Canada, 1994). These numbers may be subject to uncertainty and should be used only as an indication of the order of magnitude of e.g. the concentration in stack gases that is to be expected from experimental values. Actual information on PAH emissions from oil- and gas-fired power plants was not available. PAH emissions from coal-fired power plants have been claimed to be negligible in Germany due to the installation of appropriate filter systems, despite the vast amount of stack gases produced (Zimmermeyer et al., 1991; Ministers for the Environment, 1992).

(ii) *Incinerators*

Numerous PAH are formed under simulated incinerator conditions from plastics such as polystyrene, polyethylene, polyvinyl chloride, and their mixtures (Hawley-Fedder et al., 1984a,b,c, 1987). PAH were detected at the following concentrations in the stack gases from a British municipal incinerator: pyrene, 1.6 µg/m^3; benz[*a*]anthracene plus chrysene, 0.72 µg/m^3; fluorene, 0.58 µg/m^3; benzo[*ghi*]perylene, 0.42 µg/m^3; benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, 0.32 µg/m^3; perylene, 0.18 µg/m^3; indeno[1,2,3-*cd*]pyrene, 0.18 µg/m^3; coronene, 0.04 µg/m^3; and benzo[*a*]pyrene plus benzo[*e*]pyrene, 0.02 µg/m^3 (Davies et al., 1976). When

Table 14. Concentrations of polycyclic aromatic hydrocarbons (ng/m³) in stack gases of coal- and oil-fired power plants

Compound	Fuel	[1]	[2]	[3]	[4]	[5]	[6][a]
Acenaphthene	Coal	NR	NR	NR	NR	NR	ND–24
Anthracene	Coal	NR	0.5	< 10–1800	0.4–100	2–65	19–120
Anthanthrene	Coal	NR	NR	NR	NR	< 0.2–< 0.6	NR
Benz[*a*]anthracene	Coal	NR	0.6	< 20–1400	NR	1–40	NR
Benzo[*a*]pyrene	Coal	< 0.1–0.7[b] < 0.5[c]	1.3	0.5–790	0.1–120	0.1–1.9	NR
	Oil	< 0.5–7	NR	NR	NR	NR	NR
Benzo[*b*]fluoranthene	Coal	< 0.1–3[b,d] < 0.1–0.4[c,d]	2.0	30/40[k] (1/880[e])	NR	0.3–12	NR
	Oil	< 0.1–39[d]	NR	NR	NR	NR	NR
Benzo[*b*]fluorene	Coal	NR	NR	NR	NR	< 2–< 6	NR
Benzo[*c*]phenanthrene	Coal	NR	NR	0.2	NR	NR	NR
Benzo[*e*]pyrene	Coal	NR	ND	< 10–810	NR	3–< 18	NR
Benzo[*ghi*]perylene	Coal	NR	NR	< 10–1400	NR	NR	NR
	Coal	< 0.5–3[b] < 0.5[c]	1.2	< 10–< 100	3–22	< 2–< 6	NR
	Oil	< 0.5–40	NR	NR	NR	NR	NR
Benzo[*j*]fluoranthene	Coal	NR	NR	NR	NR	< 5–< 13	NR
Benzo[*k*]fluoranthene	Coal	< 0.1–2[b] < 0.1–1.3[c]	0.9	20	NR	1.7–2.5	NR
	Oil	< 0.1–29	NR	NR	NR	NR	NR

Table 14 (contd)

Compound	Fuel	[1]	[2]	[3]	[4]	[5]	[6][a]
Chrysene	Coal	NR	1.8 < 10–310[e] 3.8[g]	< 10–< 600	0.1–28	1–41	ND–56
Coronene	Coal	1–3[b] < 2[c]	0.9	< 100	NR	NR	NR
	Oil	< 2–36	NR	NR	NR	NR	NR
Dibenz[*a,h*]anthracene	Coal	< 0.5–2[b] < 0.5[c]	NR	< 100	NR	NR	NR
	Oil	< 0.5–26	NR	NR	NR	NR	NR
Fluoranthene	Coal	NR	4.1	< 10–22 100	0.5–240	20–720	NR
Fluorene	Coal	NR	1.9	NR	NR	NR	2–140
Indeno[1,2,3-*cd*]pyrene	Coal	NR	1.7	< 10–< 100	NR	< 0.1–< 1.4	NR
1-Methylphenanthrene	Coal	NR	NR	< 20–90	NR	NR	NR
Naphthalene	Coal	NR	NR	NR	10–1800	NR	420–2100
Perylene	Coal	< 0.1–0.2[b] < 0.1[c]	NR	NR	NR	NR	NR
	Oil	< 0.1–15	ND	< 10–< 100	NR	< 0.2–0.9	NR

Table 14 (contd)

Compound	Fuel	[1]	[2]	[3]	[4]	[5]	[6][a]
Phenanthrene	Coal	NR	5.2	< 20–33 200	26–640	32–2930	NR
Pyrene	Coal	NR	1.3	9–5800	0.2–2850	5–335	ND–311
Triphenylene	Coal	NR	NR	NR	NR	20–77	NR

[1] Coal- and oil-fired power plants in the former FRG (Guggenberger et al., 1981); **[2]** One French coal-fired power plant (Masciet et al., 1984); **[3]** 10 Swedish coal-fired power plants (Warman, 1985); **[4]** One US coal-fired power plant (Junk et al., 1986); **[5]** One Dutch coal-fired power plant (Kanij, 1987); **[6]** One German coal-fired power plant with circulating fluid bed combustion (Wienecke et al.,1992)

NR, not reported; ND, not detected, limit of detection not given

[a] Various coal qualities

[b] Hard coal

[c] Brown coal

[d] With benzo[*e*]pyrene

[e] Isomers not specified

[f] With triphenylene

[g] With benz[*a*]anthracene

PAH were sampled at a height of about 10 m above the ground in the 110-m chimney of an incineration plant in Sweden, no measurable amounts of PAH, at a limit of detection of 10 ng/m^3, were found during normal operating conditions or during start-up in the morning; however, inactivity over a weekend resulted in detectable concentrations of individual PAH, covering three orders of magnitude up to around 100 µg/m^3 (Colmsjö et al., 1986a). Comparable results were obtained at a pilot incineration plant in Canada (Chiu et al., 1991). Only phenanthrene plus anthracene was found in measurable amounts in the stack gas (limit of detection not stated). The total release of PAH from this plant was estimated to be 80-100 ng/m^3.

The concentrations of PAH emitted in the stack gases from an Italian municipal solid waste incinerator were: 0.1–1.9 µg/m^3 indeno[1,2,3-*cd*]pyrene, 0.63 µg/m^3 acenaphthene, 0.57–2.5 µg/m^3 phenanthrene, 0.36–4.4 µg/m^3 perylene, 0.35–0.55 µg/m^3 benzo[*e*]pyrene, 0.25–3.6 µg/m^3 benz[*a*]anthracene, 0.23 µg/m^3 benzo[*k*]fluoranthene, 0.22 µg/m^3 dibenz[*a*,*h*]anthracene, 0.19 µg/m^3 benzo[*b*]fluoranthene, 0.15–0.67 µg/m^3 pyrene, 0.15–0.73 µg/m^3 acenaphthylene, 0.11–0.23 µg/m^3 chrysene, 0.08 µg/m^3 anthracene, 0.069 µg/m^3 fluorene, 0.068–1.3 µg/m^3 fluoranthene, 0.05–1.1 µg/m^3 benzo[*a*]pyrene, and 0.014–0.47 µg/m^3 benzo[*ghi*]perylene, depending on the firing conditions and the composition of the waste (Morselli & Zappoli, 1988).

The benzo[*a*]pyrene concentrations in stack gases from commercial waste incinerators in western Germany were estimated to be 1–6 µg/m^3 (Johnke, 1992).

Controlled incineration of automobile tyres for thermal and electric energy has been estimated to result in considerable release of PAH into the atmosphere. In laboratory experiments, the following concentrations were found in flue gas at an incineration temperature of 677 °C (per kg rubber): 930 mg pyrene, 760 mg fluoranthene, 390 mg phenanthrene, 290 mg anthracene, 220 mg acenaphthylene, 120 mg chrysene, 84 mg benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, 66 mg benz[*a*]anthracene, 18 mg benzo[*e*]pyrene, 11 mg benzo[*a*]pyrene, 3.8 mg perylene, 3.3 mg benzo[*ghi*]fluoranthene, 2.0 mg dibenz[*a*,*h*]anthracene, 1.5 mg benzo[*ghi*]perylene, 1.2 mg naphthalene, and 0.5 mg indeno[1,2,3-*cd*]pyrene (Jacobs & Billings, 1985). On the basis of data from Hartung & Koch (1991) on the number of tyres incinerated in western Germany in 1987, the annual emissions from this source can be calculated as follows: 160 t pyrene, 130 t fluoranthene, 70 t phenanthrene, 50 t anthracene, 40 t acenaphthylene, 20 t chrysene, 14 t benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]-fluoranthene, 10 t benz[*a*]anthracene, 3 t benzo[*e*]pyrene, 2 t benzo[*a*]pyrene, 0.5 t benzo[*ghi*]fluoranthene, 0.3 t dibenz[*a*,*h*]anthracene, 0.3 t benzo[*ghi*]-perylene, 0.2 t naphthalene, and 0.1 t indeno[1,2,3-*cd*]pyrene.

The total PAH levels in stack gases from incinerators in different countries were: Italy, 0.0075–0.21 mg/m^3; Japan, 0.002–0.04 mg/m^3; Sweden,

0.001 mg/m^3; and Canada, 0.00002–0.02 mg/m^3 (WHO, 1988). The results for traditional incinerators could not be compared with those for plants with additional abatement techniques on the basis of the available data. The total PAH emissions to the atmosphere resulting from incineration of refuse were about 0.001 t benzo[*a*]pyrene in western Germany in 1989 (Ministers for the Environment, 1992) and about 0.0003 t in 1991 (Johnke, 1992), about 50 t total PAH in the USA, 0.3 t in Norway and 2.2 t in Sweden in 1985 (Bjørseth & Ramdahl, 1985); and about 2.4 t total PAH in Canada in 1990 (Environment Canada, 1994).

In Germany, the contribution of stack gases from commercial incinerators is estimated to be < 4% of the total stack gas volume from combustion processes. One of the main confounders of and contributors to stack gases from combustion is motor vehicle traffic (Johnke, 1992), indicating that PAH released from incinerators are probably of minor importance.

(iii) *Aluminium production*

The production of coal anodes, used in the electrolytic production of aluminium, from pitch and petroleum coke may still be an important source of PAH, but confirmatory data are not available. Estimates of PAH released during the production of aluminium in the Netherlands in 1988 ranged from about 0.3 t benzo[*ghi*]perylene to 24 t naphthalene (Slooff et al., 1989). The estimated total airborne PAH released in 1985 was about 1000 t in the USA, 160 t in Norway, and 35 t in Sweden (Bjørseth & Ramdahl, 1985). In 1990, the input of total PAH from this source into the atmosphere in Canada was 930 t (Environment Canada, 1994).

In horizontal and vertical Söderberg aluminium production processes in Sweden, the emission factors per tonne of aluminium were 0.11 kg benzo[*a*]pyrene and 4.4 kg total PAH for the horizontal process and 0.01 kg benzo[*a*]pyrene and 0.7 kg total PAH for the vertical process (Alfheim & Wikström, 1984). In a Norwegian vertical Söderberg aluminium production plant, the emission factors were 0.005–0.015 kg/t aluminium for benzo[*a*]pyrene and 0.3–0.5 kg/t for total PAH (European Aluminium Association, 1990).

(iv) *Iron and steel production*

The total emissions of PAH resulting from iron and steel production with carbon electrodes containing tar and pitch in Norway was estimated to be about 34 t in 1985 (Bjørseth & Ramdahl (1985), but the database for this estimate is limited. The release of total PAH from metallurgical processes in Canada where similar electrodes were used, including ferro-alloy smelters but excluding aluminium production, was estimated to be 19 t in 1990 (Environment Canada, 1994).

(v) *Foundries*

PAH are formed during casting by thermal decomposition of carbonaceous ingredients in foundry moulding sand, and they partly vaporize under the extremely hot reducing conditions at the mould-metal interface. Thereafter, the compounds are adsorbed onto soot, fume, or sand particles. Organic binders, coal powder, and other carbonaceous additives are the predominant sources of PAH in iron and steel foundries (IARC, 1984b).

In pyrolysis experiments with green-sand additives, the highest PAH levels were found in coal-tar pitch, with values per kilogram of additive of 3100 mg benzo[*a*]pyrene, 3000 mg benzo[*b+j+k*]fluoranthenes, 3000 mg pyrene, and 2900 mg fluoranthene; the lowest levels were found in vegetable product additives, such as maize starch: 26 mg pyrene, 16 mg fluoranthene, 3 mg benzo[*b+j+k*]fluoranthenes, and 2 mg benzo[*a*]pyrene (Novelli & Rinaldi, 1979). Less than 0.002 mg/kg benzo[*a*]pyrene was found in foundry moulding sand when petrol resin, polystyrol, or polyethylene was used as the carrier and 7.5 mg/kg when hard coal was used as the carrier. The PAH content was directly correlated with the amount of hydrocarbon carrier in the sand (Schimberg et al., 1981).

The following levels of PAH were found in the stack gases of one French automobile foundry: fluoranthene, 980 ng/m^3; benz[*a*]anthracene, 830 ng/m^3; benzo[*a*]pyrene, 570 ng/m^3; benzo[*b*]fluoranthene, 460 ng/m^3; indeno[1,2,3-*cd*]pyrene, 370 ng/m^3; anthracene, 250 ng/m^3; benzo[*k*]fluoranthene, 220 ng/m^3; perylene, 160 ng/m^3; benzo[*ghi*]perylene, 130 ng/m^3; chrysene, 110 ng/m^3; coronene, 28 ng/m^3; and pyrene, 15 ng/m^3. No further information was given about the sampling site (Masclet et al., 1984). The total emission of PAH into the atmosphere from iron foundries in the Netherlands was estimated to be about 1.3 t in 1988 (Slooff et al., 1989).

(vi) *Other industrial sources*

The estimated release of 10 PAH into the atmosphere in the Netherlands in 1988 was about 1.3 t from sinter processes and 0.2 t/year from phosphorus production (Slooff et al., 1989).

(b) Emissions to the hydrosphere

(i) *Aluminium production*

PAH levels in wastewater from aluminium production in Norwegian plants are shown in Table 15. At the beginning of the 1970s, the release of anthracene and phenanthrene into the aqueous environment from aluminium production in western Europe was estimated to be 180 t/year (Palmork et al., 1973). About 0.6 t/year are released into water by the aluminium producing industry in the Netherlands (Slooff et al., 1989).

Table 15. Polycyclic aromatic hydrocarbon concentrations [µg/litre] in wastewater from aluminium production in Norway

Compound	[1]	[2]	[3]
Acenaphthene	NR	NR	5
Acenaphthylene	NR	NR	1
Anthracene	1.1–2.8	0.9	10
Anthenthrene	< 1–3.2	NR	NR
Benzo[*b*+*k*]fluoranthenes	6.8-38.1	NR	NR
Benzo[*j*+*k*]fluoranthenes	NR	10.5	5
Benz[*a*]anthracene	2.5–5.6	14.6	11
Benzo[*a*]fluorene	1.5–3.4	8.2	13
Benzo[*a*]pyrene	1.3–7.4	13.5	4
Benzo[*b*]fluoranthene	NR	21.2	9
Benzo[*b*]fluorene	1.3–3.0	7.2	2
Benzo[*c*]phenanthrene	NR	NR	3
Benzo[*e*]pyrene	2.6–16.4	17.0	5
Benzo[*ghi*]perylene	NR	8.3	2
Chrysene and triphenylene	5.8–16.0	27.3	17
Coronene	< 1-2.0	NR	NR
Dibenz[*a,h*]anthracene	NR	NR	1
Fluoranthene	12.4–20.8	7.5	124
Fluorene	NR	NR	3
Indeno[1,2,3-*cd*]pyrene	NR	8.1	2
1-Methylphenanthrene	NR	0.4	NR
Naphthalene	NR	NR	1
Perylene	NR	3.2	1
Phenanthrene	14.0–23.1	1.8	34
Pyrene	5.6–15.3	6.4	76

[1] Two samples of wastewater with two runs each from one aluminium production plant (Kadar et al., 1980); **[2]** Wastewater from one aluminium production plant; no further information (Olufsen, 1980); **[3]** Effluent from gas washers from one aluminium smelter (Berglind, 1982)
When the water samples were filtered through solid sorbents, the results may be underestimates of the actual content (see section 2.4.1.4).
NR, not reported

(ii) *Other industrial sources*

No recent data were available on PAH emissions into the aqueous environment from coal- or oil-fired power plants. PAH were found in the final effluent from a British municipal incinerator at concentrations ranging from

< 0.01 µg/litre each for coronene and indeno[1,2,3-*cd*]pyrene to 0.62 µg/litre fluoranthene. The calculated daily output of single compounds was in the low milligram range, with a maximum of 16 mg/d. Actual data were not available (Davies et al., 1976).

Numerous PAH were detected in the final effluent from a Norwegian ferro-alloy smelter in which the wastewater from gas scrubbers was treated by chemical flocculation. The concentrations were 50 µg/litre phenanthrene, 45 µg/litre pyrene, 40 µg/litre fluoranthene, 39 µg/litre acenaphthylene, 27 µg/litre fluorene, 17 µg/litre acenaphthene, 13 µg/litre chrysene plus triphenylene, 11 µg/litre anthracene, 10 µg/litre naphthalene, 10 µg/litre benz[*a*]anthracene, 9 µg/litre benzo[*b*]fluoranthene, 6 µg/litre benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, 6 µg/litre benzo[*e*]pyrene, 6 µg/litre benzo[*a*]pyrene, 3 µg/litre benzo[*c*]phenanthrene, 3 µg/litre indeno[1,2,3-*cd*]pyrene, 3 µg/litre benzo[*ghi*]perylene, 2 µg/litre benzo[*a*]fluorene, 2 µg/litre benzo[*b*]fluorene, 2 µg/litre perylene, and 1 µg/litre dibenz[*a,h*]-anthracene. The PAH contents of wastewater from gas washers in one Norwegian steel production plant were of the same order of magnitude (Berglind, 1982).

The release of 10 PAH into water from different industries in the Netherlands was estimated to be 4 t/year (Slooff et al., 1989).

(c) Emissions to the geosphere

The levels of PAH in ash samples from various incinerators are shown in Table 16. The values given by Eiceman et al. (1979) were based on the gas chromatographic responses of pyrene and benzo[*a*]pyrene. The concentrations of PAH in ashes from coal-fired power plants were of the same magnitude as the background levels of these compounds in soil, but fly ash from municipal waste incinerators may contain significantly higher levels (Guerin, 1977; Kanij, 1987). The total PAH content of filter residues in incinerators was about 0.20–0.5 µg/g. The compounds are assumed to be tightly bound to particle surfaces and not mobile in an aqueous environment in the absence of organic solvents (WHO, 1988). In a comparison of 26 incineration plants, combustion conditions were shown to have a marked influence on PAH release (Wild et al., 1992).

The material dredged from harbour areas may have a significant PAH content (see also sections 5.3.3 and 5.3.4). The annual load of naphthalene, anthracene, phenanthrene, fluoranthene, benz[*a*]anthracene, chrysene, benzo[*k*]-fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, and indeno[1,2,3-*cd*]pyrene in material dredged from Rotterdam harbour was about 12 t (year not given). The main PAH were fluoranthene and benz[*a*]anthracene (Slooff et al., 1989).

Table 16. Polycyclic aromatic hydrocarbon concentrations (µg/kg) in ash samples from coal-fired power plants and municipal waste and sewage sludge incinerators

Compound	Coal-fired power plants		Municipal waste incinerators						Sewage sludge incinerators
	Netherlands [1]	USA [2]	Canada [3]	Japan [3]	Netherlands [3]	Canada [4]	UK [5] (mean)	Italy [6]	(UK) [5]
Acenaphthene + fluoranthene	NR	NR	NR	NR	NR	NR	1–258 (7.8)	289–1022[c]	NR
Acenaphthylene	NR	NR	NR	NR	NR	3.35	NR	5–1394	NR
Anthracene	< 0.14–0.5	NR	10/500	10/10	200	NR	1–62 (2.3)	42–651	NR
Anthanthrene	< 0.24–< 0.5	NR	NR	NR	NR	NR	NR		NR
Benz[*a*]anthracene	< 0.6–< 1.2	NR	NR	NR	NR	NR	1–1646[a] (12)	280–1278	3[a]
Benzo[*a*]fluoranthene	NR	36.8	NR	NR	NR	NR	NR		NR
Benzo[*a*]pyrene	< 0.29–< 1.8	NR	ND/400	ND/ND	ND	NR	1–596 (8.2)	1014–3470	3
Benzo[*b*]fluoranthene	< 0.6–< 0.29	NR	NR	NR	NR	NR	1–873 (5.7)	1818	6
Benzo[*b*]fluorene	< 2.0–< 4	11.8	NR	NR	NR	NR	NR		NR
Benzo[*e*]pyrene	< 2.9–< 6	NR	NR	NR	NR	NR	NR	458–1786	NR
Benzo[*ghi*]perylene	< 1.6–< 1.7	NR	NR	NR	NR	NR	10–9507 (62.3)	700–2377	135
Benzo[*j*]fluoranthene	< 4.5–< 9	NR	ND/400[b]	ND/ND[b]	ND[b]	NR	NR		NR
Benzo[*k*]fluoranthene	< 0.15–< 2.8	NR	NR	NR	NR	NR	1–276 (1.5)	1535	NR
Chrysene	< 1.5–< 3	NR	NR	NR	NR	NR	NR	570–1973	NR
Coronene	NR	NR	NR	NR	NR	NR	3–238 (31.3)		36
Dibenz[*a,h*]anthracene	< 4.2–< 8.2	NR	NR	NR	NR	NR	1–167 (5.2)	57/69	1

Table 16 (contd)

Compound	Coal-fired power plants		Municipal waste incinerators						Sewage sludge incinerators (UK)
	Netherlands	USA	Canada	Japan	Netherlands	Canada	UK	Italy	
	[1]	[2]	[3]	[3]	[3]	[4]	[5] (mean)	[6]	[5]
Fluoranthene	1.1–5.2	< 13.4	2/500	3/ ND	20	2.14–43.2	1–765 (8.6)	1684–10 890	1
Fluorene	NR	NR	ND/10	ND/ND	60	2.57/4.41	NR	45–522	NR
Indeno[1,2,3-*cd*]pyrene	< 0.82–< 1.6	NR	NR	NR	NR	NR	NR	478–1343	NR
Naphthalene	NR	8.3	NR	NR	NR	NR	4/15 (0.2)		NR
Perylene	< 0.16–< 0.3	NR	NR	NR	NR	NR	NR	259	NR
Phenanthrene	4.0–43	17.6	NR	NR	NR	8.76–154[c]	2–5402 (36.5)	1616–7823	6
Pyrene	0.72–2.9	< 19.0	1/500	1/ ND	10	2.47–19.6	1–3407 (45.3)	1863–8799	10
Triphenylene	< 2.5–< 5.0	NR	NR	NR	NR	12.7[a]	NR		NR

[1] Pulverized coal ash (Kanij, 1987); **[2]** Fly ash (Guerin, 1977); **[3]** Fly ash (Eiceman et al., 1979); **[4]** Fly ash (Chiu et al., 1991); **[5]** Fly ash; 26 incinerators with different firing techniques (Wild et al., 1992); **[6]** Fly ash from electrostatic precipitator and scrubber (Morselli & Zappoli, 1988)

NR, not reported; ND, not detected; /, single measurements

[a] With chrysene
[b] Isomers not specified
[c] With anthracene
[d] Only acenaphthene

3.2.7.2 Other diffuse sources

(a) Atmosphere

(i) *Mobile sources*

PAH are released into the atmosphere by motor vehicle traffic. The profile of the PAH released and the quantity of PAH in the exhaust are fairly similar, independently of the type of engine and the PAH content of the fuel, indicating that the emitted compounds are formed predominantly during combustion (Meyer & Grimmer, 1974; Janssen, 1980; Stenberg, 1985; Williams et al., 1989). PAH accumulate in used engine oil, but the importance of the PAH content of engine oil on emissions is still under discussion. Janssen (1980), Pischinger & Lepperhoff (1980), and Stenberg (1985) assumed that the PAH content of the oil played only a minor role, but Williams et al. (1989) showed in tests with diesel fuel that it may contribute considerably to the release of particulate PAH. There is also doubt about whether PAH emissions are indepen-dent of the aromaticity of the fuel. Janssen (1980) stated that release of PAH into the atmosphere is not increased if the aromaticity does not exceed a concentration of 50% volume (see also Schuetzle & Frazier, 1986). According to Stenberg (1985), the release of PAH by automobile traffic is dependent on the:

- *aromaticity of the fuel*;
- *starting temperature*: Starting at −10 °C results in threefold higher PAH emissions than a standardized cold start (+ 23 °C); the emission factors measured by Larssen (1985) were significantly higher in winter than in summer.
- *ambient temperature*: Low ambient temperatures (5–7 °C) increase PAH emissions from petrol-fuelled vehicles by five to 10 times, depending on the engine used.
- *test conditions*: Three standardized test cycles are in general use: a test developed by the Economic Commission for Europe of the United Nations (ECE) in Europe; the Federal Test Procedure (FTP) in the USA; and the Japanese test cycle in Japan. Emissions at cold start may be lower and those at hot start slightly higher in the FTP than in the ECE test, but overall agreement between the tests is good.
- *air:fuel ratio* (l): Small variations around l = 1, representing stoichiometric levels of fuel and air, do not affect PAH emissions significantly; richer mixtures lead to increasing PAH emissions, and bad ignition at l = 0.8 causes a sharp increase in PAH emissions.
- *type of fuel*: Emissions of the sum of phenanthrene, fluoranthene, pyrene, benzo[*ghi*]fluoranthene, cyclopenta[*cd*]pyrene, benz[*a*]-anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]-pyrene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]-perylene, and coronene decreased in the FTP cycle as follows: diesel

(total PAH; 960 μg/km) > petrol (170 μg/km) > petrol containing methanol or ethanol (43–110 μg/km) > methanol = liquefied petroleum gas = catalyst-equipped petrol-fuelled vehicles (6–9 μg/km) (Stenberg, 1985). In comparable measurements, similar results were obtained but with a much lower average emission rate for diesel-fuelled vehicles: 186 μg/km for total PAH, including fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]-fluoranthene, benzo[*e*]-pyrene, benzo[*a*]pyrene, perylene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]-perylene, and coronene. It was not stated whether the difference in the emission rates was due to the numbers of PAH chosen for analysis (Lies et al., 1986).

PAH emissions in the exhaust from spark-ignition automobile engines can be reduced by operation with lean air:fuel ratios, smaller quenching distances in the combustion chamber, and increased cylinder wall temperatures in the engine (Pischinger & Lepperhoff, 1980; Lepperhoff, 1981). Diesel-fuelled engines with low emissions of total unburnt gaseous hydrocarbons have low rates of PAH emission. Control can therefore be achieved by using conventional techniques for reducing unburnt gaseous hydrocarbons (Williams et al., 1989).

Fluoranthene and pyrene constitute 70–80% of total PAH emissions from vehicles (Lies et al., 1986; Volkswagen AG, 1989; see also Table 17), whereas the emissions from one diesel-fuelled truck consisted mainly of naphthalene and acenaphthene (Nelson, 1989). Although cyclopenta[*cd*]pyrene is emitted at a high rate from petrol-fuelled engines, its concentration in diesel exhaust is just above the limit of detection, probably because the oxidizing conditions in diesel-fuelled engines decompose this relatively reactive compound (Lies et al., 1986).

The amounts of PAH released from vehicles with three-way catalytic converters are much lower than those from vehicles without catalysts (Table 18). The total amount of PAH was increased by a factor of about 40 between new and used catalytic converters (Hagemann et al., 1982). PAH emissions from diesel-fuelled vehicles can be reduced by > 90% by a combination of a catalytic converter and a particulate trap, as shown by experiments with a heavy-duty diesel-fuelled truck (Westerholm et al., 1989). Westerholm et al. (1991) found benz[*a*]anthracene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]-pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]perylene, fluoranthene, pyrene, anthracene, and coronene in much lower amounts than other investigators, while some other PAH that were not measured by other investigators, especially phenanthrene and 1-methylphenanthrene, were detected at quite high concentrations. These differences are possibly due to the driving cycle used.

Measurements made on particulate matter in the exhausts of light- and heavy-duty diesel-fuelled vehicles with different fuel qualities showed concentrations of 1 mg/kg each of benz[*a*]anthracene, benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene, benzo[*a*]pyrene plus benzo[*e*]pyrene, and

Table 17. Polycyclic aromatic hydrocarbon emission factors (μg/km) for petrol-fuelled vehicles

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
Anthracene	NR	0.7/0.7[a]	NR	2/99[b] 37/1988[c]	NR	21–42	0.6
Anthanthrene	NR	0.2/1.3	NR	NR	NR	NR	NR
Benzo[*b*+*j*+*k*]fluoranthene	NR	NR	NR	NR	NR	NR	7.6
Benzo[*b*+*k*]fluoranthene	NR	3.9/7.0	NR	NR	0.23–0.54/2.55–9.20	NR	NR
Benz[*a*]anthracene	NR	5.7/5.9	3.5–9.0	NR	0.06–0.35/2.5–8.0	5–16	5.1
Benzo[*a*]pyrene	NR	1.9/4.51	1.5–14.5	0.06–2/1–12[b]	0.06–0.62/1.30–10.4	2–11	3.7
Benzo[*e*]pyrene	NR	2.6/6.2	NR	0.2/2–14[b]	0.08–0.54/2.54–9.20	NR	5.1
Benzo[*ghi*]fluoranthene	NR	5.6/12	NR	NR	NR	NR	8.8
Benzo[*ghi*]perylene	NR	5.9/13	NR	NR	0.19–0.75/1.45–17.5	5–21	18.9
Benzo[*j*]fluoranthene	NR	1.1/0.9	NR	NR	NR	NR	NR
Benzo[*k*]fluoranthene	NR	NR	NR	NR	NR	0–5	NR
Chrysene	NR	6.7/8.7	NR	NR	0.12–0.73/2.78–23.1	11–42	7.7
Coronene	NR	6.5/12	1.5–20.0	NR	NR	NR	29.5
Cyclopenta[*cd*]pyrene	NR	2.9/12	NR	NR	NR	NR	16.5
Fluoranthene	NR	14/20	NR	3/139–211[b] ND/186–280[c]	2.7/43.3[d]	11–158	10.4
Indeno[1,2,3-*cd*]pyrene	NR	1.7/3.6	NR	NR	0.06–0.43/0.83–6.67	5–21	4.2
Naphthalene	8100–8600[a]	NR	NR	NR	NR	2300[f] 210–2651	NR

Table 17 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
Perylene	NR	0.3/0.5	NR	NR	0.01–0.06/0.25–1.82	NR	NR
Phenanthrene	NR	2.6/2.9	NR	NR	NR	84–210	1.8
Pyrene	NR	28/31	43–184	4–16/12-268[b] ND/124–360[c]	2.9/43.0[b]	NR	19.2

NR, not reported; ND not detected (detection limit not stated); /, single measurements
[a] Two driving distances
[b] Only particulate phase considered
[c] Only gaseous phase considered
[d] Average
[e] Depending on analytical conditions
[f] With converter

[1] From measurements in tunnel with converters (Hampton et al., 1983); **[2]** One vehicle without converter (Alsberg et al., 1985); **[3]** Various tests conducted mainly in the 1970s, some unstandardized, different numbers of vehicles, without converters (Stenberg, 1985); **[4]** FTP cycle only, number of vehicles not given; year of manufacture 1980-85 = petrol-engine vehicles with converter; 1973-81 = petrol-engine vehicles without converter (Schuetzle & Frazier, 1986); **[5]** Various standardized test procedures; four petrol-engine vehicles without, seven with three-way-converter for each test, all with four or five cylinders (Volkswagen AG, 1988); **[6]** No information about test cycle or number of cars tested; city roads, motorways and other roads tested; no distinction between vehicles with and without converter, unless otherwise stated (Slooff et al., 1988,1989); **[7]** One petrol-engine vehicle without converter in USFTP test cycle (Strandell et al., 1994)

Table 18. Polycyclic aromatic hydrocarbon emission factors (μg/km) for diesel-fuelled vehicles

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Acenaphthene	NR	NR	NR	NR	NR	NR	41–128	NR
Anthracene	17/63	65–273[a] 1305–5568[c]	1.2/3.0	NR	21–73[b]	3.3	2.9–26	4.6
Benzo[*b*+*j*+*k*]fluoranthene	NR	NR	NR	NR	NR	NR	1.7–12[d]	5.0
Benzo[*b*+*k*]fluoranthene	2.6/47	NR	3.9/6.1	5.57–14.96	NR	0.29	NR	NR
Benz[*a*]anthracene	8/43[a]	NR	4.0/7.0	2.73–3.91	11–21[b]	0.47	0.7–9.6	2.0
Benzo[*a*]fluorene	NR	NR	NR	NR	NR	2.4	NR	NR
Benzo[*a*]pyrene	< 1/20	0.6–34[a]	1.6/2.2	2.09–7.23	1–5	< 0.06	0.5–3.2	1.5
Benzo[*e*]pyrene	3/38	2–40[a]	2.5/4.1	2.40–52.8	NR	0.15	1.1–9.9	4.0
Benzo[*ghi*]fluoranthene	NR	NR	4.0/12	NR	NR	1.5	NR	10.6
Benzo[*ghi*]perylene	< 1/18	NR	1.9/3.1	2.84–26.3	9[e]	< 0.13	0.5–3.7	2.0
Chrysene	14/67	NR	11/25	4.7–21.1	16–42[b]	2.8[f]	3.5–28	3.7
Coronene	NR	NR	0.3/20.7	NR	NR	< 0.01	NR	NR
Cyclopenta[*cd*]pyrene	NR	NR	3.6/3.9	NR	NR	0.18	NR	4.0
Fluoranthene	58/200	139–580[a] 186–771[c]	13/38	70[g]	21–105[b]	17	14–34	43.7
Fluorene	NR	NR	NR	NR	NR	NR	38–228	NR
Indeno[1,2,3-*cd*]pyrene	NR	NR	1.5/2.3	0.89–7.52	9[e]	< 0.04	NR	1.2
1-Methylphenanthrene	NR	NR	NR	NR	NR	41	NR	NR
Naphthalene	NR	NR	NR	NR	2100–6302[b]	NR	1030–1805	NR

Table 18 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Perylene	< 1/2	NR	NR	0.23–1	NR	< 0.01	NR	NR
Phenanthrene	295/524	NR	4.6/25	NR	NR	2.9	79–308	54.8
Pyrene	< 0.9/22	24–734[a] 702–982[c]	20/104	66.9[g]	NR	11	9–30	35.4

NR, not reported; /, single measurements; **[1]** ECE test; two passenger cars with < 50 000 and > 100 000 km odometer readings (Scheepers & Bos, 1992); **[2]** FTP cycle; number of vehicles not given; year of manufacture, 1980-85 (Schuetzle & Frazier, 1986); **[3]** Chassis dynamometer; one heavy-duty vehicle (Westerholm et al., 1986); **[4]** Various standardized testing procedures; seven vehicles with four or five cylinders for each test (Volkswagen AG, 1988); **[5]** No information on test cycle or number of cars tested; three traffic situations (Slooff et al., 1989); **[6]** Bus cycle simulating public transport (duration 29 min; driving distance, 11.0 km; average speed, 22.9 km/h); one heavy-duty truck; measurement of particle phase (Westerholm et al., 1991); **[7]** Bus cycle (duration, about 10 min after warm-up, each ramp consisting of 10s acceleration, 10 s constant speed of 12 km/h, 4.5 s deceleration, 7 s idling); three trucks and two buses without particle trap, two buses with particle trap (Lowenthal et al., 1994); **[8]** US FTP cycle; one passenger car (Strandell et al., 1994)

[a] Particle phase
[b] Automobiles and trucks
[c] Gas phase
[d] Isomers not specified
[e] Trucks
[f] With triphenylene
[g] Average

benzo[*ghi*]perylene and 290 mg/kg pyrene. The results were strongly dependent on the driving cycle and individual engine conditions (CONCAWE, 1992).

The PAH concentrations measured in the exhaust gases of different vehicles are shown in Table 19. The differences in PAH emissions from petrol- and diesel-fuelled vehicles are still under discussion. When the data of Behn et al. (1985) are compared with those of Klingenberg et al. (1992), diesel-fuelled vehicles emitted larger amounts of PAH than petrol-fuelled vehicles. Benzo[*a*]pyrene was emitted at a rate of 6 µg/km from a petrol-fuelled vehicle without a catalyst and at 5 µg/km from a diesel-fuelled vehicle (Gibson, 1982). When the PAH emissions from 10 petrol- and 20 diesel-engined vehicles were measured under three urban cycles, the mean emission factors (µg/km) for benzo[*a*]pyrene were 12 with petrol and 0.56 with diesel in a cold, low-speed cycle, 0.50 with petrol and 0.37 with diesel in a hot, low-speed cycle, and 0.37 with petrol and 0.24 with diesel in a hot, free-flow cycle (Combet et al., 1993). Considerably higher emission rates were found from four petrol-fuelled passenger cars without catalysts, 11 with catalysts, and eight diesel-fuelled passenger cars, two of which had oxidation catalysts, on a chassis dynamometer at the USA FTP 75 cycle. The diesel-fuelled vehicles emitted about as much benzo[*a*]pyrene as the petrol-fuelled vehicles without catalysts (5–25 µg/km), while the petrol-fuelled vehicles with catalysts had emission rates significantly below 2 µg/km. The diesel-fuelled vehicles with oxidation catalysts had emissions of about 5 µg/km (Klingenberg et al., 1992).

The following emission factors were given for motorcycles and two-stroke mopeds: 1000 µg/km naphthalene, < 32–650 µg/km phenanthrene, < 11–170 µg/km anthracene, < 5–110 µg/km fluoranthene, < 2–11 µg/km chrysene, < 2–11 µg/km indeno[1,2,3-*cd*]pyrene, < 1–1200 µg/km benz[*a*]anthracene, 0–63 µg/km benzo[*ghi*]perylene, 0–16 µg/km benzo[*a*]pyrene, and 0–11 µg/km benzo[*k*]fluoranthene (Slooff et al., 1989).

Further PAH emissions may result from the abrasion of asphalt by vehicle traffic, so that PAH in asphalt and bitumens (see section 3.2.1) may contribute considerably to the total PAH emissions due to automobile traffic. The abrasion caused by spiked tyres in winter was estimated to be 20–50 mg/km (Lygren et al., 1984).

Another source of PAH from motor vehicle traffic is clutch and break linings, which are subject to considerable thermal stress, sometimes resulting in pyrolytic decomposition of abraded particles. Numerous PAH were found in the abraded dust of brake and clutch linings in one study, but the values show large standard deviations, due, presumably, to the fact that the substances are adsorbed onto asbestos fibres from which they are difficult to separate (Knecht et al., 1987). Total PAH release from clutch and brake linings cannot be estimated from the available data.

Rubber vehicle tyres contain highly aromatic oils as softeners. These oils, which can contain up to 20% PAH, are used at concentrations of 15–20% in

Table 19. Polycyclic aromatic hydrocarbon concentrations (μg/m³) in the exhaust gases of different vehicles

Compound	[1]	[2]	[3]
Acenaphthene	NR	NR	< 0.02–0.81
Acenaphthylene	NR	NR	< 0.02–4.16
Anthracene	NR	NR	< 0.02–6.45
Anthanthrene	0.02–0.07	0.11–0.12	NR
Benz[*a*]anthracene	1.91–2.24	3.53–4.64	NR
Benzo[*a*]pyrene	0.46–0.76	2.03–2.33	< 0.02–4.97
Benzo[*b*]fluoranthene	1.53–2.04[a]	7.37–8.58[a]	0.06–6.63
Benzo[*b*]fluorene	NR	NR	0.11–12.7
Benzo[*e*]pyrene	1.07–1.24	2.46–2.90	0.09–6.16
Benzo[*ghi*]fluoranthene	0.46–0.59	4.81–7.19	NR
Benzo[*ghi*]perylene	0.76–1.04	3.42–4.41	0.22–1.81
Benzo[*k*]fluoranthene	NR	NR	< 0.02–2.68
Chrysene	2.37–2.97[b]	7.37–8.58[b]	0.07–25.48
Coronene	0.26–0.30	1.82–2.32	< 0.02–1.80
Cyclopenta[*cd*]pyrene	1.86/2.26	5.80–6.09	NR
Dibenz[*a,h*]anthracene	0.04–0.07	0.32–0.35	< 0.02–0.44
Fluoranthene	11.83–13.09	20.90–25.30	0.16–35.94
Fluorene	NR	NR	0.06–2.16
Indeno[1,2,3-*cd*]pyrene	0.30–0.41	2.89–4.06	< 0.02–0.80
Perylene	0.10–0.26	0.21–0.33	0.13–5.55
Phenanthrene	NR	NR	< 0.02–4.16
Pyrene	6.86–8.96	12.20–15.20	0.06–21.31

NR, not reported; /, single measurements; **[1]** One vehicle with spark-ignition engine on chassis dynamometer at 75% of maximum engine performance (velocity, about 50 km/h) with varying test periods (Behn et al., 1985); **[2]** One turbo-charged diesel-fuelled vehicle on chassis dynamometer at75% of maximum engine perfomance (velocity, about 50 km/h) and a test period of 0.5 h; three tests for each component (Behn et al., 1985); **[3]** Two diesel-fuelled truck engines at different engine speeds (Moriske et al., 1987)

[a] With benzo[*k*]fluoranthene
[b] With triphenylene

rubber blends (Duus et al., 1994). In Sweden, it was considered that the input of PAH to the atmosphere from rubber particles was important (National Chemicals Inspectorate, 1994).

According to estimates for Belgium, western Germany, and the Netherlands in 1985, the annual PAH input into the atmosphere from vehicle traffic ranges from < 10 t/year for benzo[*ghi*]fluoranthene, benz[*a*]anthracene, benzo[*k*]-

fluoranthene, benzo[*a*]pyrene, and indeno[1,2,3-*cd*]pyrene, to < 10–20 t/year for anthracene, fluoranthene, and chrysene, to 10–70 t/year for phenanthrene, to about 100–1000 t/year for naphthalene (Slooff et al., 1989). Values of the same order of magnitude were reported for emissions of naphthalene in 1987 (Society of German Chemists, 1989) and benzo[*a*]pyrene in 1989 in western Germany (Ministers for the Environment, 1992) and for total PAH in 1985 in Norway and Sweden (Bjørseth & Ramdahl, 1985). The total annual PAH input from vehicle traffic in the USA in 1985 was about 2200 t/year (Bjørseth & Ramdahl, 1985). In Canada, the total PAH input was estimated to be about 200 t in 1990; 155 t were assumed to be due to diesel-fuelled and 45 t to petrol-fuelled vehicles (Environment Canada, 1994).

Measurements of PAH concentrations in a Belgian highway tunnel in 1991 were used to calculate emission factors of 2 µg/km for indeno[1,2,3-*cd*]pyrene and coronene and 32 µg/km for benzo[*ghi*]perylene. The corresponding annual PAH emissions in Belgium were estimated to be 0.11 t/year for perylene and anthanthrene and 1.3 t/year for benzo[*ghi*]perylene; the combined release of pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*e*]pyrene, perylene, anthanthrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene, dibenzo[*a,c*]anthracene, dibenzo[*a,h*]anthracene, and coronene was 8.3 t/year (De Fré et al., 1994).

The importance of PAH released by aircraft is also under discussion. While Bjørseth & Ramdahl (1985) classified the maximum emission in Norway and Sweden in 1985 of 0.1 t/year as small, Slooff et al. (1989) estimated that the release of naphthalene, anthracene, phenanthrene, fluoranthene, benz[*a*]anthracene, chrysene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, and indeno[1,2,3-*cd*]pyrene was 51 t/year in 1985. The following concentration ranges were measured in the exhaust gases from two US by-pass turbine engines at various power settings: naphthalene, 0.77–4.7 µg/m^3; phenanthrene, 0.46–1.3 µg/m^3; pyrene, 0.15–0.61 µg/m^3; fluoranthene, 0.13–0.51 µg/m^3; acenaphthene, 0.03–0.21 µg/m^3; anthracene, 0.029–0.12 µg/m^3; benzofluoranthenes (unspecified), 0.028–0.096 µg/m^3 (isomers not specified); chrysene, 0.026–0.064 µg/m^3; benzo[*a*]pyrene, 0.021–0.073 µg/m^3; benz[*a*]anthracene, 0.019–0.16 µg/m^3; acenaphthylene, 0.017–0.31 µg/m^3; benzo[*e*]pyrene, 0.017–0.057 µg/m^3; dibenz[*a,h*]anthracene, 0.011–0.064 µg/m^3; indeno[1,2,3-*cd*]pyrene, 0.011–0.054 µg/m^3; and benzo[*ghi*]perylene, 0.011–0.045 µg/m^3. Cyclopenta[*cd*]pyrene was not detected (limit of detection not stated) (Spicer et al., 1992).

(ii) *Domestic residential heating*

The main PAH released by domestic slow-combustion furnaces and hard-coal and brown-coal coal stoves were fluoranthene, pyrene, and chrysene, which comprised 70-80% of the total PAH in model experiments (Ahland &

Mertens, 1980). The specific emission factors for various fuels used in residential heating are shown in Table 20 for coal stoves and Table 21 for wood stoves (Bjørseth & Ramdahl, 1985).

Few data are available on the release of PAH from oil stoves. Benzo[*a*]pyrene was detected at a concentration of <0.05 µg/kg in one burner-boiler combination (Meyer et al., 1980), and 0.006 and 4 µg/kg benzo[*a*]pyrene and 0.02 and 15 µg/kg benzo[*e*]pyrene were found during testing of atomizer and vaporizer oil heating techniques, respectively (Ahland et al., 1985). PAH emissions from residential oil heating seem to be about one order of magnitude lower than those from coal stoves.

Numerous PAH, including acenaphthene, acenaphthylene, fluorene, phenanthrene, anthracene, 1-methylphenanthrene, fluoranthene, pyrene, benzo[*a*]fluorene, benzo[*ghi*]fluoranthene, benzo[*c*]phenanthrene, cyclopenta[*cd*]pyrene, benz[*a*]anthracene, chrysene plus triphenylene, benzo[*b*]-fluoranthene, benzo[*j*]fluoranthene, benzo[*j*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, perylene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]perylene, and anthanthrene, were detected in atmospheric emissions from straw-burning residential stoves, at concentrations mainly in the range of 10 µg/kg to 19 mg/kg (Ramdahl & Moller, 1983).

The total PAH content of barbecue briquettes was 2.5–13 µg/g sample. PAH were found in coal and charcoal briquettes but not in lava stones or pressed sawdust briquettes (Kushwaha et al., 1985).

The PAH content of soot from domestic open fires was 3–240 mg/kg benzo[*a*]pyrene, 2–190 mg/kg chrysene, 2–100 mg/kg benz[*a*]anthracene, 1–77 mg/kg indeno[1,2,3-*cd*]pyrene, 2–39 mg/kg benzo[*e*]pyrene, 1–29 mg/kg benzo[*ghi*]perylene, 1–18 mg/kg coronene, 1–14 mg/kg perylene, and 1–12 mg/kg anthracene (Cretney et al., 1985).

The amounts of PAH emitted from coal-fired domestic stoves seem to depend on the quality of the coal used and on the firing technique. Generally, hard coal has a higher energy content than other fuels; thus, less total PAH is emitted per unit of gained energy. The lowest specific emission factors for benzo[*a*]pyrene and benzo[*e*]pyrene were found with anthracite and the

Table 20. Specific polycyclic aromatic hydrocarbon emission factors (mg/kg) for residential coal stoves

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Acenaphthene	NR	NR	NR	NR	65	NR
Acenaphthylene	NR	NR	NR	0.427	NR	7.74
Anthracene	0.0039	NR	> 0.595	2.113	26[a,b]	1.49
Anthanthrene	NR	NR	0.03–0.08	0.665	NR	NR
Benz[a]anthracene	NR	NR	1.04–3.68	7.181	NR	0.61
Benzo[a]fluorene	0.0009	NR	NR	1.366	NR	NR
Benzo[a]pyrene	0.0003	0.014–17.4	0.043–1.3	4.303	5[c]	NR

Table 20 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Benzo[*b*]fluoranthene	0.0002	NR	2.028[d]	6.102	NR	NR
Benzo[*b*]fluorene	0.0007	NR	NR	0.874	NR	NR
Benzo[*c*]phenanthrene	NR	NR	1.462[e]	2.215	4	NR
Benzo[*e*]pyrene	0.0005	0.09–16.2	0.40–1.70	3.994	NR	NR
Benzofluoranthenes[f]	NR	NR	0.90–3.20	NR	6	NR
Benzo[*ghi*]fluoranthene	NR	NR	NR	3.323	NR	0.67
Benzo[*ghi*]perylene	0.0001	NR	0.30–0.50	3.855	NR	NR
Benzo[*j*]fluoranthene	NR	NR	NR	6.782	NR	NR
Benzo[*k*]fluoranthene	NR	NR	0.569	NR	NR	NR
Chrysene	0.0016[g]	NR	2.09 1.39–5.60[g]	9.571	6[h]	0.68
Coronene	NR	NR	0.081	1.898	NR	NR
Cyclopenta[*cd*]pyrene	NR	NR	0.145	3.590	NR	NR
Dibenz[*a,h*]anthracene	NR	NR	0.113	NR	5	NR
Fluoranthene	0.016	NR	3.30–17.0	28.4	9[a]	3.47
Fluorene	NR	NR	< 0.065	1.05	44	1.64
Indeno[1,2,3-*cd*]pyrene	0.0002	0.20–0.60	4.60	NR	4	NR
1-Methylphenanthrene	NR	NR	NR	2.217	NR	NR
Naphthalene	NR	NR	NR	NR	254	35.7
Perylene	NR	NR	0.20–0.50	1.134	NR	NR
Phenanthrene	0.046	NR	> 3.69	3.984	NR	7.42
Pyrene	0.020	NR	2.98–12.0	26.589	8	3.38
Triphenylene	NR	NR	0.804	NR	NR	NR

NR, not reported; **[1]** One new residential stove fuelled with charcoal (Ramdahl et al., 1982); **[2]** Five coal types: hard-coal and brown-coal briquettes and anthracite (Ahland et al., 1985); **[3]** Burning of brown coal in different domestic stoves: single values refer to one slow-combustion stove; ranges refer to one slow-combustion stove and one permanent built-in combustion stove at medium load (Grimmer et al., 1983a); **[4]** One slow-combustion stove fuelled with hard-coal briquettes (Grimmer et al.,1985); **[5]** One warm-air furnace and one hot-water boiler fuelled with three different bituminous coals (Hughes & DeAngelis, 1982); **[6]** Samples from chimney of a detached family house with brown-coal heating in Leipzig, Germany (Engewald et al., 1993)

[a] In particulate phase
[b] With phenanthrene
[c] With benzo[*e*]pyrene and perylene
[d] With benzo[*j*]fluoranthene
[e] With benzo[*ghi*]fluoranthene
[f] Isomers not specified
[g] With triphenylene
[h] With benz[*a*]anthracene

Table 21. Specific polycyclic aromatic hydrocarbon emission factors (mg/kg) for residential wood stoves

Compound	[1]	[2]	[3]
Anthracene	0.119–1.859	10.4–146.3[a]	130/3600
Benz[*a*]anthracene	0.060–0.781	NR	55/740
Benzo[*a*]fluorene	0.018–0.845	NR	NR
Benzo[*a*]pyrene	0.046–0.617	1.1–11.6[b]	NR
Benzo[*b*]fluoranthene	0.108–1.016	NR	NR
Benzo[*b*]fluorene	0.011–0.393	NR	NR
Benzo[*c*]phenanthrene	NR	0.2–2.3	NR
Benzo[*e*]pyrene	0.035–0.350	NR	NR
Benzofluoranthenes[c]	NR	1.5–15.9	NR
Benzo[*ghi*]fluoranthene	NR	0.4–6.7	NR
Benzo[*ghi*]perylene	0.034–0.544	1.1–9.9	NR
Chrysene	0.481–0.829[d]	1.3–37.1[e]	67/770[d]
Cyclopenta[*cd*]pyrene	0.04–0.720	0.5–8.9	15/800
Fluoranthene	0.296–3.245	1.2–31.6	190/2300
Indeno[1,2,3-*cd*]pyrene	0.033–0.415	NR	NR
1-Methylphenanthrene	0.141–2.213	NR	NR
Perylene	0.023–0.274	NR	NR
Phenanthrene	0.834–8.390	NR	480/7500
Pyrene	0.232–3.822	1.3–24.0	160/2100

NR, not reported; /, single measurements; **[1]** One small residential wood stove burning spruce and birch; normal and slow burning of each kind of wood (Ramdahl et al., 1982); **[2]** One zero-clearance fireplace with heat circulation and two airtight wood stoves (baffled and non-baffled) fuelled with red oak and yellow pine with different moisture contents (Peters et al., 1981); **[3]** One wood-burning stove with and without catalytic combustor (Tan et al., 1992)

[a] With phenanthrene
[b] With benzo[*e*)]pyrene and perylene
[c] Isomers not specified
[d] With triphenylene
[e] With benz[*a*]anthracene

highest with gas coal and gas-flame coal (Ahland et al., 1985). Model experiments with a slow-combustion stove showed that pitch-bound hard-coal briquettes emitted about 10 times more PAH than bitumen-bound briquettes (Ratajczak et al., 1984). The use of pitch-bound hard-coal briquettes for domestic heating may thus be an important source of PAH in the atmosphere.

Use of this fuel was restricted by law to permanent combustion stoves in western Germany in 1974, and since 1976 only bitumen-bound hard-coal briquettes have been produced there (Ratajczak et al., 1984). There is no comparable information for other countries. The levels of airborne PAH from a permanent combustion stove burning brown coal were two to four times higher than those from a slow-combustion stove with a medium load (Grimmer et al., 1983a).

About 25–1000 times more PAH are produced from burning wood than from the same mass of charcoal. Since the yield of energy per unit mass is similar, burning wood also produces more PAH per unit of energy. Burning conditions are apparently the major determinant of emission and are much more important than the kind of wood (Ramdahl et al., 1982). In areas where domestic heating is predominantly by wood burning, most airborne PAH may come from this source, especially in winter (e.g.Cooper, 1980). Using benz[*a*]pyrene as an indicator in extensive measurements in New Jersey, USA, the amounts emitted were found to be more than 10 times higher during the heating period than in seasons when heating is not required. An assessment of combustion source also showed that residential combustion of wood was the decisive factor (Harkov & Greenberg, 1985). About 43–47% of the total PAH released in winter in Fairbanks, Alaska, came from residential wood stoves (Guenther et al., 1988).

The PAH concentrations in gases in the chimney stacks of residential coal and oil furnaces are given in Table 22. The highest levels were found during the start of the burning process (Brockhaus & Tomingas, 1976). Measurements with five qualities of coal showed that Extrazit®, a specially treated coal, emitted smaller quantities of smoke and the lowest PAH levels, and anthracite briquettes emitted the highest levels. Presumably, the high PAH emissions from anthracite briquettes are due to the binding agent, hard coal-tar, which has an especially high PAH content. Furnaces with atomizer oil burners seemed to emit less PAH than those with vaporizers. Measurements in a slow-combustion stove and a tiled stove showed that the highest concentrations of PAH were associated with dust of a particle size of < 2.1 µm. As for residential heating with wood, in areas where the predominant form of domestic heating is coal burning, a major proportion of airborne PAH may come from this source, especially in winter (Moriske et al., 1987).

Estimates of annual PAH emissions due to residential heating are available for a few countries:

— In western Germany, the benzo[*a*]pyrene emissions were about 10 t in 1981 (Ahland et al., 1985), 7 t in 1985, and 2.5 t in 1988, mainly resulting from coal heating. The reduction in the release of PAH into the atmosphere due to domestic heating resulting from increasing use of oil and gas during the last 30-40 years was estimated to be 90–99% (Zimmermeyer et al., 1991).

Table 22. Polycyclic aromatic hydrocarbon concentrations (μg/m³) in stack gases from residential coal and oil stoves

Compound	Coal	Oil
Benz[*a*]anthracene	0.0157–2630	0.2–0.6
Benzo[a]pyrene	0.0016–1270	0.19–0.67
Benzo[*b*]fluoranthene	0.0188–3270	0.004–0.68
Benzo[e]pyrene	0.0261–3430	0.4–6.9
Benzo[*ghi*]perylene	0.010–1670	0.41–3.4
Benzo[*k*]fluoranthene	0.0044–1250	0.18–0.36
Chrysene	0.0142–2590	0.1–0.5
Coronene	0.003–710	0.15–0.47
Dibenz[*a,h*]anthracene	0.002–410	NR
Fluoranthene	0.0393–6830	0.0134
Perylene	0.0015–2730	0.31–0.8
Pyrene	0.0066–1650	0.1–0.9

From Brockhaus & Tomingas (1976); one permanent combustion stove burning anthracite and brown-coal briquets and vaporizer and atomizer oil burners; NR, not reported

— In the Netherlands, the estimated release in 1985 was < 1 t/year each for benzo[*k*]fluoranthene and indeno[1,2,3-*cd*]pyrene, < 10 t/year each for anthracene, fluoranthene, benz[*a*]anthracene, chrysene, benzo[*a*]-pyrene, and benzo[ghi]perylene, and 48–70 t/year each for naphthalene and phenanthrene, mainly resulting from wood heating (Slooff et al., 1989).
— The total PAH input, mainly from coal and wood heating, was about 63 t in Norway, 130 t in Sweden, and 720 t in the USA in 1985 (Bjørseth & Ramdahl, 1985).
— In Canada in 1990, the total PAH released due to residential heating, mainly wood burning, was about 500 t (Environment Canada, 1994).

(iii) *Open burning*

PAH may be released to the atmosphere during forest and agricultural fires, burning of accidentally spilled oil, disposal of road vehicles and especially automobile tyres, open burning of coal refuse and domestic and municipal waste, and open fires. The release of PAH into the atmosphere from the burning of wastes, including road vehicles, in the open is decreasing in industrialized countries due to comprehensive regulations.

Laboratory experiments with pine needles gave the following specific PAH emission factors (per kg pine needle): 980–20 000 μg pyrene, 690–15 000 μg fluoranthene, 580–12 000 μg anthracene plus phenanthrene,

540–29 000 µg chrysene plus benz[*a*]anthracene, 420–6200 µg benzo[*ghi*]-perylene, 170–4300 µg indeno[1,2,3-*cd*]pyrene, 140–8800 µg benzo[*c*]-phenanthrene, 130–13 000 µg benzofluoranthenes (isomers not specified), 61–800 µg benzo[*e*]-pyrene, 38–3500 µg benzo[*a*]pyrene, and 24–2100 µg perylene, depending on the amount of needles, area, and type of fire. Fires moving with the wind and low fuel loading resulted in significantly smaller amounts of PAH than fires moving against the wind and high fuel loading (McMahon & Tsoukalas, 1978). The emission factor for acenaphthene was 230–1000 µg/kg dry straw (Ramdahl & Møller, 1983) and 660 µg/kg dry wood (Alfheim et al., 1984).

In model experiments with crude oil spilled on water, numerous PAH were found, including acenaphthene, acenaphthylene, phenanthrene, anthracene, 1-methylphenanthrene, fluoranthene, pyrene, fluorene, benzo[*a*]fluorene, benzo[*b*]fluorene, benz[*a*]anthracene, chrysene plus triphenylene, benzo[*b*]-fluoranthene, benzo[*ghi*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, perylene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]perylene, and coronene, at concentrations of ≤ 1000 mg/kg individual substance in both the soot and the burn residue (Benner et al., 1990). Even though the open burning of oil spilled on water results in a lower PAH content than in crude oil (see Table 8), this source may be of local importance, e.g. near tanker accidents.

Between the early and the mid-1970s, the total release of PAH (including nitrogen-containing analogues and quinone degradation products) into the atmosphere in the USA due to open burning was estimated to be about 4000 t/year (Agency for Toxic Substances and Disease Registry, 1990). The total PAH input from forest and agricultural fires in 1985 was estimated to be 13 t in Norway, 1.3 t in Sweden, and 1000 t in the USA, and that from open fires to be 0.4 t in Norway and 100 t in the USA (Bjørseth & Ramdahl, 1985). The release of all PAH into the atmosphere from the burning of scrap electrical cable in 1988 was about 17 t (Slooff et al., 1989). In Canada in 1990, the total PAH emissions from agricultural burning and open-air fires were estimated to be about 360 t and those from forest fires to be about 2000 t (Environment Canada, 1994).

(iv) *Other diffuse sources*

The total PAH released into the atmosphere in the Netherlands from roofing tar and asphalt in 1988 was estimated at 0.5 t/year (Slooff et al., 1989).

(c) Emissions to the hydrosphere

(i) *Motor vehicle traffic*

The main source of PAH in the aqueous environment as a result of motor vehicle traffic is highway run-off, which contains asphalt and soot particles and

is washed by rainfall and storm water or snow into surface waters and soil (see also 3.2.7.2 (a) (i)). The available data are summarized in Table 23. Higher PAH concentrations were found in highway run-off in winter than in summer; this was attributed to the increased abrasion of the road surface due to use of steel-studded tyres in winter (Berglind, 1982).

It was estimated that an average of < 10 µg/km per vehicle per day of total PAH are transported via pavement runoff water. Most is transported to nearby surroundings as small particles of dust (see also section 3.2.7.2; Lygren et al., 1984). In contrast, storm water runoff near a US highway was of considerable importance for adjacent water bodies. In the test area, over 50% of the total PAH input into a nearby river came from highway runoff. The runoff loading factor was given as 24 mg/km per vehicle (Hoffman et al., 1985).

(ii) *Sewage treatment*

The concentrations of PAH in final effluents from municipal sewage treatment facilities are generally in the low microgram per litre range and are almost always < 0.1 µg/litre (Nicholls et al., 1979; Young et al., 1983; van Luin & van Starkenburg, 1984; Kröber & Häckl, 1989). Maximum values of 29 µg/litre naphthalene and 7 µg/litre acenaphthene were detected in one US sewage treatment plant, and 8 µg/litre benzo[*a*]pyrene were found in one German plant (Young et al., 1983; Kröber & Häckl, 1989), but no explanation was given for these unusually high concentrations. It was concluded that final effluents contain PAH at a background level (van Luin & van Starkenburg, 1984).

Naphthalene was found at a concentration of 9.3 kg/year in the final effluent from one US municipal sewage plant (Hoffman et al., 1984). The annual emissions of naphthalene, anthracene, phenanthrene, fluoranthene, benz[*a*]anthracene, chrysene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, and indeno[1,2,3-*cd*]pyrene from Dutch sewage treatment plants into surface waters were estimated to be about 0.6 t. The amount of these PAH transported into the Netherlands from other European countries via the Rhine, Meuse, and Scheldt rivers was estimated to be 65 t/year (year and database not given). The main compounds were fluoranthene (18 t/year) and naphthalene (15 t/year) (Slooff et al., 1989).

(iii) *Other sources*

PAH have been found in wastewaters from power stations, from garages with car-wash devices, and from a German car-wash storage tank at the following concentrations: fluoranthene, 1.3–7.7 µg/litre; pyrene, 3.5–28 µg/litre; benz[*a*]anthracene, 0.49–1.9 µg/litre; chrysene, 1.2–6.0 µg/litre; benzo[*e*]pyrene, 4.7–16 µg/litre; benzo[*a*]pyrene, 0.40–8.8 µg/litre; benzo[*b*]fluoranthene, 1.2–3.6 µg/litre; and benzo[*k*]-

Table 23. Polycyclic aromatic hydrocarbon concentrations (µg/litre) in highway runoff

Compound	[1]	[2]	[3]	[4]	[5]
Acenaphthene	0.016/0.087	0.195/5.126	NR	NR	NR
Acenaphthylene	0.045	0.557/16.804	NR	NR	NR
Anthracene	0.042–0.214	0.486/8.917	0.379	0.165	0.246
Benzo[*j*+*k*]fluoranthene	0.089/0.277	NR	NR	NR	0.207
Benz[*a*]anthracene	0.031–0.139	0.341/0.863	0.677	0.228	NR
Benzo[*a*]fluorene	0.018–0.170	0.587	NR	0.179	0.396
Benzo[*a*]pyrene	0.061–0.120	0.537/1.255	0.602	0.250	NR
Benzo[*b*]fluoranthene	0.129/0.157	NR	NR	0.799	1.501
Benzo[*b*]fluorene	0.033/0.097	0.356/0.366	NR	NR	0.192
Benzo[*c*]phenanthrene	NR	0.250	NR	NR	NR
Benzo[*e*]pyrene	0.108/0.202	0.238/1.665	0.609	0.360	0.630
Benzofluoranthenes[a]	0.401/0.695	1.087/2.712	1.171	NR	NR
Benzo[*a*]perylene	0.100–0.299	NR	0.551	0.391	0.319
Chyrsene + triphenylene	0.194–0.433	1.472/2.752	1.147	0.665	1.070
Fluoranthene	0.321–1.573	4.065/15.322	2.665	1.820	3.143
Fluorene	0.0088–0.564	0.432/11.093	0.096	0.485	1.237
Indeno[1,2,3-*cd*]pyrene	0.061–0.154	0.344/0.666	NR	NR	NR
1-Methylphenanthrene	0.030–1.073	0.637/2.308		1.366	2.117
Naphthalene	NR	2.59	NR	0.123	0.195
Perylene	0.048	NR	NR	NR	NR
Phenanthrene	0.068–2.668	3.297/38.10	1.385	4.055	6.787
Pyrene	0.363–1.449	3.026/12.094	2.002	1.886	3.066

NR, not reported; /, single measurements; **[1]** Run-off samples from a Norwegian highway north of Oslo in summer and winter 1980–82 (Berglind, 1982); **[2]** Snow 20 and 50 m from the same highway in February 1981 (Berglind, 1982); **[3]** Snow from a frozen Norwegian lake 50 m from a highway with high traffic density in winter 1981–82 (Gjessing et al., 1984); **[4]** Snow from a Norwegian highway south of Oslo with concrete pavement, February 1972 (Lygren et al., 1984); **[5]** Snow from a Norwegian highway south of Oslo with asphalt pavement, February 1972 (Lygren et al., 1984)

When the water samples were filtered through solid sorbents, the results may be underestimates of the actual content (see section 2.4.1.4).

[a] Isomers not specified

fluoranthene, 0.51–0.72 µg/litre (Baumung et al., 1985). Wastewaters from power stations could be an important local source of PAH.

Numerous PAH were detected in leachate plumes from refuse landfills in western Germany and the USA (Grimmer et al., 1981b; Götz, 1984; Reinhard et al., 1984). Concentrations < 0.1 µg/litre were detected of benzo[*ghi*]-fluoranthene, benz[*a*]anthracene, benzo[*c*]phenanthrene, chrysene,

benzofluoranthenes (isomers not specified), benzo[*a*]pyrene, benzo[*e*]pyrene, perylene, anthanthrene, benzo[*ghi*]perylene, and indeno[1,2,3-*cd*]pyrene (Grimmer et al., 1981b). Naphthalene was found at a concentration > 100 µg/litre, and acenaphthene, fluorene, anthracene, phenanthrene, and pyrene were found at concentrations of 1–30 µg/litre (Götz, 1984; Reinhard et al., 1984). The importance of this source for groundwater pollution cannot be estimated from the available data.

(c) Emissions to the geosphere

(i) *Motor vehicle traffic*

PAH were deposited within 100 m of a highway at a concentration of 100–200 µg/km per vehicle per day in winter as small particles of dust resulting from the abrasion of asphalt by steel-studded tyres (Lygren et al., 1984). Studies of adsorption on various soil types showed that most PAH in highway runoff is retained on the soil surface (Gjessing et al., 1984).

(ii) *Open burning*

Phenanthrene, fluoranthene, triphenylene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene, and coronene were determined in the soil of burning sites in western Oregon, USA. Before burning, the PAH concentrations in the top 2 cm of the soil layer ranged from 0.8 ng/g dry weight for benzo[*a*]pyrene to 4.4 ng/g for fluoranthene and triphenylene. One week after burning, the concentrations ranged from 0.9 ng/g for benzo[*k*]fluoranthene to 19 ng/g for triphenylene. The finding that the PAH levels did not increase appreciably after burning indicates that the bulk of the PAH were retained within the litter rather than passing into the soil (Sullivan & Mix, 1983).

(iii) *Disposal of sewage sludge and fly ash from incineration*

When sewage sludge is applied to soils, adsorbed PAH are added to the geosphere. The PAH concentrations in municipal aerobic and anaerobic sewage sludge are given in Table 24.

In a detailed survey of the PAH concentrations in soil to which anaerobic sludges had been applied between 1942 and 1961 in the United Kingdom, the total PAH content increased to over 125 mg/kg up to 1948 but had decreased to about 29 mg/kg by 1961. The authors attributed the declining levels to a decrease in atmospheric PAH contamination from smoke emissions (Wild et al., 1990). No seasonal variation in the content or profile of PAH was detected in western Germany by Grimmer et al. (1980), but Süss (1980) found the highest PAH load in sewage sludge in January–April and the lowest in July and

Table 24. Polycyclic aromatic hydrocarbons concentrations (mg/kg dry weight) in municipal sewage sludge

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Acenaphthene	NR	NR	NR	NR	NR	NR	ND	NR
Anthracene	NR	NR	NR	0.89–44	NR	NR	ND–10.0	NR
Anthanthrene	0.00–2.10	0.03–1.8	NR	NR	NR	NR	NR	NR
Benz[*a*]anthracene	0.62–19.0	0.91–17.3	NR	NR	NR	NR	ND–2.1	NR
Benzo[*a*]fluorene	0.28–9.00	0.56–7.9	NR	NR	NR	NR	NR	NR
Benzo[*a*]pyrene	0.54–13.3	0.41–14.3	0.12–9.14	NR	NR	0.29–2.00	ND–0.64	NR
Benzo[*b*]fluoranthene	NR	NR	0.06–9.14	NR	< 1–1.3	0.29–1.80	ND–1.100	NR
Benzo[*e*]pyrene	0.53–12.4	0.48–12.3	NR	NR	NR	NR	NR	NR
Benzofluoranthenes[a]	1.07–23.7	1.02–24.8	NR	NR	NR	NR	NR	NR
Benzo[*ghi*]perylene	0.40–8.70	0.34–10.9	0.06–9.14	NR	NR	< 0.1–3.41	ND–1.21	NR
Benzo[*k*]fluoranthene	NR	NR	0.06–4.57	NR	NR	0.15–1.00	ND–0.500	NR
Chrysene	0.78–23.7	1.24–22.2	NR	0.25–13	NR	NR	NR	NR
Dibenz[*a,h*]anthracene	NR	NR	NR	13	NR	NR	ND–0.25	NR
Fluoranthene	0.61–51.6	4.10–28.2	0.34–11.45	0.35–7.1	< 1–10.4	0.54–7.67	0.216–5.14	5.2/5.6[b]
Fluorene	NR	NR	NR	NR	NR	NR	ND–2.9[c]	3.5/5.8
Indeno[1,2,3-*cd*]pyrene	0.30–7.40	0.28–9.4	0.06–6.68	NR	NR	0.24–2.08	ND–0.640	NR
Naphthalene	NR	NR	NR	0.9–70	NR	NR	NR	4.5/8.6

Table 24 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Perylene	0.14–6.40	0.09–3.1	NR	NR	NR	NR	NR	NR
Phenanthrene	NR	NR	NR	0.89–44	NR	NR	0.30–40	15.2/18.6[d]
Pyrene	0.90–47.2	3.20–25.3	NR	0.33–18N	R	NR	ND–7.6	NR

NR, not reported; /, single measurements; ND, not detected (limits of detection, 0.2–1 mg/kg); **[1]** Samples from 25 sewage treatment plants in western Germany 1976–78 (Grimmer et al., 1980); **[2]** Samples from three sewage treatment facilities in western Germany before 1979 (Süss, 1980); **[3]** Samples from 12 British sewage treatment works (McIntyre et al., 1981); **[4]** Samples from 20 US sewage treatment works (Naylor & Loehr, 1982); **[5]** Samples from six Dutch municipal sewage treatment plants (van Luin & van Starkenburg, 1984); **[6]** 31 sludge samples from different sewage treatment works in western Germany (Witte et al., 1988); **[7]** Anaerobic sludge samples from 13 sewage treatment plants in western Germany 1985–88 (Kröber & Häckl, 1989); **[8]** Anaerobic sludge samples from one Spanish sewage treatment facility in spring 1985 and autumn 1986 (Gonzalez-Villa et al., 1988).

[a] Isomers not specified
[b] With pyrene
[c] With acenaphthylene
[d] With anthracene

October. Human faeces seemed to contribute little to the PAH content of sewage sludge (Grimmer et al., 1980). The most important emission sources could not be identified, but McIntyre et al. (1981) concluded that the PAH content of sewage sludge originating from British treatment works with significant flows of industrial effluent was higher than that in works dealing with predominantly domestic effluents.

After application of compost over three years to an agricultural soil in Spain, no accumulation of PAH was observed (Gonzalez-Vila et al., 1988). It was shown, however, that the extent of accumulation is dependent on the duration, frequency, and concentration of application. After 10 years of sludge spreading, considerable quantities of PAH were detected in both a sandy loam and a clay soil (Dierexsens & Tarradellas, 1987). The annual addition of PAH to soil from sewage sludge in the Netherlands was estimated as follows: 0.1 t naphthalene, 0.1 t anthracene, 1.5 t phenanthrene, 2.3 t fluoranthene, 0.6 t benzo[*a*]anthracene, 0.6 t chrysene, 0.4 t benzo[*k*]fluoranthene, 0.6 t benz[*a*]pyrene, 0.6 t benzo[*ghi*]-perylene, and 0.6 t indeno[1,2,3-*cd*]pyrene (year and database not given; Slooff et al., 1989).

The annual contribution of PAH to landfill in the United Kingdom from fly ash from coal combustion (see also Table 16) exceeded that from municipal solid-waste incineration by a factor of about 10, with the exception of naphthalene, the level of which was about 20 000-fold higher in fly ash from coal combustion than in that from solid-waste incineration. The annual PAH loads from solid-waste incineration were about 0.01 kg naphthalene and 3.5 kg benzo[*ghi*]perylene, whereas those from coal combustion were about 15 kg each of anthracene, benzo[*k*]fluoranthene, and dibenz[*a*,*h*]anthracene and 1200 kg pyrene (Wild et al., 1992).

(iv) *Waste dumping*

Soil cores taken from a hazardous waste disposal site in Spain containing petroleum tar residues and lubricating oils as the major organic wastes contained 62 mg/kg 1-methylphenanthrene, 53 mg/kg naphthalene, 52 mg/kg benzo[*a*]fluorene, 30 mg/kg benzo[*ghi*]fluoranthene, 25 mg/kg benzo[*c*]-phenanthrene, 0.5–0.71 mg/kg acenaphthene, 0.2–48 mg/kg fluorene, 0.2–390 mg/kg phenanthrene, 0.110 mg/kg anthanthrene, 0.1–210 mg/kg pyrene, 0.1–200 mg/kg acenaphthylene, 0.1–140 mg/kg anthracene, 0.1–140 mg/kg benzo[*e*]pyrene, 0.1–145 mg/kg benzo[*a*]pyrene, 0.1–50 mg/kg benzo[*b*]fluorene, 0.08–130 mg/kg chrysene plus triphenylene, 0.08–90 mg/kg indeno[1,2,3-*cd*]pyrene, 0.06–130 mg/kg benz[*a*]anthracene, 0.05–290 mg/kg fluoranthene, 0.03–75 mg/kg benzo[*ghi*]perylene, 0.03–0.2 mg/kg perylene, and 0.01–0.4 mg/kg dibenz[*a*,*h*]anthracene (Navarro et al., 1991).

There can be appreciable movement of PAH into soil from waste dumping, especially of hazardous refuse. The dumping conditions are decisive for the

amount of PAH released. Annual emissions of PAH in the Netherlands in 1987 due to the spreading of contaminated composts onto soils were estimated to be 1 t benz[*a*]anthracene, 1 t chrysene, 1 t benzo[*k*]fluoranthene, 0.5 t benzo[*ghi*]-perylene, 0.5 t indeno[1,2,3-cd]pyrene, and 0.4 t benzo[*a*]pyrene (Slooff et al., 1989).

(d) Biosphere

Perch (*Perca fluviatilis*) were not significantly contaminated after an oil spill in Finland due to a tanker accident. The concentrations of acenaphthene, acenaphthylene, fluorene, phenanthrene, anthracene, 1-methylphenanthrene, fluoranthene, pyrene, benzo[*a*]fluorene, benzo[*b*]fluorene, chrysene, triphenylene, and benzofluoranthenes in both contaminated and control groups were between ≤ 0.1 and 0.2 µg/kg each in muscle and ≤ 0.1 and 16 µg/kg in bile. The investigators concluded that the fish with the highest load would probably not have survived and others had moved to less contaminated areas. Additionally, the cold climate caused clumping of the spilled oil, which then drifted to the coast (Lindström-Seppä et al., 1989; see also sections 4.1.5.1 and 5.1.7.1).

4. ENVIRONMENTAL TRANSPORT, DISTRIBUTION, AND TRANSFORMATION

Appraisal

The transport and distribution of polycyclic aromatic hydrocarbons (PAH) in the environment depend on their physicochemical properties of very low solubility in water and low vapour pressure, and high partition coefficients for n-octanol:water (log K_{ow}) and organic carbon:water (log K_{oc}). PAH are stable towards hydrolysis as they have no reactive groups. In the gaseous phase, PAH and particularly those of higher molecular mass, are mainly adsorbed to particulate matter and reach the hydrosphere and geosphere by dry and wet deposition. Little is volatilized from water phases owing to their low Henry's law constants. The log K_{oc} values indicate strong adsorption to the organic matter of soils, so that migration does not usually occur. The log K_{ow} values indicate high bioaccumulation.

Few experimental data are available on the biodegradation of PAH. In general, they are biodegradable under aerobic conditions, and the biodegradation rates decrease drastically with the number of aromatic rings. Under anaerobic conditions, biodegradation appears to be very slow.

The bioconcentration factors measured in the water phase vary widely according to the technique used. High values are seen for some algae, crustaceans, and molluscs, but those for fish are much lower owing to rapid biotransformation. The bioaccumulation factors for aquatic and terrestrial organisms in sediment and soil are generally very low, probably because of the strong adsorption of PAH onto the organic matter of soils and sediments, resulting in low bioavailability.

The photodegradation of PAH in air and water has been studied intensively. The most important degradation process in both media is indirect photolysis under the influence of radicals like OH, O_3 and NO_3. The measured degradation rate constants vary widely according to the technique used. Under laboratory conditions, the half-life of the reaction of PAH with airborne OH radicals is about one day. Adsorption of high-molecular-mass PAH onto carbonaceous particles in the environment has a stabilizing effect. Formation of nitro-PAH has been reported from two- to four-ring PAH in the vapour phase during photooxidation with NO_3. For some PAH, photodegradation in water seems to be more rapid than in air.

According to model calculations based on physicochemical and degradation parameters, PAH with four or more aromatic rings persist in the environment.

4.1 Transport and distribution between media

4.1.1 *Physicochemical parameters that determine environmental transport and distribution*

The transport and distribution of PAH in the environment are the result of the following physicochemical parameters:

- *Aqueous solubility*: PAH are hydrophobic compounds with very low solubility in water under environmental conditions: the maximum at room temperature is 32 mg/litre for naphthalene, and the minimum is 0.14 µg/litre for coronene (see Table 4).
- *Vapour pressure*: The vapour pressure of PAH under environmental conditions is very low: the maximum at room temperature is 10.4 Pa for naphthalene, and the calculated minimum is 3×10^{-12} Pa for dibenzo[*a*,*i*]pyrene (see Table 4).
- n-*Octanol*:water partition coefficient (log K_{ow}): The affinity of PAH to organic phases is much higher than that for water. The log K_{ow} values range from 3.4 for naphthalene to 7.3 for dibenzo[*a*,*i*]pyrene (see Table 4), indicating that the potential for bioaccumulation is high.
- *Organic carbon:water partition coefficient (log* K_{oc}): The sorption coefficients of PAH to the organic fraction of sediments and soils are summarized in Table 25. The high values indicate that PAH sorb strongly to these fractions. The wide variation in the results for individual compounds are due to the very long exposure necessary to reach steady-state or equilibrium conditions, which can lead to underestimation of sorption coefficients; furthermore, degradation in the overlying aqueous phase can lead to overestimates of the actual values.

4.1.2 *Distribution and transport in the gaseous phase*

PAH are emitted mainly to the atmosphere (see Section 3), where they can be both transported in the vapour phase and adsorbed onto particulate matter. The distribution between air and particulate matter under normal atmospheric conditions depends on the lipophilicity, vapour pressure, and aqueous solubility of the substance. Generally, PAH with few (two to four) aromatic rings occur in the vapour phase and are adsorbed, whereas PAH consisting of more aromatic rings exist mainly in the adsorbed state (Hoff & Chan, 1987; McVeety & Hites, 1988; Baker & Eisenreich, 1990). PAH are usually adsorbed onto particles like fly ash and soot that are emitted during combustion.

PAH are ubiquitous in the environment, probably because they are distributed for long distances without significant degradation (Lunde, 1976;

Table 25. Organic carbon normalized sorption coefficients (K_{oc}) of polycyclic aromatic hydrocarbons

Compound	log K_{oc}	Comments	Reference
Acenaphthene	5.38	Average on sediments	Kayal & Connell (1990)
	3.79	RP-HPLC on CIHAC	Szabo et al. (1990)
	3.59	RP-HPLC on PIHAC	Szabo et al. (1990)
Acenaphthylene	3.83	RP-HPLC on CIHAC	Szabo et al. (1990)
	3.75	RP-HPLC on PIHAC	Szabo et al. (1990)
Anthracene	4.42	Average sorption isotherms on sediment	Karickhoff et al. (1979)
	3.74	Suspended particulates	Herbes et al. (1980)
	4.20	Soil, shake flask, UV	Karickhoff (1981)
	3.95/4.73	Lake Erie with 9.6 mg C/litre	Landrum et al. (1984a)
	4.87/5.70	Huron river with 7.8 mg C/litre	Landrum et al. (1984a)
	4.20	Soil, shake flask, LSC	Nkedi-Kizza et al. (1985)
	4.93	Fluorescence, quenching interaction with humic acid	Gauthier et al. (1986)
	4.38	HPLC	Hodson & Williams (1988)
	5.76	Average on sediments	Kayal & Connell (1990)
	4.41	RP-HPLC	Pussemier et al. (1990)
	4.53	RP-HPLC on CIHAC	Szabo et al. (1990
	4.42	RP-HPLC on PIHAC	Szabo et al. (1990)
Benz[*a*]anthracene	4.52	Suspended particles	Herbes et al. (1980)
	6.30	Average on sediments	Kayal & Connell (1990)
	7.30	Specified particulate	Broman et al. (1990)

Table 25 (contd)

Compound	log K_{oc}	Comments	Reference
Benzo[*a*]pyrene	6.66	LSC	Eadie et al. (1990)
	6.26	Average on sediments	Kayal & Connell (1990)
	8.3	Specified particulate	Broman et al. (1990)
	4.0	Predicted to be dissolved	Broman et al. (1990)
Benzo[*e*]pyrene	7.20	Specified particulate	Broman et al. (1990)
	4.00	Predicted to be dissolved	Broman et al. (1990)
Benzo[*k*]fluoranthene	5.99	Average on sediments	Kayal & Connell (1990)
	7.00	Specified particulate	Broman et al. (1990)
	4.00	Predicted to be dissolved	Broman et al. (1990)
Chrysene	6.27	Average on sediments	Kayal & Connell (1990)
	6.90	Specified particulate	Broman et al. (1990)
	4.0	Predicted to be dissolved	Broman et al. (1990)
Coronene	7.80	Specified particulate	Broman et al. (1990)
	5.0	Predicted to be dissolved	Broman et al. (1990)
Dibenz[*a,h*]anthracene	6.31	Average of 14 soil or sediment samples,shake flask,LSC	Means et al. (1980)
Fluoranthene	6.38	Average on sediments	Kayal & Connell (1990)
	4.74	RP-HPLC on CIHAC	Szabo et al. (1990)
	4.62	RP-HPLC on PIHAC	Szabo et al. (1990)
	6.30	Specified particulate	Broman et al. (1990)
	4.0	Predicted to be dissolved	Broman et al. (1990)

Table 25 (contd)

Compound	log K_{oc}	Comments	Reference
Fluorene	5.47	Average on sediments	Kayal & Connell (1990)
	3.76	RP-HPLC	Pussemier et al. (1990)
	4.15	RP-HPLC on CIHAC	Szabo et al. (1990)
	4.21	RP-HPLC on PIHAC	Szabo et al. (1990)
Naphthalene	3.11	Average sorption isotherms on sediments	Karickhoff et al. (1979)
	2.38	Suspended particulates	Herbes et al. (1980)
	2.94		Karickhoff (1981)
	3.0		McCarthy & Jimenez (1985); McCarthy et al. (1985)
	2.73-3.91	Aquifer materials	Stauffer et al. (1989)
	3.15/2.76		Podoll et al. (1989)
	5.00	Average on sediments	Kayal & Connell (1990)
	2.66	Average on sediments	Kishi et al. (1990)
	3.11	Soil, RP-HPLC	Szabo et al. (1990)
	3.29	Sandy surface soil	Wood et al. (1990)
Phenanthrene	4.36	Average sorption isotherms on sediments	Karickhoff et al. (1979)
	4.28		Hodson & Williams (1988)
	6.12	Average on sediments	Kayal & Connell (1990)
	4.22	RP-HPLC on CIHAC	Szabo et al. (1990)
	4.28	RP-HPLC on PIHAC	Szabo et al. (1990)
	4.42	Sandy surface soil	Wood et al. (1990)

Table 25 (contd)

Compound	log K_{oc}	Comments	Reference
Pyrene	4.92	Average isotherms on sediments	Karickhoff et al. (1979)
	4.90	Sediment, shake flask, sorption isotherm	Karickhoff et al. (1979)
	4.81	Average of soil and sediment Shake flask, LSC, sorption isotherms	Means et al. (1979)
	4.80	Average of 12 soils and sediments Shake flask, LSC, sorption isotherms	Means et al. (1980)
	4.78	Soil and sediment; calculated K_{ow}	Means et al. (1980)
	4.83	Sorption isotherms	Karickhoff (1981)
	3.11/3.46	Sediment suspensions	Karickhoff & Morris (1985)
	4.80/5.13		Hodson & Williams (1988)
	5.65	LSC	Eadie et al. (1990)
	5.29	Soil	Jury et al. (1990)
	6.51	Average on sediments	Kayal & Connell (1990)
	4.83	RP-HPLC	Pussemier et al. (1990)
	4.82	RP-HPLC on CIHAC	Szabo et al. (1990)
	4.77	RP-HPLC on PIHAC	Szabo et al. (1990)
	6.50	Specified particulate	Broman et al. (1990)
	4.0	Predicted particulate	Broman et al. (1990)
Triphenylene	6.90	Specified particulate	Broman et al. (1990)
	4.00	Predicted to be dissolved	Broman et al. (1990)

RP-HPLC, reversed-phase high-performance liquid chromatography; CIHAC, chemical-induced humic-acid column; PIHAC, physical-induced humic-acid column; UV, ultraviolet; C, carbon; LSC, liquid scintillation chromatography

De Wiest, 1978; Bjørseth & Sortland Olufsen, 1983; McVeety & Hites, 1988), e.g. from the United Kingdom and the European continent to Norway and Sweden during winter (Bjørseth & Lunde, 1979). Washout ratios calculated from measurements in rain and snow in the area of northern Lake Superior, during one year showed that airborne PAH adsorbed onto particulate matter result in effective wet deposition, while gaseous PAH are removed to only a minor degree (McVeety & Hites, 1988).

4.1.3 *Volatilization*

Henry's law constant gives a rough estimate of the equilibrium distribution ratio of concentrations in air and water but cannot predict the rate at which chemicals are transported between water and air. The constants for PAH are very low, ranging from 49 Pa·m³/mol for naphthalene to 0.000449 Pa·m³/mol for dibenzo[*a*,*i*]pyrene (see Table 4). The rates of removal and volatilization of PAH (Table 26) are strongly dependent on environmental conditions such as the depth and flow rate of water and wind velocity. Although PAH are released into the environment mainly in air, considerably higher concentrations are found in aqueous samples because of the low vapour pressure and Henry's law constants of PAH.

The volatilization half-life for naphthalene from a 22.5-m water body was found experimentally to be 6.3 h, whereas the calculated value was 2.1 h (Klöpffer et al., 1982). Calculations based on a measured air:water partition coefficient for river water 1 m deep with a water velocity of 0.5 m/s and a wind velocity of 1 m/s gave a volatilization half-life of 16 h for naphthalene (Southworth, 1979). The value calculated for evaporative loss of naphthalene from a 1-m water layer at 25 °C was of the same order of magnitude (Mackay & Leinonen, 1975). Naphthalene was volatilized from soil at a rate of 30% after 48 h, with neglible loss of PAH with three or more rings (Park et al., 1990).

4.1.4 *Adsorption onto soils and sediments*

PAH are adsorbed strongly to the organic fraction of soils and sediments (see section 4.1.1 and Table 25). Some PAH may be degraded biologically in the aerobic soil layer, but this process is slow, because sorption to the organic carbon fraction of the soil reduces the bioavailability. For the same reason, leaching of PAH from the soil surface layer to groundwater is assumed to be negligible, although detectable concentrations have been reported in groundwater (see section 5.1.2.2).

4.1.5 *Bioaccumulation*

The ability of a substance to bioconcentrate in organisms in the aqueous phase is expressed as the bioconcentration factor. For substances like PAH,

Table 26. Rates of volatilization of polycyclic aromatic hydrocarbons

Compound	Rate constant	Half-life (h)[a]	Comments	Reference
Anthracene			Removal rate constants (estimated) from water column	Southworth (1977)
			At 25 °C in midsummer sunlight:	
	0.002 h^{-1}	347	– in deep, slow, somewhat turbid water	
	0.001 h^{-1}	693	– in deep, slow, muddy water	
	0.002 h^{-1}	347	– in deep, slow, clear water	
	0.042 h^{-1}	17	– in shallow, fast, clear water	
	0.179 h^{-1}	3.9	– in very shallow, fast, clear water	
		62	Calculated half-life for a river 1m deep with water velocity of 0.5 m/s and wind velocity of 1 m/s	Southworth (1979)
Benz[*a*]anthracene		500	Calculated half-life for a river 1 m deep with water velocity of 0.5 m/s and wind velocity of 1 m/s	Southworth (1979)
Benzo[*a*]pyrene		1550	Calculated half-life for a river 1 m deep with water velocity of 0.5 m/s and wind velocity of 1 m/s	Southworth (1979)
	$<1 \times 10^{-5}$ s^{-1}	> 19	Sublimation rate constant from glass surface at 24 °C at an airflow of 3 litre/min	Cope & Kalkwarf (1987)
Naphthalene	1.675×10^{-9} $mol.cm^{-2}h^{-1}$		Rate of evaporation estimated at 20 °C and air flow of 50 litre/h	Gückel et al. (1973)
		7.15	Calculated half-life from 1 m depth of water	Mackay & Leinonen (1975)

Table 26 (contd)

Compound	Rate constant	Half-life (h)[a]	Comments	Reference
Naphthalene (contd)		16	Half-life for surface waters	Southworth (1979)
		200	In a lake, considering current velocity and wind speed in combination with typical re-aeration rates	
Perylene	$<1 \times 10^{-5}$ s^{-1}	> 19	Sublimation rate constant from glass surface at 24 °C at an air flow of 3 litre/min	Cope & Kalkwarf (1987)
Pyrene	1.1×10^{-4} s^{-1}	1.8	Sublimation rate constant as loss from glass surface at 24 °C at an air flow of 3 litre/min	Cope & Kalkwarf (1987)

For comparison of results for which only rate constants are reported, half-lives have been estimated from the equation:

$$t_{1/2} = \frac{\ln 2}{k}$$

where $t_{1/2}$ is the half-life and k is the rate constant. The calculated values are reported in italics.

with high *n*-octanol:water partition coefficients, long exposures are necessary to achieve equilibrium conditions, so that results obtained under non-equilibrium conditions can result in underestimates of the bioconcentration factor. Bioaccumulation may also vary with the metabolic capacity of the organism (see section 4.2.1.2).

Bioconcentration can also be calculated as the ratio between the rates of uptake (k_1) and depuration (k_2). This method has the advantage that relatively short exposures can be used. It is therefore preferred for PAH, as constant concentrations of compounds like benzo[*a*]pyrene are very difficult to maintain over a long period.

4.1.5.1 Aquatic organisms

Aquatic organisms may accumulate PAH from water, sediments, and their food. In general, PAH dissolved in pore water are accumulated from sediment (McElroy & Sisson, 1989), and digestion of sediment may play an important role in the uptake of PAH by some species. Although organisms can accumulate PAH from food, the relative importance of uptake from food and water is not clear (Farrington, 1991).

The bioconcentration factors of PAH in different species are shown in Table 27; this is not a comprehensive presentation of all of the available data but provides examples of the accumulation of some PAH in different groups of organisms. Species that metabolize PAH to little or no extent, like algae, oligochaetes, molluscs, and the more primitive invertebrates (protozoans, porifers, and cnidaria), accumulate high concentrations of PAH, as would be expected from their log K_{ow} values, whereas organisms that metabolize PAH to a great extent, like fish and higher invertebrates such as arthropods, echinoderms, and annelids, accumulate little or no PAH (James, 1989). Remarkably high bioconcentration factors have been measured for phenanthrene, anthracene, pyrene, benzo[*a*]anthracene, and benzo[*a*]pyrene in the amphipod *Pontoporeia hoyi*, which has a 20-50% lipid content by wet weight and no capacity to biotransform PAH (Landrum, 1988).

The ratio of the concentration of an individual PAH in a bottom-dwelling organism and in the sediment, the bioaccumulation factor, is usually < 1 when expressed as wet weight. In a coastal area, the bioaccumulation factors for 16 PAH in polychaete species varied from 4.9 to 21.8 on a dry-weight basis (Bayona et al., 1991). Measurements of the concentrations of PAH in *P. hoyi* and in the sediment at three sites with different organic carbon contents gave bioaccumulation factors close to 1 on a wet-weight basis, corrected for the 64-mm sieved fraction (Eadie et al., 1982). The lipid- and organic carbon-based bioaccumulation factors in clams (*Macoma baltica*) for naphthalene and chrysene added to sediment were 0.78 and 0.16, respectively (Foster et al., 1987). In a study in which clams were exposed for 28 days to six sediments

Table 27. Measured bioconcentration factors of polycyclic aromatic hydrocarbons in aquatic organisms

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/ depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Acenaphthene							
Fish							
Lepomis macrochirus	^{14}C	S	8.94	28 d	387	Equi	Barrows et al. (1980)
Anthracene							
Algae							
Chlorella fusca	HPLC	S	50	1 d	7 770[a]	NS	Geyer et al. (1984)
Crustaceans							
Daphnia magna	^{14}C, TLC	S	35	1 d	511	k_1/k_2	McCarthy et al. (1985)
Daphnia magna	HPLC	S	15	1 d	970	NS	Newsted & Giesy (1987)
Daphnia magna	HPLC	S	5.58	24 h	2 699	NS	Oris et al. (1990)
Daphnia pulex	Spect	S	6	24 h	917	k_1/k_2	Southworth et al. (1978)
Hyalella azteca	^{14}C	F	0.0082	8 h/7 h	2 089	k_1/k_2	Landrum & Scavia (1983)
	^{14}C, TLC				1 800	k_1/k_2	
	^{14}C	F	0.0066	8 h/7 h	10 985	k_1/k_2	
	^{14}C, TLC				9 096	k_1/k_2	

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Anthracene (contd)							
Pontoporeia hoyi	^{14}C, TLC	F	4–17	8 h/7 d	16 857	k_1/k_2	Landrum (1982)
Pontoporeia hoyi	^{14}C, TLC	F	4.6–16.9	6 h/14 d	39 727	k_1/k_2	Landrum (1988)
Oligochaetes							
Stylodrilus heringianus	^{14}C, TLC	F	< 6	6 h/8 d	5 051	k_1/k_2	Frank et al. (1986)
Fish							
Lepomis macrochirus	^{14}C	S	0.7	4 h/60 h	900	k_1/k_2	Spacie et al. (1983)
	^{14}C, TLC				675	k_1/k_2	
Leuciscus idus melanotus	^{14}C	S	50	3 d	910	NS	Freitag et al. (1985)
Oncorhynchus mykiss	^{14}C, HPLC	R	12	18 h	190	NS	Linder & Bergman (1984)
	^{14}C, HPLC	R	12	18 h	270	NS	
Oncorhynchus mykiss	^{14}C, HPLC	R	50	72 h/144 h	9 000	k_1/k_2	Linder et al. (1985)
					9 200		
Pimephales promelas	HPLC	S	6.61	24 h	1 016	NS	Oris et al. (1990)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/ depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Benz[*a*]anthracene							
Algae							
Chlorella fusca	^{14}C	S	50	1 d	3 180	NS	Freitag et al. (1985)
Crustaceans							
Daphnia magna	^{14}C, TLC	S	0.8	1 d	2 920	k_1/k_2	McCarthy et al. (1985)
Daphnia pulex	Spect	S	6	1 d	10 109	k_1/k_2	Southworth et al. (1978)
Daphnia pulex	HPLC	S	1.8	1 d	10 226	NS	Newsted & Giesy (1987)
Pontoporeia hoyi	^{14}C, TLC	F	0.62–1.11	6 h/14 d	63 000	k_1/k_2	Landrum (1988)
Fish							
Leuciscus idus melanotus	^{14}C	S	50	3 d	350	NS	Freitag et al. (1985)
Benzo[*a*]fluorene							
Crustaceans							
Daphnia magna	HPLC	S	4.8	1 d	3 668	NS	Newsted & Giesy (1987)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Benzo[*b*]fluorene							
Crustaceans							
Daphnia magna	HPLC	S	2.2	1 d	7 725	NS	Newsted & Giesy (1987)
Benzo[*a*]pyrene							
Algae							
Periphyton	^{14}C	F	1	1 d	9 000	NS	Leversee et al. (1981)
Crustaceans							
Daphnia magna	^{14}C	S/F	1	6 h	2 440	k_1/k_2	Leversee et al. (1981)
Daphnia magna	^{14}C				3 050	NS	Leversee et al. (1981)
	^{14}C, HPLC				2 837	k_1/k_2	
Daphnia magna	^{14}C, TLC	S	0.63	1 d	5 770	k_1/k_2	McCarthy et al. (1985)
Daphnia magna	HPLC	S	1.5	1 d	12 761	NS	Newsted & Giesy (1987)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Benzo[*a*]pyrene (contd)							
Daphnia pulex	^{14}C	S	1.20	24 h	458	NS	Trucco et al.
	^{14}C	S	0.47	24 h	745	NS	(1983)
	^{14}C	S	5.42	24 h	803	NS	
	^{14}C	S	3.21	24 h	1 106	NS	
	^{14}C	S	2.20	24 h	1 259	NS	
	^{14}C	S	1.50	24 h	2 720	NS	
Pontoporeia hoyi	^{14}C, TLC	S	0.002–2.6	6 h/14 d	73 000	k_1/k_2	Landrum (1988)
Oligochaetes							
Stylodrilus heringianus	^{14}C, TLC	F	< 0.03	6 h/8 d	7 048	k_1/k_2	Frank et al. (1986)
Molluscs							
Mysis relicta	^{14}C	F	–	6 h/10–26d	8 297	k_1/k_2	Evans & Landrum (1989)
Ostrea edulis	^{14}C, GLC	S	65.7	3 d	58	NS	Riley et al.
Ostrea edulis	^{14}C, GLC	S	65.7	3 d	59	NS	(1981)
Ostrea edulis	^{14}C, GLC	S	65.7	3 d	62	NS	
Physa sp.	^{14}C, GLC	S	2.5	3 d	2 177	NS	Lu et al. (1977)
Rangia cuneata	^{14}C	S	30.5	24 h	236	NS	Neff & Anderson
	^{14}C	S	30.5	24 h	187	NS	(1975)

Table 27 (contd)

Species	Analysis	Test	Concen-system in water (µg/litre)	Duration of tration or uptake/ depuration period	Biocon-exposure factor (in wet weight)	Type centration	Reference
Benzo[*a*]pyrene (contd)							
Insects							
Chironomus	^{14}C	S	1	8 h/48 h	970	k_1/k_2	Leversee et al.
riparius	^{14}C				600	NS	(1981)
	^{14}C, HPLC				166	NS	
Culex pipiens quinquefasciatus	^{14}C, GLC	S	2.5	3 d	37	NS	Lu et al. (1977)
Hexagenia limbata	^{14}C, TLC	F	-	6 h/14 d	5 870	k_1/k_2	Landrum & Poore (1988)
Fish							
Lepomis macrochirus	^{14}C-extraction	F	1	2 d/4 d	3 208	k_1/k_2	Jimenez et al. (1987)
Lepomis	^{14}C	S/F	1	4 h/4 h	4 700	k_1/k_2	Leversee et al.
macrochirus	^{14}C			4 h	120	NS	(1981)
	^{14}C, HPLC			4 h	12.5	NS	
Lepomis	^{14}C	S	1	4 h/20 h	4 900	k_1/k_2	Spacie et al.
macrochirus	^{14}C, TLC				490	k_1/k_2	(1983)
Lepomis macrochirus	^{14}C, TLC	S	0.5	5 h/100 h	2 657	k_1/k_2	McCarthy & Jimenez (1985)
Leuresthes tenuis	Spect	S	2	15 d	241	Equi	Winkler et al. (1983)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/ depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Benzo[*a*]pyrene (contd)							
Oncorhynchus mykiss	GC-HPLC	F	0.4	10 d	920	NS	Gerhart & Carlson (1978)
Salmo salar	^{14}C	S	1	48 h/96 h	2 310	k_1/k_2	Johnsen et al. (1989)
Benzo[*e*]pyrene							
Crustaceans							
Daphnia magna	HPLC	S	0.7	1 d	25 200	NS	Newsted & Giesy (1987)
Benzo[*ghi*]perylene							
Crustaceans							
Daphnia magnia	HPLC	S	0.2	1 d	28 288	NS	Newsted & Giesy (1987)
Benzo[*k*]fluoranthene							
Crustaceans							
Daphnia magna	HPLC	S	1.4	1 d	13 225	NS	Newsted & Giesy (1987)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Chrysene							
Crustaceans							
Daphnia magna	^{14}C	S	48	48 h/40 h	5 500	NS	Eastmond et al. (1984)
Daphnia magna	HPLC	S	0.7	1 d	6 088	NS	Newsted & Giesy (1987)
Dibenz[*a*,*h*]anthracene							
Algae							
Chlorella fusca	^{14}C	S	50	1 d	2 398	NS	Freitag et al. (1985)
Crustaceans							
Daphnia magnia	HPLC	S	0.4	1 d	50 119	NS	Newsted & Giesy (1987)
Fish							
Leuciscus idus melanotus	^{14}C	S	50	3 d	10	NS	Freitag et al. (1985)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/ depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Fluoranthene							
Crustaceans							
Crangon septemspinosa	HPLC	F	2.4	4 d/14 d	180	k_1/k_2	McLeese & Burridge (1987)
Daphnia magna	HPLC	S	9	1 d	1 742	NS	Newsted & Giesy (1987)
Molluscs							
Mya arenaria	HPLC	F	2.4	4 d/14 d	4 120	k_1/k_2	McLeese & Burridge (1987)
Mytilus edulis	HPLC	F	2.4	4 d/14 d	5 920	k_1/k_2	McLeese & Burridge (1987)
Polychaetes							
Neins virens	HPLC	F	2.4	4 d/14 d	720	k_1/k_2	McLeese & Burridge (1987)
Fish							
Oncorhynchus mykiss	GC-HPLC	F	3.31	21 d	378	Equi	Gerhart & Carlson (1978)
Fluorene							
Crustaceans							
Daphnia magna	HPLC	S	17	1 d	506	NS	Newsted & Giesy (1987)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Fluorene (contd)							
Fish							
Lepomis	–	IF	20, 37	30 d	1 800	Equi	Finger et al.
macrochirus	–	IF	86	30 d	700	Equi	(1985)
	–	IF	175, 353	30 d	200	Equi	
Naphthalene							
Algae							
Selenastrum capricornutum	GC	S	2,000	1 d	18 000[b]	NS	Casserly et al. (1983)
Chlorella fusca	^{14}C	S	50	1 d	130[a]	NS (1984)	Geyer et al.
Insects							
Somatochlora	Spect	S	10	48 h	1 548	NS	Correa & Coler
cingulata	Spect	S	100	48 h	178	NS	(1983)
Crustaceans							
Daphnia magna	^{14}C, HPLC	S	1 000	1 d	19.3	k_1/k_2	McCarthy et al. (1985)
Daphnia magna	^{14}C	S	1 800	48 h/40 h	50	NS	Eastmond et al. (1984)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/ depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Naphthalene (contd)							
Daphnia pulex	Spect	S	1 000	1 d	131	k_1/k_2	Southworth et al. (1978)
Daphnia pulex	^{14}C	S	2 292	4 h	677	NS	Trucco et al. (1983)
	^{14}C	S	0.45	24 h	10 844	NS	
	^{14}C	S	2.742	4 h	2 337	NS	
Fish							
Fundulus heteroclitus	^{14}C	S	20	4 h	2.2	NS	DiMichele & Taylor (1978)
Lepomis macrochirus	^{14}C, HPLC	S	1 000	24 h/36 h	310	k_1/k_2	McCarthy & Jimenez (1985)
	^{14}C, HPLC	S	100	24 h/36 h	320	k_1/k_2	
Oncorhynchus mykiss	^{14}C	S	23	8 h/24 h	253	k_1/k_2	Melancon & Lech (1978)
Perylene							
Algae							
Chlorella fusca	^{14}C	S	50	1 d	2 010	NS	Freitag et al. (1985)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Perylene (contd)							
Crustaceans							
Crangon septemspinosa	HPLC	F	0.4	4 d/14 d	175	k_1/k_2	McLeese & Burridge (1987)
Daphnia magnia	HPLC	S	0.6	1 d	7 190	NS	Newsted & Giesy (1987)
Molluscs							
Mya arenaria	HPLC	F	0.4	4 d/14 d	100 000	k_1/k_2	McLeese & Burridge (1987)
Mytilus edulis	HPLC	F	0.4	4 d/14 d	105 000	k_1/k_2	McLeese & Burridge (1987)
Polychaetes							
Neiris virens	HPLC	F	0.4	4 d/14 d	180	k_1/k_2	McLeese & Burridge (1987)
Fish							
Leuciscus idus melanotus	^{14}C	S	50	3 d	< 10	NS	Freitag et al. (1985)
Phenanthrene							
Bacteria							
Mixed	Spect	S	30-300	2 h	6 300[c]	NS	Steen & Karickhoff (1981)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Phenanthrene (contd)							
Algae							
Selenastrum capricornutum	GC	S	1000	1 d	36 970[b]	NS	Casserly et al. (1983)
Chlorella fusca	^{14}C	S	50	1 d	1 760[a]	NS	Geyer et al. (1984)
Insects							
Hexagenia limbata	^{14}C	F	-	6 h/14 d	1640	k_1/k_2	Landrum & Poore (1988)
Crustaceans							
Crangon septemspinosa	HPLC	F	4.3	4 d/14 d	210	k_1/k_2	McLeese & Burridge (1987)
Daphnia magna	HPLC	S	40.1	1 d	323	NS	Newsted & Giesy (1987)
Daphnia magna	^{14}C	S	60	48 h/40 h	600	NS	Eastmond et al. (1984)
Daphnia pulex	^{14}C	S	6.01	24 h	1 165	NS	Trucco et al. (1983)
	^{14}C	S	3.10	24 h	1 032	NS	
	^{14}C	S	3.45	24 h	1 424	NS	

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/ depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Phenanthrene (contd)							
Daphnia pulex	Spect	S	30	1 d	325	k_1/k_2	Southworth et al. (1978)
Pontoporeia hoyi	^{14}C-TLC	F	0.7–7.1	6 h/14 d	28 145	k_1/k_2	Landrum (1988)
Oligochaetes							
Stylodrilus heringianus	^{14}C-TLC	F	< 200	6 h/8 d	5 055	k_1/k_2	Frank et al. (1986)
Molluscs							
Mya arenaria	HPLC	F	4.3	4 d/14 d	1 280	k_1/k_2	McLeese & Burridge (1987)
Mytilus edulis	HPLC	F	4.3	4 d/14 d	1 240	k_1/k_2	McLeese & Burridge (1987)
Polychaetes							
Neiris virens	HPLC	F	4.3	4 d/14 d	500	k_1/k_2	McLeese & Burridge (1987)
Pyrene							
Bacteria							
Mixed	Spect	S	1-20	2 h	24 600[c]	NS	Steen & Karickhoff (1981)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/ depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Pyrene (contd)							
Algae							
Selenastrum capricornutum	GC	S	500	1 d	55 800[b]	NS	Casserly et al. (1983)
Crustaceans							
Crangon septemspinosa	HPLC	F	1.7	4 d/14 d	225	k_1/k_2	McLeese & Burridge (1987)
Daphnia magna	HPLC	S	5.7	24 h	2 702	NS	Newsted & Giesy (1987)
Daphnia pulex	Spect	S	50	24 h	2 702	k_1/k_2	Southworth et al. (1978)
Pontoporeia hoyi	^{14}C-TLC	F	0.002–0.011	6 h/14 d	16 600	k_1/k_2	Landrum (1988)
Molluscs							
Mya arenaria	HPLC	F	1.7	4 d/14 d	6 430	k_1/k_2	McLeese & Burridge (1987)
Mytilus edulis	HPLC	F	1.7	4 d/14 d	4 430	k_1/k_2	McLeese & Burridge (1987)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Pyrene (contd)							
Oligochaetes							
Stylodrilus heringianus	^{14}C-TLC	F	< 26.4	6 h/8 d	6 588	k_1/k_2	Frank et al. (1986)
Polychaetes							
Neiris virens	HPLC	F	1.7	4 d/14 d	700	k_1/k_2	McLeese & Burridge (1987)
Fish							
Oncorhynchus mykiss	GC-HPLC	F	3.89	21 d	72.2	Equi	Gerhart & Carlson (1978)
Triphenylene							
Crustaceans							
Crangon septemspinosa	HPLC	F	0.5	4 d/14 d	270	k_1/k_2	McLeese & Burridge (1987)
Daphnia magna	HPLC	S	1.7	1 d	9 066	NS	Newsted & Giesy (1987)
Molluscs							
Mya arenaria	HPLC	F	0.5	4 d/14 d	5 540	k_1/k_2	McLeese & Burridge (1987)

Table 27 (contd)

Species	Analysis	Test system	Concentration in water (µg/litre)	Duration of exposure or uptake/depuration period	Bioconcentration factor (in wet weight)	Type	Reference
Triphenylene (contd)							
Mytilus edulis	HPLC	F	0.5	4 d/14 d	11 390	k_1/k_2	McLeese & Burridge (1987)
Polychaetes							
Neiris virens	HPLC	F	0.5	4 d/14 d	2 560	k_1/k_2	McLeese & Burridge (1987)

^{14}C, measurement of radioactivity in a liquid scintillation counter; as parent compounds cannot be differentiated from metabolites with this method, additional extraction is usually performed.
S, static exposure system; Equi, at equilibrium C_{org}/C_w; HPLC, high-performance liquid chromatography; NS, not steady-state C_{org}/C_w; TLC, thin-layer chromatography; k_1/k_2, kinetics: uptake rate/depuration rate; Spect, spectroscopy; F, flow-through system; R, static renewal system; GLC, gas-liquid chromatography; GC, gas chromatography; IF, intermittent flow system

[a] Based on dry weight (5 x wet weight)

[b] Based on total suspended solids

[c] Based on dry weight

contaminated with different concentrations of PAH (and other organic pollutants) and with an organic carbon content of 0.86–7.4%, the bioaccumulation factors (normalized with respect to lipid content and organic carbon content) ranged from 0.15 to 0.85 (Ferraro et al., 1990).

Species that can biotransform PAH have internal concentrations well below the concentration in the sediment. The average bioaccumulation factors (normalized with respect to lipid content and organic carbon content) for eel, pike, and roach at two locations were 0.1 and 0.015. The lowest bioaccumulation factor was found at the site with the highest PAH concentration (128 mg/kg, organic carbon-based), probably due to the inductive capability of the fish to biotransform PAH. This was confirmed by the finding of increased hepatic metabolic activity for PAH in the fish (Van der Oost et al., 1991).

4.1.5.2 *Terrestrial organisms*

Little information is available on the accumulation of PAH in terrestrial organisms. The bioaccumulation factors of 22 PAH in the earthworm *Eisenia foetida* at six sites varied from 0.23 to 0.6 on an ash-free dry-weight basis (Rhett et al., 1988).

The half-life of labelled benzo[*a*]pyrene in crickets (*Acheta domesticus*) was 13 h; after 48 h, 36% of the injected dose was unchanged benzo[*a*]pyrene. After topical application of piperonyl butoxide, a known inhibitor of the mixed-function oxidase system, the level of polar metabolites in the excreta had decreased by approximately 75% within 8 h of injection of benzo[*a*]pyrene. After articular application of benzo[*a*]pyrene at 0.29 ng/µl in hexane, some of the dose accumulated internally; the highest level of polar metabolites was found after 24 h (Kumi et al., 1991).

The concentration of PAH in vegetation is generally considerably lower than that in soil, the bioaccumulation factors ranging from 0.0001–0.33 for benzo[*a*]pyrene and from 0.001-0.18 for 17 other PAH tested. It was concluded that some terrestrial plants take up PAH through their roots and/or leaves and translocate them to various other parts (Edwards, 1983).

When bush beans (*Phaseolus vulgaris* Pr.) were exposed to radiolabelled anthracene in a nutrient solution for 30 days during flowering and seed production, more than 90% of the compound was metabolized. Of the total ^{14}C radiolabel, 60% was found in the roots, 3% in the stems, 3% in the leaves, 0.1% in the pods, and 17% in the nutrient solution; 16% was unaccounted for (Edwards, 1986).

4.1.6 *Biomagnification*

Biomagnification, the increase in the concentration of a substance in animals in successive trophic levels of food chains, has been determined in a number of studies. When *Daphnia pulex* were exposed to water or algae

contaminated with naphthalene, phenanthrene, benz[*a*]anthracene, or benzo[*a*]pyrene, naphthalene accumulated to the greatest extent from algal food, (bioconcentration factor, 11 000), whereas benz[*a*]anthracene and benzo[*a*]pyrene accumulated more from water (bioconcentration factors, 1100 and 2700, respectively). It must be emphasized that because of the short exposure (24 h), the last two compounds would not have reached equilibrium (Trucco et al., 1983).

In a study of bioaccumulation and biomagnification in closed laboratory model ecosystems, green algae (*Oedogonium cardiacum*), *D. magna*, mosquito larvae (*Culex pipiens quinquefasciatus*), snails (*Physa* sp.), and mosquito fish (*Gambusia affinis*) were exposed for three days to 2 µg/litre of ^{14}C-benzo[*a*]pyrene. Of the radiolabel accumulated, 88% was attached to parent compound in snails, 22% in mosquito larvae, and none in fish. The parent compound represented 46% of the total extractable radiolabel in mosquito larvae and 90% in *Daphnia*. The bioconcentration factors were 5300 for algae, 12 000 for mosquito larvae, 82 000 for snails, 140 000 for *Daphnia*, and 930 for fish. Despite the apparent absence of bioconcentration in fish, accumulation is assumed to be due to food-chain transfer, as no accumulation of benzo[*a*]pyrene was found in a study of uptake from water. Biomagnification was also studied in a terrestrial-aquatic system, by adding ^{14}C-benzo[*a*]pyrene to *Sorghum vulgare* seedlings and allowing them to be eaten by fourth-instar salt-marsh caterpillar larvae (*Estigmene acrea*); the labelled products entered the terrestrial and aquatic phases as products such as faeces. The food-chain organisms were the same as in the model aquatic ecosystem. After a 33-day interaction period, the concentrations of benzo[*a*]pyrene were 0.01 µg/litre water and 36.1 µg/kg algae, with bioconcentration factors of 3600, 490, 2100, and 30, respectively. Most of the radiolabel was found on polar products or as unextractable radioactivity, which comprised 25% of the total in snails, 63% in fish, 67% in mosquito larvae, and 79% in algae (Lu et al., 1977).

Trophic transfer of benzo[*a*]pyrene metabolites between benthic organisms was studied by feeding *Nereis virens* ^{14}C-benzo[*a*]pyrene and harvesting them five days later. The worm homogenate contained 14% parent compound, 7.2% organic-soluble metabolites, 58% water-soluble metabolites, and 21% bound material. Flounder (*Pseudopleuronectes americanus*) were then given doses of 4.8–19 µg of either pure benzo[*a*]pyrene homogenized in unexposed *Nereis* or the worm-metabolite mixture by gavage and analysed after 24 h of incubation. On the basis of the radiolabel recovered from the fish tissues, assuming comparable accumulation efficiency, flounder appear to have at least a limited ability to accumulate polar, conjugated, and bound metabolic products of benzo[*a*]pyrene from the diet. The parent compound represented 5–15% of the radiolabel in liver and 6–7% in intestine; conjugated metabolites represented 40–60% of the label in liver and 60–70% in intestine; and bound metabolic products represented 30% in liver and 10–20% in intestine (McElroy & Sisson, 1989).

4.2 Transformation

On the basis of model calculations, Mackay et al. (1992) classified some PAH according to their persistence in air, water, soil, and sediment (Table 28).

Table 28. Suggested half-life classes of polycyclic aromatic hydrocarbons in various environmental compartments

Class	Half-life (h)	
	Mean	Range
1	17	10–30
2	55	30–100
3	170	100–300
4	550	300–1000
5	1 700	1000–3000
6	5 500	3000–10 000
7	17 000	10 000–30 000
8	55 000	> 30 000

Compound	Air	Water	Soil	Sediment
Acenaphthylene	2	4	6	7
Anthracene	2	4	6	7
Benz[*a*]anthracene	3	5	7	8
Benzo[*a*]pyrene	3	5	7	8
Benzo[*k*]fluoranthene	3	5	7	8
Chrysene	3	5	7	8
Dibenz[*a*,*h*]anthracene	3	5	7	8
Fluoranthene	3	5	7	8
Fluorene	2	4	6	7
Naphthalene	1	3	5	6
Perylene	3	5	7	8
Phenanthrene	2	4	6	7
Pyrene	3	5	7	8

From Mackay et al. (1992)

4.2.1 Biotic transformation

4.2.1.1 Biodegradation

Information on the biodegradation of PAH in water and soil under aerobic and anaerobic conditions is summarized in Table 29. The few results available from standard tests for biodegradation in water show that PAH

Table 29. Biodegradation of polycyclic aromatic hydrocarbons (PAH)

Compound	Rate constant	Half-life	Comments	Reference
Acenaphthene		100% degradation after 7 d	Significant degradation with rapid adaptation; static flask screening; settled domestic waste as inoculum; experiments with 5 and 10 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		295–2448 h	Aerobic half-life; aerobic soil column	Kincannon & Lin (1985)
		1180–9792 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation half-life	Howard et al. (1991)
		0% degradation after 7 d	Japanese Ministry of Trade and Industry test with 100 mg/litre PAH and 30 mg/litre sludge	Japanese Ministry of International Trade and Industry (1992)
		< 3.2 year	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Acenaphthylene		98% degradation after 7 d	Significant degradation with rapid adaptation; statis flask screening; settled domestic waste as inoculum; 5 or 10 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		1020–1440 h	Aerobic half-life; soil column	Kincannon & Lin (1985)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Acenaphthylene (contd)		4080–5760 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		0% degradation after 4 weeks	Japanese Ministry of Trade and Industry test with 100 mg/litre PAH and 30 mg/litre sludge	Japanese Ministry of International Trade and Industry (1992)
Anthracene	0.061 h^{-1}	*10 h*	Microbial degradation in Third Creek water incubated 18 h at 25 °C: Removal rate constants from water column at 25 °C in midsummer sunlight:	Southworth (1977)
	0.060 h^{-1}	*12 h*	— in deep, slow, somewhat turbid water	
	0.030 h^{-1}	*23 h*	— in deep, slow, muddy water	
	0.061 h^{-1}	*11 h*	— in deep, slow, clear water	
	0.061 h^{-1}	*11 h*	— in shallow, fast, clear water	
	0.061 h^{-1}	*11 h*	— in very shallow, fast, clear water	
	0.035 h^{-1}	*20 h*	Microbial degradation rate constant	Herbes et al. (1980)
		51–92% degradation after 7 d	Significant degradation with gradual adaptation; static flask screening; settled domestic waste as inoculum; experiments with 5 and 10 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Anthracene (contd)		1200–11 040 h	Aerobic half-life; aerobic soil die-away	Coover & Sims (1987)
			200 g dry weight of soil at –0.33 bar [33 kPa] soil moistureat 25 °C:	Park et al. (1990)
	0.0052 d^{-1}	3200 h	— Kidman sandy loam; initial test concentration, 210 mg/kg	
	0.0138 d^{-1}	1200 h	— McLaurin sandy loam; initial test concentration, 199 mg/kg	
		4800–44 160 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation half-life	Howard et al. (1991)
		1.9% degradation after 2 weeks	Japanese Ministry of Trade and Industry test with 100 mg/litre PAH and 30 mg/litre sludge	Japanese Ministry of International Trade and Industry (1992)
Anthracene		33% after 16 months	Degradation in soil in co-metabolic closed bottle with 1-phenyldecane as primary substrate; 20 °C; initial test concentration, 1 mg/g; abiotic loss, 60%	Bossert & Bartha (1986)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Anthracene (contd)		5% after 56 d	Batch test with river water; initial concentration, 20 mg/litre related to dissolved organic carbon; no mineralization during first 19 days; 20 °C	Fedorak et al. (1982)
		10–60% after 64 d	Serum bottle radiorespirometry in five soils contaminated with hydrocarbons: — initial concentration, 31.3 ng/g — Inoculated with enriched culture of *Mycobacterium* sp. and initial test concentration of 37.7 ng/g; biodegradation rate without enriched culture, 18% after 64 d	Grosser et al. (1995)
		100% after 3 d 90% after 20 d	Static test in bioreactor in enriched mixed culture; anthracene oil (38 g/litre) which also contained 62 mg/g fluorene; 30 °C: — under aerobic conditions — under anaerobic conditions	Walter et al. (1990)
		17–45 d	Aerobic degradation in surface Donneybrook sandy loam from Canadian pasture; initial test concentrations, 5 and 50 mg/kg; up to 400 days' exposure at 20 °C and water-holding capacity of 60% of the soil	Bulman et al. (1987)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Anthracene (contd)		7.9 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Benz[*a*]anthracene		2448–16 320 h	Aerobic soil die-away at 10–30 °C	Groenewegen & Stolp (1976); Coover & Sims (1987)
		0% degradation after 7 d	No significant degradation under conditions of method; static flask sceening; settled domestic waste as inoculum; experiment with 5 and 10 mg/littre PAH at 25 °C; detection by GC	Tabak et al. (1981)
	0.0026 d^{-1}	*6400 h*	Kidman sandy loam	Park et al. (1990)
		9792–65 280 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		16% after 16 months	Degradation in soil in co-metabolic closed bottle with 1-phenyldecane as primary substrate; 20 °C; initial test concentration, 1 mg/g; abiotic loss, 18%	Bossert & Bartha (1986)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Benz[a]anthracene (contd)		0–40% after 64 d	Serum bottle radiorespirometry in five soils contaminated with hydrocarbons; initial concentration, 31.3 ng/g	Grosser et al. (1995)
		130–240 d	Aerobic degradation in surface samples of Donneybrook sandy loam from Canadian pasture; initial test concentrations, 5 and 50 mg/kg; up to 400 days' exposure at 20 °C and water-holding capacity of 60% of the soil	Bulman et al. (1987)
		8.1 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Benzo[a]pyrene	0.2–0.9 $\mu mol.h^{-1}mg^{-1}$		Aquatic fate rate for bacterial protein	Bamsley (1975)
	3.5×10^{-6} h^{-1}	*19 800 h*	Estimated rate constant in soil and water	Ryan & Cohen (1986)
		1368–12 702 h	Aerobic half-life at 10–30 °C; soil die-away	Coover & Sims (1987)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Benzo[*a*]pyrene (contd)			200 g dry weight of soil at –0.33 bar [33 kPa] soil moisture; 33 mg/kg at 25 °C:	Park et al. (1990)
	0.0022 d^{-1}	7416 h	— Kidman sandy loam	
	0.0030 d^{-1}	5496 h	— McLaurin sandy loam	
		5472–50 808 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Coover & Sims (1987)
		< 8% after 160 d	Serum bottle radiorespirometry in five soils contaminated with hydrocarbons; initial concentration, 105 ng/g	Grosser et al. (1995)
		218–347 d	Aerobic degradation in surface samples of Donneybrook sandy loam from Canadian pasture; initial test concentrations, 5 and 50 mg/kg; up to 400 days' exposure at 20 °C and water-holding capacity of 60% of the soil	Bulman et al. (1987)
		8.2 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Benzo[*b*]fluoranthene		8640–14 640 h	Aerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Coover & Sims (1987)
			200 g dry weight of soil at −0.33 bar [33 kPa] soil moisture; initial test concentration, ± 38 mg/kg at 25 °C:	Park et al. (1990)
	0.0024 d^{-1}	7056 h	— Kidman sandy loam	
	0.0033 d^{-1}	5064 h	— McLaurin sandy loam	
		34 560–58 560 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		9 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Benzo[*ghi*]perylene		14 160–15 600 h	Aerobic half-life; aerobic soil dieaway at 10–30 °C	Coover & Sims (1987)
		56 640–62 400 h	Anaerobic half-life; aerobic soil dieaway at 10–30 °C	Coover & Sims (1987)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Benzo[*ghi*]perylene (contd)		9.1 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Benzo[*k*]fluoranthene		21 840–51 360 h	Aerobic half-life; aerobic soil dieaway	Coover & Sims (1987)
		87 360–205 440 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		8.7 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Chrysene		59% degradation after 7 d	Significant degradation with gradual adaptation; static flask screening; settled domestic waste as inoculum; experiment with 5 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		38% degradation after 7 d	No significant degradation under conditions of method; static flask sceening; settled domestic waste as inoculum; experiment with 10 mg/littre PAH at 25 °C; detection by GC	Tabak et al. (1981)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Chrysene (contd)		8904–24 000 h	Aerobic half-life; aerobic soil dieaway	Coover & Sims (1987)
			200 g dry weight of soil at –0.33 bar [33 kPa] soil moisture; initial test concentration, ± 100 mg/kg at 25 °C:	Park et al. (1990)
	0.0019 d^{-1}	8904 h	— Kidman sandy loam	
	0.0018 d^{-1}	9288 h	— McLaurin sandy loam	
		35 616–96 000 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		11% after 16 months	Degradation in soil in co-metabolic closed bottle with 1-phenyldecane as primary substrate; 20 °C; initial test concentration, 1 mg/g; abiotic loss, 5%	Bossert & Bartha (1986)
		224–328 d	Aerobic degradation in surface samples of Donneybrook sandy loam from Canadian pasture; initial test concentrations, 5 and 50 mg/kg; up to 400 days' exposure at 20 °C and water-holding capacity of 60% of the soil	Bulman et al. (1987)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Chrysene (contd)		8.1 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Coronene		16.5 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Dibenz[*a*,*h*]anthracene		8664–22 560 h	Aerobic half-life; aerobic soil die-away	Coover & Sims (1987); Park et al. (1990)
			200 g dry weight of soil at –0.33 bar [33 kPa] soil moisture; initial test concentration, ± 13 mg/kg at 25 °C:	Park et al. (1990)
	0.0019 d^{-1}	8664 h	— Kidman sandy loam	
	0.0017 d^{-1}	10 080 h	— McLaurin sandy loam	
		No degradation after 16 months	Degradation in soil in co-metabolic closed bottle with 1-phenyldecane as primary substrate; 20 °C; initial test concentration, 1 mg/g	Bossert & Bartha (1986)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Fluoranthene	2.2×10^{-3} µmol $h^{-1} mg^{-1}$		Aquatic fate rate with bacterial protein	Bamsley (1975)
		100% degradation after 7 d	Significant degradation with gradual adaptation; static flask screening; settled domestic waste as inoculum; experiment with 5 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		0% degradation after 7 d	No significant degradation under conditions of method; static flask screening; settled domestic waste as inoculum; experiment with 10 mg/littre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		3360–10 560 h	Aerobic half-life; aerobic soil dieaway	Coover & Sims (1987)
	$0.19\ h^{-1}$	*3.6 h*	In atmosphere	Dragoescu & Friedlander (1989)
			200 g dry weight of soil at –0.33 bar [33 kPa] soil moisture; initial test concentration, 900 mg/kg at 25 °C:	Park et al. (1990)
	$0.0018\ d^{-1}$	9048 h	— Kidman sandy loam	
	$0.0026\ d^{-1}$	6432 h	— McLaurin sandy loam	

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Fluoranthene (contd)		13 440–42 240 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		34–39 d	Aerobic degradation in surface samples of Donneybrook sandy loam from Canadian pasture; initial test concentrations, 5 and 50 mg/kg; up to 400 days' exposure at 20 °C and water-holding capacity of 60% of the soil	Bulman et al. (1987)
		7.8 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Fluorene		45–77% degradation after 7 d	Significant degradation with gradual adaptation; static flask screening; settled domestic waste as inoculum; experiment with 5 and 10 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
			Degradation of 30 µg/litre in natural river water (Skidway River; salinity, 20‰):	Lee & Ryan (1976)
		100% after 1000 d	— Turnover time in June at incubation time of 48 h	
		0% after 72 h	— February or May	

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Fluorene (contd)		30% after 1 week	Degradation of non-autoclaved groundwater samples of ± 0.06 mg/litre by microbes	Lee et al. (1984)
		768–1440 h	Aerobic half-life; aerobic soil diaway	Coover & Sims (1987)
		3072–5760 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		0% degradation after 4 weeks	Japanese Ministry of Trade and Industry test with 100 mg/litre PAH and 30 mg/litre sludge	Japanese Ministry of International Trade and Industry (1992)
		100% after 36 h	Batch test with enriched culture of *Arthrobacter* sp.; initial test concentration, 483 µmol/litre; 22 °C	Grifoll et al. (1992)
		< 3.2 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Indeno[1,2,3-*cd*]pyrene			200 g dry weight of soil at –0.33 bar [33 kPa] soil moisture; initial test concentration, ± 8 mg/kg at 25 °C:	Park et al. (1990)
	0.0024 d^{-1}	6912 h	— Kidman sandy loam	
	0.0024 d^{-1}	6936 h	— McLaurin sandy loam	

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Naphthalene			Degradation in natural river water (Skidway River; salinity, 20‰):	Lee & Ryan (1976)
		500 d	— Turnover time in February at incubation time of 48 h; test concentration, 40 µg/litre	
		46 d	— Turnover time in May at incubation time of 24 h; test concentration, 40 µg/litre	
		79 d	— Turnover time in May at incubation time of 8 h; test concentration, 40 µg/litre	
		30 d	— Turnover time in May at incubation time of 24 h; test concentration, 130 µg/litre	
		330 d	Degradation of 130 µg/litre in natural water offshore with salinity of 35%: turnover time in May at incubation time of 24 h	Lee & Ryan (1976)
	0.04–3.3 x 10^{-6} g/litre per d		At depth of 5–10 m in laboratory water basin	Lee & Anderson (1977)
		100% after 8 d	In gas–oil-contaminated groundwater circulated through sand inoculated with groundwater under aerobic conditions	Kappeler & Wuhrmann (1978)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Naphthalene (contd)		168 h 576 h 1500 h 40 800 h	In oil-polluted estuarine stream In clean estuarine stream In coastal waters In the Gulf Stream	Lee (1977)
		12h	Aerobic half-life; die-away in oil-polluted creek	Walker & Colwell (1976)
		 600 h 6200 h	Anaerobic half-life: at pH 8 at pH 5	Hambrick et al. (1980)
		24–216 h	In deep, slowly moving, contaminated water	Herbes (1981); Wakeham et al. (1983)
	0.23 h^{-1}	*3.0 h*	Microbial degradation rate constant	Herbes et al. (1980)
		100% degradation after 7 d	Significant degradation with rapid adaptation; static flask screening; settled domestic waste as inoculum; experiments with 5 and 10 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Naphthalene (contd)		100% degradation after 7 d	Degradation of non-autoclaved groundwater samples of ± 0.04 mg/litre by microbes	Lee et al. (1984)
	0.024 d^{-1}	*693 h*	Groundwater with nutrients and acclimatized microbes	Vaishnav & Babeu (1987)
	0.013 d^{-1}	*1279 h*	River water with acclimatized microbes	
	0.018 $^{-1}$	*924 h*	River water with nutrients and acclimatized microbes	
			200 g dry weight of soil at –0.33 bar [–0.0032 kPa] soil moisture; initial test concentration, 101 mg/kg at 25 °C:	Park et al. (1990)
	0.377 d^{-1}	50 h	— Kidman sandy loam	
	0.308 d^{-1}	53 h	— McLaurin sandy loam	
		2% degradation after 4 weeks	Japanese Ministry of Trade and Industry test with 30 mg/litre PAH and 100 mg/litre sludge	Japanese Ministry of International Trade and Industry (1992)
		< 2.1 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Perylene		No degradation after 16 months	Degradation in soil in co-metabolic closed bottle with 1-phenyldecane as primary substrate; 20 °C; initial test concentration, 1 mg/g	Bossert & Bartha (1986)
Phenanthrene		100% degradation after 7 d	Significant degradation with rapid adaptation; static flask screening; settled domestic waste as inoculum; experiments with 5 and 10 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		383–4800 h	Aerobic half-life; aerobic soil die-away	Coover & Sims (1987)
			200 g dry weight of soil at –0.33 bar [–0.0032 kPa] soil moisture; initial test concentration, 900 mg/kg at 25 °C:	Park et al. (1990)
	0.0447 d^{-1}	384 h	— Kidman sandy loam	
	0.0196 d^{-1}	840 h	— McLaurin sandy loam	
		1536–19 200 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Phenanthrene (contd)		96 h 264 h	Inorganic solution Kendaia soil	Manilal & Alexander (1991)
		54% degradation after 4 weeks	Japanese Ministry of Trade and Industry test with 100 mg/litre PAH and 30 mg/litre sludge	Japanese Ministry of International Trade and Industry (1992)
		> 62 % after 16 months	Degradation in soil in co-metabolic closed bottle with 1-phenyldecane as primary substrate; 20 °C; initial test concentration, 1 mg/g; abiotic loss significant	Bossert & Bartha (1986)
		38–55% after 64 d 80% after 32 d	Serum bottle radiorespirometry in five soils contaminated with hydrocarbons: — initial concentration, 31.3 ng/g — inoculated with enriched culture of *Mycobacterium* sp. and an initial test concentration of 17.9 ng/g	Grosser et al. (1995)
		9.7–14 d	Aerobic degradation in surface samples of Donneybrook sandy loam from Canadian pasture; initial test concentrations, 5 and 50 mg/kg; up to 400 days' exposure at 20 °C and water-holding capacity of 60% of the soil	Bulman et al. (1987)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Phenanthrene (contd)		5.7 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)
Pyrene		100% degradation after 7 d	Significant degradation with rapid adaptation; static flask screening; settled domestic waste as inoculum; experiment with 5 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		0% degradation after 7 d	No significant degradation under conditions of method; static flask screening; settled domestic waste as inoculum; experiments with 5 and 10 mg/litre PAH at 25 °C; detection by GC	Tabak et al. (1981)
		5040–45 600 h	Aerobic half-life at 10–30 °C; aerobic soil die-away	Coover & Sims (1987)
	0.29 h^{-1}	*2.4 h*	In atmosphere	Dragoescu & Friedlander (1989)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Pyrene (contd)			200 g dry weight of soil at –0.33 bar [33 kPa] soil moisture; initial test concentration, ± 690 mg/kg at 25 °C:	Park et al. (1990)
	0.0027 d^{-1}	6240 h	— Kidman sandy loam	
	0.0035 d^{-1}	4776 h	— McLaurin sandy loam	
		20 160–182 400 h	Anaerobic half-life; estimated unacclimatized aqueous aerobic biodegradation	Howard et al. (1991)
		70% after 16 months	Degradation in soil in co-metabolic closed bottle with 1-phenyldecane as primary substrate; 20 °C; initial test concentration, 1 mg/g; abiotic loss, 27%	Bossert & Bartha (1986)
			Serum bottle radiorespirometry in five soils contaminated with hydrocarbons:	Grosser et al. (1995)
		25–70% after 64 d	— initial concentration, 8.5 ng/g	
		54% after 32 d	— inoculated with enriched culture of *Mycobacterium* sp. and an initial test concentration of 7.7 ng/g	
		52.4% after 96 h	Mineralization test with *Mycobacterium* sp.; 24 °C; initial test concentration, 0.5 mg/litre	Heitkamp et al. (1988)

Table 29 (contd)

Compound	Rate constant	Half-life	Comments	Reference
Pyrene (contd)		48–58 d	Aerobic degradation in surface Donneybrook sandy loam from Canadian pasture; initial test concentrations, 5 and 50 mg/kg; up to 400 days' exposure at 20 °C and water-holding capacity of 60% of the soil	Bulman et al. (1987)
		8.5 years	Field tests of rural British soils amended with metal-enriched sewage sludges with 0.1–15.1 mg/kg PAH	Wild et al. (1991)

GC, gas chromatography

In order to compare numbers when only rate constants are reported, the half-lives were estimated ffrom the formula:

$$t_{1/2} = \frac{\ln 2}{k}$$

where $t_{1/2}$ is the half-life and k is the biodegradation rate constant. The calculated values are given in italics.

with up to four aromatic rings are biodegradable under aerobic conditions but that the biodegradation rate of PAH with more aromatic rings is very low. Biodegradation under anaerobic conditions is slow for all components (Neff, 1979). The reactions normally proceed by the introduction of two hydroxyl groups into the aromatic nucleus, to form dihydrodiol intermediates. Bacterial degradation produces *cis*-dihydrodiols (from a dioxetane intermediate), whereas metabolism in fungal or mammalian systems produces *trans*-dihydrodiol intermediates (from an arene oxide intermediate). The differences in the metabolic pathways are due to the presence of the cytochrome P450 enzyme system in fungi and mammals. Algae have been reported to degrade benzo[*a*]pyrene to oxides, peroxides, and dihydroxydiols (see below). Owing to the high biotransformation rate (see also section 4.2.1.2), the concentrations of PAH in organisms and water are usually not in a steady state. Freely dissolved PAH may be rapidly degraded under natural conditions if sufficient biomass is available and the turnover rates are fairly high (see Table 29).

Biodegradation is the major mechanism for removal of PAH from soil. PAH with fewer than four aromatic rings may also be removed by volatilization and photolysis (see also sections 4.1.4 and 4.2.2.1). The rate of biodegradation in soil depends on several factors, including the characteristics of the soil and its microbial population and the properties of the PAH present. Temperature, pH, oxygen content, soil type, nutrients, and the presence of other substances that can act as co-metabolites are also important (Sims & Overcash, 1983). Biodegradation is further affected by the bioavailability of the PAH. Sorption of PAH by soil organic matter may limit the biodegradation of compounds that would normally undergo rapid degradation (Manilal & Alexander, 1991); however, no significant difference was found in the biodegradation rate of anthracene in water with 10 and 1000 mg/litre suspended material (Leslie et al., 1987). In Kidman sandy loam, the biodegradation rates varied between 0.23 h^{-1} (or 5.5 d^{-1}) for naphthalene and 0.0018 d^{-1} for fluoranthene (see Table 29). In a study with sandy loams, forest soil, and roadside soil partially loaded with sewage sludge from a municipal treatment plant, the following half-lives (in days) were found: 14–48 for naphthalene, 44–74 for acenaphthene plus fluorene, 83–193 for phenanthrene, 48–210 for anthracene, 110–184 for fluoranthene, 127–320 for pyrene, 106–313 for benz[*a*]anthracene plus chrysene, 113–282 for benzo[*b*]fluoranthene, 143–359 for benzo[*k*]fluoranthene, 120–258 for benzo[*a*]pyrene, 365–535 for benzo[*ghi*]perylene, and 603–2030 for coronene (Wild & Jones, 1993).

After biodegradation of pyrene by a *Mycobacterium* sp., *cis*- and *trans*-4,5-pyrene dihydrodiol and pyrenol were the initial ring oxidation products. The main metabolite was 4-phenathroic acid. The ring fission products were 4-hydroxyperinaphthenone and cinnamic and phthalic acids (Heitkamp et al., 1988).

The pyrene-metabolizing *Mycobacterium* sp. can also use phenanthrene and fluoranthene as the sole source of carbon. Phenanthrene was degraded and 1-hydroxy-2-naphthoic acid, *ortho*-phthalate, and protocatechuate were detected as metabolites. 1-Hydroxy-2-naphthoic acid did not accumulate, indicating that it is further metabolized (Boldrin et al., 1993).

A strain of *Arthobacter* sp. was isolated that was capable of metabolizing fluorene as a sole energy source: 483 nmol/ml were degraded completely within 36 h, and four major metabolites were detected: 9-fluorenol, 9*H*-fluoren-9-one, 3,4-dihydrocoumarin, and an unidentified polar-substituted aromatic compound. Fluorenol was not degraded further, suggesting that it and fluorenone are products of a separate metabolic pathway from that which produces dihydrocoumarin, the polar compound, and the energy for cell growth. The bacteria could also degrade phenanthrene (Grifoll et al., 1992).

The degradation of PAH was studied in a culture made from activated sludge, polychlorinated biphenyl-degrading bacteria, and chlorophenol-degrading mixed cultures, adapted to naphthalene. The metabolites of naphthalene were 2-hydroxybenzoic acid and 1-naphthalenol, those of phenanthrene were 1-phrenanthrenol and 1-hydroxy-2-naphthalenecarboxylic acid, and that of anthracene was 3-hydroxy-2-naphthalenecarboxylic acid. The authors concluded that the biotransformation pathway proceeds via initial hydroxylation to ring cleavage, to yield the *ortho* or *meta* cleavage intermediates, which are further metabolized via conventional metabolic pathways (Liu et al., 1992).

The metabolism of PAH by fungi is similar to that by mammalian cells. For example, *Cunninghamella elegans* in culture metabolizes benzo[*a*]pyrene to the *trans*-7,8-diol, the *trans*-9,10-diol, 3,6-quinone, 9-hydroxybenzo[*a*]pyrene, 3-hydroxybenzo[*a*]pyrene, and 7,8-dihydro-7,8-dihydroxybenzo[*a*]pyrene (Cerniglia, 1984). In a further experiment, *C. elegans* metabolized about 69% of added fluorene after 24 h. The major ethyl acetate-soluble metabolites were 9-fluorenone (62%), 9-fluorenol, and 2-hydroxy-9-fluorenone (together, 7%). The degradation pathway was similar to that in bacteria, with oxidation at the C9 position of the five-member ring to form an alcohol and the corresponding ketone. 2-Hydroxy-9-fluorenone had not been found as a metabolite previously (Pothuluri et al., 1993).

4.2.1.2 *Biotransformation*

Biotransformation is often advanced as an explanation for the differences in PAH profiles seen in aquatic organisms and in the medium to which they were exposed. Furthermore, all of the metabolites of PAH may not have been identified or quantified. This section addresses biotransformation in organisms other than bacteria and fungi, which is discussed in section 4.2.1.1, above.

The uptake of naphthalene and benzo[*a*]pyrene was studied in three species of marine fish: the mudsucker or sand goby (*Gillichthys mirabilis*), the sculpin

(*Oligocottus maculosus*), and the sand dab (*Citharichthys stigmaeus*). In all three species, biotransformation took place rapidly in the liver. The uptake of naphthalene was greater than that of benzo[*a*]pyrene. The major metabolite of benzo[*a*]pyrene appeared to be 7,8-dihydroxy-7,8-dihydroxy benzo[*a*]pyrene, while the major metabolite of naphthalene was 1,2-dihydro-1,2-dihydroxy-naphthalene. The gall-bladder was the major storage site for the PAH and their metabolites. Naphthalene and its metabolites were removed at a higher rate than benzo[*a*]pyrene and its metabolites (Lee et al., 1972).

Transformation of naphthalene and benzo[*a*]pyrene in the bluegill sunfish *Lepomis macrochirus* took place very rapidly, benzo[*a*]pyrene having the highest rate (McCarthy & Jimenez, 1985). *L. macrochirus* were exposed in a flow-through system to 4 nmol/litre benzo[*a*]pyrene for 48 h, followed by a 96-h depuration period, at 13 or 23 °C in the presence or absence of food. Both polar and nonpolar metabolites were found. After 48 h, the polar metabolites comprised 10% of the benzo[*a*]pyrene metabolites in fed fish at 13 °C, 20% in unfed fish at 23 °C, and 30% in fed fish at 23 °C (Jimenez et al., 1987). In rainbow trout (*Oncorhynchus mykiss*) exposed to naphthalene at 0.5 mg/litre for 24 h, the bile contained 65–70% metabolites, the liver contained 5–10%, and muscle < 1% (Melancon & Lech, 1978).

In *L. macrochirus* exposed to 8.9 ± 2.1 µg/litre acenaphthene for 28 days, the half-life for metabolism was less than one day. No information was given on metabolites (Barrows et al., 1980).

The depuration of anthracene was investigated in *O. mykiss* during simulated day and night cycles of 16 and 8 h, respectively. After a 96-h clearance period, the metabolites contributed 2-3% of the depurated substance, half of which came from the bile. No specific metabolites were reported (Linder & Bergman, 1984). After *L. macrochirus* had been exposed to anthracene at 8.9 µg/litre or benzo[*a*]pyrene at 0.98 µg/litre for 4 h, the rates of biotransformation were 0.26 and 0.082 nmol/g per h, respectively, and 8% of the anthracene and 88% of the benzo[*a*]pyrene were metabolized (Spacie et al., 1983).

Benzo[*a*]pyrene is transformed in the Japanese medaka (*Oryzias latipes*) and the guppy (*Poecilia reticulata*), the main metabolite being the 7,8-diol-9,10-epoxide (Hawkins et al., 1988).

Two benthic organisms, the European fingernail clam (*Sphaerium corneum*) and larvae of the midge *Chironomus riparius*, both metabolized benzo[*a*]pyrene. In the larvae, the main metabolite appeared to be 3-hydroxybenzo[*a*]pyrene; a quinone isomer was also found. Only a very small amount of 3-hydroxy-benzo[*a*]pyrene was found in the clam. No diol metabolites were found in either species (Borchert & Westendorf, 1994). After exposure of the benthic oligochaete *Stylodrilus heringianus* to either anthracene and pyrene or phenanthrene and benzo[*a*]pyrene, 2% degradation of each PAH was reported within 24 h (Frank et al., 1986).

The half-lives for metabolism in *D. magna* were 0.5 h for 1.8 mg/litre naphthalene, 9 h for 0.06 mg/litre phenanthrene, and 18 h for 0.023 mg/litre chrysene (Eastmond et al., 1984).

In amphipod *Hyalella azteca* was exposed to 0.043 nmol/ml anthracene for 8 h, the rates of biotransformation were 2.2 ± 0.5 nmol/g dry weight per h with no substratum, 3.0 ± 0.8 in the presence of washed sand from a local lake, and 1.0 ± 0.15 in the presence of sediment from the lake (Landrum & Scavia, 1983).

The amphipod *Rhepoxynius abronius* metabolizes benzo[*a*]pyrene (Plesha et al., 1988). When two marine amphipods were exposed to a sediment containing 5.1 ng/mg of this compound, *R. abronius* metabolized 49% and *Eohaustorius washingtonianus* metabolized 27% of the benzo[*a*]pyrene after one day. The main metabolites appeared to be 7,8-dihydro-7,8-dihydroxybenzo[*a*]pyrene, 9,10-dihydro-9,10-dihydroxybenzo[*a*]pyrene, 3-hydroxybenzo[*a*]pyrene, and 9-hydroxybenzo[*a*]pyrene. The ratio of 7,8-dihydro-7,8-dihydroxybenzo[*a*]pyrene to 9,10-dihydro-9,10-dihydroxybenzo[*a*]pyrene in normal-phase high-performance liquid chromatography was 1.2 for *R. abronius* and 0.7 for *E. washingtonianus* (Reichert et al., 1985).

No biotransformation of benzo[*a*]pyrene or phenanthrene was found in mayflies (*Hexagenia limbata*) or in the amphipod *Pontoreia hoyi* (Landrum & Poore, 1988).

In a study of the route of metabolism of benzo[*a*]pyrene in green algae (*Selenastrum capricornutum*) exposed to 1.2 µg/litre for four days, with simulated day and night periods, the major dihydrodiol metabolites identified were the *cis*-4,5-diol (< 1%), the *cis*-7,8-diol (13%), the 9,10-diol (36%), and the *cis*-11,12-diol (50%), demonstrating the presence of a dioxygenase enzyme for this type of algae (Lindquist & Warshawsky, 1985), as suggested by Cody et al. (1984). Payne (1977) reported, however, that aryl hydrocarbon hydroxylase was not present in *Fucus* and *Ascophyllum* sp. of marine algae.

Benzo[*a*]pyrene was not biotransformed in periphyton after 0.25 or 4 h. In cladocerans (*D. magna*) exposed to 1.0 µg/litre benzo[*a*]pyrene, the biotransformation rate after exposure for 6 h was 1.07 ± 0.20 nmol/g dry weight per h. In midge larvae (*C. riparius*) exposed to 0.6-1.5 µg/litre, the biotransformation rate was 3.6 ± 0.7 nmol/g dry weight per h after exposure for 1 h and 2.7 ± 0.3 after 4 h. In *L. macrochirus* exposed to 1.0 µg/litre, the biotransformation rate was 0.20 ± 0.03 nmol/g dry weight per h after 1 h and 0.37 ± 0.04 after 4 h. In chironomids, 3-hydroxybenzo[*a*]pyrene was the major metabolite after 8 h, representing 4.4% of the total water activity; smaller amounts of 7-hydroxybenzo[*a*]pyrene and the 9,10- and 7,8-dihydroxydiols of benzo[*a*]pyrene were also found (Leversee et al., 1981).

After exposure of benthic species to benzo[*a*]pyrene for one to four weeks, the following percentages of metabolites were found: *E. washingtonianus*, 22% in the whole body; *R. abronius*, 74% in the whole body; clams (*Macoma nasuta*), < 5% in the body and < 5 in the hepatopancreas; shrimp (*Pandalus*

platyceros), 94% in the hepatopancreas; and the English sole (*Parophrys vetulus*), 94% in the body, 99% in the liver and > 99% in the bile (Varanasi et al., 1985).

Mosquito larvae (*C. pipens quinquefasciatus*) were exposed for three days to 0.002 mg/litre benzo[*a*]pyrene in the presence or absence of the mixed-function oxidase inhibitor piperonyl butoxide at 0.0025 mg/litre. Parent benzo[*a*]pyrene represented 22% of the excreted PAH in the absence of piperonyl butoxide and 86% in its presence. After three days' exposure of snails (*Physa* sp.) to the same concentration of benzo[*a*]pyrene with or without piperonyl butoxide at 0.0025 mg/litre, parent benzo[*a*]pyrene represented 88% in the absence of the inhibitor and 85% in its presence. The authors suggested that snails are deficient in microsomal oxidases. In mosquito fish (*G. affinis*) exposed similarly, no parent benzo[*a*]pyrene was found in the absence of piperonyl butoxide but 21% in its presence (Lu et al., 1977).

In an aquatic ecosystem, plankton, green algae (*Oedogonium cardiacum*), *D. magna*, mosquito larvae (*C. pipiens quinquefasciatus*), snails (*Physa* sp.), and mosquito fish (*G. affinis*) were exposed to 0.002 mg/litre benzo[*a*]pyrene for three days. Parent benzo[*a*]pyrene represented 83, 90, 46, 70, and 55% in the four organisms, respectively. The substance was metabolized to unidentified hydroxylated polar compounds. The finding of 55% parent benzo[*a*]pyrene in the fish was attributed to food-chain transfer, as none was found after direct exposure. A terrestrial-aquatic ecosystem was also exposed to benzo[*a*]pyrene by applying 0.2 mg of radiolabelled compound to *Sorghum vulgare* seedlings to simulate atmospheric fall-out and allowing them to be consumed by fourth-instar salt-marsh caterpillar larvae (*E. acrea*). Faecal products then entered the terrestrial and aquatic ecosystem described above, which was left for 33 days. The maximum radiolabel (0.005 ppm) was detected in the aquatic phase after 14 days. Unmetabolized benzo[*a*]pyrene accounted for 7.1% of the total extractable radiolabel in fish, 19% in snails, 32% in algae, and 34% in mosquitoes. Addition of the mixed-function oxidase inhibitor, piperonyl butoxide, resulted in 12% parent benzo[*a*]pyrene in fish, 34% in snails, 48% in the algae, and no change in mosquitoes (Lu et al., 1977).

The biotransformation of 19 PAH was studied in the food chain seston (plankton)→blue mussel (*Mytilus edulis* L.)→common eider duck (*Somateria mollissima* L.) in the open, northern Baltic Sea. The concentrations of the PAH in the eider duck showed the distribution gallbladder > adipose tissue ≥ liver. There was a high flux of the PAH in the food chain, but the concentration did not increase with increasing trophic level, indicating that the PAH were biotransformed rapidly. There was little biotransformation in the plankton. The distribution of the PAH in blue mussels was different from that in plankton, perhaps due to metabolic activity in the mussel. Biotransformation of PAH with a relative molecular mass of 252 was rapid in the ducks (Broman et al., 1990).

In beans (*Phaseolus vulgaris* L.) exposed to 15 µg anthracene per plant, uptake via the roots was rapid, 90% being metabolized within 30 days (Edwards, 1986).

These investigations are summarized in Table 30. As the rate of metabolism depends not only on the species but also on factors such as temperature, pH, and other experimental conditions, the results are difficult to compare. Some general conclusions can, however, be drawn:

- The biotransformation potential of aquatic organisms depends on the activity of cytochrome P450-dependent mixed-function oxidases, which are important for oxidation, the first step in the metabolism of xenobiotics such as PAH (James, 1989).
- The tissues in which biotransformation mainly takes place are liver, lung, kidney, placenta, intestinal tract, and skin (Cerniglia, 1984).
- The initial transformation step in invertebrates usually occurs more slowly than in vertebrates (James, 1989). Monoxygenation of PAH is faster in higher invertebrates like arthropods, echinoderms, and annelids and slowest in more primitive invertebrates like protozoa, profina, cnidaria, and molluscs (Neff, 1979).
- In general, invertebrates excrete PAH metabolites inefficiently (James, 1989).
- In higher organisms and algae, metabolites are usually produced by monooxygenase activity, resulting in the formation of epoxides, phenols, diols, tetrols, quinones, and conjugates.
- It is not clear whether molluscs have cytochrome P450 activity (Moore et al., 1989).
- In crustaceans, biotransformation differs greatly between species and for different PAH. Biotransformation of naphthalene, anthracene, phenanthrene, and chrysene appears to occur rapidly, while that of benzo[*a*]pyrene is generally slower. Only Reichert et al. (1985) reported significant degradation in *R. abronius* (49%) and *E. washingtonianus* (27%) within one day.
- It is not clear how rapidly biotransformation occurs in insects.
- Too little information was available on algae, plants, and fungi for conclusions to be drawn.

4.2.2 *Abiotic degradation*

Abiotic processes may account for the removal of 2–20% of two- and three-ring PAH from soil (Park et al., 1990). In soils partly amended with PAH-containing sewage sludge, 24–100% was removed, and naphthalene was eliminated almost completely by volatilization and photodegradation (Wild & Jones, 1993).

Table 30. Biotransformation of polycyclic aromatic hydrocarbons by various organisms

Species	Compound	Biotransformation rate	Reference
Fungi			
Cunninghamella elegans	Benzo[*a*]pyrene	No information	Cerniglia (1984)
Algae			
Selenastrum capricornutum	Benzo[*a*]pyrene	Relatively fast	Lindquist & Warshawsky (1985)
Oedogenium cardiacum	Benzo[*a*]pyrene	15% after 3 d in aquatic ecosystem	Lu et al. (1977)
Fucus sp.	Various	None	Payne (1977)
Ascophyllum sp.	Various	None	
Molluscs			
Sphaerium corneum	Benzo[*a*]pyrene	Very fast (no carcinogenic metabolites)	Borchert & Westendorf (1994)
Physa sp.	Benzo[*a*]pyrene	12% after 3 d	Lu et al. (1977)
Mytilus edulis L.	Different	No information	Broman et al. (1990)
Crustaceae			
Hyalella azteca	Anthracene	2.2 nmol/g dw/h in water	Landrum & Scavia (1983)
Hyalella azteca	Anthracene	3.0 nmol/g dw/h in water/ sediment	Landrum & Scavia (1983)
Daphnia magna	Benzo[*a*]pyrene	1.07 nmol/g dw/h after 6 h	Leversee et al. (1981)
Daphnia magna	Benzo[*a*]pyrene	10% after 3 d in aquatic ecosystem	Lu et al. (1977)

Table 30 (contd)

Species	Compound	Biotransformation rate	Reference
Crustaceae (contd)			
Pontoporeia hoyi	Benzo[a]pyrene	None	Landrum & Poore (1988)
Pontoporeia hoyi	Benzo[a]pyrene	None after 48 h	Evans & Landrum (1989)
Mysis relicta	Benzo[a]pyrene	No information	Evans & Landrum (1989)
Rhepoxynius abronius	Benzo[a]pyrene	No information	Plesha et al. (1988)
Rhepoxynius abronius	Benzo[a]pyrene	74% after 1–4 weeks	Varanasi et al. (1985)
Rhepoxynius abronius	Benzo[a]pyrene	49% after 1 d	Reichert et al. (1985)
Eohaustorius washingtonianus	Benzo[a]pyrene	27% after 1 d	Reichert et al. (1985)
Eohaustorius washingtonianus	Benzo[a]pyrene	22% after 1–4 weeks	Varanasi et al. (1985)
Pandalus platyceros	Benzo[a]pyrene	< 5% after 1–4 weeks	Varanasi et al. (1985)
Parophrys vetulus	Benzo[a]pyrene	94% after 1–4 weeks	Varanasi et al. (1985)
Daphnia magna	Chrysene	50% after 18 h	Eastmond et al. (1984)
Daphnia magna	Naphthalene	50% after 0.5 h	Eastmond et al. (1984)
Daphnia magna	Phenanthrene	50% after 9 h	Eastmond et al. (1984)
Fish			
Lepomis macrochirus	Acenaphthene	Half-life, < 1 d	Barrows et al. (1980)
Lepomis macrochirus	Anthracene	8% after 4 h	Spacie et al. (1983)
Oncorhynchus mykiss	Anthracene	2–3% after 24 h	Linder & Bergman (1984)
Gillichthys mirabilis	Benzo[a]pyrene	Rapid in liver	Lee et al. (1972)
Oligocottus maculosus	Benzo[a]pyrene	Rapid in liver	Lee et al. (1972)
Citharichthys stigmaeus	Benzo[a]pyrene	Rapid in liver	Lee et al. (1972)
Lepomis macrochirus	Benzo[a]pyrene	Very fast	McCarthy & Jimenez (1981)

Table 30 (contd)

Species	Compound	Biotransformation rate	Reference
Fish (contd)			
Lepomis macrochirus	Benzo[*a*]pyrene	88% after 4h	Spacie et al. (1983)
Lepomis macrochirus	Benzo[*a*]pyrene	0.20–0.37 nmol/g dry weight per h	Leversee et al. (1981)
Oryzias latipes	Benzo[*a*]pyrene	No information	Hawkins (1988)
Poecilia reticulata	Benzo[*a*]pyrene	No information	Hawkins (1988)
Rhepoxynius abronius	Benzo[*a*]pyrene	None	Plesha et al. (1988)
Gambusia affinis	Benzo[*a*]pyrene	100% after 3 d in water 40% after 3 d in aquatic ecosystem	Lu et al. (1977)
Gillichthys mirabilis	Naphthalene	Rapid in liver	Lee et al. (1972)
Oligocottus maculosus	Naphthalene	Rapid in liver	Lee et al. (1972)
Citharichthys stigmaeus	Naphthalene	Rapid in liver	Lee et al. (1972)
Lepomis macrochirus	Naphthalene	Very fast	McCarthy & Jimenez (1981)
Worm			
Stylodrilus heringianus	Various	None	Franck et al. (1986)
Insects			
Chironomus riparius	Benzo[*a*]pyrene	Very fast (no carcinogenic metabolites)	Bochert & Westendorf (1994)
Chironomus riparius	Benzo[*a*]pyrene	2.7–3.6 nmol/g dry weight per h	Leversee et al. (1981)
Hexagenia limbata	Benzo[*a*]pyrene	None	Landrum & Poore (1983)

Table 30 (contd)

Species	Compound	Biotransformation rate	Reference
Insects (contd)			
Culex pipiens quinquefasciatus	Benzo[a]pyrene	78% after 3 d	Lu et al. (1977)
Somatochlora cingulata	Naphthalene	No information	Correa & Coler (1990)
Bird			
Somateria mollissima L.	Various	Fast for PAH with molecular mass > 252	Broman et al. (1990)
Plant			
Phaseolus vulgaris L.	Anthracene	90% after 30 d	Edwards (1986)

4.2.2.1 Photodegradation in the environment

PAH can be expected to be photodegraded in air and water but to a very low extent in soils and sediments, owing to low light intensity. In natural waters, photodegradation takes place only in the upper few centimetres of the aqueous phase. Information on the photodegradation of PAH in air and water is summarized in Table 31; however, as the testing conditions varied widely, general conclusions cannot be drawn.

PAH are photodegraded in air and water by two processes: direct photolysis by light with a wavelength < 290 nm and indirect photolysis by least one oxidizing agent such as OH, O_3, and NO_3 in air and ROO radicals in water. In general, indirect photolysis—photooxidation—is the more important process. The reaction rates of PAH with airborne OH radicals measured under standard conditions are given in Table 32, which shows that most of the calculated half-lives are one day or less. Under environmental conditions, PAH of higher molecular mass, i.e. those with more aromatic rings, are almost completely adsorbed onto fine particles (see section 4.1.2); this reduces the degradation rate markedly.

Degradation half-lives of 3.7–30 days were reported for the reaction with NO_x of various PAH adsorbed onto soot. The degradation was much slower in the absence of sunlight. PAH did not react significantly with SO_2 (Butler & Crossley, 1981). PAH in wood smoke and gasoline exhaust did not degrade significantly during winter in extreme northern and southern latitudes owing to low temperatures and the low angle of the sun (Kamens et al., 1986a). In summer, however, at a temperature of 20 °C, the half-lives of individual PAH were in the range of 30–60 min (Kamens et al., 1986b). The degradation rate increased further with increasing humidity (Kamens et al., 1991).

In a study of the fate of 18 PAH on 15 types of fly ash, carbon black, silica gel, and alumina, the PAH were stabilized, depending on the colour, which is related to the carbon content: the higher the carbon content, the more stable the PAH. The authors suggested that radiation energy is adsorbed by the organic matter of particulates, and PAH therefore do not achieve the excited state in which they can be degraded (Behymer & Hites, 1988). The half-lives for direct photolysis of various PAH adsorbed onto silica gel are in the range of hours (Vu-Duc & Huynh, 1991).

A two-layer model has been proposed for the behaviour of naturally occurring PAH on airborne particulate matter, in which photooxidation takes place in the outer layer, and much slower, 'dark' oxidation takes place in the inner layer (Valerio et al., 1987). This model is in line with the results of Kamens et al. (1991), who reported that PAH on highly loaded particles degrade more slowly than those on particles with low loads. As PAH occur mainly on particulate matter with a high carbon content, their degradation in the atmosphere is slower than that of PAH in the vapour phase under laboratory

Table 31. Photodegradation of polycyclic aromatic hydrocarbons

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Acenaphthene	Air, particles			Determined in rotary photoreactor with 25 µg/g on:	Behymer & Hites (1985)
			2.0	— silica gel	
			2.2	— alumina	
			44	— fly ash	
	Water	0.23 h^{-1}	3.0	Rate constant in distilled water	Fukuda et al. (1988)
Acenaphthylene	Air, particles			Determined in rotary photoreactor with 25 µg/g on:	Behymer & Hites (1985)
			0.7	— silica gel	
			2.2	— alumina	
			44	— fly ash	
Anthracene	Air, water		0.58	Measured in atmosphere and water from aqueous photolysis rate constant for midday summer sunlight at 35° N	Southworth (1979)
	Air, particles			Determined with 25 µg/g on:	Behymer & Hites (1985)
			2.9	— silica gel	
			0.5	— alumina	
			48	— fly ash	

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Anthracene (contd)	Water			Removal rate constants from water at 25 °C in midsummer sunlight:	Southworth (1979)
		0.004 h^{-1}	*173*	— in deep, slow, somewhat turbid water	
		< 0.001 h^{-1}	*> 700*	— in deep, slow, muddy water	
		0.018 h^{-1}	*38*	— in deep, slow, clear water	
		0.086 h^{-1}	*8*	— in shallow, fast, clear water	
		0.238 h^{-1}	*3*	— in very shallow, fast, clear water	
	Water			Half-lives calclulated from average light intensity over 24 h:	Southworth (1977)
			1.6	— in summer	
			4.8	— in winter	
	Water			Half-lives calculated for direct sunlight at 40° N at midday in midsummer:	Zepp & Schlotzhauer (1979)
			0.75	— near surface water	
			108	— inland water	
			125	— inland water with sediment partitioning	
			0.75	— direct photochemical transformation near water surface	

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Anthracene (contd)	Water	0.66 h^{-1}	1.0	In distilled water	Fukuda et al. (1988)
Benz[*a*]anthracene	Air, particles			First-order daytime decay rate constants with soot particle loading of:	Kamens et al. (1988)
		0.0125 min^{-1}	*0.9*	— 1000–2000 ng/mg	
		0.0250 min^{-1}	*0.5*	— 30–350 ng/mg	
	Air, particles			Determined with ± 25 µg/g on:	Behymer & Hites (1985)
			4.0	— silica gel	
			2.0	— alumina	
			38	— fly ash	
	Water			Calculated rate constant in pure water:	Mill et al. (1981)
		13.4 x $10^{-5}s^{-1}$	*1.4*	— at 366 nm and in sunlight at 23–28 °C, early March	
		2.28 x $10^{-5}s^{-1}$	*8.4*	— at 313 nm with 1% acetonitrile in filter-sterilized natural water	
			5	Early March	

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Benzo[*a*]pyrene	Air, particles			Determined with 25 µg/g on:	Behymer & Hites (1985)
			4.7	silica gel	
			1.4	— alumina	
			31	— fly ash	
	Air, particles			First-order daytime decay rate constants with soot particle loading of:	Kamens et al. (1988)
		0.0090 min^{-1}	*1.3*	— 1000–2000 ng/mg	
		0.0211 min^{-1}	*0.54*	— 30–350 ng/mg	
	Air, particles	< 6.1 x 10^{-4} m/s		Ozonization rate constant measured at 24 °C with O_3 = 0.16 ppm and light intensity of 1.3 kW/m^2	Cope & Kalkwarf (1987)
	Air		0.37–1.1	Estimated	Lyman et al. (1982)
	Air		1	Sunlight in mid-December	Mill & Mabey (1985)
	Air, water			Calculated rate constants for direct photolysis:	Mill et al. (1981)
		3.86 x $10^{-6}s^{-1}$	0.69	— in pure water at 366 nm and in sunlight at 23–28 °C, late January	
		1.05 x $10^{-5}s^{-1}$	1.1	— at 313 nm with 1–20% acetonitrile in filter-sterilized natural water, mid-December	

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Benzo[*a*]pyrene (contd)	Water			Computed near-surface half-life for direct photochemical transformation of a natural water body:	Zepp & Schlotzhauer (1979)
			0.54	– latitude 40° N, midday, midsummer	
			77	– no sediment:water partitioning	
			312	– sediment:water partitioning in a 5-m deep inland water body	
	Air		> 1 Days	Summer Winter	Valerio et al. (1991)
	Methanol		2	Irradiated at 254 nm	Lu et al. (1977)
Benzo[*b*]fluoranthene	Air, particles			First-order daytime decay rate constants with soot particle loading of:	Kamens et al. (1988)
		0.0065 min^{-1}	*1.8*	— 1000–2000 ng/mg	
		0.0090 min^{-1}	*1.3*	— 30–350 ng/mg	
	Air, water		8.7–720	Based on measured rate of photolysis in heptane irradiated with light at > 290 nm	Lane & Katz (1977); Muel & Saguem (1985)

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Benzo[*ghi*]perylene	Air, particles			Determined with 25 µg/g on:	Behymer & Hites (1985)
			7.0	—silica gel	
			2.2	— alumina	
			29	— fly ash	
	Air, particles			First-order daytime photodegradation rate constants for adsorption on wood soot particles in an outdoor Teflon chamber for soot loading of:	Kamens et al. (1988)
		0.0077 min^{-1}	*1.5*	— 1000–2000 ng/mg	
		0.0116 min^{-1}	*1.0*	— 30–350 ng/mg	
Benzo[*k*]fluoranthene	Air, particles			First-order daytime decay constants for soot loading of:	Kamens et al. (1988)
		0.0047 min^{-1}	*2.5*	— 1000–2000 ng/mg	
		0.0013 min^{-1}	*8.9*	— 30–350 ng/mg	
	Air, water		3.8–499	Based on measured rate of photolysis in heptane under November sunlight, adjusted by ratio of sunlight photolysis half-lives in water: heptane	Muel & Saguem (1985)

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Chrysene	Air, particles			Determined with 25 µg/g on:	Behymer & Hites (1985)
			100	— silica gel	
			78	— alumina	
			38	— fly ash	
	Air, particles			First-order daytime decay constants for soot loading of:	Kamens et al. (1988)
		0.0056 min^{-1}	*2.1*	— 1000–2000 ng/mg	
		0.0090 min^{-1}	*1.3*	— 30–350 ng/mg	
	Air, water		4.4	Calculated for direct photochemical transformation near surface of a water body at 40° N at midday in midsummer	Zepp & Schlotzhauer (1979)
	Water		13	Estimated on basis of photolysis in water in winter	Lyman et al. (1982)
Dibenzo[*a*,*h*]anthracene	Air, water		782	Based on measured rate of photolysis in heptane in November sun	Muel & Saguem (1985)
			6	After adjusting ratio of sunlight photolysis in water:heptane	

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Fluoranthene	Air, particles			Determined with 25 µg/g on:	Behymer & Hites (1985)
			74	— silica gel	
			23	— alumina	
			44	— fly ash	
	Air, water		63	Computed, adjusted for approximate winter sunlight intensity	Lyman et al. (1982)
	Air, water			Calculated photochemical transformation near surface of water body:	Zepp & Schlotzhauer (1979)
			21	— at 40° N, midday, midsummer	
			3800	— 5-m deep inland water body with no sediment:water partitioning	
			4800	— with sediment:water partitioning	
	Water		3800	Summer sunlight in surface water	Mill & Mabey (1985)
Fluorene	Air, particles			Determined in rotary photoreactor with 25 µg/g on:	Behymer & Hites (1985)
			110	— silica gel	
			62	— alumina	
			37	— fly ash	

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Naphthalene	Water		13 200	Calculated, 5-m deep inland water	Zepp & Schlotzhauer (1979)
	Water	0.028 h^{-1}	*25*	Half-life in distilled water	Fukuda et al. (1988)
Perylene	Air, particles			Determined with 25 µg/g on:	Behymer & Hites (1985)
			3.9	— silica gel	
			1.2	— alumina	
			35	— fly ash	
	Air, glass	< 4.7×10^{-5} m/s		Ozonization rate constant measured from glass surface at 24 °C with O_3 = 0.16 ppm and light intensity of 1.3 kW/m^2	Cope & Kalkwarf (1987)
Phenanthrene	Air, particles			Determined with 25 µg/g on:	Behymer & Hites (1985)
			150	— silica gel	
			45	— alumina	
			49	— fly ash	
	Water		3	Based on measured aqueous photolysis quantum yields, midday, midsummer, 40° N	Zepp & Schlotzhauer (1979)

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Phenanthrene (contd)	Air, water		25	Adjusted for approximate winter sunlight intensity	Lyman et al. (1982)
	Air, water			Calculated, direct sunlight photolysis, midday, midsummer, 40° N:	Zepp & Schlotzhauer (1979)
			8.4	— near surface water	
			1400	— 5-m deep inland water body with no sediment:water partitioning	
			1650	— with sediment:water partitioning	
	Water	0.11 h^{-1}	*6.3*	Half-life in distilled water	Fukuda et al. (1988)
Pyrene	Air, particles			Determined with 25 µg/ml on:	Behymer & Hites (1985)
			21	— on silica gel	
			31	— on alumina	
			46	— on fly ash	
	Air, particles			Adsorption on airborne particles by sunlight:	Valerio et al. (1991)
			1	— in summer	
			Days	— in winter	
	Air, water	1.014 h^{-1}	0.68	Based on measured aqueous photolysis quantum yields, midday, summer, 40° N	Zepp & Schlotzhauer (1979)

Table 31 (contd)

Compound	Compartment	Photolysis rate constant	Half-life (h)	Comments	Reference
Pyrene (contd)	Air, water		2.04	Based on measured aqueous photolysis quantum yields, adjusted for approximate winter sunlight intensity	Lyman et al. (1982)
	Air, glass	< 1.05×10^{-4} m/s		Ozonization rate on glass surface at 24 °C with O_3 = 0.16 ppm and light intensity of 1.3 kW/m²	Cope & Kalkwarf (1987)
	Water			Calculated, direct sunlight photolysis, midday, midsummer, 40° N:	Zepp & Schlotzhauer (1979)
			0.58	— near surface water	
			100	— 5-m deep inland water body with no sediment:water partitioning	
			142	— with sediment:water partitioning	
	Water		100	Summer sunlight photolysis in surface water	Mill & Mabey (1985)

In order to compare numbers reported only as rate constants, half-lives were estimated from the formula:

$$t_{1/2} = \frac{\ln 2}{k}$$

where $t_{1/2}$ is the half-life and k is the rate constant. The calculated values are reported in italics.

Table 32. Reactions of polycyclic aromatic hydrocarbons with hydroxy radicals

Compound	Oxidation rate constant	Photooxidation half-life (h)	Comments	Reference
Acenaphthene	1×10^{-10}	0.879–8.79	Based on estimated reaction rate constant with hydroxy radical in air	Atkinson (1987)
Acenaphthylene	1.1×10^{-10}	0.191–1.27	Based on estimated rate constant for reaction in air	Atkinson (1987)
Anthracene	1.1×10^{-12} cm^3 molec^{-1}s^{-1}	58–580	Rate constant for gas-phase reaction with hydroxy radicals at 298 ± 1 K, based the relative rate technique for propene	Biermann et al. (1985)
		0.501–5.01	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Benz[*a*]anthracene		0.801–8.01	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Benzo[*a*]pyrene		0.428–4.28	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Benzo[*b*]fluoranthene		1.43–14.3	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Benzo[*ghi*]perylene		0.321–3.21	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Benzo[*k*]fluoranthene		1.1–11	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Chrysene		0.802–8.02	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)

Table 32 (contd)

Compound	Oxidation rate constant	Photooxidation half-life (h)	Comments	Reference
Dibenz[*a*,*h*]anthracene		0.428–4.28	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Fluoranthene		2.02–20.2	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Fluorene	1.3×10^{-11}	6.81–68.1	Based on estimated rate constant for reaction with hydroxy radical in air	Atkinson (1987)
Naphthalene	2.16×10^{-11} cm^3 $molec^{-1}s^{-1}$	*2.7–27*	Rate constant for reaction with hydroxy radicals using relative rate technique at 294 K	Atkinson (1989)
	2×10^{-19} cm^3 $molec^{-1}s^{-1}$	*19–321*	Upper limit was obtained for reaction with O_3	
	2.35×10^{-11} cm^3 $molec^{-1}s^{-1}$	*2.7–27*	Rate constant for gas-phase reaction with hydroxy radicals at 298 K, based on relative rate technique from propene	Biermann et al. (1985)
Phenanthrene	3.4×10^{-11} cm^3 $molec^{-1}s^{-1}$	*2–20*	Rate constant for gas-phase reaction with hydroxy radicals at 298 K, based on relative rate technique for propene	Biermann et al. (1985)
	3.1×10^{-11}	2.01–20.1	Half-life based on measured rate constants for reaction with hydroxy radical in air	Atkinson (1987)

Table 32 (contd)

Compound	Oxidation rate constant	Photooxidation half-life (h)	Comments	Reference
Pyrene		0.802-8.02 h	Based on estimated rate constant for reactions with hydroxy radical in air and with hydroxy radical and ozone	Atkinson (1987); Atkinson & Carter (1984)

To allow comparison when only rate constants are reported, half-lives were estimated from the following formula:

$$t_{1/2} = \frac{\ln 2}{[x] \times k}$$

where $t_{1/2}$ is the half-life, $[x]$ is the concentration of the radical with which the compounds react (i.e. hydroxyl or ozone), and k is the rate constant. The calculated values are reported in italics.

For the concentrations of the radicals, the following ranges of values were used; the lower values are estimates for rural areas and the higher ones for urban areas (Howard et al., 1991):

$[OH]_{air} = 3–30 \times 10^{6}$ radicals/cm^3
$[O_3]_{air} = 3–50 \times 10^{12}$ molecules/cm^3
$[OH]_{water} = 5–200 \times 10^{-17}$ mol/litre
$[RO_2]_{water} = 1–50 \times 10^{-11}$ mol/litre
$[^1O_2]_{water} = 1–100 \times 10^{-15}$ mol/litre

conditions or adsorbed on synthetic materials like alumina and silica gel that have no or a low carbon content.

Formation of nitro-PAH was found from the low-molecular-mass two- to four-ring PAH that occur in the atmosphere, predominantly in the vapour phase. The rate constants range from 5.5×10^{-12} cm³/molecule x s for acenaphthylene to 3.6×10^{-28} cm³/molecule x s for naphthalene, with corresponding half-lives ranging from 6 min to 1.5 years. The yields were 1% or less (Atkinson et al., 1991; Atkinson & Arey, 1994).

The rate of degradation of absorbed individual PAH seems to be independent of their physicochemical characteristics but dependent on their molecular structure. Thus, activated carbon from graphite particles effectively stabilized pyrene, phenanthene, fluoranthene, anthracene, and benzo[*a*]pyrene adsorbed onto coal fly ash against photochemical decomposition, but no stabilization was seen for fluorene, benzo[*a*]fluorene, benzo[*b*]fluorene, 9,10-dimethylanthracene, or 4-azafluorene. The authors suggested that PAH that contain benzylic carbon atoms are less reactive than others (Hughes et al., 1980).

PAH with vinylic bridges appear to degrade by direct photolysis more rapidly than those with only aromatic rings, both in air and in the aquatic environment (Hites, 1981).

In measurements of the photodegradation of benz[*a*]anthracene and benzo[*a*]pyrene, addition of humic acids and purging of the solution with nitrogen reduced the reaction rates significantly (Mill et al., 1981). The authors concluded that light screening and quenching occurred with humic acids. The reduction in rate with exclusion of oxygen was probably due to a decrease in photooxidative processes. The first metabolites were mainly quinones.

4.2.2.2 *Hydrolysis*

PAH are chemically stable, with no functional groups that result in hydrolysis. Under environmental conditions, therefore, hydrolysis does not contribute to the degradation of PAH (Howard et al., 1991).

4.3 Ultimate fate after use

The main sinks for PAH are sediment and soil. The available information indicates that high-molecular-mass PAH are especially persistent in groundwater, soil, and sediment under environmental conditions.

5. ENVIRONMENTAL LEVELS AND HUMAN EXPOSURE

Appraisal

Polycyclic aromatic hydrocarbons (PAH) occur in all environmental compartments. Ambient air, residential heating, and vehicle traffic are the main sources. The levels of individual substances vary over several orders of magnitude but are generally in the range < 0.1–100 ng/m³.

Surface waters are contaminated by PAH mainly through atmospheric deposition, urban runoff, and industrial activities such as coal coking and aluminium production. Apart from highly industrial polluted rivers, the concentrations of individual substances are generally ≤ 50 ng/litre. High concentrations of PAH have been measured in rainwater and especially in snow and fog. The concentrations of PAH in sediments are in the low microgram per kilogram range.

PAH levels in soils near industrial sources (e.g. coal coking) are especially high, sometimes up to grams per kilogram. In contrast, soils contaminated by atmospheric deposition or runoff have concentrations of 2–5 mg/kg of individual PAH, and the concentrations in unpolluted areas are in the low microgram per kilogram range.

PAH have been detected in vegetables but are mainly formed during food processing, roasting, frying, or baking. The highest levels were detected in smoked meat and fish, at up to 200 µg/kg food for individual PAH.

Five-fold increases in the concentrations of PAH in soil have been observed over a 150-year period, although there are indications that the concentrations of some PAH are decreasing. Similar findings have been reported for sediments, perhaps because of measures to reduce emissions.

Aquatic animals are known to adsorb and accumulate PAH. Especially high concentrations were found in aquatic organisms from highly polluted rivers, at levels up to milligrams per kilogram. Of the terrestrial animals, earthworms are a good indicator of soil pollution with PAH. The benzo[a]pyrene concentrations in the faeces of earthworms living in a highly industrialized region were in the low milligrams per kilogram range.

The main sources of exposure for the general population appear to be food and air. The estimated intake of individual PAH in the diet is 0.1–8 µg/d. The main contribution appears to be that of cereals and cereal products, due to the large amounts consumed. In ambient air, the main sources are residential heating and environmental tobacco smoke; exposure to PAH from environmental tobacco smoke in indoor air is estimated to be 6.4 µg/day.

Occupational exposure to PAH occurs via the lung and skin. High exposure occurs during the processing and use of coal and mineral oil products, such as in coal coking, petroleum refining, road paving, asphalt

roofing, and impregnation of wood with creosotes; high concentrations are also found in the air of aluminium production plants and steel and iron foundries. No measurements were available for the primary production and processing of PAH.

5.1 Environmental levels

5.1.1 *Atmosphere*

Relevant data on the occurrence of PAH in ambient air are compiled in Tables 33–36. The concentrations were determined mainly by gas chromatography and high-performance liquid chromatography, usually with enrichment by filtration through a solid sorbent. The amount of particle-bound PAH is therefore given. In studies in which vapour-phase PAH were also sampled, the results for the vapour and particulate phases were combined (for reviews, see Grimmer, 1979; Ministry of Environment, 1979; Grimmer, 1983b; Lee & Schuetzle, 1983; Daisey et al., 1986; Baek et al., 1991; Menichini, 1992a).

5.1.1.1 *Source identification*

Qualitative indications of different sources can be obtained by comparing the PAH profiles, i.e. the ratio between the total PAH concentration and that of a selected PAH, in air with those of samples representative of the emitting sources or by determining PAH that are emitted mainly from a specific source (Menichini, 1992a). Quantitative assignments are difficult to make, however, owing to the complexity of factors that affect the variability of PAH concentrations and profiles.

Measurements were made at selected sources of PAH in the area of Chicago, USA, in 1990–92, in order to identify them: Five samples were taken 100 m directly downwind of a coke plant in an area that was not affected by steel-making facilities, four samples from diesel buses at a parking garage, three samples from petrol vehicles under warm-engine operating conditions at a public parking garage, five samples in heavily travelled tunnels during evening rush hours, and two samples from the roof directly downwind of the chimney of fireplaces burning seasoned oak. The authors give a source distribution pattern in percent related to the total mass of 20 PAH. Naphthalene made by far the largest contribution to petrol engine and coke oven emissions (55 and 89%, respectively). The three-ring compounds acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, and retene were detected in large amounts in diesel motor emissions (56%) and in wood combustion exhausts (69%). The four-ring fluoranthene, pyrene, benz[*a*]anthracene, chrysene, and triphenylene and the five-ring cyclopenta[*cd*]pyrene,

benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*e*]pyrene, and dibenzo[*ghi*]perylene together contributed 28% to diesel engine emissions, 25% to petrol engine emissions, and 20% to wood combustion emissions (Khalili et al., 1995).

The winter levels of PAH are higher than the summer levels (Gordon, 1976; Lahmann et al., 1984; Greenberg et al., 1985; Chakraborti et al., 1988; Catoggio et al., 1989), due to more intensive domestic heating and to meteorological (lower inversions during the winter) and physicochemical factors (temperature-dependent partition between gaseous and particulate phases). The ratios of benzo[*a*]pyrene:CO, in which CO was used as an 'inert' tracer of automotive emissions, in Los Angeles, USA, were higher at night (0.18–0.34) than in the day (0.12–0.14), and substantially more so during winter (0.14–0.34) than in summer (0.12–0.18), consistent with daytime loss of PAH by chemical degradation (Grosjean, 1983).

In studies of sources of PAH at commercial, industrial, and urban sampling sites in Athens, Greece, the effects of wind velocity and thermal inversion were studied. There seemed to be no direct correlation between benzo[*a*]pyrene and lead levels, which would be expected if exhaust from cars run on leaded petrol were the preponderant source of PAH (linear regression coefficient, 0.32–0.38) (Viras et al., 1987).

Differences in the composition of profiles of PAH from different sources can also be standardized by giving the concentrations relative to that of a specific PAH. For particle-bound PAH, benzo[*e*]pyrene has often been used as a reference compound, since it is photochemically stable and found mainly in the particulate phase (Baek et al., 1991).

Cyclopenta[*cd*]pyrene is emitted particularly from petrol-fuelled automobiles (Grimmer et al., 1981c). Fluoranthene, pyrene, benzo[*ghi*]perylene, and coronene are also found in higher concentrations in condensates of vehicle exhausts (Baek et al., 1991). The contribution of vehicles and domestic heating has also been estimated as the ratio of indeno[1,2,3-*cd*]pyrene to benzo[*ghi*]-perylene concentrations. The ratio should be 0.37 for the PAH profile in traffic exhaust and 0.90 for domestic heating (Lahmann et al., 1984; Jaklin & Krenmayr, 1985). In a comparison of the PAH ratios determined in New Jersey, USA, with those reported in the literature for samples collected under similar conditions in street tunnels, the ratios coronene:benzo[*a*]pyrene and benzo[*ghi*]perylene:benzo[*a*]pyrene indicated that vehicle traffic was the major source of PAH during the summer (Harkov et al., 1984).

Measurements in ambient air in North Rhine Westphalia, Germany, in 1990 indicated that coronene is the most characteristic PAH for automobile traffic. At a ratio of benzo[*a*]pyrene:coronene of < 3.5, vehicle traffic is the dominant PAH source, whereas emissions with ratios > 3.5 are influenced by other sources. The benzo[*a*]pyrene levels were 0.66–5.0 ng/m^3, and those of coronene 0.57–2.5 ng/m^3 (Pfeffer, 1994).

In a study of the PAH concentrations during weekdays and weekends in South Kensington, London, United Kingdom, no distinct differences were observed in winter, but the average concentrations were 1.5–2.5 times higher during the week than during the weekends in summer. Likewise, the diurnal variations appeared to be less distinct during winter than summer (Back et al., 1992).

Measurements in streets with high traffic density in Stockholm, Sweden, showed that the concentration of PAH decreased by 25–50% during holidays in comparison with weekdays. Benzo[*a*]pyrene in street air was all particle-bound, while chrysene and lighter PAH occurred both on particles and in the vapour phase (Östman et al., 1991, 1992a,b).

In a study of 15 PAH in the air of various areas in an industrial city in Germany with 700 000 inhabitants, the highest levels were detected in air affected by a coke plant, where benzo[*a*]pyrene was found at 1.4–400 ng/m^3 and cyclopenta[*cd*]pyrene at none detected to 120 ng/m^3. The concentrations measured in air affected by vehicle traffic were 11–110 ng/m^3 benzo[*a*]pyrene and 0.1–440 ng/m^3 cyclopenta[*cd*]pyrene. Within 4 km, the average concentration of 88 ng/m^3 cyclopenta[*cd*]pyrene had dropped to 1.6 ng/m^3. The levels were lower in areas where hand-stoked residential coal heating predominated (0.37 µg/m^3 benzo[*a*]pyrene and none detected to 39 µg/m^3 cyclopenta[*cd*]pyrene) and where oil heating predominated (0.2–66 ng/m^3 and none detected to 15 ng/m^3, respectively). The concentration of PAH was three to four times higher between 7:43 and 10:00 than between 10:00 and 15:46. Benzo[*c*]phenanthrene, cyclopenta[*cd*]pyrene, benzo[*ghi*]perylene, and coronene dominated the PAH in areas with heavy traffic, whereas chrysene, benzo[*b*]fluoranthene, and benzo[*a*]pyrene occurred at the highest concentrations in an area surrounding a coke plant (Grimmer et al., 1981c).

The use of receptor-source apportionment modelling was examined, despite its limited applicability to reactive species, for the PAH profiles of emissions from a variety of sources (Daisey et al., 1986; Pistikopoulos et al., 1990). In one study, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene, and coronene were measured in the ambient air of the centre of Paris, France. The concentrations of PAH varied from 42% in winter to 72% in summer for petrol-fuelled vehicles, from 25 to 40% for diesel-fuelled vehicles, and from about 30 to 2% for domestic heating. The winter-summer differences were due mainly to different emission patterns and not to changes in the rate of decay of PAH (Pistikopoulos et al., 1990). In another study, the contributions of PAH from five sources to ambient air were distinguished by use of fuzzy clustering analysis (Thrane & Wikström, 1984).

The information on PAH levels in ambient air is discussed below according to possible source: background and rural, industrial emissions, and diffuse sources like automobile traffic and residential heating. Attribution of different

studies to these sections was difficult because the sources of PAH emissions are often mixed. For example, Seifert et al. (1986) determined PAH in Dortmund 200 m from a coke plant; this study was deemed to relate to PAH levels resulting from industrial emissions. The concentrations of PAH attributable to mobile sources can be estimated by monitoring near areas with heavy traffic in the summer, but it is difficult to estimate the contribution of home heating, because in winter PAH in ambient air derive from both mobile sources and home heating. Furthermore, emissions from mobile sources may differ in winter from those in the summer because of meteorological and physicochemical factors (Greenberg et al., 1985; see also section 5.1.1.3).

5.1.1.2 *Background and rural levels*

The levels in ambient air of rural areas are summarized in Table 33. Background levels were measured about 25 km from La Paz, Bolivia, at an altitude of 5200 m (Cautreels & van Cauwenberghe, 1977) and on the island of Mallorca, Spain, at an altitude of 1100 m (Simó et al., 1990). The concentrations were generally 0.01–0.1 ng/m^3. The average values in rural areas are usually 0.1–1 ng/m^3. Average concentrations of 0.34 and 0.27 ng/m^3 benzo[*a*]pyrene were measured in two rural areas in Japan in 1989, with a maximum concentration of 1.1 ng/m^3 (Okita et al., 1994).

5.1.1.3 *Industrial sources*

PAH levels in ambient air resulting mainly from industrial emissions are summarized in Table 34. The average concentrations of individual PAH at ground level were 1–10 ng/m^3. In general, aluminium smelters and industrial processes for the pyrolysis of coal, such as coking operations and steel mills, result in higher levels of PAH than most other point industrial sources. Furthermore, the levels of PAH are much higher downwind from major sources than upwind.

The highest levels of individual PAH were measured near an aluminium smelter in Hoyanger, Norway, with maximum concentrations of 10–100 ng/m^3. Phenanthrene was present at very high levels in ambient air contaminated by industrial emissions (Thrane, 1987). In Sundsvall, Sweden, near an aluminium production facility, 310 ng/m^3 phenanthrene, 190 ng/m^3 naphthalene, 120 ng/m^3 pyrene, and 84 ng/m^3 fluorene were detected (Thrane & Wikström, 1984).

The concentration of benzo[*a*]pyrene in ambient air near an oil processing plant in Moscow was up to 13 ng/m^3 (Khesina, 1994). Benzo[*a*]pyrene was detected at 15–120 ng/m^3 and perylene at 3–37 ng/m^3 at 39 measuring stations in the heavily polluted area of Upper Silesia, Poland. The maximum values were 950 ng/m^3 for benzo[*a*]pyrene and 270 ng/m^3 for perylene (Chorazy et al., 1994).

Table 33. Polycyclic aromatic hydrocarbon concentrations (ng/m³) in ambient air of background and rural areas

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Acenaphthene									0.32	6.3–23	
Anthracene		0.004				0.05		0.03	< 0.05	1.2–3.9	ND–0.05
Anthanathrene				0.004–0.16		0.08	0.07			ND–0.2	ND–0.04
Benz[*a*]anthracene		0.005	0.12			0.4		0.40	0.07	1.8–3.2	0.16–0.39
Benzo[*a*]fluorene										0.8–3.3	
Benzo[*a*]pyrene		0.006	0.005	0.002–0.12	0.33/0.47	0.6	ND–0.52	0.45	0.08	0.8–2.5	0.41–0.45
Benzo[*b*]fluoranthene			0.02					1.2			0.45–0.58
Benzo[*b*]fluorene								0.24		0.5–2.4	
Benzo[*c*]phenanthrene											0.15–0.20
Benzo[*e*]pyrene		0.022	0.006	0.007–0.26		0.6		0.59		1.8–5.8	0.44–0.65
Benzo[*ghi*]fluoranthene										ND–0.2	
Benzo[*ghi*]perylene		0.009	0.002	0.005–0.40		0.6	ND–0.58			1.4–3.0	0.89–1.4
Benzo[*k*]fluoranthene			0.02	0.002–0.088				0.48			0.17–0.25
Chrysene			0.07[a]					1.0			0.13–0.19
Coronene				0.005–0.23		0.24	ND–0.22			0.4–0.9	0.16–0.26
Cyclopenta[*cd*]pyrene						0.2					0.16–0.39
Dibenzo[*a,h*]pyrene						0.14					0.02–0.07
Dibenzo[*a,l*]pyrene								0.53			
Fluoranthene	0.041	0.030	0.18		0.20/0.26	1.2	ND	0.93	1.3	11–47	0.19–0.23
Fluorene			0.45						0.66	14–32	
Indeno[1,2,3-*cd*]pyrene		0.006	0.02			0.7		0.72			0.43–0.65
1-Methylphenanthrene						0.09				0.7–2.8	

Table 33 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Naphthalene									ND	3.0-98	
Perylene				0.001- 0.026		0.09		0.08		ND-0.4	
Phenanthrene		0.026	2.66			0.4	ND- 0.43		4.2	26-70	ND-0.03
Pyrene	0.034	0.024	0.34	0.010- 0.15	0.15/0.15	1.3	ND	0.60	0.73	8.8-26	0.16-0.26

ND, not detected; /, single measurements; **[1]** About 25 km from La Paz, Bolivia, at 5200 m (Cautreels & van Cauwenberghe, 1977); **[2]** Mallorca, Spain, 1989 (Simó et al.,1991); **[3]** Lake Superior, USA, 1986; sum of vapour and particulate phases (Baker & Eisenreich,1990); **[4]** Latrobe Valley, Australia, (Lyall et al.,1988); **[5]** Belgium, (Van Vaeck et al.,1980); **[6]** Denmark (Nielsen, 1984); **[7]** Western Germany, 1981 (Pflock et al.,1983); **[8]** Oostvoorne, Netherlands, (De Raat et al.,1987b); **[9]** Canada, 1989–91 (Environment Canada, 1994); **[10]** Sidsjön, Sweden, 1980–81, sum of vapour and particulate phases (Thrane & Wikström, 1984); **[11]** Folkestone, Ashford, United Kingdom, 1986 (Baek et al., 1992)

[a] With triphenylene

Analysed by high-performance liquid chromatography or gas chromatography; only particulates sampled, unless otherwise stated

Table 34. Polycyclic aromatic hydrocarbon concentrations (ng/m³) in ambient air near industrial emissions

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Acenaphthene					23	9.8–372	15–122		3.7		
Acenaphthylene					747				0.01		
Anthracene		2.9/3.4			158	4.5–6.1	4.1–43	0.12/0.15	0.01–3.4		0.08–0.19
Anthanthrene	0.001/3.0	0.2/1.1				ND–3.0		0.15/0.15			0.13–0.22
Benz[*a*]anthracene		0.28/1.2			7.6	2.0–158	2.5–58	0.8/3.1	0.02–1.2		1.3–4.7
Benzo[*a*]fluorene						1.1–179					
Benzo[*a*]pyrene	0.002/1.5	0.5/3.5	25/37	6.3–6.7	5.3	1.1–61	2.1–36	0.14/0.11	0.20–0.11	1.8–3.1	1.1–2.6
Benzo[*b*]fluoranthene		0.9/1.8			4.8						2.7–6.4
Benzo[*b*]fluorene						0.7–122					0.61–1.4
Benzo[e]pyrene	0.004/1.4	1.8/3.2			11.6	2.5–86					1.3–3.1
Benzo[*ghi*]fluoranthene						ND-0.5		0.26/0.35			
Benzo[*ghi*]perylene	0.003/1.5	4.2/7.1			0.7	2.2–45		0.35/0.33	0.25		
Benzo[*j*]fluoranthene		0.3/0.8									
Benzo[*k*]fluoranthene	0.001/0.67	0.3/1.3			8.0						1.0–2.2
Chrysene		1.6/3.8			14.7			0.22/0.29	0.01–1.6		2.5–7.5
Coronene	0.003/1.5	3.2/2.8		1.3–1.5	ND	0.6–9.0		0.25/0.26			
Cyclopenta[cd]pyrene					2.2						
Dibenzo[*a,h*]pyrene					ND				0.77		
Dibenzo[*a,l*]pyrene											1.0–1.5
Fluoranthene		0.8/3.4			88.3	20–812	22–272	0.12/0.20	0.02–10		2.3–3.3
Fluorene					502	27–419	16–46		0.02–0.86		
Indeno[1,2,3-*cd*]pyrene		0.4/0.3			1.1	3.8–38		0.28/0.27	0.10–7.7		1.4–2.4
1-Methylphenanthrene						2.5–58					

Table 34 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Naphthalene					22 400	9.0–193	3.1–26	0.03–0.06			
Perylene	0.001/ 0.2	0.3/1.2				0.1–8.3		0.05/0.05	22		0.23–0.61
Phenanthrene					500	54–1760	58–390	0.11/0.16	0.02–152		
Pyrene		1.4/3.8			56.3	16–491	14–207	0.17/0.35	0.006–28		1.6–2.1

ND, not detected; /, single measurements; **[1]** Three sampling sites near various industries in Latrobe Valley, Australia (Lyall et al., 1988); **[2]** Near various industries, USA, 1971–72 (Gordon & Bryan, 1973); **[3]** Near a coke plant, Dortmund, Germany, 1982–83 (Seifert et al., 1986); **[4]** Near a coke plant, Dortmund, Germany, 1989 (Buck, 1991); **[5]** 100 m directly downwind of a coke plant, Chicago, USA,1990–92 (Khalili et al., 1995); **[6]** Near aluminium smelters, Norway and Sweden, 1980–82 (analytical method not given) (Thrane, 1987); vapour and particulate phase (Thrane & Wikström, 1984); **[7]** Near aluminium smelter, Canada, 1989–91 (Environment Canada, 1994); **[8]** Near incineration plant, Sweden (Colmsjö et al., 1986a,b); **[9]** Near refinery, USA, 1981–83 (Karlesky et al., 1987); **[10]** Brown coal industry area, western Germany, 1983 (Seifert et al., 1986); **[11]** Near harbours, Netherlands (De Raat et al., 1987b)

Analysed by high-performance liquid chromatography or gas chromatography; only particulates sampled, unless otherwise stated

In Ontario, Canada, up to 140 ng/m³ benzo[*k*]fluoranthene, 110 ng/m³ perylene, 110 ng/m³ benzo[*a*]pyrene, 90 ng/m³ benzo[*ghi*]perylene, and 43 ng/m³ fluoranthene were found near a steel mill (Potvin et al., 1980). The benzo[*a*]pyrene concentrations near coke ovens in urban areas of the USA were more than double those in urban areas without coke ovens (Faoro & Manning, 1981). These results are consistent with those of Grimmer et al. (1981c), who detected maximum levels of benzo[*a*]pyrene, chrysene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, and benzo[*k*]fluoranthene in the area surrounding a coke plant.

The PAH concentrations in ambient air 900 and 2500 m from a municipal incineration plant were of the same order of magnitude, and no significant contribution from the plant to the ambient PAH concentrations was observed (Colmsjö et al., 1986a).

The PAH levels in an industrial area of Ahmedabad City, India, were significantly higher than those in a residential area. The highest levels were found during winter, and the rate of degradation of airborne PAH was predicted to be lowest in the monsoon season. The most striking finding was the high concentration of dibenz[*a,h*]anthracene in urban air (5.3–23 ng/m³) (Raiyani et al., 1993a). The limited resolution of PAH may have resulted in overestimation: for instance, the concentrations of benzo[*ghi*]perylene and indeno[1,2,3-*cd*]pyrene reported are one order of magnitude higher than that of dibenz[*a,h*]anthracene.

5.1.1.4 *Diffuse sources*

A special situation of local importance was the pollution of ambient air in Kuwait after the war in the Persian Gulf, due to burning of oil fields. The mean concentrations of benzo[*a*]pyrene at three sampling sites were 0.27–9.2 ng/m³, and the maximum was 26 ng/m³ (Okita et al., 1994). These values are within the range of those detected in urban areas (see below).

(a) Motor vehicle traffic

The concentrations of PAH in the ambient air of various urban areas are listed in Table 35. The average levels of individual PAH were 1–30 ng/m³. Relatively high concentrations of benzo[*a*]pyrene, benzo[*ghi*]perylene, phenanthrene, fluoranthene, and pyrene were measured.

Total PAH concentrations of 43–640 ng/m³ were measured in London, United Kingdom, in 1991, nearly 80% of which consisted of phenanthrene, fluorene, and fluoranthene; benzo[*a*]pyrene and benz[*a*]anthracene were present at 1% or less (Clayton et al., 1992).

In Delft, the Netherlands, benzo[*a*]pyrene levels of up to 140 ng/m³ were measured on a foggy day with low wind velocity near a major road. High concentrations of pyrene (220 ng/m³), benzo[*ghi*]perylene (130 ng/m³), and

Table 35. Polycyclic aromatic hydrocarbon concentrations (ng/m^3) in ambient air of urban areas

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Acenaphthene						0.4–101			2.7–6	
Acenaphthylene						0.9–39			4.4–130	
Anthracene	34	0.6–36				0.3–2.1			3.5–25	
Anthanthrene	2.5	0.1–4.7		30		< 0.1–0.6	0.003–0.76			
Benz[*a*]anthracene	10	0.3–27			1.2–13	0.2–1.4	0.10–25		0.3–7.6	
Benzo[*a*]fluorene						0.1–0.9			0.8–6.9	
Benzo[*a*]pyrene	9.3	0.3–20		29	1.2–11	< 0.1–1.9	0.074–15		0.2–5.7	
Benzo[*b*]fluoranthene				43			1.0–36			
Benzo[*b*]fluorene						0.1–0.8			0.6–7.3	
Benzo[*c*]phenathrene	4.0	0.2–5.0								
Benzo[*e*]pyrene	8.4	0.4–17		16	1.7–15	< 0.1–1.2	0.40–27		0.4–6.5	
Benzo[*ghi*]fluoranthene	12	0.3–5.0				0.1–1.5			0.5–7	
Benzo[*ghi*]perylene	14	0.5–12	1.6/13	27	2.1–11	0.2–3.5	0.45–31	0.9–2.4	0.6–18	
Benzo[*j*]fluoranthene							0.17–13			
Benzo[*k*]fluoranthene				23			0.29–25			
Chrysene						0.3–2.5	0.56–29	3.6–5.6		3.3
Coronene	10	0.3–5.5		12	0.88–2.0	0.1–2.4	0.22–3.3		0.4–19	
Cyclopenta[*cd*]pyrene	11	0.1–4.8		71		< 0.1–1.1			0.1–6	
Dibenzo[*a,h*]pyrene					0.22–3.4		0.29–2.8			

Table 35 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Fluoranthene	72	6.2–108	0.40/14			1.4–10	0.80–14	1.3–2.0	6.9–38	15
Fluorene						1.3–61			16–86	
Indeno[1,2,3-*cd*]pyrene	8.6	0.4–12		31		< 0.1–2.9	0.39–30		0.4–7.6	
1-Methylphenanthrene						0.3–2.5			5–16	
Naphthalene									14–63	
Perylene	2.3	0.1–4.3		4.8		< 0.1–0.4	0.011–4.4		0.1–1.3	
Phenanthrene	153	18–223				3.6–41			32–105	111
Pyrene	74	2.9–67	0.34/12			1.2–5.5	0.34–10		5.5–45	20
Triphenylene							0.15–6.9			

ND, not detected; /, single measurements; **[1]** Vienna, Austria, 1983–84; vapour and particulate phase (Jaklin & Krenmayr, 1985); **[2]** Linz, Austria, 1985; vapour and particulate phase (Jaklin et al., 1988); **[3]** Antwerp, Belgium (Van Vaeck et al., 1980); **[4]** Berlin, western Germany, 1984–85 (Seifert et al., 1986); **[5]** Rhein/Ruhr area, western Germany, 1985–88; analytical method not stated (Buck et al., 1989); **[6]** Kokkola, Finland (Pyysalo et al., 1987); **[7]** St Denis, France, 1979–80 (Muel & Saguem, 1985); **[8]** Various cities, Greece, 1984–85 (Viras et al., 1987); **[9]** Oslo, Norway, 1981–83, vapour and particulate phase (Larssen, 1985); **[10]** Barcelona, Spain, 1988–89, vapour and particulate phase (Albaiges et al., 1991)

Analysed by high-performance liquid chromatography or gas chromatography; only particulates sampled, unless otherwise stated

Table 35 (contd)

Compound	[11]	[12]	[13]	[14]	[15]	[16]	[17]	[18]	[19]	[20]
Acenaphthene					0.07–3.58	0.05–31.1				
Acenaphthylene	9.1	0.8					0.9			
Anthracene	21	1.4		2.8	0.01–8.28	0.20–39.8	0.1–0.9		ND–4.8	6.1/11
Anthanthrene				0.63						
Benz[*a*]anthracene	4.1	0.4		1.4	0.24–10.6	0.12–18.5	0.2–5.8	5–21	0.07–2.1	
Benzo[a]fluorene	5.0	0.7								
Benzo[a]pyrene	2.9	0.2	0.99/1.4	1.6	0.01–7.02	0.18–13.7	0.3–3.4	1–17	0.04–3.2	0.6/1.6
Benzo[*b*]fluoranthene				1.8	0.01–3.04	0.13–14.8	0.2–3.7	5–30	0.10–3.7	
Benzo[c]phenathrene				2.8						
Benzo[*e*]pyrene	3.5	0.4	1.1/2.0	2.3						2.1/2.1
Benzo[*ghi*]fluoranthene	7.3	0.8								
Benzo[*ghi*]perylene	6.6	0.5	2.9/3.3	3.3	0.02–6.90	0.15–85.3				
Benzo[*k*]fluoranthene				0.75		0.23–16.5	0.3–0.8	3–22	0.07–0.85	
Chrysene	5.1	0.8		1.6	0.04–4.97	0.13–24.3	0.2–5.5		ND–2.3	
Coronene	4.1	0.3	2.4/1.7	1.7	0.02–3.72	0.17–6.92			ND–16	
Cyclopenta[*cd*]pyrene	3.9	0.11		4.1						
Dibenz[*a,h*]pyrene				0.12						

Table 35 (contd)

Compound	[11]	[12]	[13]	[14]	[15]	[16]	[17]	[18]	[19]	[20]
Fluoranthene	24	3.9		3.5		2.03–62.4	2.2–23	14–54	0.24–2.0	8.0/9.7
Fluorene					0.07–27.6	0.07–161				
Indeno[1,2,3-*cd*]pyrene	3.8	0.5		1.6			0.3–4.4	4–24		
Naphthalene										15/75
Perylene	1.0	0.1								0.2/0.5
Phenanthrene	76	11		5.1	0.06–111	2.25–492	0.1–2.4			78/81
Pyrene	28	3.2		3.8	0.39–17.4	0.33–64.4	0.1–7.5		0.48–3.6	8.0/12
Triphenylene										

ND, not detected; /, single measurements; **[11]** Stockholm, Sweden, April 1991; vapour and particulate phases (Östman et al.,1992a,b); **[12]** Stockholm, Sweden; 1992 vapour and particulate phases (Östman et al.,1992a,b); **[13]** London, United Kingdom, 1985–87 (Baek et al.,1992); **[14]** London, United Kingdom, 1987; vapour and particulate phases (Baek et al.,1992); **[15]** Manchester, United Kingdom, 1990–91; vapour and particulate phases (Clayton et al.,1992); **[16]** Various cities, United Kingdom, 1991–92; vapour and particulate phases (Halsall et al.,1994); **[17]** Lake Baikal shore, Russian Federation, 1993–94 (Grachev et al.,1994); **[18]** Zagreb, Croatia, 1977–82; determined by thin-layer chromatography and fluorescence detector (Bozicevic et al.,1987); **[19]** Los Angeles, USA, 1981–82 (Grosjean, 1983); **[20]** Los Angeles basin, USA, 1986; vapour and particulate phases (Arey et al.,1987)

Table 35 (contd)

Compound	[21]	[22]	[23]	[24]	[25]	[26]	[27]	[28]	[29]	[30]
Acenaphthene		3.3–9.0	0.06–5.2						0.6	
Acenaphthylene		< 11–47							1.9	
Anthracene		1.9–4.5	0.45–3.8				0.17–0.57	0.12–0.52	0.2	2.5–5.5
Anthanthrene				0.006–3.3	1–11					
Benz[*a*]anthracene	0.07–1.4	0.19–0.40	0.19–4.4				0.99–7.0	0.37–1.7	1.9	20–66
Benzo[a]fluorene										1.8–6.3
Benzo[a]pyrene	0.11–1.6	ND–0.03	0.09–1.7	0.006–1.8	8–38		1.6–8.4	ND–2.3	3.4	30–120
Benzo[*b*]fluoranthene	0.17–1.7						3.1–12		3.0	109–200
Benzo[*b*]fluorene							0.19–0.94			
Benzo[*e*]pyrene	0.03–11	ND–0.04		0.016–2.3	4–19		2.7–9.0		2.3	49–182
Benzo[*ghi*]fluoranthene	0.12–1.3									
Benzo[*ghi*]perylene	0.24–2.7			0.027–4.7	11–33		3.2–12		3.4	34–141
Benzo[*j*]fluoranthene	0.08–1.1									22–66
Benzo[*k*]fluoranthene	0.09–0.97			0.005–0.85			1.8–7.7		2.7	
Chrysene	0.22–5.3	0.38–0.57			3–15			0.29–1.4	2.4	
Coronene	0.14–1.6			0.020–2.3	5–16					
Dibenzo[*a,e*]pyrene	0.06–2.7									
Dibenzo[*a,h*]pyrene							0.46–1.2		5.3–23	
Dibenzo[*a,l*]pyrene	0.05–0.35									

Table 35 (contd)

Compound	[21]	[22]	[23]	[24]	[25]	[26]	[27]	[28]	[29]	[30]
Fluoranthene		5.7–10	1.6–11			14–79	1.5–8.3		1.0	11–26
Fluorene		7.4–14	0.94–5.5				0.08–0.15	0.31–1.2	2.8	
Indeno[1,2,3-*cd*]pyrene	0.20–2.9				6–24		2.6–12		3.1	
Naphthalene		280–940	ND					4.5–13		
Perylene	0.01–0.15			0.001–0.24	2–9		0.51–1.2			
Phenanthrene		21–35	2.2–35				0.79–2.6	0.52–2.4	0.7	12–21
Pyrene	0.12–2.8	4.8–10	1.4–6.9	0.008–0.66		16–69	1.5–9.0	0.46–4.0	3.8	20–44
Triphenylene										22–60

ND, not detected; /, single measurements; **[21]** New Jersey, USA, 1981–82 (Greenberg et al.,1985); **[22]** Portland, Oregon, USA, 1984 (Ligocki et al.,1985); **[23]** Urban area (not specified), Canada, 1989–91 (Environment Canada,1994); **[24]** Latrobe Valley, Australia (Lyall et al.,1988); **[25]** Christchurch, New Zealand, 1979 (Cretney et al.,1985); [26] Osaka, Japan, 1977–78; vapour and particulate phases (Yamasaki et al.,1982); **[27]** Osaka, Japan, 1981–82 (Matsumoto & Kashimoto, 1985); **[28]** La Plata, Argentina, 1985 (Catoggio et al.,1989); **[29]** Ahmedabad City, India, 1984–85 (Raiyani et al.,1993a); **[30]** Calcutta, India, 1984 (Chakraborti et al.,1988)

Table 35 (contd)

Compound	[31]	[32]	[33]	[34]	[35]	[36]	[37]	[38]	[39]	[40]
Acenaphthene									4.5	
Anthracene			14–16	2.5	1.8			ND–34	8.7–23	
Anthanthrene		0.15–0.63				0.001–0.21				2–24
Benz[*a*]anthracene	2.9–4.8		99–139	23	6.5	0.028–4.8		3.1–9.8		
Benzo[*a*]pyrene	3.8–5.5	0.005–1.3	67–73	15	5.6	0.023–4.6	Trace–9.3	ND–44	1.9–7.7	19–72
Benzo[*b*]fluoranthene		1.0–3.1	130–133			0.46–16				
Benzo[*b*]fluorene		0.07–0.18								
Benzo[*c*]phenathrene			33–37							
Benzo[*e*]pyrene	5.5–7.4	0.016–3.3	96	19	9.1	0.18–8.8	0.17–4.2	ND–370		9–41
Benzo[*ghi*]fluoranthene	3.0–4.9	0.024–0.98	30–33							
Benzo[*ghi*]perylene	7.0–13	0.004–3.2	49–61	12	7.9	0.21–12		ND–74		11–49
Benzo[*j*]fluoranthene						2.6–5.5				
Benzo[*k*]fluoranthene	3.4–5.0					0.12–7.4				
Chrysene	4.3–6.5	0.34–0.49	237–261	43	16	0.22–8.9	0.22–6.4	ND–170		7–71
Coronene		0.002–1.4	14–16	3.1	2.8	0.14–2.1	Trace–2.1	8–96		4–18
Cyclopenta[*cd*]pyrene			ND	3.1	1.6					
Dibenzo[*a,h*]pyrene						0.012–0.98				

Table 35 (contd)

Compound	[31]	[32]	[33]	[34]	[35]	[36]	[37]	[38]	[39]	[40]
Fluoranthene	3.4–4.9	0.14–1.2				0.32–8.6		8–520	15–51	
Fluorene									15–26	
Indeno[1,2,3-*cd*]pyrene	5.1–9.1	0.022–2.0	57	11	5.5	0.16–9.6				9–43
Naphthalene									44	
Perylene		0.01–0.20	7.6–10			0.004–0.88		ND–28		3–21
Phenanthrene		0.002–1.1						4–170	50–271	
Pyrene	3.6–6.6	0.002–0.58				0.13–6.7	0.21–8.6	ND–540	12–49	
Triphenylene	1.4–1.9	0.07–0.24				0.11–2.9		ND–50		

ND, not detected; /, single measurements; **[31]** Various cities, China (Chen et al.,1981); **[32]** Various cities, China, 1986–88; determined by thin-layer chromatography and gas chomatography–mass spectroscopy (Chang et al., 1988; Simoneit et al., 1991); **[33]** Various locations with predominantly coal heating; Germany (analytical method not given) (Grimmer, 1980); **[34]** Essen, Germany, predominantly coal heating, 1978–79 (Buck, 1983); **[35]** Essen, Germany, predominantly oil heating, 1978–79 (Buck, 1983); **[36]** Antony, France, 1979–80 (Muel & Saguem, 1985); **[37]** Sutton Coldfield, United Kingdom, 1976–78 (Butler & Crossley, 1982); **[38]** Barrow, USA, fossil fuel combustion area, 1979 (Daisey et al., 1981); **[39]** Wood-heating area, Canada, 1989–91 (Environment Canada, 1994); **[40]** Christchurch, New Zealand, 1979 (Cretney et al., 1985)

coronene (21 ng/m^3) were also found. At border crossings between the Netherlands and Germany on days with heavy traffic, the maximum levels of individual PAH were 1-54 ng/m^3 (Brasser, 1980).

PAH concentrations were determined in the centre of Paris, France, at the top of a 55-m tower and thus less likely than ground-level samples to be affected by traffic emissions and street dust; they can therefore be considered to be homogeneous and representative. The maximum levels found were 98 ng/m^3 benzo[*ghi*]perylene, 60 ng/m^3 indeno[1,2,3-*cd*]pyrene, 34 ng/m^3 coronene, 28 ng/m^3 benzo[*b*]fluoranthene, 13 ng/m^3 benzo[*a*]pyrene, and 13 ng/m^3 benzo[*k*]fluoranthene (Pistikopoulos et al., 1990).

The average concentration of individual PAH in particulate and vapour phases during a nine-day photochemical pollution episode in California, USA, in 1986 was 1 ng/m^3. The maximum levels of acenaphthene, acenaphthylene, fluorene, and phenanthrene ranged from 30 to 64 ng/m^3 (Arey et al., 1991).

In 1989, the average benzo[*a*]pyrene concentrations in five Japanese cities (Sapporo, Tokyo, Kawasaki, Nagoya, and Osaka) were 1.2–3.1 ng/m^3. A maximum level of 15 ng/m^3 was detected in Tokyo (Okita et al., 1994). A detailed examination was undertaken of the molecular composition of PAH in street-dust samples collected from the Tokyo metropolitan area. Unsubstituted ring systems (i.e. parent PAH) ranging from phenanthrene with three rings to benzo[*ghi*]perylene with six rings were the primary components, three- and four-ring PAH (i.e. phenanthrene, fluoranthene, and pyrene) predominating. The concentrations of total PAH were of the order of a few micrograms per gram of dust. On the basis of the PAH profile, it was suggested that PAH in the dust of busy streets arose mainly from automobile exhausts, while residential areas received a greater contribution from stationary sources. In both types of dust, asphalt was thought to contribute to only a minor extent (Takada et al., 1990). Giger & Schaffner (1978) had come to the same conclusion some 20 years earlier.

Benzo[*a*]pyrene was detected in ambient air in Moscow, Russian Federation, at concentrations of 5.4 ng/m^3 at a regular traffic site and 20 ng/m^3 at a crossroads with heavy traffic (Khesina, 1994).

(b) Road tunnels

In road tunnels, the concentrations of individual PAH were usually 1–50 ng/m^3 (Table 36). Higher levels were reported in tunnels in western Germany, with concentrations of 84 and 96 ng/m^3 cyclopenta[*cd*]pyrene (Buck (1983) and 76 ng/m^3 (Brasser, 1980) and 110 ng/m^3 pyrene (Benner et al., 1989).

PAH were found at levels of up to 4 ng/m^3 in an underground bus terminal in Stockholm, Sweden; and 21 ng/m^3 fluoranthene, 11 ng/m^3 pyrene, and 8.1 ng/m^3 phenanthrene were found in a subway station (Colmsjö et al., 1986b).

Table 36. Polycyclic aromatic hydrocarbon concentrations (ng/m³) in ambient air polluted predominantly by vehicle exhaust

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Acenaphthene									168		
Acenaphthylene					32				445		
Anthracene	8.6/9.8		2.3	55				0.6–12	177		
Anthanthrene				7.2		1 500				0.1–4.5	2–82
Benz[*a*]anthracene	37/44		0.6–1.9	16	20	12 000	102	1.9–2.9	90.2		
Benzo[*a*]fluorene					18	2 800					
Benzo[*a*]pyrene	30	2–14	0.2–0.8	16	12	9 600	66	1.3–26	62.6	0.1–14	1–57
Benzo[*b*]fluoranthene				2.3	8.8	12 000			43.6		
Benzo[*e*]pyrene	28/32				11	9 600	69	1.5–19	55.5	0.1–12	3–43
Benzo[*ghi*]fluoranthene				29	18			3.2–26			
Benzo[*ghi*]perylene	40/47	4–16	0.4–2.6	44	30	19 000	85	1.8–18	17.0	0.6–27	20–213
Benzo[*k*]fluoranthene				8.1	9.7	9 000			41.2		
Chrysene	54/58			25	15	9 500			77.9		
Coronene	26/27	2–17	0.3–1.1	29	20	7 500		1.0–10	ND	0.3–14	9–156
Cyclopenta[*cd*]pyrene	84/96			40	31			7.6–65	100		
Dibenzo[*a,h*]pyrene									14.7		
Fluoranthene					35	83	93	6.4–69	117		
Fluorene									406		
Indeno[1,2,3-*cd*]pyrene	18/22		0.3–1.3	16	13	9 400		0.3–15	20.0		6–70
1-Methylphenanthrene								2.6–43			
Naphthalene									8030		
Perylene				3.4	3.1	1 500					1–18

Table 36 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Phenanthrene				8.1	243			4.4–56	300		
Pyrene		33–114		47	122	16 000	120	9.7–76	193	0.2–29	

ND, not detected; /, single measurements; **[1]** Street tunnel (location not specified), western Germany, 1978–79 (Buck, 1983); **[2]** Coen Tunnel, Netherlands (Brasser, 1980); **[3]** Street tunnel in Lincoln, Netherlands, 1981 (Kebbekus et al., 1983); **[4]** Klara Tunnel, Sweden, 1983 (Colmsjö et al., 1986b); **[5]** Söderleds Tunnel, Sweden, 1991; vapour and particulate phases (Östman et al., 1991); **[6]** Craeybeckx Highway Tunnel, Belgium, 1991 (De Fré et al., 1994); **[7]** Baltimore Harbor Tunnel, USA, 1975 (Fox & Staley, 1976); **[8]** Baltimore Harbor Tunnel, USA, 1985–86 (Benner et al., 1989); **[9]** Heavily travelled tunnel, Chicago area, USA, 1990–92 (Khalili et al., 1995); **[10]** Diesel bus garage, United Kingdom, 1979 (Waller et al., 1985); **[11]** Inside car park, New Zealand (Cretney et al., 1985)

Analysed by high-performance liquid chromatography or gas chromatography; only particulates sampled, unless otherwise stated

Very high concentrations of PAH were found in the air of the Craeybeckx Highway Tunnel in Belgium, which was used daily by an average of 45 000 vehicles, of which 60% were petrol-fuelled passenger cars, 20% diesel-fuelled cars, and 20% trucks. Of the cars, only 3% had three-way catalysts (De Fré et al., 1994).

(c) Residential heating

The PAH levels in ambient air resulting mainly from residential heating are included in Table 35, as the source cannot be identified properly (see section 5.1.1.1).

The use of wood and coal for heating was the source of high levels of benzo[*a*]pyrene in Calcutta, India (up to 120 ng/m^3; Chakraborti et al., 1988). The concentrations of individual PAH in Calcutta ranged from 1.3 to 200 ng/m^3, the highest levels being those of benzo[*e*]pyrene, benzo[*ghi*]perylene, and benzo[*b*]fluoranthene. The average levels of individual PAH resulting from domestic heating in Christchurch, New Zealand were 1–210 ng/m^3, benzo[*ghi*]perylene and coronene showing the highest levels (Cretney et al., 1985), and up to 43 ng/m^3 were measured in Essen-Vogelheim, Germany (Buck, 1983). High concentrations of individual PAH were determined in a residential area heated primarily by coal, with levels of up to 260 ng/m^3 chrysene, benz[*a*]anthracene, and benzo[*b*]fluoranthene (Grimmer, 1980).

The following PAH levels were measured on a roof directly downwind of the chimney of a fireplace burning seasoned oak in the Chicago area, USA: 1.8 µg/m^3 acenaphthylene, 0.40 µg/m^3 naphthalene, 0.35 µg/m^3 anthracene, 0.22 µg/m^3 phenanthrene, 0.20 µg/m^3 benzo[*a*]pyrene, 0.20 µg/m^3 benzo[*e*]pyrene, 0.13 µg/m^3 fluorene, 0.10 µg/m^3 pyrene, 0.096 µg/m^3 fluoranthene, 0.052 µg/m^3 acenaphthene, 0.045 µg/m^3 benzo[*k*]fluoranthene, 0.033 µg/m^3 chrysene, 0.030 µg/m^3 cyclopenta[*cd*]pyrene, 0.023 µg/m^3 benzo[*b*]fluoranthene, and 0.019 µg/m^3 benz[*a*]anthracene. The levels of indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, and coronene were below the limit of detection (Khalili et al., 1995).

In a comparison of the PAH concentrations in ambient air in eastern and western Germany, the concentrations in rural areas were 3–12 times higher in eastern than in comparable western parts of the country. The PAH profiles were slightly different: the concentrations of the lower-boiling-point PAH fluoranthene and pyrene were 110 and 68 ng/m^3 in eastern and 36 and 28 ng/m^3 in western Germany. The differences may be due to the different types of brown and hard coal burnt (Jacob et al., 1993a).

In 1991, PAH were determined in the air of Berchtesgaden, a national park in Germany, and of the Oberharz (Ministry of Environment, 1993). The concentration of phenanthrene, fluoranthene, and pyrene (about 14 ng/m^3) in the Oberharz was two to three times higher than in Berchtesgaden, due to the

use of brown coal for heating. The levels of the other PAH were of the same order of magnitude: benz[*a*]anthracene and benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, about 5 ng/m^3; and benzo[*ghi*]fluoranthene, benzo[*c*]phenanthrene, benzo[*e*]pyrene, benzo[*a*]-pyrene, indeno(1,2,3-*cd*)pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, anthanthrene, and coronene, < 1 ng/m^3.

A model calculation for Germany showed that 5000 oil-heated houses contributed to the pollution of ambient air by benzo[*a*]pyrene to the same extent as one coal-heated house. It was assumed that one German household consumes annually about 5000 litre of heating oil, producing a maximum of 5 mg of benzo[*a*]pyrene (about 1 µg/litre combusted oil). On the basis of a consumption of a similar amount of hard coal, the same household would have an output of 25 g benzo[*a*]pyrene (about 5000 µg/kg combusted hard coal) annually (J. Jacob, 1994, personal communication).

5.1.2 *Hydrosphere*

PAH are found in the hydrosphere (Borneff & Kunte, 1983; Müller, 1987), mostly as a result of urban runoff, with smaller particles from atmospheric fallout and larger ones from asphalt abrasion (Hoffman et al., 1984). Long-range atmospheric transport of PAH has been well documented in different countries (Lunde & Bjørseth, 1977; see also section 4.1.2). After PAH are emitted into the atmosphere, for example in motor vehicle exhaust, they are transferred into water by direct surface contact or as a result of rainfall (Grob & Grob, 1974; Van Noort & Wondergem, 1985a,b; Kawamura & Kaplan, 1986). The higher levels of PAH that are found during winter months reflect increased emissions resulting from domestic heating (Quaghebeur et al., 1983; Thomas, 1986; see also section 5.1.1.1); however, the major source of PAH varies for each body of water.

Anthropogenic combustion and pyrolysis and urban runoff containing atmospheric fallout, asphalt particles, tyre particles, automobile exhaust condensate and particulates, and lubricating oils and greases were the major sources of PAH in lakes in Switzerland (Wakeham et al., 1980a,b).

Comparisons between the levels of individual PAH in precipitation and those in surface water showed that all of the precipitation samples were more highly polluted with PAH, because they had been 'washed out' of the atmosphere. Nearly all of the samples contained > 100 ng/litre of fluoranthene, benzo[*b*]fluoranthene, pyrene, indeno[1,2,3-*cd*]pyrene, phenanthrene, and naphthalene. The highest levels of PAH in rainwater were found in Leidschendam, the Netherlands, where pyrene concentrations ≤ 2000 ng/litre, fluoranthene concentrations ≤ 1700 ng/litre, and benzo[*a*]pyrene and benzo[*b*]fluoranthene concentrations ≤ 390 ng/litre were detected (van Noort & Wondergem, 1985b).

Most surface water samples contained concentrations of ≤ 50 ng/litre of individual PAH. The levels in rainwater were 10–200 ng/litre, whereas those in snow were ≤ 1000 µg/kg, with a maximum of 6800 µg/kg for an individual PAH (Lygren et al., 1984). In one fog sample, benzo[*a*]pyrene was found at 880 ng/litre and fluoranthene at 3800 ng/litre (Schrimpff, 1983; see section 5.1.2.4).

In sediment the levels of individual PAH were usually 1000–10 000 µg/kg dry weight, which are one order of magnitude higher than those in precipitation. Triphenylene was detected in samples of sediment from the Mediterranean Sea (France) at 2–600 µg/kg (Milano et al., 1985) and in samples from Lake Geneva (Switzerland) at 25 µg/kg (Dreier et al., 1985; see section 5.1.3).

5.1.2.1 *Surface and coastal waters*

The levels of individual PAH found in surface and coastal waters at various locations are summarized in Table 37. Rivers in Germany contained some PAH at concentrations of 1–50 ng/litre (Grimmer et al., 1981b; Ernst et al., 1986; Regional Office for Water and Waste Disposal, 1986; Kröber & Häckl, 1989) and fluoranthene, pyrene, chrysene, benzo[*a*]pyrene, and benzo[*e*]pyrene at concentrations ≤ 100 ng/litre. The PAH levels in seawater from the German coast varied over one order of magnitude depending on the sampling site. In open seawater, the concentrations of two- to four-ring PAH—naphthalene, fluorene, phenanthrene, fluoranthene, and pyrene—were 0.1–5 ng/litre, and those of five- to six-ring PAH ranged from < 0.01 to 0.2 ng/litre. Near the coast, the concentration of five- to six-ring PAH increased with the content of particles, to which they have greater affinity than two- to four-ring PAH (German Federal Office for Sea Navigation and Hydrography, 1993).

The maximum levels of PAH in the Rivers Thames and Trent in the United Kingdom were > 130 ng/litre. The highest levels of individual PAH in the River Thames were 360 ng/litre fluoranthene, 350 ng/litre benzo[*a*]pyrene, 210 ng/litre indeno[1,2,3-*cd*]pyrene, 160 ng/litre benzo[*ghi*]perylene, 140 ng/litre benzo[*k*]fluoranthene, and 130 ng/litre perylene (Acheson et al., 1976). More recent data were not available.

In Norway, the levels of most individual PAH were > 100 ng/litre. For example, surface water from Bislet Creek near Oslo contained fluoranthene, pyrene, phenanthrene, methylphenanthrene, naphthalene, acenaphthene, acenaphthylene, and fluorene at concentrations > 1000 ng/litre (Berglind, 1982).

The highest concentrations of PAH in water in Canada were reported for water samples from ditches next to utility and railway lines near Vancouver. The highest mean concentrations were measured near utility poles treated with creosote, with values of 2000 µg/litre for fluoranthene, 1600 µg/litre for phenanthrene, and 490 µg/litre for naphthalene (Environment Canada, 1994). Four individual PAH were detected in seawater from Green Island, Australia. The highest levels of PAH found were 53 ng/litre pyrene, 25 ng/litre anthracene,

Table 37. Polycyclic aromatic hydrocarbon concentrations (ng/m³) in surface and coastal waters

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Acenaphthene									14–1232	
Acenaphthylene						0.4–0.9			12–1024	
Anthracene	1						10		18–932	
Anthanthrene				0.2–0.5	15/1.8					
Benz[*a*]anthracene	ND		0.16	2.2–6.8	24/66		40/10	71–582		
Benzo[*a*]fluorene									43/330	
Benzo[*a*]pyrene	1–23	0.8	0.39	1.2–7.3	87/25	18	10/60	ND–40	19–311	0.9
Benzo[*b*]fluoranthene		0.1–0.5	0.07				80/20	ND–42	70–678	0.5–0.9
Benzo[*b*]fluorene	38								17	
Benzo[*c*]phenanthrene				2.3–4.2	13/34				23–172	
Benzo[*e*]pyrene	2–40		0.06	7.1–11	108/36				40–551	
Benzo[*ghi*]fluoranthene										
Benzo[*ghi*]perylene	ND	ND	< 0.05	3.7–7.0	61/16		50/10	ND–61	33–636	ND
Benzo[*k*]fluoranthene		0.7–0.8	0.02	3.6–6.1	59/22		40/10	ND–24		0.2–0.5
Chrysene				11–15	36/87	14	10/10			
Coronene				ND–2.4	15/4.3					
Cyclopenta[*cd*]pyrene				ND	ND					
Dibenzo[*a,h*]pyrene			< 0.03				30/10			
Fluoranthene	4–616	1.0–3.5	0.35	5.2/9.1	28/102	2.3–13	50/130	2–110	285–3269	3.4–5.1
Fluorene	2		0.63			0.6–1.2			25–1995	
Indeno[1,2,3-*cd*]pyrene		Trace	< 0.03	2.8–6.1	63/13		50/20	ND–39	17–299	ND
1-Methylphenanthrene									30–1281	

Table 37 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
5-Methylcholanthrene										
Naphthalene	4								50–2090	
Perylene				0.8–1.4	27		20		9/28	
Phenanthrene	3–136		3.5			1.5–9.1			101–5656	
Pyrene	5–402		0.28	4.8/8.5	25/90	2.2–13	100/30		485–3099	
Triphenylene										

ND, not detected; /, single measurements; **[1]** Lake water, Norway, 1981–82 (Gjessing et al., 1984); **[2]** Lake water, Switzerland (Vu Duc & Huynh, 1981); **[3]** Lake Superior, USA, 1986 (Baker & Eisenreich, 1990); **[4]** Elbe River, Germany, 1980 (Grimmer et al., 1981b); **[5]** Elbe River, main drainage channel, Germany, 1980 (Grimmer et al., 1981b); **[6]** Water in various rivers, Germany, 1981–83 (Ernst et al., 1986); **[7]** Water in various rivers, Germany, 1985; analytical method not given (Regional Office for Water and Waste Disposal, 1986); **[8]** Water in various rivers, Germany, 1985–86; analytical method not given (Kröber & Häckl (1989); **[9]** River water, Norway, 1979 (Berglind, 1982); **[10]** River water, Switzerland (Vu Duc & Huynh, 1981)

Analysed by high-performance liquid chromatography or gas chromatography, unless otherwise stated. The results of studies in which water samples were filtered through solid sorbents may be underestimates of the actual PAH content (see section 2.4.1.4).

Table 37 (contd)

Compound	[11]	[12]	[13]	[14]	[15]	[16]	[17]	[18]	[19]	[20]
Acenaphthene				ND–3	10			0.08–1.1		50–100
Acenaphthylene				ND–5				0.02–1.7		80–1300
Anthracene				ND–4	0.2	0.8–9.5		0.01–1.5	< 1–25	ND
Anthanthrene										NR
Benz[*a*]anthracene				ND–5	0.3	ND–9.6		0.04–6.8		ND
Benzo[*a*]fluorene										NR
Benzo[*a*]pyrene	0.1–1.8	130–150	0.1/0.2	ND–10	0.2–1.0			0.03–8.8		ND
Benzo[*b*]fluoranthene				ND–8				0.04–12		
Benzo[*b*]fluorene						4.0–19				NR
Benzo[c]phenanthrene										NR
Benzo[*e*]pyrene								0.02–8.8		ND
Benzo[*ghi*]fluoranthene										NR
Benzo[*ghi*]perylene	0.2–11	30–160	0.7/0.8	ND–10				0.02–3.8	< 0.3–16	50
Benzo[*k*]fluoranthene	0.1–1.7	80–140	0.2/0.3	ND–13				0.02–7.7		
Chrysene				ND–12						NR
Coronene								0.01–1.4		NR
Dibenzo[*a,h*]pyrene				ND–1						100
Fluoranthene	0.7–508	20–360	1.1/3.7	3–12	0.8	10–25	1.4–2.6	0.40–14		NR
Fluorene				ND–2	0.7–15		1.9–5.2	0.33–3.2		70–2500
Indeno[1,2,3-*cd*]pyrene	0.1–8.0	50–210	ND/0.2	ND–8				0.01–3.5		NR
1-Methylphenanthrene										NR

Table 37 (contd)

Compound	[11]	[12]	[13]	[14]	[15]	[16]	[17]	[18]	[19]	[20]
5-Methylcholanthrene										NR
Naphthalene				4–34	3.6			0.4–9.2		NR
Perylene		40–130						0.01–5.7		NR
Phenanthrene				6–34	2.1–18	6.0–93	2.4–2.7	0.24–5.8	< 1–3	ND
Pyrene		50–260		1–15	0.3–15	8.8–25	0.82–1.7	0.12–15	< 1–53	10–65
Triphenylene										NR

ND, not detected; /, single measurements; **[11]** River water, United Kingdom, 1974 (Lewis, 1975); **[12]** Water in various rivers, United Kingdom, analytical method not given (Acheson et al.,1976); **[13]** Water in various rivers, United Kingdom; analytical method not given (Sorrell et al., 1980); **[14]** River water, USA, 1984 (De Leon et al., 1986); **[15]** Surface water, Canada (Environment Canada, 1994); **[16]** River water, China, 1981 (Wu et al., 1985); **[17]** Coastal water, Germany, 1982 (Ernst et al., 1986); **[18]** Seawater, Germany, 1990 (German Federal Office for Sea Navigation and Hydrography, 1993); **[19]** Coastal water, Australia, 1983 (Smith et al., 1987); **[20]** Water (no further specification), Japan, 1974–91 (Environment Agency, Japan, 1993)

Analysed by high-performance liquid chromatography or gas chromatography, unless otherwise stated. The results of studies in which water samples were filtered through solid sorbents may be underestimates of the actual PAH content (see section 2.4.1.4).

16 ng/litre benzo[*ghi*]perylene, and 3 ng/litre phenanthrene, (Smith et al., 1987).

The total content of phenanthrene, anthracene, fluoranthene, pyrene, benzo[*b*]fluorene, and benz[*a*]anthracene in the Yellow River, China, was 170 ng/litre (Wu et al., 1985; for individual PAH concentrations, see Table 37).

The PAH levels found in the River Rhine in Germany and the Netherlands and in some of its tributaries are summarized in Table 38. Many investigators have detected PAH in the Rhine. The lowest concentrations of benzo[*a*]pyrene, < 10–20 ng/litre, were found in the Rhine at Lobith and Hagestein in Germany and at Lek in the Netherlands in 1987–90 (Association of Rhine and Meuse Water Supply Companies, 1987–90), when the levels of fluoranthene were 70–140 ng/litre. In 1976–79, the Rhine at Lek and Waal contained < 10–580 ng/litre of benzo[*a*]pyrene (Association of Rhine and Meuse Water Supply Companies, 1976–79), so that the levels had decreased by one order of magnitude within 14 years. The sum of fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, and indeno[1,2,3-*cd*]pyrene) was 9–40 ng/litre at km 30 and 130–5700 ng/litre at km 853, indicating that the level of pollution increased markedly between the source and the estuary (Borneff & Kunte (1983). The average concentrations of individual PAH were 1–50 ng/litre, although individual PAH were found at concentrations in the range 100–200 ng/litre near Mainz, an industrialized town (Borneff & Kunte, 1964, 1965). In general, the PAH levels in the Rhine decreased by a factor of 3 between 1979 and 1989.

The Emscher and Ruhr waterways in Germany have been heavily polluted (see Table 38). In 1985, the Emscher River contained 6400 ng/litre fluoranthene, 6000 ng/litre pyrene, 2000 ng/litre benz[*a*]anthracene, 1100 ng/litre dibenz[*a,h*]anthracene, 910 ng/litre benzo[*a*]pyrene, 880 ng/litre chrysene, 630 ng/litre indeno[1,2,3-*cd*]pyrene, 510 ng/litre benzo[*ghi*]perylene, 270 ng/litre anthracene, 220 ng/litre perylene (Regional Office for Water and Waste Disposal, 1986), but by 1989 the levels had decreased by about one order of magnitude (Regional Office for Water and Waste Disposal, 1990). The PAH concentrations in the Emscher were three times higher than those in the Rhine near Mainz. Between 1985 and 1989, the PAH levels in the Emscher decreased further by a factor of 15; however, the levels in the Ruhr remained about the same or increased slightly between 1979 and 1985 (Regional Office for Water and Waste Disposal, 1986, 1988, 1990).

The PAH levels in the main drainage channels of the River Elbe, Germany, were one order of magnitude higher than in the river water (Grimmer et al., 1981b), owing to the high input of rainwater to the channels.

5.1.2.2 *Groundwater*

The PAH concentrations in uncontaminated groundwater in the Netherlands generally did not exceed 0.1 µg/litre, but levels of about 30 µg/litre naphthalene,

Table 38. Polycyclic aromatic hydrocarbon concentrations (ng/m³) in the River Rhine and some highly polluted tributaries

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Anthracene				10		270	25–260		10
Anthanthrene	0.9–11							1.3	
Benz[*a*]anthracene	6.1–31			11–50		1970	100–780	13	20
Benzo[*a*]pyrene	0.8–36	ND–7	6–30	12–40	< 10–20	910	59–280	15	30
Benzo[*b*]fluoranthene		ND–8	7–30	12–40	< 10–30	880	62–310		40
Benzo[*c*]phenanthrene	1.5–9.1							1.9	
Benzo[*e*]pyrene	18–31							33	
Benzo[*ghi*]fluoranthene	1.0–11							2.2	
Benzo[*ghi*]perylene	15–29	ND–8	6–30	9–30	< 10–20	510	30–210	17	30
Benzo[*k*]fluoranthene		ND–4	2–14	6–20	< 10–40	440	36–150		20
Chrysene	21–62					1080		27	30
Dibenzo[*a,h*]pyrene				10–40		1100	32–310		30
Fluoranthene		4–18	15–61	25–77	20–140	6420	207–1700	60	
Indeno[1,2,3-*cd*]pyrene	9.5–27	ND–6	2–26	10–40	< 10–20	630	28–220	17	30
Perylene	ND–8.1			10		220	13/80	2.1	10
Pyrene				20–50		6010	155–1100		50

ND, not detected; /, single measurements; **[1]** Rhine, Germany, 1979 (Grimmer et al.,1981b); **[2]** Rhine, Germany, 1985–88, analytical method not given (Kröber & Häckl, 1989); **[3]** Rhine, Netherlands, 1985–88 (Netherlands' Delegation, 1991); **[4]** Rhine, Germany, 1987–89, analytical method not given (Regional Office for Water and Waste Disposal, 1988, 1989, 1990); **[5]** Rhine, Netherlands; 1987–90, analytical method not given (Association of Rhine and Meuse Water Supply Companies, 1987–90); **[6]** Emscher, Germany,1985, analytical method not given (Regional Office for Water and Waste Disposal, 1986); **[7]** Emscher, Germany, 1987–89, analytical method not given (Regional Office for Water and Waste Disposal, 1988, 1989, 1990); **[8]** Ruhr, Germany, 1979 (Grimmer et al., 1981b); **[9]** Ruhr, Germany, 1985, analytical method not given (Regional Office for Water and Waste Disposal, 1986)

10 µg/litre fluoranthene, and 1 µg/litre benzo[*a*]pyrene were reported in contaminated groundwater (Luitjen & Piet, 1983).

Benzo[*a*]pyrene levels in groundwater in western Germany ranged from 0.1 to 0.6 ng/litre and those of total PAH from 34 to 140 ng/litre (Andelman & Suess, 1970). Benzo[*a*]pyrene was also detected at levels of 0.1-5.0 ng/litre in groundwater (Woidich et al., 1976). More recent data were not available.

Groundwater in the USA contained maximum concentrations of 0.38–1.8 ng/litre naphthalene, 0.02–0.04 ng/litre acenaphthene, and 0.008–0.02 ng/litre fluorene (Stuermer et al., 1982). Near a refinery at Pincher Creek, Alberta, Canada, the pyrene concentrations in groundwater showed a maximum of 300 µg/litre (median, 30 µg/litre); the maximum concentration of fluorene was 230 µg/litre (median, 40 µg/litre). At Newcastle, New Brunswick, Canada, naphthalene was detected at concentrations up to 2.8 µg/litre and benzo[*a*]pyrene up to 0.32 µg/litre in groundwater near a wood-preserving plant (Environment Canada, 1994).

5.1.2.3 *Drinking-water and water supplies*

PAH levels were determined in drinking-water in samples from Canada, Scandinavia, and the USA up to 1982. The concentration of naphthalene was 1.2-8.8 ng/litre, that of benzo[*a*]pyrene was 0.2-1.6 ng/litre, and that of the sum of the six 'standard WHO' PAH (fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, and indeno[1,2,3-*cd*]pyrene) was 0.6–24 ng/litre. The highest levels of naphthalene (1300 ng/litre), benzo[*a*]pyrene (77 ng/litre), and the six WHO standard PAH (660 ng/litre) were detected in raw water sources in the USA and in the Great Lakes area of Canada (Müller, 1987). More recent measurements are given in Table 39. Most samples contained 0.38–16 ng/litre naphthalene and < 0.04–2.0 ng/litre benzo[*a*]pyrene. In one set of water samples from the Netherlands, no PAH were detected, with a limit of detection for individual PAH of 4 ng/litre (de Vos et al., 1990).

In a study of the changes in PAH concentrations after passage of water through tar-coated major distribution pipes, the level increased from an initial concentration of none detected–13 ng/litre to none detected–62 ng/litre. The finding that water in a few distribution lines had lower concentrations of PAH may be due to sorption of PAH on the surfaces of distribution pipes, chemical interaction with oxidants in water, or a dilution effect (Basu et al., 1987).

Of 101 German drinking-water samples analysed in 1994, four exceeded the German drinking-water standard of 0.2 µg/litre for the sum of fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]-perylene, and indeno[1,2,3-*cd*]pyrene. Heavy contamination had occurred after repairs to a pipeline coated with tar, and one drinking-water sample taken in a household contained 2.7 µg/litre of these PAH, in addition to phenanthrene at 2.8 µg/litre and pyrene at 1.2 µg/litre (State Chemical Analysis Institute,

Table 39. Polycyclic aromatic hydrocarbon concentrations (ng/litre) in drinking-water

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Acenaphthene		0.6–4.0	7.4–14						
Acenaphthylene		0.4–4.4	0.40–1.6						
Anthracene		0.5–7	< 1.3–9.7						
Anthanthrene		0.2							
Benz[*a*]anthracene	ND–1.9	0.4–5.5	0.12–1.5						
Benzo[*a*]fluoranthene		0.1–3.3	0.05–4.2						
Benzo[*a*]pyrene	0.1–0.7	< 0.1–2.0	< 0.04–0.29		Trace–1.9	0.2–0.3		0.2–1.6	< 5.0
Benzo[*b*]fluoranthene	0.5–1.3	2.4–4.0	0.05–0.34		0.1–14				< 5–40
Benzo[*b*]fluorene		0.9	0.04–< 1.4						
Benzo[*c*]phenanthrene		0.9–1.5	0.28						
Benzo[*e*]pyrene		0.2–4	< 0.1–0.41						
Benzo[*ghi*]fluoranthene			0.36						
Benzo[*ghi*]perylene	0.3–0.9	0.4–1.1			ND	0.4–0.7		0.4–4.0	< 5.0
Benzo[*j*]fluoranthene			0.03–0.14					0.2–1.2	
Benzo[*k*]fluoranthene	0.2–0.8		0.02–0.10		0.2–4.9	0.1–0.3		0.1–0.7	< 5–40
Chrysene	21–62					1080		27	30
Dibenz[*a,h*]anthracene		1.2							
Fluoranthene	3.5–6.5	1.7–18	< 0.58–24		0.7–3400	3.4–4.2	5–24	2.4–9.0	< 5–623
Fluorene		0.9–4	< 1.1–21				4–16		
Indeno[1,2,3-*cd*]pyrene	Trace–0.7	0.4–1.2			ND–1.1	< 0.5		0.7–2.2	< 5.0
1-Methylphenanthrene		0.5–1.0	0.14–13						
Naphthalene		1.8–5	< 6.3–8.8	8			6–16		

Table 39 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Perylene	Trace–0.2	0.2							
Phenanthrene		2.5–46	< 2.2–64				24–90		
Pyrene	1.6–3.7	1.1–15	< 0.30–12						40/40

ND, not detected; /, single measurements; **[1]** Austria; analytical method, in-situ fluorescence determination (Woidich et al., 1976); **[2]** Norway, 1978–80 (Berglind, 1982); **[3]** Norway, 1980–81 (Kveseth et al., 1982); **[4]** Switzerland, 1973 (Grob & Grob, 1974); **[5]** Switzerland (Vu Duc & Huynh, 1981); **[6]** United Kingdom; water reservoirs after treatment, 1974 (Lewis, 1975); **[7]** USA, 1976; analytical method, high-performance liquid chromtography and gas chromatography (Thruston, 1978); **[8]** USA, 1976–77; analytical method, thin-layer chromatography and gas–liquid chromatography with flame ionization detection (Basu & Saxena, 1978a,b); **[9]** Canada, treated drinking-water, 1987–90 (Environment Canada, 1994)

Analysed by high-performance liquid chromatography or gas chromatography, unless otherwise stated. The results of studies in which water samples were filtered through solid sorbents may be underestimates of the actual PAH content (see section 2.4.1.4).

Freiburg, 1995). The report stated that abrasion of particles from tar-coated drinking-water pipelines poses a hazard that is often difficult to judge since it is often not known what material was used decades previously.

In Canada, the PAH concentrations in drinking-water were usually below or near the detection limits of 1–5 ng/litre, although concentrations of 5.0–21 ng/litre benzo[*ghi*]perylene, 1.0–12 ng/litre fluoranthene, 1.0–5.0 ng/litre benzo[*b*]fluoranthene, 1.0–3.0 ng/litre benzo[*k*]fluoranthene, and 1.0–3.0 ng/litre benzo[*a*]pyrene were detected in some areas (Environment Canada, 1994).

5.1.2.4 *Precipitation*

(a) Rain

The concentrations of PAH found in precipitation in 1979–91 are summarized in Table 40. The levels of benzo[*a*]pyrene were < 1–390 ng/litre. In an analysis of PAH in rainfall in Hanover, Germany, between July 1989 and March 1990, fluoranthene was the dominant component, followed by pyrene. The average concentration of all PAH increased from 351 ng/litre in summer to 765 ng/litre in the autumn of 1989, while a slight decrease was observed in the winter of 1989–90. These results indicate that the increase in the level of PAH in precipitation in cold weather is due to an increase in residential heating and a slower rate of photochemical degradation (Levsen et al., 1991).

The concentrations of phenanthrene and fluoranthene in rainwater were noticeably higher than those at 200 m when sampled simultaneously, but no significant differences in the concentrations of benzo[*k*]fluoranthene, benzo[*b*]fluoranthene, benzo[*a*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, or indeno[1,2,3-*cd*]pyrene were found. The authors suggested that scavenging in and below clouds was responsible for the presence of PAH in rainwater (Van Noort & Wondergem, 1985b).

The deposition rates of individual PAH in Cardiff, London, Manchester, and Stevenage, United Kingdom, were 0.3–20 μg/m^2 per day. Anthracene accounted for about 25% of the deposition in London, followed by pyrene (16%), benzo[*b*]fluoranthene (16%), and benz[*a*]anthracene (13%) (Clayton et al., 1992).

The rate of precipitation containing PAH after gravitational deposition by rain, snow, and particles was not affected by the type or structure of the receiving surface. Precipitation in a beech and spruce stand contained concentrations of 23–52 ng/litre fluoranthene, 8.9–30 ng/litre benzo[*ghi*]-perylene, 6.4–27 ng/litre indeno[1,2,3-*cd*]pyrene, and 2.0–8.4 ng/litre benzo[*a*]pyrene. The deposition of PAH is in general higher under spruce stands because the rates of interception are higher than those in beech stands. Substantial amounts of PAH are transferred to the soil by litterfall, indicating adsorption of PAH on the surfaces of leaves and needles (Matzner, 1984).

Table 40. Polycyclic aromatic hydrocarbon concentrations (ng/litre) in rainwater

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Acenaphthene							3.2	1.2/16	2.5–8.5
Acenaphthylene			130–200				14	4.7/55	23–59
Anthracene							8–19	0.88/23	2.0–7.9
Benz[*a*]anthracene		1.2–86	140	6–100	9–33	7–17	20–65		1.6–4.5
Benzo[*a*]fluoranthene							14–52		
Benzo[*a*]pyrene	5–17	1.1–187		ND–390	10–37	7–26	5–36		ND–0.18
Benzo[*b*]fluoranthene		2.9–166		15–390	45–70	17–65			
Benzo[*b*]fluorene								15	
Benzo[*c*]phenanthrene								8–32	
Benzo[*e*]pyrene		< 0.5[a]–149	217–290				7–62		ND–0.51
Benzo[*ghi*]perylene	7–29	1.7–109			40–70	15–56	22		
Benzo[*k*]fluoranthene		1.0–142		6–190	17–30	9–28			
Chrysene		2.9–141	30–120		ND–67	21–29			3.3–12
Dibenz[*a,h*]anthracene		< 0.5[a]–12			7–20	3–12			
Fluoranthene	23–66	23–392	240–270	14–1650	66–180	87–189	115–162	1.7/110	28–70
Fluorene			10–200				6–50	3.2/43	9.1–22
Indeno[1,2,3-*cd*]pyrene		< 0.5[a]–137		ND–80	50–110	24–72	12		
1-Methylphenanthrene							8–26		
Naphthalene							8–77	20/72	46–140

Table 40 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Perylene							2		
Phenanthrene			130–600		30–133	79–113	158–238	24/140	61–130
Pyrene		9.5–304	25–60	ND–2000	ND–37	36–108	77–175		24–56

ND, not detected; /, single measurements; **[1]** Bavaria, Germany,1979–80; analytical method, high-performance thin-layer chromatography (Thomas, 1986); **[2]** Hanover, Germany, 1989–90 (Levsen et al., 1991); **[3]** Italy (Morselli & Zappoli, 1988); **[4]** Leidschendam, Netherlands, 1982 (Van Noort & Wondergem, 1985b); **[5]** Rotterdam, Netherlands, 1983 (Van Noort & Wondergem, 1985b); **[6]** Netherlands, 1983 (Den Hollander et al., 1986); **[7]** Oslo, Norway, 1978 (Berglind, 1982); **[8]** Oregon, USA, 1982 (Pankow et al., 1984); **[9]** Portland, USA, 1984 (Ligocki et al., 1985)

[a] Detection limit for benzo[*a*]pyrene

Analysed by high-performance liquid chromatography or gas chromatography, unless otherwise stated. The results of studies in which water samples were filtered through solid sorbents may be underestimates of the actual PAH content (see section 2.4.1.4).

(b) Snow

The concentrations of PAH in snow samples are summarized in Table 41. A sample collected in Hanover, Germany, contained fluoranthene at 55 ng/litre, pyrene at 31 ng/litre, and other PAH at concentrations up to 9 ng/litre (Levsen et al., 1991). A sample of snow from Bavaria contained 200 ng/litre fluoranthene, 50 ng/litre benzo[*ghi*]perylene, and 29 ng/litre benzo[*a*]pyrene (Schrimpff et al., 1979).

In Norwegian snow samples, the average concentrations of individual PAH were 10–100 ng/litre, but levels up to 6800 ng/litre were found of phenanthrene, 1-methylphenanthrene, fluoranthene, benzo[*b*]fluoranthene, and fluorene (Berglind, 1982; Gjessing et al., 1984; Lygren et al., 1984). Snow taken near a steel plant in Canada contained average levels of 50–500 ng/litre of individual PAH but higher amounts of phenanthrene, fluoranthene, and pyrene (Boom & Marsalek, 1988).

(c) Hail

The PAH levels in a hail sample collected in Hanover, Germany, were of the same order of magnitude as those in rain samples: fluoranthene, 170 ng/litre; pyrene, 98 ng/litre; benzo[*b*]fluoranthene, 58 ng/litre; chrysene, 47 ng/litre; benzo[*e*]pyrene, 40 ng/litre; indeno[1,2,3-*cd*]pyrene, 29 ng/litre; benzo[*ghi*]perylene, 27 ng/litre; benzo[*k*]fluoranthene, 19 ng/litre; benz[*a*]anthracene, 16 ng/litre; benzo[*a*]pyrene, 12 ng/litre; and dibenz[*a,h*]anthracene, 3.3 ng/litre (Levsen et al., 1991).

(d) Fog

The concentrations of PAH in fog are higher than those in rain. A fog sample collected in western Germany contained 360-3800 ng/litre fluoranthene and 130–880 ng/litre benzo[*a*]pyrene (Schrimpff, 1983).

In fog samples collected during the autumn of 1986 in Zürich, Switzerland, the average concentrations of PAH found were 4400 ng/litre fluoranthene, 2700 ng/litre benzo[*b*]fluoranthene, 2500 ng/litre pyrene, 2200 ng/litre phenanthrene, 2100 ng/litre benzo[*e*]pyrene, 1400 ng/litre benz[*a*]anthracene, 1400 ng/litre indeno[1,2,3-*cd*]pyrene, 1200 ng/litre benzo[*a*]pyrene, 920 ng/litre anthracene, 860 ng/litre 1-methylphenanthrene, 750 ng/litre benzo[*b*]fluorene, 750 ng/litre perylene, 590 ng/litre benzo[*k*]fluoranthene, 540 ng/litre benzo[*ghi*]perylene, 340 ng/litre anthanthrene, 260 ng/litre fluorene, and 160 ng/litre benzo[*a*]fluorene (Capel et al., 1991).

5.1.3 *Sediment*

PAH levels in sediments from rivers, lakes, seas, estuaries, and harbours are summarized in Tables 42–46.

Table 41. Polycyclic aromatic hydrocarbon concentrations (ng/litre) in snow

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Acenaphthene			10–13			<50–98
Acenaphthylene			19–47			<50–153
Anthracene			13–28	9–379	165–246	
Benz[*a*]anthracene		2.6	21–47	15–677	228	
Benzo[*a*]fluoranthene			13		179–396	
Benzo[*a*]pyrene	29	3.0	23–77	54–602	250	<100–558
Benzo[*b*]fluoranthene		9.2			799–1501	<100–647
Benzo[*b*]fluorene			11		192	
Benzo[*e*]pyrene		5.5	30–64	609	360–630	
Benzo[*ghi*]perylene	50	4.8	29–85	98–551	319–391	<100–466
Benzo[*k*]fluoranthene		2.8				<100–990
Chrysene		6.2				
Dibenz[*a,h*]anthracene		<0.5[a]				
Fluoranthene	200	55	108–211	86–2665	1820–3143	<50–7020
Fluorene			13–85	96	485–1237	<50–237
Indeno[1,2,3-*cd*]pyrene		<0.5[a]	20–82			<100–496
1-Methylphenanthrene					1366–2117	
Naphthalene			50–94	36–67	123–195	
Perylene			12			
Phenanthrene			119–276	45–1385	4055–6787	<50–3560
Pyrene		31	68–143	55–2002		<50–3750

Analysed by high-performance liquid chromatography or gas chromatography, unless otherwise stated. The results of studies in which water samples were filtered through solid sorbents may be underestimates of the actual PAH content (see section 2.4.1.4).

[a] Detection limit for benzo[a]pyrene

[1] Bavaria, Germany, 1978; analytical method, high-performance thin-layer chromatography and gas chromatography–mass spectroscopy (Schrimpff et al., 1979); **[2]** Hanover, Germany, 1990 (Levsen et al.,1991); **[3]** Norway, 1979–81 (Berglind, 1982); **[4]** Norway, 1981–82 (Gjessing et al., 1984); **[5]** Norway (Lygren et al., 1984); **[6]** Near steel plant, Canada, 1986 (Boom & Marsalek, 1988)

5.1.3.1 *River sediment*

The concentrations of individual PAH in river sediments in 1987–91 (Table 42) varied over a wide range; the maximum values were in the high nanogram per gram range.

Table 42. Polycyclic aromatic hydrocarbon concentrations (μg/kg) in river sediments

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]	[12]
Acenaphthene	ND–140						14.5			1 100	ND	0.04–130
Acenaphthylene	ND						9.7			1 540	ND	0.7–671
Anthracene	ND–1010	80–640	670/NR	ND/NR			82.1	8–200	152	4 700		10–1200
Benz[*a*]anthracene		620–1700	10000/NR	50–90/NR			450	ND–100	541	6 600		3.2–2100
Benzo[*a*]pyrene	Q	400–1250	ND–8000/ ND–5300	20–90/10–80	1–760	70–11 960	454	ND–80	570	4 400		5–3700
Benzo[*b*]fluoranthene		460–1290	ND–8700/ ND–5600	50–190/ 26–150			620	ND–50				
Benzo[*e*]pyrene									596	4 900		0.9–1800
Benzo[*ghi*]fluoranthene									253			NR
Benzo[*ghi*]perylene	Q–578	340–750	ND–2900/ ND–1900	ND	10–70	60–7480	358	ND	353	7 400		3–1310
Benzo[*j*]fluoranthene									749			
Benzo[*k*]fluoranthene		230–650	ND–4000/ ND–2700	20–90/10–80			408	ND–60	608			
Chrysene	ND–1549		6700/NR	ND–30/NR			597		904			NR
Coronene					20–260	150–2460			284			NR
Cyclopenta[*cd*]pyrene									15	1 100		NR
Dibenz[*a,h*]anthracene		500–1070	2600/NR	ND–20/NR			21	ND–200		2 800		8.1–340
Fluoranthene	ND–4455	900–2470	ND–19 000/ ND–12 600	100–380/ 52–310	2–2360	190–29 300	904	100–400		1013	13 000	ND–60
NR												
Fluorene	ND–260						25.4	ND–2	26	3 000	ND/50	3–130

Table 42 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]	[12]
Indeno[1,2,3-*cd*]pyrene		360-910	ND-6300/ ND-4200	ND/ND			332		486	16 000		NR
1-Methylphenanthrene									145			NR
Naphthalene	ND-2630						7.0			3 800		ND
Perylene		120-320						ND-100		2 400		NR
Phenanthrene	ND-220		3300/NR	ND-40/NR			361	10-400	563	10 000	ND/220	9-2800
Pyrene	ND-2526	680-3450	17000/NR	ND-130/NR			736	80-300	940	9 200	ND-160	20-3900
Triphenylene								10-80				NR

ND, not detected; /, single measurements; NR, not reported; Q, qualitative; **[1]** Czechoslovakia, 1988; reference weight not given (Holoubek et al., 1990); **[2]** Rhine, Germany, 1982–83 and 1987–88; analytical method and reference weight not given (Regional Office for Water and Waste Disposal, 1989); **[3]** Neckar, Germany, 1985–88; fine, unsieved sediment; analytical method not given (Kröber & Häckl, 1989); **[4]** Gersprenz, Germany, 1985–88; fine, unsieved sediment; analytical method not given (Kröber & Häckl, 1989); **[5]** Wildbach, Germany, 1989 (Lampe et al., 1991); **[6]** Haarbach, Germany, 1989 (Lampe et al., 1991); **[7]** River, Bremen, Germany, 1994 (Riess & Wefers, 1994); **[8]** Rhône, France, 1985 (Milano & Vernet, 1988); **[9]** Sweden, 1985 (Broman et al., 1987); **[10]** Black River, USA, 1984 (Fabacher et al., 1991); **[11]** Rainy River, Canada, 1986; reference weight not given (Merriman, 1988); **[12]** Japan, 1974–91 (Environment Agency, Japan, 1993)

Analysed by high-performance liquid chromatography or gas chromatography and concentration in micrograms per kilogram dry weight

The levels of individual PAH in sediments from German rivers were about 4000 µg/kg for benzo[*a*]pyrene, fluoranthene, and benzo[*b*]fluoranthene and about 1500 µg/kg for pyrene, indeno[1,2,3-*cd*]pyrene, and benz[*a*]anthracene. The levels of other PAH generally did not exceed 500 µg/kg (Kröber & Häckl, 1989; Regional Office for Water and Waste Disposal, 1989). PAH were determined in many German river sediments. Table 42 gives data for three rivers: the Rhine and Neckar rivers are highly polluted, whereas the Gersprenz is relatively uncontaminated.

The concentrations of PAH in the sediments of rivers around Aachen, Germany, were determined in different size fractions, which allowed the authors to locate where the sediment became contaminated (Lampe et al., 1991).

The PAH concentrations in sediment from the River Elbe in Germany in 1991 were of the same order of magnitude as those in Lake Plöner and Lake Constance, but the river sediment contained more PAH with a low boiling-point than the lake sediments. The ratio of fluoranthene to benzo[*e*]pyrene, taken as a marker of the emission of PAH from the combustion of brown coal, was 2.8–5.1, similar to those found in the Elbe sediment. It was concluded that the PAH in the sediment were due mainly to brown-coal combustion (German Ministry of Environment, 1993).

The maximum levels of individual PAH in sediments in Czechoslovakia were 4500 µg/kg fluoranthene, 2600 µg/kg naphthalene, 2500 µg/kg pyrene, 1500 µg/kg chrysene, 1000 µg/kg anthracene, 580 µg/kg benzo[*ghi*]perylene, 260 µg/kg fluorene, 220 µg/kg phenanthrene, and 140 µg/kg acenaphthene (Holoubek *et al.*, 1990).

The levels of individual PAH in sediments from some of the most polluted areas in continental USA were summarized by Bieri et al. (1986). The levels usually ranged from 1000 to 10 000 µg/kg, but that in sediment from the Elizabeth River, Virginia, contained concentrations up to 42 000 µg/kg. Up to 39 000 µg/kg wet weight were found in the Detroit River (Fallon & Horvath, 1985).

The concentrations of individual PAH in sediments from the Trenton Channel of the Detroit River, a waterway in a highly industrialized area, connecting Lake St Clair with Lake Erie, varied from not detected (< 4 µg/kg) to 22 000 µg/kg in different locations. Sediments from the southwest shore of Grosse Ile had low levels of contamination, while those in the vicinity of Monguagon Creek had high levels (Furlong et al., 1988).

5.1.3.2 *Lake sediment*

The concentrations of individual PAH found in lake sediments in 1984–91 (Table 43) ranged from 1 to about 30 000 µg/kg dry weight. The total PAH concentrations in surface sediments from Lake Michigan, USA, were 200–6200 µg/kg dry weight (Helfrich & Armstrong, 1986).

Table 43. Polycyclic aromatic hydrocarbon concentrations (µg/kg) in lake sediments

Compound	[1]	[2]	[3]	[4]
Anthracene	160		41–620	
Benz[*a*]anthracene	ND		150–1700	41
Benzo[*a*]pyrene			180–2000	45
Benzo[*b*]fluoranthene				200
Benzo[*e*]pyrene	80		140–1500	75
Benzo[*ghi*]fluoranthene		75	18–270	
Benzo[*ghi*]perylene			21–1600	107
Benzo[*k*]fluoranthene				126
Chrysene		250		124
Coronene		1		
Dibenz[*a,h*]anthracene				70
Fluoranthene	66–248	390	330–3900	103
Fluorene				5.9
Indeno[1,2,3-*cd*]pyrene		100	25–1500	279
Naphthalene	ND			
Perylene		50	47–540	
Phenanthrene	70–180	100	300–6600	81
Pyrene	110–122	340	210–3500	60
Triphenylene		25		

ND, not detected; **[1]** Lake Padderudvann, Norway; 1981–82; reference weight not given (Gjessing et al., 1984); **[2]** Lake Geneva, Switzerland (Dreier et al., 1985); **[3]** Cayuga Lake, USA, 1978; concentrations are given as ng/g deepwater (Heit, 1985); **[4]** Lake Superior, USA (Hamburg Environment Office, 1993)

Analysed by high-performance liquid chromatography or gas chromatography; concentration in micrograms per kilogram dry weight

5.1.3.3 *Marine sediment*

The concentrations of individual PAH in marine sediments in 1985–91 (Table 44) varied widely, with maximum values up to about 4000 µg/kg.

Sediments near power-boat moorings at the coral reef around Green Island, Australia, were found to contain measurable amounts of several PAH, strongly suggesting that they originated from fuel spillage or exhaust emissions (Smith et al., 1987).

The benzo[*a*]pyrene level was 10^4–10^6 times higher in bottom sediments from the Baltic Sea than in water at the same location. The bottom sediments also contained more individual PAH than the corresponding water samples (Veldre & Itra, 1991).

Table 44. Polycyclic aromatic hydrocarbon concentrations (µg/kg) in sea sediments

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Acenaphthene		ND–6				NR		
Acenaphthylene		ND–2000	0.6–4.3			NR		
Anthracene		3–800	0.3–2.1	6–42		5–313		< 0.06–1.0
Anthanthrene				29–74		NR		
Benz[*a*]anthracene	5–39	1–900	0.8–19	9–150		15–250		< 0.01–6.0
Benzo[a]fluoranthene				2–41		NR		
Benzo[a]pyrene	16–25	6–2200	0.4–13	7–160	1100	14–265	0.2–460	< 0.004–4.3
Benzo[*b*]fluoranthene	13–26	ND–3800			1300	51–490		
Benzo[*b*]fluorene				2–38		NR		
Benzo[*e*]pyrene	5.8–18		0.6–15	9–125		21–345	0.4–396	< 0.1–0.6
Benzo[*ghi*]perylene		ND–400		12–225	700	< 10–623		< 0.01–2.6
Benzo[*k*]fluoranthene	4.0–9.8	ND–3400			600	10–180		< 0.001–2.5
Chrysene	49		1.0–12	8–165		21–398		< 0.04–0.8
Coronene				11–36		NR		
Dibenzo[*a,e*]pyrene				7–79		NR		
Dibenz[*a,h*]anthracene	2–7	ND–400	0.5–4.2	4–74		NR		
Fluoranthene	ND/159	4–2000	0.4–31	12–230	2300	36–1913		< 0.1–7.2
Fluorene		ND–100	0.5–3.1	1–12		NR		
Indeno[1,2,3-*cd*]pyrene				8–200		17–510		
Naphthalene		ND–100	0.7–8.6	1–2[b]		18–1074		
Perylene		1–2200		5–105		24–178		
Phenanthrene		1–1500	0.8–29	23–93		11–971		< 0.06–4.2

Table 44 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Pyrene	8–160	5–1600	1.6–40	10–145		30–1697		< 0.1–15
Triphenylene		2–600				NR		

ND, not detected; /, single measurements; NR, not reported; **[1]** Baltic Sea, Estonia, reference weight not given (Veldre & Itra, 1991); **[2]** Mediterranean Sea, France (Milano et al., 1985); **[3]** Adriatic Sea, Italy, 1983 (Marcomini et al., 1986); **[4]** Ligurian Sea, Italy (Desideri et al., 1988); **[5]** Ketelmeer, Netherlands, 1987 (Netherlands' Delegation, 1991); **[6]** North Sea, Netherlands, within 70 km from the coast; 1987–88 (Compaan & Laane, 1992); **[7]** North Sea, United Kingdom, 1980 (Massie et al., 1985); **[8]** Great Barrier Reef, Australia, 1983 (Smith et al., 1987)

Analysed by high-performance liquid chromatography or gas chromatography

Maximum levels of 460 µg/kg benzo[*a*]pyrene and 400 µg/kg benzo[*e*]pyrene were determined in northern North Sea sediments in the vicinity of oil fields. The hydrocarbon concentrations were above the background levels only in water and sediments within a 2-km radius of platforms, where diesel-coated drill cuttings were dumped. The contribution of five- and six-ring compounds to the total PAH in sediments was unexpectedly high in samples unlikely to be contaminated by oil. Their source was probably windborne combustion products (Massie et al., 1985).

The following background concentrations have been reported in North Sea sediments: <0.01–20 µg/kg dry weight benzo[*a*]pyrene, <30 µg/kg fluoranthene, < 6 µg/kg benzo(*b*)fluoranthene plus benzo(*k*)fluoranthene, < 5 µg/kg benzo[*ghi*]-perylene, and < 3 µg/kg indeno[1,2,3-*cd*]pyrene (Compaan & Laane, 1992).

5.1.3.4 *Estuarine sediments*

The concentrations of individual PAH in estuarine sediments in 1981–92 (Table 45) varied widely, with maximum values in the high microgram per gram range. Measurements in sediments from the Continental Shelf of the Atlantic Ocean and the Gironde Estuary, France, showed relatively little contamination with PAH when compared with sediments from more polluted European estuaries (Garrigues et al., 1987). The levels of PAH in estuarine sediments in the United Kingdom were 10–500 µg/kg. Higher amounts of fluoranthene (1000–1900 µg/kg) and pyrene (790 µg/kg) were reported in estuaries of the River Mersey and the River Tamar (Readman et al., 1986).

The total PAH concentrations in sediments from the Penobscot Bay region of the Gulf of Maine, USA, ranged from 290 to 8800 µg/kg, with a distinct gradient that decreased seawards. The PAH composition was uniform throughout Penobscot Bay. Particulates of combustion products transported in the atmosphere were suggested to be a major source of PAH contamination. The levels in Penobscot Bay sediments were significantly higher than expected for an area previously considered to be uncontaminated and fell within the range found in industrialized regions throughout the world (Johnson et al., 1985).

The Saguenay Fjord is the major tributary that empties into the St Lawrence River estuary, and the area is highly industrialized. The PAH concentrations were maximal near the aluminium smelting plants that dominate the industrial sector and which were considered to be the major source of PAH, and the levels decreased with distance from this industrial zone. The concentrations of benzo[*a*]pyrene, benzo[*e*]pyrene, fluoranthene, benzo[*b*]fluoranthene, benzo[*j*]-fluoranthene, benzo[*k*]fluoranthene, chrysene and triphenylene, pyrene, indeno[1,2,3-*cd*]pyrene, benz[*a*]anthracene, dibenz[*a,h*]anthracene, perylene, benzo[*ghi*]perylene, and dibenzo[*a,e*]pyrene in sediments from the Saguenay Fjord ranged from 2000 to 3800 µg/kg (dry or wet weight basis not given) (Martel et al., 1986).

Table 45. Polycyclic aromatic hydrocarbon concentrations (μg/kg) found in estuarine sediments

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Acenaphthene			NR		NR		210–670		310		
Acenaphthylene			NR		NR		<10–160				
Anthracene		0.1–18	10–50	30–210	11–93	ND–49	60–860		610		
Benz[*a*]anthracene	10–790	0.2–68	30–160	30–650	23–189	14–540	70–3200	5–140	2000		
Benzo[*a*]fluoranthene			NR		NR			2–150			
Benzo[*a*]pyrene	10–560	<0.1–52	30–210	30–760	33–313	10–540	160/7200	4–150	2300	60–6800	20–60
Benzo[*b*]fluoranthene		0.2–79	100–500		53–346	17–1000					
Benzo[*e*]pyrene	10–620	103	40–180	30–550	56–323		120–8200	1–150	2500		
Benzo[*ghi*]perylene		1–72	120–490	70–410	66–403	23–641	<70–4200	3–96	1300		
Benzo[*k*]fluoranthene		<0.1–24	20–100		33–189	14–696					
Chrysene	20–1210	0.2–46	30–180		37–263	9–578			2900		
Cyclopenta[*cd*]pyrene			NR		NR		300/830				
Dibenz[*a,h*]anthracene		0.5–12	NR		8–50	2–120	550–4900		470		
Fluoranthene	30–1920	1–100	50–180	80–1880	85–506	156–3700	60–7200	14–410	3900		
Fluorene		15	40–120		NR		15–1500		390		
Indeno[1,2,3-*cd*]pyrene	20–630	61	60–240	30–420	50–343	9–228	<130–9000			1800	
1-Methylphenanthrene			NR		NR				240		
Naphthalene		43	NR		NR		80–2200		400		

Table 45 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]
Perylene		2–52	NR		NR		270/880		650	60–4200	50–60
Phenanthrene	30–1470	0.5–74	40–130	60–790	119–413	17–252	60–8700	5–300	2400		
Pyrene	20–1980	0.5–102	50–220	60–1510	93–425	16–539	50–5400	4–380	4800		

ND, not detected; /, single measurements; NR, not reported; **[1]** Estuarine sediment of the River Elbe, Germany (Japenga et al., 1987); **[2]** Continental Shelf and Gironde estuary, France (Garrigues et al., 1987); **[3]** Wadden Sea, Netherlands, 1988 (Compaan & Laane, 1992); **[4]** Mersey, Dee and Tamar estuaries, United Kingdom, 1984 (Readman et al., 1986); **[5]** Humber Estuary/the Wash, United Kingdom, 1990 (Compaan & Laane, 1992); **[6]** Gulf of Maine, Penobscot Bay, USA, 1982 (Johnson et al., 1985); **[7]** Great Lake tributaries, USA, 1984 (Fabacher et al., 1991); [8] Chesapeake Bay; USA, 1984–86 (Huggett et al., 1988); [9] Puget Sound, USA (Varanasi et al., 1992); **[10]** Yarra River estuary, Australia, 1976; analytical method: thin-layer chromatography with fluorescence detector (Bagg et al., 1981); **[11]** Mallacoota Inlet, Australia, 1976; analytical method: thin-layer chromatography with fluorescence detector (Bagg et al., 1981)

Analysed by high-performance liquid chromatography or gas chromatography and concentration in micrograms per kilogram dry weight, unless otherwise stated

5.1.3.5 *Harbour sediment*

The levels of individual PAH found in harbour sediments (Table 46) were higher than those in rivers, lakes, or oceans, concentrations ≤ 650 µg/g being reported.

5.1.3.6 *Time trends of PAH in sediment*

The PAH levels in sediments taken at various depths indicate changes and trends in the sources of PAH, e.g. from coal combustion to oil and gas heating. Measurements in sediments from Plöner Lake, Germany, showed that the concentration of PAH in samples from the northern part of the lake, which is in a populated region situated near a railway, had increased fivefold since 1920, whereas those in the southern part had remained constant. The increase in the northern part was attributed to an increase in the number of PAH emitters. As most of the PAH in the sediment originated from coal combustion, the concentrations decreased when coal-fired railway engines were replaced in this area. The benzo[*a*]pyrene levels ranged from 240 to 2400 µg/kg dry weight (Grimmer & Böhnke, 1975). These findings are consistent with the results of time-dependent analyses of sediments from Lake Constance (Müller et al., 1977).

A general trend in decreasing PAH concentrations from north to south was found in bottom sediments from the main stem of Chesapeake Bay, USA, thought to be due to the higher human population density in the northern region. Most of the compounds appeared to be derived from the combustion or high-temperature pyrolysis of carbonaceous fuels rather than from crude or refined oils. The levels of PAH remained relatively constant over the period 1979–86 at the locations examined. Naturally occurring PAH usually comprised less than 20% of the total; the finding of higher proportions may reflect riverine transport of older sediments to the area or scouring and removal of recently deposited sediments. The benzo[*a*]pyrene concentrations were 12–150 µg/kg dry weight (Huggett et al., 1988). Similar results were reported for sediments from Buzzard's Bay, USA (Hites et al., 1977).

In a study of PAH in sediment samples from the lagoon of Venice, Italy, a historical reconstruction of the PAH depositions in a dated drilling core made it possible to distinguish between natural and anthropogenic combustion and between different PAH sources, including direct petroleum spills and sedimentary diagenesis. The predominance of unsubstituted homologues and the relative abundance of some individual components suggested combustion as the predominant source. The lowest values determined in the deepest strata were assumed to be background concentrations resulting from pre-industrial pyrolytic sources, such as forest fires and wood burning. The benzo[*a*]pyrene levels were 2.2–17 µg/kg dry weight (Pavoni et al., 1987).

Table 46. Polycyclic aromatic hydrocarbon concentrations (µg/kg) found in harbour sediments

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Acenaphthene			<260–2509					50	3800
Acenaphthylene			<240–2700						
Anthracene			<30–27 200	1800/1700	ND–507		110–17 000	120	10 900
Benz[a]anthracene			<50–1991	3400/3400			310–20 000	240	8800/414 000
Benzo[*a*]pyrene	600–1500	400	<30–16 486	1800/2100	<70–94 984		300–19 000	340	8900/109 000
Benzo[*b*]fluoranthene		450	<35–17 182		ND–4103		410–15 000		
Benzo[*e*]pyrene				930/930			120–11 000		
Benzo[*ghi*]perylene		300	<35–1079				210–12 000		
Benzo[*k*]fluoranthene		200	<35–1430				150–22 000		
Chrysene			<30–13 900	3900/3800			580–21 000		
Coronene						130			
Fluoranthene	2000–3600	850	<70–21 566	900/5800	<5–84 514			640	34 200/60 700
Fluorene			<60–24 530			370	810–65 000	100	7000
Indeno[1,2,3-*cd*]pyrene		300	<50–372				180–14 000	160	157 000/715 000
1-Methylphenanthrene				2100/2300					
Naphthalene			<310–1564	1300/2000	<10–43 628			400	198 000

Table 46 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Perylene				1100/1200					
Phenanthrene			<50–5001	4200/4000	45–63 683			510	26 000/655 000
Pyrene			<70–5179	6300/6400	196–66 831		610–40 000	740	22 800/413 000

ND,not detected; /, single measurements; **[1]** Rotterdam, Netherlands (Japenga et al., 1987); **[2]** Rotterdam, Netherlands, 1990 (Netherlands' Delegation, 1991); **[3]** Hampton Roads, USA, 1982 (Alden & Butt, 1987); **[4]** New York Bight, USA, 1979; reference weight not given (Boehm & Fiest, 1983); **[5]** Boston, USA (Shiaris & Jambard-Sweet, 1986); **[6]** Black Rock, USA (Rogerson, 1988); **[7]** Various harbours of the Rhine, Germany, 1990 (Hamburg Environment Office, 1993); **[8]** Vancouver Harbour, Canada (Environment Canada, 1994); **[9]** Various harbours near steel mills, Canada (Environment Canada, 1994)

Analysed by high-performance liquid chromatography or gas chromatography and concentration in micrograms per kilogram dry weight, unless otherwise stated

5.1.4 Soil

A rough distinction can be made between local sources of pollution (point sources) and diffuse sources. Point sources can obviously give rise to significant local contamination of soil, whereas diffuse sources usually affect more widespread areas, though to a lesser extent. The main sources of PAH in soil are:

— atmospheric deposition after local emission, long-range transport, and pollution from combustion gases emitted by industry, power plants, domestic heating, and automotive exhausts (Hembrock-Heger & König, 1990; König et al., 1991) and from natural combustion like forest fires (Hites et al., 1980);
— deposition from sewage (sewage sludge and irrigation water) and particulate waste products (compost) (Hembrock-Heger & König, 1990; König et al., 1991); and
— carbonization of plant material (Grimmer et al., 1972).

The extent of soil pollution by PAH also depends on factors such as the cultivation and use of the soil, its porosity, its lipophilic surface cover, and its content of humic substances (Windsor & Hites, 1978). There is a correlation between the organic content of a soil and the PAH concentration: humus contains more PAH than a soil with little humic content, such as sand (Grimmer et al., 1972; Matzner et al., 1981; Grimmer, 1993).

This section addresses PAH in soil resulting mainly from industrial sources, automobile exhaust, and other diffuse sources and gives background values. Attribution of a study to a particular section was difficult, as the sources of PAH emissions are often mixed.

5.1.4.1 *Background values*

Table 47 gives background levels of PAH in soil in rural areas. In non-polluted areas, PAH concentrations were usually in the range 5–50 µg/kg.

5.1.4.2 *Industrial sources*

The PAH levels in soil resulting mainly from industrial sources are given in Table 48.

The PAH levels were determined in soil near one American plant where animal by-products and brewer's yeast had been processed since 1957. The operation had subsequently expanded to include the handling of solvents, flue dust, chips, acids, cyanides, and a wide variety of industrial waste. Extremely high PAH concentrations were found in the soil (Aldis et al., 1983).

PAH were detected in the soil at the sites of former coking plants in Canada (Environment Canada, 1994). For example in Lasalle, Quebec, the benzo[*a*]-pyrene levels in 1985 ranged from none detected to 1300 µg/g dry weight. The

Table 47. Polycyclic aromatic hydrocarbon concentrations (µg/kg dry weight) in soil of background and rural areas

Compound	[1]	[2]	[3]	[4]
Acenaphthene	1.7	< 1–21		
Acenaphthylene			ND/3.0	
Anthracene			1.2/4.2	
Benzo[*a*]pyrene	15	6–12	13/22	ND–4.0
Benzo[*b*]fluoranthene			14/25	
Benzo[*ghi*]perylene			49/28	ND–3.3
Benzo[*k*]fluoranthene				0.2–3.3
Fluoranthene	22	8–28	35/73	ND–28
Fluorene	ND	< 1–10		
Indeno[1,2,3-*cd*]pyrene				0.5–4.0
Naphthalene	46	13–60	3.8/11	
Phenanthrene	30	17–21	18/39	ND–76
Pyrene	20	9–25	29/42	

ND, not detected; /, single measurements; **[1]** Norway (depth, 0–10 cm), reference weight not given (Vogt et al., 1987); **[2]** Norway (Aamot et al., 1987); **[3]** Wales, United Kingdom (depth, 5 cm) (Jones et al., 1987); **[4]** Green Mountain (depth, 0–5 cm), USA (Sullivan & Mix, 1985)

Analysed by high-performance liquid chromatography or gas chromatography

facility closed in 1976, and by 1991 the benzo[*a*]pyrene concentration was below 10 000 µg/kg. In Pincher Creek, Alberta, high levels of alkylated PAH were measured after a refinery was dismantled. Maximum concentrations of 260 µg/g dry weight each of fluoranthene and pyrene were measured; benzo[*a*]pyrene was not detected.

PAH profiles were found to depend on the depth of soil from which the samples were taken. A comparison of soil samples from an area of clean air and from an industrialized area showed that the concentrations of PAH with lower boiling-points (fluoranthene, chrysene, and pyrene) decreased with depth, whereas those of PAH with higher boiling-points (indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, and coronene) were relatively greater. The opposite would have been expected on the basis of the solubility of these PAH (Jacob et al., 1993b).

5.1.4.3 *Diffuse sources*

(a) Motor vehicle and aircraft exhaust

The concentrations of individual PAH in soil resulting mainly from motor vehicle exhaust (Table 49) usually range between 1 and 2000 µg/kg. The PAH

Table 48. Polycyclic aromatic hydrocarbon concentrations (µg/kg dry weight) in soil near industrial emissions

Compound	[1]	[2]	[3]	[4]
Acenaphthene		54	5 090 000	
Anthracene	144 000		1 600	70
Benz[*a*]anthracene	79 000		200 000	
Benzo[*a*]pyrene	38 000	321		100
Benzo[*b*]fluoranthene				200
Benzo[*e*]pyrene	35 000			
Benzo[*ghi*]perylene				100
Benzo[*k*]fluoranthene			130 000	100
Chrysene			1 210 000	
Fluoranthene	340 000	573	234 000	200
Fluorene		80	8 600 000	
Indeno[1,2,3-*cd*]pyrene				100
Naphthalene		48	5 200	2.4
Perylene	12 000			
Phenanthrene	506 000	353	20 000 000	40
Pyrene	208 000	459	16 000 000	100

[1] Near coal gasification plant, Netherlands, concentrations in µg/kg wet weight (de Leeuw et al., 1986); **[2]** Norway, reference weight not given (Vogt et al., 1987); **[3]** Near processing plant, USA, 1982; maximum (Aldis et al., 1983); values, analytical method, and reference weight not given; **[4]** Area of an abandoned coal gasification plant, USA; reference weight not given (Dong & Greenberg, 1988)

Analysed by high-performance liquid chromatography or gas chromatography

content of soil often decreased with increasing depth (Matzner et al., 1981; Wang & Meresz, 1982; Butler et al., 1984). Near a motorway in the Midlands, United Kingdom, PAH were determined at depths of 0–4 cm and 4–8 cm. Extremely high concentrations were found in the surface layer, but soil at a depth of 4–8 cm was two times less contaminated (Butler et al., 1984). The pollution may have been a result of airborne transport or of microbial or photochemical degradation (Hembrock-Heger & König, 1990). Comparably high levels of PAH were found at Reykjavik Airport, Iceland (Grimmer et al., 1972; see Table 49).

(b) Other diffuse sources

Table 50 gives the levels of PAH from unpecified sources in soil. Benzo[*a*]pyrene levels of 800 µg/kg were found in humus, 100–800 µg/kg in garden soil, 35 µg/kg in forest soil, and 0.8–10 µg/kg in sand (Fritz, 1971).

Table 49. Polycyclic aromatic hydrocarbon concentrations (µg/kg dry weight) in soil of areas predominantly polluted by vehicle exhaust

Compound	[1]	[2]	[3]	[4]	[5]
Acenaphthylene				71	
Anthracene	0.2			13	11
Anthanthrene	0.4	149			
Benz[*a*]anthracene	2.3	430	169–3297		13
Benzo[*a*]pyrene	3.2	785	165–3196	38	24
Benzo[*b*]fluoranthene				41	
Benzo[*e*]pyrene	4.5	870	159–2293		29
Benzo[*ghi*]perylene	7.1	1450		168	46
Benzo[*k*]fluoranthene					78
Chrysene	4.1	436	251–2703		39
Coronene	1.8	410	40–322		37
Dibenz[*a,h*]anthracene	1.1	351			2
Fluoranthene	6.5	1290	200–3703	91	37
Fluorene					5
Indeno[1,2,3-*cd*]pyrene					36
Naphthalene					3
Perylene	0.6	157			6
Phenanthrene	17	1735		92	45
Pyrene	3.5	1610	145–4515	72	61

[1] Iceland (depth, 20 cm; reference weight not given) (Grimmer et al., 1972); **[2]** Reykjavik Airport, Iceland (surface soil; reference weight not given) (Grimmer et al., 1972); **[3]** United Kingdom, surface soil near motorway; analytical method, adsorbance measurement, reference weight not given) (Butler et al., 1984); **[4]** United Kingdom (urban soil; depth, 5 cm) (Jones et al., 1987); **[5]** Brisbane, Australia (Pathirana et al., 1994)

Analysed by high-performance liquid chromatography or gas chromatography

The PAH concentrations of cultivated soil were slightly higher than those in virgin soil. For example, the benzo[*a*]pyrene concentrations were 65–87 µg/kg in cultivated soil and 54 µg/kg in virgin soil (Wang & Meresz, 1982). The PAH levels in cultivated soils from German gardens at a maximum depth of 25 cm decreased from industrial areas (fluoranthene, 590–2500 µg/kg; benzo[*a*]pyrene, 220–1400 µg/kg) to rural areas (fluoranthene, 100–390 µg/kg; benzo[*a*]pyrene, 30–150 µg/kg) and with soil depth (benzo[*a*]pyrene concentration, 280–3000 µg/kg at 0–30 cm, 60–4600 µg/kg at 30–60 cm, and 10–7900 µg/kg at 60–100 cm). High PAH concentrations were found at a depth of 100 cm in soil from an old industrial area and from an area filled with contaminated soil. In compost soil, benzo[*a*]pyrene was present

Table 50. Polycyclic aromatic hydrocarbon concentrations (µg/kg dry weight) in soil from areas polluted by various diffuse sources

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Acenaphthylene					NR	NR		3.8		
Anthracene					NR	NR	ND–1.4		22–70	
Anthanthrene					27	0.50	ND		10–38	
Benz[*a*]anthracene					80	0.60	ND		47–101	
Benzo[*a*]pyrene	273	10/6.2	24	0.8–357	116	1.50	ND–1.4	157	54–108	
Benzo[*b*]fluoranthene									49–97	
Benzo[*e*]pyrene	23	20/22	50		143	3.10	ND–5.0		47–116	
Benzo[*ghi*]perylene	106	15/33	32	0.9–339	98	3.0	ND		64–147	
Benzo[*k*]fluoranthene									31–62	
Chrysene					NR	NR	ND–2.1		50–128	
Coronene					49	0.70	ND–1.7		32–66	
Dibenz[*a*,*h*]anthracene	266	8.4/22	44		44	0.60	ND–1.4		11–29	
Fluoranthene				2.5–444	254	2.1	ND–2.1	83	73–170	0.3–75
Fluorene					NR	NR		14		
Indeno[1,2,3-*cd*]pyrene	30	6.4/7.9	21.4	1.2–545	127	3.3			32–80	
Naphthalene					NR	NR		58		

Table 50 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Perylene	3537	4.0/8.5	5.0		NR	NR	ND		19–71	
Phenanthrene					NR	NR	ND–18	78	31–106	
Pyrene					150	0.80	ND–0.5	90	80–183	0.1–64

ND, not detected; /, single measurements; NR, not reported; **[1]** Germany, birch tree peat (Ellwardt, 1976); **[2]** Germany, black and white peat (Ellwardt, 1976); **[3]** Germany, sandy loam (Ellwardt, 1976); **[4]** Solling mountain, Germany; depth, 0–15 cm; analytical method, high-performance thin-layer chromatography; reference weight not given (Matzner et al., 1981); **[5]** Germany, forest, brown soil, surface layer (Bachmann et al., 1994); **[6]** Germany, forest, brown soil; depth, 0–2 cm (Bachmann et al., 1994); **[7]** Iceland; depth, 3–30 cm; reference weight not given (Grimmer et al., 1972); **[8]** Norway, bog soil; depth, 0–10 cm; reference weight not given (Vogt et al., 1987); **[9]** Toronto, Canada, virgin and cultivated soil; reference weight not given (Wang & Meresz, 1982); **[10]** Nova Scotia, Canada (Windsor & Hites, 1978)

Analysed by high-performance liquid chromatography or gas chromatography

at a concentration of 0.10–2.5 mg/kg in 1986 and 0.02–1.3 mg/kg in 1987 (Crössmann & Wüstemann, 1992).

Fluoranthene and pyrene were measured in soil samples, from a wooded area in Maine, a marshy area of South Carolina, a grassy, uncultivated meadow in Nebraska, a mossy area with pine needles in Wyoming, and a sandy desert area in Nevada, USA, and in dark brown, red clay, and light brown loam from Samoa. The highest levels of individual PAH (up to 80 µg/kg) were found in the soil from the wooded area in Maine. The levels in the marshy and grassy soils of South Carolina and Nebraska were 8.4–26 µg/kg. The other soils sampled contained fluoranthene and pyrene at levels < 1 µg/kg (Hites et al., 1980).

In Iceland, the concentrations of individual PAH in lava soil at depths of 3 and 25 cm were near the limit of detection. Similar levels were found in vegetable soil at depths of 10 and 30 cm, but the concentrations at 10 cm were twice as high as those at 30 cm (Grimmer et al., 1972).

Higher levels of PAH were found in the humus layer of spruce and beech forest ecosystems than in the mineral soil, but the spruce stand contained and stored more PAH than the beech stand (Matzner et al., 1981). Forest soils in Germany contain many PAH in large amounts; Table 48 shows the PAH concentrations in one forest brown soil. The first humic layer of the soil had the highest PAH concentration, and the level decreased with depth to below the limit of detection in layers below 10 cm (Bachmann et al., 1994).

The concentrations of PAH were no higher in soil that had been treated with sewage sludge than in untreated soil, indicating that sewage sludge is not a major source of PAH (Hembrock-Heger & König, 1990; König et al., 1991).

5.1.4.4 *Time trends of PAH in soil*

Soil samples collected from Rothamsted Experimental Station in southeast England over a period of about 140 years (1846–1980) were analysed for PAH (Jones et al., 1987). All of the soils were collected from the plough layer (0–3 cm) of an experimental plot for which atmospheric deposition was the only source of PAH. The total PAH burden of the plough layer had increased by approximately fivefold since 1846. The concentrations of most of the individual PAH (anthracene, anthanthrene, fluorene, benzo[*a*]pyrene, benzo[*e*]pyrene, fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, chrysene, pyrene, indeno[1,2,3-*cd*]pyrene, phenanthrene, and benz[*a*]-anthracene) had increased by about one order of magnitude. For example, the benzo[*a*]pyrene level was 18 µg/kg in 1846 and 130 µg/kg in 1980, and the anthracene level was 3.6 µg/kg in 1846 and 13 µg/kg in 1980. The levels of coronene, acenaphthylene, acenaphthene, perylene, and benzo[*ghi*]perylene remained approximately the same, whereas the naphthalene content decreased from 39 µg/kg in 1846 to 23 µg/kg in 1980.

5.1.5 Food

In the past, benzo[*a*]pyrene was the most common PAH determined in foods and was used as an indicator of the presence of PAH (Tilgner, 1968). The earliest measurements of PAH, in particular of benzo[*a*]pyrene, date to 1954; these were reviewed by Lo & Sandi (1978) and by Howard & Fazio (1980). The levels of individual PAH in foods in more recent studies are summarized in Tables 51–56.

5.1.5.1 *Meat and meat products*

The concentrations of individual PAH found in meat are shown in Table 51. In a comparison of home and commercially smoked meats in Iceland, very little benzo[*a*]pyrene was detected in smoked sausage and mutton, but considerable amounts of benzo[*a*]pyrene and other PAH were found in home-smoked mutton and lamb, independently of whether they were covered with cellophane or muslin. About 60–75% of the total benzo[*a*]pyrene was detected in the superficial (outer) layers of the meat (Thorsteinsson, 1969). These findings are in agreement with those of Rhee & Bratzler (1970) for smoked bologna and bacon and with those of Tilgner (1958) and Gorelova et al. (1960).

The amount of PAH formed during roasting, baking, and frying depends markedly on the conditions (Lijinsky & Shubik, 1964). In an investigation of the effect of the method of cooking meat, including broiling (grilling) on electric or gas heat, charcoal broiling, and broiling over charcoal in a no-drip pan, it was shown that the formation of PAH can be minimized by avoiding contact of the food with flames, cooking meat at lower temperatures for a longer time, and using meat with minimal fat (Lijinsky & Ross, 1967). The most likely source of PAH is melted fat that drips onto the heat and is pyrolysed (Lijinsky & Shubik, 1965). The exact chemical mechanism for the formation of PAH is unknown.

In one study, the highest concentration of benzo[*a*]pyrene (130 µg/kg) in cooked meat was found in fatty beef, and the concentration appeared to be proportional to the fat content (Doremire et al., 1979). Levels of about 50 µg/kg were detected in a charcoal-grilled T-bone steak (Lijinsky & Ross, 1967), in heavily smoked ham (Toth & Blaas, 1972), and in various other cooked meats (Potthast, 1980). Usually, benzo[*a*]pyrene levels up to 0.5 µg/kg have been found (Prinsen & Kennedy, 1977).

In meat, poultry, and fish in Canada, benzo[*k*]fluoranthene was detected at concentrations up to 0.30 µg/kg and benzo[*a*]pyrene up to 1.1 µg/kg (Environment Canada, 1994).

Benzo[*a*]pyrene was found in some German meat products in 1994 at concentrations generally < 1 µg/kg. The highest concentration, 9.2 µg/kg, was found in a ham from the Black Forest (State Chemical Analysis Institute, Freiburg, 1995).

Table 51. Polycyclic aromatic hydrocarbon concentrations (μg/kg fresh weight) in meat and meat products

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]	[12]	[13]
Anthracene		0.9									20–31[a]	ND–2	0.5–133
Anthanthrene											5–8	ND	ND–66.5
Benz[a]anthracene	0.5	0.5	0.02–0.64	0.03		Trace–0.33[a]	0.02–0.03	0.04–0.38	0.04–0.13	0.05	16–37	ND–1	0.2–144
Benzo[*a*]fluorene											17–28	1–2	ND–174
Benzo[*a*]pyrene	0.1	0.6	0.02–0.45	0.02	0.05	0.01–0.14	0.01–0.04	0.04–0.26	0.03–0.26	0.05	26–42	ND–1	ND–212
Benzo[*b*]fluoranthene	0.3	1.0			0.30					0.04	16–24		ND–92.3
Benzo[*b*]fluorene											10–12	2–7	ND–71.9
Benzo[*c*]phenathrene		1.4	0.03–0.36	0.06		Trace– 0.18	0.03–0.04	0.05–0.21	0.05–0.10				
Benzo[*e*]pyrene										0.03	6–9	ND–2	ND–80.9
Benzo[*ghi*]perylene	0.2	0.6	0.03–0.31	0.03	3.75	Trace– 0.12	0.03–0.04	0.06–0.27	0.05–0.19	0.05	10–17	ND–2	ND–153
Benzo[*j*]fluoranthene											5–7		
Benzo[*k*]fluoranthene	0.2	0.2			0.05					0.01	8–14		ND–172[b]
Chrysene		0.6								0.15			0.3–140[a]
Dibenz[*a,h*]anthracene										0.01			ND–8.8
Fluoranthene	0.9	1.1			7.8					0.48	57–103	6–9	1.1–376
Indeno[1,2,3-*cd*]pyrene	0.2	0.7	0.04–0.38	0.03	2.5	Trace–0.11	0.01–0.03	0.04–1.40	0.05/0.1		15–22	ND–5	ND–171
1-Methylphenanthrene											4–5	ND–3	0.5–57.6
Perylene											ND–3	ND	ND–27.9

Table 51 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]	[12]	[13]
Phenanthrene		3.0									22–64	10–16	3.5–618
Pyrene										0.55	38–63	5–7	1.2–452

ND, not detected; /, single measurements; **[1]** Poultry and eggs, Netherlands, reference weight not given (de Vos et al., 1990); **[2]** Meat and meat products, Netherlands, reference weight not given (de Vos et al., 1990); **[3]** Smoked beef, Netherlands, reference weight not given (de Vos et al., 1990); **[4]** Unsmoked beef, Netherlands (de Vos et al., 1990); **[5]** Bacon, United Kingdom (Crosby et al., 1981); **[6]** Smoked meat, United Kingdom (McGill et al., 1982); **[7]** Unsmoked meat, United Kingdom (McGill et al., 1982); **[8]** Smoked sausages, United Kingdom (McGill et al., 1982); **[9]** Unsmoked sausages, United Kingdom (McGill et al., 1982); **[10]** Meat, United Kingdom, reference weight not given (Dennis et al., 1983); **[11]** Mesquite wood cooked pattie (70–90 % lean), USA, reference weight not given (Maga, 1986); **[12]** Hardwood charcoal cooked pattie (70–90% lean), USA, reference weight not given (Maga, 1986); **[13]** Grilled sausages, Sweden, reference weight not given (Larsson et al., 1983)

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[a] In sum with triphenylene

[b] In sum with benzo[*j*]fluoranthene

5.1.5.2 *Fish and other marine foods*

Benzo[*a*]pyrene was found at levels ranging from none detected to 18 µg/kg in smoked fish. The differences were probably due to factors such as the type of smoke generator, the temperature of combustion, and the degree of smoking (Draudt, 1963). The highest concentration of benzo[*a*]pyrene (130 µg/kg) in seafood was found in mussels from the Bay of Naples (Bourcart & Mallet, 1965), and a level of about 60 µg/kg was detected in smoked eel skin. Most of the fish analysed contained 0.1–1.5 µg/kg (Steinig, 1976). Benzo[*a*]pyrene was also detected at levels up to 3.3 µg/kg in 21 samples of smoked fish, oysters, and mussels of various origins (Prinsen & Kennedy, 1977). The levels of individual PAH are summarized in Table 52.

5.1.5.3 *Dairy products: cheese, butter, cream, milk, and related products*

PAH were detected in considerable amounts in smoked cheese (Prinsen & Kennedy, 1977; Lintas et al., 1979; McGill et al., 1982; Osborne & Crosby, 1987a). The benzo[*a*]pyrene content of a smoked Italian Provola cheese was 1.3 µg/kg (Lintas et al., 1979). Concentrations of 0.01–5.6 µg/kg fresh weight fluoranthene, benz[*a*]anthracene, benzo[*c*]phenanthrene, benzo[*a*]pyrene, benzo[*ghi*]perylene, and indeno[1,2,3-*cd*]pyrene were found in a smoked cheese sample and 0.01–0.06 µg/kg in unsmoked cheese from the United Kingdom (McGill et al., 1982). In other unsmoked cheese samples from the United Kingdom, the individual PAH levels were between < 0.01 µg/kg for dibenz[*a*,*h*]anthracene and 1.5 µg/kg for pyrene. Similar concentrations of PAH were found in British butter and cream samples (Dennis et al., 1991).

In Finnish butter samples, most of the measured PAH (phenanthrene, 1-methylphenanthrene, fluoranthene, pyrene, benzo[*a*]fluorene, benzo[*ghi*]-fluoranthene, cyclopenta[*cd*]pyrene, perylene, anthanthrene, benzo[*ghi*]pyrene, and indeno[1,2,3-*cd*]pyrene) occurred at levels ≤ 0.1 µg/kg. The maximum level was 1.4 µg/kg fluoranthene (Hopia et al., 1986).

The concentrations of fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*e*]pyrene, perylene, benzo[*ghi*]pyrene, indeno[1,2,3-*cd*]pyrene, and dibenz[*a*,*h*]anthracene were measured in milk, milk powder, and other dairy products in Canada (Lawrence & Weber, 1984), the Netherlands (de Vos et al., 1990), and the United Kingdom (Dennis et al., 1983, 1991). The concentrations ranged from < 0.01 µg/kg for benzo[*k*]fluoranthene and dibenz[*a*,*h*]anthracene to 2.7 µg/kg for pyrene.

Canadian infant formula was found to contain 8.0 µg/kg fluoranthene, 4.8 µg/kg pyrene, 1.7 µg/kg benz[*a*]anthracene, 0.7 µg/kg benzo[*b*]fluoranthene, 1.2 µg/kg benzo[*a*]pyrene, 0.6 µg/kg perylene, 0.3 µg/kg anthanthrene, and 1.2 µg/kg indeno[1,2,3-*cd*]pyrene (Lawrence & Weber, 1984). Slightly lower levels were detected in British samples in 1982–83 (Dennis et al., 1991).

Table 52. Polycyclic aromatic hydrocarbon concentrations (µg/kg) found in fish and marine foods

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Acenaphthene						
Anthracene	0.9	1.3–64.3	1.4–49.6			
Benz[*a*]anthracene	1.3	ND–11.2	ND–6.3		ND–86	Trace–0.09
Benzo[*a*]pyrene	1.4	ND–5.5	ND–5.4	0.10	ND–18	Trace–0.35
Benzo[*b*]fluoranthene	2.0	ND–3.9	ND–3.6	0.35		
Benzo[*c*]phenanthrene					ND–15	0.01–0.09
Benzo[*e*]pyrene		ND–2.8	ND–3.0			
Benzo[*ghi*]perylene	0.9	ND–2.8	ND–1.8	4.3	ND–25	Trace–0.39
Benzo[*k*]fluoranthene	0.7	ND–6.7[a]	ND–5.1[a]	0.10		
Chrysene	2.9	ND–13.0[b]	ND–9.4[b]			
Dibenz[*a,h*]anthracene						
Fluoranthene	1.8	1.4–79.9	1.7–48.4	2.4		
Fluorene						
Naphthalene						
Indeno[1,2,3-*cd*]pyrene	1.6	ND–7.1	ND–2.4	2.7	ND–37	ND–0.33
Perylene		ND–1.2	ND–1.0			
Phenanthrene	3.5	5–330	10.4–277			
Pyrene		1.3–67.8	2.1–38.4			

ND, not detected; NR, not reported; **[1]** Fish, Netherlands (de Vos et al., 1990); **[2]** Herring, whitefish, mackerel, eel, salmon, salmon trout, various fillets; all smoked; Sweden (Larsson, 1982); **[3]** Fish and fish products: sprats, herring, rainbow trout, caviar, herring paste, salmon paste; all smoked or canned; Sweden (Larsson, 1982); **[4]** Kippers, United Kingdom (Crosby et al., 1981); **[5]** Fish (smoked), United Kingdom, concentration in µg/kg wet weight (McGill et al., 1982); **[6]** Fish, unsmoked, United Kingdom, concentration in µg/kg wet weight (McGill et al., 1982)

PAH were detected at levels of 0.003–0.03 µg/kg in human milk (Heeschen, 1985).

5.1.5.4 *Vegetables*

The levels of PAH found in vegetables in recent studies are listed in Table 53.

Fluoranthene, but no other PAH, was reported to have been found in unspecified fruits and vegetables in Canada at levels of none detected to 1.8 µg/kg (Environment Canada, 1994). Kale was found to contain high concentrations of fluoranthene (120 µg/kg), pyrene (70 µg/kg), chrysene (62 µg/kg), and benz[*a*]anthracene (15 µg/kg), and PAH concentrations up to 7 µg/kg were determined in other vegetables (Vaessen et al., 1984). The differences in PAH content have been attributed to variations in the ratio of

Table 52 (contd)

Compound	[8]	[9]	[10]	[11]	[12]	[13]	[14]
Acenaphthene			< 2–5.13	2.22–22.3			
Anthracene			< 2–78.4	ND –5.88	ND–0.6	ND–1.9	< 0.05
Benz[*a*]anthracene	0.14	1.6–7.5	< 2	0.14–5.31	0.8–3.0	0.8–20.9	
Benzo[*a*]pyrene	0.13	t–4.5	< 2–7.63	ND–5.33	0.4–1.0	0.2–12.2	< 0.004
Benzo[*b*]fluoranthene	0.13			0.13–5.77	4.5–12.2[c]	1.2–24.3[c]	
Benzo[*e*]pyrene	0.12				2.4–6.3	0.7–7.6	
Benzo[*ghi*]perylene	0.12			0.17–30.9	0.4–0.8	0.3–5.7	
Benzo[*k*]fluoranthene	0.04				NR	NR	< 0.002
Chrysene	0.65		< 2	ND–15.9	3.2–8.8[b]	3.9–30.8[b]	< 0.03
Dibenz[*a,h*]anthracene	0.03			0.21–39.3	0.1–0.2[d]	<0.1–0.5[d]	
Fluoranthene	0.1		< 2–123.5	ND–32.7	5.1–17.5	4.5–18.7	
Fluorene			< 2–18.5	ND–65.7			
Naphthalene			< 2–67.4	2.06–156.1			
Indeno[1,2,3-*cd*]pyrene				0.28–28.6	0.3–0.6	0.2–6.4	
Perylene					0.2–2.7	0.1–3.1	< 0.05
Phenanthrene			< 2–100.8	5.84–87.2	2.1–4.2	1.9–19.6	
Pyrene	0.79		< 2–144.9	ND–68.0	3.1–12.4	2.6–11.2	< 0.03

ND, not detected; NR, not reported; **[8]** Fish, United Kingdom (Dennis et al., 1983); **[9]** Fish, Nigeria (Emerole et al., 1982); **[10]** Fresh fish from the Arabian Gulf: *andag, sheim, gato, sheiry, faskar, chaniedah*; after an oil spill (Al-Yakoob et al., 1993); **[11]** Fresh fish and shrimps, Kuwait, after Gulf war (Saeed et al., 1995); **[12]** Fresh oysters, various origins, concentration in µg/kg wet weight (Speer et al., 1990); **[13]** Canned or smoked oysters and mussels, various origins, concentration in µg/kg wet weight (Speer et al., 1990); **[14]** Clam, Australia; analytical method: fluorescence spectrophotometry; concentration in µg/kg wet weight (Smith et al., 1987)

Analysed by high-performance liquid chromatography or gas chromatography; reference weight not given, unless otherwise stated

[a] In sum with benzo[*j*]fluoranthene
[b] In sum with triphenylene
[c] Benzo[*b+j+k*]fluoranthenes
[d] Dibenz[*a,h+a,c*]anthracenes

surface area:weight, in location (rural or industrialized), and in growing season. Washing (at 20 °C) vegetables contaminated by vehicle exhausts did not reduce the PAH contamination (Grimmer & Hildebrandt, 1965).

In a comparison of the PAH contents of terrestrial plants grown in chambers containing 'clean air' and in the open field, the contamination was shown to be due almost exclusively to airborne PAH, which were not synthesized by the plants (Grimmer & Düvel, 1970).

Table 53. Polycyclic aromatic hydrocarbon concentrations (µg/kg) in vegetables

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Anthracene		0.09–0.19		<0.1–0.3				
Benz[*a*]anthracene	15			0.7–4.6	0.05–3.17	0.05–3.2	0.4	0.3
Benzo[*a*]fluoranthene		0.08–2.6						
Benzo[*a*]pyrene	4.2	0.05–1.4	5.6	0.3–6.2	ND–1.42	0.05–3.0		0.2
Benzo[*b*]fluoranthene			6.1	0.5–7.3		0.9–3.2	0.2	
Benzo[*b*]fluorene		0.11–2.8						
Benzo[*c*]phenanthrene	9.2				0.05–1.5			
Benzo[*e*]pyrene	7.9	0.07–2.2		0.5–6.7				0.2
Benzo[*ghi*]perylene	7.7	0.13–2.1	10	0.5–10.8	ND–1.39	3.7–10	0.1	
Benzo[*k*]fluoranthene			3.7			ND–17	0.1	
Chrysene	62					2.4–4.0	0.8	0.5
Dibenz[*a,h*]anthracene	1.0							0.04
Dibenzo[*a,h*]pyrene	0.7							
Dibenzo[*a,i*]pyrene	0.3							
Fluoranthene	117	1.1–28	28	2.8–9.1		9.2–17		
Indeno[1,2,3-*cd*]pyrene	7.9	0.14–0.72	2.4	0.3–8.3	ND–1.92	1.8–4.2		
1-Methylphenanthrene		0.10–2.1		0.7–1.6				
Perylene		0.05–0.75		<0.1–1.7				
Phenanthrene		0.47–12		1.8–7.5				
Pyrene	70	0.9–18		3.4–10.4				

ND, not detected; **[1]** Kale, Netherlands (Vaessen et al., 1984); **[2]** Lettuce, Finland, concentration in µg/kg fresh weight (Wickström et al., 1986); **[3]** Lettuce, Germany, from an industrial area (Ministry of Environment, 1994); **[4]** Lettuce, Sweden, concentration in µg/kg fresh weight (Larsson & Sahlberg, 1982); **[5]** Lettuce and cabbage, United Kingdom, concentration in µg/kg fresh weight (McGill et al., 1982); **[6]** Lettuce, India (Lenin, 1994); **[7]** Potatoes, Netherlands (de Vos et al., 1990); **[8]** Tomatoes, Netherlands (Vaessen et al., 1984)

Analysed by high-performance liquid chromatography or gas chromatography; reference weight not given, unless otherwise stated

The benzo[*a*]pyrene levels in potatoes in eastern Germany were 0.2–400 µg/kg. The highest concentrations were detected in the peel of potatoes grown in soil containing 400 µg/kg benzo[*a*]pyrene, 750 µg/kg benzo[*e*]pyrene, 1000 µg/kg benz[*a*]anthracene, 600 µg/kg chrysene, 160 µg/kg dibenz[*a*,*h*]anthracene, 1000 µg/kg benzo[*b*]fluoranthene, 2300 µg/kg phenanthrene, 1800 µg/kg pyrene, 220 µg/kg benzo[*k*]fluoranthene, 500 µg/kg indeno[1,2,3-*cd*]pyrene, 2500 µg/kg fluoranthene, and 120 µg/kg anthracene (Fritz, 1971, 1972, 1983).

High concentrations of PAH were detected in lettuce grown close to a highway; the levels of individual PAH decreased with distance from the road. Washing the vegetables reduced their content of high-molecular-mass PAH but not of phenanthrene (Larsson & Sahlberg, 1982). In another study, the profiles of PAH in lettuce were similar to those in ambient air, indicating that deposition of airborne particles was the main source of contamination (Wickström et al., 1986).

PAH concentrations were determined in fenugreek, spinach beet, spinach, amaranthus, cabbage, onion, lettuce, radish, tomato, and wheat grown on soil that had been treated with sewage sludge. The levels of individual PAH in lettuce leaves (Table 53) were one to two orders of magnitude lower than those in the sewage sludge and the soil on which the lettuce was grown (Lenin, 1994).

The PAH levels in carrots and beans grown near a German coking plant were below 0.5 µg/kg wet weight. The levels of fluoranthene were 1.6–1.7 µg/kg and those of pyrene 1.0–1.1 µg/kg. Vegetables with large, rough leaf surfaces, such as spinach and lettuce, had PAH levels that were 10 times higher, perhaps due to deposition from ambient air (Crössmann & Wüstemann, 1992).

5.1.5.5 *Fruits and confectionery* (Table 54)

Higher concentrations of PAH were found in fresh fruit than in canned fruit or juice, and especially high concentrations of phenanthrene (17 µg/kg) and chrysene (69 µg/kg) were found in nuts (de Vos et al., 1990). In 1982–83 in the United Kingdom, high PAH levels were found in samples of puddings, biscuits, and cakes, which were probably derived from vegetable oil. Similar concentrations of individual PAH were detected in samples of British chocolate (Dennis et al., 1991).

5.1.5.6 *Cereals and dried foods*

Wheat, corn, oats, and barley grown in areas near industries contained higher levels of PAH than crops from more remote areas. Drying with combustion gases increased the contamination by three- to 10-fold; use of coke as fuel resulted in much less contamination than oil (Bolling, 1964). Cereals contained benzo[*a*]pyrene at levels of 0.2–4.1 µg/kg (Table 55). The highest

concentrations, up to 160 µg/kg, were found in smoked cereals (Tuominen et al., 1988).

The PAH concentration in rye grown near a highway with high traffic density decreased slightly 7–25 m away from the road (Larsson, 1982).

5.1.5.7 *Beverages*

Benzo[*a*]pyrene was found at 0.8 µg/kg in coffee powder, 0.01 µg/litre in brewed coffee, 9.51 µg/kg in tea leaves, and 0.02 µg/litre in brewed tea (Lintas et al., 1979). In 40 samples of tea leaves from India, China, and Morocco, the concentration of benzo[*a*]pyrene was generally 2.2–60 µg/kg, although concentrations up to 110 µg/kg were found in smoked teas (Prinsen & Kennedy, 1978).

In samples of whisky and beer, the concentrations of six of 11 PAH (benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]-perylene, dibenz[*a,h*]anthracene, and indeno[1,2,3-*cd*]pyrene) were below or slightly above 0.01 µg/kg. The highest level determined (0.24 µg/kg) was that of pyrene (Dennis et al., 1991). The PAH content of the water used in the preparation of whisky and beer was not described.

5.1.5.8 *Vegetable and animal fats and oils*

The levels of PAH in oil products, butter, and margarine are listed in Table 56. Vegetable oils are reported to be naturally free of PAH, and contamination is due to technological processes like smoke drying of oil seeds or environmental sources such as exhaust gases from traffic. The PAH content of native olive oils was particularly high (Speer et al., 1990). The PAH content of coconut, soya bean, maize, and rapeseed oil was radically reduced during refining, particularly by treatment with activated charcoal (Larsson et al., 1987). This method is now widely used (Dennis et al., 1991).

Benzo[*a*]pyrene was detected in 30 vegetable oils from Italy and France in 1994, including 17 grape-seed oils and one pumpkin-seed oil. The average concentration was 59 µg/kg, and the maximum value was 140 µg/kg. Benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, dibenz[*a,h*]anthracene, and indeno[1,2,3-*cd*]pyrene were also found in measurable amounts. The source of these high levels was the smoke in drying ovens (State Chemical Analysis Institute, Freiburg, 1995).

Lard and dripping were found to contain levels of individual PAH ranging from < 0.01 µg/kg dibenz[*a,h*]anthracene) to 6.9 µg/kg fluoranthene (Dennis et al., 1991). High PAH levels were found in margarine samples in studies in Finland (Hopia et al., 1986), the Netherlands (Vaessen et al., 1988), New Zealand (Thomson et al., 1996), and the United Kingdom (Dennis et al., 1991) (see Table 56).

Table 54. Polycyclic aromatic hydrocarbon concentrations (µg/kg) in fruits and confectionery

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
Anthracene				0.4		0.3	
Benz[*a*]anthracene	0.5		0.11	4.2	0.2	4.2	0.08–2.73
Benzo[*a*]pyrene		0.1	0.07	0.2	0.3	0.4	0.04–2.20
Benzo[*b*]fluoranthene	0.1	0.1	0.06	0.4	0.4	3.5	0.03–1.27
Benzo[*c*]phenanthrene	0.5			12		2.2	
Benzo[*e*]pyrene			0.03				0.08–2.92
Benzo[*ghi*]fluoranthene	0.9			0.9			
Benzo[*ghi*]perylene		0.1	0.06	0.4	1.1	0.2	0.11–2.55
Benzo[*k*]fluoranthene	0.1	0.1	0.02	0.1	0.1	0.5	0.04–1.36
Chrysene	0.5		0.23	69	0.5	36	0.09–2.84
Dibenzo[*a,h*]pyrene		0.01					< 0.01–0.23
Fluoranthene	3.6	1.0	0.93	3.0	1.9	2.3	0.52–3.57
Indeno[1,2,3-*cd*]pyrene				0.4	0.4	0.2	0.10–3.18
Phenanthrene	7.8			17	2.9	3.2	
Pyrene			0.83				0.59–2.37

[1] Fresh fruit, Netherlands (de Vos et al., 1990);**[2]** Canned fruit and juices, Netherlands (de Vos et al., 1990);**[3]** Fruit and sugar, United Kingdom (Dennis et al., 1983); **[4]** Nuts, Netherlands (de Vos et al., 1990);**[5]** Biscuits, Netherlands (de Vos et al., 1990); **[6]** Sugar and sweets, Netherlands (de Vos et al., 1990); **[7]** Puddings, biscuits and cakes, United Kingdom (Dennis et al., 1991)

Analysed by high-performance liquid chromatography or gas chromatography; reference weight not given

Table 55. Polycyclic aromatic hydrocarbon concentrations (μg/kg) in cereals and dried foods

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Acenaphthene		1.6	NR	NR				0.7	2.3
Anthracene		9.4	NR	NR				1.3	19/150
Anthanthrene			NR	NR					
Benz[*a*]anthracene	0.1–4.2	11	0.69	0.11–0.21	2.5/3.7	0.6	0.3	<0.1/0.2	6.3/110
Benzo[*a*]pyrene	ND–0.3	5.4	0.40	0.10–0.12	0.5/0.8	0.2		0.3/0.4	0.6/160
Benzo[*b*]fluoranthene	0.1–0.5		0.28	0.07–0.09	0.9	0.2	0.1		
Benzo[*e*]pyrene			0.42	0.06–0.17				0.1/0.7	
Benzo[*ghi*]perylene			0.54	0.13–0.20					
Benzo[*k*]fluoranthene			0.50	0.1–0.14					
Dibenz[*a,h*]anthracene	ND–1.2		0.06	0.01–0.02	3.6				
Fluoranthene	0.8–26	130	0.71	0.58–0.69	18/28	1.9	1.4	1.5/13	70/790
Fluorene		5.9	NR	NR				2.3/2.7	6.4/87
Indeno[1,2,3-*cd*]pyrene	ND–0.4		1.08	0.24–0.33	1.4	0.2			
Perylene	0.1-0.4	0.7	NR	NR		94	NR	NR	14/2983/1
Pyrene	1.1-48	47	0.10	0.38-0.62	20/21	2.2	3.4	1.6/5.4	60/630

ND, not detected; /, single measurements; NR, not reported; **[1]** Barley malt, Canada (Lawrence & Weber, 1984); **[2]** Bran, Finland (Tuominen et al., 1988); **[3]** Bran, United Kingdom (Dennis et al., 1991); **[4]** High bran and granary bread, United Kingdom (Dennis et al., 1991); **[5]** Bran, Canada (Lawrence & Weber, 1984); **[6]** Corn bran, Canada (Lawrence & Weber, 1984); **[7]** Flaked milled corn, Canada (Lawrence & Weber, 1984); **[8]** Oats, Finland (Tuominen et al., 1988); **[9]** Smoked oats, barley and beans, Finland (Tuominen et al., 1988)

Analysed by high-performance liquid chromatography or gas chromatography; reference weight not given

Table 55 (contd)

Compound	[10]	[11]	[12]	[13]	[14]	[15]	[16]	[17]
Acenaphthene				0.6/0.7		NR	NR	0.6
Anthracene				0.5		NR	NR	
Anthanthrene			0.05–0.08			NR	NR	
Benz[*a*]anthracene	0.4	ND–0.2	0.14–0.25	< 0.1/<0.1	0.3–0.8	0.06–0.15	0.33–1.26	0.1
Benzo[*a*]pyrene		< 0.1	0.17–0.30	0.2/0.4	0.1	0.03–0.05	0.15–0.34	
Benzo[*b*]fluoranthene					0.1/0.2	0.02–0.05	0.1–0.27	
Benzo[*c*]phenanthrene						NR	NR	
Benzo[*e*]pyrene		ND–0.1	0.16–0.29	0.2/0.4		0.06–0.16	0.28–0.81	
Benzo[*ghi*]fluoranthene			0.05			NR	NR	
Benzo[*ghi*]perylene			0.20–0.35			0.06–0.08	0.15–0.28	
Benzo[*k*]fluoranthene		ND–0.2[a]				0.02–0.07	0.15–0.31	
Chrysene		0.3–0.7[b]				NR	NR	
Coronene			0.03–0.06			NR	NR	
Cyclopenta[*cd*]pyrene			0.07–0.13			NR	NR	
Dibenz[*a,h*]anthracene			0.03–0.05		3.0	< 0.01	0.01–0.02	
Fluoranthene	2.9	0.9–1.3	0.32–0.57	1.8/3.0	1.5–7.4	0.22–0.60	0.82–6.17	3.8
Fluorene				1.3/1.7		NR	NR	2.0
Indeno[1,2,3-*cd*]pyrene			0.16–0.29		3.0	0.08–0.15	0.30–0.65	
1-Methylphenanthrene		0.3						
Perylene	0.1			< 0.1/0.1	0.1–0.3	NR	NR	

Table 55 (contd)

Compound	[10]	[11]	[12]	[13]	[14]	[15]	[16]	[17]
Phenanthrene		1.3-1.5		9.9/10		NR	NR	14
Pyrene	2.8	1.6-2.3	0.22-0.39	1.6/5.5	2.6-8.5	0.26-1.18	1.41-10.86	2.6

ND, not detected; /, single measurements; NR, not reported; **[10]** Whole grain oats, Canada (Lawrence & Weber, 1984); **[11]** Whole-grain rye, Sweden, concentration in µg/kg fresh weight (Larsson, 1982); **[12]** Wheat grain, United Kingdom (Jones et al., 1989b); **[13]** Wheat, Finland (Tuominen et al., 1988); **[14]** Wheat, Canada (Lawrence & Weber, 1984); **[15]** Breakfast cereal, United Kingdom (Dennis et al., 1991); **[16]** Bran-enriched cereals, United Kingdom (Dennis et al., 1991); **[17]** Bolted rye flour, Finland (Tuominen et al., 1988)

Analysed by high-performance liquid chromatography or gas chromatography; reference weight not given, unless otherwise specified

[a] Benzofluoranthenes
[b] In sum with triphenylene

Table 55 (contd)

Compound	[18]	[19]	[20]	[21]	[22]	[23]	[24]	[25]
Acenaphthene	NR	NR		NR				
Anthracene	NR	NR		NR				
Anthanthrene	NR	NR		NR				
Benz[*a*]anthracene	0.04–0.19	0.64	0.8	0.10–0.14	0.5	0.1	0.4	
Benzo[*a*]pyrene	0.02–0.09	0.43	0.8	0.05–0.15	0.2	0.3	0.8	
Benzo[*b*]fluoranthene	0.02–0.06	0.25	1.2	0.04–0.06	0.5	0.6	1.0	0.05
Benzo[*c*]phenanthrene	NR	NR		NR			0.7	
Benzo[*e*]pyrene	0.10–0.23	0.35		0.06–0.12				
Benzo[*ghi*]fluoranthene	NR	NR		NR				
Benzo[*ghi*]perylene	0.06–0.19	0.39	0.5	0.04–0.21	0.5	0.9	0.6	
Benzo[*k*]fluoranthene	0.03–0.08	0.35	0.6	0.04–0.1	0.1	0.3	0.5	0.08
Chrysene	NR	NR	1.0	NR	2.0		1.3	0.4
Coronene	NR	NR		NR				
Cyclopenta[*cd*]pyrene	NR	NR		NR				
Dibenz[*a,h*]anthracene	<0.01–0.01	0.05		<0.01–0.01				
Fluoranthene	0.07–0.40	0.66	2.8	0.23–2.03	3.7	0.6	2.5	
Fluorene	NR	NR		NR				
Indeno[1,2,3-*cd*]pyrene	0.06–0.24	0.84	0.6	0.11–0.25	0.3	0.6	0.5	
1-Methylphenanthrene								
Perylene	NR	NR		NR				

Table 55 (contd)

Compound	[18]	[19]	[20]	[21]	[22]	[23]	[24]	[25]
Phenanthrene	NR	NR	3	NR	4.2	3.0	2.1	
Pyrene	0.04–0.88	0.67		0.23–0.87				

NR, not reported; **[18]** White flour, United Kingdom (Dennis et al., 1991); **[19]** Granary flour, United Kingdom (Dennis et al., 1991); **[20]** Bread, Netherlands (de Vos et al., 1990); **[21]** White bread, 1982–83, United Kingdom (Dennis et al., 1991); **[22]** Noodles, pizza, Netherlands (de Vos et al., 1990); **[23]** Potato products, Netherlands (de Vos et al., 1990); **[24]** Rice, macaroni, Netherlands (de Vos et al., 1990); **[25]** Soups, Netherlands (de Vos et al., 1990)

Analysed by high-performance liquid chromatography or gas chromatography; reference weight not given

Table 56. Polycyclic aromatic hydrocarbon concentrations (µg/kg) in vegetable oils and related products

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Acenaphthene	NR	< 0.02–45	NR	NR	NR	NR	0.29	NR	NR	< 0.1–11
Anthracene	NR	< 0.02–460	<0.1–0.1	ND–4.8	ND–8	NR	0.04–0.92	NR	NR	< 0.2–5.6
Anthanthrene	Trace–0.1	NR	NR	NR	NR	NR	0.03–0.53	NR	NR	< 0.1–2.7
Benz[*a*]anthracene	NR	NR	0.7–6.1	ND–6.1	ND	0.30–7.46	NR	0.22–3.98	0.28–0.96	< 0.1–5.2
Benzo[*a*]fluorene	NR	< 0.02–130	NR	NR	ND–2	NR	0.07–3.8	NR	NR	NR
Benzo[*a*]pyrene	Trace–0.3	< 0.02–24	0.5–2.3	ND–4.1	ND	0.29–4.92	0.05–2.2	0.19–6.0	0.17–0.83	< 0.2–5.2
Benzo[*b*]fluoranthene	Trace–0.1	< 0.02–91[a]	NR	ND–8.9[a]	ND	0.20–2.39	NR	0.16–3.0	0.09–0.37	< 0.2–9.2
Benzo[*b*]fluorene	NR	< 0.02–45	NR	NR	ND	NR	0.03–2.1	NR	NR	NR
Benzo[*e*]pyrene	NR	< 0.02–23	0.7–2.4	ND–3.8	ND	0.26–6.06	0.09–2.1	0.42–6.11	0.36–0.87	NR
Benzo[*ghi*]fluoranthene	NR	< 0.02–1.3	NR	NR	ND	NR	0.14–4.9	NR	NR	NR
Benzo[*ghi*]perylene	NR	< 0.02–10	0.5–1.7	ND–4.2	NR	0.06–5.23	0.02–1.4	0.38–5.21	0.17–1.16	< 0.2–10.6
Benzo[*k*]fluoranthene	NR	NR	NR	NR	ND	0.24–3.17	NR	0.20–3.40	0.16–0.55	< 0.1–11.4
Chrysene	NR	NR	NR	0.1–8.6[b]	ND	0.39–10.3	NR	0.26–7.36	0.31–0.97	< 0.2–7.5
Coronene	NR	< 0.02	NR	NR	NR	NR	NR	NR	NR	NR
Cyclopenta[*cd*]pyrene	NR	< 0.02–1.4	NR	NR	ND	NR	0.10–1.1	NR	NR	NR
Dibenz[*a,h*]anthracene	0.7–1.1	< 0.02–1.1[c]	NR	ND–0.2[c]	NR	<0.01–0.82	NR	0.05–1.02	0.04–0.11	< 0.1–9.2
Fluoranthene	0.2–7.5	< 0.02–460	1.2–4.8	0.2–18.2	3–15	0.21–12.4	0.52–9.0	0.09–4.50	0.44–1.56	< 0.1–1.6
Fluorene	NR	< 0.02–200	NR	NR	ND–7	NR	0.08–1.6	NR	NR	< 0.2–2.1
Indeno[1,2,3-*cd*]pyrene	Trace–0.5	< 0.02–0.85	0.3–1.7	ND–4.3	NR	0.59–6.78	0.03–1.1	0.49–9.14	0.43–1.17	< 0.2–9.7
Naphthalene	NR	NR	NR	NR	NR	NR	NR	NR	NR	< 0.2–52
1-Methylphenanthrene	NR	< 0.02–190	NR	NR	NR	NR	0.08–1.8	NR	NR	NR
Perylene	Trace–0.2	< 0.02–5.9	0.1–0.4	ND–0.9	NR	NR	0.02–0.57	NR	NR	NR

Table 56 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Phenanthrene	NR	0.09–1400	0.9–1.6	ND–69.4	4–38	NR	0.29–6.0	NR	NR	< 0.2–4.6
Pyrene	0.2–1.4	< 0.02–330	1.1–4.2	0.1–13.6	2–14	0.58–17.2	0.59–15	0.29–6.03	0.44–1.88	< 0.1–1.7

ND, not detected; /, single measurements; NR, not reported; **[1]** Corn oil, canola, soya bean oil (Lawrence & Weber, 1984); **[2]** Corn oil, coconut oil (crude and deodorized), olive oil, soya bean oil, sunflower oil, sesame oil, flaw oil, wheatseed oil (Hopia et al., 1986); **[3]** Coconut oil (pure) (Sagredos et al., 1988); **[4]** Various olive oils, safflower oils, sunflower oils, maize germ oils, sesame oil, linseed oil, wheat germ oil (all native) (Speer et al., 1990); **[5]** Various olive oils (Menichini et al., 1991b); **[6]** Various unspecified oils (Dennis et al., 1991); **[7]** Four cooking margarines, seven table margarines (Hopia et al., 1986); **[8]** Margarine (Dennis et al., 1991); **[9]** Low-fat spread (Dennis et al., 1991); **[10]** Margarine (Thomson et al., 1996)

Analysed by high-performance liquid chromatography or gas chromatography

[a] Benzo[*b*+*j*+*k*]fluoranthenes
[b] In sum with triphenylene
[c] Dibenz[*a*,*h*+*a*,*c*]anthracenes

5.1.6 *Plants*

PAH with low molecular masses are more readily taken up by vegetation than those with higher molecular masses (Wang & Meresz, 1982).

In a study of PAH levels in soil (see section 5.1.4), leaf litter, and soil fauna (see section 5.1.7) from a roadside in Brisbane, Australia, vegetation height, soil depth, and distance from the roadside were found to be important in the distribution of PAH. The concentration of PAH in leaf litter declined exponentially with distance from the roadway, few PAH being detectable 30 m away. A decrease in PAH levels with height was found in the roadside vegetation canopy. In leaf litter, fluorene, phenanthrene, fluoranthene, pyrene, chrysene, benzo[*k*]fluoranthene, and benzo[*ghi*]perylene were present at concentrations of about 100 µg/kg wet weight. Naphthalene, benz[*a*]anthracene, benzo[*e*]pyrene, benzo[*a*]pyrene, and indeno[1,2,3-*cd*]pyrene were present at about 50 µg/kg wet weight, whereas anthracene was present at concentrations below 10 µg/kg wet weight. Perylene and dibenz[*a,h*]anthracene were not detectable. The tree *Casuarina littorina* contained high levels of pyrene and chrysene (100 µg/kg wet weight each) and benzo[*k*]fluoranthene (72 µg/kg wet weight); the concentrations of fluoranthene, phenanthrene, and benzo[*ghi*]-perylene were about 40 µg/kg wet weight. Perylene, indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, and coronene were not detectable (Pathirana et al., 1994).

The benzo[*a*]pyrene levels in spruce sprouts from a rural area of Germany (Bornhövede, Schleswig-Holstein) decreased from 2.6 µg/kg in 1991 to 1.3 µg/kg in 1993. The concentrations of PAH with low boiling-points significantly decreased between 1991 and 1993: for example, that of fluoranthene decreased from 44 µg/kg in 1991 to 11 µg/kg in 1993, perhaps due to a decrease in coal burning. The levels of phenanthrene, fluoranthene, pyrene, and benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene were about 10 µg/kg; those of benzo[*ghi*]fluoranthene, benzo[*c*]phenanthrene, benz[*a*]anthracene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]perylene, and coronene were about 2 µg/kg; and those of anthracene, dibenz[*a,h*]anthracene, and anthanthrene were < 1 µg/kg. The PAH levels in spruce sprouts from the Saarland, an industrial area in Germany, were about 10 times higher than those in the Bornhöveder area, although these levels also decreased between 1991 and 1993: from 5.9 to 4.1 µg/kg for benzo[*a*]pyrene and 97 to 58 µg/kg for fluoranthene. the concentrations of pyrene were 40–50 µg/kg, those of benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene were 20 µg/kg, and those of benzo[*ghi*]perylene, benzo[*c*]phenanthrene, benz[*a*]anthracene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, anthanthrene, and coronene were < 10 µg/kg (Jacob & Grimmer, 1994, 1995). In 1994, the PAH levels had decreased further. Overall, a 25% decrease in the

PAH levels in spruce sprouts was seen over the previous 10 years (Jacob & Grimmer, 1995).

The PAH profiles in spruce sprouts and poplar leaves were reasonably similar in areas with clean air (Bavarian forests) and in industrialized areas (Saarland) of Germany, indicating that one emission source is predominantly responsible for air pollution by PAH. Hard-coal combustion resulted in a characteristic PAH profile (Jacob et al., 1993a).

The concentrations of PAH in pine needles from Dübener Heide near Leipzig (Saxony, Germany) were similar to those from the Bornhöveder area (Schleswig-Holstein, Germany), with an average benzo[*a*]pyrene level of 2.3 µg/kg (Jacob & Grimmer, 1995).

Beech leaves from the Harz mountains in Germany contained fluoranthene at a level of 5 µg/kg, whereas the concentrations of phenanthrene, pyrene, benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, anthracene, benz[*a*]anthracene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, anthanthrene, and coronene were all < 2 µg/kg. Beech sprouts in an industrial area in eastern Germany contained 10–15 times higher levels of PAH, with fluoranthene at about 60 µg/kg, pyrene at about 30 µg/kg, benzo[*b*]fluoranthene plus benzo[*j*]-fluoranthene plus benzo[*k*]fluoranthene at about 10 µg/kg, and anthracene, benz[*a*]anthracene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]perylene, coronene, dibenz[*a,h*]anthracene, and anthanthrene at < 2 µg/kg (Jacob & Grimmer, 1995).

Comparable results were obtained in poplar leaves: those from the Saarland analysed in 1989, 1991, and 1993 had 10 times lower concentrations of PAH than those in Dübener Heide. The concentrations of phenanthrene, fluoranthene, and pyrene were about 20 µg/kg, those of benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene were about 10 µg/kg, and those of anthracene, benz[*a*]anthracene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, anthanthrene, and coronene were < 5 µg/kg (Jacob & Grimmer, 1995).

5.1.7 *Animals*

5.1.7.1 *Aquatic organisms*

Aquatic invertebrates are known to adsorb and accumulate PAH from water (see section 4.1.5). The concentrations of PAH in aquatic organisms collected from various sites are listed in Tables 57–64. All of the sampling sites listed in Tables 57–60 were contaminated with industrial effluents, the major components of the PAH profile being benzo[*b*]fluoranthene, benz[*a*]anthracene, benzo[*a*]pyrene, benzo[*e*]pyrene, fluoranthene, pyrene, and phenanthrene. The average levels of PAH in aquatic organisms from these sites ranged from

1 to 100 µg/kg; the differences in levels generally corresponded to the degree of industrial and urban development and shipping movements.

The levels in holothurians from urban sewage were 1–15 mg/kg (Milano et al., 1986).

Concentrations of 1–5 mg/kg individual PAH were found in limpets (*Patella vulgata*) in the North Sea (Knutzen & Sortland, 1982). The PAH concentrations in two species of bivalves in Saudafjorden (Norway) near an iron alloy smelter decreased rapidly with distance from the source, but the compounds could still be detected more than 15 km away. High levels of individual PAH were reported in mussels (*Modiolus modiolus*), with maximum levels of 57 000 µg/kg benzo[*b*]fluoranthene, 25 000 µg/kg benz[*a*]anthracene, 23 000 µg/kg benzo[*e*]pyrene, 21 000 µg/kg benzo[*a*]pyrene, 20 000 µg/kg fluoranthene, 8200 µg/kg pyrene, 6000 µg/kg benzo[ghi]perylene, 4000 µg/kg perylene, 2900 µg/kg benzo[*a*]fluorene, 2300 µg/kg benzo[*b*]fluorene, 2200 µg/kg dibenz[*a*,*h*]anthracene, 2000 µg/kg benzo[*c*]phenanthrene, 1100 µg/kg phenanthrene, 524 µg/kg anthracene, and 360 µg/kg anthanthrene (Bjørseth, 1979). A very high level of anthracene (243 µg/kg) was found in mussels (*Mytilus edulis* L.) in the North Sea near the Dutch coast (Boom, 1987). Mussels in the USA frequently contained up to 500 µg/kg of individual PAH (Heit et al., 1980; Mix & Schaffer, 1983).

The levels of PAH in pooled mussel samples in 1986, 1988, and 1990 in Germany were about 10 µg/kg for fluoranthene, pyrene, chrysene plus triphenylene, benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, and benzo[*e*]pyrene and < 4 µg/kg for benzo[*ghi*]fluoranthene plus benzo[*c*]phenanthrene, benz[*a*]anthracene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, dibenz[*a*,*h*]anthracene, benzo[*ghi*]perylene, anthanthrene, and coronene. The levels were high in the winter months and low in summer, with minima in June and April. The authors concluded that this seasonal variation was due to more intensive metabolic activity (Jacob & Grimmer, 1994).

During 1978–79, the average total PAH concentrations in two subpopulations of softshell clams were 555 µg/kg in the industrialized bayfront area of Coos Bay, Oregon, and 76 µg/kg in a more remote environment. During 1979–80, low-molecular-mass, readily water-soluble PAH were one or two orders of magnitude more concentrated then high-molecular-mass, less water-soluble PAH in mussels (*M. edulis*) (Mix & Schaffer, 1983).

Individual PAH levels of 1–20 mg/kg were found in the hepatopancreas of lobsters (*Homarus americanus*) in the south arm of Sydney Harbour, Canada, near a coking plant (Sirota et al., 1983), and levels of the same order of magnitude were found in the digestive gland (Uthe & Musial, 1986). The levels in digestive gland, tail muscle, and hepatopancreas from lobsters from other areas of Canada were 100–1000 µg/kg (Sirota & Uthe, 1981; Sirota et al., 1983; Uthe & Musial, 1986).

Table 57. Polycyclic aromatic hydrocarbon concentrations (µg/kg dry weight) in bivalves and gastropods; main source, industrial emissions

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
Acenaphthene	ND	ND	7				2.1/8.8
Anthracene			9				9.0/25
Benz[*a*]anthracene	172	203	3		5–41	25–229	
Benzo[*a*]pyrene	12	21	1	8.1	2–8	Trace–28	2.6/2.8
Benzo[*b*]fluoranthene	23	25			3–30	48–90	
Benzo[*e*]pyrene	17	10			Trace–30	231–356	
Benzo[*ghi*]perylene	ND		4		5		
Benzo[*j*]fluoranthene		1.3					
Benzo[*k*]fluoranthene	2.3						
Chrysene	209	205					
Coronene			4				
Dibenzo[*a,e*]pyrene			2				
Dibenzo[*a,i*]pyrene			4				
Dibenzo[*a,l*]pyrene			Trace				
Fluoranthene	18	62	7		43–407	300–4992	26/61
Fluorene			2				1.3/6.3
1-Methylphenanthrene				2.9			
Naphthalene							15/3
Perylene			8				
Phenanthrene	733	462	9	4.4	115-258	55–2542	66/194
Pyrene	85	131	4		32-204	141–3128	23/40
Triphenylene	ND						

ND, not detected; *f*, single measurement; **[1]** Whole cooked clam (*Mya arenaria*); oil-contaminated area (tanker accident), Canada, 1979; concentration in µg/kg wet weight (Sirota & Uthe, 1981); **[2]** Whole cooked mussel (*Mytilus edulis*); oil-contaminated area (tanker accident), Canada, 1979; concentration in µg/kg wet weight (Sirota & Uthe, 1981); **[3]** Whole mussel (*Mytilus galloprovincialis*); Thermaikos Gulf, Aegean Sea, Greece (agricultural and industrial area); concentration in µg/kg wet weight (Iosifidou et al., 1982); **[4]** Whole scallop (*Amusium pleuronectes*); Gulf of Thailand, Thailand; reference weight not given (Hungspreugs et al., 1984); **[5]** Whole periwinkle (*Littorina littorea*); moderately polluted parts of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[6]** Whole limpet (*Patella vulgata*); moderately polluted parts of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[7]** Whole snails (*Thais haemostoma*), Pensacola Bay, USA; creosote contaminated; concentration in µg/kg wet weight (Rostad & Pereira, 1987)

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Table 58. Polycyclic aromatic hydrocarbon concentrations (µg/kg dry weight) in algae and water plants; main source, industrial emissions

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Benz[*a*]anthracene	5	4	31–325	45–431	3–40	
Benzo[*a*]pyrene	4	5	Trace–64	Trace–<2	2–20	
Benzo[*b*]fluoranthene	4	5	7–76	6-12	5–31	
Benzo[*e*]pyrene	7	14	Trace–100	Trace-8	8–50	410
Benzo[*ghi*]perylene		4				79
Fluoranthene	45	32	40–412	15–900	<4–236	
Phenanthrene	87	34	31–325	45–431	109–146	
Pyrene	36	20	28–286	15–388	<4–224	260

[1] *Laminaria saccharina* (whole); moderately polluted parts of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[2]** *Ceramium rubrum* (whole), moderately polluted parts of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[3]** Bladder wrack (*Fucus vesiculosus*, whole), moderately polluted parts of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[4]** Knotted wrack (*Ascophyllum nodosum*, whole), moderately polluted parts of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[5]** Toothed wrack (*Fucus serratus*, whole), moderately polluted parts of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[6]** Wakame seaweed, Japan (Obana et al., 1981a)

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High PAH levels were found in oysters (*Crassostrea virginica*) in Chesapeake Bay, USA, with maximum levels of 650 µg/kg pyrene, 450 µg/kg benzo[*e*]pyrene, 450 µg/kg fluoranthene, 290 µg/kg benzo[*a*]pyrene, 130 µg/kg benz[*a*]anthracene, 130 µg/kg perylene, 73 µg/kg benzo[*ghi*]-perylene, 70 µg/kg benzo[*c*]phenanthrene, 48 µg/kg naphthalene, 45 µg/kg phenanthrene, 40 µg/kg anthracene, and 20 µg/kg dibenz[*a,h*]anthracene. The levels of PAH in clams (*Rangia cuneata*) from Chesapeake Bay were 170 µg/kg benzo[*a*]pyrene, 170 µg/kg pyrene, 52 µg/kg fluoranthene, 15 µg/kg phenanthrene, 10 µg/kg perylene, 10 µg/kg benzo[*ghi*]perylene, 9 µg/kg benzo[*c*]phenanthrene, and 6 µg/kg benz[*a*]anthracene (Bender & Huggett, 1988).

Phenanthrene was found at 15 mg/kg in lampreys (*Petromyzon* sp.) in the Hersey River, USA, which was polluted with creosote used for wood preservation (Black et al., 1981).

The viviparous blenny (*Zoarces viviparus*) fish contained 0.06 µg/kg benzo[*a*]pyrene and 0.2–3.9 µg/kg phenanthrene and fluoranthene; the concentrations of other PAH were below the detection limit (0.01 µg/kg). In bream (*Abramis brama*) the levels were < 0.01–0.15 µg/kg benzo[*a*]pyrene

Table 59. Polycyclic aromatic hydrocarbon concentrations (µg/kg wet weight) in lobsters; main source, industrial emissions

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Acenaphthene	ND	ND				
Benz[*a*]anthracene	684	ND/23	1620-23 400	34–604	762–32 700	17–900
Benzo[*a*]pyrene	24	0.2/2.6	35-1000	2.0–40	711–1430	27–43
Benzo[*b*]fluoranthene	24	1	155–2350	6–78	1020–3820	29–835
Benzo[*e*]pyrene	57	5/8	415–9330	15–165	1550–3600	35–36
Benzo[*ghi*]perylene	ND	ND/2	7–493	1.6–31	232–769	10–20
Benzo[*k*]fluoranthene	7.6	0.3/0.6	43–588	1.6–25	502–955	15–26
Chrysene	445	ND	360–5050	5–79	252–1240	15–24
Fluoranthene	318	ND/0.2	1910–12400	103–545	4220–15 200	68–442
Indeno[1,2,3-*cd*]pyrene		5	38–855	3–45	486–931	12–40
Phenanthrene	1588	ND	Trace–3470	Trace–650		
Pyrene	488	ND	730–6710	32–265	2910–13 100	59–333
Triphenylene		ND/244	2520–23100	Trace–330		

ND, not detected; /, single measurements; **[1]** *Homarus americanus* (digestive gland), oil-contaminated area (tanker accident), Canada, 1979 (Sirota & Uthe, 1981); **[2]** *Homarus americanus* (tail muscle), oil-contaminated area (tanker accident), Canada, 1979 (Sirota & Uthe, 1981); **[3]** *Homarus americanus* (hepatopancreas), Sydney Harbour, near coking plant, Canada (Sirota et al., 1983); **[4]** *Homarus americanus* (tail muscle), Sydney Harbour, near coking plant, Canada (Sirota et al., 1983); **[5]** *Homarus americanus* (digestive gland), Sydney Harbour, near coking plant, Canada, 1982–84 (Uthe & Musial, 1986); **[6]** *Homarus americanus* (tail muscle), Sydney Harbour, near coking plant, Canada, 1982–84 (Uthe & Musial, 1986)

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Table 60. Polycyclic aromatic hydrocarbon levels (µg/kg dry weight) in fish and other aquatic species; main source, industrial emissions

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Acenaphthene	39			Trace–0.9	130	< 25			
Acenaphthylene	270			0.1–0.2					
Anthracene			ND	0.1–0.2	460	< 22		1000	
Benz[a]anthracene	22	ND–40	ND–< 0.1	0.1–88	1000	< 21	1–2	800	5
Benzo[*a*]fluorene				0.2–0.6				500	
Benzo[*a*]pyrene	7	0.07–8.4	ND–< 0.1	0.1–0.5	570	< 20		ND	8
Benzo[*b*]fluoranthene			< 0.1[a]						28
Benzo[*b*]fluorene				0.1–0.2					
Benzo[*c*]phenanthrene				Trace					
Benzo[*e*]pyrene	14		ND–< 0.1	0.1–1.6	840	< 25			25
Benzo[*ghi*]perylene			ND–< 0.1	0.2–18	75	< 25			23
Chrysene	61		< 0.1–2.1[b]		1500	< 22			
Dibenz[*a,h*]anthracene			ND–< 0.1[c]		< 100	< 25			
Fluoranthene	1800		0.1–9.1	1.2–5.6	4800	< 20	13–18	800	48
Fluorene				0.2–2.4	200	< 25		ND[c]	
Indeno[1,2,3-*cd*]pyrene			ND–< 0.1	0.3–3.7	150	< 25			
1-Methylphenanthrene					85	< 20			
Naphthalene				2.5–11	610	< 25			
Perylene	8		ND–< 0.1	Trace–0.2	75	< 20			

Table 60 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Phenanthrene	2700	28-15 313	0.1–2.4	0.7–9.1	1400	< 20	32–50	900	71
Pyrene	1500		ND–10.0	0.7–3.7	2300	< 20	10–8	800	39
Triphenylene								800	

ND, not detected; **[1]** Bullhead catfish (*Ictalurus nebulosus*, whole); Black River, USA, near coking plant; concentration in µg/kg wet weight (Vassilaros et al., 1982); **[2]** Whole fish (unspecified); Hersey River, USA, creosote polluted; concentration in µg/kg wet weight (Black et al., 1981); **[3]** Bream (fillet and liver); River Elbe, Germany, industrial region of city of Hamburg (Speer et al., 1990); **[4]** Dabs (*Limanda limanda*, muscle, North Sea, United Kingdom, near Beatrice oil platform; concentration in µg/kg wet weight (McGill et al., 1987); **[5]** English sole (*Parophrys vetulus*, stomach contents); Mukilteo, USA, near petroleum storage tanks (Malins et al., 1985); **[6]** English sole (*Parophrys vetulus*, liver); Mukilteo, USA, near petroleum storage tanks (Malins et al., 1985); **[7]** Whole starfish (*Asterias rubens*), moderately polluted areas of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982); **[8]** Whole holothurians, Toulon, France; urban sewage (Milano et al., 1986); **[9]** Whole crumb-of-bread sponge (*Halichondria panicea*); moderately polluted areas of North Sea coast, Norway, 1978–79 (Knutzen & Sortland, 1982)

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[a] Benzo[*b*+*j*+*k*]fluoranthenes
[b] In sum with triphenylene
[c] Dibenz[*a*,*h*+*a*,*c*]anthracenes

Table 61. Polycyclic aromatic hydrocarbon concentratrations (µg/kg dry weight) in bivalves (mussels and clams); background values

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Acenaphthene	NR							24/46
Acenaphthylene	NR							34/130
Anthracene	0.7–19	9–15	149–243					36/43
Benz[*a*]anthracene	NR			0.1–0.8	2.9/42	< 1	31/94	
Benzo[*a*]pyrene	4.6–451	3/5	<0.8–2		3.5/8.7	< 1	1.3/26	
Benzo[*b*]fluoranthene	3.0–120				1.5/12		2.5/18	
Benzo[*c*]phenanthrene	5.3–280				3.1/55	< 1	26/94	
Benzo[*e*]pyrene	NR		5–25					
Benzo[*ghi*]perylene	3.4–57				5.4/4.2	3	0.4/8.1	
Benzo[*k*]fluoranthene	1.0–43		1–2		2.6/9.6		1.7/17	
Chrysene	NR				7.6/27		86	
Coronene			< 10–24		1.3/2.7		0.7/4.6	
Dibenz[*a,h*]anthracene	NR				4.7/6.9		2.1/9.6	
Fluoranthene	16–288	23/43	8–23	0.7–7.2	11/111	17	47/180	72
Indeno[1,2,3-*cd*]pyrene	ND–9.9				5.9/3.9		0.3/5.7	
1-Methylphenanthrene	22–708							
Naphthalene	NR	5–4						51/120
Perylene	4.2–59		< 5–26			36		

Table 61 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Phenanthrene	21–570	7–109		0.1–1.7	12/155	18	108/216	
Pyrene	6.6–394	9–77	15–38	0.3–6.6	6.2/62	23	25/109	
Triphenylene	7.5–300				7.9/43		27/106	

ND, not detected; /, single measurements; NR, not reported; **[1]** Mussel (*Mytilus edulis*), Danish, German and Dutch Wadden Sea, 1989 (Compaan & Laane, 1992); **[2]** Mussel (*Mytilus edulis*); Finnish archipelago, Finland,1978–79; concentration in µg/kg wet weight (Rainio et al., 1986); **[3]** Mussel (*Mytilus edulis* L.); North Sea coast, Netherlands; concentration in µg/kg wet weight (Boom, 1987); **[4]** Hard shell clam (*Mercenaria mercenaria*), Rhode Island (seafood stores), USA; concentration in µg/kg wet weight (Pruell et al., 1984); **[5]** Softshell clam (*Mya arenaria*), Coos Bay, Oregon, USA, 1978–79; reference weight not given (Mix & Schaffer, 1983); **[6]** Clam (*Mya mercenaria*); Chesapeake Bay, USA, 1984 (Bender & Huggett, 1988); **[7]** Mussel (*Mytilus edulis*); Yaquina Bay, USA, 1979–80; concentration in µg/kg wet weight (Mix & Schaffer, 1983); **[8]** *Rangia cuneata*; Lake Pontchartrain, USA, 1980; concentration in µg/kg wet weight(McFall et al., 1985)

Table 61 (contd)

Compound	[9]	[10]	[11]	[12]	[13]	[14]	[15]
Acenaphthene							16
Acenaphthylene							18
Anthracene			< 0.05–3.2	<0.05			
Benz[*a*]anthracene	< 1–6	< 10			1.0–1.8	ND–2.3	
Benzo[*a*]pyrene	30–168	< 10	< 0.003–0.02	< 0.004	0.41–1.8	0.40–2.6	1.0
Benzo[*b*]fluoranthene					1.0–1.8	0.83–1.9	
Benzo[*c*]phenanthrene	< 1–9						
Benzo[*e*]pyrene							
Benzo[*ghi*]perylene	< 1–10		< 0.05–0.3	<0.05	0.53–1.9	0.83–2.3	
Benzo[*k*]fluoranthene			< 0.002–0.02	< 0.002	0.29–0.80	0.32–1.2	
Chrysene			< 0.03–1.4	<0.03			
Coronene							
Dibenz[*a,h*]anthracene							
Fluoranthene	< 1/52	< 1–370	< 0.04–0.70				
Fluorene							
Indeno[1,2,3-*cd*]pyrene							
1-Methylphenanthrene							
Naphthalene							
Perylene	< 1–10	< 10–300	< 0.01–0.08				

Table 61 (contd)

Compound	[9]	[10]	[11]	[12]	[13]	[14]	[15]
Phenanthrene	< 1–15	< 1–60					4.4
Pyrene	17/165	< 1–450	< 0.03–1.4	<0.03			
Triphenylene							

[9] *Rangia cuneaya*, Chesapeake Bay, USA, 1984 (Bender & Huggett, 1988); **[10]** *Lampsilus radiata*, *Elliptio complanatus*, *Anodonata grandis*; Lake George, Hearts Bay, USA (Heit et al., 1980); **[11]** *Tridacna maxima*; Great Barrier Reef, Australia, 1980–82; concentration in µg/kg wet weight (Smith et al., 1984); **[12]** Clam; Green Island, Great Barrier Reef, Australia, concentration in µg/kg wet weight (Smith et al., 1984); **[13]** Shortnecked clam; near Miyagi Prefecture, Japan, concentration in µg/kg wet weight (Takatsuki et al., 1985); **[14]** Mussel; near Miyagi Prefecture; Japan, reference weight not given (Takatsuki et al., 1985); **[15]** *Perna viridis*; Gulf of Thailand (mussel farm),Thailand, reference weight not given (Hungspreugs et al., 1984)

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Table 62. Polycyclic aromatic hydrocarbon concentrations (µg/kg wet weight) in bivalves (oysters); background values

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
Acenaphthene	46			< 0.2–2.0			16
Acenaphthylene	36			< 0.4–3.0			
Anthracene	44		< 1–40	< 0.08–0.9		< 0.25–4.2	
Benz[*a*]anthracene	9.9	0.3–12	< 1–135		1.1	1.5–10	
Benzo[*a*]pyrene		0.5–1.6	50–285	< 0.01–5	0.6–2.6	0.78	3.5
Benzo[*b*]fluoranthene		0.3–5.2		< 0.03–6	3.0–20	2.2	
Benzo[*c*]phenanthrene			< 1–70				
Benzo[*e*]pyrene			< 1–453		2.8–32		
Benzo[*ghi*]perylene		0.4–1.2	< 1–73	< 0.05–5	0.87	< 0.20–2.8	
Benzo[*k*]fluoranthene	12	0.1–0.9	< 0.06–5.1		1.2		< 0.01–< 3
Chrysene	58	1.3–14		< 0.1–3			
Dibenz[*a,h*]anthracene			< 1–20	< 0.01–< 4		< 0.06	
Fluoranthene	80	0.9–94	< 1–450	0.4–22			470
Fluorene	21			0.1–0.8			
Indeno[1,2,3-*cd*]pyrene		1.7		< 0.01–5			
1-Methylphenanthrene							3.5
Naphthalene	35		5–48	0.8–7			
Perylene			< 1–130				

Table 62 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
Phenanthrene	220	4.9–77	< 1–45	2–38			6.7
Pyrene	200	1.6–50	< 1–645	< 0.4–15	7.0–52		
Triphenylene							0.03

[1] *Crassostrea virginica*; Lake Pontchartrain, USA, 1980 (McFall et al., 1985); **[2]** *Crassostrea virginica*; Palmetto Bay (Marina), USA (Marcus & Stokes, 1985); **[3]** *Crassostrea virginica*; Chesapeake Bay, USA, 1983–84; concentration in µg/kg dry weight (Bender & Huggett, 1988); **[4]** *Saccostrea cucculata*; Mermaid Sound, Australia, 1982 (Kagi et al., 1985); **[5]** Oyster, Japan (local market); 1977-78 (Obana et al., 1981a); **[6]** Oyster, near Miyagi Prefecture, Japan; reference weight not given (Takatsuki et al., 1985); **[7]** *Ostrea plicatula*; Gulf of Thailand, Thailand; reference weight not given (Hungspreugs et al., 1984)

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Table 63. Polycyclic aromatic hydrocarbon concentrations (µg/kg wet weight) in crustacea (lobsters); background values

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Acenaphthene	ND	ND				
Benz[*a*]anthracene	655	179	9–38	Trace–133	6–79	6–17
Benzo[*a*]pyrene	18	3.8	0.4–2.1	Trace–2	1.6–8	ND–1.6
Benzo[*b*]fluoranthene	17	28	3–6.5	Trace–5.3	7–16	ND–0.8
Benzo[*e*]pyrene	ND	170	12–23	ND–22	15–29	ND–3.6
Benzo[*ghi*]perylene	11	63	1.4–6.8	Trace–2.0	2.4–10	ND–0.8
Benzo[*k*]fluoranthene	2	4.4	0.8–1.9	Trace–1.6	1.9–8	ND–0.8
Chrysene	140	113	2.5–12	ND–14	2–43	ND
Fluoranthene	ND	147	46–407	5.5–12	90–162	ND–34
Fluorene	ND	194				
Indeno[1,2,3-*cd*]pyrene	22	77	2.1–5.0	ND–3.7	Trace–5	ND–0.8
Phenanthrene	ND	1197	20–345	ND–15		
Pyrene	ND	174	ND–197	ND–5	35–46	ND–22
Triphenylene	ND	1373	ND–141	ND–Trace		

ND, not detected; **[1]** *Homarus americanus* (digestive gland); Port Hood, Canada, 1979 (Sirota & Uthe, 1981); **[2]** *Homarus americanus* (digestive gland); Brown Bank (offshore), Canada, 1979 (Sirota & Uthe, 1981); **[3]** *Homarus americanus* (hepatopancreas); Morien Bay and Mira Bay, Canada (Sirota et al.,1983); **[4]** *Homarus americanus* (tail muscle); Morien Bayand, Mira Bay, Canada (Sirota et al., 1983); **[5]** *Homarus americanus* (digestive gland); Port Morien, Canada, 1982–84 (Uthe & Musial, 1986); **[6]** *Homarus americanus* (tail muscle); Port Morien, Canada, 1982–84 (Uthe & Musial, 1986)

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and 1.3–15 µg/kg phenanthrene. Mussels (*Mytilus* sp.) were shown to accumulate PAH and were thus a better marker for PAH contamination (Jacob & Grimmer, 1994, 1995).

The concentrations of individual PAH in English sole (*Paraphrys vetulus*) taken from near petroleum storage tanks were 1–5 mg/kg (Malins et al., 1985).

5.1.7.2 *Terrestrial organisms*

The liver of wild deer mice (*Peromyscus maniculatus*) trapped at a PAH-contaminated site in South Carolina, USA (Whidbey Island Naval Air Station) had levels of PAH ranging from 0.075 for benzo[*b*]fluoranthene to 4.6 mg/kg for benz[*a*]anthracene. Acenaphthylene, acenaphthene, fluorene, benz[*a*]-anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, dibenz[*a,h*]anthracene, and indeno[1,2,3-*cd*]pyrene were detected. Liver from mice at an uncontaminated reference site contained measurable amounts of

Table 64. Polycyclic aromatic hydrocarbon concentrations (µg/kg wet weight) in fish and other aquatic species (background values)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Acenaphthene		ND–83		11		7		1–500		
Acenaphthylene						43		0.8–24		
Anthracene				10			2.0–2.2	ND		20
Benz[*a*]anthracene						4	4.0–26	1.2	1.6–7.5	20
Benzo[*a*]fluorene										ND
Benzo[*a*]pyrene					0.04–0.84	1	1.9–15	8	Trace–4.5	5
Benzo[*b*]fluoranthene							3.2–17			
Benzo[*e*]pyrene								ND		
Benzo[*ghi*]perylene							2.0–14	16		
Benzo[*k*]fluoranthene							2.1–11			
Chrysene			6			3	3.4–26	NR		
Dibenz[*a,h*]anthracene							1.2–4.13			
Fluoranthene	4–95		4	85		9	ND–732			20
Fluorene				8.9			ND–15	1–370		ND
Indeno[1,2,3-*cd*]pyrene							ND–15	NR		
1-Methylphenanthrene								NR		
Naphthalene	45–215	ND–117								
Perylene								NR		

Table 64 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]
Phenanthrene	8–142		2	157	2.3–35	36	23–43	ND		40
Pyrene	2–62		4	30		31	2.4–74	1.3–9.6		ND
Triphenylene										20

ND, not detected; NR, not reported; **[1]** Various seafish (muscle, liver, gall), Finnish archipelago, Finland, 1979 (Rainio et al., 1986); **[2]** Edible tissues of various seafish, Arabian Gulf, Iraq (DouAbdul et al., 1987); **[3]** Whole bullhead catfish (*Ictalurus nebulosus*), Buckeye Lake, USA (Vassilaros et al., 1982); **[4]** Whole bullhead catfish (*Ictalurus nebulosus*; whole), Black River, USA (West et al., 1985); **[5]** Whole fish, Hersey River, USA (Black et al., 1981); **[6]** Whole striped bass (*Morone saxatillis*); Potomac River, USA (Vassilaros et al., 1982); **[7]** White suckers (*Catastomus commersoni*); stomach contents; Lake Erie, USA (Maccubbin et al., 1985); **[8]** Various fish, Japan, 1974–91 (Environment Agency, Japan, 1993); **[9]** Fish bought in market, Ibadan, Nigeria; reference weight not given (Emerole et al., 1982); **[10]** Whole holothurians, France; concentration in µg/kg dry weight (Milano et al., 1986)

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only benz[*a*]anthracene (0.55 mg/kg) and acenaphthylene (2.2 mg/kg) (Dickerson et al., 1994).

In a study of PAH levels in terrestrial organisms from a roadside in Brisbane, Australia, 16 PAH were determined: naphthalene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, perylene, indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, benzo[*ghi*]perylene, and coronene. In the beetle *Laxta granicollis*, pyrene and benzo[*ghi*]perylene were present at the highest levels, at 20 µg/kg wet weight each; phenanthrene and fluoranthene were present at about 10 µg/kg; and the concentrations of other PAH were < 5 µg/kg. Naphthalene, anthracene, dibenz[*a,h*]anthracene, and coronene were not detected. Fluorene, at a concentration of 11 µg/kg wet weight, was the most abundant PAH in the beetle *Platyzosteria nitida*; the concentrations of other PAH were < 5 µg/kg; whereas naphthalene, dibenz[*a,h*]anthracene, and coronene were not detected. In millipedes (myriapods), benzo[*k*]fluoranthene was the most abundant PAH (19 µg/kg wet weight); the pyrene concentration was 12 µg/kg; those of other PAH were < 5 µg/kg wet weight; and dibenz[*a,h*]anthracene and coronene were not detected. In centipedes (*Myriaod* sp.), no PAH were detected. In slugs (*Arion ater*), benzo[*k*]fluoranthene showed the highest concentration, at 19 µg/kg wet weight; the pyrene and naphthalene levels were about 10 µg/kg; those of other PAH were < 5 µg/kg wet weight; and anthracene, perylene, dibenz[*a,h*]anthracene, and coronene were not detected. In earthworms (*Lumbricus terrestris*), benzo[*ghi*]perylene was the most abundant PAH (28 µg/kg wet weight); phenanthrene, fluoranthene, pyrene, chrysene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene were present at about 10 µg/kg; and naphthalene and dibenz[*a,h*]anthracene were not detected (Pathirana et al., 1994).

The PAH concentrations in earthworms did not seem to be affected by the location in which the worms lived, but those in the faeces showed a significant dependence on location. In a survey of earthworm faeces from the Bornhöveder Lake district in 1988, the concentrations of phenanthrene, fluoranthene, pyrene, and benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene were in the range of 45 µg/kg; those of benz[*a*]anthracene, chrysene plus triphenylene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, and benzo[*ghi*]perylene were about 20 µg/kg; and those of anthracene, benzo[*ghi*]fluoranthene plus benzo[*c*]phenanthrene, dibenz[*a,h*]anthracene, anthanthrene, and coronene were < 5 µg/kg. Earthworm faeces in the Saarland contained 250–770 µg/kg benzo[*a*]pyrene, and *Allolobophora longa* earthworm faeces from a highly industrialized region of eastern Germany (Halle, Leipzig) contained even higher concentrations: 37–2100 µg/kg benzo[*a*]pyrene and 36–1700 µg/kg benzo[*e*]pyrene. The faeces of the earthworm *Lumbricus terrestris* contained 4.6–55 µg/kg benzo[*a*]pyrene and 6.5–50 µg/kg benzo[*e*]pyrene (Jacob & Grimmer, 1995).

In insects near the Hersey River, USA, the maximum concentrations of PAH were 5500 µg/kg phenanthrene, 2900 µg/kg benz[*a*]anthracene, and 730 µg/kg benzo[*a*]pyrene (Black et al., 1981).

The lipid fraction of liver from herring gulls (*Larus argentatus*) from Pigeon Island and Kingston, Ontario, Canada, contained 0.15 µg/kg anthracene, 0.082 µg/kg fluoranthene, 0.076 µg/kg pyrene, 0.05 µg/kg naphthalene, 0.044 µg/kg fluorene, 0.038 µg/kg acenaphthene, and 0.038 µg/kg benzo[*a*]pyrene (Environment Canada, 1994). The concentrations of PAH in pooled samples taken from the eggs of herring gulls (*Larus argentatus*) on the German North Sea islands Mellum and Trischen during 1992–93 were below the limit of detection, except for that of phenanthrene, which was 1 µg/kg wet weight (Jacob & Grimmer, 1994).

5.2 Exposure of the general population

Possible sources of nonoccupational exposure to PAH are:

- polluted ambient air (main emission sources: vehicle traffic, industrial plants, and residential heating with wood, coal, mineral oil) (see section 5.1.1);
- polluted indoor air (main emission sources: open stoves and environmental tobacco smoke) (see Table 65);
- tobacco smoking (see Table 66);
- contaminated food and drinking-water (see sections 5.1.5 and 5.1.2.3)
- use of products containing PAH (coal-tar skin preparations and coal-tar-containing hair shampoos);
- ingestion of house dust; and
- dermal absorption from contaminated soil and water.

5.2.1 *Indoor air, tobacco smoke, and environmental tobacco smoke*

PAH are found in indoor air (Table 65) mainly as a result of tobacco smoking and residential heating with wood, coal, or, in some developing countries, rural biomass. The PAH levels in indoor air usually range from 1 to 50 ng/m^3. The most abundant PAH were phenanthrene and naphthalene, with levels of up to 2300 ng/m^3. Homes with gas heating systems had higher indoor levels than those with electric heating systems (Chuang et al., 1991), and even higher levels were detected in indoor air near open fireplaces (Alfheim & Ramdahl, 1984). Airtight residential wood-burning stoves seemed to have a minor effect on the indoor air concentration of PAH (Alfheim & Ramdahl, 1984; Traynor et al., 1987), but in homes with non-airtight wood stoves, 2–46 times higher PAH concentrations were found during heating periods than during periods without heating (Daisey et al., 1989).

Emissions from unvented kerosene heaters can significantly affect indoor air quality in mobile homes, with a maximum value for naphthalene of

2300 ng/m³. Four of eight heaters investigated emitted PAH-containing particles at levels that exceeded the USA ambient air standards for airborne particles, with a 50% cutoff at the aerodynamic diameter of 10 µm. When the kerosene heaters were in operation, the concentrations of carcinogenic PAH (with four rings or more) in the mobile homes were increased by 10-fold (Mumford et al., 1991).

Emissions from coal and wood combustion in open fires for cooking purposes in unvented rooms in Xuan Wei County, China, contained extremely high PAH concentrations (see also section 8). The highest concentration (benzo[*a*]pyrene at 15 000 ng/m³) was measured in fumes from smoky coal combustion. Coal combustion in open fires in Xuan Wei homes emitted 15 µg/m³ of carcinogenic PAH, while wood combustion emitted 3.1 µg/m³ (Mumford et al., 1987).

Cooking with rural biomass in open fires also led to high PAH levels in indoor air, as measured in rural Indian households. Benzo[*a*]pyrene was measured at a concentration of about 4 µg/m³ during the cooking period, which occupied about 10% of the household activities over the year. The cooking fuels included *baval*, *neem*, mango, *rayan*, and crop residues (Smith et al., 1983). The total release of PAH into indoor air from this source is unknown but may be of major importance, especially in developing countries. Very low PAH emissions were found when liquid petroleum gas was used as a fuel for cooking (Raiyani et al., 1993b). In contrast, the PAH content of kitchen air in Berlin, in the industrialized part of Germany, was similar to that encountered in ambient air (Seifert et al., 1983).

House dust may be another important source of indoor pollution with PAH. In a study of the homes of four smokers and four nonsmokers in Columbus, Ohio, USA, the sum of the concentrations of naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, retene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, cyclopenta[*cd*]pyrene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, dibenz[*a*,*h*]anthracene, benzo[*ghi*]perylene, and coronene in house dust and in soil from the entryway, the pathway, and the foundation of the houses was 16–580 mg/kg. The concentrations in house dust correlated well with those in the entryway soil samples, and a weaker correlation was found with the pathway soil samples, but the relationships were not statistically significant (Chuang et al., 1995).

A special source of exposure to PAH is wood-heated saunas. The highest concentrations were found in a smoke sauna, the second highest in a preheated sauna where the flues were closed before use, and the lowest concentrations in a sauna heated by continuous burning of wood. Pyrene, fluoranthene, benz[*a*]anthracene, and phenanthrene were present at the highest levels (100–330 µg/m³ air); other PAH were present at < 50 µg/m³. The concentrations decreased from benzo[*e*]pyrene > benzo[*a*]pyrene > benzo[*a*]fluorene > anthracene > benzo[*b*]fluorene > fluorene (Häsänen et al., 1983).

Table 65. Polycyclic aromatic hydrocarbon concentrations (ng/m³) in indoor air; main source, residential heating

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Acenaphthene						NR			589–1649
Acenaphthylene						NR			60–592
Anthracene		5–30	408	5–15	84	NR			9.9–11
Benz[*a*]anthracene		3–9	2–6	3–13	145	NR			0.9–5.5
Benzo[*a*]pyrene	13–370	0.3–12	1–7	3–23	150	< 0.009–1.34	0.34–3.5	2.0–490	8.5–29
Benzo[*b*]fluoranthene						< 0.007–0.68	0.17–3.8	1.4–420	5.6–21
Benzo[*e*]pyrene						< 0.06–1.36			
Benzo[*ghi*]perylene	14–340	0.4–10	1–7	3–30	125	< 0.01–6.20	0.37–3.7	2.8–450	0.4–7.5
Benzo[*k*]fluoranthene	5–150	0.07–7	0.6–3	2–10	63	0.005–0.48	0.07–1.9	0.67–200	0.7–21
Chrysene		2–12	3–6	4–13	115	NR			
Coronene						NR			
Cyclopenta[*cd*]pyrene						NR			
Dibenzo[*a,e*]pyrene						NR			
Dibenz[*a,h*]anthracene						NR			3.3–25
Fluoranthene		16–56	16–24	16–50	208	0.07–1.18			87–268
Fluorene						NR			
Indeno[1,2,3-*cd*]pyrene	20–560	1–16	1–8	3–22	130	< 0.02–3.54	1.1–6.1	3.9–740	2.3–11

Table 65 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Phenanthrene		120–400	120–200	140–290	555	NR			31–64
Pyrene						0.02–1.53			1.0–20

ND, not determined; NR, not reported; /, single measurements; **[1]** Wood-burning open fire-place, Netherlands (Slooff et al., 1989); **[2]** Wood in multi-burner, Netherlands (Slooff et al., 1989); **[3]** Coal, Netherlands (Slooff et al., 1989); **[4]** Briquettes, Netherlands (Slooff et al., 1989); **[5]** 'Icopower' heating, Netherlands (Slooff et al., 1989); **[6]** Wood heating in seven homes, USA (Daisey et al., 1989); **[7]** Wood burning in one home; volume, 236 m^3; airtight stove, Truckee, USA, (elevation, 1800 m) (Traynor et al., 1987); **[8]** Wood burning in one home; volume, 236 m^3; non-airtight stove, Truckee, USA (elevation, 1800 m) (Traynor et al., 1987); **[9]** Wood burning in one home with four different heaters, USA (Knight & Humphreys, 1985)

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Table 65 (contd)

Compound	[10]	[11]	[12]	[13]	[14]	[15]	[16]	[17]
Acenaphthene				NR				1–258
Acenaphthylene	10–120	21/68	25–36	NR				1–753
Anthracene	1.5–15		4.2–5.9	NR				0.1–80
Benz[*a*]anthracene	0.24–3.4	0.72/2.8	0.55–1.0	ND–3.81	25 100	1000	4000	5–1021
Benzo[*a*]pyrene	0.28–3.3	0.24/2.0	0.54–1.0	ND–4.13	14 700	600	3100	8–1645
Benzo[*b*]fluoranthene				NR				2–930
Benzo[*e*]pyrene	0.33–10		1.4–3.0	NR				5–1106
Benzo[*ghi*]perylene	0.32–2.5	0.22/3.7	0.72–1.0	ND–5.4				4–802
Benzo[*k*]fluoranthene				ND–7.81[a]				4–824
Chrysene	0.58–7.2	1.5/3.1	1.4–2.2	0.18–8.61				7–1439
Coronene	0.31–1.4	0.07/2.3	0.55–0.58	ND–4.75				NR
Cyclopenta[*cd*]pyrene	0.18–2.0	0.49/4.2	0.36–0.59	ND–2.38	10 700	400	5600	NR
Dibenzo[*a,e*]pyrene				NR	11700	600	200	NR
Dibenz[*a,h*]anthracene				NR				8–958
Fluoranthene	6.2–23	16/11	11	2.4–37.4				5–1095
Fluorene				NR				3–275
Indeno[1,2,3-*cd*]pyrene	0.24–1.8	0.15/1.3	0.48–0.79	ND–3.53	8400	500	2000	4–670
5-Methylcholanthrene				NR	7300	200	200	NR
Naphthalene	750–2200	2300/950	1200–1600	NR				NR

Table 65 (contd)

Compound	[10]	[11]	[12]	[13]	[14]	[15]	[16]	[17]
Phenanthrene	55–210	48/34	93–110	9.2–210				3–667
Pyrene	3.6–17	9.7/13	6.9–7.6	1.4–18.1				7–850

[10] Gas or electricity, USA (Wilson & Chuang, 1991); **[11]** Kerosene; unvented heaters in mobile homes, Apex, USA (Mumford et al., 1991); **[12]** Various heating in eight homes, Columbus, USA (Chuang et al., 1991); **[13]** Various heating in 33 homes, USA (Wilson et al., 1991); **[14]** Smoky coal, Xuan Wei, China (Mumford et al., 1987); **[15]** Smokeless coal, Xuan Wei, China (Mumford et al., 1987); **[16]** Wood, Xuan Wei, China (Mumford et al., 1987); **[17]** Various cooking fuels (cattle dung, wood, kerosene, liquid petroleum gas) in 60 homes, India (Raiyani et al., 1993b)

[a] Sum of benzofluranthenes

The protocol of a study of total human environmental exposure included direct monitoring of exposure to benzo[*a*]pyrene by inhalation and ingestion during three periods of 14 days. The range and magnitude of dietary exposure (2–500 ng/day) was much greater than that by inhalation (10–50 ng/day). The levels of benzo[*a*]pyrene in indoor air were closely correlated with the ambient levels in most homes (Waldman et al., 1991).

Indoor air concentrations of individual PAH due mainly to cigarette smoke are shown in Table 66, and the levels in mainstream and sidestream smoke of cigarettes are listed in Table 67. The average PAH levels ranged from 1 to 50 ng per cigarette, and the major components were phenanthrene, naphthalene, benzo[*a*]pyrene, benzo[*e*]pyrene, fluoranthene, and pyrene. Sidestream smoke was found to contain 10 times more PAH than mainstream smoke. The levels in sidestream smoke were 42–2400 ng per cigarette (Grimmer et al., 1987). The PAH concentrations in the mainstream smoke from filter cigarettes increased with increasing puff volume (Funcke et al., 1986). In a pilot study in Columbus, Ohio, USA, naphthalene was the most abundant PAH; environmental tobacco smoke appeared to be the most significant source of indoor pollution (Chuang et al., 1991).

In studies in eight healthy male smokers, aged 20–40 years, the benzo[*a*]pyrene intake from the smoking of 20 cigarettes per day was calculated to be 150–750 ng/d, assuming a deposition rate for particulate matter of 75% (Scherer et al., 1990).

The total concentration of 14 PAH (fluoranthene, pyrene, benzo[*a*]fluorene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, perylene, dibenz[*a,h*]-anthracene, benzo[*ghi*]perylene, and anthanthrene) measured in a 36-m^3 room into which sidestream smoke from five German cigarettes was introduced every hour, with one air change per hour, was 429 ng/m^3. Assuming that the daily inhalation volume for adults is 18 m^3 and that 20 h/d are spent indoors, the volume of indoor air inhaled daily is 18 m^3 x 20/24 = 15 m^3. Thus, passive smokers are exposed daily to 15 x 429 = 6435 ng PAH, including 15 x 22 = 330 ng benzo[*a*]pyrene (Ministry of Environment, 1979). An intake of 11 ng benzo[*a*]pyrene was estimated in another study on the basis of an assumed breath volume of 0.5 m^3/h, a deposition rate for particulate matter of 11%, and an exposure time of 8 h, after monitoring in an unventilated, 45-m^3, furnished room (Scherer et al., 1990).

5.2.2 *Food*

Smoked and barbecued food in particular can contain PAH (Grimmer & Düvel, 1970; McGill et al., 1982; de Vos et al., 1990; Menichini et al., 1991b; see also section 5.1.5 and Tables 51–56). Preparation of food with contaminated drinking-water (see section 5.1.2.3) may also lead to exposure to PAH.

Table 66. Polycyclic aromatic hydrocarbon concentrations (ng/m³) in indoor air; main source, environmental tobacco smoke

Compound	[1]	[2]	[3]	[4]	[5]	[6]
Acenaphthene	2.5	36				
Acenaphthylene	14	177				
Anthracene	2.8	25	1.5	< 1		
Anthanthrene	0.5	1.5	< 1	2.5		3
Benz[*a*]anthracene	1.3	12	15	13		
Benzo[*a*]fluorene			5.5			39
Benzo[*a*]pyrene	1.8	7.3	14	4.5	0.04–0.16	22
Benzo[*b*]fluoranthene					0.06–0.08	
Benzo[*b*]fluorene			2.5			
Benzo[*e*]pyrene	2.3	7.1	11	4.5		18
Benzo[*ghi*]fluoranthene	4.3	18	8.5	14		
Benzo[*ghi*]perylene	2.5	5.8	7	2	0.09–0.36	17
Benzo[*k*]fluoranthene					0.02–0.06	
Coronene	2.0	3.1				
Fluoranthene	14	41	5	16		99
Indeno[1,2,3-*cd*]pyrene	2.3	5.8	1	1.5	0.13–0.45	
1-Methylphenanthrene	6.6	38	< 1	3.5		
Perylene	0.5	0.8	4	2.5		11
Phenanthrene	38	168	3	1		
Pyrene	13	32	6.3	21		66

[1] Office room (volume, 88 m³; ventilation, 176 m³/h; background sample after weekend, Finland; vapour and particulate phase (Salomaa et al., 1988); **[2]** Office room (volume, 88 m³; ventilation, 176 m³/h; 6 h; 96 cigarettes, American type, 10 different brands, both medium- and low tar, Finland; vapour and particulate phase (Salomaa et al., 1988); **[3]** House in a forest (room volume, 65 m³; air exchange, 2.0–2.3 turnovers/h); background sample, Norway (Alfheim & Ramdahl, 1984); **[4]** House in a forest (room volume, 65 m³; air exchange, 2.0–2.3 turnovers/h); with tobacco smoking, Norway (Alfheim & Ramdahl, 1984); **[5]** House in a residential, wooded area of Truckee, USA (elevation, 1800 m); volume, 236 m³; no stove (Traynor et al., 1987); **[6]** Model room (volume, 36 m³); one air exchange/h, smoking of five cigarettes/h (Ministry of Environment, 1979))

High-performance liquid chromatography or gas chromatography; concentration of particulate phase, unless otherwise stated

Table 67. Concentrations of selected polycyclic aromatic hydrocarbons in cigarette smoke

Compound	Mainstream smoke (μg/100 cigarettes)	Sidestream smoke (μg/100 cigarettes)
Anthracene	2.3–23.5	
Anthanthrene	0.2–2.2	3.9
Benz[*a*]anthracene	0.4–7.6	
Benzo[*b*]fluoranthene	0.4–2.2	
Benzo[*j*]fluoranthene	0.6–2.1	
Benzo[*k*]fluoranthene	0.6–1.2	
Benzo[*ghi*]fluoranthene	0.1–0.4	
Benzo[*a*]fluorene	4.1–18.4	75.0
Benzo[*b*]fluorene	2.0	
Benzo[*ghi*]perylene	0.3–3.9	9.8
Benzo[*c*]phenanthrene	Present	
Benzo[*a*]pyrene	0.5–7.8	2.5–19.9
Benzo[*e*]pyrene	0.2–2.5	13.5
Chrysene	0.6–9.6	
Coronene	0.1	
Dibenz[*a,h*]anthracene	0.4	
Dibenzo[*a,e*]pyrene	Present	
Dibenzo[*a,h*]pyrene	Present	
Dibenzo[*a,i*]pyrene	0.17–0.32	
Dibenzo[*a,l*]pyrene	Present	
Fluoranthene	1.0–27.2	126.0
Fluorene	Present	
Indeno[1,2,3-*cd*]pyrene	0.4–2.0	
5-Methylcholanthrene	0.06	
Perylene	0.3–0.5	3.9
Phenanthrene	8.5-62.4	
Pyrene	5.0–27	39.0–101.0
Triphenylene	Present	
1-Methylphenanthrene	3.2	

Adapted from International Agency for Research on Cancer (1985)

In 1989 and 1990, the levels of naphthalene and alkylated derivatives, acenaphthene, acenaphthylene, fluorene, phenanthrene, anthracene, fluoranthene, 1-methylphenanthrene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, perylene, indeno[1,2,3-*cd*]pyrene, dibenz[*a,h*]anthracene, and benzo[*ghi*]-perylene were measured in salmon, herring, cod, rockfish, and halibut in the area of the Gulf of Alaska where oil spilled from the tanker *Exxon Valdez*. As

only the sums of the concentrations were considered, there was no apparent difference from those in fish samples taken from unpolluted control sites in 1989. In 1990, slightly elevated PAH concentrations were found at the polluted sampling site. Nevertheless, the fish from the area were considered to be safe for human consumption by these investigators (Saxton et al., 1993).

In another special exposure situation, the average daily PAH intake of the inhabitants of Kuwait due to consumption of seafood after the war in the Persian Gulf was calculated to be 0.23 µg/day on the basis of the concentrations monitored in local fish and shrimps (Saed et al., 1995).

5.2.3 *Other sources*

Benzo[*a*]pyrene was detected in coal-tar-containing hair shampoos at levels of 7000–61 000 µg/kg, and a tar bath lotion contained 150 000 µg/kg benzo[*a*]pyrene. No PAH were detected in hair shampoos made from wood tar (State Chemical Analysis Institute Freiburg, 1995). PAH are absorbed from coal-tar shampoos through the skin during hair washing. Exposure during one washing with this type of shampoo, which contains benzo[*a*]pyrene at 56 mg/kg, for anti-dandruff therapy results in absorption of 0.45 µg/kg body weight, assuming 20 g coal-tar, 70 kg body weight, and 3% dermal absorption (van Schooten et al., 1994; see also section 8).

5.2.4 *Intake of PAH by inhalation*

Estimates of PAH intake from air are summarized in Table 68.

In an assessment of the risk for cancer due to air pollution in Germany, the average volume of air inhaled during heavy work was assumed to be 140 m^3 per person per week. The maximum intake of airborne benzo[*a*]pyrene per week was thus estimated to be 0.21 µg/week in rural areas, 0.84 µg/week in industrial areas, and 7 µg/week near emission sources (State Committee for Air Pollution Control, 1992).

On the basis of an average inhalation of 15 m^3 air per day, exposure to benzo[*a*]pyrene was calculated to be 0.05 µg/d. In industrial areas, the exposure was calculated to be four times higher (0.19 µg/d) (Raiyani et al., 1993a).

5.2.5 *Intake of PAH from food and drinking-water*

Estimates of PAH intake from food are shown in Table 69. The values for benzo[*a*]pyrene range from 0.14–1.6 µg/d.

The total dietary intake of some PAH in the United Kingdom was estimated to be (µg/person per day): 1.1 for pyrene, 0.99 for fluoranthene, 0.50 for chrysene, 0.25 for benzo[*a*]pyrene, 0.22 for benz[*a*]anthracene, 0.21 for benzo[*ghi*]perylene, 0.18 for benzo[*b*]fluoranthene, 0.17 for benzo[*e*]pyrene,

0.06 for benzo[*k*]fluoranthene, and 0.03 for dibenz[*a*,*h*]anthracene. The major contributors of PAH to the total dietary intake appeared to be oils and fats, with 28% from butter, 20% from cheese, and 17% from margarine, in respective dietary survey groups; cereals provided 56% from white bread and 12% from flour. The oils and fats had the highest individual PAH levels. Although cereals did not contain high levels of individual PAH, they were the main contributor by weight to the total in the diet. Fruits and vegetables contributed most of the rest of the PAH in the diet, while milk and beverages were of minor importance. Smoked meat and smoked fish made very small contributions to the food groups to which they belonged, which themselves were not major components of the diet (Dennis et al., 1983).

In Sweden, the annual intake per person of the sum of fluoranthene, pyrene, benz[*a*]anthracene, chrysene, triphenylene, benzo[*b*]fluoranthene, benzo[*j*]-fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, and indeno[1,2,3-*cd*]pyrene was about 1 mg. Cereals again seemed to be the main contributor (about 34%), followed by vegetables (about 18%) and oils and fats (about 16%). Although smoked fish and meat products had the highest PAH levels, they made a modest contribution since they are minor components of the usual Swedish diet (Larsson, 1986).

5.3 Occupational exposure

PAH have been measured in the air at various workplaces. Studies in which measurements were reported only as the benzene-soluble fraction or some other summarizing parameter affected mainly by PAH are not covered because they do not refer to individual substances. The presence of PAH metabolites in biological samples (urine, blood) from workers has been used as a biomarker, and 1-hydroxypyrene seems to be a suitable marker in some workplaces (see section 8.2.3). No data were available on occupational exposure during production and use.

Occupational exposure to PAH occurs by both inhalation and dermal absorption. In coke-oven workers, 75% of their exposure to total pyrene and 51% of that to benzo[*a*]pyrene occurs by cutaneous transfer (Van Rooij et al., 1993a; see also section 6). The exposure of workers due to deposition of airborne pyrene on the skin, detected in wipe samples, can be summarized as follows: in refineries, < 0.0045 μg/cm² (detection limit), 26 samples below detection limit; in hot-mix asphalt facilities, < 0.0045 μg/cm², 25 samples below detection limit; during paving, < 0.13-0.31 μg/cm² found in two of nine samples (assuming a body area of 1.8 m², equivalent to 5600 μg/person per day); in asphalt roofing manufacture, < 0.0045–0.0091 μg/cm² found in 1 of 29 samples (assuming a body area of 1.8 m², equivalent to 170 μg/person per day); in application of asphalt roofing, < 0.0045 μg/cm², 10 samples below detection limit; in a wood preserving plant, 47–1500 μg pyrene per person per

Table 68 Estimated intake of polycyclic aromatic hydrocarbons (μg/day per person) from ambient air

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Anthracene			0.005				0.001		
Anthanthrene			0.015						
Benz[*a*]anthracene			0.030				0.013		
Benzo[*a*]pyrene	0.01–0.03[a] 0.02–0.12[b] 0.06–1.0[c]	0.0025–0.025	0.025	0.034[a]	0.0095–0.0435	0.004[a]	0.017	0.03–0.05	0.0005–0.20
Benzo[*b*]fluoranthene			0.060				0.029		
Benzo[*b*]fluorene			0.002				0.002		
Benzo[*e*]pyrene			0.035				0.022		
Benzo[*ghi*]perylene			0.030				0.027		
Benzo[*j*]fluoranthene			0.010						
Benzo[*k*]fluoranthene			0.015				0.015		
Chrysene			0.035						
Coronene			0.025						
Dibenz[*a,h*]anthracene			0.020				0.004		
Fluoranthene			0.040				0.016		
Fluorene							0.0005		
Indeno[1,2,3-*cd*]pyrene			0.030				0.024		
Perylene			0.015				0.003		

Table 68 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]
Phenanethrene			0.200				0.007		
Pyrene			0.040				0.017		
Triphenylene			0.020						

[1] Germany (maximum concentrations) (State Committee for Air Pollution Control, 1992); **[2]** Italy (Menichini, 1992a); **[3]** Netherlands (maximum concentrations) (Guicherit & Schulting, 1985); **[4]** United Kingdom (maximum concentrations) (Butler & Crossley, 1979); **[5]** USA (Santodonato et al., 1980); **[6]** USA (WHO, 1987); **[7]** Japan (maximum concentrations) (Matsumoto & Kashimoto, 1985); **[8]** China (Chen et al., 1980); **[9]** India (Chakraborti et al., 1988)

[a] Rural areas
[b] Industrial areas
[c] Near emission source

Table 69. Estimated intake of polycyclic aromatic hydrocarbons (µg/day per person, maximum values) from food

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
Anthracene	5.6							
Anthanthrene	0.30							
Benz[*a*]anthracene	0.14							
Benzo[*a*]pyrene	0.36	0.14–1[a]	0.1–0.3[b] 0.2[c]	0.12–0.42	0.5	0.5	0.48	0.16–1.6
Benzo[*b*]fluoranthene	1.0							
Benzo[*ghi*]perylene	7.6				0.3	0.9		
Benzo[*j*]fluoranthene	0.90							
Benzo[*k*]fluoranthene	0.30							
Chrysene	0.90					5.0		
Coronene	0.09							
Dibenz[*a,h*]anthracene	0.10							
Fluoranthene	4.3				3	10		
Indeno[1,2,3-*cd*]pyrene	0.31				0.4	<0.3		
Perylene	0.20							
Phenanethrene	2.0							
Pyrene	4.0					5.1		

[1] Austria (Pfannhauser, 1991); **[2]** Germany (State Committee for Pollution Control, 1992); **[3]** Italy (Menichini, 1992a); **[4]** Netherlands (de Vos et al., 1990); **[5]** Market basket study, Netherlands (Vaessen et al., 1984); **[6]** Duplicate diet study, Netherlands (Vaessen et al., 1984); **[7]** United Kingdom (Dennis et al., 1983); **[8]** USA (Santodonato et al., 1980)

[a] Concentration in µg/week
[b] Adult non-smoker (70 kg)
[c] Mean concentration

day. These data indicate that skin penetration is an important factor in estimating total body exposure to PAH.

5.3.1 *Occupational exposure during processing and use of of coal and petroleum products*

The following section is based on data obtained up to the early 1980s which were compiled by the IARC (1984b, 1985, 1989b). More recent studies are presented in detail.

5.3.1.1 *Coal coking*

In studies of pollution of the atmosphere near coke-oven batteries, the concentration of benzo[*a*]pyrene varied from < 0.1 in administrative buildings and a pump house to 100–200 μg/m³ on the machinery and discharge side of a battery roof. At the top of a coke battery, the following concentrations of particulate and gaseous PAH were measured by stationary sampling: naphthalene, 0–4.4 (particulate)/ 280–1200 (gaseous) μg/m³; acenaphthene, 0–17/6.0–100 μg/m³; fluorene, 0–58/23–130 μg/m³; phenanthrene, 27–890/ 6.7–280 μg/m³; anthracene, 9.6–310/6.0–91 μg/m³; 1-methylphenanthrene, 2.7–21/0–7.0 μg/m³; fluoranthene, 45–430/0–24 μg/m³; pyrene, 35–320/ 0–14 μg/m³; benzo[*a*]fluorene, 9.7–90/0–6.8 μg/m³; benzo[*b*]fluorene, 3.1–61/0–0.3 μg/m³; benzo[*c*]phenanthrene, 2.6–49 μg/m³ (particulate); benz[*a*]anthracene, 5.4–160/< 0.4–1.6 μg/m³; benzo[*b*]fluoranthene, 5.5–67/ 0–0.7 μg/m³; benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, 0–35/0–0.7 μg/ m³; benzo[*e*]pyrene, 8–73/0–0.2 μg/m³; benzo[*a*]pyrene, 14–130/0–1.5 μg/m³; perylene, 3.3–19/0–0.1 μg/m³; benzo[*ghi*]perylene, 8.7–45 μg/m³ (particulate); anthanthrene, 2.6–62 μg/m³ (particulate); and coronene, 1.0–19 μg/m³ (particulate) (IARC, 1984b).

At eight sites in a German coke plant in 1981, including the top of the oven and the cabin of a lorry driver, the following PAH concentrations were measured: 2.7 μg/m³ fluoranthene, 1.9–170 μg/m³ pyrene, 0.38–37 μg/m³ benzo[*c*]phenanthrene, 0.22–21 μg/m³ cyclopenta[*cd*]pyrene, 1.2–120 μg/m³ benz[*a*]anthracene, 0.71–79 μg/m³ benzo[*c*]pyrene, 0.88–89 μg/m³ benzo[*a*]pyrene, 0.21–14 μg/m³ perylene, 0.37–27 μg/m³ benzo[*ghi*]perylene, 0.18–17 μg/m³ anthanthrene, and 0.93–6.5 μg/m³ coronene. The authors pointed out that the concentrations may have been much higher previously (Manz et al., 1983).

Measurements with personal air samplers in Germany and Sweden showed benzo[*a*]pyrene concentrations varying from 0.16–33 μg/m³ for coke-oven operators to 4.7–17 μg/m³ for lorry drivers. The ranges of exposure to all PAH at different workplaces in the 1970s were: lorry driver, 170–1000 μg/m³; coke-car operator, 4.8–73 μg/m³; jamb cleaner, 62–240 μg/m³; door cleaner,

9.1–17 μg/m³; push-car operator, 9.4–62 μg/m³; sweeper, 110 μg/m³; quench-car operator, 5.7 μg/m³; and wharf man, 360 μg/m³ (IARC, 1984b).

Personal air samples taken from 56 Dutch coke-oven workers in 1986 showed pyrene levels of < 0.6 μg/m³ (detection limit) to 9.8 μg/m³ (Jongeneelen et al., 1990). The results of more recent measurements in personal air samples are shown in Table 70.

5.3.1.2 *Coal gasification and coal liquefaction*

The levels of individual PAH in area air samples in Norwegian and British coal gasification plants between the late 1940s and the mid 1950s were in the low microgram per cubic millilitre range. In modern gasification systems, the concentrations of total PAH are usually ≤ 1 μg/m³, but in one of three plants examined the total aerial PAH load was about 30 μg/m³. Personal samples taken in modern coal gasification plants showed similar PAH concentrations (IARC, 1984b).

In a pilot coal liquefaction plant in the United Kingdom, monitoring of five operators for vapour-phase PAH gave following results: 1900–3300 ng/m³ phenanthrene, 340–670 ng/m³ pyrene, 270–380 ng/m³ fluoranthene, 29–130 ng/m³ anthracene, 22–1700 ng/m³ fluorene, < 1–1800 ng/m³ naphthalene, < 1–1000 ng/m³ acenaphthene, and < 1–8 ng/m³ acenaphthylene. The higher-molecular-mass PAH were not detected (limit of detection, 1 ng/m³). Pyrene was detected in the particulate phase at concentrations of 630–2900 ng/m³ (Quinlan et al., 1995a).

5.3.1.3 *Petroleum refining*

Personal samples from operators of catalytic cracker units and reaction and fractionation towers in a petroleum refinery showed total PAH levels of 2.6–470 μg/m³. During performance and turn-round operations on reaction and fractionation towers, naphthalene and its methyl derivatives accounted for more than 99% of the total PAH measured; exposure to anthracene, pyrene, chrysene, and benzo[*a*]pyrene was ≤ 1 μg/m³. Area monitoring for these PAH during normal activities and during shut-down, leak-testing, and start-up operations after turn-rounds gave total PAH concentrations up to 400 μg/m³, most of the measurements being < 100 μg/m³ (IARC, 1989b).

The results of personal air sampling of workers at six jobs in seven American refineries in 1990–91 were as follows (mean and range): 5.5 (< 0.25–10) μg/m³ naphthalene, 3.3 (< 0.44–24) μg/m³ acenaphthene, 3.3 (< 0.19–26) μg/m³ acenaphthylene, 0.98 (< 0.085–7.9) μg/m³ fluoranthene, 0.82 (< 0.055–6.7) μg/m³ phenanthrene, 0.78 (< 0.13–5.3) μg/m³ benzo[*e*]pyrene, 0.65 (< 0.055–5.2) μg/m³ benzo[*b*]fluoranthene, 0.47 (< 0.14–2.7) μg/m³ fluorene, 0.29 (< 0.11–1.4) μg/m³ indeno[1,2,3-*cd*]pyrene,

Table 70. Workplace exposures to polycyclic aromatic hydrocarbons in the atmosphere of coke-oven batteries (μg/m³), determined from personal air samples

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
Acenaphthene						3.8	
Acenaphthylene						28	
Anthracene				55		16	
Anthanthrene						2.4	
Benz[*a*]anthracene		0.11–33.19		96		7.5	
Benzo[*a*]fluorene				70		3.7	
Benzo[*a*]pyrene	< 0.01–31.15[a] 0.01–22.91[b]	0.03–12.63	0.9–46.02	38	0.1–29	7.3	1300
Benzo[*b*]fluoranthene				42			1500
Benzo[*b*]fluorene					4.3		
Benzo[*c*]phenanthrene					1.4		
Benzo[*e*]pyrene						4.7	
Benzo[*ghi*]fluoranthene						1.6	
Benzo[*ghi*]perylene						4.4	
Benzo[*k*]fluoranthene				42			
Chrysene		0.08–13.17		72			
Coronene						3.2	
Cyclopenta[*cd*]pyrene						1.9	
Fluoranthene	0.12–17.00[a]			144		22	4400
Fluorene				109		14	
Indeno[1,2,3-*cd*]pyrene						4.5	

Table 70 (contd)

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
1-Methylphenanthrene						3.4	
Naphthalene	28–445[a]			650			
Perylene						1.8	
Phenanthrene	0.07–8.53[a]			195		49	
Pyrene			2.36–98.63			17	Trace

[1] Finland; samples from one plant, 1988–90 (Yrjänheikki et al., 1995); **[2]** Italy; samples from 69 workers, six workplaces (Assennato et al., 1993a); **[3]** Italy; samples from three workplaces at battery top (Cenni et al., 1993); **[4]** Sweden; one typical sample (Andersson et al., 1983); **[5]** United Kingdom; samples from 12 plants (Davies et al., 1986); **[6]** USA; samples from topside coke-oven workers (Haugen et al., 1986); **[7]** India; samples from top of coke oven (Rao et al., 1987)

[a] Area air samples
[b] Personal air samples

0.18 (< 0.085–0.69) μg/m³ benz[*a*]anthracene, 0.16 (< 0.11–< 0.59) μg/m³ benzo[*a*]pyrene, 0.063 (< 0.028–0.26) μg/m³ anthracene, < 0.11–< 0.2 μg/m³ pyrene, < 0.085–< 0.15 μg/m³ chrysene, < 0.085–< 0.15 μg/m³ benzo[*k*]fluoranthene, < 0.11–< 0.2 μg/m³ benzo[*ghi*]perylene, and < 0.11–< 0.2 μg/m³ dibenz[*a*,*h*]anthracene. Dermal wipe samples from the back of the hand or from the forehead of workers showed PAH levels of < 0.0011–0.29 μg/cm², with the highest level for naphthalene and the lowest for anthracene (Radian Corp., 1991).

5.3.1.4 *Road paving*

In early studies on road paving operations, the total PAH concentrations reported in personal air samples were 4–190 μg/m³, and the mean in area air samples was 0.13 μg/m³. The benzo[*a*]pyrene concentration in stationary samples was < 0.05–0.19 μg/m³ (IARC, 1985).

The concentrations of individual PAH in fume condensates from paving asphalt were generally < 2 mg/kg condensate, varying by about seven times depending on the source of crude oil. The levels of benzo[*a*]pyrene, for example, were between 0.09 and 2.0 mg/kg (Machado et al., 1993).

Fourteen stationary air samples from a road paving site in New Zealand in 1983 contained: 0.14–52 μg/m³ benz[*a*]anthracene plus chrysene, 0.2–14 μg/m³ benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, 0.15–9.0 μg/m³ benzo[*a*]pyrene, 0.31–5.4 μg/m³ benzo[*e*]pyrene, 0.039–2.2 μg/m³ perylene, 0.24–5.4 μg/m³ benzo[*ghi*]perylene, and 0.03–6.3 μg/m³ indeno[1,2,3-*cd*]pyrene plus dibenz[*a*,*h*]anthracene (Swallow & van Noort, 1985). The concentrations in 17 stationary air samples from a road paving operation in New Zealand in another study (year not given) were: 1.2–18 μg/m³ benz[*a*]anthracene plus chrysene, 1.1–11 μg/m³ benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene plus benzo[*k*]fluoranthene, 0.9–9.0 μg/m³ benzo[*a*]pyrene, 0.7–5.4 μg/m³ benzo[*e*]pyrene, and 0.7–6.3 μg/m³ indeno[1,2,3-*cd*]pyrene (Darby et al., 1986). Concentrations of up to 1.3 μg/m³ were found for acenaphthene, ≤ 0.13 μg/m³ for anthracene, and ≤ 0.54 μg/m³ pyrene in road-paving operations. The workers, and especially the machine driver, were exposed to a mixture of bitumen fumes and diesel exhaust gases for 4–6 h per day (Monarca et al., 1987).

The PAH concentrations in personal air samples obtained from workers at six jobs in six paving operations in the USA in 1990 were (mean and range): 6.5 (1.3–15) μg/m³ naphthalene, 2 (< 0.54–6.9) μg/m³ acenaphthene, 2 (< 0.24–8.1) μg/m³ acenaphthylene, 0.58 (< 0.19–0.98) μg/m³ fluorene, 0.55 (< 0.085–1.3) μg/m³ phenanthrene, 0.26 (< 0.11–0.37) μg/m³ fluoranthene, 0.17 (< 0.13–< 0.31) μg/m³ pyrene, 0.16 (< 0.13–0.27) μg/m³ benzo[*e*]pyrene, 0.13 (< 0.099–< 0.2) μg/m³ chrysene, 0.052 (< 0.034–0.11) μg/m³ anthracene, < 0.099–< 0.12 μg/m³ benz[*a*]anthracene, < 0.064–< 0.085 μg/m³

benzo[*b*]fluoranthene, < 0.099–< 0.12 μg/m³ benzo[*k*]fluoranthene, <0.13–<0.25 μg/m³ benzo[*a*]pyrene, <0.13–<0.16 μg/m³ benzo[*ghi*]perylene, < 0.13–< 0.16 μg/m³ indeno[1,2,3-*cd*]pyrene, and < 0.13–< 0.16 μg/m³ dibenz[*a*,*h*]anthracene. Dermal wipe samples from the back of the hand and from the forehead of workers contained PAH at < 0.00004–0.43 μg/cm², with the highest level for naphthalene and the lowest for anthracene and pyrene (Radian Corp., 1991).

Measurements in the air in France during road paving with different bitumens and tars showed the highest benzo[*a*]pyrene concentrations with hard-coal tar (1–6 μg/m³) and the lowest with petroleum-based bitumen (0.004–0.007 μg/m³). In general, the benzo[*a*]pyrene levels in the workplace atmosphere were two to three orders of magnitude higher during paving operations with tar products than with bitumen products (Barat, 1991).

5.3.1.5 *Roofing*

The concentrations of PAH measured during roofing and roofing manufacture are shown in Table 71.

The concentrations of individual PAH in fume condensates from roofing asphalt generated at 232 and 316 °C the were usually < 10 mg/kg condensate, with higher levels only for naphthalene. They varied with the source of crude oil: those for benzo[*a*]pyrene were between 0.6 and 2.8 mg/kg (Machado et al., 1993).

Acenaphthene was detected at concentrations of 1.4–2.1 μg/m³ in personal samples from roofing workers at two US roofing sites in 1985 (Zey & Stephenson, 1986); 0.8–22 μg/m³ phenanthrene were measured at one US roofing site in 1981 (Reed, 1983). Pyrene was measured at ≤ 190 μg/m³ at three roofing sites in Canada (year not given) (Malaiyandi et al., 1986). Personal air samples from 12 roofers at one US roofing site contained benzo[*a*]pyrene at 0.53–2.0 μg/m³ in 1987 (Herbert et al., 1990a). The workplace concentrations during bitumen and coal-tar pitch roofing, waterproofing, and flooring operations were of the same order of magnitude (IARC, 1985).

Significant, 10-fold differences were found in the levels of anthracene, fluoranthene, pyrene, benz[*a*]anthracene, benzo[*b*]fluoranthene, benzo[*k*]-fluoranthene, benzo[*a*]pyrene, and benzo[*ghi*]perylene on skin wipes from the forehead taken before and after a shift in 10 US roofers in 1987 (Wolff et al., 1989a). Comparable results for benzo[*a*]pyrene levels were obtained for 12 roofers at another US roofing site (Herbert et al., 1990a,b).

Dermal wipe samples from the back of the hand or the forehead of workers at six asphalt roofing manufacturing sites in the USA showed PAH levels of < 0.12–5.5 μg/cm², with the highest level for acenaphthylene and the lowest for fluoranthene, benz[*a*]anthracene, benzo[*k*]fluoranthene, and chrysene. Similar samples from workers at six asphalt roofing sites in the USA in 1990–91

Table 71. Exposure to polycyclic aromatic hydrocarbons (μg/m³) during roofing and roofing manufacture

Compound	[1]	[2]	[3]	[4]
Acenaphthene			< 0.52–3.2 (0.87)	< 0.6–6.7 (1.5)
Acenaphthylene			< 0.23–29 (7.1)	< 0.26–12 (2.9)
Anthracene		0.5 / 1.5	< 0.033–0.069 (0.043)	< 0.037–0.042
Anthanthrene	< 0.030			
Benz[*a*]anthracene	< 0.03–0.130	1.3 / 2.5	< 0.099–< 0.13	< 0.11–< 0.13
Benzo[*a*]fluorene	0.03–0.080			
Benzo[*a*]pyrene	< 0.03–0.037	0.9 / 1.5	< 0.13–< 0.18	< 0.11–< 0.13
Benzo[*b*]fluoranthene	< 0.03–0.093[a]	0.8 / 1.2	< 0.065–< 0.38 (0.13)	< 0.078–< 0.085
Benzo[*b*]fluorene	0.051–0.093			
Benzo[*e*]pyrene	< 0.03–0.110		< 0.13–3 (0.61)	< 0.15–< 0.17
Benzo[*ghi*]fluoranthene	< 0.03			
Benzo[*ghi*]perylene	< 0.03–0.069	0.6 / 0.9	< 0.13–< 0.18	< 0.15–< 0.17
Benzo[*k*]fluoranthene		0.4 / 0.7	< 0.099–< 0.13	< 0.099–< 0.12
Chrysene	0.038–0.214		< 0.099–< 0.13	< 0.11–< 0.13
Coronene	< 0.03			
Dibenz[*a,h*]anthracene	< 0.03		< 0.13–< 0.18	< 0.15–< 0.17
Fluoranthene	0.084–0.234	3.1 / 7	< 0.099–4 (0.64)	< 0.11–0.13
Fluorene			< 0.16–14 (2.5)	< 0.19–1.1 (0.44)
Indeno[1,2,3-*cd*]pyrene	< 0.030		< 0.13–< 0.18	< 0.15–0.94 (0.16)
Naphthalene			< 0.22–9.2 (5.2)	1.2–25 (7.5)

Table 71 (contd)

Compound	[1]	[2]	[3]	[4]
Perylene	< 0.030			
Phenanthrene			< 0.065–1.7 (0.53)	< 0.078–1.4 (0.38)
Pyrene	0.035–0.183	2.6 / 5.4	< 0.13–3.4 (0.76)	< 0.15–< 0.73 (0.25)

/, single determinations; mean values shown in parentheses; **[1]** Germany; personal and area air samples from one bitumen roofing site (Schmidt, 1992); **[2]** USA; personal air samples from nine workers; 1987 (Wolff, M.S. et al., 1989); **[3]** USA; personal air samples from six asphalt roofing sites; 1990 (Radian Corp., 1991); **[4]** USA; personal air samples from six roofing manufacturing sites; 1990 (Radian Corp., 1991)

[a] Benzo[*b*+*j*+*k*]fluoranthenes

showed PAH levels of < 0.0011–0.0045 μg/cm², with the highest levels for pyrene, chrysene, and benzo[*a*]pyrene and the lowest for anthracene (Radian Corp., 1991).

5.3.1.6 *Impregnation of wood with creosotes*

Concentrations of PAH ranging from 0.05 μg/m³ benzo[*a*]pyrene to 650 μg/m³ naphthalene were detected during the handling of creosote-impregnated wood for railroad ties in Sweden. Naphthalene, fluorene and phenanthrene were by far the most abundant compounds (> 100 μg/m³) (Andersson et al., 1983). Concentrations of 0.04–0.28 μg/m³ anthracene and 0.11–7.7 μg/m³ pyrene were found at workplaces in Finland where railroad ties were manufactured (Korhonen & Mulari, 1983), and concentrations of 1–19 μg/m³ anthracene, 6.5–61 μg/m³ phenanthrene, and 0.6–13 μg/m³ pyrene were measured in one plant where railroad sleepers were impregnated and in another where poles were preserved (year not given) (Heikkilä et al., 1987). In measurements of personal air samples from 10 workers in a Dutch plant for impregnation of railroad sleepers in 1991, 0.3–1.3 μg pyrene/m³ was measured in the breathing zone and 47–1500 μg/d in pads placed on various areas of the skin of the workers. Dermal exposure was shown to be reduced by up to 90% by the use of protective clothing (Van Rooij et al., 1993b).

5.3.1.7 *Other exposures*

In area air samples taken near the bitumen processing devices of refineries, the total PAH levels varied from 0.004 to 50 μg/m³ (IARC, 1985, 1989b).

The use of lubricating oils may result in exposure to PAH. At two Italian glass manufacturing plants, phenanthrene, anthracene, pyrene, and fluoranthene were found in personal air samples at concentrations ≤ 3 μg/m³ (year not given) (Menichini et al., 1990). The pyrene levels resulting from use of lubricating oils in Italian earthenware factories were 0.02–0.09 μg/m³; the benzo[*a*]pyrene concentration was below the limit of detection (Cenni et al., 1993). Measurable concentrations of individual PAH were detected in indoor air above asphalt floor tiles in e.g. warehouses, factories, and manufacturing plants. The concentrations at six sampling sites in Germany were between < 0.01 ng/m³ for benzo[*ghi*]perylene and 3.3 ng/m³ for chrysene. The concentrations of phenanthrene, pyrene, fluoranthene, chrysene, and benzo[*b*]fluorene in particular were higher than those in outdoor air (Luther et al., 1990).

In two Swiss plants for the production of silicon carbide, personal air samples from four and five workers, respectively, contained the following PAH levels: 4–140 ng/m³ acenaphthylene, 8–86 ng/m³ acenaphthene, 11–500 ng/m³ fluorene, 88–1400 ng/m³ phenanthrene, 3–250 ng/m³ anthracene, 20–1100 ng/m³ fluoranthene, 30–2500 ng/m³ pyrene, 7–6400 ng/m³ benz[*a*]-

anthracene, 37–14 000 ng/m³ chrysene, 11–3700 ng/m³ benzo[*b*]fluoranthene plus benzo[*j*]fluoranthene, 3–470 ng/m³ benzo[*k*]fluoranthene, 18–3800 ng/m³ benzo[*e*]pyrene, 4–630 ng/m³ benzo[*a*]pyrene, 2–250 ng/m³ indeno[1,2,3-*cd*]pyrene, 2–520 ng/m³ dibenz[*a,h*]anthracene, 4–550 ng/m³ benzo[*ghi*]-perylene, and 4–34 ng/m³ coronene (Petry et al., 1994).

5.3.2 *Occupational exposure resulting from incomplete combustion of mineral oil, coal, and their products*

5.3.2.1 *Aluminium production*

Early measurements of atmospheric benzo[*a*]pyrene at workplaces in the aluminium industry showed concentrations of 0.02–970 µg/m³ in personal air samples and 0.03–5.3 µg/m³ in area air samples. In the atmosphere of an aluminium production plant, naphthalene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[*a*]fluorene, benzo[*b*]fluorene, benzo[*c*]phenanthrene, benz[*a*]anthracene, chrysene, triphenylene, benzo[*b*]fluoranthene plus benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, benzo[*ghi*]perylene, anthanthrene, and coronene were found at concentrations ≤ 400 µg/m³. The most abundant compounds were phenanthrene, naphthalene, fluorene, fluoranthene, and pyrene, at concentrations ≥ 100 µg/m³. The other substances occurred at concentrations ≤ 10 µg/m³ (IARC, 1984b).

The following concentrations of PAH were found in four stationary air samples from an aluminium smelter in New Zealand in 1979: 0.37–9.6 µg/m³ benz[*a*]anthracene plus chrysene, 0.34–7.6 µg/m³ benzo[*b*+*j*+*k*]fluoranthenes, 0.12–2.6 µg/m³ benzo[*e*]pyrene, 0.19–4.1 µg/m³ benzo[*a*]pyrene, 0.05–1.5 µg/m³ perylene, 0.13–2.7 µg/m³ indeno[1,2,3-*cd*]pyrene plus dibenz[*a,h*]anthracene, and 0.12–3.3 µg/m³ benzo[*ghi*]perylene (Swallow & van Noort, 1985).

Similar levels were found in a typical personal air sample from a Söderberg aluminium plant in Sweden (year not given) with, in addition, 27 µg/m³ phenanthrene, 20 µg/m³ fluoranthene, 2.8 µg/m³ fluorene, 2.8 µg/m³ anthracene, 2.8 µg/m³ benzo[*a*]fluorene, and < 1.0 µg/m³ naphthalene (Andersson et al., 1983).

In personal air samples from 38 workers in the Söderberg potroom of an aluminium smelter in the humid tropics (location not given), mean concentrations of < 1.0–48 µg/m³ benzo[*a*]pyrene and 3.5–130 µg/m³ pyrene were detected (Ny et al., 1993).

The arithmetic mean concentrations of PAH in workplace air samples from the Canadian aluminium industry were 1100 µg/m³ naphthalene, 130 µg/m³ acenaphthene, 45 µg/m³ fluorene, 30 µg/m³ phenanthrene, 4.5 µg/m³ anthracene, 1.1 µg/m³ fluoranthene, and 0.58 µg/m³ pyrene. The concentrations of benz[*a*]anthracene, chrysene, benzo[*a*]pyrene, and benzo[*e*]pyrene were < 0.01 µg/m³ (Lesage et al., 1987).

Personal air samples from 18 workers in a US plant producing anodes for use in aluminium reduction (year not given) showed pyrene concentrations of 1.2–7.4 μg/m³ (Tolos et al., 1990).

Urine samples from 11 workers in Norwegian Söderberg aluminium plants contained very low levels of unchanged PAH, although the concentrations in the workplace air greatly exceeded the concentrations in urban air. The total concentration of PAH metabolites in the samples was 1.5–6 greater than that in a control group (Becher & Bjørseth, 1983).

The PAH concentrations in the air of aluminium plants is reduced dramatically by the use of tempered anodes instead of Söderberg anodes. Measurements of benzo[*a*]pyrene levels in French factories showed 1–36 μg/m³ in potrooms with Söderberg anodes and 0.004–0.6 μg/m³ in potrooms with tempered anodes (Barat, 1991).

5.3.2.2 *Foundries*

In personal air samples from workers in 10 Canadian foundries, mean concentrations of 0.14–1.8 μg/m³ benz[*a*]anthracene plus chrysene, 0.09–1.2 μg/m³ benzo[*a*]pyrene, and 0.09–1.9 μg/m³ dibenz[*a,h*]anthracene were measured. The benzo[*a*]pyrene levels in stationary air samples from six Finnish foundries were 0.01–1.3 μg/m³, depending on whether coal-tar pitch or coal powder was used as the moulding sand additive (IARC, 1984b).

In another study, the highest individual PAH levels were found in coke making, moulding, and furnaces (Gibson et al., 1977). Personal air samples from 67 Finnish foundry workers in 1990–91 showed benzo[*a*]pyrene concentrations of 2–60 ng/m³ with a mean of 8.6 ng/m³ (Perera et al., 1994). Depending on the foundry process and sand binder, the mean benzo[*a*]pyrene level in 29 French foundries varied from 3 to 2300 ng/m³ (Lafontaine et al., 1990).

Concentrations of PAH measured in foundries are shown in Table 72.

5.3.2.3 *Other workplaces*

Personal air samples from German chimney sweeps (year not given; 115 samples) showed an average benzo[*a*]pyrene level of 0.09 μg/m³, but eight of the samples exceeded 2 μg/m³. With an inhaled air volume of 10 m³ per working day, the daily intake of benzo[*a*]pyrene was estimated to be 0.24–2.7 μg, with a median value of 1.3 μg (Knecht et al., 1989).

In an Italian pyrite mine, pyrene levels of 0.03–0.21 μg/m³ were measured in personal and area air samples. The benzo[*a*]pyrene concentrations were below the limit of detection (Cenni et al., 1993). Area air samples taken in China showed total PAH levels of 3–40 μg/m³ in two iron mines and 4–530 μg/m³ in four copper mines. Individual compounds were not identified,

Table 72. Exposure to polycyclic aromatic hydrocarbons (μg/m³) in the atmosphere of foundries

Compound	[1]	[2]	[3]
Acenaphthene			0.03
Acenaphthylene			ND
Anthracene		2.31	0.05
Anthanthrene		0.64	
Benz[*a*]anthracene	0.008–0.221	0.67	0.01
Benzo[*a*]fluorene		0.48	
Benzo[*a*]pyrene	0.049–0.152	0.47	0.02
Benzo[*b*]fluoranthene		0.87[a]	0.003
Benzo[*b*]fluorene		0.41	
Benzo[*e*]pyrene		0.48	
Benzo[*ghi*]fluoranthene		0.15	
Benzo[*ghi*]perylene		0.72	0.05
Benzo[*k*]fluoranthene	0.037–0.458		0.02
Chrysene		0.82[b]	0.02
Coronene		0.21	
Dibenz[*a,h*]anthracene		0.20	ND
Fluoranthene		1.56	0.13
Fluorene			0.08
Indeno[1,2,3-*cd*]pyrene		0.81	ND
Naphthalene			9.68
Perylene		0.21	
Phenanthrene		4.46	0.32
Pyrene		1.74	0.01

ND, not detected; /, single measurements; **[1]** Canada, steel foundry; coke making, moulding, furnaces, finishing, and cranes (Gibson et al., 1977); **[2]** Western Germany, one foundry, area air samples (Knecht et al., 1986); **[3]** Denmark, 70 workers, personal air samples; melting, machine moulding, casting, sand preparation (Omland et al., 1994)

[a] In sum with benzo(j+k)fluoranthene
[b] In sum with triphenylene

but the main components were naphthalene and acenaphthene in the iron mines and naphthalene, benz[*a*]anthracene, benzo[*b*]fluoranthene, benzo[*a*]pyrene, benzo[*e*]pyrene, and dibenz[*a,h*]anthracene in the copper mines. The PAH concentrations probably resulted from the drilling of holes with hydraulic or pneumatic drills and by the transport of broken ore in diesel-powered scoops (Wu et al., 1992).

Area and personal air samples from workers in a railway tunnel in Italy showed pyrene levels of 0.04–0.30 μg/m³. The benzo[*a*]pyrene concentrations ranged from below the limit of detection to 0.04 μg/m³ (Cenni et al., 1993).

In the air of fish and meat smokehouses in Denmark (year not given), the maximum concentration of naphthalene in stationary air samples was about 2900 μg/m³. The most abundant compounds were naphthalene, phenanthrene, pyrene, fluorene, anthracene, and fluoranthene (> 100 μg/m³) (Nordholm et al., 1986). The minimal values were ≤ 1 μg/m³, benzo[*a*]pyrene being detected at minimal levels of 0.08 μg/m³ in meat smokehouses and 0.4 μg/m³ in fish smokehouses (Hansen et al., 1991b), with a maximum concentration of 78 μg/m³ (Nordholm et al., 1986).

In a further study in nine Danish meat smokehouses, naphthalene was detected at 21 μg/m³, fluorene at 6.9 μg/m³, fluoranthene at 6.6 μg/m³, phenanthrene at 5.6 μg/m³, acenaphthene at 5.2 μg/m³, chrysene at 1.2 μg/m³, anthracene at 1.1 μg/m³, pyrene at 0.2 μg/m³, and benzo[*ghi*]perylene at 0.2 μg/m³ (Hansen et al., 1992).

The concentrations of naphthalene, fluorene, anthracene, phenanthrene, pyrene, benzo[*a*]fluorene, chrysene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*e*]pyrene, benzo[*ghi*]perylene, and dibenz[*a,h*]anthracene in cooking fumes in a Finnish food factory, three restaurants, and one bakery (year not given) during the frying of meat and during deep-frying ranged between < 0.02 μg/m³ (the limit of detection) and 26 μg/m³. Naphthalene occurred at by far the highest concentration. Stationary air was sampled as close as possible to the active working area and the workers' breathing zone (Vainiotalo & Matveinen, 1993).

6. KINETICS AND METABOLISM IN LABORATORY MAMMALS AND HUMANS

Appraisal

Polycyclic aromatic hydrocarbons (PAH) are lipophilic compounds and can be absorbed through the lungs, the gastrointestinal tract, and the skin. In studies of the distribution of PAH in rodents, both the parent compounds and their metabolites were found in almost all tissues and particularly those rich in lipids. As a result of mucociliary clearance and hepatobiliary excretion, they were present, for example, in the gastrointestinal tract even when administered by other routes.

The metabolism of PAH to more water-soluble derivatives, which is a prerequisite for their excretion, is complex. Generally, the process involves epoxidation of double bonds, a reaction catalysed by cytochrome P450-dependent mono-oxygenases, rearrangement or hydration of the epoxides to yield phenols or diols, respectively, and conjugation of the hydroxylated derivatives. The reaction rates vary widely: interindividual variations of up to 75-fold have been observed, for example, with human macrophages, mammary epithelial cells, and bronchial explants from different donors.

All aspects of the absorption, metabolism, activation, and excretion of benzo[a]pyrene have been covered exhaustively in the published literature, but there is a dearth of information on many of the other PAH considered in this publication, particularly in humans. Thus, this overview sets out general principles and describes pathways relevant to benzo[a]pyrene in greater detail.

Most biotransformation leads to detoxification products that are conjugated and excreted in the urine and faeces. The human body burden of PAH has not been extensively studied, but tissue samples taken at autopsy were found in one study to contain benzo[a]pyrene at an average of 0.3 µg/100 g dry tissue; lung contained 0.2 µg/100 g. In contrast, the pathways by which several PAH are metabolized to reactive intermediates that bind covalently to nucleic acids have been examined in great detail. Although the commonest mechanism in animals and humans appears to involve the formation of diol epoxides, radical cations and sulfate esters of hydroxymethyl derivatives may also be important in certain cases.

6.1 Absorption

PAH are lipophilic compounds, soluble in organic solvents, that are usually devoid of ionizable or polar groups. Like many other xenobiotic substances, they would be expected to dissolve readily in, and be transported through, the external and internal lipoprotein membranes of mammalian cells. This is

confirmed by the uptake of PAH *in vitro* from media in which cells are maintained in culture and modified metabolically by enzymes of the endoplasmic reticulum. Furthermore, PAH are known to be able to cause biological effects *in vivo* in cells and tissues that are distant from their site of uptake by the organism.

In humans, the major routes of uptake of PAH are thought to be through (i) the lungs and the respiratory tract after inhalation of PAH-containing aerosols or of particulates to which a PAH, in the solid state, has become absorbed; (ii) the gastrointestinal tract after ingestion of contaminated food or water; and (iii) the skin as a result of contact with PAH-bearing materials.

6.1.1 Absorption by inhalation

Investigations of the pulmonary absorption of PAH have frequently been clouded by the existence of the mucociliary clearance mechanism, by which hydrocarbons absorbed onto particulates that have been inhaled are swept back up the pulmonary tree and are swallowed, thus entering the organism through the gastrointestinal tract. Use of isolated perfused rat lungs, however, provided a clear demonstration that benzo[*a*]pyrene is absorbed directly through the pulmonary epithelia. After intratracheal administration, both the hydrocarbon and its metabolites were detected in effluent perfusion fluid (Vainio et al., 1976). Other studies have shown that benzo[*a*]pyrene administered *in vivo* as an aerosol is cleared from the lungs of rats by a biphasic process in which an initial rapid phase (tracheal clearance) is followed by a much slower second phase (alveolar clearance) (Mitchell, 1982). PAH absorbed onto particles may take very much longer to be cleared from rodent lungs, however, than the free hydrocarbons, and the factors that affect this clearance rate include the structure of the hydrocarbon and the dimensions and chemical nature of the particles onto which the PAH are absorbed (Henry & Kaufman, 1973; Creasia et al., 1976; Nagel et al., 1976). For example, while 50% of the benzo[*a*]pyrene coated onto carbon particles of 15-30 μm was cleared from hamster lungs within 60 h, it took only 10 h to clear 50% of the benzo[*a*]pyrene that had been coated onto 0.5–1.0-μm carbon particles. In a comparable experiment, however, when ferric oxide particles of either 0.5–10 or 15–20 μm were used as carriers for benzo[*a*]pyrene, 50% of the hydrocarbon was cleared in just over 2 h, and carrier particle size did not affect the clearance rates (Henry & Kaufman, 1973).

Benzo[*a*]pyrene was metabolized by the epithelia lining the nasal cavities of hamsters, dogs, and monkeys when ^{14}C-labelled hydrocarbon was instilled as an aqueous suspension (Dahl et al., 1985; Petridou-Fischer et al., 1988). From their studies with hamsters, the authors concluded that when frequent small doses of 650 ng at 10-min intervals were instilled into the nasal cavity, so as to imitate inhalation, some 50% of the benzo[*a*]pyrene was metabolized;

a large fraction of the metabolites could be recovered from the mucus on the epithelial surfaces; and the nasal epithelia were comparable to those of the trachea and lungs in their ability to metabolize benzo[*a*]pyrene. Metabolites produced nasally would be expected to be swallowed and then absorbed in the gastrointestinal tract.

In humans, the concentrations of benzo[*a*]pyrene and pyrene present in association with soot particles in the lungs were much lower than would have been expected from the soot content. Thus, only a trace of benzo[*a*]pyrene was found in one of 11 lung samples examined, in which the expected benzo[*a*]pyrene content ranged from 9 to 200 µg; in the other 10 samples, no benzo[*a*]pyrene was detected. Pyrene disappeared more slowly: all 11 lung samples contained the compound, at levels of 0.9–4.9 µg, whereas 3–190 µg might have been expected (Falk et al., 1958). The ability of pulmonary epithelial cells to metabolize PAH such as chrysene and benzo[*a*]pyrene to a variety of hydroxylated derivatives (Jacob et al., 1992) may facilitate the absorption and clearance of PAH from the lungs.

6.1.2 *Absorption in the gastrointestinal tract*

Indirect evidence for the gastrointestinal absorption of PAH was provided by Shay et al. (1949), who found that repeated intragastric instillation of 3-methylcholanthrene led to the development of mammary cancer. Mammary tumours can also be induced in rats by intracolonic adminstration of 7,12-dimethylbenz[*a*]anthracene (Huggins et al., 1961). (3-Methylcholanthrene and 7,12-dimethylbenz[*a*]anthracene are synthetic PAH that are potent carcinogens.) More direct investigations by Rees et al. (1971) showed rapid absorption of intragastrically administered benzo[*a*]pyrene; the highest levels of hydrocarbon were found in the thoracic lymph some 3–4 h after administration. In a report of studies of intact rats and intestinal sacs to examine the mechanisms involved in benzo[*a*]pyrene absorption, Rees et al. (1971) proposed that two sequential steps were involved, in which a phase of absorption by the mucosa is followed by diffusion through the intestinal lining. In a study with Sprague-Dawley rats, the presence of bile was found to increase intestinal absorption of PAH such as benzo[*a*]pyrene and 7,12-dimethylbenz[*a*]anthracene to a greater degree than that of anthracene and pyrene. The effect may be related to differences in the aqueous solubility of the PAH examined (Rahman et al., 1986). The composition of the diet also affects intestinal absorption of co-administered benzo[*a*]pyrene. Of the dietary components studied, soya bean oil and triolein gave rise to the highest levels of absorption of ^{14}C-benzo[*a*]pyrene given orally at a dose of 8.7 µg to Wistar rats, while cellulose, lignin, bread, rice flake, and potato flake suppressed it (Kawamura et al., 1988).

6.1.3 *Absorption through skin*

PAH and PAH-containing materials have been applied dermally in solution in solvents such as acetone and tetrahydrofuran. Dermal transfer without use of a solvent was achieved by use of reconstituted vapour–particulate phases emitted from coal-tar and bitumen (Genevois et al., 1995) and by application in oil (Ingram et al., 1995).

Absorption of PAH through the skin was observed indirectly when it was found that repeated topical application of 3-methylcholanthrene led to the appearance of mammary tumours in mice (Maisin & Coolen, 1936; Englebreth-Holm, 1941). The percutaneous mechanism of absorption is not universal, however, since although almost all of a dose of ^{14}C-benzo[*a*]pyrene applied to mouse skin appeared in the faeces within two weeks, very little dibenz[*a,h*]anthracene was absorbed in this way and most was lost through epidermal sloughing (Heidelberger & Weiss, 1951). Benzo[*a*]pyrene has been shown to be absorbed percutaneously *in vitro*, by absorption from soil into human skin (Wester et al., 1990) and, after application as a solution in acetone, into discs of human, mouse, marmoset, rat, rabbit, and guinea-pig skin (Kao et al., 1985). In the latter experiments, marked interspecies differences were noted: 10% of the applied dose (10 µg/5 cm^2) of ^{14}C-benzo[*a*]pyrene permeated mouse skin, 3% crossed human skin, and < 0.5% crossed guinea-pig skin within 24 h. It was concluded that both diffusional and metabolic processes are involved in the percutaneous absorption of benzo[*a*]pyrene.

In Wistar rats that received ^{14}C-pyrene as a solution in acetone on areas of shaved dorsal skin, the rate of uptake was relatively rapid (half-life, 0.5–0.8 d). The concentrations of pyrene were highest in the liver, kidneys, and fat, but those of pyrene metabolites were highest in the lungs. About 50% of an applied dose of 2, 6, or 15 mg/kg bw was excreted in the urine and faeces during the first six days after treatment (Withey et al., 1993).

In studies with ^{32}P-postlabelling for the detection of DNA adducts, when complex mixtures of PAH, such as that present in used lubricating oil from petrol engines, in coal-tar, or in juniper-tar, were applied directly to mouse skin, appreciable, persistent levels of DNA adducts (50–750 amol/µg DNA [1 amol/µg DNA equivalent to 3.3 adducts/10^{10} nucleotides]) were formed in the lungs (Schoket et al., 1989, 1990). The level of adducts in mouse skin was inversely related to the viscosity of the oil applied (Ingram et al., 1995).

Evidence for percutaneous absorption of PAH has also been obtained in humans *in vivo*. When 2% coal-tar in petroleum jelly was applied topically, phenanthrene, anthracene, pyrene, and fluoranthene were detected in peripheral blood samples (Storer et al., 1984). In addition, volunteers treated topically with creosote (100 µl) or pyrene (500 µg, applied as a solution in toluene) and a psoriasis patient who used a coal-tar shampoo excreted 1-hydroxypyrene in their urine. In each case, maximal excretion occurred 10–15 h after treatment (Viau & Vyskocil, 1995).

6.2 Distribution

The whole-body distribution of PAH has been studied in rodents. The levels found in individual tissues depend on a number of factors, including the PAH, the route of administration, the vehicle, the times after treatment at which tissues are assayed, and the presence or absence of inducers or inhibitors of hydrocarbon metabolism within the organism. The investigations have shown that (i) detectable levels of PAH occur in almost all internal organs, (ii) organs rich in adipose tissue can serve as storage depots from which the hydrocarbons are gradually released, and (iii) the gastrointestinal tract contains high levels of hydrocarbon and metabolites, even when PAH are administered by other routes, as a result of mucociliary clearance and swallowing or hepatobiliary excretion (Heidelberger & Jones, 1948; Heidelberger & Weiss, 1951; Kotin et al., 1959; Bock & Dao, 1961; Takahashi & Yasuhira, 1973; Takahashi, 1978; Mitchell, 1982). ^{14}C-Benzo[*a*]pyrene injected intravenously at 11 µg/rat was cleared rapidly from the bloodstream, with a half-life of < 1 min (Kotin et al., 1959), as confirmed by Schlede et al. (1970a,b), who also noted that the rate of clearance was increased when animals were pretreated with 20 mg/kg bw non-radioactive benzo[*a*]pyrene or 37 mg/kg bw phenobarbital, both of which can induce metabolism.

The distribution of 3-methylcholanthrene in mice and their fetuses was studied by whole-body autoradiography. When 1 mg of ^{14}C-labelled hydrocarbon is injected intravenously, it is not only widely distributed in maternal tissues but also crosses the placenta and can be detected in the fetuses (Takahashi & Yasuhira, 1973; Takahashi, 1978), in which it induces pulmonary tumours (Tomatis, 1973; see also Section 7). The distribution of inhaled and intragastrically or intravenously administered benzo[*a*]pyrene and 7,12-dimethylbenz[*a*]anthracene in rats and mice has also been studied, with similar results (Shendrikova & Aleksandrov, 1974; Shendrikova et al., 1973, 1974; Neubert & Tapken, 1988; Withey et al., 1992). Rapid transfer of radioactive benzo[*a*]pyrene across the placenta was confirmed in experiments in which the appearance of radioactivity in the umbilical vein of pregnant guinea-pigs was measured (Kelman & Springer, 1982).

Samples of placenta, maternal blood, umbilical cord blood, and milk from 24 women in south India were examined for the presence of selected PAH. Although umbilical cord blood and milk showed the highest levels (benzo[*a*]pyrene, 0.005–0.41 ppm; dibenz[*a,c*]anthracene, 0.013–0.60 ppm; chrysene, 0.002–2.8 ppm), only 50% of the samples examined contained detectable levels. The authors concluded that developing fetuses and newborn infants were exposed to these PAH, probably from the maternal diet (Madhavan & Naidu, 1995).

After intratracheal administration to mice and rats, the distribution of PAH was essentially similar to that found after intravenous or subcutaneous injection

(Kotin et al., 1959), except for the expected high pulmonary levels. Detailed time-concentration curves for several organs have been obtained after inhalation of ^{3}H-benzo[*a*]pyrene aerosols at 500 µg/litre of air (Mitchell, 1982). For example, 1 h after the end of administration, the highest levels were present in the stomach and small intestine; as these declined, the amounts of radioactivity in the large intestine and caecum increased. The elimination half-times in the respiratory tract were 2–3 h for the initial rapid phase and 25–50 h for the subsequent slow phase.

6.3 Metabolic transformation

The metabolism of PAH follows the general scheme of xenobiotic metabolism originally outlined by Williams (1959). The hydrocarbons are first oxidized to form phase-I metabolites, including primary metabolites, such as epoxides, phenols, and dihydrodiols, and then secondary metabolites, such as diol epoxides, tetrahydrotetrols, and phenol epoxides. The phase-I metabolites are then conjugated with either glutathione, sulfate, or glucuronic acid to form phase-II metabolites, which are much more polar and water-soluble than the parent hydrocarbons.

The metabolism of PAH has been studied *in vitro*, usually in microsomal fractions prepared from rat liver, although many other tissue preparations have also been used. Metabolism in such systems might be expected to be simpler than that in whole animals because the enzymes and co-factors necessary for sulfate, glutathione, or glucuronide conjugate formation may be removed, depleted, or diluted during tissue fractionation. Use of these systems appears to be justified, however, because the same types of phase-I metabolites are formed when animals are treated with simple hydrocarbons such as naphthalene as when the same hydrocarbon is incubated with hepatic microsomes or tissue homogenates (Boyland et al., 1964). The metabolism of PAH has thus been studied extensively in cells and tissues in culture, which metabolize hydrocarbons to both phase-I and phase-II metabolites and which probably better represent the metabolism of PAH that occurs *in vivo* (for reviews see Conney, 1982; Cooper et al., 1983; Dipple et al., 1984; Hall & Grover, 1990; Shaw & Connell, 1994).

Particular attention has been paid to the metabolism of PAH in human tissues that might be exposed to hydrocarbons present in food and in the environment and which are, therefore, potential targets for the carcinogenic action of PAH (Autrup & Harris, 1983). The cells and tissues examined include the bronchus, the colon, mammary cell aggregates, keratinocytes, monocytes, and lymphocytes. The metabolism of PAH by human pulmonary macrophages has also received attention (Autrup et al., 1978a; Harris et al., 1978a; Marshall et al., 1979) because it is conceivable that metabolism by these cells might be responsible, at least in part, for the high incidence of bronchial cancer in

smokers (Wynder et al., 1970). Macrophages can engulf particulate matter that reaches the terminal airways of the lung and thus would be expected, especially in smokers, to contain PAH (Hoffmann et al., 1978). The macrophages and engulfed particulate matter can then be transported to the bronchi where proximate and ultimate carcinogens, formed by metabolism in the macrophages, could leave the macrophages and enter the epithelial cells lining the bronchi (Autrup et al., 1978a; Harris et al., 1978a). This is an attractive theoretical mechanism which could account for the high incidence of respiratory tumours at the junctions of the large bronchi and which is supported by experimental evidence.

Extracts of organic material from isolated perfused lung tissues of rabbits that had been exposed intratracheally to benzo[*a*]pyrene with or without ferric oxide were analysed for benzo[*a*]pyrene metabolites and for mutagenicity. Extracts of lung tissue exposed to benzo[*a*]pyrene only were mutagenic and contained benzo[*a*]pyrene metabolites. When ferric oxide was co-administered, only the macrophage extracts were mutagenic, owing to relatively large amounts of unmetabolized benzo[*a*]pyrene. These experiments demonstrate that ferric oxide particles enhance the uptake of benzo[*a*]pyrene by lung macrophages and slow its metabolism beyond the 3-h period during which perfused lung systems can be maintained (Schoeny & Warshawsky, 1983).

Administration of particles *in vitro* enhances both the uptake and metabolism of benzo[*a*]pyrene by hamster alveolar macrophages (Grieie et al., 1988). Metabolites were found in both the cells and the culture medium. Subsequent studies showed that concurrent administration of benzo[*a*]pyrene and ferric oxide particles resulted in increased benzo[*a*]pyrene metabolism and release of superoxides (Greife & Warshawsky, 1993). In particular, the dihydrodiol fraction was increased. These studies indicate that particulates may act in lung cancer by changing the time frame for metabolism, shifting the site of metabolism to macrophages and enhancing the production of metabolites that are on the pathway to putative ultimate carcinogenic forms. In this context, it has been demonstrated that particles of various sorts exert different toxic effects on rat and hamster pulmonary macrophages *in vitro*: ferric oxide and aluminium oxide particulates were toxic, while crystalline silica was not (Warshawsky et al., 1994).

The conclusion that the macrophage is the principal metabolizing cell is further supported by the studies of Ladics et al. (1992a,b), who demonstrated that the macrophage population was the only one in murine spleen that could metabolize benzo[*a*]pyrene, while the other splenic cell types examined, including B cells, T cells, polymorphonuclear cells, and the splenic capsule, did not produce benzo[*a*]pyrene metabolites above the background level.

Although the same types of metabolite are formed from PAH in many of the cell and tissue preparations examined in culture, the relative levels and the rates of formation of these metabolites depend on the type of tissue or cell that

is being studied and on the species and strain of animal from which the metabolizing systems are prepared. With heterogeneous populations such as humans, the rate of metabolism depends on the individual from whom the tissues or cells are prepared. For example, a 75-fold variation in the extent of hydrocarbon activation was reported in studies of human bronchus (Harris et al., 1976), and similar variations were observed among human mammary cell aggregates (Grover et al., 1980; MacNicoll et al., 1980) and macrophages (Autrup et al., 1978a). The pattern and role of metabolism can also be varied by adding inhibitors of the enzymes that are responsible for metabolism or by pretreating either cells in culture or the animals from which the metabolizing systems are prepared with enzyme inducers.

6.3.1 *Cytochromes P450 and metabolism of PAH*

The cytochromes P450 (CYP) are a superfamily of haemoproteins that catalyse the oxidation of various endogenous molecules as well as xenobiotics, including PAH. To date, about 250 genes that encode these enzymes have been identified in various organisms. For classification purposes, the CYP have been organized into families and subfamilies according to their structural homology (Nelson et al., 1993).

Certain CYP belonging to families 1, 2, and 3 are expressed in mammalian cells and are particularly important in xenobiotic metabolism, and one or more member of each family is capable of metabolizing one or more PAH (Guengerich & Shimada, 1991; Gonzalez & Gelboin, 1994). Most studies to compare the catalytic properties of different CYP have been carried out with model compounds such as benzo[*a*]pyrene. They show that the catalytic properties (e.g. the V_{max}) of different CYP in PAH metabolism can differ essentially (Shou et al., 1994).

In considering the contribution of a CYP enzyme to PAH metabolism *in vivo*, two other parameters in addition to the catalytic properties should be taken into account: the mode of regulation and tissue specificity in its expression. Combinations of the three factors should give an idea of the relative importance of an enzyme in PAH metabolism.

6.3.1.1 *Individual cytochrome P450 enzymes that metabolize PAH*

CYP1A: CYP1A appears to be the only enzyme with metabolic capability towards a wide variety of PAH molecules. It is expressed in various tissues but at a generally low constitutive level (Guengerich & Shimada, 1991). The induction of CYP1A1 is controlled by the Ah (aryl hydrocarbon) receptor, a transcription factor that can be activated by several ligands such as 2,3,7,8-tetradichlorobenzo-*para*-dioxin (TCDD) and PAH, with variable potency (Negishi et al., 1981). Thus, PAH and material containing PAH can regulate their own metabolism by inducing CYP1A1. After induction, CYP1A1

expression may reach high levels, e.g. in the placenta, lung, and peripheral blood cells; however, in the liver, the principal organ of xenobiotic metabolism, the level of expression is low even after induction, and other CYP appear to be more important, at least in the metabolism of benzo[*a*]pyrene (Guengerich & Shimada, 1991).

CYP1A2: The other member of the CYP1A family, CYP1A2, also metabolizes PAH; however, its capacity to metabolize benzo[*a*]pyrene to the 3-hydroxy metabolite, for example, is about one-fifth that of CYP1A1 (Shou et al., 1994). Human CYP1A2 is nevertheless very active in forming benzo[*a*]pyrene 7,8-dihydrodiol (Bauer et al., 1995) and in forming diol epoxides from the 7,8-dihydrodiol (Shou et al., 1994). There is also evidence that CYP1A2 can activate 7,12-dimethylbenz[*a*]anthracene to mutagenic species, albeit at a low rate (Aoyama et al., 1989).

The expression of CYP1A2 is also regulated by the Ah receptor, but in not exactly the same way as CYP1A1 (Negishi et al., 1981). In the liver, for example, the level of CYP1A2 expression is much higher than that of CYP1A1 (Guengerich & Shimada, 1991). While the capacity of CYP1A2 to oxidize various PAH is more limited than that of CYP1A1, its role in reactions like diol epoxide formation from benzo[*a*]pyrene in the liver could be important because of its high level of expression.

CYP1B: The CYP1B subfamily was discovered only recently. Once the enzyme had been isolated, it was found to be capable of metabolizing PAH. Interestingly, its expression is also under the control of the Ah receptor. Only limited information is available on its expression and catalytic properties in different tissues, but it seems to be expressed at least in mouse embryo fibroblasts (Savas et al., 1994), rat adrenal glands (Bhattacharyya et al., 1995), and several human tissues (Sutter et al., 1994). A number of PAH may act as substrates for this enzyme (Shen et al., 1994).

CYP2B: When recombinant gene technology was used to express human CYP2B6 cDNA in a human lymphoblastoid cell line, this enzyme was shown to be capable of metabolizing benzo[*a*]pyrene to 3- and 9-phenols and *trans*-dihydrodiols (Shou et al., 1994). In addition, CYP2B enzymes may be involved in the metabolism of 7,12-dimethylbenz[*a*]anthracene (Morrison et al., 1991a).

The constitutive levels of CYP2B enzymes are extremely low in human liver, but they are strongly induced by phenobarbital and phenobarbital-type inducers of CYP. Accordingly, immunological studies of inhibition have shown that the CYP2B enzymes may play a significant role in the metabolism of PAH, only when they are induced (Hall et al., 1989; Honkakoski & Lang, 1989).

CYP2C: The CYP2C subfamily contains several members, some of which are expressed at high levels in human liver. More than one member of this subfamily may be capable of metabolizing PAH; thus, human CYP2C9 and, to a lesser extent, CYP2C8 metabolize benzo[*a*]pyrene to 3- and 9-phenols and

trans-dihydrodiols (Shou et al., 1994). In addition, CYP2C enzymes may play an essential role in the metabolism of benzo[*a*]pyrene and 7,12-dimethylbenz[*a*]anthracene, particularly in phenobarbital-induced liver (Morrison et al., 1991a,b; Todorovic et al., 1991). In view of the relative abundance of CYP in human liver and their role in the metabolism of PAH, it has been suggested that some CYP2C enzymes play an essential role in hepatic PAH metabolism (Morrison et al., 1991b; Yun et al., 1992).

CYP3A: CYP3A is one of the most abundant CYP enzymes in human liver, and it can metabolize benzo[*a*]pyrene and some of its dihydrodiols to several metabolic products (Shimada et al., 1989; Yun et al., 1992; Shou et al., 1994; Bauer et al., 1995). In one study, human CYP3A4 was the most important single enzyme in the hepatic 3-hydroxylation of benzo[*a*]pyrene (Yun et al., 1992).

6.3.1.2 *Regulation of cytochrome P450 enzymes that metabolize PAH*

All of the enzymes discussed above are inducible, and their level of expression can be enhanced by external stimuli. CYP1A and CYP1B are under the transcriptional control of the Ah receptor, which can be activated by numerous PAH and other planar hydrocarbons, including dioxins (Negishi et al., 1981; Guengerich & Shimada, 1991)

CYP2B enzymes can also be induced by foreign compounds but not through the Ah receptor. The mechanism of induction of these enzymes is not well understood, but their prototype inducer is phenobarbital; several other drugs used clinically have similar effects (Gonzalez & Gelboin, 1994).

The regulation of CYP2C enzymes is complicated, and both endogenous factors such as steroid hormones and exogenous factors such as phenobarbital may be involved. Furthermore, different members of this subfamily are regulated differently. The CYP3A are also regulated by endogenous and exogenous factors; typical inducers of this subfamily are rifampicin, dexamethasone, certain macrolide antibiotics, and steroid hormones (Guengerich & Shimada, 1991).

Genetic polymorphisms of CYP1A1, CYP1A2, and some CYP2C and CYP3A enzymes have also been described. Some of the genetic defects leading to the polymorphism have been identified and can be used to predict an individual's capacity to metabolize drugs, for example by the polymerase chain reaction. Genetic polymorphism may lead to dramatic changes in the capacity to metabolize PAH (Raunio & Pelkonen, 1994).

Studies with a few prototype compounds such as benzo[*a*]pyrene and its metabolites and 7,12-dimethylbenz[*a*]anthracene indicate that several CYP are involved in PAH metabolism. As each has its own metabolic capacity, mode of regulation, and tissue-specific expression, the one that plays a key role in PAH metabolism *in vivo* at any one time may vary and will depend on the

compound being metabolized, pre-exposure to inducers of the CYP, the tissue and cell type where the metabolism is taking place, and the genotype of the individual in cases of genetic polymorphism.

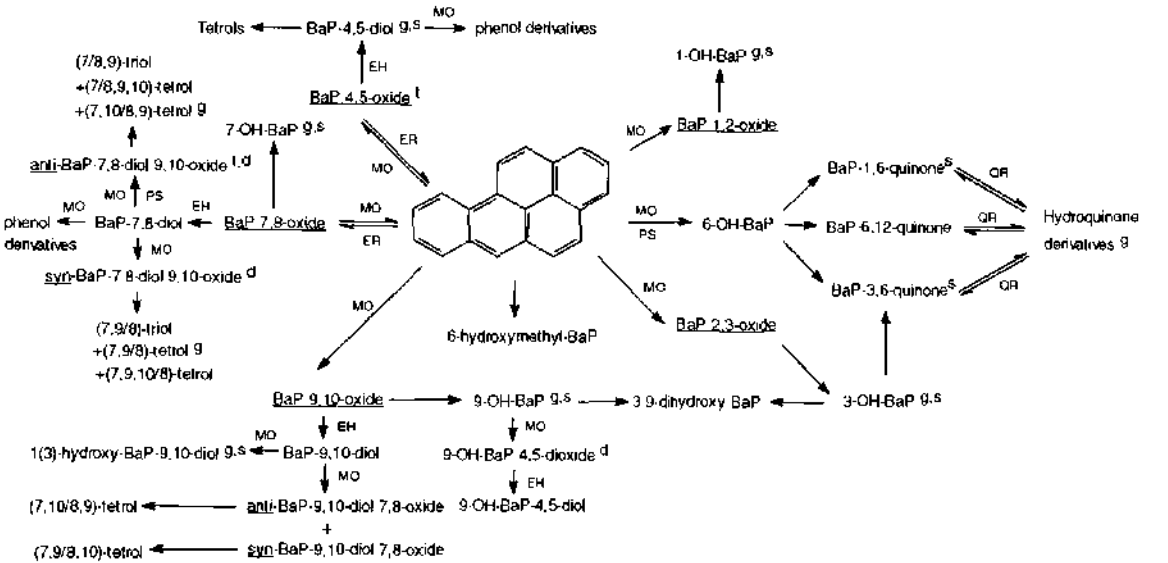
Many PAH that are metabolized by the CYP-dependent mono-oxygenases also induce the enzyme system. This ability of hydrocarbons to induce their own metabolism usually results in lower tissue levels and more rapid excretion of the hydrocarbon (Schlede et al., 1970b; Aitio, 1974). Although CYP1A1 is mainly responsible for activation of PAH in the lung and CYP1A2 in the liver, most recent investigations have shown that other CYP isoforms may also contribute to the metabolism of PAH in mammals (Jacob et al., 1996). Thus, pretreatment of animals with inducers of mono-oxygenase systems is frequently associated with a decreased tumour incidence (Wattenberg, 1978). Conversely, studies with strains of mice that differ genetically in the capacity of their mono-oxygenase systems to be induced by PAH indicate that inducibility may also be associated with an increased tumorigenic or toxicological response (Nebert, 1980). Induction of the mono-oxygenase system by different types of inducers can result in different profiles of hydrocarbon metabolites, although the extent of the effect appears to be variable (Holder et al., 1974; Jacob et al., 1981a,b; Schmoldt et al., 1981). The metabolism of benzo[*a*]pyrene has been investigated in more detail than that of other hydrocarbons and is used here as an example.

6.3.2 *Metabolism of benzo[a]pyrene*

In early studies, the PAH metabolites isolated from or excreted by experimental animals were shown to consist of hydroxylated derivatives, commonly in the form of conjugates. Thus, the general scheme of xenobiotic metabolism outlined above applies to PAH. One of the principal interests in hydrocarbon metabolism arose, however, from the realization that hydrocarbons, like many other environmental carcinogens, are chemically unreactive and that their adverse biological effects are probably mediated by electrophilic metabolites capable of covalent interaction with critical macromolecules such as DNA. Identification of the biologically active metabolites of PAH, coupled with advances in both the synthesis of known and potential hydrocarbon metabolites and the analysis of metabolites by high-performance liquid chromatography, has led in the last two decades to a greatly enhanced appreciation of the complexity of hydrocarbon metabolism. Most of these metabolic interrelationships are illustrated for benzo[*a*]pyrene in Figure 3; the structures of some types of metabolites are given in Figure 4. The metabolism of benzo[*a*]pyrene and other PAH has been reviewed (for example, Sims & Grover, 1974, 1981; Conney, 1982; Cooper et al., 1983; Dipple et al., 1984; Hall & Grover, 1990).

Benzo[*a*]pyrene is metabolized initially by the microsomal CYP-dependent mono-oxygenase system to several epoxides (Figure 3). Once formed, these

Figure 3. Reactions and pathways involved in the metabolism of benzo[a]pyrene



From Cooper et al. (1983), with permission; BaP, benzo[a]pyrene; EH, epoxide hydralase; ER, epoxide reductase; MO, mono-oxygenase; PS; prostaglandin endoperoxide synthase; QR, quinone reductase

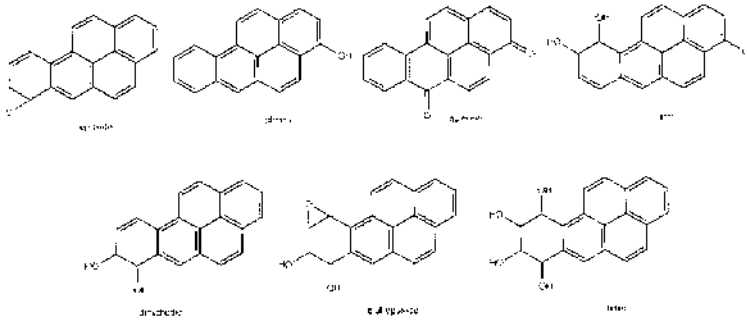
[a] Can be conjugated with glutathione

[b] Can be conjugated with sulfate

[c] Can be conjugated with glucuronic acid

[d] Can contribute to the covalent reactions of hydrocarbons with nucleic acids in cells or tissues treated with benzo[a]pyrene

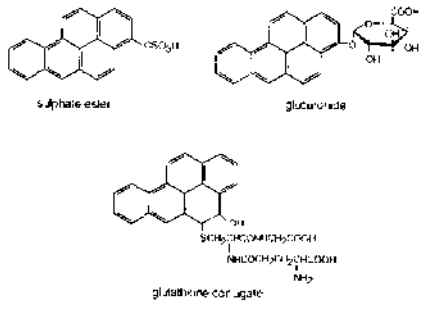
Figure 4. Structures of some types of metabolite of benzo[a]pyrene



epoxides (Sims & Grover, 1974) may spontaneously rearrange to phenols, be hydrated to dihydrodiols in a reaction that is catalysed by epoxide hydrolase (see review by Oesch 1973), or react covalently with glutathione, either chemically or in a reaction catalysed by glutathione *S*-transferase (Chasseaud, 1979). 6-Hydroxybenzo[*a*]pyrene is further oxidized either spontaneously or metabolically to the 1,6-, 3,6-, or 6,12-quinone, and this phenol is also a presumed intermediate in the oxidation of benzo[*a*]pyrene to the three quinones that is catalysed by prostaglandin H synthase. Two additional phenols may undergo further oxidative metabolism: 3-hydroxybenzo[*a*]pyrene is metabolized to the 3,6-quinone, and 9-hydroxybenzo[*a*]pyrene is oxidized to the K-region 4,5-oxide, which is hydrated to the corresponding 9-hydroxy 4,5-dihydrodiol (Jernström et al., 1978; for a formula showing a K-region, see Figure 11). Phenols, quinones, and dihydrodiols can all be conjugated to yield glucuronides and sulfate esters, and the quinones may also form glutathione conjugates (Figure 5).

In addition to being conjugated, dihydrodiols can undergo further oxidative metabolism. The mono-oxygenase system metabolizes benzo[*a*]pyrene 4,5-diol to a number of metabolites, while the 9,10-dihydrodiol is metabolized predominantly to its 1- and 3-phenol derivatives, only minor quantities of a 9,10-diol-7,8-epoxide being formed. In contrast to 9,10-dihydrodiol metabolism, the principal route of oxidative metabolism of benzo[*a*]pyrene 7,8-dihydrodiol is to a 7,8-diol 9,10-epoxide, and triol formation is a minor pathway. The diol epoxides can themselves be further metabolized to triol epoxides and pentols (Dock et al., 1986) and can become conjugated with glutathione either through chemical reaction or via a glutathione *S*-transferase-catalysed reaction (Cooper et al., 1980; Jernström et al., 1985; Robertson et al., 1986). They may also spontaneously hydrolyse to tetrols, although epoxide hydrolase does not appear to catalyse this hydration. Further oxidative metabolism of benzo[*a*]pyrene 7,8-diol can also be catalysed by prostaglandin H synthase (Marnett et al., 1978; Eling et al., 1986; Eling & Curtis, 1992), by a myeloperoxidase system (Mallett et al., 1991), or by lipoxygenases (Hughes et

Figure 5. Structures of some benzo[*a*]pyrene conjugates

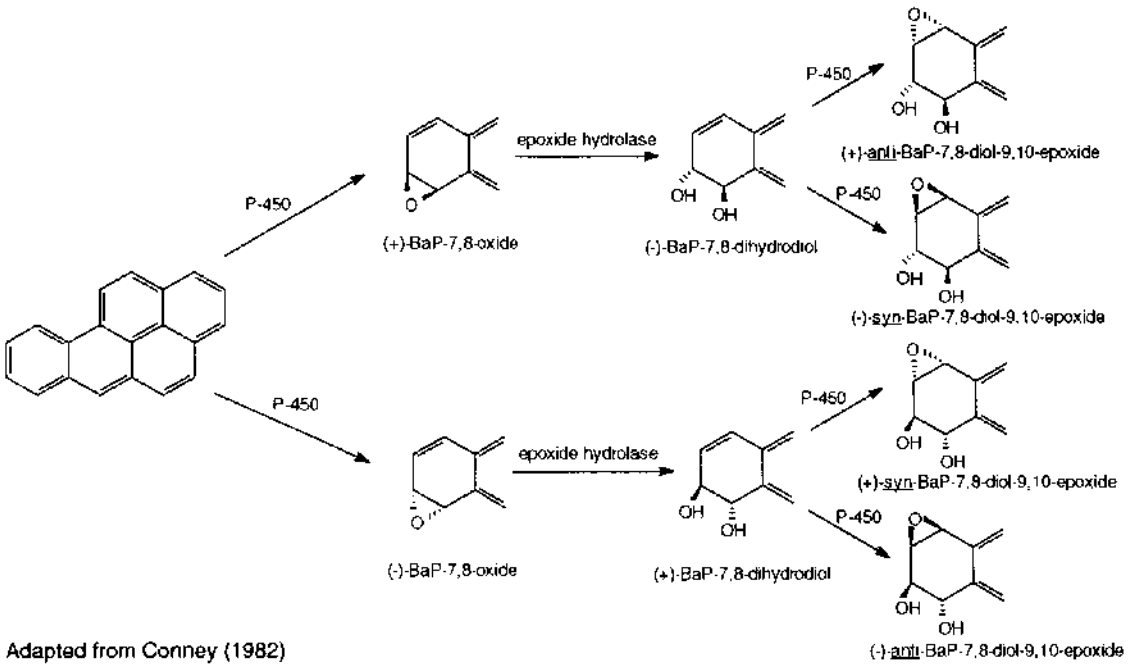


al., 1989). These reactions may be of particular importance in situations in which there are relatively low levels of CYP (i.e. in uninduced cells and tissues) or when chronic irritation and/or inflammation occurs, as during cigarette smoking (Kensler et al., 1987; Ji & Marnett, 1992). The products detected have included diol epoxides (Mallet et al., 1991; Ji & Marnett, 1992) and tetrols (Sivarajah et al., 1979). Taken together, these reactions illustrate that benzo[*a*]pyrene in particular, and PAH in general, can undergo a multitude of simultaneous or sequential metabolic transformations; they also illustrate the difficulty in determining which metabolites are responsible for the various biological effects resulting from treatment with the parent PAH.

An additional complexity of hydrocarbon metabolism stems from the fact that the compounds are metabolized to optically active products. Figure 6 illustrates the stereoselective metabolism of benzo[*a*]pyrene to the 7,8-diol-9,10-epoxides. Four isomers may be generated, since each diastereomer can be resolved into two enantiomers. In rat liver microsomes, the (+) 7,8-epoxide of benzo[*a*]pyrene is formed in excess relative to the (–) isomer, such that more than 90% of the benzo[*a*]pyrene 7,8-oxide formed consists of the (+) enantiomer (Levin et al., 1982). The epoxide is then metabolized stereospecifically by epoxide hydrolase to the (–) 7,8-dihydrodiol. This metabolically predominant dihydrodiol is metabolized in turn, primarily to a single diol epoxide isomer, the (+)*anti*-benzo[*a*]pyrene 7,8-diol-9,10-epoxide. The biological significance of the stereoselective formation of the 7,8-diol-9,10-epoxide isomers is that the metabolically predominant isomer is also the isomer with the highest tumour-inducing activity and that found predominantly to be covalently bound to DNA in a variety of mammalian cells and organs that have been exposed to benzo[*a*]pyrene.

Benzo[*a*]pyrene metabolism has been examined extensively in human tissue preparations, including human cells, explant cultures, tissue homogenates, and microsomal preparations. Table 73 lists some studies of the metabolism of benzo[*a*]pyrene in human tissues that included metabolites soluble in organic

Figure 6. Stereoselective pathways involved in the formation of 'bay-region' vicinal diol epoxides of benzo[*a*]pyrene



Adapted from Conney (1982)

Table 73. Metabolites of benzo[*a*]pyrene formed by human tissues and cells

Tissue or cell type	Type of metabolite detected					References
	Dihydrodiols	Phenols	Quinones	Tetrols	Conjugates	
Bronchus	+	+	+	+	+	Pal et al. (1975); Cohen et al. (1976); Harris et al. (1977); Autrup et al. (1978a, 1980)
Colon	+	+	+	+	+	Autrup et al. (1978b); Autrup (1979)
Endometrium	+	+	+			Mass et al. (1981)
Fibroblasts	+					Baird & Diamond (1978)
Kidney	+	+	+			Prough et al. (1979)
Liver	+	+	+		+	Selkirk et al. (1975); Prough et al. (1979); Pelkonen et al. (1977); Diamond et al. (1980)
Lung	+	+	+	+	+	Cohen et al. (1976); Stoner et al. (1978); Mehta et al. (1979); Prough et al. (1979); Sipal. et al. (1979)

Table 73 (contd)

Tissue or cell type	Type of metabolite detected					References
	Dihydrodiols	Phenols	Quinones	Tetrols	Conjugates	
Lymphocytes	+	+	+			Booth et al. (1974); Selkirk et al. (1975); Vaught et al. (1978); Okano et al. (1979); Gurtoo et al. (1980)
Macrophages	+	+	+	+	+	Autrup et al. (1978a); Harris et al. (1978a,b); Autrup et al. (1979); Marshall et al. (1979)
Mammary epithelium	+					Grover et al. (1980); MacNicoll et al. (1980)
Monocytes	+	+	+			Vaught et al. (1978); Okano et al. (1979)
Oesophagus	+	+	+	+		Harris et al. (1979)
Placenta	+	+	+			Namkung & Juchau (1980); Pelkonen & Saarni (1980)
Skin	+	+	+	+	+	Fox et al. (1975); Vermorken et al. (1979); Parkinson & Newbold (1980); Kuroki et al. (1980)

solvents and water-soluble conjugates. The results show that the metabolites produced by different human tissues are qualitatively similar and that the metabolites detected are the same as those formed in a variety of animal tissues. The metabolic profiles reported in human tissues are almost all identical to those seen for other eukaryotes, indicating the involvement of similar enzyme systems. The same types of reactive electrophilic intermediates found in other experimental systems also appear to be formed in human tissues (Autrup & Harris, 1983). So far, no differences in the metabolism or activation of benzo[*a*]pyrene have been reported that might account for differences in the susceptibility of different animal and human tissues to its carcinogenic properties (see Section 7). Studies with cultured cells and other substrates such as benz[*a*]anthracene, however, indicate that bioactivation of PAH is species-dependent (Jacob, 1996).

6.4 Elimination and excretion

Most metabolites of PAH are excreted in faeces and urine. As complete breakdown of the benzene rings of which unsubstituted PAH are composed does not occur to any appreciable extent in higher organisms, very little of an administered dose of an unsubstituted hydrocarbon would be expected to appear as carbon dioxide in expired air.

The urinary excretion of PAH metabolites has been studied more extensively than faecal excretion, but the importance of the enterohepatic circulation of metabolites has led to increased research on the latter. Detailed studies of the metabolism and excretion of PAH in whole animals have been restricted mainly to the simpler compounds. Because of the toxicity of the larger hydrocarbons and the complexity of their metabolism, most studies on these compounds have been carried out in hepatic homogenates and microsomal preparations or with cultured cells (see above).

Metabolism and excretion in whole animals have been examined with regard to naphthalene (Bourne & Young, 1934; Young, 1947; Booth & Boyland, 1949; Corner & Young, 1954; Corner et al., 1954; Boyland & Sims, 1958; Sims, 1959), anthracene (Boyland & Levi, 1935, 1936a,b; Sims, 1964), phenanthrene (Boyland & Wolf, 1950; Sims, 1962; Boyland & Sims, 1962a,b; Jacob et al., 1990b; Grimmer et al., 1991a), pyrene (Harper, 1957, 1958a; Boyland & Sims, 1964a; Jacob et al., 1989, 1990b), benz[*a*]-anthracene (Harper 1959a,b; Boyland & Sims, 1964b), and chrysene (Grimmer et al., 1988b, 1990). A limited number of studies have been published on more complex compounds such as benzo[*a*]pyrene (Berenblum & Schoental, 1943; Weigert & Mottram, 1946; Harper, 1958b,c; Falk et al., 1962; Raha, 1972; Jacob et al., 1990b), dibenz[*a*,*h*]anthracene (Dobriner et al., 1939; Boyland et al., 1941; La Budde & Heidelberger, 1958), and 3-methylcholanthrene (Harper, 1959a; Takahashi & Yasuhira, 1972; Takahashi, 1978). Much of the earlier

qualitative work was reviewed by Boyland & Weigart (1947) and by Young (1950). The absorption and excretion of different hydrocarbons *in vivo* can differ. For example, while almost all of a topically applied dose of benzo[*a*]pyrene appeared in mouse faeces (Heidelberger & Weiss, 1951), little dibenz[*a*,*h*]-anthracene was excreted by this route.

In rats given PAH either singly or as mixtures, the faecal elimination of chrysene (25% of the dose) was not affected by co-administration of benz[*a*]anthracene, but that of benz[*a*]anthracene was doubled, from 6 to 13% of the dose, when chrysene was given (Bartosek et al., 1984). Such effects are relevant to human pharmacokinetics, since exposure is almost always to mixtures of PAH. In workers in a coke plant exposed to mixtures of PAH, the amounts of phenanthrene, pyrene, and benzo[*a*]pyrene inhaled and the amounts of their principal metabolites excreted in the urine were correlated (Grimmer et al., 1994).

In rats, the amount of benzo[*a*]pyrene 7,8-diol excreted in the urine is related to the susceptibility of individual animals to the carcinogenic effects of benzo[*a*]pyrene (Likhachev et al., 1992; Tyndyk et al., 1994). In studies of the disposition of benzo[*a*]pyrene in rats, hamsters, and guinea-pigs after intratracheal administration, the distribution of the hydrocarbon was qualitatively similar but quantitatively different. In Sprague-Dawley and Gunn rats and in guinea-pigs, the rate of excretion was dependent on the dose administered, but in hamsters the rate of excretion was independent of dose (0.16 or 350 µg ^{3}H-benzo[*a*]pyrene) (Weyand & Bevan 1986, 1987a). Evidence for enterohepatic circulation of benzo[*a*]pyrene metabolites was obtained in Sprague-Dawley rats with bile-duct cannulae treated by intratracheal instillation with 1 µg/kg bw ^{3}H-benzo[*a*]pyrene (Weyand & Bevan, 1986). The results of a study of the pharmacokinetics and bioavailability of pyrene in rats strongly suggested that enterohepatic recycling took place after oral or intravenous administration of ^{14}C-labelled compound at 2–15 mg/kg bw (Withey et al., 1991).

Other studies on the enterohepatic circulation of PAH in rats and rabbits have also shown that the significant amounts of metabolites excreted in the bile persist *in vivo* because of enterohepatic circulation (Chipman et al., 1981; Chipman, 1982; Boroujerdi et al., 1981). For example, while some 60% of an intravenous dose of 3 µmol/kg bw ^{14}C-benzo[*a*]pyrene was excreted in bile, only 3% appeared in urine within the first 6 h after injection (Chipman et al., 1981). Biliary metabolites of xenobiotic compounds are usually polar and nonreactive, but mutagenic or potentially mutagenic derivatives may be excreted by this route into the intestine (for a review, see Chipman, 1982). Glucuronic acid conjugates of biliary metabolites can be hydrolysed by some intestinal flora to potentially reactive species (Renwick & Drasar, 1976; Chipman et al., 1981; Boroujerdi et al., 1981; Chipman, 1982). Thio-ether conjugates of hydrocarbons may also be involved in enterohepatic circulation (Hirom et al., 1983; Bakke et al., 1983), although there is no evidence that these

represent a mutagenic or carcinogenic hazard to the tissues through which they pass.

In a controlled study in humans, a 100–250-fold increase in dietary exposure to PAH, as measured by benzo[*a*]pyrene intake, resulted in a 4–12-fold increase in urinary excretion of 1-hydroxypyrene. The authors concluded that dietary exposure to PAH is as substantial as some occupational exposures (Buckley & Lioy, 1992).

6.5 Retention and turnover

Very little is known about the retention and turnover of PAH in mammalian species. It can be deduced from the few data available on hydrocarbon body burdens (see below) that PAH themselves do not persist for long periods and must therefore turn over reasonably rapidly. During metabolism, PAH moieties become covalently bound to tissue constituents such as proteins and nucleic acids. Protein-bound metabolites are likely to persist, therefore, for periods that do not exceed the normal lifetime of the protein itself. Nucleic acid adducts formed from reactions of PAH metabolites can be expected to differ in their persistence in the body according to whether they are RNA or DNA adducts. Although most DNA adducts are removed relatively rapidly by repair, small fractions can persist for long periods. The persistence of these adducts in tissues such as mouse skin is of considerable interest since one of the basic features of the two-stage mechanism of carcinogenesis (Berenblum & Shubik, 1947) is that application of the tumour promoter can be delayed for many months without markedly reducing the eventual tumour yield.

The persistence of adducts is also consistent with multistage theories of carcinogenesis, in which multiple steps in neoplastic transformation are dependent on the mutagenic and other actions of carcinogens.

6.5.1 *Human body burdens of PAH*

Since the effects of chemical carcinogens are likely to be related to both the dose and the duration of exposure, it is important to determine the human body load of carcinogens during a lifetime. It has been estimated that the total intake of PAH over a 70-year lifespan may amount to the equivalent of 300 mg of benzo[*a*]pyrene (Lutz & Schlatter, 1992); however, inhabitants of conurbations are likely to inhale additional amounts of PAH. Of course, much of the intake of PAH is metabolized and excreted. Thus, the pulmonary tissues of elderly town dwellers in Russia contained 1000 times less benzo[*a*]pyrene (< 0.1 µg per individual) than might have been expected from the estimated intake figures alone (Shabad & Dikun, 1959). Some experiments with cows and domestic fowl fed diets containing added benzo[*a*]pyrene tend to confirm this finding, since the meat, milk, and eggs produced were, after a suitable delay,

reported to be much less heavily contaminated than might have been expected from the amounts of benzo[*a*]pyrene administered (Gorelova & Cherepanova, 1970). More recent data are not available.

The average benzo[*a*]pyrene levels (measured by ultraviolet spectroscopy) in tissues taken at autopsy from normal people of a wide age range were 0.32 µg/100 g dry tissue weight in liver, spleen, kidney, heart, and skeletal muscle and 0.2 µg/100 g in lung (Gräf, 1970; Gräf et al., 1975).

When cancer-free liver and fat from six individuals were assayed for nine hydrocarbons by co-chromatography with authentic standards, pyrene, anthracene, benzo[*b*]fluoranthene, benzo[*ghi*]perylene, benzo[*k*]fluoranthene, and benzo[*a*]pyrene were detected at average levels of 380 ppt (0.38 µg/kg wet weight) in liver and 1100 ppt (1.1 µg/kg wet weight) in fat. Pyrene was the most abundant PAH present (Obana et al., 1981b).

Samples of 24 bronchial carcinomas, taken during surgery or at autopsy from smokers and nonsmokers with a variety of occupations, were analysed for the presence of 12 PAH by thin-layer chromatography and fluorescence spectroscopy. Benzo[*a*]pyrene, benzo[*b*]fluoranthene, fluoranthene, and perylene were detected. Benzo[*a*]pyrene was present, but the other three PAH were found in only some of the samples. The average concentrations of benzo[*a*]pyrene were 3.5 µg/g in carcinoma tissue and 0.09 µg/g in tumour-free tissue (Tomingas et al., 1976).

6.6 Reactions with tissue components

The reactions of metabolites of PAH with tissue constituents (Weinstein et al., 1978) are relevant because they may indicate the mechanisms by which the hydrocarbons exert biological effects that include toxicity and carcinogenesis.

6.6.1 Reactions with proteins

Covalent interactions of PAH with protein in whole animals were first noted in 1951 (Miller, 1951). It was proposed that reactions with specific proteins might be involved in the initiation of malignancy in liver (Miller & Miller, 1953), skin (Abell & Heidelberger, 1962), and transformable cells in culture (Kuroki & Heidelberger, 1972). These findings were supported by evidence that hydrocarbon metabolites can react covalently with protein in microsomal incubates (Grover & Sims, 1968), in preparations of nuclei (Vaught & Bresnick, 1976; Pezzuto et al., 1976, 1977; Hemminki & Vainio, 1979), and in cells and tissues maintained in culture, including human tissues (Harris et al., 1978b; MacNicoll et al., 1980). Although hydrocarbon metabolites often react at much greater rates with protein than with nucleic acids in the same biological system, relatively little attention has been paid to the nature of the hydrocarbon metabolites involved or to the specificity of these reactions, in

terms of which proteins are most extensively modified and where and the effect that such modification might have on protein function. The evidence suggests, however, that the reactive species involved include diol epoxides. Thus, when protein isolated from the skin of mice that had been treated with benzo[*a*]pyrene was hydrolysed, tetrols were liberated, and the patterns of specific tetrols indicated that both *syn* and *anti* isomers of the benzo[*a*]pyrene 7,8-diol 9,10-oxides are involved in covalent reactions with protein (Koreeda et al., 1978). Studies of the covalent interactions of diol epoxides with nuclear proteins show that a variety of histones and non-histone proteins are modified (Kootstra & Slaga, 1979; Kootstra et al., 1979; Whitlock, 1979).

6.6.2 *Reactions with nucleic acids*

The covalent interactions of electrophilic metabolites of PAH with nucleic acids have been studied in much greater detail than those with protein, partly because characterization of the products might, in theory, be expected to be simpler, partly because the cellular nucleic acids are, as nucleophiles, more 'homogeneous' than proteins, but mainly because it has long been suspected that nucleic acid modifications could lead to a permanent alteration of cell phenotype.

The covalent binding of a PAH (dibenz[*a*,*h*]anthracene) to DNA *in vivo* was first reported by Heidelberger & Davenport in 1961. Subsequent studies with naphthalene, dibenz[*a*,*c*]anthracene, dibenz[*a*,*h*]anthracene, benzo[*a*]-pyrene, 3-methylcholanthrene, and 7,12-dimethylbenz[*a*]anthracene showed that the levels of DNA binding in mouse skin are correlated with carcinogenic potency, as measured by Iball's index (Brookes & Lawley, 1964).

6.7 Analytical methods

Of the methods used for the detection of carcinogen–DNA adducts (Phillips, 1990; Strickland et al., 1993; Weston, 1993), one of the most widely used is ^{32}P-postlabelling, in which DNA is hydrolysed to nucleotides, modified nucleotides (i.e. adducts) are labelled with ^{32}P-phosphate, and the post-labelled adducts separated by thin-layer chromatography and/or high-performance liquid chromatography (for reviews of the method, see Phillips, 1991, and Phillips et al., 1993). The main advantages of the ^{32}P-postlabelling assay are its high sensitivity and the fact that radiolabelled carcinogens and/or their metabolites need not be synthesized beforehand.

A variety of physical methods have been described for the detection of adducts, including fluorescence line narrowing spectroscopy, synchronous fluorescence spectroscopy, and some specialized gas chromatography-mass spectrometry procedures (Weston, 1993). The physical methods combine high sensitivity with no requirement for prior radiolabelling of the carcinogens or their adducts and may be nondestructive. Sensitive methods involving antisera

specific for carcinogen–DNA adducts have also been developed. These include radioimmunoassays, enzyme-linked immunosorbent assays, and immunoaffinity chromatography (Poirier, 1994).

Information on the pathways thought to be involved in the metabolic activation of several PAH is given in Table 74. For PAH that have been extensively investigated, reviews are cited. In order to provide an overall view of activation, the Table also includes data on PAH not covered elsewhere in this monograph.

Most of the metabolites that have been found to react with nucleic acids are vicinal diol epoxides, and most of these are diol epoxides of the 'bay-region' type, although there are certain exceptions (Table 74). For example, activation of benzo[*j*]fluoranthene in mouse skin involves a diol epoxide that is not of the bay-region type (Weyand et al., 1993). Additionally, methyl-substituted PAH may become bound to hydroxymethyl derivatives which, when conjugated, yield electrophilic sulfate esters (Surh et al., 1989, 1990a,b).

The sites of attack on nucleic acid bases are usually the extranuclear amino groups of guanine and adenine. When the reactions of the *syn* and *anti* isomers of benzo[*a*]pyrene 7,8-diol-9,10-oxide with RNA, DNA, and homopolymers were examined in experiments in which the epoxide was incubated with the nucleic acid in a predominantly aqueous solution, RNA, DNA, poly G, poly A, poly C, poly (dG), poly (dA), and poly (dC) were modified, but there was little reaction with poly U, poly I, or poly (dT) (Weinstein et al., 1976; Jennette et al., 1977). Although many of the hydrocarbon-deoxyribonucleoside adducts formed in human cells and tissues treated with PAH have not been completely characterized, the available evidence, which is mostly chromatographic, suggests that in human bronchial epithelium, colon, mammary cells in culture, and skin the patterns of adducts formed are very similar to those formed in corresponding rodent tissues (Autrup et al., 1978a,b; Harris et al., 1979; Autrup et al., 1980; MacNicoll et al., 1980; Weston et al., 1983). The rates of reaction of diol epoxides with nucleic acids was in the general order: poly G > DNA > poly A > poly C (Jennette et al., 1977).

Diol epoxides are also strongly suspected to react frequently with the N7 position of guanine. This type of modification has not been detected more often because N7-alkylated adducts are thought to have a relatively short half-life at pH 7 and would therefore be lost during the isolation and hydrolysis of DNA. In experiments in which care was taken to avoid adduct loss, reactions of benzo[*a*]pyrene diol epoxide with both the N2 and N7 positions of guanine residues in DNA were detected (Osborne et al., 1978). N7 adducts were not, however, detected in cells treated with *anti*-benzo[*a*]pyrene 7,8-diol-9,10-oxide (King et al., 1979).

In studies of the role of radical cations in the activation of PAH *in vitro*, adducts were formed in which the 6 position of benzo[*a*]pyrene was covalently linked to the C8 and N7 positions of guanine and the N7 position of adenine,

Table 74. Pathways involved in the metabolic activation of polycyclic aromatic hydrocarbons to form ultimate carcinogens

Compound	Derivatives with highest levels of biological activity	Putative ultimate carcinogen	Reference
Aceanthrylene		1,2-Oxide[a]	Nesnow et al. (1991)
Benz[*j*]aceanthrylene		? 1,2-Oxide[b]	Bartczak et al. (1987); Nesnow et al. (1988)
Benz[*l*]aceanthrylene		? 1,2-Oxide[b,c]	Nesnow et al. (1984); Bartzczak et al. (1987); Nesnow et al. (1988)
Benz[*a*]anthracene	3,4-Diol[d,e,f,g] 8,9-Diol[d]	3,4-Diol 1,2-oxide[a,b,c,f,g] 8,9-Diol 10,1-oxide[e,h]	Sims & Grover (1981); Conney (1982); Wood et al. (1983a)
Benzo[*b*]fluoranthene	9,10-Diol[d,f,i]	? 910-Diol-11,12-oxide and 5/6-hydroxy-9,10-diol-11,12-oxide	Geddie et al. (1987); Pfau et al. (1992)
Benzo[*j*]fluoranthene	? 9,10-Diol[f,j]	? 9,10-Diol 11,12-oxide[a]	Rice et al. (1987); Weyand et al. (1993)
	? 4,5-Diol[a]	? 4,5-Diol 6,6a-oxide[a]	Weyand et al. (1987)

Table 74 (contd)

Compound	Derivatives with highest levels of biological activity	Putative ultimate carcinogen	Reference
Benzo[*c*]phenanthrene	3,4-Diol[d,e,f,g]	3,4-Diol 1,2-oxide[a,b,c,f,g]	Conney (1982); Levin et al. (1986); Agarwal et al. (1987); Dipple et al. (1987); Pruess-Schwartz et al. (1987)
Benzo[*a*]pyrene	7,8-Diol[d,e,f,h]	7,8-Diol 9,10-oxide[a,b,c,g]	Cooper et al. (1983); Osborne & Crosby (1987a)
Benzo[*e*]pyrene	9,10-Diol[f]	? 9,10-Diol 11,12-oxide[g]	Osborne & Crosby (1987b)
Chrysene	1,2,-Diol[d,e,f] 9-Hydroxy 1,2-diol[d,e]	1,2-Diol 3,4-oxide[a,b,c,h] 9-Hydroxy-1,2-diol 3,4-oxide[b,c]	Conney (1982); Hodgson et al. (1983); Glatt et al. (1986)
Cyclopenta[*cd*]pyrene	–	? 3,4-oxide[b,c,h]	Gold & Eisenstadt (1980); Gold et al. (1980)
15,16-Dihydro-11-methylcyclopenta[*a*]phenanthren-17-one	3,4-Diol[d,f]	3,4-Diol 1,2-oxide[a]	Coombs & Bhatt (1987)

Table 74 (contd)

Compound	Derivatives with highest levels of biological activity	Putative ultimate carcinogen	Reference
15,16-Dihydro-1,11-methano-cyclopenta[*a*]phenanthren-17-one	3,4-Diol[a]	3,4-Diol 1,2-oxide	Coombs & Bhatt (1987)
Dibenz[*a,c*]anthracene	10,11-Diol[d]	? 10,11-Diol 12,13-oxide	Sims & Grover (1981)
Dibenz[*a,h*]anthracene	3,4-Diol[d,f,g,h]	? 3,4-Diol 1,2-oxide and 3,4:10,11-bis-diol-epoxides	Conney (1982); Lecoq et al. (1991, 1992); Carmichael et al. (1993); Nesnow et al. (1994)
Dibenzo[*a,e*]fluoranthene (1983,1984);	12,13-Diol[d,i]	12,13-Diol 10,11-oxide[a]	Perin-Roussel et al.
	3,4-Diol[d,i]	3,4-Diol 1,2-oxide[a]	Saguem et al. (1983a,b); Zajdela et al. (1987)
Dibenzo[*a,h*)]pyrene	1,2-Diol[f,g]	? 1,2-Diol 3,4-oxide[g]	Chang et al. (1982)
Dibenzo[*a,l*]pyrene	? 11,12 Diol[f]	? 11,12-Diol 13,14-oxide	Cavalieri et al. (1991)
Dibenzo[*a,l*]pyrene	3,4-Diol[f,g]	? 3,4-Diol 1,2-oxide[g]	Chang et al. (1982)

Table 74 (contd)

Compound	Derivatives with highest levels of biological activity	Putative ultimate carcinogen	Reference
7,12-Dimethylbenz[*a*]anthracene	3,4-Diol[d,e,f,h]	3,4-Diol 1,2-oxide[a]	Sims & Grover (1981); Conney (1982); Sawicki et al. (1983); Dipple et al.; 1984)
7-Ethylbenz[*a*]anthracene	3,4-Diol[d]	? 3,4-Diol 1,2-oxide[a,b]	McKay et al. (1988); Glatt et al. (1989)
Fluoranthene	2,3-Diol[d]	2,3-Diol 1,10b-oxide[a]	La Voie et al. (1982a); Rastetter et al. (1982); Babson et al. (1986a); Hecht et al. (1995)
Indeno[1,2,3-*cd*]pyrene	1,2-oxide[b,f] 1,2-Diol[f] 8-Hydroxy[d] 9-Hydroxy[d]	?	Rice et al. (1985) Rice et al. (1986)
7-Methylbenz[*a*]anthracene	3,4-Diol[d,e,f,h]	3,4-Diol 1,2-oxide[a,b]	Sims & Grover (1981); McKay et al. (1988); Glatt et al. (1989)

Table 74 (contd)

Compound	Derivatives with highest levels of biological activity	Putative ultimate carcinogen	Reference
3-Methylcholanthrene	9,10-Diol[d,f,h] diol 7,8-oxide	? 9,10-Diol 7,8-oxide[a,i] ? 3-Hydroxymethyl-9,10-DiGiovanni et al. (1985);	Sims & Grover (1981); Conney (1982); Osborne et al. (1986)
5-Methylchrysene	1,2-Diol[d,l]	1,2-Diol 3,4-oxide[a,c,h]	Hecht et al. (1986); Brookes et al. (1986); Reardon et al. (1987); Hecht et al. (1987)

[a] DNA adducts characterized
[b] Directly acting mutagen in *S. typhimurium*
[c] Directly acting mutagen in V79 Chinese hamster cells
[d] Mutagenic to *S. typhimurium* with metabolic activation
[e] Mutagenic to V79 Chinese hamster cells with metabolic activation
[f] Tumour initiator in mouse skin
[g] Induces tumours in newborn mice
[h] Transforms cells in culture
[i] Not detected as a metabolite; activation may therefore occur via a different pathway.
[l] Although the 4,5-diol is the most active derivative so far tested, there is some evidence that adducts arise from the 9,1-diol.

and the 7-methyl position of 7,12-dimethylbenz[*a*]anthracene was covalently linked to the N7 positions of guanine and adenine (see Figure 7; Cavalieri et al., 1993; Rogan et al., 1993). All of these adducts are depurination adducts, which may explain why they were not detected earlier *in vivo*. The formation of apurinic sites in DNA could lead to strand nicking (Gamper et al., 1977, 1980). When the positions of the nicks produced as a result of modification by benzo[*a*]pyrene 7,8-diol-9,10-oxide were investigated with DNA of a defined sequence, nicking appeared to be the result of the loss of purines and pyrimidines that had been modified at the N7 position of guanine or at the N3 position of adenine and cytosine (Haseltine et al., 1980).

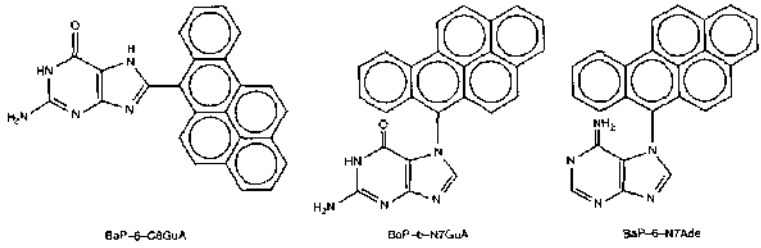
In studies of the distribution of covalently bound benzo[*a*]pyrene moieties in chromatin, more was bound to the inter-nucleosomal spacer regions of DNA than to DNA in nucleosomes (Jahn & Litman, 1977, 1979; Kootstra & Slaga, 1980). One explanation for this finding may be that nucleosomal DNA is better protected from modification by the presence of nucleoproteins; results consistent with this suggestion have been obtained with mitochondrial DNA. Graffi (1940a,b,c) suggested that lipophilic PAH accumulate in lipid-rich mitochondria. Allen & Coombs (1980) and Backer & Weinstein (1980) showed much higher levels of modification of mitochondrial than nuclear DNA in cultured cells treated with either benzo[*a*]pyrene or the *anti*-benzo[*a*]pyrene 7,8-diol-9,10-oxide.

The molecular properties of adducts of benzo[*a*]pyrene 7,8-dihydrodiol-9,10-epoxides with DNA have been described (Geacintov 1988; Jernström & Gräslund, 1994). Although the biological effectiveness of all types of hydrocarbon-nucleic acid adducts has not been determined, it has been shown that differences in the biological activities of 7-ethyl- and 7-methylbenz[*a*]-anthracene are not due to differences in the mutagenic potential of the adducts formed (Glatt et al., 1989). Similar conclusions were drawn from work with a series of bay-region and fjord-region diol epoxides (Phillips et al., 1991; see section 7.10 for a description of a fjord region). At present, therefore, all hydrocarbon-deoxyribonucleoside adducts should be regarded as potentially damaging to the organism.

The relationships between DNA adduct formation and tumour incidence were examined by Poirier & Beland (1992) on the basis of data from long-term studies in rodents administered carcinogens. The tumour incidence was compared with adduct levels measured in target tissues during the first two months of exposure. In most cases, linear increases in DNA adduct levels with dose were reflected in linear increases in tumour incidence, although there were exceptions.

In a comparison of the incidence of lung adenomas in strain A/J mice 240 days after they had received a single intraperitoneal injection of benzo[*a*]pyrene, dibenz[*a,h*]anthracene, benzo[*b*]fluoranthene, 5-methyl-chrysene, or cyclopenta[*cd*]pyrene with the levels of DNA adducts detected in

Figure 7. Benzo[a]pyrene adducts in which the 6 position is covalently linked to the C8 and N7 positions of guanine and the N7 position of adenine



the lungs by ^{32}P-postlabelling between days 1 and 21 after treatment, time-integrated DNA adduct levels were calculated and plotted against lung adenoma frequency. The slopes obtained were essentially similar for benzo[*a*]pyrene, benzo[*b*]fluoranthene, 5-methylchrysene, and cyclopenta[*cd*]pyrene but were different for dibenz[*a,h*]anthracene. The authors concluded that 'essentially identical induction of adenomas as a function of [time-integrated DNA adduct levels] for these PAH suggests that the formation and persistence of DNA adducts determines their carcinogenic potency' (Ross et al., 1995).

7. EFFECTS ON LABORATORY MAMMALS AND *IN VITRO*

Appraisal

*Single doses of polycyclic aromatic hydrocarbons (PAH) have moderate to low toxicity, with LD_{50} values generally > 100 mg/kg bw after intraperitoneal or intravenous injection and > 500 mg/kg bw after oral administration. Because most of the experimental studies have addressed the carcinogenicity of PAH, the database on their short- and long-term toxicity is quite small. In short-term studies, effects on the haematopoietic system were observed, e.g. benzo[*a*]pyrene caused myelotoxicity and dibenz[*a,h*]anthracene caused haemolymphatic alterations in mice. Anaemia is a typical effect of naphthalene. Values for a no-observed-adverse-effect level (NOAEL) and a lowest-observed-adverse-effect level (LOAEL) have been obtained in 90-day studies by oral administration. The NOAEL values based on haematological effects and hepato- and nephrotoxicity were 75–1000 mg/kg bw per day for the noncarcinogenic PAH acenaphthene, anthracene, fluoranthene, fluorene, and pyrene.*

*Few studies have been conducted on dermal or ocular irritation. PAH do, however, have adverse effects after dermal administration, such as hyperkeratosis, which are correlated with their carcinogenic potency. Anthracene and naphthalene were reported to cause mild ocular irritation. The ocular toxicity of naphthalene is characterized by cataract formation. Benzo[*a*]pyrene caused skin hypersensitization. Anthracene and benzo[a]pyrene have been shown to have phototoxic potential and benzo[*a*]pyrene, dibenz[a,h]anthracene, and fluoranthene to have immunotoxic potential.*

*PAH can cross the placenta and induce adverse effects on the embryo and fetus. Benz[*a*]anthracene, benzo[*a*]pyrene, dibenz[*a,h*]anthracene, and naphthalene were found to be embryotoxic. Benzo[a]pyrene also reduced female fertility and had effects on oocytes and on postnatal development. Studies on the effects of benzo[*a*]pyrene in mice with different genotypes demonstrated the importance of the genetic predisposition of animals or embryos for the development of overt toxic effects. A crucial genetic property is the presence or absence of the arylhydrocarbon (Ah) receptor, which induces the monooxygenase system; organisms can thus be divided into Ah responders and Ah non-responders.*

Mutagenicity has been investigated intensively in a broad range of assays. The only compounds that are clearly not mutagenic are naphthalene, fluorene, and anthracene. The evidence for five PAH is considered to be questionable because of a limited database, while the remaining 25 PAH are mutagenic (see Table 87). Mutagenicity is strictly dependent on metabolic activation of parent compounds. In bacteria and other cell systems that have no metabolizing

system, a 9000 x g microsomal preparation of liver (S9 mix) must be added as a metabolic activator.

*Comprehensive work on the carcinogenicity of these compounds has yielded negative results for fluorene, anthracene, 1-methylphenanthrene, pyrene, triphenylene, perylene, anthanthrene, and benzo[*ghi*]perylene, some of which have been shown to be mutagenic. The evidence for a further eight PAH was classified as questionable, while the other 17 compounds were carcinogenic. Generally, the site of tumour development depends on the route of administration but is not restricted to those sites. Tissues such as the skin can metabolize PAH to their ultimate metabolites, thus becoming target organs themselves, and metabolites formed in the liver can reach various sites of the body via the bloodstream. The carcinogenic potency of PAH differs by three orders of magnitude, and toxic equivalence factors have been used to rank individual PAH (see Appendix I).*

The various theories for the mechanism of the carcinogenicity of PAH take into account chemical structure and ionization potential. The most prevalent theories are those involving the bay region and radical cations. The bay-region theory is based on the assumption that diol epoxides of the parent compounds are the ultimate carcinogens, which react with electrophilic epoxide groups on N atoms of DNA purines. The radical cation theory postulates the one-electron oxidation of PAH to form strong electrophiles which then react with DNA bases. These theories have been confirmed experimentally by detection of the corresponding DNA adducts in the PAH that have been investigated. Nevertheless, there is general agreement that any one theory cannot cover the mechanisms of action of all PAH.

7.1 Toxicity after a single exposure

Few studies are available on the acute toxicity of PAH, except for naphthalene. The LD_{50} values (Table 75) indicate that the acute toxicity is moderate to low. The results of all of these studies are summarized, even when a study was old and followed a non-systematic protocol, in the absence of alternatives.

7.1.1 Benzo[a]pyrene

In young rats, a single intraperitoneal injection of 10 mg benzo[*a*]pyrene per animal caused an immediate, sustained reduction in the growth rate (Haddow et al., 1937). In mice, a single intraperitoneal injection (dose not specified) resulted in small spleens, marked cellular depletion, prominent haemosiderosis, and follicles with large lymphocytes, leading to death (Shubik & Della Porta, 1957). After a single application of 0.05 ml of a 1% solution in acetone to the interscapular area of hairless mice (hr/hr strain), the mitotic rate of epidermal cells was increased (Elgjo, 1968).

Table 75. Toxicity of single doses of polycyclic aromatic hydrocarbons

Compound	Species	Route of administration	LD_{50} (mg/kg) or LC_{50} (mg/litre)	Reference
Anthracene	Mouse	Oral	18 000	Montizaan et al. (1989)
	Mouse	Intraperitoneal	> 430	Salamone (1981)
Benzo[*a*]pyrene	Mouse	Oral	> 1 600	Awogi & Sato (1989)
	Mouse	Intraperitoneal	~ 250	Salamone (1981)
	Mouse	Intraperitoneal	> 1 600	Awogi & Sato (1989)
	Rat	Subcutaneous	50	Montizaan et al (1989)
Chrysene	Mouse	Intraperitoneal	> 320	Simmon et al. (1979)
Fluoranthene	Rat	Oral	2 000	Smyth et al. (1962)
	Rabbit	Dermal	3 180	Smyth et al. (1962)
	Mouse	Intravenous	100	Montizaan et al. (1989)
Naphthalene	Rat	Oral	1 250	Sax & Lewis (1984)
	Rat (M)	Oral	2 200	Gaines (1969)
	Rat (F)	Oral	2 400	Gaines (1969)
	Rat	Oral	9 430	US Environmental Protection Agency (1978a)
	Rat	Oral	1 110	Montizaan et al. (1989)
	Rat	Oral	490	Montizaan et al. (1989)
	Rat	Oral	1 800	Montizaan et al. (1989)
	Rat (M)	Dermal	> 2 500	Gaines (1969)
	Rat (F)	Dermal	> 2 500	Gaines (1969)
	Rat	Intraperitoneal	~ 1 000	Bolonova (1967)

Table 75 (contd)

Compound	Species	Route of administration	LD_{50} (mg/kg) or LC_{50} (mg/litre)	Reference
Naphthalene (contd)	Rat (M)	Intraperitoneal	~ 1 600	Plopper et al. (1992)
	Rat	Inhalation	> 0.5 mg/litre (8 h)	US Environmental Protection Agency (1978a)
	Mouse (F)	Oral	354	Plasterer et al. (1985)
	Mouse (M)	Oral	533	Shopp et al. (1984)
	Mouse (F)	Oral	710	Shopp et al. (1984)
	Mouse	Subcutaneous	5 100	Sandmeyer (1981); Shopp et al. (1984)
	Mouse	Subcutaneous	969	Sax & Lewis (1984)
	Mouse	Intraperitoneal	150	Sax & Lewis (1984)
	Mouse	Intraperitoneal	380	Warren et al. (1982)
	Mouse (M)	Intraperitoneal	~ 400	Plopper et al. (1992)
	Mouse	Intravenous	100	Sax & Lewis (1984)
	Hamster (M)	Intraperitoneal	~ 800	Plopper et al. (1992)
	Guinea-pig	Oral	1 200	Sax & Lewis (1984)
Phenanthrene	Mouse	Oral	700	Montizaan et al. (1989)
	Mouse	Oral	1 000	Montizaan et al. (1989)
	Mouse	Intraperitoneal	700	Simmon et al. (1979)
	Mouse	Intravenous	56	Montizaan et al. (1989)
Pyrene	Mouse	Intraperitoneal	514 (7 d)	Salamone (1981)
	Mouse	Intraperitoneal	678 (4 d)	Salamone (1981)

LC_{50}, median lethal concentration; LD_{50}, median lethal dose; M, male; F, female

7.1.2 *Chrysene*

In young rats, single intraperitoneal injections of 30 mg chrysene per animal did not reduce growth (Haddow et al., 1937).

7.1.3 *Dibenz[a,h]anthracene*

One or two intraperitoneal injections of 3–90 mg dibenz[*a*,*h*]anthracene per animal within two days led to a reduction in the growth rate of young rats that persisted for at least 15 weeks (Haddow et al., 1937).

7.1.4 *Fluoranthene*

In young rats, a single intraperitoneal injection of 30 mg fluoranthene per animal did not inhibit growth (Haddow et al., 1937).

7.1.5 *Naphthalene*

After oral administration of 1–4 g/kg bw naphthalene to dogs or 1–3 g/kg bw to cats, diarrhoea was observed. Rabbits given 1–3 g/kg bw showed corneal clouding (Flury & Zernik, 1935). After intravenous injection of 1–6 mg napthalene to white male rabbits weighing 3–4 kg, no haemolytic effect was seen (Mackell et al., 1951)

In mice, Clara cells of the bronchiolar epithelium are the primary targets of low doses of naphthalene. Dose-dependent bronchiolar epithelial cell necrosis was detected after intraperitoneal injection of a single dose of 50, 100, or 200 mg/kg bw per day to mice (O'Brien et al., 1989). Severe bronchiolar epithelial cell necrosis was also seen in mice within 2-4 h after intraperitoneal injection of 200–375 mg/kg bw; hepatic and renal necrosis were not observed (Warren et al., 1982). Alterations in the morphology of Clara cells were observed as early as 6 h after intraperitoneal injection of 64 mg/kg bw; ciliated cells were also affected after 24 and 48 h and at doses up to 256 mg/kg bw. After a 4-h inhalation of 1.0 mg/litre naphthalene, bronchiolar necrosis was detected in mice but not in rats (Buckpitt & Franklin, 1989; see also section 7.2.1).

After single injections of 50–400 mg/kg bw to mice, 100–800 mg/kg bw to hamsters, and 200–1600 mg/kg bw to rats, Clara cells in mice showed the effects described above; those of rats showed no significant effects, and minor effects were observed in hamsters. The trachea and lobar bronchi showed swelling and vacuolation of non-ciliated cells in mice, no effects in rats, and cytotoxic changes in hamsters. In the nasal cavity, cytotoxicity to the olfactory epithelium with necrosis was observed in mice and hamsters at 400 mg/kg bw and in rats at 200 mg/kg bw (Plopper et al., 1992).

Mice injected intraperitoneally with 200–600 mg/kg bw naphthalene showed dose-dependent abnormalities in the bronchial region (Clara cells) in

studies in which the lungs were examined by scanning electron micrography. No pulmonary damage was detected at 100 mg/kg bw. Depletion of pulmonary glutathione, which protects against the toxicity of xenobiotics, was observed within 6 h of naphthalene administration (Honda et al., 1990).

The doses and detailed findings of experiments with single doses of naphthalene are summarized in Table 76.

7.1.6 Phenanthrene

After acute intraperitoneal injection to rats (dose not specified), liver congestion with a distinct lobular pattern was observed as well as alterations in some serum parameters (Yoshikawa et al., 1987).

7.1.7 Pyrene

In young rats, single intraperitoneal injections of 10 mg pyrene per animal did not lead to a reduction in growth rate (Haddow et al., 1937).

7.2 Short-term toxicity

7.2.1 Subacute toxicity

7.2.1.1 Acenaphthene

Four of five mice given 500 mg/kg bw per day acenaphthene intraperitoneally for seven days survived (Gerarde, 1960).

7.2.1.2 Acenaphthylene

Nine of 10 mice given 500 mg/kg bw per day acenaphthylene for seven days survived (Gerarde, 1960).

7.2.1.3 Anthracene

Nine of 10 mice given 500 mg/kg bw per day anthracene for seven days survived (Gerarde, 1960). Oral administration of 100 mg/kg bw per day to rats for four days increased carboxylesterase activity in the intestinal mucosa by 13% (Nousiainen et al., 1984).

7.2.1.4 Benzo[a]pyrene

Death due to myelotoxicity was observed after daily oral administration of benzo[*a*]pyrene at 120 mg/kg bw to poor-affinity Ah receptor DBA/2N mice for one to four weeks, whereas high-affinity C57 Bl/6N mice survived with no

Table 76. Toxicity of single doses of naphthalene

Species (strain)	Sex (no./sex per group)	Route of administration	Dose (purity)	Effects	Reference
Dog		Oral	1000–2000, 4000 or 5000 mg/dog	1000–2000: Light diarrhoea; 4000 mg: lethal; 5000 mg: heavy diarrhoea	Flury & Zernick (1935)
Cat		Oral	1000–3000 mg/kg bw	Lethal	Flury & Zernick (1935)
Rabbit		Oral	1000–3000 and 3000 mg/kg bw	1000–3000 mg: corneal clouding; 3000 mg: death after 24 h	Flury & Zernick (1935)
Dog	(1)	Oral	400 and 1600 mg/kg bw	400 mg: weakness, severe anaemia; 1600 mg: weakness, vomiting, diarrhoea, slight anaemia; complete recovery within 1–2 weeks	Zuelzer & Apt (1949)
Mouse		Inhalation	0.1 mg/litre, 4 h	Bronchiolar necrosis	Buckpitt & Franklin (1989)
Mouse (Swiss-Webster	M	Intraperitoneal	50, 100, 200, 300 mg/kg bw	Dose-dependent bronchiolar epithelial-cell necrosis	O'Brien et al. (1989)

Table 76 (contd)

Species (strain)	Sex (no./sex per group)	Route of administration	Dose (purity)	Effects	Reference
Mouse (Swiss-Webster)	M (4–35)	Intraperitoneal	50, 100, 200, 300, and 400 mg/kg bw (> 99.9%)	Dose-dependent bronchiolar necrosis; 300 mg/kg: swollen cells in trachea 400 mg/kg: cytotoxicity in olfactory epithelium	Plopper et al. (1992)
Rat (Sprague-Dawley)	M (4–11)	Intraperitoneal	200, 400, 800, and 1600 mg/kg bw (> 99.9%)	Bronchiolar necrosis not observed; no changes in trachea; 200 mg/kg: complete necrosis of olfactory epithelium	Plopper et al. (1992)
Rat (Wistar)	M	Intraperitoneal	400–1600 mg/kg bw	No damage to lungs, liver, or kidneys	O'Brien et al. (1985)
Hamster (Syrian golden)	M (4–6)	Intraperitoneal	100, 200, 400 and 800 mg/kg bw (99.9%)	800 mg/kg: minor alterations in terminal bronchioles; cytotoxic changes in trachea; 400 mg/kg: necrosis of olfactory epithelium	Plopper et al. (1992)
Rabbit (white)	M	Intraperitoneal	0.3–1.7 mg/kg bw	No haemolytic effects	Mackell et al. (1951)

M, male

myelotoxicity for at least six months under these conditions (Legraverend et al., 1983).

Rats given 50 or 150 mg/kg bw per day of benzo[*a*]pyrene orally for four days showed suppressed carboxylesterase activity in the intestinal mucosa. The NOAEL with respect to gastric, hepatic, and renal effects was 150 mg/kg bw per day (Nousiainen et al., 1984)

In Fischer 344/Crl rats exposed by inhalation to 7.7 mg/m^3 of benzo[*a*]pyrene dust for 2 h/day, five days per week for four weeks, no respiratory tract lesions were observed, as measured by lung lavage, clearance of tagged particles, and histopathological findings (Wolff, R.K. et al., 1989).

7.2.1.5 *Benz[a]anthracene*

When benz[*a*]anthracene was given orally to rats daily for four days, the NOAEL with respect to gastric, hepatic, and renal effects was 150 mg/kg bw per day. Carboxylesterase activity in the intestinal mucosa was suppressed (Nousiainen et al., 1984).

7.2.1.6 *Dibenz[a,h]anthracene*

Adverse haemolymphatic changes, including the appearance of extravascular erythrocytes in the lymph spaces and large pigmented cells, were reported after subcutaneous injection of male rats with 0.28 mg per animal on five days per week for four weeks (Lasnitzki & Woodhouse, 1944).

7.2.1.7 *Fluoranthene*

All of 10 mice that received 500 mg/kg bw per day fluoranthene intraperitoneally for seven days survived (Gerarde, 1960).

7.2.1.8 *Naphthalene*

Anaemia was induced in three dogs by single oral doses of 3 or 9 g or a total dose of 10.5 g per animal given over seven days. All three animals showed neurophysiological symptoms and slight to very severe changes in haematological parameters. Full recovery was observed within 7–14 days (Zuelzer & Apt, 1949).

No immunosuppressive effects were observed in a number of test systems. Tolerance to the effects of naphthalene was reported in mice after intraperitoneal injection for seven days. A sharp contrast between single and multiple doses was observed in the effects on the morphology of the bronchiolar epithelium. When naphthalene was given intraperitoneally at a dose of 50, 100, or 200 mg/kg bw per day as a single injection, dose-dependent bronchiolar epithelial cell necrosis was detected; however, when these doses were given

daily for seven days, no significant effects were observed. Addition of 300 mg/kg bw on day 8 had no effect, whereas recovered sensitivity was observed with increasing time between the last dose and the challenge dose. A single dose of 300 mg/kg bw without pretreatment resulted in substantial denudation of the bronchiolar epithelium. This pattern was attributed to a reduction in metabolic activation of naphthalene due to a decrease in cytochrome P450 mono-oxygenase activity after multiple dosing. A rough correlation was observed in mouse lung (but not liver microsomes) between induction of tolerance and decreased metabolic formation of the 1*R*, 2*S*-epoxide enantiomer, which is responsible for tissue-selective toxicity. Such toxicity was demonstrated in mice both *in vivo* and in isolated perfused lung (Buckpitt & Franklin, 1989). These studies are summarized in Table 77.

7.2.1.9 Phenanthrene

Oral administration of 100 mg/kg bw per day phenanthrene to rats for four days induced a 30% increase in carboxylesterase activity in the intestinal mucosa (Nousiainen et al., 1984).

7.2.1.10 Pyrene

Four of five mice injected intraperitoneally with 500 mg/kg bw per day pyrene for seven days survived (Gerarde, 1960).

7.2.2 Subchronic toxicity

7.2.2.1 Acenaphthene

Administration of 175 mg/kg bw per day acenaphthene to mice by gavage for 90 days resulted in a NOAEL of 175 mg/kg bw per day and a LOAEL of 350 mg/kg bw per day for hepatotoxicity (US Environmental Protection Agency, 1989a).

7.2.2.2 Anthracene

Four of five rats given 5 mg per animal anthracene subcutaneously for four months survived (Gerarde, 1960).

Anthracene was administered to groups of 20 male and female CD-1 (ICR) BR mice by gavage at a dose of 0, 250, 500, or 1000 mg/kg bw per day for at least 90 days. No treatment-related effects were noted on mortality, clinical signs, body weights, food consumption, ophthalmological findings, the results of haematology and clinical chemistry, organ weights, organ-to-body weight ratios, and gross pathological and histopathological findings. The no-observed-effect level (NOEL) was the highest dose tested, 1000 mg/kg bw per day (US Environmental Protection Agency, 1989b).

Table 77. Subacute and subchronic effects of naphthalene

Species (strain)	Sex (no./sex per group)	Route of administration	Dose (purity)	Effects	Reference
Mouse (CD-1)	M,F (40–112)	Oral	27, 53, and 267 mg/kg bw, 7 d/ week, 14 d	In all groups, slight alterations in haematological parameters; humoral immune response not affected. 27 and 53 mg/kg: no significant effects; 267 mg/kg: 5–10% mortality (m/f); significantly decreased terminal body weight (m/f); 30% decrease in thymus weight (m); significant decrease in weight of spleen (f); increase in lung weight (f)	Shopp et al. (1984)
Mouse (CD-1)	M,F	Oral	5.3, 53, and 133 mg/kg bw, 7 d/ week, 90 d	No obvious pulmonary effects or immunotoxicity; significantly decreased relative spleen weights (f); tolerance	Shopp et al. (1984)
Mouse (Swiss-Webster)	M	Intraperitoneal	50, 100, and 200 mg/kg, 7 d	No significant alterations in lung morphology; tolerance to 300 mg/kg on day 8	Buckpitt & Franklin (1989); O'Brien et al. (1989)
Rat		Diet	2 g/kg diet, 100 d	Inhibition of growth; enlarged, fatty livers	White & White (1939)
Dog	(1)	Oral	0.22 g/kg bw per day, 7 d	Diarrhoea, weakness, lack of appetite, ataxia, very severe anaemia; complete recovery within 1–2 weeks	Zuelzer & Apt (1949)

M, male; F, female

7.2.2.3 *Benzo[a]pyrene*

Male Syrian golden hamsters were exposed by inhalation to 9.8 or 44.8 mg/m³ benzo[*a*]pyrene for 4.5 h/day, five days per week for 16 weeks. No neoplastic response was observed in the respiratory tract (Thyssen et al., 1980).

The growth of rats was inhibited by feeding a diet enriched with benzo[*a*]pyrene at 1.1 g/kg for more than 100 days (White & White, 1939).

7.2.2.4 *Fluorene*

Groups of 25 male and 25 female CD-1 mice were given 0, 125, 250, or 500 mg/kg bw per day fluorene suspended in corn oil by gavage for 13 weeks. Increased salivation, hypoactivity, and abdomens wetted with urine were observed in all treated males. The percentage of hypoactive mice was dose-related. In mice exposed at 500 mg/kg bw per day, laboured respiration, ptosis (drooping eyelids), and an unkempt appearance were also observed. A significant decrease in erythrocyte count and packed cell volume were observed in females treated with 250 mg/kg bw per day fluorene and in males and females treated with 500 mg/kg bw per day. The latter also showed a decreased haemoglobin concentration and an increased total serum bilirubin level. A dose-related increase in relative liver weight was observed in treated mice, and a significant increase in absolute liver weight was observed in the mice treated with 250 or 500 mg/kg bw per day. Significant increases in absolute and relative spleen and kidney weights were observed in males and females exposed to 500 mg/kg bw per day and in males at 250 mg/kg bw per day. The increases in absolute and relative liver and spleen weights in animals at the high dose were accompanied by increases in the amounts of haemosiderin in the spleen and in Kupffer cells of the liver. No other histopathological lesions were observed. The LOAEL for haematological effects was 250 mg/kg bw per day, and the NOAEL was 125 mg/kg bw per day (US Environmental Protection Agency, 1989c).

In a similar study, fluorene at 35, 50, and 150 mg/kg bw increased the weight of the liver by about 20% in a dose-dependent fashion and the mitotic index of hepatocytes by sixfold after 48 h (Danz et al., 1991).

7.2.2.5 *Fluoranthene*

Groups of 20 male and 20 female CD-1 mice were given 0, 125, 250, or 500 mg/kg bw per day fluoranthene by gavage for 13 weeks. A fifth group of 30 male and 30 female mice was used to establish baseline levels in blood. Body weight, food consumption, and haematological and serum parameters were recorded regularly throughout the experiment. At the end of 13 weeks, the animals were killed and autopsied; organs were weighed and a histological evaluation was made. All treated mice had dose-dependent nephropathy, increased salivation, and increased liver enzyme activities, but these effects

were either not significant, not dose-related, or not considered adverse at 125 mg/kg bw per day. Mice exposed to 500 mg/kg bw per day had increased food consumption and increased body weight. Mice exposed to the two higher doses had statistically increased alanine aminotransferase activity and increased absolute and relative liver weights. Treatment-related microscopic liver lesions (indicated by pigmentation) were observed in 65% of mice at 250 mg/kg bw per day and 88% of those at the highest dose. On the basis of the increased alanine aminotransferase activity, pathological effects in the kidney and liver, and clinical and haematological changes, the LOAEL was 250 mg/kg bw per day and the NOAEL 125 mg/kg bw per day (US Environmental Protection Agency, 1988).

7.2.2.6 *Naphthalene*

In a 90-day study in mice, naphthalene at oral doses up to 133 mg/kg bw caused neither mortality nor serious changes in organ weights (Shopp et al., 1984). These authors did not observe haemolytic anaemia in CD-1 mice after oral uptake, although this effect had been seen in human patients (Konar et al., 1939; Zuelzer & Apt, 1949; see Section 8). It was suggested that glucose-6-phosphate dehydrogenase deficiency in erythrocytes, a prerequisite of haemolytic anaemia in adult humans, was not present in the mice (Shopp et al., 1984).

In rats that ingested 150 mg/kg bw per day naphthalene for the first three weeks and 200-220 mg/kg bw per day for a further 11 weeks, reduced weight gain and food intake were observed. Later, the liver was found to be enlarged, with cell oedema and congestion of the liver parenchyma, and the kidneys showed signs of inflammation (Kawai, 1979).

The presence of 1 g/kg naphthalene in the feed of rats and rabbits for 46–60 days led to cataracts (US Environmental Protection Agency, 1984b; see also section 7.8).

Administration to rabbits of 0.1–1 mg/kg bw per day naphthalene by subcutaneous injection for 123 days resulted in severe oedema and a high degree of vacuolar and collicular degeneration in the brain; necrosis of nerve cells also occurred. The author suggested that hypoxaemia resulting from haemolytic anaemia was responsible for this finding (Suja, 1967; cited by Kawai, 1979).

Subacute and subchronic studies with naphthalene are summarized in Table 77.

7.2.2.7 *Pyrene*

The growth of rats was inhibited by feeding a diet enriched with benzo[*a*]pyrene at 2 g/kg for more than 100 days. The livers were enlarged and had a fatty appearance indicating hepatic injury (White & White, 1939).

Groups of 20 male and 20 female CD-1 mice were given 0, 75, 125, or 250 mg/kg bw per day pyrene in corn oil by gavage for 13 weeks and then examined for changes in body weight, food consumption, mortality, clinical pathological manifestations in major organs and tissues, and changes in haematology and serum chemistry. Nephropathy, characterized by the presence of multiple foci of renal tubular regeneration, often accompanied by interstitial lymphocytic infiltrates and/or foci of interstitial fibrosis, was present in four male control mice, one at the low dose, one at the medium dose, and nine the high dose. Similar lesions were seen in two, three, seven, and 10 female mice, respectively. The renal lesions in all groups were described as minimal or mild. Relative and absolute kidney weights were reduced in mice at the two higher doses. On the basis of nephropathy and decreased kidney weights, the low dose (75 mg/kg bw per day) was considered to be the NOAEL and 125 mg/kg bw per day the LOAEL (US Environmental Protection Agency, 1989d).

7.3 Long-term toxicity

Almost all of the long-term studies reported were designed to assess the carcinogenic potency of PAH and are therefore summarized in section 7.7. Information about the non-carcinogenic effects, such as growth inhibition, liver damage, and irritation, which occurred at concentrations that also caused carcinogenic effects is presented here. General effects, such as on mortality, body weight, and pathological findings at sacrifice, were not considered useful.

7.3.1 Anthracene

A group of 28 BD I and BD III rats received anthracene in the diet from the age of about 100 days, at a daily dose of 5–15 mg per rat. The experiment was terminated when a total dose of 4.5 g per rat had been achieved, on day 550. The rats were observed until they died; some lived for more than 1000 days. No treatment-related effects on lifespan or on gross or histological appearance of tissues were observed; haematological parameters were not measured (Schmähl, 1955).

After weekly subcutaneous injections of anthracene at 0.25 mg per animal for 40 weeks, mice showed deposition of iron in lymph glands and a reduced number of lymphoid cells (Hoch-Ligeti, 1941).

7.3.2 Benz[a]anthracene

Weekly subcutaneous injection of 0.25 mg per mouse for 40 weeks resulted in deposition of iron in lymph glands and a reduced number of lymphoid cells (Hoch-Ligeti, 1941).

7.3.3 *Dibenz[a,h]anthracene*

Mice given weekly subcutaneous injections of 0.25 mg per animal for 40 weeks had pale, soft, enlarged livers with signs of fatty degeneration. There was deposition of iron in lymph glands, and the number of lymphoid cells was reduced (Hoch-Ligeti, 1941).

7.4 Dermal and ocular irritation and dermal sensitization

The adverse dermatological effects observed in animals after acute and subchronic dermal exposure to PAH included destruction of sebaceous glands, dermal ulceration, hyperplasia, hyperkeratosis, and alterations in epidermal cell growth. Perylene, benzo[*e*]pyrene, phenanthrene, pyrene, anthracene, naphthalene, acenaphthalene, fluorene, and fluoranthene did not suppress the sebaceous gland index; benz[*a*]anthracene, dibenz[*a,h*]anthracene, and benzo[*a*]pyrene resulted in indices > 1 (Bock & Mund, 1958). In Swiss mice treated daily for three days with solutions of benzo[*a*]pyrene in acetone, a concentration of 0.1% destroyed less than half of the sebaceous glands, whereas 0.2% destroyed more than 50% (Suntzeff et al., 1955).

7.4.1 *Anthracene*

Anthracene is a primary irritant, and its fumes can cause mild irritation of the skin, eyes, mucous membranes, and respiratory tract. At a concentration of 4.7 mg/m^3, mild skin irritation was found in 50% of exposed mice (Montizaan et al., 1989). The median value for dermal irritant activity (ID_{50}) in the mouse ear was 6.6×10^{-4} mmol or 118 µg/ear; in comparison, the ID_{50} for benzo[*a*]pyrene was 5.6×10^{-5} mmol per ear (Brune et al., 1978). Anthracene increases the sensitivity of skin to solar radiation (Gerarde, 1960). No contact sensitivity to anthracene was observed (Old et al., 1963).

7.4.2 *Benzo[a]pyrene*

Four adult female guinea-pigs were injected with a total of 250 µg benzo[*a*]pyrene in Freund's adjuvant, and two to three weeks later were tested for contact sensitivity with solutions of 0.001, 0.01, 0.1, or 1% benzo[*a*]pyrene in acetone and olive oil. After 24 h, a slight to severe (0.001–1%) contact hypersensitivity was observed (Old et al., 1963).

C3H mice were given an epicutaneous administration of 100 µg benzo[*a*]pyrene in 0.1% acetone solution into the abdominal skin. Five days later, contact hypersensitivity was elicited by applying 20 µg benzo[*a*]pyrene to the dorsal aspect of the ear. The response was quantified by ear thickness, which reaced a maximum three to five days after challenge. The LOAEL for allergic contact sensitivity was thus 120 µg (Klemme et al., 1987).

The ID_{50} value for dermal irritant activity in the mouse ear was 5.6×10^{-5} mmol per ear (Brune et al., 1978).

7.4.3 Naphthalene

A single dose of 100 mg naphthalene to the rabbit eye was slightly irritating, whereas application of 495 mg to rabbit skin, without occlusion, caused mild irritation (Sax & Lewis, 1984).

7.4.4 Phenanthrene

No contact sensitization to phenanthrene was observed (Old et al., 1963).

7.5 Reproductive effects, embryotoxicity, and teratogenicity

The mechanistic aspects of reproductive and embryotoxic effects are presented in detail and the results summarized in Tables 78–80. The genotype of mice is decisive for the manifestation of effects.

Studies have been reported on anthracene, benz[*a*]anthracene, benzo[*a*]-pyrene, chrysene, dibenz[*a,h*]anthracene, and naphthalene. Embryotoxicity was reported in response to benz[*a*]anthracene, benzo[*a*]pyrene, dibenz[*a,h*]-anthracene, and naphthalene. Benzo[*a*]pyrene also had adverse effects on female fertility, reproduction, and postnatal development. In a study in young mice, an NOEL of 150 mg/kg bw per day was obtained for benzo[*a*]pyrene on the basis of effects on fertility (sperm in lumen of testes, size of litters) and embryotoxicity (malformations) (Rigdon & Neal, 1965).

7.5.1 Benzo[a]pyrene

7.5.1.1 Teratogenicity in mice of different genotypes

Benzo[*a*]pyrene is embryotoxic to mice, and the effect is partly dependent on the genetically determined induction of the cytochrome P450 mono-oxygenase receptor, Ah, of the mother and fetus by PAH (see also section 7.10). In the case of an inducible mother (Ah^b/Ah^b and Ah^b/Ah^d, B groups), the genotype of the fetus is not crucial because the active metabolites formed in the mother appear to cross the placenta, causing fetal death or malformation. In contrast, when the mother is non-inducible (Ah^d/Ah^d, D group), the genotype of the fetus is important; one litter may contain both inducible and non-inducible fetuses. Another decisive factor is the route by which benzo[*a*]pyrene is given to the mother. Three studies of the genetic expression of effects are summarized below.

Intraperitoneal injection of benzo[*a*]pyrene at 50 or 300 mg/kg bw on day 7 or 10 of gestation was more toxic and teratogenic *in utero* in genetically inducible C57Bl/6 (Ah^b/Ah^b) than in non-inducible AKR inbred mice (Ah^d/Ah^d).

In AKR x (C57Bl/6)(AKR)F_1 and (C57Bl/6)(AKR)F_1 x AKR back-crosses (father x F_1 mother), allelic differences at the *Ah* locus in the fetus correlated with dysmorphogenesis. The inducible fetal Ah^b/Ah^d genotype results in more stillborn and resorbed fetuses, decreased fetal weight, increased frequency of congenital anomalies, and enhanced P1-450-mediated covalent binding of benzo[*a*]pyrene metabolites to fetal protein and DNA, when compared with fetuses of the non-inducible Ah^d/Ah^d genotype (not-inducible) from the same uterus (see Table 78). In the case of an inducible mother (Ah^b/Ah^d), however, these parameters do not differ in Ah^b/Ah^d and Ah^d/Ah^d individuals in the same uterus, presumably because the increased benzo[*a*]pyrene metabolism in maternal tissues and placenta cancels them out (Shum et al., 1979).

An inducible genotype is not the only factor involved in the reproductive toxicity of benzo[*a*]pyrene. In a study in which C57Bl/6 female mice (*Ah* inducible) were mated with C57Bl/6, DBA/2, or BDF_1 male mice (B groups), and DBA/2 females (non-inducible) were mated with C57Bl/6, DBA/2, or BDF_1 males (D groups) and received intraperitoneal injections of benzo[*a*]-pyrene, fetal mortality increased dose-dependently in all groups except the DBA/2 x DBA/2. Fetal body weight was reduced dose-dependently in all experimental groups, but the effect was more pronounced in D than B groups, as was a dose-dependent increase in the frequency of cervical ribs (for experimental details, see Table 78). These results suggest that *Ah*-inducible fetuses are more sensitive to lethal events, whereas those of non-inducible dams are more susceptible to a decrease in body weight and an increased incidence of cervical ribs. The incidence of external malformations may, however, differ in mice of different genotypes after treatment with benzo[*a*]-pyrene, even if both dams and fetuses are inducible (Hoshino et al., 1981).

The toxicity of benzo[*a*]pyrene *in utero* was investigated in pregnant Ah^d/Ah^d x Ah^b/Ah^d F_1 and Ah^b/Ah^d x Ah^d/Ah^d F_1 back-crossed mice fed benzo[*a*]pyrene in the diet at 120 mg/kg daily on days 2–10 of gestation. Embryos of D females (Ah^d/Ah^d genotype; non-inducible) showed more signs of toxicity and malformations than Ah^d/Ah^d embryos. Fetuses of B females (Ah^b/Ah^d genotype) did not show these changes. The authors suggested that reduced benzo[*a*]pyrene metabolism in the intestine had caused high concentrations in the embryos, and more toxic metabolites (benzo[*a*]pyrene-1,6- and -3,6-quinones) were detected in the Ah^d/Ah^d embryos than in Ah^b/Ah^d embryos (Legraverend et al., 1984). These results were in contrast to those reported after intraperitoneal injection by Shum et al. (1979) and Hoshino et al. (1981). The route of administration can thus affect the magnitude of the observed effects (see also section 7.8.2.2).

7.5.1.2 *Reproductive toxicity*

A single intraperitoneal injection of benzo[*a*]pyrene reduced fertility and destroyed primordial oocytes of DBA/2N mice in a dose-dependent manner (Mattison et al., 1980; see also Table 79).

Table 78. Embryotoxicity of polycyclic aromatic hydrocarbons in experimental animals

Species	No. per group	Route of administration	Duration, dose	Effects	Reference
Anthracene					
Rat Sprague-		Gavage 60 mg/kg bw	Day 19 of gestation,	F_1: no induction of BaP hydroxylase in liver compared with control (< 0.2 vs <0.2 units in controls)	Welch et al. (1972)
Benz[a]anthracene					
Rat	2	Subcutaneous	Day 1–11 or 1–15 of gestation, 5 mg/animal per day	F_0: Day 10 and 12: severe vaginal haemorrhage; Day 14: intraplacental haemorrhage F_1: fetal death and resorption up to day 18	Wolfe & Bryan (1939)
Rat Sprague-Dawley		Gavage	Day 19 of gestation, 60 mg/kg bw	F_1: induction of BaP hydroxylase in liver (12 vs < 0.2 units in controls)	Welch et al. (1972)
Benzo[a]pyrene					
Mouse White Swiss	9	Diet	Day 5 or 10 of gestation until delivery,150 mg/kg bw	F_1: no malformations	Rigdon & Neal (1965)
Mouse C57Bl/6N, AKR/J, and back-crosses (reciprocal)	6–17	Diet	Day 2–10 of gestation, 120 mg/kg per day	F_1: increased intrauterine toxicity and malformations in Ah^d/Ah^d embryos compared with Ah^b/Ah^d embryos in pregnant Ah^d/Ah^d mice (effect not seen in pregnant Ah^b/Ah^d mice)	Legraverend et al. (1984)

Table 78 (contd)

Species (strain)	No. per group	Route of administration	Duration, dose	Effects	Reference
Benzo[a]pyrene (contd)					
Mouse C57Bl/6, AKR and back-crosses (reciprocal)	5–30	Intraperitoneal	Day 7, 10, or 12 of gestation, 50–300 mg/kg bw	200 mg/kg bw: F_1: increase in stillbirths, resorptions, malformations (4-fold higher in pregnant C57Bl than in AKR mice)	Shum et al. (1979)
Mouse C57Bl/6, DBA/2, and back-crosses (reciprocal)	20	Intraperitoneal	Day 8 of gestation, 150 or 300 mg/kg	150 and 300 mg/kg bw: F_0: increased fetal mortality (except DBA/2 x DBA/2 offspring); reduced fetal body weight; increased number of cervical ribs 300 mg/kg: F_1: increased malformations (C57Bl/6 x C57Bl/6)	Hoshino et al. (1981)
Mouse CD-1		Gavage	Day 7–16 of gestation, 10, 40, 160 mg/kg bw per day	F_0: no toxicity F_1: no toxicity	MacKenzie & Angevine (1981)
Rat	17	Subcutaneous	Day 1–11 or 16 of gestation, 5 mg/animal per day	F_0: Days 10 and 12: profuse vaginal haemorrhage; day 14: intraplacental haemorrhage; F_1: fetal death and resorption up to day 18	Wolfe & Bryan (1939)

Table 78 (contd)

Species (strain)	No. per group	Route of administration	Duration, dose	Effects	Reference
Benzo[a]pyrene (contd)					
Rat Sprague-Dawley		Gavage	Day 19 of gestation, 60 mg/kg bw	F_1: induction of BaP-hydroxylase in liver (20 vs < 0.2 units in controls)	Welch et al. (1972)
Rat Sprague-Dawley	10–15	Subcutaneous	Day 6–8 or 6–11 of gestation, 50 mg/kg bw per day	F_1: significant increase in number of resorptions and fetal wastage (dead fetuses plus resorption); fetal weight reduced	Bui et al. (1986)
Chrysene					
Rat Sprague-Dawley		Gavage 60 mg/kg bw	Day 19 of gestation,	F_1: induction of BaP hydroxylase in liver (6 vs < 0.2 units in controls)	Welch et al. (1972)
Dibenzo[a,h]anthracene					
Rat Sprague-Dawley		Gavage	Day 19 of gestation, 60 mg/kg bw	F_1: induction of BaP hydroxylase in liver (15 vs < 0.2 units in controls)	Welch et al. (1972)
Rat	38	Subcutaneous	Day 1–8 or 1–18 of gestation, 5 mg/animal per day	F_0: Days 10 and 12: profuse vaginal haemorrhage; day 14: intraplacental haemorrhage F_1: fetal death and resorption up to day 18	Wolfe & Bryan (1939)

Table 78 (contd)

Species	No. per	Route of (strain) stration	Duration, dose group	Effects admini-	Reference
Naphthalene					
Mouse CD-1	50	Gavage	Day 7–14 of gestation, 300 mg/kg bw per day	F_0: significant15% increase in mortality; significant reduction in weight gain F_1: significant reduction in number of live offspring; no malformations	Plasterer et al. (1985)
Mouse CD-1		Gavage	Day 6–13 of gestation, 300 mg/kg bw per day	F_0 increased mortality 10/50; control: 0/50); significant reduction in weight gain F_1: significant reduction in liveborns per litter	Hardin et al. (1987)
Rat Sprague-Dawley	10–15	Intraperitoneal	Day 1–15 of gestation, 395 mg/kg per day	F_0: no toxicity F_1: no toxicity	Hardin et al. (1981)

For genotypes of the mouse strains used see section 7.5.1.1

In experiments with B6 (*Ah*-inducible) and D2 (non-inducible) mice, primordial oocytes of B6 mice underwent more rapid destruction after treatment with benzo[*a*]pyrene than those of D2 mice. This effect corresponded to a two- to threefold increase in ovarian arylhydrocarbon hydroxylase (AHH) activity in B6 mice after treatment. This correlation was not found in analogous experiments with D2B6F$_1$ mice, in which AHH activity was increased by two- to threefold, but the oocyte destruction was similar to that observed in D2 mice. This demonstrates an inconsistent consequence of strain differences in genotype (Mattison & Nightingale, 1980; see also Table 79). The sum of activation, detoxification, and repair seems to be decisive for the process of oocyte destruction (Figure 8).

Benzo[a]pyrene and its three metabolites, benzo[*a*]pyrene 7,8-oxide, benzo[*a*]pyrene 7,8-diol, and benzo[a]pyrene diol epoxide, were administered by injection at a single dose of 10 µg into the right ovary of B6, D2, and D2B6F$_1$ mice. Ovarian volume, weight, and follicle numbers were measured after two weeks; various reductions were observed in all strains. There was also compesatory hypertrophy of the left ovary (Mattison et al., 1989; see also Table 79).

7.5.1.3 *Effects on postnatal development*

Three studies of the postnatal effects of benzo[*a*]pyrene on mouse offspring, with administration dermally, intraperitoneally, or orally, showed adverse effects, including an increased incidence of tumours, immunological suppression, and reduced fertility (see also Table 80).

7.5.1.4 *Immunological effects on pregnant rats and mice*

Benzo[*a*]pyrene given to pregnant rats on day 15 or 19 of gestation caused alterations at the thymic glucocorticoid receptors in the offspring, suggesting binding to the pre-encoded hormone receptors and interference with receptor maturation (Csaba et al., 1991; Csaba & Inczefi-Gonda, 1992; see also section 7.8.2.6).

Strong suppression of immunological parameters was found in the progeny of mice that had been treated intraperitoneally with benzo[a]pyrene at mid-gestation (Urso & Johnson, 1987; see also section 7.8.2.6).

7.5.2 *Naphthalene*

7.5.2.1 *Embryotoxicity*

Naphthalene was administered by gavage at 50, 150, or 450 mg/kg bw per day to pregnant Sprague-Dawley rats on days 6–15 of gestation, i.e. during the main period of organogenesis. The dams showed signs of toxicity including

Table 79. Effects of benzo[*a*]pyrene on fertility in experimental animals

Species (strain)	Sex/No. per group	Route of administration	Duration, dose	Effects	Reference
Mouse White	M 5	Diet	Up to 30 days before mating, 37.5, 75, or 150 mg/kg bw per day	NOEL: 150 mg/kg bw per day Parameters: sperm in lumen of testes; number of offspring	Rigdon & Neal (1965)
Mouse White Swiss	F 5–65	Diet	20 days before mating 37.5, 75, or 150 mg/kg bw per day	NOEL: 150 mg/kg bw per day Parameter: number of offspring	Rigdon & Neal (1965)
Mouse DBA/2N	F 15	Intraperitoneal	Day 14 before mating, 10, 100, 200, or 500 mg/kg bw once	10, 100 mg/kg bw: dose-dependent decrease in number of pups 200, 500 mg/kg bw: completely infertile; threshold: 3.4 mg/kg bw; 50% effect dose: 25.5 mg/kg bw	Mattison et al. (1980)
Mouse DBA/2N	F	Intraperitoneal	Day 21 before sacrifice, 5, 10, 50, 100, or 500 mg/kg bw once	Dose-dependent increase in primordial oocyte destruction; 500 mg/kg: 100% destruction; threshold: 2.7 mg/kg bw; 50% effect dose: 24.5 mg/kg bw	Mattison et al. (1980)

Table 79 (contd)

Species (strain)	Sex/No. per group	Route of administration	Duration, dose	Effects	Reference
Mouse B6 and D2	F	Intraperitoneal 5	Day 13 before sacrifice, 100 mg/kg bw once	100 mg/kg bw: significant increase in primordial oocyte destruction in both genotypes; effects in B6 mice greater than in D2 mice	Mattison & Nightingale (1980)
Mouse C57Bl/6N (B6), DBA/2N (D2), $D2B6F_1$ (F_1)	F	Intra-ovarian injection	Day 14 before sacrifice, 10 µg/right ovary once	10 µg: decreased ovarian weight (D2); decreased ovarian volume (D2 and F1); decreased antral follicles (F1) decreased number of small follicles (D2 and F1)	Mattison et al. (1989)
Mouse C57Bl/6N	F 5	Intraperitoneal	1, 2, 3, and 4 weeks before sacrifice; 1, 5, 10, 50, 100, or 500 mg/kg bw	500 mg/kg: 35% mortality 1–500 mg/kg bw: dose- and time-dependent decrease in ovarian volume, total volume and number of corpora lutea/ovary (for last parameter, after 1 week threshold was about 1 mg/kg bw and ED_{50} 1.6 mg/kg bw); effect transitory in low-dose groups, but not reversible in two highest by four weeks	Swartz & Mattison, 1985); Miller et al. (1992)

For genotypes of the mouse strains used see section 7.5.1.1

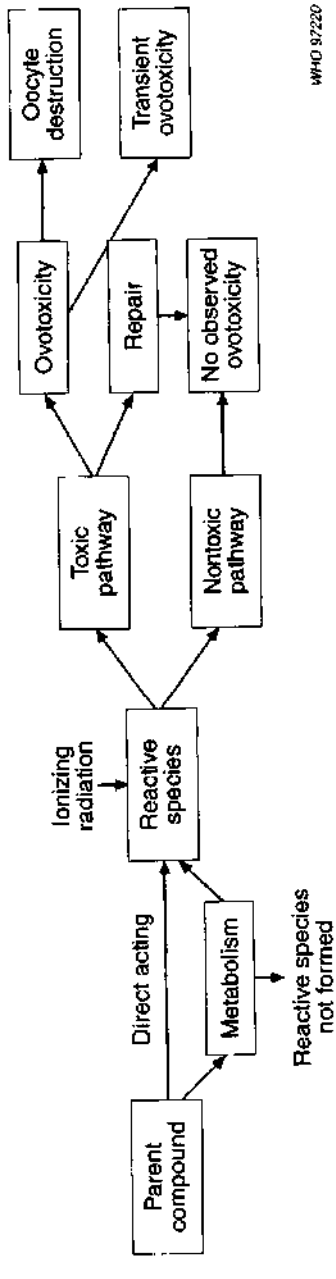


Figure 8. Biological mechanism of ovotoxicity

Table 80. Effects of benzo[*a*]pyrene on postnatal development in experimental animals

Species (strain)	Sex/No. per group	Route of administration	Duration, dose	Effects	Reference
Mouse non-inbred	F	Dermal	Entire gestation period 1 drop of 0.5% solution, twice per week; F_1–F_4 treated with BaP, m 1x/week, f 2x/week	F_1– F_4: sensitization of offspring: increased incidence of papillomas and carcinomas in all generations compared with animals not treated *in utero*	Andrianova (1971)
Mouse C3H/Anf	F 25	Intraperitoneal	Day 11–13 or 16–18 of gestation, 100 or 150 mg/kg	F_1: no difference in birth rate, litter size of progeny compared to controls; severe suppression of anti-SRBC PFC response up to 78 weeks of life (see also section 7.8.2.6); 11–13-fold increase in tumour incidence (liver, lung, ovaries) after 56–78 weeks	Urso & Gengozian (1980)
Mouse CD-1	F	Gavage	Days 7–16 of gestation, 10, 40, 160 mg/kg bw per day	F_1: 10 mg/kg: markedly impaired fertility (by 20%) and reduced testis weight (by 40%), 34% sterility of females; 40 and 160 mg/kg: fertility impaired by > 90%/100%; testis weight reduced by > 80%; 100%/100% sterility of females	MacKenzie & Angevine (1981)

anti-SRBC PFC, anti-sheep red blood cell antibody (plaque)-forming cells

lethargy, slow breathing, prone body posture, and rooting, and these effects persisted after the end of dosing with the high dose. The body-weight gain of treated animals was reduced by 31 and 53% in the groups at the two higher doses. Naphthalene did not induce fetotoxic or teratogenic effects, and the numbers of corpora lutea per dam, implantation sites per litter, and live fetuses per litter were within the range in controls. The maternal NOAEL was < 50 mg/kg bw per day (National Toxicology Program, 1991).

In a second study, doses of 0, 20, 80, or 120 mg/kg bw per day were given to rabbits by gavage during days 6–19 of gestation. There were no signs of maternal toxicity, fetotoxicity, or developmental toxicity (National Toxicology Program, 1992a).

7.5.2.2 *Toxicity in cultured embryos*

Mice injected intraperitoneally on day 2 of gestation with 14 or 56 mg/kg bw naphthalene were sacrificed 36 h later, and embryos were cultured *in vitro*. Maternal doses below the $LD5_0$ value inhibited the viability and implantation capacity of the embryos, and attachment and embryonic growth *in vitro* were markedly decreased (Iyer et al., 1990).

In a subsequent study, three-day-old whole mouse embryos were collected at the blastocyst stage, cultured in NCTC 109 medium, and exposed to naphthalene at 0.16, 0.2, 0.39, or 0.78 mmol/litre for 1 h with and without S9. They were then transferred to toxicant-free medium, cultured for 72 h, and evaluated microscopically. Naphthalene was not directly embryotoxic, but growth and viability were decreased in the presence of S9, with 100% embryolethality at doses > 0.2 mmol/litre; furthermore, hatching and attachment rates were significantly decreased. The approximate LC_{50} in S9-supplemented media was 0.18 mmol/litre (Iyer et al., 1991).

7.6 Mutagenicity and related end-points

Benzo[*a*]pyrene has been used extensively as a positive control in a variety of short-term tests. It is active in assays for the following end-points: bacterial DNA repair, bacteriophage induction, and bacterial mutation; mutation in *Drosophila melanogaster*; DNA binding, DNA repair, sister chromatid exchange, chromosomal aberration, point mutation, and transformation in mammalian cells in culture; and tests in mammals *in vivo*, including DNA binding, sister chromatid exchange, chromosomal aberration, sperm abnormalities, and somatic mutation at specific loci (Hollstein et al., 1979; De Serres & Ashby, 1981). Positive effects were seen in most assays for the mutagenicity of benzo[*a*]pyrene.

A selection of these studies is summarized in Tables 81–88. All of the data available on the other PAH considered in this monograph were taken into account. Because of the amount of data, the purities of the chemicals tested and

details of the assay conditions are omitted from the tables, but they do show the results obtained when S9 was used. Variations in the S9 metabolic activation component of the assay system, e.g. the age, sex, and strain of the rats used as a source of liver and any pretreatment with enzyme inducers such as Aroclor, 3-methylcholanthrene, or phenobarbital, may markedly affect the results and may account for apparent discrepancies.

DNA binding of benzo[*a*]pyrene was observed in various species. For example, adducts were found in cells from hamsters, mice (Arce et al., 1987), rats (Moore et al., 1982), and chickens (Liotti et al., 1988), in calf thymus DNA (Cavalieri et al., 1988a), and in human cell systems (Moore et al., 1982; Harris et al., 1984). Formation of DNA adducts was inhibited in the presence of scavengers of active oxygen species like superoxide dismutase, catalase, and citrate-chelated ferric iron, indicating that reactive oxygen species such as superoxide, OH radicals, and singlet oxygen may be involved in DNA binding (Bryla & Weyand, 1991). Benzo[*a*]pyrene at a total dose of 10 mg/kg bw induced gene mutations in mice, as seen in the coat-colour spot test (Davidson & Dawson, 1976).

The results of tests for reverse mutation in *Salmonella typhimurimum* (Ames test) and for forward mutation in *S. typhimurimum* strain TM677 are presented in Table 81. Bacterial tests for DNA damage *in vitro* are shown in Table 82. The results of tests for mutagenicity in yeasts and *Drosophila melanogaster*, including host-mediated assays, are shown in Table 83. The results of various assays carried out on mammalian cells *in vitro* are summarized in Tables 82–86. The results of tests *in vivo* are shown in Tables 87 and 88.

The activity of PAH in short-term tests is summarized in Table 89, which gives the evaluations of IARC (1983; see also Section 12) and the results of studies reported after 1983. Only three of the 33 PAH considered, i.e. anthracene, fluorene, and naphthalene were inactive in all short-term tests; 16 had mutagenic effects. Eight PAH showed a tendency for mutagenic activity, but the data are still too sparse to permit a final judgement. The available information on acenaphthene, acenaphthylene, benzo[*a*]fluorene, and coronene is still inadequate. As phenanthrene and pyrene showed inconsistent results in various experiments, they could not be clearly classified as mutagenic.

7.7 Carcinogenicity

Most of the studies that have been conducted on PAH were designed to assess their carcinogenicity. Studies on various environmentally relevant matrices such as coal combustion effluents, vehicle exhaust, used motor lubricating oil, and sidestream tobacco smoke showed that PAH are the agents predominantly responsible for their carcinogenic potential (Grimmer et al., 1991b). Because of the abundance of literature, only studies involving the administration of single PAH have been taken into account in this monograph.

Table 81. Mutagenicity of polycyclic aromatic hydrocarbons to *Salmonella typhimurium*

Compound Strain	Result with metabolic activation	Reference
Acenaphthene		
TA98, TA100	–	Florin et al. (1980)
TM677	+	Kaden et al. (1979)
TA98, TA100	+	Epler et al. (1979)
TA100	–	Pahlman & Pelkonen (1987)
Acenaphthylene		
TA98, TA100	–	Florin et al. (1980)
TM677	+	Kaden et al. (1979)
TA98, TA100	–	Bos et al. (1988)
Anthanthrene		
TA98	+	Hermann (1981)
TA100	+	LaVoie et al. (1979); Andrews et al. (1978)
TA98	–	Tokiwa et al. (1977)
TM677	+	Kaden et al. (1979)
Anthracene		
TA98, TA100	–	Purchase et al. (1976)
TA98, TA100	–	Epler et al. (1979)
TA100	–	LaVoie et al. (1979); Gelboin & Ts'o (1978)
TA98, TA100, TA1535, TA1537, TA1538	–	McCann et al. (1975a); Salamone et al. (1979); Ho et al. (1981); Purchase et al. (1976)
TA98, TA100	–	Bridges et al. (1981)
TA98, TA100, TA1535, TA1536, TA1537, TA1538	–	Simmon (1979)
TM677	–	Kaden et al. (1979)
TA97	+	Sakai et al. (1985)
TA98, TA100	–	Probst et al. (1981)
TA100	+	Carver et al. (1986)
TA98, TA100	–	LaVoie et al. (1983a (1985)
TA1535, TA1538	–	Rosenkranz & Poirier (1979)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Anthracene (contd)		
TA100	–	Pahlman & Pelkonen (1987)
TA98, TA100	–	Bos et al. (1988)
TA98, TA100	–	Florin et al. (1980)
Benz[a]anthracene		
TA100	+	Epler et al. (1979); Bartsch et al. (1980)
TA98, TA100	+	McCann et al. (1975a); Coombs et al. (1976); Simmon (1979); Salamone et al. (1979)
TA1535, TA1538	–	Rosenkranz & Poirier (1979)
TA100	+	Pahlman & Pelkonen (1987)
TA98, TA100	+	Hermann (1981); Carver et al. (1986)
TA100	+	Bartsch et al. (1980)
TM677	+	Kaden et al. (1979)
TA100	+	Baker et al. (1980)
TA98, TA100	+	Bos et al. (1988)
TA98, TA100, TA1535, TA1537	+	Probst et al. (1981)
TA98, TA100, TA1537, TA1538	±	Dunkel et al. (1984)
TA1535	–	Dunkel et al. (1984)
TA98, TA100	+	Florin et al. (1980)
TA1537, TA1538	–	Teranishi et al. (1975)
TA98	+	Tokiwa et al. (1977)
Benzo[b]fluoranthene		
TA98	+	Hermann (1981)
TA100	+	LaVoie et al. (1979); Hecht et al. (1980)
TA100	+	Amin et al. (1985a)
TA98, TA100	–	Mossanda et al. (1979)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Benzo[j]fluoranthene		
TA100	+	LaVoie et al. (1980); Hecht et al. (1980)
TM677	+	Kaden et al. (1979)
Benzo[k]fluoranthene		
TA100	+	LaVoie et al. (1980); Hecht et al. (1980)
TA98	+	Hermann et al. (1980)
Benzo[ghi]fluoranthene		
TA98	±	Karcher et al. (1984)
TA100	+	Karcher et al. (1984)
TA98, TA100	+	LaVoie et al. (1979)
Benzo[a]fluorene		
TA98, TA100, TA1535, TA1537, TA1538	–	Salamone et al. (1979)
TA100	+	Epler et al. (1979)
TA100	–	LaVoie et al. (1980)
TA98, TA100	–	Bos et al. (1988)
TA98	+	Tokiwa et al. (1977)
Benzo[b]fluorene		
TA98, TA100	–	LaVoie et al. (1980)
TA98, TA100, TA1535, TA1537, TA1538	–	Salamone et al. (1979)
TM677	+	Kaden et al. (1979)
TA98, TA100	+	Bos et al. (1988)
Benzo[ghi]perylene		
TA98, TA1538	+	Mossanda et al. (1979); Tokiwa et al. (1977); Katz et al. (1981)
TA100	+	Andrews et al. (1978); Katz et al. (1981); LaVoie et al. (1979); Salamone et al. (1979)
TA1537, TA1538	+	Poncelet et al. (1978)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Benzo[ghi]perylene (contd)		
TM677	+	Kaden et al. (1979)
TA97	+	Sakai et al. (1985)
TA100	+	Carver et al. (1986)
Benzo[c]phenanthrene		
TA98	+	Salamone et al. (1979); Wood et al. (1980)
TA100	+	Carver et al. (1986)
TA100	+	Wood et al. (1980)
TA98, TA100	+	Bos et al. (1988)
Benzo[a]pyrene		
TA98	+	Epler et al. (1979)
TA100	+	Andrews et al. (1978)
TA98, TA100	+	LaVoie et al. (1979)
TA98, TA100, TA1537, TA1538	+	McCann et al. 1975a,b)
TM677	+	Kaden et al. (1979)
TM677	+	Rastetter et al. (1982)
TM677	+	Babson et al. (1986b)
TA97, TA98, TA100	+	Sakai et al. (1985)
TA98, TA100	+	Prasanna et al. (1987)); Simmon (1979)); Glatt et al. (1987)
TA1535, TA1538	+	Rosenkranz & Poirier (1979)
TA100	+	Norpoth et al. (1984)); Alzieu et al. (1987)); Carver et al. (1986)); Bos et al. (1988); Hermann (1981); Bruce & Heddle (1979); Marino (1987); Alfheim & Ramdahl (1984)
TA98	+	Lee & Lin (1988)
TA100	+	Pahlman & Pelkonen (1987)
TA97, TA98,TA100	+	Marino (1987)
TA97, TA98, TA100	+	Sakai et al. (1985)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Benzo[a]pyrene (contd)		
TA98, TA100	+	Devanesan et al. (1990)
TM677	+	Skopek & Thilly (1983)
TA98, TA100, TA1535, TA1537, TA1538	+	Dunkel et al. (1984)
TA98, TA100	+	Löfroth et al. (1984)
TA98, TA100	+	Florin et al. (1980)
TA98	+	Tokiwa et al. (1977)
Benzo[e]pyrene		
TA98	+	LaVoie et al. (1979); Hermann (1981)
TA100	±	Salamone et al. (1979)
TA100	+	Andrews et al. (1978); LaVoie et al.,1979)
TA100	±	McCann et al. (1975a)
TA1535, TA1538	–	Rosenkranz & Poirier (1979)
TM677	+	Kaden et al. (1979)
TA100	–	Epler et al. (1979)
TA98, TA100, TA1538	+	Simmon (1979)
TA97, TA100	+	Sakai et al. (1985)
TA98, TA100, TA1535, TA1537, TA1538	±	Dunkel et al. (1984)
TA 100	+	Carver et al. (1986)
TA100	–	Pahlman & Pelkonen (1987)
TA1537,TA1538	–	Teranishi et al. (1975)
TA98	+	Tokiwa et al. (1977)
Chrysene		
TA100	+	McCann et al. (1975a); LaVoie et al. (1979)
TA98	+	McCann et al. (1975a)
TA100	+	Wood et al. (1977)
TA100	+	Epler et al. (1979); LaVoie et al. (1979)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Chrysene (contd)		
TA100	+	Salamone et al. (1979)
TA1535, TA1536, TA1537, TA1538	–	Simmon (1979)
TA98, TA100	+	Bhatia et al. (1987)
TM677	+	Kaden et al. (1979)
TA1535, TA1538	–	Rosenkranz & Poirier (1979)
TA97, TA100	+	Sakai et al. (1985)
TA98, TA100	+	Bos et al. (1988)
TA98	+	Hermann (1981)
TA100	+	Carver et al. (1986)
TA100	+	Pahlman & Pelkonen (1987)
TA100	+	Florin et al. (1980)
TA98	+	Tokiwa et al. (1977)
Coronene		
TA98	+	Mossanda et al. (1979)
TA98	+	Hermann (1981)
TA98	±	Salamone et al. (1979)
TA98	+	Florin et al. (1980)
TA98, TA1537, TA1538	+	Poncelet et al. (1978)
TA97	±	Sakai et al. (1985)
TM677	–	Kaden et al. (1979)
Cyclopenta[cd]pyrene		
TA98	+	Wood et al. (1980)
TA98, TA100, TA1537, TA1538	+	Eisenstadt & Gold (1978)
TM677	+	Kaden et al. (1979); Cavalieri et al. (1981a)
TA98	+	Reed et al. (1988)
Dibenz[a,h]anthracene		
TA100	+	Andrews et al. (1978); Epler et al. (1979); McCann et al. (1975a,b)
TA100	+	Salamone et al. (1979)
TA98	+	Baker et al. (1980)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Dibenz[a,h]anthracene (contd)		
TA98	+	Hermann (1981)
TM677	+	Kaden et al. (1979)
TA100	+	Wood et al. (1978)
TA100	+	Pahlman & Pelkonen (1987); Carver et al., 1986)
TA98, TA100, TA1537, TA1538	+	Probst et al. (1981)
TA100	+	Platt et al. (1990)
TA100	+	Lecoq et al. (1989)
TA1537,TA1538	–	Teranishi et al. (1975)
Dibenzo[a,e]pyrene		
TA100	+	LaVoie et al. (1979)
TA1537,TA1538	+	Teranishi et al. (1975)
TA98, TA100	+, ±	Devanesan et al. (1990)
Dibenzo[a,h]pyrene		
TA100	±	LaVoie et al. (1979)
TA98, TA100	+	Wood et al. (1981)
Dibenzo[a,i]pyrene		
TA100	+	LaVoie et al. (1979); McCann et al. (1975a)
TA100	+	Baker et al. (1980)
TA98	+	Hermann (1981)
TA98	+	Wood et al. (1981)
TA1537,TA1538	+	Teranishi et al. (1975)
Not specified	+	Sardella et al. (1981)
Dibenzo[a,l]pyrene		
TA98, TA100	+	Karcher et al. (1984)
TA98	+	Hermann (1981)
TA98, TA100	+, ±	Devanesan et al. (1990)
Fluoranthene		
TA98	+	Hermann et al. (1980)
TA98	+	Epler et al. (1979)
TA100	–	LaVoie et al. (1979)
TA100	+	LaVoie et al. (1982a)
TA98, TA100,	–	Salamone et al. (1979)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Fluoranthene (contd)		
TA1535, TA1537, TA1538		
TA98, TA100	+	Poncelet et al. (1978)
TA98, TA100	+	Mossanda et al. (1979)
TM677	+	Kaden et al. (1979)
TM677	+	Rastetter et al. (1982)
TM677	+	Babson et al. (1986b)
TA97, TA98, TA100	+	Sakai et al. (1985)
TA98, TA100	+	Bos et al. (1988)
TA100	+	Carver et al. (1986); Hermann (1981); LaVoie et al., 1979)
TA98, TA100	+	Bos et al. (1987)
TA97, TA102, TA1537	±	Bos et al. (1987)
TA1535	–	Bos et al. (1987)
TA98, TA100	+	Bhatia et al. (1987)
TA98, TA100	–	Florin et al. (1980)
TA98	–	Tokiwa et al. (1977)
Fluorene		
TA98, TA100, TA1535, TA1537	–	McCann et al. (1975a); LaVoie et al. (1979, 1980, 1981a)
TM677	–	Kaden et al. (1979)
TA97	–	Sakai et al. (1985)
TA98, TA100	–	Bos et al. (1988)
TA100	–	Pahlman & Pelkonen (1987)
Indeno[1,2,3-cd]pyrene		
TA98	+	Hermann et al. (1980)
TA100	+	LaVoie et al. (1979)
TA100	+	Rice et al. (1985)
5-Methylcholanthrene		
TA100	+	Coombs et al. (1976); Gelboin & Ts'o (1978); LaVoie et al. (1979); McCann et al. (1975a); Hecht et al. (1978)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
5-Methylcholanthrene (contd)		
TA100	+	Amin et al. (1979)
TA100	+	El-Bayoumy et al. (1986)
1-Methylphenanthrene		
TA100	+	LaVoie et al. (1981b)
TM677	+	Kaden et al. (1979)
TA97, TA98, TA100	+	Sakai et al. (1985)
TA98, TA100	+	LaVoie et al. (1983b)
Naphthalene		
TA98, TA100, TA1535, TA1537	–	Florin et al. (1980)
TA98, TA100, TA1535, TA1537, TA1538	–	McCann et al. (1975a)
TA98, TA100, TA1535, TA1538	–	Purchase et al. (1976)
TA98	–	Ho et al. (1981)
TM677	–	Kaden et al. (1979)
G46, *E. coli* K12	–	Krämer et al. (1974)
TA98, TA100	–	Epler et al. (1979)
TA98, TA100	–	Mamber et al. (1984)
TA97, TA98, TA100	–	Sakai et al. (1985)
TA100	–	Pahlman & Pelkonen (1987)
TA98, TA100	–	Bos et al. (1988)
Perylene		
TA98	+	Ho et al. (1981)
TA100	+	LaVoie et al. (1979)
TA98, TA100, TA1535, TA1537, TA1538	–	Salamone et al. (1979)
TA98	+	Hermann (1981)
TA98	+	Florin et al. (1980)
TM677	+	Kaden et al. (1979); Penman et al. (1980)
TA100	+	Carver et al. (1986)
TA97, TA100	+	Sakai et al. (1985)
TA98, TA100	+	Löfroth et al. (1984)
TA100	-	Pahlman & Pelkonen (1987)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Phenanthrene		
TA100	+	Oesch et al. (1981)
TA100	-	Wood et al. (1979)
TA98	+	Epler et al. (1979)
TA98	–	LaVoie et al. (1979, 1980)
TA100	–	LaVoie et al. (1981b)
TA98, TA100	–	Probst et al. (1981)
TA100	–	LaVoie et al. (1979); LaVoie et al. (1980); Gelboin & Ts'o (1978); McCann et al. (1975a)
TA98, TA100, TA1535, TA1537	–	McCann et al. (1975a)
TA100	+	Carver et al. (1986)
TM677	–	Kaden et al. (1979)
TA97	+	Sakai et al. (1985)
TA98, TA100	±	Bos et al. (1988)
TA1535, TA1536, TA1537, TA1538	–	Simmon (1979)
TA1535, TA1538	–	Rosenkranz & Poirier (1979)
TA100	–	Pahlman & Pelkonen (1987)
TA98, TA100, TA1535, TA1537, TA1538	–	Dunkel et al. (1984)
TA98, TA100	-	Florin et al. (1980)
Pyrene		
TA98	-	Ho et al. (1981); Rice et al. (1988a)
TA98, TA100,	-	McCann et al. (1975a); LaVoie et al. (1979);
TA1535, TA1537		Ho et al. (1981)
TA1537	+	Bridges et al. (1981)
TA98, TA100	-	Salamone et al. (1979)
TA98, TA100	-	Probst et al. (1981)
TA1537	+	Epler et al. (1979)
TM677	+	Kaden et al. (1979)
TA97	+	Sakai et al. (1985)
TA98, TA100	±	Bos et al. (1988)

Table 81 (contd)

Strain	Result with metabolic activation	Reference
Pyrene (contd)		
TA100	-	Carver et al. (1986); Hermann (1981)
TA98, TA100	+	Bhatia et al. (1987)
TA98, TA100, TA1535, TA1537, TA1538	–	Dunkel et al. (1984)
TA100	–	Pahlman & Pelkonen (1987)
TA98, TA100	–	Florin et al. (1980)
Triphenylene		
TA98	+	Epler et al. (1979)
TA98	+	Tokiwa et al. (1977)
TA98, TA100	+	Mossanda et al. (1979); Wood et al. (1980)
TA98	+	Hermann (1981)
TA98, TA100	+	Poncelet et al. (1978)
TM677	+	Kaden et al. (1979)
TA98, TA100	+	Bos et al. (1988)
TA100	+	Pahlman & Pelkonen (1987)

TA, used to test reverse mutation to histidine non-auxotrophic mutants); TM, used to test forward mutation to 8-azaguanine-resistant mutants +, positive); -, negative); ±, inconclusive

Table 82 DNA damage induced by polycyclic aromatic hydrocarbons *in vitro*

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Prokaryotes				
Anthracene				
E. coli pol A⁻	R	+	–	Rosenkranz & Poirier (1979)
E. coli WP2, *E. coli* WP100	R	+	–	Mamber et al. (1983)
E. coli WP2, *E. coli* WP67, *E. coli* CM871	R	+/–	–	Tweats (1981)
E. coli PQ37	R	+/–	–	Mersch-Sundermann et al. (1992)
E. coli WP2s(λ) (lambda prophage induction)	R	+/–	+	Rossman et al. (1991)
B. subtilis	R	+/–	–	Ashby & Kilby (1981)
B. subtilis	R	+/–	–	McCarroll et al. (1981)
E. coli GY5027 (prophage induction)	R	+	-	Mamber et al. (1984)
Anthranene				
E. coli PQ37	R	+/-	+	Mersch-Sundermann et al. (1992)
Benz[a]anthracene				
E. coli pol A⁻	R	+	–	Rosenkranz & Poirier (1979)
E. coli WP2 uvrA	R	+	–	Dunkel et al. (1984)
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Benzo[b]fluoranthene				
E. coli PQ37	R	+/-	+	Mersch-Sundermann et al. (1992)
Benzo[ghi]fluoranthene				
E. coli PQ37	R	+/-	+	Mersch-Sundermann et al. (1992)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Benzo[j]fluoranthene				
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Benzo[a]fluoranthene				
E. coli PQ37	R	+/–	–	Mersch-Sundermann et al. (1992)
Benzo[b]fluoranthene				
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Benzo[ghi]perylene				
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Benzo[a]pyrene				
E. coli WP2, *E. coli* WP100	R	+	+	Mamber et al. (1983)
E. coli GY5027	R	+	+	Mamber et al. (1983)
E. coli pol A⁻	R	+	+	Rosenkranz & Poirier (1979)
E. coli WP2, *E. coli* WP67, *E. coli* CM871	R	+/–	+	Tweats (1981)
E. coli WP2 uvrA	R	+	–	Dunkel et al. (1984)
E. coli PQ37	R	+/–	+/+	Mersch-Sundermann et al. (1992)
B. subtilis	R	+/–	+	McCarroll et al. (1981)
E. coli WP2s(λ) (lambda prophage induction)	R	+/–	+	Rossman et al. (1991)
Benzo[e]pyrene				
E. coli pol A⁻	R	+	-	Rosenkranz & Poirier (1979)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Benzo[e]pyrene (contd)				
E. coli WP2 *uvrA*	R	+	–	Dunkel et al. (1984)
E. coli WP2s(λ) (lambda prophage induction)	R	+/–	+	Rossman et al. (1991)
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Chrysene				
E. coli pol A⁻	R	+	–	Rosenkranz & Poirier (1979)
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Coronene				
E. coli PQ37	R	+/–	–	Mersch-Sundermann et al. (1992)
Dibenz[a,h]anthracene				
E. coli	R	+/–	+	Ichinotsubo et al. (1977)
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
B. subtilis	R	+/–	+	McCarroll et al. (1981)
E. coli WP2s(λ) (lambda prophage induction)	R	+/–	+	Rossman et al. (1991)
Dibenzo[a,i]pyrene				
E. coli	R	+/–	+	Ichinotsubo et al. (1977)
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
B. subtilis	R	+/–	+	McCarroll et al. (1981)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Dibenzo[a,h]pyrene				
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Dibenzo[a,l]pyrene				
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Fluoranthene				
E. coli WP2s(λ) (lambda prophage induction)	R	+/–	–	Rossman et al. (1991)
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Fluoranthene				
E. coli WP2, E. coli WP100	R	+	–	Mamber et al. (1983)
E. coli GY5027	R	+	–	Mamber et al. (1984)
E. coli PQ37	R	+/–	–	Mersch-Sundermann et al. (1992)
Indeno[1,2,3-cd]pyrene				
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
Naphthalene				
E. coli WP2, *E. coli* WP100	R	+	–	Mamber et al. (1983)
E. coli GY5027	R	+	–	Mamber et al. (1984)
E. coli PQ37	R	+/–	–	Mersch-Sundermann et al. (1992)
Perylene				
E. coli PQ37	R	+/–	–	Mersch-Sundermann et al. (1992)

Table B2 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Phenanthrene				
E. coli pol A⁻	R	+	–	Rosenkranz & Poirier (1979)
E. coli WP2 *uvrA*	R	+	–	Dunkel et al. (1984)
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)
E. coli WP2s(λ) (lambda prophage induction)	R	+/–	+	Rossman et al. (1991)
B. subtilis	R	+/–	–	McCarroll et al. (1981)
Pyrene	R	+/–		
E. coli			–	Ashby & Kilbey (1981; De Serres & Ashby, 1981)
E. coli WP2, *E. coli* WP100	R	+	–	Mamber et al. (1983)
E. coli GY5027	R	+	–	Mamber et al. (1984)
E. coli WP2 *uvrA*	R	+	–	Dunkel et al. (1984)
E. coli WP2, *E. coli* WP67, *E. coli* CM871	R	+/–	–	Tweats (1981)
E. coli PQ37	R	+/–	–	Mersch-Sundermann et al. (1992)
B. subtilis	R	+/–	–	Ashby & Kilbey (1981)
B. subtilis	R	+/–	–	McCarroll et al. (1981)
E. coli WP2s(λ) (lambda prophage induction)	R	+/–	–	Rossman et al. (1991)
Triphenylene				
E. coli PQ37	R	+/–	+	Mersch-Sundermann et al. (1992)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Eukaryotes				
Acenaphthene				
Rat liver or lung	DA	–	–	Beach & Gupta (1991)
Anthracene				
Primary rat hepatocytes	UDS	–	–	Williams, 1977; Probst et al. (1981)
Primary rat hepatocytes	R	–	–	Tong et al. (1983)
HeLa cells	UDS	+/–	–	Martin et al. (1978; Martin & McDermid (1981)
Human skin fibroblasts	R		–	Milo et al. (1978)
Primary rat hepatocytes	UDS	–	–	Probst et al. (1981)
Human peripheral blood lymphocytes	DA	–	–	Gupta et al. (1988)
Benz[a]anthracene				
Primary rat hepatocytes	UDS	–	+	Probst et al. (1981)
Primary rat hepatocytes	R	–	+	Tong et al. (1983)
HeLa cells	UDS	+/–	+	Martin et al. (1978)
Rat or human mammary epithelial cells	DS	–	±	Mane et al. (1990)
Hamster buccal pouch (epithelial cells (inhibition of DNA synthesis)	DS	–	–	Nagabhushan et al. (1990)
Human peripheral blood lymphocytes	DA	–	+	Gupta et al. (1988)
Benzo[b]fluoranthene				
Rat buccal mucosa epithelial cells	DA	–	+	Autrup & Autrup (1986)
Human leukocytes	DA	+	+	Roggeband et al. (1994a)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Benzo[j]fluoranthene				
Rat buccal mucosa epithelial cells	DA	–	+	Autrup & Autrup (1986)
Benzo[k]fluoranthene				
Human leukocytes	DA	+	+	Roggeband et al. (1994a)
Benzo[a]pyrene				
Primary rat hepatocytes	UDS	–	+	Probst et al. (1981)
Primary rat hepatocytes	R	–	+	Williams et al. (1982)
C3H/10T1/2 mouse clone 8 (DNA breaks)		–	±	Lubet et al. (1983a)
Human leukocytes	DA	+	+	Roggeband et al. (1994a)
Hamster or rat trachea epithelial cells	DA, UDS	–	+	Roggeband et al. (1994b)
HeLa cells	UDS	+/–	+	Martin et al. (1978)
Human skin fibroblasts	R	–	+	Milo et al. (1978)
Human mammary cells (oxidative DNA damage)		–	+	Leadon et al. (1988)
Human fibroblasts	UDS	+	+	Agrelo & Amos (1981)
Human fibroblasts WI-38	UDS	+/–	+	Robinson & Mitchell (1981)
Rat or human mammary epithelial cells	R	–	+	Mane et al. (1990)
Human bronchial cells	DA	–	+	Harris et al. (1984)
Syrian hamster embryo cells	R	–	+	Casto (1979)
Hamster buccal pouch epithelial cells (inhibition of DNA synthesis)		–	+	Nagabhushan et al. (1990)
Rat buccal mucosa epithelial cells	DA	–	+	Autrup & Autrup (1986)
Human peripheral lymphocytes	DA	–	+	Gupta et al. (1988)
Primary rat hepatocytes	DA	–	+	Monteith & Gupta (1992)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Benzo[a]pyrene (contd)				
Primary human hepatocytes	DA	–	+	Monteith & Gupta (1992)
Calf thymus DNA	DA	–	+	Bryla & Weyand (1991)
Primary mouse epidermal keratinocytes	DA, UDS	–	+	Gill et al. (1991)
Primary rat hepatocytes (SCE)	R	–	+	Tong et al. (1983)
Benzo[e]pyrene				
Primary rat hepatocytes (SCE)	R	–	–	Tong et al. (1983)
HeLa cells (UDS)	UDS	+/–	+	Martin et al. (1978)
Primary rat hepatocytes	R	–	–	Williams et al. (1982)
Rat mammary epithelial (DNA synthesis)		–	–	Mane et al. cells (1990)
Syrian hamster embryo cells	R	–	–	Casto (1979)
Human skin fibroblasts	R	–	–	Milo et al. (1978)
Chrysene				
Primary rat hepatocytes	R	–	–	Tong et al. (1983)
Human leukocytes	DA	+	+	Roggeband et al. (1994a)
Cyclopenta[cd]pyrene				
Rat liver or lung tissue	DA	–	+	Beach & Gupta (1991)
Calf thymus DNA	DA	+	+	Beach & Gupta (1994)
Dibenz[a,h]anthracene				
Primary human foreskin epithelial cells	UDS	–	+	Lake et al. (1978)
HeLa cells	UDS	+/–	+	Martin et al. (1978)
Syrian hamster embryo cells	R	–	–	Casto (1979)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Dibenz[a,h]anthracene (contd)				
Primary rat hepatocytes	UDS	–	+	Probst et al. (1981)
Mouse liver DNA	DA	+	+	Lecoq et al. (1991)
Human bronchial cells	DA	–	+	Harris et al. (1984)
Hamster embryonic cells			+	Kuroki & Heidelberger (1972)
C3H10T1/2 mouse clone 8 cells	DA	–	+	Nesnow et al. (1994)
Dibenzo[a,i]pyrene				
Primary rat hepatocytes	UDS	–	–	Probst et al. (1981)
Fluorene				
Primary rat hepatocytes	UDS	–	–	Probst et al. (1981)
Human leukocytes	DA	+	+	Roggeband et al. (1994a)
5-Methylcholanthrene				
Primary rat hepatocytes	UDS	–	+	Tong et al. (1981a)
1-Methylphenanthrene				
Primary rat hepatocytes	UDS	–	+	Tong et al. (1981a)
Chinese hamster ovary cells	DA	+	+	Dunn & Douglas (1991)
Perylene				
Human peripheral blood lymphocytes	DA	–	–	Gupta et al. (1988)
Syrian hamster embryo cells	R	–	–	Casto (1979)
Phenanthrene				
Syrian hamster embryo cells	R	–	–	Casto (1979)

Table 82 (contd)

Test system	End-point	Metabolic[a] activation	Result[b]	Reference
Phenanthrene (contd)				
Human foreskin epithelial cells	UDS	–	–	Lake et al. (1978)
Primary rat hepatocytes (1981)	UDS	–	–	Probst et al.
Human skin fibroblasts	R		–	Milo et al. (1978)
Pyrene				
Syrian hamster embryo cells	R	–	–	Casto (1979)
Human foreskin epithelial cells	UDS	–	–	Lake et al. (1978)
Primary rat hepatocytes	UDS	–	–	Probst et al. (1981)
HeLa cells	UDS	+/–	–	Martin et al. (1978)
Human fibroblast cell line WI38	UDS	+/–	+	Robinson & Mitchell (1981)
Primary rat hepatocytes	R	–	–	Williams et al. (1982)
Human skin fibroblasts	R		–	Milo et al. (1978)
Human skin fibroblasts	UDS	+	–	Agrelo & Amos (1981)
Primary rat hepatocytes	R	–	–	Tong et al. (1983)
Human peripheral blood lymphocytes	DA	–	–	Gupta et al. (1988)
Triphenylene				
Human peripheral blood lymphocytes	DA	–	+	Gupta et al. (1988)

R, DNA repair; DA, DNA adducts; UDS, unscheduled DNA synthesis; SCE, sister chromatid exchange

[a] +, tested with metabolic activation; -, tested without metabolic activation; +/-, tested with and without metabolic activation

[b] Result: +, positive; -, negative; ±, inconclusive; positive results shown if positive only with activation

Table 83. Mutagenicity of polycyclic aromatic hydrocarbons in yeasts and other eukaryotes, host-mediated mutagenicity, and mutagenicity in *Drosophila*

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Yeasts and other eukaryotes				
Anthracene				
Saccharomyces cerevisiae D4-RDII	MGC	–	–	Siebert et al. (1981)
Saccharomyces cerevisiae	MR	–	–	De Serres & Hoffman (1981)
Benzo[a]pyrene				
Saccharomyces cerevisiae D4-RDII	MGC	–	–	Siebert et al. (1981)
Saccharomyces cerevisiae	NMR	–	+	De Serres & Hoffmann (1981)
Paramecium tetraurelia (survival)		+	+	Smith-Sonneborn (1983)
Chrysene				
Saccharomyces cerevisiae D4-RDII	MGC	–	–	Siebert et al. (1981)
Dibenz[a,h]anthracene				
Neurospora crassa		–	+	Barratt & Tatum (1958)
Saccharomyces cerevisiae D4-RDII	MGC	–	–	Siebert et al. (1981)
Naphthalene				
Paramecium tetraurelia (survival)		+	–	Smith-Sonneborn (1983)
Phenanthrene				
Saccharomyces cerevisiae D4-RDII	MGC	–	–	Siebert et al. (1981)
Pyrene				
Saccharomyces cerevisiae; *S. pombe*	NMR; FM	–	–	De Serres & Hoffman (1981)
Host-mediated mutagenicity				
Anthracene				
Salmonella typhimurium TA1530, TA1535, TA1538		–	±	Simmon et al. (1979)
Saccharomycescerevisiae	NMR	–	–	Simmon et al. (1979)

Table 83 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Benz[a]anthracene				
Salmonella typhimurium TA1530, TA1535, TA1538		–	+	Simmon et al. (1979)
Saccharomyces cerevisiae	NMR	–	–	Simmon et al. (1979)
Benzo[a]pyrene				
Salmonella typhimurium TA1530, TA1535, TA1538		–	–	Simmon et al. (1979); Glatt et al. (1985)
Saccharomyces cerevisiae	NMR	–	–	Simmon et al. (1979)
Benzo[e]pyrene				
Salmonella typhimurium TA1538		–	–	Simmon et al. (1979)
Chrysene				
Salmonella typhimurium TA1530, TA1535, TA1538		–	–	Simmon et al. (1979)
Saccharomyces cerevisiae	NMR	–	–	Simmon et al. (1979)
Phenanthrene				
Salmonella typhimurium TA1530, TA1535		–	–	Simmon et al. (1979)
Saccharomyces cerevisiae	NMR	–	–	Simmon et al. (1979)
Drosophila melanogaster				
Anthracene	R		–	Fujikawa et al. (1993)
Benz[a]anthracene	SLRL		+	Fahmy & Fahmy (1973)
Somatic mutation			–	Fahmy & Fahmy (1980)
	SLRL		–	Zijlstra & Vogel (1984)
	SMART		+	Frölich & Würgler (1990)
	R		+	Fujikawa et al. (1993)

Table 83 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Drosophila melanogaster (contd)				
Benzo[a]pyrene	SLRL		±	Vogel et al. (1983)
Somatic mutation			+	Fahmy & Fahmy (1980)
	SLRL		–	Zijlstra & Vogel (1984)
	SLRL		–	Valencia & Houtchens (1981)
Somatic mutation			+	Batiste-Alentorn et al. (1991)
	SMART		+	Frölich & Würgler (1990)
	SLRL		–	Valencia & Houtchens (1981)
	R		+	Fujikawa et al. (1993)
Benzo[e]pyrene	R		–	Fujikawa et al. (1993)
Fluorene	R		–	Fujikawa et al. (1993)
Pyrene	R		±	Fujikawa et al. (1993)

MGC, mitotic gene conversion; NMR, number of mitotic recombinants; MR, mitotic recombination; SLRL, sex-linked recessive lethal mutation; R, DNA repair; FM, forward mutation; SMART, somatic mutation and recombination test

[a] +, tested with metabolic activation; -, tested without metabolic activation; +/-, tested with and without metabolic activation

[b] Result: +, positive; -, negative; ±, inconclusive; positive results shown if positive only with activation

Table 84. Mutagenicity of polycyclic aromatic hydrocarbons in mammalian cells *in vitro*

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Anthracene				
Chinese hamster V79	HPRT	+/–	–	Knaap et al. (1981)
Mouse lymphoma L5178Y	TK	+	–	Amacher & Turner (1980); Amacher et al. (1980)
Human lymphoblastoid TK6	TK	+	–	Barfknecht et al. (1981)
Fischer rat embryo	OR	+	–	Mishra et al. (1978)
Human epithelial EUE cells	DTR	–	–	Rocchi et al. (1980)
Mouse lymphoma L5178Y	TK	+/–	+	Myhr & Caspary (1988)
Benz[a]anthracene				
Chinese hamster V79	HPRT	+	+	Krahn & Heidelberger (1977); Slaga et al. (1978)
Chinese hamster V79	HPRT	+	–	Huberman (1975)
Human lymphoblasts TK6	TK	+	+	Barfknecht et al. (1982)
Human epithelial EUE cells	DTR	–	–	Rocchi et al. (1980)
Human keratinocytes	HPRT	–	–	Allen–Hoffmann & Rheinwald (1984)
Mouse lymphoma L5178Y	TK	+	+	Amacher & Turner (1980); macher et al. (1980)
Mouse lymphoma L5178Y (+ hamster hepatocytes)	TK	–	–	Amacher & Paillet (1983)
Mouse lymphoma L5178Y (+ hamster hepatocytes)	TK	–	+	Amacher & Paillet (1982)
Rat liver epithelial ARL18	HPRT	–	–	Tong et al. (1981a)
Mouse lymphoma L5178Y	TK	+/–	+	Myhr & Caspary (1988)

Table 84 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Benzo[b]fluoranthene				
Chinese hamster V79	HPRT	+	–	Huberman (1975)
Benzo[a]pyrene				
Chinese hamster V79	HPRT	+	+	Arce et al.(1987); Diamond et al. 1980); Huberman (1975)
Chinese hamster V79	HPRT	+	+	Krahn & Heidelberger (1977)
Mouse lymphoma L5186Y (+ hamster hepatocytes)	TK	–	+	Amacher & Paillet (1982)
Chinese hamster ovary	HPRT	+/–	+	Gupta & Singh (1982)
Fischer rat embryo	OR	–	+	Mishra et al. (1978)
Mouse lymphoma L5178Y (+ hamster hepatocytes)	TK	–	+	Amacher & Paillet (1983)
Mouse lymphoma L5178Y	TK	+/–	+	Clive et al. (1979)
Mouse lymphoma L5178Y	TK	+	+	Amacher & Turner (1980); Amacher et al. (1980); Arce et al. (1987)
Mouse lymphoma L5178Y	TK	+	+	Wangenheim & Bolcsfoldi (1988)
Human lymphoblasts AHH	TK	–	+	Crespi & Thilly (1984)
Human lymphoblasts K6	TK	+/–	+	Crespi et al. (1985)
Human epithelial EUE cells	DTR	–	+	Rocchi et al. (1980); Barfknecht et al. (1982)
Human fibroblasts HS172	DTR	+/–	+	Gupta & Goldstein (1981)
Human keratinocytes	HPRT	–	+	Allen-Hoffmann & Rheinwald (1984)
Rat liver epithelial cells ARL 18		–	+	Tong et al. (1981a)

Table 84 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Benzo[a]pyrene (contd)				
Chinese hamster ovary-AS52 (chromosomal mutation)			+	Oberly et al. (1992)
Human epithelial teratoma P3 (cocultivated with human carcinoma BJ cells)	HPRT	–	+	Huberman et al. (1984)
Chinese hamster lung cells V79	HPRT	–	+	Baird et al. (1984)
Mouse lymphoma L5178Y	TK	+/–	+	Myhr & Caspary (1988)
Mouse lymphoma L5178Y	TK	+/–	+	Rees et al. (1989)
Mouse Balb/c-3T3	OR	–	+	Lubet et al. (1990)
Mouse lymphoma L5178Y	TK	+/–	+	Jotz & Mitchell (1981)
Benzo[e]pyrene				
Chinese hamster V79	HPRT	+	–	Hubermann (1978)
Rat liver epithelial ARL18	HPRT	–	–	Tong et al. (1981a)
Mouse C3H10T1/2	OR	–	–	Gehly et al. (1982)
Human epithelial teratoma P3	HPRT	–	–	Huberman et al. (1984)
Chinese hamster lung cells V79	HPRT	–	–	Baird et al. (1984)
Fischer rat embryo cells	OR	+	–	Mishra et al. (1978)
Mouse lymphoma L5178Y	TK	+/–	+	Myhr & Caspary (1988)
Mouse lymphoma L5178Y	TK	+/–	–	Clive et al. (1979)
Mouse Balb/c-3T3	OR	–	–	Lubet et al. (1990)
Chrysene				
Chinese hamster V79	HPRT	+	–	Huberman & Sachs (1976)
Human lymphoblasts TK6	TK	+	+	Barfknecht et al. (1982)
Human epithelial EUE	DTR	–	–	Rocchi et al. (1980)

Table 84 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Chrysene (contd)				
Human epithelial teratoma P3 (cocultivated with human carcinoma BJ cells)	HPRT	–	+	Huberman et al. (1984)
Cyclopenta[cd]pyrene				
Human lymphoblastoid HH-4	HPRT	+	+	Skopek et al. (1979)
Mouse lymphoma L5178Y	TK	+/–	+	Gold et al. (1980)
Human lymphoblasts TK6	TK	+	+	Barfknecht et al. (1982)
Human lymphoblasts AHH1	TK	–	+	Crespi & Thilly (1984)
Chinese hamster V79 (+ hamster embryo fibroblasts)	HPRT	–	+	Raveh et al. (1982)
Dibenz[a,h]anthracene				
Chinese hamster V79	HPRT	+	+	Huberman & Sachs (1976); Huberman (1978)
Chinese hamster V79	HPRT	+	+	Krahn & Heidelberger (1977)
Human epithelial EUE	DTR	–	±	Rocchi et al.,1980)
Fluoranthene				
Human lymphoblastoid HH-4	HPRT	+	+	Thilly et al. (1980)
Human lymphoblasts AHH1	TK	–	–	Crespi & Thilly (1984)
Human lymphoblasts TK6	TK	+	+	Barfknecht et al. (1982)
Fluorene				
Mouse lymphoma L5178Y	TK	+/–	+	Wangenheim & Bolcsfoldi (1988)
1-Methylphenanthrene				
Human lymphoblastoid TK6	TK	+	+	Barfknecht et al. (1981)
Human lymphoblasts AHH1	TK	–	+	Crespi & Thilly (1984)

Table 84 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Perylene				
Human lymphoblastoid TK6	HPRT	+	–	Penman et al. (1980)
Phenanthrene				
Chinese hamster V79	HPRT	+	–	Huberman & Sachs (1976)
Human lymphoblastoid TK6	TK	+	+	Barfknecht et al. (1981)
Fischer rat embryo	OR	+	–	Mishra et al. (1978)
Pyrene				
Mouse lymphoma L5178Y	TK	+/–	+	Jotz & Mitchell (1981)
Fischer rat embryo	OR	+	+	Mishra et al. (1978)
Chinese hamster V79	HPRT	+	–	Huberman (1975)
Mouse lymphoma L5178Y	TK	+	–	Amacher et al. (1980)
Human lymphoblasts TK6	TK	+	–	Barfknecht et al. (1982)
Chinese hamster ovary	HPRT	+/–	–	Heflich et al. (1990)
Human epithelial teratoma P3	HPRT	–	–	Huberman et al. (1984)
Mouse lymphoma L5178Y	TK	+/–	+	Myhr & Caspary (1988)
Mouse lymphoma L5178Y	TK	+/–	+	Wangenheim & Bolcsfoldi (1988)
Rat liver epithelial ARL18	HPRT	–	–	Tong et al. (1981a)
Mouse Balb/c-3T3	OR	–	±	Lubet et al. (1990)
Triphenylene				
Human lymphoblasts	TK	+	+	Barfknecht et al. (1982)

HPRT, hypoxanthine-guanine phosphoribosyl transferase reversion; TK, thymidine kinase reversion; OR, ouabain resistance; DTR, diphtheria toxin resistance

[a] +, tested with metabolic activation; –, tested without metabolic activation; +/–, tested with and without metabolic activation

[b] Result: +, positive; –, negative; ±, inconclusive; positive results shown if positive only with activation

Table 85. Chromosomal effects of polycyclic aromatic hydrocarbons in mammalian cells *in vitro*

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Anthracene				
Chinese hamster D6	CA, SCE	–	–	Abe & Sasaki (1977a)
Rat liver epithelial ARL18	SCE	–	–	Tong et al. (1981b)
Rat liver RL1	CA	–	–	Dean (1981)
Benz[a]anthracene				
Chinese hamster ovary	SCE	–	+	Pal (1981)
Rat liver epithelial ARL18	SCE	–	±	Tong et al. (1981b)
Chinese hamster V79 (coincubation with rat mammary epithelial cells)	SCE	–	±	Mane et al. (1990)
Benzo[a]pyrene				
Rat liver RL1	CA	–	+	Dean (1981)
Chinese hamster V79-4 (+ feeder cells)	CA, SCE	–	–	Popescu et al. (1977)
Chinese hamster lung	CA	+/–	+	Matsuoka et al. (1979)
Mouse lymphoma L5178Y (+ hamster embryo cells)	CA	–	+	Arce et al. (1987)
Human fibroblasts WI-38	CD	+/–	+	Weinstein et al. (1977)
Chinese hamster V79 (+ hamster embryo cells)	SCE	–	+	Arce et al.(1987); Wojciechowski et al. (1981)
Chinese hamster Don-6	SCE	–	+	Abe et al. (1983a)
Chinese hamster ovary	SCE	+/–	+	Husgafvel-Pursiainen et al., 1986)
Chinese hamster ovary	SCE	+/–	+	Evans & Mitchell (1981)
Rat pleural mesothelial cells	SCE	–	+	Achard et al. (1987)
Rat liver epithelial ARL18	SCE	–	+	Tong et al. (1981b)
Rat hepatoma Reuber H4-II-E	SCE	–	+	Dean et al. (1983a)

Table 85 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Benzo[a]pyrene (contd)				
Rat oesophageal tumour R1	SCE	–	+	Abe et al. (1983a)
Rat ascites hepatoma AH66-B	SCE	–	+	Abe et al. (1983a)
Human fibroblasts TIG-II	SCE	–	+	Huh et al. (1982)
Human hepatoma cells	SCE	–	+	Huh et al. (1982)
Human hepatoma C-HC-4 and C-HC-20	SCE	–	+	Abe et al. (1983a,b)
Chinese hamster V79 (coincubation rat/human mammary epithelial cells)	SCE	–	+	Mane et al. (1990)
Primary mouse epidermal keratinocytes	UDS	–	+	Gill et al. (1991)
Human hepatoma (strain Hep G2)	SCE, MN	–	+	Natarajan & Darroudi (1991)
Mouse spleen lymphocytes	SCE	–	+	Wielgosz et al. (1991)
Mouse C3H/10T1/2 clone 8	SCE	–	+	Krolewski et al. (1986)
Human epithelial teratoma P3 (coincubation with rat hepatoma RL-12 cell line)	SCE	–	+	Murison (1988)
Chinese hamster epithelial liver	SCE	–	+	DeSalvia et al. (1988)
Human lymphocytes	CD	+/–	+	Rees et al. (1989)
Benzp[e]pyrene				
Rat liver epithelial ARL18	SCE	–	–	Tong et al. (1981b)
Mouse C3H 10T1/2	CA, SCE	–	–	Gehly et al. (1982)
Chinese hamster V79 cells (coincubation with rat mammary epithelial cells)	SCE	–	–	Mane et al. (1990)
Human epithelial teratoma P3 (coincubation with human breast carcinoma cells BJ-015)	SCE	–	–	Murison (1988)

Table 85 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Cyclopenta[cd]pyrene				
Mouse C3H/10T1/2 clone 8	SCE	–	+	Krolewski et al. (1986)
Human epithelial teratoma P3 (coincubation with human breast carcinoma cells BJ-015)	SCE	–	+	Murison (1988)
Dibenz[a,h]pyrene				
Chinese hamster ovary	SCE	–	+	Pal (1981)
Fluoranthene				
Chinese hamster CHO-1	SCE	+/–	+	Palitti et al. (1986)
Chinese hamster epithelial liver	SCE	–	–	DeSalvia et al. (1988)
Fluorene				
Chinese hamster lung CHL	CA	+/–	+	Matsuoka et al. (1991)
Naphthalene				
Mouse embryos (*in vitro*)	CA	+	+	Gollahon (1991); Gollahon et al. (1990)
Perylene				
Chinese hamster V79	CA	–	+	Popescu et al. (1977)
Chinese hamster V79	SCE	–	–	Popescu et al. (1977)
Phenanthrene				
Chinese hamster V79-4 (+ hamster feeder cells)	SCE	–	–	Popescu et al. (1977)
Chinese hamster V79-4 (+ hamster feeder cells)	CA	–	+	Popescu et al. (1977)
Chinese hamster Don	CA, SCE		–	Abe & Sasaki (1977b)
Chinese hamster lung CHL	CA		–	Ishidate & Odashima (1977)

Table 85 (contd)

Test system	End-point	Metabolic activation[a]	Result[b]	Reference
Phenanthrene (contd)				
Chinese hamster lung CHL	CA	+/–	–	Matsuoka et al. (1979)
Pyrene				
Rat liver epithelial ARL18	SCE	–	–	Tong et al. (1981b)
Chinese hamster D6	CA, SCE	–	–	Abe & Sasaki (1977a)
Chinese hamster ovary	SCE	+/–	+	Evans & Mitchell (1981)
Chinese hamster ovary	SCE	+/–	+	Perry & Thomson (1981)
Chinese hamster V79-4	CA	–	+	Popescu et al. (1977)
Chinese hamster V79-4 (+ hamster feeder cells)	SCE	–	–	Popescu et al. (1977)
Rat liver RL1	CA	–	–	Dean (1981)
Human fibroblasts WI-38	CD	+/–	–	Weinstein et al. (1977)
Human hepatoma (strain Hep G2)	MN, SCE	–	–	Natarajan & Darroudi (1991)
Chinese hamster epithelial liver	SCE	–	–	DeSalvia et al. (1988)

SCE, sister chromatid exchange; MN, micronucleus formation; CA, chromosomal aberration; CD, chromosomal damage

[a] +, tested with metabolic activation; –, tested without metabolic activation; +/–, tested with and without metabolic activation

[b] Result: +, positive; –, negative; ±, inconclusive; positive results shown if positive only with activation

Table 86.Morphological transformation of mammalian cell *in vitro* by polycyclic aromatic hydrocarbons

Test system	Result[a]	Reference
Anthracene		
Balb/c3T3 mouse cells	–	DiPaolo et al. (1972)
Guinea-pig fetal cells	–	Evans & DiPaolo (1975)
Neonatal Syrian golden hamster kidney fibroblasts BHK21 C13	–	Purchase et al. (1976)
Syrian hamster embryo cells	–	Pienta et al. (1977)
Hamster BHK21 clone 13 cells	–	Greb et al. (1980)
Syrian hamster embryo cells	–	Dunkel et al. (1981)
Balb/3T3 mousecells	–	Dunkel et al. (1981)
Balb/3T3 mousecells	–	Peterson et al. (1981); Lubet et al. (1983a)
Fischer rat embryo cells	–	Mishra et al. (1978)
Fischer rat embryo cells (leukaemia virus-infected)	–	Dunkel et al. (1981)
Fischer rat embryo cells (Rauscher leukaemia virus-infected)	±	Freeman et al. (1973)
C3H/10T1/2 mouse clone 8	–	Dunkel et al. (1988)
Benz[a]anthracene		
Syrian hamster embryo cells	+	Pienta et al. (1977); DiPaolo et al. (1969, 1971)
Mouse prostate C3HG23 cells	+	Marquardt & Heidelberger (1972)
C3H/10T1/2 mouse cells	–	Nesnow & Heidelberger (1976)
Hamster BHK21 clone 13 cells	+	Greb et al. (1980)
Hamster embryo cells	–	Grover et al. (1971)
Syrian hamster embryo cells	+	Dunkel et al. (1981)
Syrian hamster lung cells FSHL	+	Emura et al. (1980)
Mouse ventral prostate C3H clone G23 cells	–	Marquardt et al.(1972); Grover et al. (1971)
Balb/3T3 mouse clone 1-13 cells	+	Rundell et al. (1983)
Balb/3T3 mouse cells	±	Dunkel et al. (1981)
Fischer rat embryo cells (Rauscher leukaemia virus infected)	±	Freeman et al. (1973)
Fischer rat embryo cells (leukaemia virus-infected)	+	Dunkel et al. (1981)
Syrian hamster embryo cells	+	DiPaolo et al. (1985)

Table 86 (contd)

Test system	Result[a]	Reference
Benzo[b]fluoranthene		
Hamster BHK21 clone 13 cells	+	Greb et al. (1980)
Syrian hamster lung FSHLcells	+	Emura et al. (1980)
Benzo[k]fluoranthene		
Syrian hamster lung FSHLcells	–	Emura et al. (1980)
Benzo[ghi]perylene		
Syrian hamster embryo cells	-	DiPaolo et al. (1985)
Benzo[a]pyrene		
Golden hamster embryo cells	+	Mager et al. (1977)
Hamster BHK21 clone 13 cells	+	Greb et al. (1980)
Hamster embryo cells (SA7 virus-transformed)	+	Casto et al. (1977)
Syrian hamster embryo cells	+	DiPaolo et al. (1969, 1971)
Syrian hamster embryo cells	+	Dunkel et al. (1981)
Syrian hamster embryo cells	+	Casto et al. (1977)
Syrian hamster lung FSHL	+	Emura et al. (1980, 1987)
Syrian hamster SHE (SA7 virus-transformed)	+	Arce et al. (1987)
C3H/10T1/2 mouse	+	Arce et al. (1987); Lubet et al. (1983b); Peterson et al. (1981)
Balb/3T3 mouse	+	Dunkel et al. (1981)
Balb/3T3 mouse clone A31-1-1	+	Little & Vetrovs (1988)
Fischer rat embryo cells	+	Mishra et al. (1978)
Rat embryo cells (SA7 virus-transformed)	+	DiPaolo & Casto (1976)
Fischer rat embryo cells (Rauscher leukaemia virus-infected)	+	Freeman et al. (1973)
Fischer rat embryo cells (leukaemia virus-infected)	+	Dunkel et al. (1981)
Balb/c 3T3 mouse clone A31 cells	+	Albini et al. (1991)
C3H/10T1/2 mouse clone 8 cells	+	Dunkel et al. (1988)
C3H/10T1/2 mouse clone 8 cells	+	Krolewski et al. (1986)
Balb/c3T3 mouse clone 1-13 cells	+	Rundell et al. (1983)
Benzo[e]pyrene		
C3H/10T1/2 mouse cells	–	Gehly et al. (1982)
Fischer rat embryo cells	–	Mishra et al. (1978)

Table 86 (contd)

Test system	Result[a]	Reference
Benzo[b]fluoranthene		
Syrian hamster embryo cells	–	Pienta et al. (1977)
Syrian hamster embryo cells	+	DiPaolo et al. (1969)
C3H/10T1/2 mouse clone 8 cells	±	Dunkel et al. (1988)
Balb/c-3T3 mouse	–	Lubet et al. (1990)
Syrian hamster lung FSHLcells	±	Emura et al. (1980)
Syrian hamster embryo cells	–	Dunkel et al. (1981)
Balb/3T3 mouse	–	Dunkel et al. (1981)
Hamster BHK21 clone 13 cells	+	Greb et al. (1980)
Chrysene		
Syrian hamster embryo cells	+	Pienta et al. (1977)
Mouse prostate C3HG23 cells	–	Marquardt et al. (1972)
Hamster BHK21 clone 13 cells	+	Greb et al. (1980)
Hamster epithelial lung cell line M3E3/C3	+	Jacob et al. (1993c); Riebeimre et al. (1993)
Cyclopenta[cd]pyrene		
C3H10T1/2 mouse clone 8 cells	+	Gold et al. (1980)
C3H/10T1/2 mouse clone 8 cells	+	Krolewski et al. (1986)
Dibenz[a,h]anthracene		
Syrian hamster embryo cells	+	DiPaolo et al. (1969); Pienta et al. (1977)
C3H 10T1/2 mouse cells	+	Reznikoff et al. (1973)
C3H mouse prostate cells	+	Chen & Heidelberger (1969)
C3HG23 mouse prostate cells	–	Marquardt et al. (1972)
Hamster embryo cells	–	Grover et al. (1971)
Fischer rat embryo cells (Rauscher leukaemia virus-infected)	+	Freeman et al. (1973)
Hamster embryo cells (SA7 virus-transformed)	+	Casto (1973); Casto et al. (1977)
Hamster BHK21 clone 13 cells	+	Greb et al. (1980)
C3H/10T1/2 mouse clone 8 cells	±	Lubet et al. (1983a,b)
Rat embryo cells (SA7 virus-transformed)	+	DiPaolo & Casto (1976)
C3H/10T1/2 mouse clone 8 cells	±	Dunkel et al. (1988)
C3H/10T1/2 mouse clone 8 cells	+	Nesnow et al. (1994)
Fluoranthene		
Fischer rat embryo cells (Rauscher leukaemia virus- infected)	–	Freeman et al. (1973)

Table 86 (contd)

Test system	Result[a]	Reference
Fluorene		
Balb/c3T3 mouse cells	±	Tonelli et al. (1979)
Indeno[1,2,3-cd]pyrene		
Syrian hamster lung FSHLcells	+	Emura et al. (1980)
Naphthalene		
Fischer rat embryo cells (Rauscher leukaemia virus-infected)	–	Freeman et al. (1973)
Human lung WI-38 cells	–	Purchase et al. (1976)
Syrian hamster kidney BHK-21C13 cells	–	Purchase et al. (1976)
Balb/c mouse mammary gland	–	Tonelli et al. (1979)
Balb/c-3T3 mouse cells	–	Rundell et al. (1983)
Perylene		
Syrian hamster embryo cells	–	DiPaolo et al. (1985)
Syrian hamster embryo cells	–	Casto (1979)
Phenanthrene		
Mouse prostate C3HG23 cells	–	Marquardt et al. (1972)
Syrian hamster embryo cells	–	Pienta et al. (1977)
Balb/3T3 mouse cells	–	Kakunaga (1973)
Fetal guinea-pig cells	–	Evans & DiPaolo (1975)
Syrian hamster embryo cells	–	DiPaolo et al. (1969); Dunkel et al. (1981)
Hamster BHK21 clone 13 cells	–	Greb et al. (1980)
Hamster embryo cells (SA7 virus-transformed)	–	Casto et al. (1977)
C3H/10T1/2 mouse cells	–	Peterson et al. (1981)
C3H/10T1/2 mouse cells	–	Lubet et al. (1983b)
Balb/3T3 mouse cells	–	Dunkel et al. (1981)
Fischer rat embryo cells	–	Mishra et al. (1978)
Fischer rat embryo cells (Rauscher leukaemia virus-infected)	±	Freeman et al. (1973)
Fischer rat embryo cells (leukaemia virus-infected)	–	Dunkel et al. (1981)
C3H/10T1/2 mouse clone 8 cells	–	Dunkel et al. (1988)
Pyrene		
Syrian hamster embryo cells	–	DiPaolo et al. (1969); Pienta et al. (1977); Casto (1979)

Table 86 (contd)

Test system	Result[a]	Reference
Pyrene (contd)		
C3H mouse prostate cells	–	Chen & Heidelberger (1969)
Balb/C-3T3 mouse cells	–	DiPaolo et al. (1972); Kakunaga (1973)
Fetal guinea pig cells	–	Evans & DiPaolo (1975)
Fischer rat embryo cells	–	Mishra et al. (1978)
Hamster embryo cells (SA7 virus-transformed)	–	Casto et al. (1977)
C3H/10T1/2 mouse clone 8 cells	±	Dunkel et al. (1988)
Balb/c-3T3 mouse	–	Lubet et al. (1990)

[a] Result; +, positive; ±, inconclusive; –, negative

Table 87. Chromosomal effects of polycyclic aromatic hydrocarbons in mammalian cell systems*in vivo*, including DNA binding and adducts and sperm abnormalities

Test system	Result[a]	Reference
Anthracene		
Chinese hamster bone marrow: CA, SCE	–	Roszinsky-Köcher et al. (1979)
Mouse bone marrow: MN	–	Salamone et al. (1981)
Mouse: sperm abnormalities	–	Topham (1980)
Chinese hamster V79 (mouse host-mediated): SCE	–	Sirianni & Huang (1978)
Mouse peripheral blood : MN	–	Oshiro et al. (1992)
Mouse skin : DNA binding	–	Reddy et al. (1984)
Benz[a]anthracene		
Chinese hamster bone marrow: SCE	+	Roszinsky-Köcher et al. (1979)
Chinese hamster bone marrow: CA (1979)	–	Roszinsky-Köcher et al.
Long-Evans rat bone marrow : CA	–	Sugiyama (1973)
Chinese hamster bone marrow : MN, CA	+	Peter et al. (1979)
NMRI mouse(in metaphase II oocytes): CA	+	Peter et al. (1979)
Mouse gastrointestinal epithelial cells: nuclear anomalies	–	Reddy et al. (1991)
Rat lung: DNA adducts,SCE, MN	+	Whong et al. (1992)
Mouse skin: DNA binding	+	Reddy et al. (1984)
Rat bone marrow and spleen cells: MN	+	Zhong et al. (1995)
Benzo[b]fluoranthene		
Chinese hamster bone marrow: SCE	+	Roszinsky-Köcher et al. (1979)
Chinese hamster bone marrow: CA	–	Roszinsky-Köcher et al. (1979)
Mouse skin: DNA binding	+	Weyand et al. (1987)
Lung, liver and peripheral lymphocytes of rats: DNA adducts	+	Ross et al. (1991); Ross et al. (1992)
Rat lung, liver; peripheral blood lymphocytes; whole blood cultures: SCE	+	Ross et al. (1991); Ross et al. (1992)
Mouse gastrointestinal epithelial cells: nuclear anomalies	+	Reddy et al. (1991)

Table 87 (contd)

Test system	Result[a]	Reference
Benzo[b]fluoranthene (contd)		
Rat peripheral blood lymphocytes: SCE, MN	+	Bryant et al. (1991)
Mouse gastrointestinal epithelial cells: nuclear anomalies	+	Reddy et al. (1991)
Mouse skin: DNA binding	+	Amin et al. (1991a)
Mouse skin: DNA adducts, MN, UDS	+	Winker et al. (1995)
Benzo[j]fluoranthene		
Mouse lung and liver cells: DNA adducts	+	Weyand & LaVoie (1988)
Mouse skin: DNA adducts	+	Weyand et al. (1993)
Benzo[k]fluoranthene		
Mouse skin: DNA binding	+	Weyand et al. (1987)
Mouse lung and liver cells: DNA adducts	+	Weyand & LaVoie (1988)
Benzo[ghi]perylene		
Mouse skin: DNA binding	+	Reddy et al. (1984)
Benzo[a]pyrene		
Mouse: dominant lethal mutation	+	Epstein (1968)
Mouse: dominant lethal mutation	+	Generoso et al. (1982)
Mouse: spot test	+	Russell (1977)
Mouse: spot test	+	Davidson & Dawson (1976)
Rat hepatocytes: UDS	–	Miralis et al. (1982)
Mouse germ cells: UDS	–	Sega (1979)
Mouse skin: DNA binding	+	Weyand et al. (1987); Rice et al. (1984)
Chinese hamster bone-marrow cells: CA, SCE	+,+	Roszinsky-Köcher et al. (1979)
Chinese hamster bone-marrow cells: CA, SCE	±,+	Bayer (1978)
Mouse: CA; heritable translocations	–	Generoso et al. (1982)
Mouse bone marrow: MN	+	Salamone et al. (1981)
Mouse bone marrow: MN	–	Bruce & Heddle (1979)
Chinese hamster bone-marrow cells: MN	–	Bayer (1978)
Mouse: sperm abnormalities	+	Topham (1980)
Mouse: sperm abnormalities	+	Bruce & Heddle (1979)

Table 87 (contd)

Test system	Result[a]	Reference
Chinese hamster V79 (mouse host-mediated): SCE	+	Sirianni & Huang (1978)
Mouse epidermal cells: DNA adducts	+	Albert et al. (1991a,b)
Mouse bone marrow: MN	+	Shimada et al. (1991)
Mouse keratinocytes: MN	+	He & Baker (1991)
Mouse lung and liver cells: DNA adducts	+	Weyand & LaVoie (1988)
Mouse liver, lung and stomach: DNA adducts	+	Cummings et al. (1991)
Rat peripheral lymphocytes: SCE	+	Li et al. (1991)
Mouse bone marrow: MN	+	Mavournin et al. (1990)
Mouse bone marrow: MN	+	Kliesch et al. (1982)
Mouse bone marrow: MN	+	Harper & Legator (1987)
Mouse peripheral blood cells: MN	±	Oshiro et al. (1992)
Mouse gastrointestinal epithelial cells: nuclear anomalies	+	Reddy et al. (1991)
Mouse bone marrow: SCE	+	Wielgosz et al. (1991)
Rat peripheral blood lymphocytes: SCE, DNA adducts	+	Willems et al. (1991)
Rat liver cells: DNA adducts	+	Willems et al. (1991)
Rat peripheral blood lymphocytes: CA	-	Willems et al. (1991)
Chinese hamster cells: CA	+	Matsuoka et al. (1979)
Mouse skin epithelial cells: DNA binding	+	Hughes & Phillips (1991)
Human peripheral lymphocytes: DNA adducts	+	Haugen et al. (1986)
Rat lung, liver and peripheral lymphocytes: DNA adducts	+	Ross et al. (1991)
Mouse bone marrow: MN	+	Awogi & Sato (1989)
Mouse skin: DNA binding	+	Reddy et al. (1984)
Mouse skin: DNA adducts	+	Oueslati et al. (1992)
Mouse and rat bone marrow: MN	+	Shimada et al. (1992)
Mouse bone marrow: CA	+	Adler & Ingwersen (1989)
Benzo[e]pyrene		
Chinese hamster bone-marrow cells: SCE	+	Roszinsky-Köcher et al. (1979)
Chinese hamster bone-marrow cells: CA	–	Roszinsky-Köcher et al. (1979)
Mouse gastrointestinal epithelial cells: nuclear anomalies	–	Reddy et al. (1991)
Mouse skin: DNA binding	±	Reddy et al. (1984)

Table 87 (contd)

Test system	Result[a]	Reference
Chrysene		
Chinese hamster bone-marrow cells: SCE	+	Roszinsky-Köcher et al. (1979)
Chinese hamster bone-marrow cells: CA	–	Roszinsky-Köcher et al. (1979)
NMRI mice: metaphase II oocytes	+	Basler et al. (1977)
Mouse keratinocytes: MN	+	He & Baker (1991)
Mouse skin: DNA binding	+	Reddy et al. (1984)
Dibenz[a,h]anthracene		
Chinese hamster bone-marrow cells: SCE	+	Roszinsky-Köcher et al. (1979)
Chinese hamster bone-marrow cells: CA	–	Roszinsky-Köcher et al. (1979)
Rat peripheral blood lymphocytes: SCE, MN	–	Bryant et al. (1990)
Mouse skin: DNA binding	+	Lecoq et al. (1991)
Mouse skin: DNA binding	+	Reddy et al. (1984)
Rat bone-marrow and spleen cells: MN	+	Zhong et al. (1995)
Rat lung: DNA adducts, MN, SCE	+	Whong et al. (1994)
Mouse skin: DNA adducts, MN, UDS	+	Winker et al. (1995)
Dibenzo[a,e]pyrene		
Mouse skin epithelial cells: DNA binding	+	Hughes & Phillips (1991)
Dibenzo[a,i]pyrene		
Rat spleen cells: MN	+	Zhong et al. (1995)
Rat lung: DNA adducts, MN, SCE	+	Whong et al. (1994)
Fluoranthene		
Mouse bone-marrow cells: SCE	–	Palitti et al. (1986)
Indeno[1,2,3-cd]pyrene		
Mouse skin: DNA binding	+	Weyand et al. (1987)
Mouse skin epithelial cells: DNA binding	+	Rice et al. (1990)
5-Methylcholanthrene		
Mouse skin: DNA adducts	+	Amin et al. (1985a)

Table 87 (contd)

Test system	Result[a]	Reference
Naphthalene		
Mouse bone-marrow cells: MN	–	Harper et al. (1984)
Perylene		
Mouse skin: DNA binding	–	Reddy et al. (1984)
Phenanthrene		
Chinese hamster bone-marrow cells: CA	–	Bayer (1978); Roszinsky-Köcher et al. (1979)
Chinese hamster bone-marrow cells: SCE	+	Bayer (1978); Roszinsky-Köcher et al. (1979)
Chinese hamster bone-marrow cells: MN	–	Bayer (1978)
Pyrene		
Mouse bone marrow cells: SCE	–	Paika et al. (1981)
Mouse bone marrow: MN	–	Salamone et al. (1981)
Mouse bone marrow cells: MN	–	Tsuchimoto & Matter (1981)
Chinese hamster V79 (mouse host-mediated): SCE	–	Sirianni & Huang (1978)
Mouse keratinocytes: MN	–	He & Baker (1991)
Mouse peripheral blood cells: MN	–	Oshiro et al. (1992)
Mouse gastrointestinal epithelial cells: nuclear anomalies	–	Reddy et al. (1991)
Mouse: sperm abnormalities	–	Topham (1980)
Mouse skin: DNA binding	–	Reddy et al. (1984)

SCE, sister chromatid exchange; MN, micronucleus assay; CA, chromosomal aberrations; UDS, unscheduled DNA synthesis

[a] Result: +, positive; ±, inconclusive; –, negative

Table 88. Effects of polycyclic aromatic hydrocarbons on morphological transformation of mammalian cells *in vivo*

Test system	Result[a]	Reference
Anthracene		
Mouse bone-marrow cells	–	Salamone et al. (1981)
Chinese hamster embryo cells	–	DiPaolo et al. (1973)
Benzo[ghi]perylene		
Hamster embryos, transplacental exposure	–	Quarles et al. (1979)
Benzo[a]pyrene		
Hamster embryos, transplacental exposure	+	Quarles et al. (1979)
Phenanthrene		
Hamster embryos, transplacental exposure	–	Quarles et al. (1979)

[a] +, positive; ±, inconclusive; –, negative

Table 89. Overview of genotoxicity of polycyclic aromatic hydrocarbons

Compound	Results
Acenaphthene	Inconsistent, limited database
Acenaphthylene	Inconsistent, limited database
Anthanthrene	Positive, limited database
Anthracene	Negative, with a few exceptions
Benz[*a*]anthracene	Positive
Benzo[*b*]fluoranthene	Positive
Benzo[*j*]fluoranthene	Positive
Benzo[*k*]fluoranthene	Positive
Benzo[*ghi*]fluoranthene	Positive, limited database
Benzo[*a*]fluorene	Inconsistent, limited database
Benzo[*b*]fluorene	Inconsistent, limited database
Benzo[*ghi*]perylene	Positive
Benzo[*c*]phenanthrene	Positive, limited database
Benzo[*a*]pyrene	Positive
Benzo[*e*]pyrene	Positive
Chrysene	Positive
Coronene	Positive, limited database
Cyclopenta[*cd*]pyrene	Positive
Dibenz[*a,h*]anthracene	Positive
Dibenzo[*a,e*]pyrene	Positive
Dibenzo[*a,h*]pyrene	Positive, limited database
Dibenzo[*a,i*]pyrene	Positive
Dibenzo[*a,l*]pyrene	Positive, limited database
Fluoranthene	Positive
Fluorene	Negative, with a few exceptions
Indeno[1,2,3-*cd*]pyrene	Positive
5-Methylchrysene	Positive
1-Methylphenanthrene	Positive
Naphthalene	Negative
Perylene	Positive
Phenanthrene	Inconsistent
Pyrene	Inconsistent
Triphenylene	Positive

Benzo[a]pyrene has been tested in a range of species, including frogs, toads, newts, trout, pigeons, rats, guinea-pigs, rabbits, ferrets, ground squirrels, tree shrews, marmots, marmosets, and rhesus monkeys. Tumours have been observed in all experiments with small animals, and the failure to induce neoplastic responses in large animals has been attributed to lack of information on the appropriate route or dose and the inability to observe the animals for a sufficient time (Osborne & Crosby, 1987a). In studies with other PAH,

benzo[*a*]pyrene was often used as a positive control and therefore administered at only one concentration. Benzo[*a*]pyrene has been shown to be carcinogenic when given by a variety of routes, including diet, gavage, inhalation, intratracheal instillation, intraperitoneal, intravenous, subcutaneous, and intrapulmonary injection, dermal application, and transplacental administration.

Assessment of the carcinogenic potency of the selected PAH is restricted for various reasons: Many of the studies performed before about 1970 were carried out without controls, without clearly defined, purified test substances, or using experimental designs and facilities considered today to be inadequate. Despite these shortcomings, all of the available studies were taken into account, except for those on dibenz[*a*,*h*]anthracene and benzo[*a*]pyrene. An overview of the results, as reported by the authors, is given in Table 90. To facilitate appraisal of the studies, the penultimate column gives a classification of the substances as positive, negative, or questionably carcinogenic; indicates whether the tumour incidence was evaluated statistically; and judges that a study is valid or provides reasons suggesting that it is unreliable. The criteria used to classify a study as valid were (i) an appropriate study protocol, i.e. use of concurrent controls (sham or vehicle), 20 or more animals per group, and study duration at least six months; and (ii) sufficient documentation, including detailed description of administration, results, and the survival of animals. As the use of concurrent controls is important for making judgements, data for these are given with the results for treated groups. If control data are not mentioned, it is because they were not given in the original paper.

In experiments by topical application, the lower, more volatile PAH partially evaporate, and therefore their doses may have varied. The substances may also decompose. Both features could lead to underestimations of carcinogenic potency if they are not taken into account.

Table 91 shows the classification of the compounds as carcinogenic, noncarcinogenic, or questionably carcinogenic. In order to make these classifications, all of the studies were summarized according to species and route of administration. In cases of doubt, the judgement was based on valid studies only. For example, despite one positive but invalid result and two questionable (one valid, one invalid) results from 17 studies, anthracene was classified as negative; however, pyrene, for which one positive, valid result and three questionable, valid results were found in 15 studies, could not be classified as negative and the compromise 'questionable' was chosen.

The PAH found not to be carcinogenic were anthracene, benzo[*ghi*]perylene, fluorene, benzo[*ghi*]fluoranthene, 1-methylphenanthrene, perylene, and triphenylene. Questionable results were obtained for acenaphthene, benzo[*a*]-fluorene, benzo[*b*]fluorene, coronene, naphthalene, phenanthrene, and pyrene. The remaining compounds were found to be carcinogenic.

The dermal route was the commonest mode of administration, followed by subcutaneous and intramuscular injection. In most studies, the site of tumour

Table 90. Carcinogenicity of polycyclic aromatic hydrocarbons in experimental animals

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Acenaphthene									
	Mouse		100	Dermal	Dissolved in 90% benzene	9 months	No tumours observed	n no/ic	Kennaway (1924)
'Pure'	Mouse, white	m	85	Dermal	3 drops, 1 x/week of ~ 3% solution, 1 year; initiation experiment	≤1 year	After 12 months 5/85 survived with a total of 2 tumours; 0.4 tumour/animal; promotor only: 0.08 tumour/animal	q no/ls	Graffi et al. (1953)
	Mouse		30	Dermal	0.3% in benzene, 2 x/week, life	Life	1/30 lung adenoma	n no/ld	Badger et al. (1940)
Recrystal-lized	Mouse, Ha/ICR/Mil	f	20	Dermal	0.05 or 0.1%, 3 x/week, 12 months	15 months	0/20 with tumours	n no/val	Hoffmann & Wynder (1966)
Recrystal-lized	Mouse, Swiss Ha/ICR/Mil	f	30	Dermal	25 µg/animal, 10 x over 20 days; initiation experiment	6 months	2/25 papillomas; promotor only: 2/26	n no/val	Hoffmann & Wynder (1966)
'Rigourous-ly purified'	Mouse, ICR/Ha	f	13	Dermal	0.25 mg/animal, 4 x; initiation experiment	65 weeks	2/13 papillomas; promotor only: 520 papillomas; control acetone: 0/20 papillomas	n no/val	Van Duuren et al. (1968)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat / Val.	Reference
Acenaphthene (contd)									
Recystallized	Mouse, Swiss	f	30	Dermal	43 µg/animal, 2 x/week, 75 weeks	≤ 100	1/30 with skin carcinoma; control: 2/30 with carcinomas	n no/**val**	Lijinsky & Garcia (1972)
98.65%	Mouse, Swiss	f	40	Dermal	109 µg/animal, 2 x/week, 30 weeks	70 weeks	47% skin-tumour-bearing animals; solvent control: 0%	p no/**val**	Cavalieri et al. (1977)
TLC-purified)	Mouse, CD-1	f	30	Dermal	0.69 mg/animal, 1x; initiation experiment	35 weeks	18% with papillomas; promoter only: 3%	p no/**val**	Scribner (1973)
> 99%	Mouse, Sencar	f	27	Dermal	221 µg/animal, 1x; initiation experiment	26 weeks	11% papillomas; solvent only: 9%	n yes/**val**	Cavalieri et al. (1989)
	Mouse, XVII	m/f	7/7	s.c.	0.6 mg/animal, 1x/month, 3 months		No local sarcomas observed	n no/ln,ld	Lacassagne et al. (1958)
99.4%	Rat Osborne-Mendel	f	35	Intrapulm.	0.65 and 3.4 mg/kg, 1x	102/88 weeks	1/35 and 19/35 with lung tumours; control: no tumours	p yes/**val**	Deutsch-Wenzel et al. (1983)
> 99%	Rat, Sprague-Dawley	f	20	Intra-mammary injection	1.1 mg/gland, 1x, 8 glands	≤ 40 weeks	1/20 with mammary tumours; control: 0/21 or 2/20	n yes/**val**	Cavalieri et al. (1989)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Anthracene									
	Mouse		2x100	Dermal	40% suspension/solution	5 months	0/100, 1/100 tumours	n no/lc	Kennaway (1924)
	Mouse		44	Dermal	5%, 3x/week	≤ 11 months	No skin tumours	n no/lc	Miescher (1942)
	Mouse, 'S'		20	Dermal	1.5 mg/animal, 2x/day, 3 days/week; total: 20 x; initiation experiment	21 weeks	3/17 with tumours; promotor only: 4/19	n yes/val	Salaman & Roe (1956)
	Mouse, Swiss Millerton	f	5	Dermal	10% solution, 3x/week, life	≤ 20 months	No skin tumours	n no/ln,ld	Wynder & Hoffmann (1959a)
TLC-purified	Mouse, CD-1	f	30	Dermal	1.8 mg/animal, 1x; initiation experiment	35 weeks	14% with papillomas; promotor only: 3%	q no/val	Scribner (1973)
	Mouse, Skh:hairless 1, outbred	m/f	24	Dermal	4 µg, 1x/day, 5 days/week, 38 weeks, then 2 h/day UV	38 weeks	No increased tumour frequency compared with controls	n yes/val	Forbes et al. (1976)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Anthracene (contd)									
	Mouse, Swiss albino (Ha/ICR)	f	20	Dermal	100 µg/animal, 10x on alternate days; initiation experiment	24 weeks	15% with tumours; solvent:: 10%	n yes/**val**	LaVoie et al., (1983a)
	Mouse, C57Bl	m/f	40–50	s.c.	5 mg/animal in tricaprylin; 1x	≤ 22–28 months	0/26 sarcomas after 5 months	n yes/ld	Steiner (1955)
	Mouse, Swiss	m	5	i.p.	1000 mg/kg, 1x	≤ 5 months	No effects observed	n no/ln	Shubik & Della Porta (1957)
	Rat		31	Oral (diet)	6 mg/animal/day, 7x/ week	33 months	22/31 alive after 1 year; no tumours after 33 months	n no/lc	Schmähl & Reuter, cited by Gerarde (1960)
Highly purified	Rat, BD I /BD III		28	Oral (diet)	5–15 mg/animal/day, 6x/week, 78 weeks	700 days	2/28 malignant tumours	n no/ld	Schmähl (1955)
	Rat		10	s.c.	1 mg/animal, 1x/week, 103 weeks	≤ 103 weeks	No subcutaneous sarcomas	n no/ln ld	Boyland & Burrows (1935)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Anthracene (contd)									
	Rat, Wistar		5	s.c.	5 mg/animal, 6–7x	10 months	No tumours observed	n no/ln	Pollia (1941)
Highly purified	Rat, BD I /BD III		10	s.c.	20 mg/animal, 1x/week, 33 weeks	≤ 29 months	5/9 tumours (fibromas) at site injection	p no/ln,ld	Druckrey & Schmähl (1955)
Highly purified	Rat, BD I /BD III		10	i.p.	20 mg/animal, 1x/week,	> 2 years 33 weeks	1/10 spindle-cell sarcoma	q	Schmähl (1955) no/ln,ld
	Rat, Osborne/Mendel	f	60	Intrapulm.	0.5 mg/animal,	1x1 year	No lung tumours; control: no tumours	n no/ld	Stanton et al. (1972)
	Rabbit		9	Cerebral implant	4-20 mg/animal, 1x	20–54 months	No glioma	n no/ln,ld	Russel (1947)
Benz[a]anthracene									
	Mouse, C57/BL		8–19	Oral	0.5 mg/animal, 1x, 8x or 16 x (highest dose), ≤ 2 months	16 months	0/13, 1/19 and 1/8 with papillomas; no carcinomas observed; control: 0/12	q yes/ln	Bock & King (1959)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benz[a]anthracene (contd)									
	Mouse, B6AF1/J, newborn	m	20 or 40	Oral	1.5 mg/animal, 3x/wk, 5 weeks	≤ 547–600 days	100% hepatomas and 95% pulmonary adenomas; solvent only: 10% hepatomas and 35% pulmonary adenomas	p no/**val**	Klein (1963)
	Mouse, B6AF1/J, newborn	m	20	Oral	1.5 mg/animal, 1x/day, 2 days	≤ 568 days	80% hepatomas and 85% lung adenomas (inadequately reported)	p no/**val**	Klein (1963)
Purified	Mouse		30	Dermal	0.3% in benzene, 2x/week,life	≤ 584 days	1/30 epitheliomas	n no/ld	Barry et al. (1935)
'Pure'	Mouse, white	m	75	Dermal	3 drops, 1x/week of 0.5% solution, 1 year; initiation experiment	≤1 year	After 12 months 9/75 survived with a total of 18 tumours; 2 tumours/animal; promotor only: 0.08 tumour/animal	p no/**val**	Graffi et al. (1953)
Recystal-lized	Mouse, albino	f	30	Dermal	66 µg/animal, 2x/week, 20 weeks	13–15.5 months	No tumours; solvent only: no tumours	n no/**val**	Miller & Miller (1963)
	Mouse, C3H		20	Dermal	0.5% solution, 2x/week, 638 days	638 days	No tumours; control: no tumours	n no/**val**	Stevenson & von Haam (1965)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benz[a]anthracene (contd)									
Recrystallized	Mouse, C3H/He		30–50	Dermal	0.0001-0,5 mg/animal in *n*-dodecane or 0.1 mg/animal in toluene, 3x/week, 50 weeks	≤ 88 weeks	Dose-dependent increase in malignant tumours; solvent control: no tumours	p yes/val	Bingham & Falk (1969)
Recystallized	Mouse, Swiss Millerton ICR/Ha	f	20	Dermal	1 mg/animal, 1x; initiation experiment	58–60 weeks	10/20 with papillomas; promotor only:1/20; solvent control: 0%	p no/val	Van Duuren et al. (1970)
TLC purified	Mouse, CD-1	f	30	Dermal	0.5 mg/animal, 1x; initiation experiment	35 weeks	62% with papillomas; promotor only: 3%	p no/val	Scribner (1973)
> 99%	Mouse, Swiss	f	40	Dermal	90 µg/animal, 2x/week, 30 weeks	70 weeks	2.6% skin-tumour bearing animals; solvent control: 0%	n no/val	Cavalieri et al. (1977)
> 99%	Mouse, CD-1	f	30	Dermal	0.46 mg/animal, 1x; initiation experiment	26 weeks	57% with papillomas; promotor only: 6%	p no/val	Slaga et al. (1978)
	Mouse, CD-1	f	30	Dermal	0.1 and 0.57 mg/animal, 1x; initiation experiment	27 weeks	14% and 36% ($p < 0.05$)with tumours; solvent control: 7%	p yes/val	Levin et al. (1984)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benz[a]anthracene (contd)									
	Mouse, CD-1	f	30	Dermal	0.23 and 0.57 mg/animal, 1x; initiation experiment	27 weeks	17% and 38% papillomas; solvent control: 4%	p yes/**val**	Weyand et al. (1990);Wood et al. (1980)
Spectro-meter	Mouse, C57Bl control	m/f	50	s.c.	5 mg/animal in tricaprylin; 1x	≤ 22 months	8/46 sarcomas after 4 months; solvent control: 3/280	p no/**val**	Steiner & Falk (1951)
	Mouse, C57Bl	m/f	40-50	s.c.	0.05, 0.2, 1, 5, or 10 mg/animal in tricaprylin; 1x	≤ 22–28 months	5/44, 11/45, 15/44, 20/36 and 5/16 sarcomas	p yes/ld	Steiner & Edgcomb (1952); Steiner (1955)
Recrystal-lized	Mouse, albino	f	30	s.c.	0.94 mg/animal, 1x	≤15 months	No sarcomas; solvent control: no tumours	n no/**val**	Miller & Miller (1963)
	Mouse C3H		20	s.c.	5 mg in tricaprylin, 1x	638 days	No tumours; control: no tumours	n no/**val**	Stevenson & von Haam (1965)
	Mouse, C57BL	m/f	10/10	s.c.	1 mg/animal, 1x/week, 10 weeks	60-80 weeks	8/10 m and 6/10 f with sarcomas; control: 0/20 m and 0/20 f	p no/**val**	Boyland & Sims (1967)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benz[a]anthracene (contd)									
	Mouse, Swiss newborn	m/f	87	s.c.	0.2 mg/animal in polyethylene glycol on days 0, 1 and 2 after birth	70–75 weeks	70 weeks: 15/15 m and 2/18 f with liver tumours, 4/15 m and 10/18 f with lung tumours; corrected control data: 4/22 m and 1/23 f with liver tumours and 3/22 m and 1/23 f with lung tumours	p no/**val**	Grover et al. (1975)
	Mouse, Swiss Webster BLU:Ha(ICR), newborn	m/f	140	i.p.	9.1, 18.2, and 36.4 µg/animal on days 1, 8, and 15 after birth	26 weeks	10/47 m and 4/38 f with pulmonary tumours; solvent control: 7/43 and 2/24	n no/**val**	Wislocki et al. (1979)
	Mouse, A		11	i.v.	10 mg/kg, 1x	20 weeks	18% lung tumours; control: 21%	n no/**val**	Shimkin & Stoner (1975)
	Mouse, C57 x IF F_1 hybrid		52	Bladder implant	About 2 mg/animal, 1x	≥ 40 weeks	17/52 bladder carcinomas and 1/52 papillomas; control: 4/89	p yes/**val**	Clayson et al. (1968)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benz[a]anthracene (contd)									
	Rat, Sprague-Dawley	f	10	Oral	200 mg/rat, 1x	60 days	No tumours in treated animals; control: 8/164 after 310 days	n no/ln,lc	Huggins & Yang (1962)
	Rat, Donryu	m	25	Dermal	Saturated solution in acetone,dropped at 2x/wk to cover 2 cm^2, 5 months	≤18 months	No tumours	n no/ld	Tawfic (1965)
Recrystal-lized	Rat, Holtzman	m	20	s.c.	1.88 mg/animal, 1x	≥ 4 months	No sarcomas; solvent control: no tumours	n no/val	Miller & Miller (1963)
TLC purified	Rat, Sprague-Dawley	f 3x	28	i.v.	2 mg/animal (=13 mg/kg), on day 50, 53, and 56 of age	98 days	No mammary tumours	n no/ld	Pataki & Huggins(1969)
TLC purified	Rat, Long-Evans	m	8	i.m.	2.5 mg/animal into hind leg, 1x on day 25 of age	270 days	No sarcomas; control: no spontaneous sarcomas	n no/ln	Pataki & Huggins(1969)
> 99%	Rat, Sprague-Dawley	f	20	Intra mammary injection	0.91 and 3.7 mg into 5th mammary gland, 1x	20 weeks	No mammary tumours; control: no tumours	n no/val	Cavalieri et al. (1988a)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benz[a]anthracene (contd)									
Chromatography control	Hamster, Syrian golden	m/f	5/5	Dermal	8 drops of a 0.5% solution, 2x/week,10 weeks	≤ 85/61 weeks	No tumours	n no/ln,ld	Shubik et al. (1960)
	Hamster, Syrian golden	m 26	5 or	Dermal (buccal pouch)	20 mmol/litre solution, painting 2x/week, 5 or 20 weeks	≤ 44 weeks	No tumours; control: no tumours	n **no/val**	Solt et al. (1987)
	Hamster Syrian golden	m	47 or 33	Intra-tracheal	0.5 or 3 mg/animal per week 30 or 15 weeks	≤ 110 weeks	No tracheal tumours; control: no tumours	n yes/**val**	Sellakumar & Shubik (1974)
Benzo[b]fluoranthene									
	Mouse, Swiss Millerton	f	20	Dermal	0.01, 0.1 and 0.5%, 3x/week, life	≤ 14, 12, and 8 months	0.01%: 5% papillomas after 14 months; 0.1%: 65% papillomas and 85% carcinomas after 12 months;0.5%: 100% carcinomas after 5 months	p no/ld	Wynder & Hoffmann (1959b)
	Mouse, Swiss ICR/Ha	f	20	Dermal	1 mg, 1x; initiation experiment	63 weeks	18/20 papillomas, 5/20 carcinomas; promotor only: 5/20, 1/20	p no/val	Van Duuren et al. (1966)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[b]fluoranthene (contd)									
> 96%	Mouse, NMRI	f	40	Dermal	3.4, 5.6, 9.2 µg/animal, 2x/week, life	≤ 2 years	5/15/54% with local tumours; control: no tumours	p yes/val	Habs et al. (1980)
	Mouse, CD-1			Dermal	10–100 µg/animal; initiation experiment	20 weeks	Dose-related skin tumour incidence	p yes/val	LaVoie et al. (1982b)
> 99%	Mouse, Cr1:CD-1 (ICR)BR	f	20	Dermal	4 and 10 nmol/animal, 10x every other day; initiation experiment	34 weeks	45 and 95% tumour incidence; solvent control: 5%	p yes/val	Amin et al. (1985a)
	Mouse, CD-1	f	20	Dermal	0.025 and 0.1 mg/animal, 10x every other day; initiation experiment	24 weeks	100 and 100% tumour incidence; solvent control: 10%	p yes/val	Weyand et al. (1990)
	Mouse, Cr1:CD1 (ICR)BR	f	20	Dermal	3 and 10 µg/animal, 10x; initiation experiment	34 weeks	65 and 100% with tumours; solvent control: 15%	p yes/val	Amin et al. (1991a)
	Mouse, XVII nc/Z	m/f	16/14	s.c.	0.6 mg/animal, 1x/month, 3 months	~ 200 days	8/16 m and 10/14 f with local sarcoma	p no/ld	Lacassagne et al. (1963a)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[b]fluoranthene (contd)									
> 99%	Mouse, CD-1 newborn	m/f	15/17	i.p.	126 µg/animal in DMSO on days 1,8, and 15 after birth (total dose)	≤ 52 weeks	53% hepatic, 18% lung tumours; control: 6% hepatic tumours, no lung tumours	p yes/**val**	LaVoie et al. (1987)
99.5%	Rat, Osborne/ Mendel	f	35	Intrapulm.	0.1, 0.3 and 1 mg/animal, 1x	110/113/ 112 weeks	0/35, 1/35, and 9/35 pulmonary carcinomas; 1/35, 2/35, and 4/35 pleomorphic sarcomas; control: no tumours	p yes/**val**	Deutsch-Wenzel et al.(1983)
	Hamster, Syrian golden	m	47	Intra-tracheal	0.5 and 0.5 mg/animal per week, 30 weeks	≤ 110 weeks	0/47 and 1/47 tracheal tumours; control: no tumours	n yes/**val**	Sellakumar & Shubik (1974)
Benzo[j]fluoranthene									
Highly purified	Mouse, Swiss	f	20	Dermal	0.1 and 0.5%, 3x/week, life	≤ 9 and 7 months	100%/95% with skin carcinomas	p no/ld	Wynder & Hoffmann (1959b)
96%	Mouse, NMRI	f	40	Dermal	3.4, 5.6, 9.2 µg/animal, 2x/week, life	≤ 2 years	3, 3, and 5% with local tumours;controls: 0%	q yes/**val**	Habs et al. (1980)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[j]fluoranthene (contd)									
> 99%	Mouse, Cr1:CD1 (ICR)BR	f	20	Dermal	3, 10, 100 µg, 10x over 20 days; initiation experiment	24 weeks	30, 55, and 95% with tumours (papillomas/keratinizing lesions);1 malignant lymphoma	p yes/**val**	LaVoie et al. (1982b)
	Mouse, CD-1	f	20	Dermal	25, 75 µg, 10x over 20 days; initiation experiment	24 weeks	70 and 90% with papillomas; vehicle control: 10%	p yes/**val**	Rice et al. (1987)
99%	Mouse, CD-1 newborn	m/f	21/18	i.p.	278 µg/animal in DMSO on days 1,8, and 15 after birth (total dose)	≤ 52 weeks	81% males and 22% females with liver and lung tumours; control: 6%:0%	p yes/**val**	LaVoie et al. (1987)
99.9%	Rat, Osborne/ Mendel	f	35	Intrapulm.	0.8, 4, and 20 mg/kg, 1x	110/117/ 89 weeks	1/35, 3/35 and 18/35 pulmonary carcinomas; control: no tumours	p yes/**val**	Deutsch-Wenzel et al.(1983)
Benzo[ghi]fluoranthene									
Highly purified	Mouse, Swiss	f	20	Dermal	0.1 and 0.5%, 3x/week, life	≤13 months	No skin tumours	n no/ld	Wynder & Hoffmann (1959b)
	Mouse, Swiss ICR/Ha	f	20	Dermal	1 mg, 1x; initiation experiment		4/20 papillomas, no carcinomas; promotor only: 5/20, 1/20	n no/**val**	Van Duuren et al. (1966)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[k]fluoranthene									
Highly purified	Mouse, Swiss	f	20	Dermal	0.1 and 0.5%, 3x/week, life	≤ 13 months	0/20 and 2/20 skin papillomas	q no/ld	Wynder & Hoffmann (1959b)
	Mouse, NMRI	f	25	Dermal	1 mg/animal (total dose) in 50 aliquots	≤ 2 years	No skin tumours; spontaneous tumours: 10%	n no/**val**	Mohr (1969)
> 96%	Mouse, NMRI	f	40	Dermal	3.4, 5.6, 9.2 µg/animal, 2x/week, life	≤ 2 years	3, 0 and 0% with local tumours; control: no tumours	n yes/**val**	Habs et al. (1980)
> 99%	Mouse, Cr1:CD1 (ICR)BR	f	20	Dermal	3, 10, 100 µg, 10x over 20 days; initiation experiment	24 weeks	5, 25, and 75% with tumours (papillomas/keratinizing lesions)	p yes/**val**	LaVoie et al. (1982b)
	Mouse XVII nc/Z	m/f	16/14	s.c.	0.6 mg/animal, 1x/month, 3 months	~200 days	8/16 m and 5/14 f with local sarcomas	p no/ld	Lacassagne et al. (1963a)
> 99%	Mouse, CD-1 newborn	m/f	16/18	i.p.	530 µg/animal in DMSO on days 1, 8, and 15 after birth (total dose)	≤ 52 weeks	19% males and 17% females with tumours; control: 6%:0% liver and lung tumours	q yes/**val**	LaVoie et al. (1987)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[k]fluoranthene (contd)									
99.5%	Rat, Osborne/ Mendel	f	27–35	Intrapulm.	0.65, 3.4, and 17 mg/kg, 1x	114/95/98 weeks	0/35, 3/31 and 12/27 pulmonary carcinomas; control: no tumours	p yes/**val**	Deutsch-Wenzel (1983)
Benzo[a]fluorene									
	Mouse, stock'		20	Dermal	0.3%, 2x/week, life	≤ 20 months	No skin tumours; 4/20 lung adenoma; 1/20 sebaceous adenoma	q no/ld	Badger et al. (1942)
> 99.5%	Mouse, Swiss Ha/ICR	f	20	Dermal	100 µg, 10x over 20 days; initiation experiment	24 weeks	2/20 skin tumours; control: 1/20	n yes/**val**	LaVoie et al. (1981c)
	Mouse, stock'		10	s.c.	5 mg/animal, at intervals of a few weeks, life	≤ 23 months	1/10 lung adenoma; no sarcomas	n no/ln,ld	Badger et al. (1942)
Benzo[b]fluorene									
99.5%	Mouse, Swiss Ha/ICR	f	20	Dermal	100 µg, 10x over 20 days; initiation experiment	24 weeks	4/20 skin tumours; control: 1/20	q yes/**val**	LaVoie et al. (1981c)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[ghi]perylene									
	Mouse, Swiss	f	50	Dermal	0.38% solution in benzene, 3x/week, life		2/50 with skin tumours; control: 1/59 skin carcinomas	n no/val	Lijinsky & Saffiotti (1965)
Chromatography purified	Mouse, Swiss Ha/ICR/Mil	f	20	Dermal	0.05% and 0.1%, 3x/week, 12 months	15 months	1/20 and 0/20 skin papillomas; solvent control: no tumours	n no/val	Hoffmann & Wynder (1966)
Chromatography purified	Mouse, Swiss Ha/ICR/Mil	f	30	Dermal	25 µg/animal, 10x over 28 days; initiation experiment	6 months	2/30 papillomas; control: 2/30 0/20	n no/val	Hoffmann & Wynder (1966)
	Mouse, NMRI	f	50	Dermal	20 µg, 2 mg and 4 mg/ animal, 2x/week, 25 weeks	≤ 22.5 months	3/50, 6/50, and 4/50 with tumours; vehicle control: 7/50	n no/val	Müller (1968)
	Mouse, NMRI	f	50	Dermal	1 and 2 mg/animal, 1x; initiation experiment	≤ 22.5 months	5/50 and 4/50 with tumours; vehicle control: 7/50	n no/val	Müller (1968)
Rigourously purified	Mouse, Swiss	f	20	Dermal	0.8 mg/animal, 1x; initiation experiment	12–13 months	3/20 papillomas, 1/20 squamous-cell carcinoma; vehicle control: 1/20 with 2 papillomas	n no/val	Van Duuren et al. (1970)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[ghi]perylene (contd)									
Highly purified	Mouse, ICR/Ha	f	50	Dermal	5.5 and 16.5 µg/animal, 3x/week, 33weeks	33 weeks	No tumours	n no/lc	Goldschmidt et al. (1973
Highly purified	Mouse, Swiss ICR/Ha	f	50	Dermal	21 µg/animal, 3x/week, 52 weeks	52 weeks	No skin tumours; solvent control: no tumours	n no/val	Van Duuren & Goldschmidt (1976)
	Mouse, NMRI	f	50	s.c.	0.83 and 16.7 mg/animal, 1x/2 weeks, 6 months	≤ 22.5 months	5/50 and 4/40 with tumours; control:4/50	n no/val	Müller (1968)
	Mouse, NMRI	f	20	s.c.	0.1, 1 and 10 mg/animal, 1x/2 weeks, 20 weeks	≤ 22 months	No skin or subcutaneous tumours;other tumours same as control	n no/val	Müller (1968)
98.5%	Rat, Osborne/ Mendel	f	34-35	Intrapulm.	0.65, 3.4, and 17 mg/kg, 1x	109/114/ 106 weeks	0/35, 1/35 and 4/34 pulmonary carcinomas effect of impurity suggested; control: no tumours	n yes/val	Deutsch-Wenzel et al.(1983)
Benzo[c]phenanthrene									
	Mouse,		20	Dermal	Not specified	≤ 676 days	7 epitheliomas, 5 papillomas	p no/ld	Barry et al. (1935)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[c]phenanthrene (contd)									
	Mouse,		40	Dermal	0.3%, 2x/week, ≤19 months	≤ 19 months	1 papilloma, 4 squamous cell	q no/ld	Badger et al. (1940)
	Mouse, C3H		20	Dermal	0.5% solution, 2x/week, 638 days	638 days	3 carcinomas, 2 sarcomas; control: no tumours	q no/**val**	Stevenson & von Haam (1965)
	Mouse, CD-1	f	30	Dermal	91 and 457 µg, 1x; initiation experiment	21 weeks	5/30 and 11/30 papillomas; control:no tumours	p	Levin et al. (1980)
	Mouse		10	s.c.	5 mg at intervals of several weeks, life	≤ 15 months	No injection-site tumours	n no/ln,ld	Badger et al. (1940)
	Mouse C3H		20	s.c.	5 mg in tricaprylin, 1 x	638 days	3 sarcomas; controls: no tumours	q no/**val**	Stevenson & von Haam (1965)
	Rat,		6	s.c.	5 mg/animal; several repeated doses	~18 months	1/6 sarcoma at injection site	q no/ln,ld	Badger et al. (1940)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene									
	Mouse, A/HeJ	f	15	Oral	3 mg/animal in sesame oil, 2x	30 weeks	Increased pulmonary tumours (16.6); control: 0.3	p yes/**val**	Wattenberg & Leong (1970)
	Mouse, A/J	f	15	Oral	2 mg/animal, 3x, every 2 weeks	26 weeks	15/15 with forestomach tumours and 15/15 with pulmonary adenomas;no control	p yes/**val**	Sparnins et al. (1986)
	Mouse, CFW	m/f	25–73	Oral (diet)	0.004–1 mg/animal per day, ≤110–165 days	140–200 days	Dose-dependent gastric tumours (0–90%);control: no tumours	p no/**val**	Neal & Rigdon (1967)
	Mouse, CFW	m/f	9–26	Oral (diet)	1–20 mg/animal/day, ≤ 1–30 days	150–300 days	Dose-dependent gastric tumours(0–100%); control: no tumours	p no/**val**	Neal & Rigdon, (1967)
	Mouse, White Swiss	m/f	60–175	Oral (diet)	0.25 and 1 mg/g food, ≤ 34 weeks	≤ 34 weeks	33 and 61% with stomach tumours; 53 and 20% with lung tumours;controls: 1 and 21%	p no/**val**	Rigdon & Neal (1966)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Mouse, Swiss	f	20–30	Dermal	0.001, 0.005, and 0.01%, 3x/week, life	≤ 21, 14, and 11	3 and 43%, 63 and 73%, and 95% and 95% with skin carcinomas/papillomas	p no/ld	Wynder & Hoffmann (1959a)
	Mouse, Swiss Millerton	f	20	Dermal	0.01, 0.05 and 0.5%, 3x/week, life months	≤12, 6, and 6	85, 95, and 75% with skin carcinomas	p no/ld	Wynder & Hoffmann (1959b)
Recrystal-lized	Mouse, Swiss Ha/ICR/Mil	f	20	Dermal	0.05 and 0.1%, 3x/week, 12 months	15 months	17/20 and 19/20 skin tumours; solvent control: no tumours	p no/**val**	Hoffmann & Wynder (1966)
	Mouse, Swiss Ha/ICR/Mil	f	30	Dermal	25 µg/animal, 10x over 28 days; initiation experiment	6 months	24/30 papillomas; promotor only: 2/30	p no/**val**	Hoffmann & Wynder (1966)
	Mouse, NMRI	f	50	Dermal	20 and 200 µg/animal, 2x/week, 25 weeks	≤ 22.5 months	50/50 and 50/50 with skin tumours;vehicle control: 7/50	p no/**val**	Müller (1968)
	Mouse, C3H/He		20–30	Dermal	(a) 0.00002% in *n*-dode-cane/decalin; (b) 0.02% in decalin, 3x/week, 50 weeks		(a) 21% malignant tumours; (b) 50% tumours (three orders of magnitude difference in dose)	p yes/**val**	Bingham & Falk (1969)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Mouse, Swiss Ha/ICR/Mil	f	30	Dermal	5 µg/animal, 10x, 20 days; initiation experiment	24 weeks	19/29 tumour-bearing animals with 67 skin tumours; control: 1/30	p no/**val**	Hoffmann et al. (1972)
	Mouse, Swiss ICR	f	20	Dermal	0.05 and 0.1 mg/animal; 60x	6 months	13 and 18 with skin tumours; no solvent control	p no/**val**	Masuda & Kagawa (1972)
	Mouse, Swiss Ha/ICR/Mil	f	20	Dermal	5 µg/animal, 3x/week, 72 weeks	≤ 72 weeks	13/20 with 22 skin tumours; 4/20 with 4 carcinomas; solvent control: no tumours	p no/**val**	Hecht et al. (1974)
	Mouse, Swiss ICR/Ha	f	50	Dermal	5 µg/animal, 3x/week, life	440 days	16 animals with 26 tumours; control: no tumours	p no/**val**	Van Duuren & Goldschmidt (1976)
	Mouse, Swiss Ha/ICR	f	20	Dermal	5 and 10 µg/animal, 3x/week, 62 weeks	62 weeks	Low dose: 10/20 with 19 skin tumours, 7/20 with 8 carcinomas; high dose:18/20 with 70 skin tumours, 14/20 with 16 carcinomas; solvent control: no tumours	p yes/**val**	Hecht et al. (1976b)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
99.9%	Mouse, Swiss	f	40	Dermal	100 µg/animal, 2x/week, 30 weeks	70 weeks	79% skin-tumour bearing animals;solvent control: no tumours	p no/**val**	Cavalieri et al. (1977)
	Mouse, NMRI	f	40	Dermal	1.7, 2.8, 4.6 µg/animal, 2x/week, life	≤ 2 years	24, 69 and, 61% with local tumours(high rate of systemic tumours);control: no tumours	p yes/**val**	Habs et al. (1980)
> 99.5%	Mouse, Swiss Ha/ICR	f	20	Dermal	30 µg/animal, 10x on alternate days; initiation experiment	24 weeks	93% with tumours; vehicle control: no tumours	p no/**val**	LaVoie et al. (1981b)
	Mouse, Cr1:CD1 (ICR)BR	f	20	Dermal	3 µg, 10x over 20 days; initiation experiment	24 weeks	85% with tumours (papillomas/keratinizing lesions)	p yes/**val**	LaVoie et al. (1982b)
> 96%	Mouse, NMRI	f	20	Dermal	2 and 4 µg/animal, 2x/week,life	648 and 528 days (mean)	45% (10% papillomas/35% carcinomas) and 85% (0%/85%) with skin tumours; control: no tumours	p yes/**val**	Habs et al. (1984)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
99.5%	Mouse, CH3/HeJ	m	50	Dermal	12.5 µg/animal, 2x/week, 99 weeks	≤ 99 weeks	94% with malignant skin tumours;solvent control: no tumours; untreated control: no tumours	p no/**val**	Warshawsky & Barkley (1987)
	Mouse, Sencar	f	24	Dermal	0.8 µmol/mouse, 1x; initiation experiment	24 weeks	Enhanced incidence of skin papillomas (80-92%)	p no/**val**	Cavalieri et al. (1988b)
	Mouse, Swiss	m	12	Dermal	1.2 mg/animal, 6 days/wk, 19 weeks	≤ 27 weeks	Multiple tumours; squamous-cell carcinomas	p no/**In**	Shubik & Della Porta (1957)
> 99%	Mouse, CD-1	f	20	Dermal	2.5 µg/animal, 10x over 20 days; initiation experiment	24 weeks	89% tumours, 5.5 skin tumours/animal; control: 5%	p yes/**val**	Rice et al. (1988b)
> 99%	Mouse, CD-1	f	25	Dermal	2.5 µg/animal, 10x over 20 days; initiation experiment	23 weeks	96% tumours, 3.4 skin tumours/animal; control: no tumours	p yes/**val**	Rice et al. (1990)
Chromato-graphy purified	Mouse, Sencar	f	23-24	Dermal	8.4, 25.2 and 75.7 µg/ animal, 1x; initiation experiment	15 weeks	10/23, 17/24 and 21/23 with tumours; control: no tumours	p yes/**val**	Cavalieri et al. (1991)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
Chromatography purified	Mouse, Sencar	f	24	Dermal	1, 5 and 25 µg/animal, 1x; initiation experiment	27 weeks	1/24, 10/24 and 22/24 with tumours; control: no tumours	p yes/val	Cavalieri et al. (1991)
Chromatography purified	Mouse, Sencar	f	24	Dermal	25 µg/animal, 1x; initiation experiment without promotion	27 weeks	1/24 with tumours	p yes/val	Cavalieri et al. (1991)
HPLC control	Mouse, ICR/Harlan	f	43-50	Dermal	16, 32, or 64 µg/animal, 1x/week, 29 weeks	≤ 35 weeks	1, 1.5 and 7.5 tumours/animal after 35 weeks	p no/val	Albert et al. (1991a)
	Mouse, Balb/c	m	20	Dermal	100 µg/animal, 2x/week, 3 weeks-5 months		Tumours from 15 weeks onwards	p no/val	Andrews et al. (1991)
	Mouse, C57Bl	m/f	40-50	s.c.	0.09 mg/animal in tricaprylin; 1x	≤ 22-28 months	16/21 sarcomas after 5 months	p yes/ld	Steiner (1955)
	Mouse, XVII	m/f	14/16	s.c.	0.6 mg/animal, 1x/month, 3 months	> 129/160 days (average latency)	13/14 m and 8/16 f with local sarcomas	p no/ld	Lacassagne et al. (1958)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Mouse, XVII nc/Z	m/f	154/ 162	s.c.	0.6 mg/animal, 1x/month, 3 months	~110/150 days	154/154 m and 112/162 f with local sarcomas	p no/ld	Lacassagne et al. (1963a)
	Mouse, NMRI		20	s.c.	0.1, 1 and 10 mg/animal, 1x/2 weeks, 20 weeks	17, 7, 6 months	All animals with sarcomas at injection site	p no/val	Müller (1968)
	Mouse, NMRI	f	90	s.c.	25, 50, 100, 200 and 400 µg/animal, 1x	≤16 months	25, 50, 55, 75 and 65% with tumours; solvent control: < 5%	p no/val	Pott et al. (1973)
	Mouse, newborn	m/f	31-38	s.c.	0.01 and 0.1 mg/animal, 1x	30 weeks	16 and 64% with lung tumours; control: 13% with lung tumours	p no/val	Rippe & Pott (1989)
> 99%	Mouse, CD-1 newborn	m/f	17/14	i.p.	278 µg/animal in DMSO on days 1, 8, and 15 after birth (total dose)	≤ 52 weeks	76% hepatic and 64% lung tumours; control: 6% hepatic tumours, no lung tumours	p yes/val	LaVoie et al. (1987)
> 99%	Mouse, Swiss-Webster BLU:Ha(ICR), newborn	m/f	28/27	i.p.	59.5 µg/animal on days 1, 8, and 15 after birth (total dose)	26 weeks	46 m, 70, f with lung tumours; vehicle control: 14 m, 7 f	p yes/val	Busby et al. (1989)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Mouse, A		10	i.v.	10 mg/kg, 1x	20 weeks	100% lung tumours; control: 21%	p no/**val**	Shimkin & Stoner (1975)
	Mouse, NMRI	f	19-22	Intra-tracheal	0.05 and 0.15 mg/animal, 20x		27 and 42% with carcinomas in the respiratory tract; control: 9%	p no/**val**	Pott et al. (1978)
99%	Mouse, ICR/Ha	f	45; 60 contr.	Intra-colonic instill-ation	1 mg/animal, 1x/week, 14 weeks	≤ 18 months	No colonic tumours, 73% lung tumours, 94% forestomach umours, 7% subcutaneous sarcomas, 23% mammary carcinomas; control: 25% lung tumours, 20% forestomach tumours, 9% mammary carcinomas, no subcutaneous sarcomas or colonic tumours	p yes/**val**	Anderson et al. (1983)
99%	Mouse, C57Bl/6	f	38; 45	Intra-colonic instill-ation	1 mg/animal, 1x/week, 14 weeks	≤ 18 months	No colonic tumours, 94% fore-stomach tumours, 16% peritoneal sarcomas, 28% lymphomas; control: 21% forestomach tumours, no sarcomas or lymphomas or colonic tumours	p yes/**val**	Anderson et al. (1983)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Rat, Sprague-Dawley	f	9	Oral	100 mg/kg, 1x	60 days	8/9 with mammary tumours; control: 8/164 in 310 days	p no/ln,lc	Huggins & Yang (1962)
	Rat, LEW/Mai	f	20	Oral	6.25 mg/animal, 1x/week, 8x; 50 mg/animal, 1x	90 weeks	67–77% with mammary tumours; control: 30%	p yes/val	McCormick et al. (1981)
	Rat, Wistar	f	50	s.c.	33, 100, 900, and 2700 µg/animal, 1x	≤16 months	10,15, 70 and 75% with tumours; solvent only: < 5%	p no/val	Pott et al. (1973)
	Rat, Wistar	f	37	i.p.	5 mg/animal; 1x; in bees' wax/tricaprylin 25/75 (a) or saline (b)	2 years	(a) 89% abdominal tumours (mesotheliomas, sarcomas); (b) 50%; vehicle controls: (a) 7%; (b) 3%	p no/val	Roller et al. (1992)
	Rat, Wistar	f	13–17	Intra-tracheal	0.5, 1, or 2 mg/animal in infusion; 1x/2 weeks; 18x	Life	7, 65, and 92% with lung tumours; control: no tumours	p yes/val	Davis et al. (1975)
	Rat, Wistar	m/f	15/15	Intra-tracheal	1 mg/animal, 1x/week, 15x	Life (mean, 491/540 days)	3/13 (m) and 3/14 (f) with malignant lung tumours (mean: 22.2%); vehicle control: 0%	p no/val	Ishinishi et al. (1976)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Rat, Wistar	f	36–40	Intra-tracheal	1 mg/animal; 20x		19% with lung tumours; control: no tumours	p no/val	Pott et al. (1987)
	Rat, Sprague-Dawley	m/f	20/20	Intra-tracheal	7 mg/kg, every 14 days, 22x (total dose: 154 mg/kg)	≤ 781 days	19/20 m and 18/20 f with lung tumours; vehicle control: 0%	p no/val	Steinhoff et al. (1991)
99.1%	Rat, Osborne/ Mendel	f	35	Intrapulm.	0.1, 0.3, or 1 mg/animal, 1x	111/77/54 weeks	4/35, 21/35 and 33/35 pulmonary carcinomas; 6/35, 2/35 and 0/35 pleomorphic sarcomas; control: no tumours	p yes/val	Deutsch-Wenzel et al.(1983)
	Rat, Osborne/ Mendel	f	35	Intrapulm.	0.05, 0.1,or 0.2 mg/animal, 1x		11, 17, and 46% with tumours; control: no tumours	p yes/val	Grimmer et al. (1987)
99.6%	Rat, Osborne/ Mendel	f	35	Intrapulm.	0.03, 0.1, or 0.3 mg/ animal,1x	≤ 135 weeks	8.6, 31.4, and 77.1% tumour incidence; control: no tumours	p yes/val	Wenzel-Hartung et al. (1990)
	Rat, Fischer 344/Du Crj	m	14–15	Intrapulm.	50, 100, or 200 µg/animal	≤100 weeks	0/10, 3/10 and 4/9 lung tumours; control: no tumours	p no/val	Horikawa et al. (1991)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Rat,		94	Intrabr-onchial pellet	~3–5 mg/animal, 1x	~5 months	Carcinoma incidence: 17%	p no/lc	Laskin et al. (1970)
> 99%	Rat, Sprague-Dawley	f	20	Intra-mammary	1 and 4 mg into 5th mammary gland,1x	20 weeks	50 and 80% with mammary tumours; control: no tumours	p no/**val**	Cavalieri et al. (1988a)
	Rat, Sprague-Dawley	f	20	Intra-mammary injection	63 and 252 µg/gland, 1x, 8 glands	≤ 24 weeks	7/20 and 9/20 with mammary tumours; control: 1/18	p yes/**val**	Cavalieri et al. (1991)
Recrystal-lized	Rat, Fischer 344	f	20	Pellet implant-ation	0.5 and 1 mg; 1x (implanted into tracheal transplants)	28 months	12 and 65% with carcinomas in tracheal transplants	p yes/**val**	Topping *et al.* (1981)
	Hamster, Syrian golden	*m/f*	13	Oral (diet)	2.5 mg/animal/day, 4 days/week, ≤14 months	≤ 14 months	9/13 with forestomach cancer; 2/13 with papillomas	p no/**val**	Chu & Malmgren (1965)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
Chromatography control	Hamster, Syrian golden	m/f	15/15	Dermal	4 drops of a 0.8% solution in mineral oil, 1x/week, 8 weeks including a 30-week interval	≤ 99/68 weeks	m:1 small nodular melanotic lesion, 2 malignant lymphomas ; f: no tumours	q no/ln,ld	Shubik et al. (1960)
Chromatography control	Hamster, Syrian golden	m/f	5/5	Dermal	6 drops of 0.01% solution in acetone, 2x/week, 40 weeks	≤ 70 weeks	No skin tumours	n no/ln,ld	Shubik et al. (1960)
	Hamster, Syrian golden	m 28	5 or	Dermal (buccal pouch)	20 mmol/litre solution, painting 2x/week, 5 or 20 weeks	≤ 44 weeks	10% buccal pouch carcinomas after 40 weeks; control: no tumours	p no/val	Solt et al. (1987)
	Hamster, Syrian golden	m	10	Inhalation	4.5 h/day, 5 days/week, 9.8 mg/m³ for 16 weeks or 44.8 mg/m³ for 10 weeks	Life	No tumours	n no/ld	Thyssen et al. (1980)
	Hamster, Syrian golden	m	24	Inhalation	2.2, 9.5, or 46.5 mg/m³, 4.5 h/day in the first 10 weeks, thereafter 3 h/day, 109 weeks	109 weeks	Dose-dependent tumours in nasal cavity, pharynx, larynx, and trachea;also in oesophagus and forestomach (papillomas, polyps, squamous-cell carcinomas); no lung tumours; larynx most affected with 0, 31 and, 52% incidence; control: no tumours	p no/val	Thyssen et al. (1981)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
Pure	Hamster, Syrian golden	m/f	30/30	Intra-tracheal	3 mg/animal, 1x/wk, 15 weeks (mixed with inert dust of haematite [ferric oxide]	≤ 45/60 weeks	14/19 and 21/21 with tumours in respiratory tract; control: no tumours	p **no/val**	Saffiotti et al. (1968)
	Hamster, Syrian golden	m	30	Intra-tracheal	3 mg/animal, 1x/week, 14 weeks	≤ 74 weeks	All with bronchioalveolar metaplasia; 5/19 squamous-cell carcinomas, 3/19 adenomas, 1/19 tracheal tumours	p no/val	Crocker et al. (1970)
Pure	Hamster, Syrian golden	m/f	30-50	Intra-tracheal	0.25, 0.5, 1, or 2 mg/animal, 1x/week, 30 wks (mixed with inert dust of ferric oxide)	Life	Dose-related increase in respiratory tract tumours; control: no tumours	p no/val	Saffiotti et al. (1972)
Pure	Hamster, Syrian golden	m	30	Intra-tracheal	0.0625, 0.125, 0.25, 0.5, and 1 mg/animal, 1x/week, 52 weeks	78 weeks	Dose-related increase in respiratory tract tumours (3–26%); controls: no tumours	p no/val	Feron et al. (1973)
	Hamster, Syrian golden	m/f	25/25	Intra-tracheal	0.9 mg/animal per week, 30 weeks	≤100 weeks	17% (8/46) tumours in respiratory tract; control: no tumours	p no/val	Henry et al. (1975)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Hamster, Syrian golden			Intra-tracheal	0.3 or 0.9 mg/animal, 1x/week,20 weeks	≤ 2 years	17 and 68% with tumours	p no/lc	Pott et al. (1978)
	Hamster, Syrian golden	m	29	Intra-tracheal	0.125, 0.25, 0.5, or 1 mg/animal,1x/week, life	Life	31, 83, 66, and 31% tumours in respiratory tract; control: no tumours	p no/val	Ketkar et al. (1979)
	Hamster, Syrian golden	m	30	Intra-tracheal	5, 20 or 40 µg/animal, every 2 weeks, life	Life	4/28, 5/27, and 7/28 with meta plasia in respiratory tract, malignant neoplasm and 1 adenoma in high-dose group; controls: 1/29 or 3/30	q no/val	Kunstler (1983)
	Hamster,		97	Intra-bronchial pellets	3-5 mg		63/97 with lung cancers	p no/val	Laskin et al. (1970)
	Hamster, Syrian golden			Tracheal insufflation	~0.83 mg/animal, 3x/week, 1 year		Tracheal papillomas and carcinomas	p no/lc	Mohr (1971)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Hamster, Syrian golden			Bronchial implants		150 days	> 90% with focal cancers	p no/lc	Benfield & Hammond (1992)
	Dog			Parenchymal implants		> 8 months	First parenchymal cancer after 8 months; 7/12 dogs with tumours	p no/lc	Benfield & Hammond (1992)
	Pig, German Edelland-schwein	m/f	1/1	i.m.	4.8 mg/kg, 1x; 6 months later 2.1 mg/kg, 1x	12 months	No sarcomas	n no/**val**	Kallistratos & Pfau (1971)
	Pig, mini	m/f	1/1	i.m.	6.3 mg/kg, 1x; 6 months later 1.9 mg/kg, 1x	12 months	No sarcomas	n no/**val**	Kallistratos & Pfau (1971)
	Cattle, German black/white	m/f	1/1 later	i.m.	0.95 mg/kg, 1x; 6 months 0.75 mg/kg, 1x	29 months	No sarcomas	n no/**val**	Kallistratos & Pfau (1971)
	Monkeys (a) *Saguinus / oedipus*; (b) *S. fusciocollis*	m/f	m1/1 f 1/1	s.c.	10 mg/animal, 1x (coadministration with 10 mg DMBA at other site)	(a) > 18 months (b) ≤ 5 weeks	(a) 1/2 with local tumours (b) death within 5 weeks	q no/ln	Noyes (1969)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[a]pyrene (contd)									
	Monkey, *Galago crassus*			s.c.	1x (not specified)		Fibrosarcomas	p no/lc	Adamson & Sieber (1983)
	Monkey, Old world		17	s.c.	30–90 mg/kg, multiple administration (not specified)	≤ 18 years	No tumours observed; survival: 9/17	n no/lc	Adamson & Sieber (1983)
	Monkey, *Galago crassust*	m/f	4/2	Intra-tracheal	3–15 mg, 1x/week (with ferric oxide), up to 69 weeks	67–69 weeks	Bronchioalveolar metaplasia; 2/3 squamous carcinomas arising from bronchus	p no/**val**	Crocker et al. (1970)
Benzo[e]pyrene									
	Mouse, Swiss Millerton	f	20	Dermal	0.1%, 3x/week, life	≤ 13 months	2/20 papillomas, 3/20 carcinomas	q no/ld	Wynder & Hoffmann (1959a)
	Mouse, Swiss ICR/Ha	f	20	Dermal	1 mg, 1x; initiation experiment	64 weeks	2/20 with papillomas; pure substance: no tumours	q no/**val**	Van Duuren et al. (1968)
TLC purified	Mouse, CD-1	f	20	Dermal	2.5 mg/animal, 1x; initiation experiment	35 weeks	85% with papillomas; promotor only: 3%	p no/**val**	Scribner (1973)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[e]pyrene (contd)									
Highly purified	Mouse, ICR/Ha	f	50	Dermal	15 µg/animal, 3x/week, 368 days	368 days	No tumours observed; control: no tumours	n no/**val**	Van Duuren & Goldschmidt (1976)
> 99%	Mouse, CD-1	f	30	Dermal	100 µg/animal, 2x/week, 30 weeks	30–40 weeks	At 30 wks: 68% papillomas, at 40 weeks: 24% carcinomas	p no/**val**	Slaga et al. (1979)
> 99%	Mouse, CD-1	f	30	Dermal	100 and 252 µg/animal, 1x; initiation experiment	30–40 weeks	High dose: at 30 weeks, 19% papillomas; at 40 weeks, no carcinomas;vehicle control: at 30 weeks, 14% papillomas	q no/**val**	Slaga et al. (1979)
99%	Mouse, CD-1	f	30	Dermal	0.25, 0.63, or 1.5 mg/ animal, 1x;initiation experiment	26 weeks	15, 11, or 14% with papillo-mas; vehicle control: 7% papillomas	q no/**val**	Buening et al. (1980)
> 95%	Mouse, Sencar	f	30	Dermal	0.5 mg/animal, 1x; initiation experiment	15 weeks	17% with papillomas; vehicle control:10%	q no/**val**	Slaga et al. (1980, 1981)
99%	Mouse, Swiss-Webster BLU:Ha(ICR), newborn	m/f	30/30	i.p.	0.1, 0.2, 0.4, or 0.2, 0.4, 0.8 mg on days 1, 8, and 15 of life	62–66 weeks	21/35 (m), 0/35 (f) or 12/30 (m), 0/30 (f) with hepatic tumours; controls:11/53 (m), 0/24 (f)	q no/**val**	Buening et al. (1980)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Benzo[e]pyrene (contd)									
99.7%	Rat, Osborne/ Mendel	f	30-35	Intrapulm.	0.8, 4.2, or 20 mg/kg, 1x	117/111/ 104 weeks	1 pulmonary sarcoma at 4.2 mg/kg; 1 squamous-cell carcinoma at 20 mg/kg; no tumours in controls	n yes/**val**	Deutsch-Wenzel et al.(1983)
Recrystal-ized	Rat, Fischer 344	f	20	Tracheal pellet	1 mg, 1x	28 months	No tumours	n yes/**val**	Topping et al. (1981)
Chrysene									
	Mouse		100	Dermal	1% in 90% benzene	≤ 11 months	No tumours	n no/ld,lc	Kennaway (1924)
Purified	Mouse			Dermal	7.5% in liquid paraffin or oleic acid, 5x/week, 78 or 50 weeks	78 or 50 weeks	6 or 18 benign, 1 or 9 malig-nant tumours	q no/lc	Bottomley & Twort (1934)
Doubtful purity	Mouse		100 20	Dermal	(a) 0.3% in benzene or (b) 0.3% in mouse fat, 2 x/week, life	≤ 704 days	(a) 1/100 papilloma and 1/100 epithelioma, (b) no tumours	n no/ld	Barry et al. (1935)
'Synthe-sized'	Mouse		20	Dermal	0.3% (pure), 2x/week, 440 days	440 days	No tumours	n no/lc,ld	Barry et al. (1935)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Chrysene (contd)									
	Mouse		50 100	Dermal	(a) 0.3% in benzene, (b) 7.5% in oleic acid, 2x/week, life	≤ 797 days	(a) 2/50 papillomas, (b) no tumours	n no/lc,ld	Barry et al. (1935)
'Pure'	Mouse		50	Dermal	In benzene, 2x/week, 276 days	≤ 276 days	After 276 days at 11/50 survivors, no tumours	n no/lc,ld	Schürch & Winterstein (1935)
	Mouse, CF1	m/f	10/10	Dermal	40 µg/animal, 2x/week, 31 weeks	31 weeks	1/15 carcinomas	n no/ld	Riegel et al. (1951)
	Mouse, Swiss	f	20	Dermal	1%, 3x/week, life	≤ 12 months	9/20 papillomas, 8/20 carcinomas; no solvent control	p no/ld	Wynder & Hoffmann (1959a)
	Mouse, Swiss ICR/Ha	f	20	Dermal	1 mg, 1x; initiation experiment	63 weeks	16/20 papillomas, 2/20 carcinomas; promotor only: 5/20, 1/20	p no/**val**	Van Duuren et al. (1966)
TLC purified	Mouse, CD-1	f	30	Dermal	1 mg/animal, 1x; initiation experiment	35 weeks	73% with papillomas; promotor only:3%	p no/**val**	Scribner (1973)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex/ group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Chrysene (contd)									
	Mouse, C3H	m	20	Dermal	75 µg/animal in decalin, 2x/week, 82 weeks	82 weeks	1/12 papillomas; solvent control:2/13 papillomas	n no/ld	Horton & Christian (1974)
	Mouse, C3H	m	20	Dermal	75 µg/animal in decalin/ dodecane 50/50, 2x/week, 82 weeks; co-carcinogenicity experiment	82 weeks	5/19 papillomas; 12/19 carcinomas; solvent control: 2/13 papillomas	p no/**val**	Horton & Christian (1974)
> 99.9%	Mouse, Swiss Ha/ICR/Mil	f	20	Dermal	0.1 mg/animal/day, 10x; initiation experiment	22 weeks	11/18 papillomas/carcinomas; chrysene only: 4/11 after 72 weeks; solvent control: no tumours	p no/**val**	Hecht et al. (1974)
	Mouse, CD-1	f	30	Dermal	0.09, 0.29 and 0.91 mg/ animal, 1x; initiation experiment	26 weeks	25, 43 and 52% papillomas; promotor only: 7%	p no/**val**	Levin et al. (1978)
95%	Mouse, CD-1	f	30	Dermal	0.46 mg/animal, 2x; initiation experiment	26 weeks	21/30 papillomas; promotor only: 1/30	p no/**val**	Wood et al. (1979)
98%	Mouse, CD-1	f	30	Dermal	0.57 mg/animal, 1x; initiation experiment	27 weeks	80% papillomas; promotor only: 4%	p yes/**val**	Wood et al. (1980)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Chrysene (contd)									
> 95%	Mouse, Sencar	f	30	Dermal	0.46 mg/animal, 1x; initiation experiment	15 weeks	21/29 papillomas; promotor only: 3/30	p no/**val**	Slaga et al. (1980, 1981)
	Mouse, CD-1	f	30	Dermal	0.09 and 0.274 mg/animal, 1x; initiation experiment	26 weeks	43, 43% (or 39%) with skin papillomas; vehicle control: 10%	p yes/**val**	Chang et al. (1983)
> 99%	Mouse, CD-1	f	20	Dermal	3.4, 11.4 and 34 µg/ animal, 10x over 20 days; initiation experiment	24 weeks	25, 90 and 95% with tumours; 0.5, 3, and 4.5 skin tumours/ animal; control: 20%	p yes/**val**	Rice et al. (1988b)
	Mouse, CD-1	f	20	Dermal	7.5 µg/animal, 1x; initiation experiment	21 weeks	10% with skin tumours; solvent control:10%	n yes/**val**	Amin et al. (1990)
	Mouse, Sencar	m/f	16/16	Dermal	365 µg/animal, 1x; initiation experiment	≤ 100 weeks	No skin tumours; solvent control: no tumours	n no/**val**	Bhatt & Coombs (1990)
Purified	Mouse		50	s.c.	2 mg/animal, 1x	≤ 35 weeks	No tumours	n no/lc	Bottomley & Twort (1934)
Purified	Mouse, Jackson A		30	s.c.	10 mg/animal, 2x (4-month interval)	15 months	No tumours	n no/ld	Shear & Leiter (1941)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Chrysene (contd)									
Spectrometer control	Mouse, C57Bl	m/f	50	s.c.	5 mg/animal in tricaprylin; 1x	≤ 22 months	4/39 sarcomas after 4 months; solvent control: 3/280	p no/**val**	Steiner & Falk (1951)
	Mouse, C57Bl	m/f	40–50	s.c.	5 mg/animal in tricaprylin; 1x	≤ 22–28 months	5/22 sarcomas after 5 months	p yes/ld	Steiner (1955)
	Mouse, C57Bl	m	20	s.c.	1 mg/animal in arachis oil, 1x/week, 10 weeks	60–80 weeks	2/20 injection site tumours; control: no tumours	p no/**val**	Boyland & Sims (1967)
	Mouse, Swiss newborn	m/f	104	s.c.	0.1 mg/animal in polyethylene glycol on days 1,2 and 3 after birth	70–75 weeks	70 weeks: 13/27m liver, 1/27 m and 1/21 f lung tumours; vehicle control: 9/30 m liver, 3/30 m and 1/15 f lung tumours	q no/**val**	Grover et al., (1975)
	Mouse,		10	s.c.	1 mg, weekly; later 2 mg at longer intervals	350 days	No tumours; control: no tumours	n no/ln,ld	Barry & Cook (1934)
Purified	Mouse		50	i.p.	2 mg/animal, 1x	≤ 45 weeks	No tumours	n no/lc	Bottomley & Twort (1934)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Chrysene (contd)									
TLC control	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	100	i.p.	Total dose 0.32 mg/animal in DMSO on days 1, 8 and 15 after birth	38–42 weeks	5/24 m and 2/11 f pulmonary tumours; 6/24 m liver tumours; 1/24 m lymphosarcoma; control: 2/21 m and 7/38 f lung tumours	q yes/**val**	Buening et al. (1979)
Repurified, 256 °C	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	80	i.p.	0.045, 0.09 and 0.18 mg/animal in DMSO on days 1, 8 and 15 after birth	39–41 weeks	Males: 4/27 lung and 6/27 liver tumours; females; 1/11 lung and 0/11 liver tumours; vehicle control: no tumours	p yes/**val**	Chang et al. (1983)
> 98%	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	20-29	i.p.	6.3 and 210 µg/animal (total dose) in 3 aliquots on day 1, 8, and 15 after birth	26 weeks	7/10% and 15/0% m/f with lung tumours; vehicle control: 14/7% m/f	n yes/**val**	Busby et al. (1989)
	Rat		10	s.c.	2 mg/animal, weekly; later 6 mg at longer intervals	≤ 626 days	4/10 tumours; control: 2/10 sarcomas	p no/ln,ld	Barry & Cook (1934)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Chrysene (contd)									
Purified	Rat		10	s.c.	1 mg/animal, weekly, 103 weeks	≤ 103 weeks	No tumours	n no/ln ld	Boyland & Burrows (1935)
	Rat, Wistar		5	s.c.	5 mg/animal, 7–9x	10 months	No tumours	n no/ln	Pollia (1941)
99.6%	Rat, Osborne/ Mendel	f	35	Intrapulm.	1 and 3 mg/animal, 1x weeks	≤ 135	14.3% and 28.6% tumour incidence; control: no tumours	p no/val	Wenzel-Hartung et al. (1990)
Coronene									
> 96%	Mouse, NMRI	f	40	Dermal	5 or 15 µg/animal, 4x/week, 104 weeks	≤ 104 weeks	Low dose:1/39, high dose: 2/40 local tumours at application site; vehicle control: no tumours	n yes/val	Habs et al. (1980)
TLC control	Mouse, Swiss ICR/Ha	f	20	Dermal	0.1 mg, 5x; initiation experiment	65 weeks	6/20 papillomas; promotor only: 5/20; coronene only: no tumours	q no/val	Van Duuren et al. (1968)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Cyclopenta[cd]pyrene									
> 96%	Mouse, NMRI	f	40	Dermal	1.7, 6.8 and 27.2 µg/ animal, 2x/week, 112 weeks	112 weeks	Low dose: no tumours; high dose: 2/38 skin carcinomas, 1/38 sarcomas;control: no tumours	q yes/**val**	Habs et al. (1980)
> 98%	Mouse, CD-1	f	30	Dermal	23, 91, 226, 566 µg/ animal, 1x; initiation experiment	27 weeks	10, 21, 30, and 37% papillomas; promotor only: 4%	p yes/**val**	Wood et al. (1980)
> 99.9%	Mouse, Swiss	f	30	Dermal	45, 136 and 407 µg/ animal, 2x/week, 30 weeks	57 weeks	Low dose:17; med. dose: 11; high dose: 7 skin tumours; control: no tumours	p no/**val**	Cavalieri et al. (1981b)
> 99.9%	Mouse, CD-1	f	30	Dermal	4.5, 14 and 41 µg/animal, every other day, 20 days; initiation experiment	44 weeks	Low dose:1/30; med. dose: 9/29; high dose: 6/29 papillomas; promotor only: 3/29	p no/**val**	Cavalieri et al. (1981b)
	Mouse, Sencar	f	30	Dermal	10, 100 and 200 µg/ animal, 1x; initiation experiment	26 weeks	Low dose:11%; med. dose: 39%; high dose: 57% papillomas; promotor only: 10%	p no/**val**	Raveh et al. (1982)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Cyclopenta[cd]pyrene (contd)									
> 99%	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	8-14	i.p.	0.35, 0.7, 1.05, 1.4, and 1.75 mg/animal (total dose) in 3 aliquots on day 1, 8, and 15 after birth	26 weeks	62, 60, 56, 70, 86, 93%, 77, 100, and 89, 100% m/f with lung tumours; vehicle control: 8, 8%	p yes/val	Busby et al. (1988)
> 99%	Rat, Sprague-Dawley	f	20	Intra-mammary	1.8 and 5.4 mg into 4th mammary gland, 1x	≤ 34 weeks	No mammary tumours; control: no tumours	n no/val	Cavalieri et al. (1988b)
Dibenz[a,h]anthracene									
	Mouse Swiss	m		Oral	1.5 mg/animal in PEG-400, 1x; initiation experiment	30 weeks	21% forestomach papillomas; promotor only: 14%	q no/lc	Berenblum & Haran (1955)
	Mouse DBA/2	m/f	21/21 control: 25/10	Drinking-water	0.8 mg/day/animal in olive oil, 8-9 mtonhs	8–9 months	14/14 m and 13/13 f with pulmonary adenomas; 14/14 m and 10/13 f with alveologenic carcinomas; control: 1 mouse with tumour	p no/val	Snell & Stewart (1962)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenz[a,h]anthracene (contd)									
	Mouse, Swiss Millerton	f	20	Dermal	0.001, 0.01, and 0.1%, 3x/week, life	≤ 21, 13 or 9 months	0.001%: 30% papillomas, 30% carcinomas; 0.01%: 95/90% papilloma/carcinoma; 0.1%: 90%/75% papilloma/ carcinoma	p no/ld	Wynder & Hoffmann (1959a)
	Mouse, Swiss albino DBA/2Jax	m	≤ 50	Dermal	0.02 and 0.16 µg/animal, 1x; initiation experiment	32 weeks	33 and 38% with skin tumours; acetone control: 13%	p yes/**val**	Klein (1960)
Chromatograph recrystallized	Mouse, Swiss	f	20	Dermal	38 µg/animal, 2x/wk, 44 weeks	≤ 60 weeks	80% with skin tumours; vehicle control: 4%	p no/**val**	Lijinsky et al. (1965)
	Mouse, IF/Bor	m/f	30/30	Dermal	m: 0.3% solution (= 1.5 mg/animal), 1x/week, 18 weeks; f: 0.5% (= 1 mg/ animal), 8x, every 2 weeks	≤ 29/22 weeks	m: 26% with papillomas after 20 weeks,100% after 29 weeks; f: 100% with breast tumours after 22 weeks	p no/**val**	Johnson (1968)
> 99%	Mouse, NMRI	f	50	Dermal	1 drop, 3x/week, 112 weeks; total doses: 37.8, 125, and 378 µg/animal	112 weeks	6%, 8% and 32% with skin tumours; controls: 2–4%	p no/**val**	Platt et al. (1990)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenz[a,h]anthracene (contd)									
> 99%	Mouse, NMRI	f	16	Dermal	83.5 and 167 µg/animal, 1x; initiation experiment	24 weeks	38 and 93% with skin tumours; vehicle control: no tumours	p no/**val**	Platt et al. (1990)
	Mouse		10	s.c./i.p.	0.2 mg/animal, 2x/week, 50 weeks alternating	Life	3/10 with subcutaneous sarcomas	p no/In ld	Boyland & Burrow, (1935)
Spectrometer control	Mouse, C57Bl	m/f	50	s.c.	0.02 mg/animal in tricaprylin; 1x	≤ 22 months	28/48 sarcomas after 4 ; months solvent control: 3/280	p no/**val**	Steiner & Falk (1951)
	Mouse, C57Bl	m/f	40–50	s.c.	0.02, 0.04 mg/animal in tricaprylin; 1x	≤ 22–28 months	7/21 and 6/18 sarcomas after 6 and 5 months	p yes/ld	Steiner (1955)
	Mouse, C57BL	m/f	20/19	s.c.	1 mg/animal, 1x/week, 10 weeks	60–80 weeks	20/20 m and 17/19 f with sarcomas; control: no sarcomas	p no/**val**	Boyland & Sims (1967)
	Mouse, NMRI	f	60	s.c.	10, 30, 90, 270 and 810 µg/animal, 1x	≤ 16 months	40, 35, 65, 75, and 90% with tumours	p no/**val**	Pott et al. (1973)
	Mouse, B6, D2	m/f	30(60)	s.c.	0.15 and 0.3 mg/animal, 1x	12 months	B6 mice: 16/30 and 14/30; D2 mice:1/30 and 0/30 with fibrosarcomas	p no/**val**	Kouri et al. (1983)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenz[a,h]anthracene (contd)									
> 99%	Mouse, NMRI	f	47–50	s.c.	10, 30, 86 µg/animal, 1x	112 weeks	52, 46, and 63% with fibro-sarcomas; controls: 2-6%	p no/**val**	Platt et al. (1990)
> 99%	Mouse, NMRI newborn	m/f	40–50	s.c.	11.1 and 111 µg/animal on day 2, 1x	40 weeks	12/35 with pulmonary tumours; controls: 2/33 and 4/41	p no/**val**	Platt et al. (1990)
	Mouse, A		10	i.v.	10 mg/kg, 1x	20 weeks	100% lung tumours; control: 21%	p no/**val**	Shimkin & Stoner (1975)
	Rat		2 x 10	s.c.	2 mg/animal, weekly; later 6 mg at longer intervals	≤ ~ 200 days	1/10 and 7/10 with tumours; control: 2/10	q no/ln,ld	Barry & Cook (1934)
	Rat		10–18 (6 exp.)	s.c./i.p. alterna-ting	1 mg/animal, 2x/week, 50 weeks	Life	3–6/10 and 9/18 with sub-cutaneous sarcomas	p no/ln ld	Boyland & Burrows (1935)
	Rat, Wistar		5	s.c.	5 mg/animal, 4–8x	10 months	2 with tumours after 8–9 months	p no/ln	Pollia (1941)
99.3%	Rat, Osborne/ Mendel	f	35	Intrapulm.	0.1 mg/animal, 1x	≤ 123 weeks	57.1% tumour incidence; control: no tumours	p no/**val**	Wenzel-Hartung et al.(1990)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenz[a,h]anthracene (contd)									
> 99%	Rat, Sprague-Dawley	f	20	Intra-mammary	1.1 and 4.5 mg into 5th mammary gland, 1x	20 weeks	No mammary tumours; control: no tumours	n no/**val**	Cavalieri et al. (1988a)
Chromato-graphy control	Hamster, Syrian golden	m/f	5/5 weeks	Dermal	8 drops of a 0.2% solution, 2x/week, 10 weeks	≤ 75	No tumours	n no/ln,ld	Shubik et al. (1960)
	Hamster, Syrian golden	m	46	Intra-tracheal	0.05 and 0.25 mg/animal, 1x/week, 30 weeks weeks	≤ 110	0/46 and 2/46 respiratory tract tumours; control: no tumours	q yes/**val**	Sellakumar & Shubik (1974)
	Hamster, Syrian golden			Intra-trachea	l0.3 and 0.9 mg/animal, 1x/week, 20 weeks	≤ 2 years	55 and 65% with tumours	p no/**val**	Pott et al. (1978)
	Monkey, Old world				Not specified		No tumours	n no/ld,lc	Adamson & Sieber (1983)
Dibenzo[a,e]pyrene									
Recystal-lized	Mouse, Swiss albino Ha/ICR/Mil	f	40/20	Dermal	0.05 and 0.1% solution, 3x/week, 12 months	15 months	16/40, 9/20 with papillomas and 9/40, 6/20 with epitheliomas; solvent control: no	p no/**val**	Hoffmann & Wynder (1966)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,e]pyrene (contd)									
Recrystallized	Mouse, Swiss albino Ha/ICR/Mil	f	28	Dermal	25 µg/animal, 10x over 20 days; initiation experiment	6 months	10/28 papillomas; promotor only: 2/30	p no/val	Hoffmann & Wynder (1966)
> 99%	Mouse, Sencar	f	21	Dermal	242 µg/animal, 1x; initiation experiment	26 weeks	24% papillomas; solvent control: 9%	p yes/val	Cavalieri et al. (1989)
	Mouse, XVII nc/Z	m/f	21/14	s.c.	0.6 mg/animal, 1x/month, 3 x	≤ 142 days m or 126 days f	18/21m and 14/14 f local sarcomas; no vehicle control	p no/val	Lacassagne et al. (1963b)
	Mouse	m/f	12/15	s.c.	0.6 mg/animal, 1x	≤ 196 days mor 220 days f	10/12 m and 10/15 f local sarcomas; no vehicle control	p no/val	Lacassagne et al. (1963b)
> 99%	Rat, Sprague-Dawley	f	19	Intra-mammary	1.2 mg/gland, 1x, 8 glands	≤ 40 weeks	1/19 with mammary tumours; control: 0/21 or 2/20	n yes/val	Cavalieri et al. (1989)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,h]pyrene									
	Mouse		74	Dermal	1 drop of a 0.15% solution alternate days, 55 or 86 times	4.5 months	50% with skin tumours	p no/lc	Kleinenberg (1939)
Recystal-lized	Mouse, Swiss albino Ha/ICR/Mil	f	20	Dermal	0.05 and 0.1% solution, 3x/week, 12 months	11, 15 months	16/20, 15/20 with papillomas and 13/20, 15/20 with epithe-liomas; solvent control: no tumours	p no/**val**	Hoffmann & Wynder (1966)
Recrystal-lized	Mouse, Swiss albino Ha/ICR/Mil	f	29	Dermal	25 µg/animal, 10x over 20 days; initiation experiment	6 months	21/29 papillomas; promotor only: 2/30	p no/**val**	Hoffmann & Wynder (1966)
96.6%	Mouse, Swiss	f	40	Dermal	120 µg/animal, 2x/week, 30 weeks	70 weeks	90% tumour incidence; solvent control: no tumours	p no/**val**	Cavalieri et al. (1977)
Pure	Mouse, CD-1	f	30	Dermal	15.1, 60.5 and 181,4 µg/animal, 1x; initiation experiment	17 weeks	55, 79, and 72% with skin tumours; controls: 0-10%	p yes/**val**	Chang et al. (1982)
> 99%	Mouse, Sencar	f	24	Dermal	242 µg/animal, 1x; initiation experiment	26 weeks	75% papillomas; solvent control: 9%	p yes/**val**	Cavalieri et al. (1989)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,h]pyrene (contd)									
	Mouse, XVII	m/f	35/10	s.c.	0.6 mg/animal, 1x/month, 3 months	> 111/128 days (average latency)	34/35 m and 1/10 f with local sarcomas	p no/ld	Lacassagne et al. (1958)
	Mouse, CD-1	f	31	s.c.	0.2 mg/animal, 1x; initiation experiment	27 weeks	26/28 with tumours; solvent control: 2/32	p no/**val**	Sardella et al. (1981)
	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	40	i.p.	3.8, 7.6 and 15.1 µg on days 1, 8 and 15 of life	49–54 weeks	97% with pulmonary and 44% with hepatic tumours; control: pulmonary tumours 27%, no hepatic tumours	p yes/**val**	Chang et al. (1982)
> 99%	Rat, Sprague-Dawley	f	20	Intra-mammary	1.2 mg/gland, 1x, 8 glands	≤ 40 weeks	19/20 with mammary tumours; control: 0/21 or 2/20	p yes/**val**	Cavalieri et al. (1989)
Dibenzo[a,i]pyrene									
	Mouse, XVII	m	23	Dermal	1 drop of a saturated solution, 2x/week	> 7 months	21/23 papillomas and 8/23 epitheliomas;solvent control: no tumours (14 months)	p no/**val**	Lacassagne et al. (1958)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,i]pyrene (contd)									
	Mouse, Swiss Millerton	f	20/10	Dermal	0.01 and 0.1%, 3x/week, 16 and 13 months	≤ 16 and 13 months	0.01%: 10% papillomas, no carcinomas;0.1%: 50% papillomas, 10% carcinomas	p no/ld	Wynder & Hoffmann (1959a)
Recrystal-lized	Mouse, Swiss albino Ha/ICR/Mil	f	20	Dermal	0.05 and 0.1% solution, 3x/week, 12 months	15 months	16/40, 16/20 with papillomas and 13/20, 15/20 with epithe-liomas; solvent control: no tumours	p no/**val**	Hoffmann & Wynder (1966)
Recrystal-lized	Mouse, Swiss albino Ha/ICR/Mil	f	30	Dermal	25 µg/animal, 10x over 20 days; initiation experiment	6 months	12/30 papillomas; promotor only: 2/30	p no/**val**	Hoffmann & Wynder (1966)
	Mouse, Swiss albino Ha/ICR	f	20	Dermal	100 and 500 µg/animal, 1x; initiation experiment	22 weeks	40 and 80% with tumours; vehicle control: no tumours	p no/**val**	Hecht et al. (1981)
Pure	Mouse, CD-1	f	30	Dermal	15.1, 60.5 and 181.4 µg/ animal, 1x; initiation experiment	17 weeks	28, 67, and 70% with skin tumours; controls: 0-10%	p yes/**val**	Chang et al. (1982)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,i]pyrene (contd)									
> 99%	Mouse, Sencar	f	24	Dermal	242 µg/animal, 1x; initiation experiment	26 weeks	63% papillomas; solvent control: 9%	p yes/**val**	Cavalieri et al. (1989)
	Mouse, XVII	m/f	17/18	s.c.	0.6 mg/animal, 1x/month, months	3> 75/82 days (average latency)	17/17 m and 16/18(f) with local sarcomas	p no/ld	Lacassagne et al. (1958)
	Mouse, XVII/C57Bl hybrids	m/f	8/8	s.c.	2 mg/animal, 1x	2–3 months	100, 100% with skin tumours; average latency: 74 days	p no/ln,ld	Waravdekar & Ranadive (1958)
	Mouse, C57BL/6	m		s.c.	0.5 mg/animal, 1x	4–5 weeks	100% fibrosarcomas; malignant cells identifiable after 4–5 weeks	p no/ld	Homburger et al. (1962)
	Mouse, CD-1	f	50	s.c.	0.1 mg/animal, 1x	75 weeks	40/41 with tumours; solvent control: no tumours	p no/**val**	Sardella et al. (1981)
	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	40	i.p.	3.8, 7.6 and 15.1 µg on day 1, 8, and 15 of life	49–54 weeks	97% with pulmonary and 54% with hepatic tumours; control: pulmonary tumours 27%, no hepatic tumours	p yes/**val**	Chang et al. (1982)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,i]pyrene (contd)									
> 99%	Rat, Sprague Dawley	f	19	Intra-mammary	1.2 mg/gland, 1x, 8 glands	≤ 40 weeks	18/19 with mammary tumours; control: 0/21 or 2/20	p yes/**val**	Cavalieri et al. (1989)
	Hamster, Syrian	m	6–10	s.c.	0.25, 0.5, 1 and 2 mg/ animal, 1x	9–14 weeks (average latency)	55, 90, 100, and 100% with fibrosarcomas; vehicle control: 0%	p no/**val**	Wodinsky et al. (1964)
	Hamster, Syrian	m/f	139/ 157	s.c.	1 mg/animal, 1x	11 weeks (average latency)	99/100% with fibrosarcomas	p no/**val**	Wodinsky et al. (1964)
	Hamster, Syrian golden	m	44/34	Intra-tracheal	0.5 and 2 mg/animal, weekly, 24 and 4 weeks, respectively	≤110 weeks	Tumours (i) 6/44 (trachea), 37/44 (bronchi), 2/34 (trachea); (ii) 1/34 (larynx), 13/34 (bronchi); control: no tumours	p yes/**val**	Sellakumar & Shubik (1974)
	Hamster, Syrian golden	m/f	24/24	Intra-tracheal	0.5 and 1 mg/animal, 1x/week, 17 and 12 weeks, respectively		65 and 75% respiratory tumours (bronchi, trachea); shortest latency: 8 weeks	p no/ld	Stenbäck & Sellakumar (1974)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,i]pyrene (contd)									
	Monkey, Old world				Not specified		No tumours	n no/lc	Adamson & Sleber (1983)
Dibenzo[a,i]pyrene									
Recrystal-ized	Mouse, Swiss albino Ha/ICR/Mil	f	20	Dermal	0.05 and 0.1% solution, 3x/week, 12 months	11, 14 months	17/20, 18/20 with papillomas and 17/20, 18/20 with epitheliomas; solvent control: no tumours	p no/**val**	Hoffmann & Wynder (1966)
Recrystal-lized	Mouse, Swiss albino Ha/ICR/Mil		30	Dermal	25 µg/animal, 10x over 20 days; initiation experiment	6 months	18/30 papillomas; 1/30 epitheliomas; promotor only: 2/30	p no/**val**	Hoffmann & Wynder (1966)
	Mouse, Swiss ICR	f	19–21	Dermal	55, 200, 240, 350 and 700 µg/animal given in 55, 40, 24, 7 and 7 applications	6 months	20, 19, 21, 19 and 16 with skin tumours; no solvent control group	p no/**val**	Masuda & Kagawa (1972)
> 99%	Mouse, Sencar	f	24	Dermal	242 µg/animal, 1x; initiation experiment	26 weeks	92% papillomas; solvent control: 9%	p yes/**val**	Cavalieri et al. (1989)
Pure, 161–162 °C)	Mouse, Sencar	f	24	Dermal	10, 30 and 90 µg/animal, 1x; initiation experiment	15 weeks	23/24, 22/24 and 24/24 with tumours;control: no tumours	p yes/**val**	Cavalieri et al. (1991)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Dibenzo[a,l]pyrene (contd)									
Pure	Mouse, Sencar	f	24	Dermal	1.2, 6 and 30 µg/animal, 1x; initiation experiment	7 weeks	22/24, 20/24 and 20/24 with tumours; 2 control: no tumours	p yes/**val**	Cavalieri et al. (1991)
Chomato-graphy purified	Mouse, Sencar	f	24	Dermal	30 µg/animal, 1x; initiation experiment without promotion	27 weeks	7/24 with tumours	p yes/**val**	Cavalieri et al. (1991)
	Mouse, XVII nc/ZE	m/f	12/12	s.c.	0.6 mg/animal, 1x/month, 2 months (some animals, 3rd injection after 2 months)	≤ 7 months	All animals with local sarcomas (mean latent period: 120 days); control: no tumours	p no/**val**	Lacassagne et al. (1968a)
> 99%	Rat, Sprague-Dawley	f	9	Intra-mammary	1.2 mg/gland, 1x, 8 glands	≤ 40 weeks	9/9 with mammary tumours; control: 0/21 or 2/20	p yes/**val**	Cavalieri et al. (1989)
	Rat, Sprague-Dawley	f	20	Intra-mammary	76 and 302 µg/gland, 1x, 8 glands	≤ 24 weeks	20/20 and 19/20 with mammary tumours; control: 1/18	p yes/**val**	Cavalieri et al. (1991)
Fluoranthene									
	Mouse,		2x10	Dermal	0.3% in benzene, 2x/week, life	≤ 501 days	No tumours	n no/**id**	Barry et al. (1935)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Fluoranthene (contd)									
	Mouse, Swiss Millerton	f	20	Dermal	0.1% solution, 3x/week, life	≤ 17 months	No papillomas or carcinomas	n no/ld	Wynder & Hoffmann (1959a)
	Mouse, Swiss Ha/ICR/Mil	f	20	Dermal	1%, 3x/week, 12 months	15 months	At 12 months 0/20 tumours; no vehicle control	n no/**val**	Hoffmann et al. (1972)
99.9%	Mouse, Swiss Ha/ICR/Mil	f	30	Dermal	0.1 mg/animal, 10x over 20 days; initiation experiment	24 weeks	1/29 skin tumours; solvent control: 1/30	n no/**val**	Hoffmann et al. (1972)
Recrystal-lized	Mouse, C3H	m	15	Dermal	250 µg/animal in decalin, 2x/eswk, 82 weeks	82 weeks	No papillomas or carcinomas; solvent control: 2/13 papillo-mas	n no/**val**	Horton & Christian (1974)
Purified, 107–109 °C)	Mouse, Swiss ICR/Ha	f	50	Dermal	40 µg/animal, 3x/week, life	440 days	No tumours observed; controls: no tumours	n no/**val**	Van Duuren & Goldschmidt (1976)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Fluoranthene (contd)									
	Mouse, Jackson A	m/f	7/7	s.c.	10 mg/animal, 5x	19 months	No tumours	n no/ld,ln	Shear (1938)
	Mouse, XVII nc/Z	m/f	10/10	s.c.	0.6 mg/animal, 1x/month, 3x		No sarcomas	n no/ld,ln	Buu-Hoi (1964)
99%	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	20-31	i.p.	0.7 and 3,5 mg/animal (total dose) in 3 aliquots on days 1, 8 and 15 after birth	24 weeks	23, 15, and 74, 38% m/f with lung tumours; vehicle control: 4, 14%	p yes/val	Busby et al. (1984)
>99.5%	Mouse, CD-1 newborn	m/f	22/30	i.p.	0.7 and 3.5 mg/animal (total dose) in 3 aliquots on days 1, 8, and 15 after birth	52 weeks	43, 35, and 65, 86% with lung tumours; 64, 0% and 100, 7% with hepatic tumours; vehicle only: 17, 12% (lung) and 17, 6% (liver)	p yes/val	La Voie et al. (1994)
Fluorene									
	Mouse		100	Dermal	Dissolved in 90% benzene	9 months	No tumours	n no/ld,lc	Kennaway (1924)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Fluorene (contd)									
	Mouse, CF1	m/f	10/10	Dermal	60 µg/animal, 2x/week, 31 weeks	31 weeks	No skin tumours	n no/ld	Riegel et al. (1951)
'Pure'	Mouse, white	m	100	Dermal	3 drops, 1x/week of ~3% solution, 1 year; initiation experiment	≤1 year	After 9 months 10/100 survived, no tumours; promotor only: 0.08 tumour/animal	n no/**val**	Graffi et al. (1953)
	Mouse, Swiss	m	5	i.p.	1000 mg/kg, 1x	≤ 5 months	No effects	n no/ld ln	Shubik & Della Porta (1957)
	Mouse, Jackson A	m	10	s.c.	10 mg/animal, 7x over 16 months	19 months	No tumours	n no/ln,ld	Shear (1938)
Highly purified	Rat, Buffalo	f	20	Oral (diet)	0.05% diet; 4.3 mg/rat per day = 796 mg/rat (total intake) over 6 months	10.7 months	2/11 carcinomas (renal pelvis, ureter); control: 4/16 with carcinomas	q no/ld	Morris et al. (1960)
Highly purified	Rat, Buffalo	f	18	Oral (diet)	0.05% diet; 4.6 mg/rat per day = 2553 mg/rat (total intake) over 18 months	≤ 20.1 months	7/18 tumours; control: 4/18 or 15/18 tumours	q no/**val**	Morris et al. (1960)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Indeno[1,2,3-cd]pyrene									
Recrystal-lized	Mouse, Swiss Ha/ICR/Mil	f	30	Dermal	25 µg/animal, 10x over 20 days; initiation experiment	6 months	5/30 papillomas; promotor: 2/30	q no/**val**	Hoffmann & Wynder (1966)
Recrystal-lized	Mouse, Swiss albino Ha/ICR/Mil	f	20	Dermal	0.05 and 0.1% solution, 3x/week, 12 months	15 months	Dioxane solvent: no tumours; acetone solvent: dose-related tumour increase	q no/**val**	Hoffmann & Wynder (1966)
> 96%	Mouse, NMRI	f	40	Dermal	3.4, 5.6, 9.2 µg/animal, 2x/week, life	≤ 2 years	3, 0, 0% with local tumours; control: no tumours	n yes/**val**	Habs et al. (1980)
	Mouse, Crl:CD1 (ICR)BR	f	25	Dermal	100 µg/animal, 10x over 20 days; initiation experiment	25 weeks	90% with skin tumours; vehicle control: < 5%	p yes/**val**	Rice et al. (1986)
> 99%	Mouse, CD-1	f	25	Dermal	110 µg/animal, 10x over 20 days; initiation experiment	23 weeks	72% tumours, 2.1 skin tumours/ animal; control: no tumours	p yes/**val**	Rice et al. (1990)
	Mouse, XVII nc/Z	m/f	14/14	s.c.	0.6 mg/animal, 1x/mth, 3 months	Average, 265 days m, 145 days f	Sarcomas: 10/14 m and 1/14 f	p no/**val**	Lacassagne et al. (1963a)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Indeno[1,2,3-cd]pyrene (contd)									
> 99%	Mouse, CD-1 newborn	m/f	11/9	i.p.	580 µg/animal in DMSO on days 1, 8 and after birth (total dose)	≤ 52 weeks	9% hepatic or 0% lung tumours; controls: 6%/0%	n yes/**val**	LaVoie et al. (1987)
99.4%	Rat, Osborne/ Mendel	f	35	Intra-pulm.	0.16, 0.83 and 4.15 mg/animal, 1x	116/109/ 92 weeks	3/35, 8/35 and 21/35 with lung tumours; control: no tumours	p yes/**val**	Deutsch-Wenzel et al. (1983)
5-Methylcholanthrene									
> 99.9%	Mouse, Swiss Ha/ICR/Mil	f	20	Dermal	0.1 mg/animal, 3x/week, 35 weeks	35 weeks (solvent control: 72 weeks)	20/20 with 85 skin tumours by 25 weeks; 20/20 with 99 tumours and 12/20 with 37 carcinomas by 35 wks; solvent control: no tumours	p no/**val**	Hecht et al. (1974)
> 99.9%	Mouse, Swiss Ha/ICR/Mil	f	20	Dermal	10, 30 and 100 µg, 10x over 20 days; initiation experiment	24 weeks	Low dose: 20/20 mice with 110 skin tumours; med dose: 20/20 with 160 skin tumours; high dose: 17/18 with 96 skin tumours; solvent control: no tumours	p no/**val**	Hecht et al. (1974)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
5-Methylcholanthrene (contd)									
	Mouse, Swiss Ha/ICR	f	20	Dermal	5 and 10 µg/animal, 3x/week, 62 weeks	62 weeks	Low dose: 9/20 with 22 skin tumours, 6/20 with 7 carcinomas; high dose: 15/20 with 38 tumours, 10/20 with 12 carcinomas; solvent control: no tumours	p yes/**val**	Hecht et al. (1976a)
Highly purified	Mouse, Swiss Ha/ICR	f	20	Dermal	1 and 3 µg, 10x over 20 days; initiation experiment	24 weeks	Low dose: 2/20 mice with 2 skin tumours; high dose: 20/20 with 45 skin tumours (1 carcinoma); solvent control: no tumours	p yes/**val**	Hecht et al. (1976a)
> 99.9%	Mouse, Swiss Ha/ICR/Mil	f	8 x 20	Dermal	3 and 10 µg, 10x over 20 days; initiation experiment	24 weeks	Low dose: 55-95% of mice with skin tumours; high dose: 80–90%; solvent control: no tumours	p yes/**val**	Hecht et al. (1978)
> 99.9%	Mouse, Swiss Ha/ICR outbred	f	20	Dermal	1 and 3 µg, 10x over 20 days; initiation experiment	24 weeks	Low dose: 75% of mice with skin tumours; high dose: 85%	p no/**val**	Hecht et al. (1979)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
5-Methylcholanthrene (contd)									
	Mouse, Swiss CD-1	f	20	Dermal	1 or x or 3 µg/animal, 10x over 20 days; initiation experiment	21 weeks	55, 75, and 90% with skin tumours; solvent control: 5%	p yes/**val**	Amin et al. (1981)
HPLC purified	Mouse, CD-1	f	20	Dermal	8 and 24 µg/animal, 1x; initiation experiment	26 weeks	80 and 90% tumour-bearing animals; solvent control: 10%	p yes/**val**	Hecht et al. (1985)
	Mouse, CD-1	f	20	Dermal	8 µg/animal, 1x; initiation experiment	21 weeks	65% with skin tumours; solvent control: 5%	p yes/**val**	Amin et al. (1985b)
	Mouse, CD-1	f	20	Dermal	24.2 µg/animal, 1x; initiation experiment	26 weeks	90% with tumours; 5.2 tumours/ animal; solvent control: 10%/0.1	p no/**val**	El-Bayoumy et al. (1986)
> 99%	Mouse, CD-1	f	20	Dermal	3.6, 12.1 and 36 µg/ animal, 10x over 20 days; initiation experiment	24 weeks	100, 100 and 100% with tumours; 9.2, 10.7 and 9.4 tumours/animal; solvent control: 20%	p yes/**val**	Rice et al. (1988b)
	Mouse, CD-1	f	20	Dermal	8 µg/animal, 1x; initiation experiment	21 weeks	85% with skin tumours; solvent control: 10%	p yes/**val**	Amin et al. (1990)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
5-Methylcholanthrene (contd)									
	Mouse, CD-1	f	20	Dermal	8 µg/animal, 1x; initiation experiment	26 weeks	65% with skin tumours; solvent control: 10%	p yes/**val**	Amin et al. (1992)
	Mouse, Swiss/C3H	m/m	20/ 2 x 10	s.c.	2 mg/animal in tricaprylin, 1x	6 months	Swiss mice: no local tumours; 16/20 mice died; C3H mice: 7/10 or 3/10 local sarcomas	q no/ld	Dunlap & Warren (1943)
Highly purified	Mouse, C57BL	m	25	s.c.	50 µg/animal in trioctanoin, 1x/2 weeks, 20 weeks	32 weeks	22/25 mice with 24 fibrosarcomas; vehicle control: no tumours	p no/**val**	Hecht et al. (1976b)
HPLC purified	Mouse, ICR/Ha newborn	m/f	35/48	i.p.	1.9 µg/animal on day 1; 3.9 µg on day 8; 7.8 µg on day 15	Weaned after 3 weeks; sacrificed after 35 weeks	20/21% with pulmonary tumours; 23/12% with hepatic tumours; solvent control: 4/7% and 2/2%	p yes/**val**	Hecht et al. (1985)
> 99%	Rat, Sprague-Dawley	f	20	Intra-mammary	0.97 and 3.9 mg into 5th mammary gland, 1x	20 weeks	No mammary tumours; control: no tumours	n no/**val**	Cavalieri et al. (1988a)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
1-Methylphenanthrene									
>99.5%	Mouse, Swiss Ha/ICR	f	20	Ddermal	100 µg, 10x over 20 days; initiation experiment	24 weeks	No tumours; vehicle control: no tumours	n no/**val**	LaVoie et al. (1981b)
Naphthalene									
	Mouse			Dermal	Several times/wk, ≤ 11 months	≤ 11 months	No skin tumours	n no/lc	Kennaway (1930)
Highly purified	Mouse, SW inbred		25; control: 21	Dermal	0.5% in benzene, 6x/week for 3 weeks, then 2x/wk for life	Life	4/25 with lymphatic leukaemia; 1/25 lymphosarcoma of thymus; 4/25 with benign tumours; benzene only: 2/21 with sarcomas; 1/21 with lung adenoma	q no/lc,ln	Knake (1956)
	Mouse, ICR/Ha	f	30	Dermal	0.25 mg/animal + 3 µg benzo[a]pyrene, 3x/wk, 78 weeks; co-carcinogenicity test	78 weeks	5/30 lymphomas; inhibitory effect on skin tumours; naphthalene only: no skin tumours	q no/**val**	Schmeltz et al. (1978)
98–99%	Mouse, A/J	f	30	Inhalation	0.05 and 0.15 mg/l, 6 h/day, 5 days/week, 6 months	6 months	29 and 30% with pulmonary tumours; control: 21% (increase in treatment groups not significant)	q yes/**val**	Adkins et al. (1986)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Naphthalene (contd)									
> 99%	Mouse, B6C3F1	m/f	75 (150)	Inhalation	0.053 and 0.16 mg/litre, 6 h/day, 5 days/week, 103 weeks	103 weeks	Significantly increased pulmonary alveolar and bronchiolar adenomas in females; no cataracts	q yes/**val**	Abdo et al. (1992); National Toxicology Program (1992b)
	Mouse		23	Bladder implant	1x (dose unspecified)	7 months	1/23 bladder carcinoma after 1 month; "inert" substance: higher rate of bladder carcinoma	–	Boyland et al. (1964)
	Rat, BDI/BDI II inbred		28	Oral (diet)	10–20 mg/animal, 6x/week, 70 weeks	Life	No tumours	n no/ld	Schmähl (1955)
	Rat, BDI/BDI II inbred		10	s.c.	20 mg/animal, 1x/week, 40 weeks	Life	No tumours	n no/ln	Schmähl (1955)
Crude, 90%	Rat, 'white'		38	s.c.	0.5 g/kg, 2x/month, 3.5 months	Life	5 malignant tumours (4/38 lmphosarcomas, 1/38 uterine sarcoma); 1 benign tumour; vehicle control: 1/38 lymphosarcoma and 1 benign tumour	q no/**val**	Knake (1956)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Naphthalene (contd)	Rat, BDI/BDI II inbred		10	i.p.	20 mg/animal, 1x/week, 40 weeks	Life	No tumours	n no/ln	Schmähl (1955)
Perylene Recrystallized	Mouse, Swiss ICR/Ha	f	20	Dermal	0.8 mg/animal, 1x; initiation experiment	58–60 weeks	3/20 papillomas; promotor only: 1/20 with papillomas; pure substance only: no tumours	n no/**val**	Van Duuren et al. (1970)
Recrystallized	Mouse, C3H	m	20	Dermal	75 µg/animal in decalin, 2x/week, 82 weeks	82 weeks	No skin tumours; solvent control: 2/13 papillomas	n no/**val**	Horton & Christian (1974)
Phenanthrene	Mouse		100	Dermal	Dissolved in 90% benzene	9 months	No tumours	n no/ld,lc	Kennaway (1924)
'Pure'	Mouse, white	m	100	Dermal	3 drops, 1x/week of ~ 3% solution, 1 year; initiation experiment	≤1 year	After 12 months 6/100 survived with a total of 1 tumour; 0.16 tumour/animal;promotor only: 0.08 tumour/animal	n no/**val**	Graffi et al. (1953)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Phenanthrene (contd)									
	Mouse, 'S'		20	Dermal	54 mg/animal, 3x/wk; total: 10x; initiation experiment	24 weeks	5/20 survivors with 12 papillomas; promotor only: 4/19 survivors/4 papillomas	q yes/**val**	Salaman & Roe (1956)
High purity	Mouse, 'stock albino'	m/f	10/10	Dermal	0.3 mg, 4x on days 0, 2, 6 and 8; initiation experiment	24 weeks	4/19 papillomas; solvent control: 2/20	q yes/**val**	Roe (1962)
TLC purified	Mouse, CD-1	f	30	Dermal	1.8 mg/animal, 1x; initiation experiment	35 weeks	40% with papillomas; promotor only: 3%	p no/**val**	Scribner (1973)
> 98%	Mouse, CD-1	f	30	Dermal	1.8 mg/animal, 1x; initiation experiment	36 weeks	5/30 papillomas; solvent control: 2/30	q no/**val**	Wood et al. (1979)
	Mouse, Swiss Ha/ICR	f	20	Dermal	100 µg, 10x over 20 days; initiation experiment	24 weeks	No skin tumours observed; vehicle control: no tumours	n no/**val**	LaVoie et al. (1981b)
	Mouse, C57Bl	m/f	40-50	s.c.	5 mg/animal in tricaprylin; 1x	≤ 22–28 months	No sarcomas after 8 months	n yes/ld	Steiner (1955)
	Mouse, 'stock albino'	m/f	10/10	s.c.	0.3 mg, 5x on days 0, 2, 4, 6 and 8; initation experiment	24 weeks	3/17 papillomas; solvent control: 2/20	n yes/**val**	Roe (1962)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Phenanthrene (contd)									
	Mouse, stock albino' newborn	m/f	57	s.c.	40 µg/animal; 1x administered to neonatal mice	≤ 62 weeks	3/49 lung adenomas; control: 8/34 and 5/38	n yes/val	Grant & Roe (1963)
> 98%	Mouse, Swiss-Webster BLU:Ha(ICR) newborn		100	i.p.	35, 70 and 140 µg/animal in DMSO on days 1, 8 and 15 after birth	38–42 weeks	6/35 pulmonary adenomas; DMSO only: 9/59	n yes/val	Buening et al. (1979)
	Rat, Sprague-Dawley	f	10	Oral	200 mg/rat, 1x; experiment on mammary tumours	60 days	No tumours at 60 days; controls: 8/164 after 310 days	n no/ln,lc	Huggins & Yang (1962)
99.9%	Rat, Osborne/ Mendel	f	35	Intrapulm.	1, 3 and 10 mg/animal, 1x	≤ 135 weeks	No tumours; control: no tumours	n no/val	Wenzel-Hartung et al.(1990)
Pyrene									
	Mouse		2x20	Dermal	1% in benzene, 2x/week, life	≤ 717 days	1/20 and 1/20 papillomas	n no/ld	Barry et al. (1935)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Pyrene (contd)									
	Mouse		40	Dermal	0.3% in benzene, 2x/ week, ≤ 680 days	≤ 680 days	No skin lesions	n no/ld	Badger et al. (1940)
'Pure'	Mouse, white	m	150	Dermal	3 drops, 1x/week of a 0.3% solution, 1 year; initiation experiment	≤ 1 year	After 6 months 18/150 survived with a total of 1 tumour; 0.06 tumour/animal; promotor only: 0.08 tumour/animal	n no/**val**	Graffi et al. (1953)
	Mouse, 'S'		20	Dermal	25 mg/animal, 3x/week; total:10x; initiation experiment	24 weeks	6/20 mice with 9 papillomas; promotor only: 4/19 mice with 4 papillomas	q yes/**val**	Salaman & Roe (1956)
	Mouse, Swiss Millerton	f	5	Dermal	10%, 3 x/week, life	≤ 18 months	No skin tumours	n no/ld,ln	Wynder & Hoffmann (1959a)
TLC purified	Mouse, CD-1	f	30	Dermal	2 mg/animal, 1x; initiation experiment	35 weeks	17% with papillomas; promotor only: 3%	q no/**val**	Scribner (1973)
High purity	Mouse, C3H	m	20	Dermal	250 µg/animal in decalin, 2x/week, 82 weeks	82 weeks	3/13 papillomas; solvent control: 2/13	q no/**val**	Horton & Christian (1974)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of administration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Pyrene (contd)									
Recrystallized	Mouse, Swiss ICR/Ha	f	50	Dermal	12 or 40 µg/animal, 3 x/week, 368 or 440 days	≤ 440 days	No skin tumours observed; control: no tumours	n no/**val**	Van Duuren & Goldschmidt (1976)
High purity	Mouse, Swiss ICR/Ha	f	50	Dermal	4 and 12 µg/animal + 5 µg benzo[*a*]pyrene, 3x/week, 33 weeks; co-carcinogenicity test	33 weeks	High dose: 13/50 papillomas, 5/50 carcinomas; benzo[*a*]pyrene only: 6/50 papillomas; pyrene only: no tumours	n no/**val**	Goldschmidt et al. (1973)
Recrystallized	Mouse, Swiss ICR/Ha	f	50	Dermal	4, 12 and 40 µg/animal + 5 µg benzo[*a*]pyrene, 3x/week, 368/368/440 days; co-carcinogenicity test	368 or 440 days	12/26/35 mice with papillomas, 6/20/26 with squamous cell carcinomas; positive control: 15, 11 tumours; solvent control: no tumours	p no/**val**	Van Duuren & Goldschmidt (1976)
> 98%	Mouse, CD-1	f	30	Dermal	20.2 and 80.9 µg/animal, 1x; initiation experiment	27 weeks	14 and 10% with tumours; vehicle control: 10%	n yes/**val**	Wood et al. (1980)
Crystals	Mouse, Jackson A	m/f	30	s.c.	10 mg/animal, 2 x at 4-month interval	≤ 18 months	No malignant tumours	n no/ld	Shear & Leiter (1941)

Table 90 (contd)

Purity	Species, strain	Sex	No./ sex / group	Route of adminis-tration	Dosage	Study duration at death/ sacrifice	Incidence and type of tumour	Result Stat./ Val.	Reference
Pyrene (contd)									
Recrystal-lized, HPLC	Mouse, Swiss-Webster BLU:Ha(ICR) newborn	m/f	23-28	i.p.	86.1 and 1750 µg/animal (total dose) in 3 aliquots on days 1, 8 and 15 after birth	26 weeks	17, 4, and 7, 12% m/f with lung tumours; vehicle control: 14, 7% m/f	n no/**val**	Busby et al. (1989)
> 99%	Hamster, Syrian golden	m	48	Intra-tracheal	3 mg/animal, 1x/week, 30 weeks	≤ 110 weeks	1/48 tumours of the trachea, 2/48 malignant lymphomas; control: 0/82 and 2/82	n yes/**val**	Sellakumar & Shubik (1974)
Triphenylene									
	Mouse		10	Dermal	0.3% in benzene, 2x/ week, life	≤ 548 days	No skin lesions	n no/ln,ld	Barry et al. (1935)
Recrystal-lized	Mouse, C3H	m	20	Dermal	250 µg/animal in decalin, 2x/week, 82 weeks	82 weeks	No skin tumours; solvent control: 2/13 papillomas	n no/**val**	Horton & Christian(1974)

Result: p(ositive), n(egative), q(uestionable); Stat, statistical evaluation: yes or no; Val, validity: **val**, valid; ld, limited design; lc, limited documentation; ls, limited survival; ln, limited number of animals
intrapulm., intrapulmonary injection; i.p., intraperitoneal injection; s.c., subcutaneous injection; i.m., intramuscular injection
m, male; f, female
TLC, thin-layer chromatography; DMSO, dimethylsulfoxide; HPLC, high-performance liquid chromatography; DMBA, 7,12-dimethylbenz[a]anthracene

Table 91. Overview of carcinogenicity of polycyclic aromatic hydrocarbons

Compound	Carcinogenicity (weight of evidence)	Species	Route of administration No. of studies with positive, negative, and questionable results																			
			Oral			Dermal			s.c./i.m.			i.p./i.v.			inh./tr.			Other				
			+	−	±	+	−	±	+	−	±	+	−	±	+	−	±	+	−	±		
Acenaphthene	Questionable	Mouse					1	1														
Acenaphthylene			No studies																			
Anthanthrene	Positive	Mouse				2	6			1					1				1			
Anthracene	Negative	Mouse					6	1		1		1										
		Rat		2					1	2			1			1						
		Rabbit																	1			
Benz[*a*]anthracene	Positive	Mouse	2		1	7	4		4	2		2						1				
		Rat		1			1			2		1							1			
		Hamster					2											1				
Benzo[*b*]fluoranthene	Positive	Mouse				7			1			1										
		Rat													1							
		Hamster														1						
Benzo[*j*]fluoranthene	Positive	Mouse				3		1				1										
		Rat													1							
Benzo[*ghi*]fluoranthene	(Negative)	Mouse					2															
Benzo[*k*]fluoranthene	Positive	Mouse				1	2	1	1				1									
		Rat													1							
Benzo[*a*]fluorene	(Questionable)	Mouse					1	1		1												

Table 91 (contd)

Compound	Carcinogenicity (weight of evidence)	Species	Route of administration No. of studies with positive, negative, and questionable results																			
			Oral			Dermal			s.c./i.m.			i.p./i.v.			inh./tr.			Other				
			+	−	±	+	−	±	+	−	±	+	−	±	+	−	±	+	−	±		
Benzo[*a*]fluorene	(Questionable)	Mouse						1														
Benzo[*ghi*]perylene	Negative	Mouse					8			2												
		Rat														1						
Benzo[*c*]phenanthrene	(Positive)	Mouse				2		2		1	1											
		Rat									1											
Benzo[*a*]pyrene	Positive	Mouse	5			26			6			3			1			2				
		Rat	2						1			1			9			3				
		Hamster	1			1	1	1							11	1	1					
		Dog																1				
		Cattle								1												
		Pig								2												
		Monkey							1	1	1				1							
Benzo[*e*]pyrene	Questionable	Mouse				2	1	5						1								
		Rat														1				1		
Chrysene	Positive	Mouse				11	9	1	3	3	1	1	2	1								
		Rat							1	2					1							
Coronene	(Questionable)	Mouse					1	1														
Cyclopenta[*cd*]pyrene	Positive	Mouse				4		1				1										
		Rat																		1		

Table 91 (contd)

Compound	Carcinogenicity (weight of evidence)	Species	Route of administration No. of studies with positive, negative, and questionable results																			
			Oral			Dermal			s.c./i.m.			i.p./i.v.			inh./tr.			Other				
			+	−	±	+	−	±	+	−	±	+	−	±	+	−	±	+	−	±		
Dibenz[*a*,*h*]anthracene	Positive	Mouse	1	1		6			8			1										
		Rat							2	1					1			1				
		Hamster					1								1	1						
		Monkey																1				
Dibenzo[*a*,*e*]pyrene	Positive	Mouse				3			2													
		Rat																1				
Dibenzo[*a*,*h*]pyrene	Positive	Mouse				6			2			1										
		Rat																1				
Dibenzo[*a*,*i*]pyrene	Positive	Mouse				7			4			1										
		Rat																1				
		Hamster							2						2							
		Monkey																1				
Dibenzo[*a*,*l*]pyrene	Positive	Mouse				7			1													
		Rat																2				
Fluoranthene	(Positive)	Mouse					6			2		3										
Fluorene	Negative	Mouse					3			1			1									
		Rat		2																		

Table 91. (contd)

Compound	Carcinogenicity (weight of evidence)	Species	Route of administration No. of studies with positive, negative, and questionable results																			
			Oral			Dermal			s.c./i.m.			i.p./i.v.			inh./tr.			Other		
			+	−	±	+	−	±	+	−	±	+	−	±	+	−	±	+	−	±
Indeno[1,2,3-*cd*]pyrene	Positive	Mouse				2	1	2	1			1								
		Rat													1					
5-Methylchrysene	Positive	Mouse				13			1		1	1	1							
		Rat																		1
1-Methylphenanthrene	(Negative)	Mouse					1													
Naphthalene	(Questionable)	Mouse					1	2									2			1
		Rat	1							1	1		1							
Perylene	(Negative)	Mouse					2													
Phenanthrene	(Questionable)	Mouse				1	3	3		3			1							
		Rat	1													1				
Pyrene	(Questionable)	Mouse				1	7	3		1			1							
		Hamster														1				
Triphenylene	(Negative)	Mouse					2													

+, positive; −, negative; ±, questionable; parentheses, limited number of studies
s.c., subcutaneous; i.m., intramuscular; i.p., intraperitoneal; i.v., intravenous; inh., inhalation; tr., intratracheal
Other: e.g. intramammary injection, bladder implant, bronchial implant

development is related closely to the route of administration, i.e. dermal application induces skin tumours, inhalation and intratracheal instillation result in lung tumours, subcutaneous injection results in sarcomas, and oral administration induces gastric tumours. Tumour induction is, however, not restricted to the obvious sites. For example, lung tumours have been observed after oral administration or subcutaneous injection of benzo[*a*]pyrene to mice and liver tumours following intraperitoneal injection. In two studies, lung tumours were found in mice after intravenous injection of benzo[*a*]pyrene and dibenz[*a,h*]anthracene. Thus, tissues such as the skin must be able to metabolize PAH to their ultimate metabolites and itself become a target organ; however, all PAH that reach the liver via the bloodstream can be metabolized there. The liver in turn is a depot from which the metabolites are distributed all over the body (Wall et al., 1991). The carcinogenic potency of the PAH differs by three orders of magnitude, and several authors have presented tables of toxic equivalence factors based on experimental results in order to quantify these differences. Carcinogenic potency cannot be based only on chemical structure but requires theoretical considerations and calculations (see section 7.10).

Although this monograph primarily addresses single PAH, it was considered necessary for risk assessment to present some information on mixtures of PAH, to which humans are almost always exposed, predominantly adsorbed onto inhalable particles.

Although the essential results of the studies of carcinogenicity are summarized in Table 90, special aspects and comparisons of individual PAH are presented in more detail below.

7.7.1 Single substances

7.7.1.1 Benzo[a]pyrene

Oral administration of benzo[*a*]pyrene to male and female CFW mice induced gastric papillomas and squamous-cell carcinomas and increased the incidence of pulmonary adenomas (Rigdon & Neal, 1966). In other studies in which mice of the same strain were fed benzo[*a*]pyrene, pulmonary adenomas, thymomas, lymphomas, and leukaemia occurred, indicating that it can cause carcinomas distal to the point of application (Rigdon & Neal, 1969). The incidence of gastric tumours was 70% or more in mice fed 50–250 ppm benzo[*a*]pyrene for four to six months. No tumours were observed in controls (Rigdon & Neal, 1966; Neal & Rigdon, 1867; see also Table 90).

In a study of the effects of benzo[*a*]pyrene given in the diet or by gavage in conjunction with caffeine, groups of 32 Sprague-Dawley rats of each sex were fed diets containing 0.15 mg/kg bw benzo[*a*]pyrene either five times per week or only on every ninth day. Tumours were observed in the forestomach, oesophagus, and larynx, at combined tumour incidences of 3/64, 3/64, and

10/64 in the controls and those at the low and high doses, respectively. In the study by gavage, groups of 32 rats of each sex were treated with benzo[*a*]pyrene at 0.15 mg/kg bw in a 1.5% caffeine solution every ninth day, every third day, or five times per week. The combined incidences of tumours of the forestomach, oesophagus, and larynx were 3/64 in controls, 6/64 in rats at the low dose, 13/64 in those at the medium dose, and 14/64 among those at the high dose (Brune et al., 1981).

In hamsters exposed to 9.5 or 46.5 mg/m^3 benzo[*a*]pyrene by inhalation for 109 weeks, a dose-response relationship was seen with tumorigenesis in the nasal cavity, pharynx, larynx, and trachea. The fact that lung tumours were not detected could not be explained (Thyssen et al., 1981). Hamster lung tissue can activate benzo[*a*]pyrene to carcinogenic derivatives (Dahl et al., 1985).

Epidermal cell kinetics, DNA adduct levels, and changes in skin morphology were measured in ICR/Harlan mice after 29 weekly topical applications of 16, 32, or 64 µg benzo[*a*]pyrene for up to 35 weeks. Initially, there was a linear increase in DNA adducts, which was much less steep at 64 µg and which did not correlate with the sharp rise in tumour response at that dose. A dose-dependent increase in the ^{3}H-thymidine labelling index, the mitotic index, and the incidence of pyknotic and dark cells indicated that benzo[*a*]pyrene induced extensive cytotoxicity and cell death, with regenerative proliferation. Virtually all of the initial tumours were papillomas, which required an average of eight weeks to progress to carcinomas, reflecting the tumour-promoting activity of benzo[*a*]pyrene in this model (Albert et al., 1991a,b).

In a study with female Sprague-Dawley rats to elucidate whether the metabolites and DNA adducts of benzo[*a*]pyrene are formed in the liver or in target tissues, animals that had received a liver transplant were compared with normal animals. The liver was found to serve as a depot for PAH, in this case infused ^{3}H-benzo[*a*]pyrene, which was converted into polar metabolites. A few hours later, polar metabolites and DNA adducts were found in target tissues of both groups (Wall et al., 1991).

7.7.1.2 *Benzo[e]pyrene*

The results of studies on benzo[*e*]pyrene are considered to be questionable or negative even though positive results were reported in two studies by dermal application, because no bay-region activation was found in liver tissue that would result in an 'ultimate' carcinogen, such as 9,10-dihydroxy-11,12-epoxy-9,10,11,12-tetrahydrobenzo[*e*]pyrene (Jacob et al., 1983).

7.7.2 *Comparative studies*

Comparative studies on the tumorigenic activity of individual PAH that have been used as the basis for comparative potency factors (see Appendix 1)

are summarized below. The detailed results of studies with individual PAH are given in Table 90. In general, the results of studies with skin painting and lung implantation were used for estimating comparative potencies in preference to those from initiation–promotion experiments and studies by intraperitoneal injection. No comparative studies have been carried out by oral administration.

7.7.2.1 *Carcinogenicity*

(a) Dermal exposure

Skin painting: Solutions of 0.5% benzo[*a*]pyrene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, or benzo[*k*]fluoranthene were applied dermally three times weekly to groups of 20 female Swiss Millerton mice for life, and the number of skin tumours was determined. The percentages of papillomas/carcinomas induced by these compounds after four months were 70/20 with benzo[*a*]pyrene, 95/10 with benzo[*b*]fluoranthene, 40/5 with benzo[*j*]-fluoranthene, and none with benzo[*k*]fluoranthene. Minimal activity (10% papillomas) was induced by benzo[*k*]fluoranthene after 11 months (Wynder & Hoffmann, 1959b). The order of potency was thus benzo[*a*]pyrene > benzo[*b*]-fluoranthene > benzo[*j*]fluoranthene > benzo[*k*]fluoranthene.

In a similar regime, 0.01% solutions of benzo[*a*]pyrene or dibenz[*a,h*]-anthracene applied dermally to groups of 20 mice induced 10/10% and 15/5% papillomas/carcinomas, respectively after six months. A 0.1% solution of dibenzo[*a,i*]pyrene induced 10/0% tumours after seven months, and a 0.1% solution of benzo[*e*]pyrene induced 5/0% tumours after 10 months. A 1% chrysene solution induced 5/5% papillomas/carcinomas after eight months. A 0.1% solution of fluoranthene and 10% solutions of anthracene and pyrene had no activity (Wynder & Hoffmann, 1959a). The order of potency was thus benzo[*a*]pyrene = dibenz[*a,h*]anthracene > dibenzo[*a,i*]pyrene > chrysene > benzo[*e*]pyrene > fluoranthene, anthracene, pyrene.

In a further study, the carcinogenicity of PAH was compared after dermal application to mice three times weekly, as above. A dose of 0.05% induced the following percentages of papillomas/carcinomas after eight months: benzo[*a*]pyrene, 17/17; dibenzo[*a,h*]pyrene, 14/9; dibenzo[*a,l*]pyrene, 10/10; dibenzo[*a,i*]pyrene, 3/0; dibenzo[*a,e*]pyrene, 2/1 after 10 months; indeno[1,2,3-*cd*]pyrene, 1/1 with 0.5% solution; and benzo[*ghi*]perylene, 0/0 (Hoffmann & Wynder, 1966). The order of potency was thus benzo[*a*]pyrene > dibenzo[*a,h*]-pyrene > dibenzo[*a,l*]pyrene > dibenzo[*a,i*]pyrene > dibenzo[*a,e*]pyrene > indeno[1,2,3-*cd*]pyrene > benzo[*ghi*]perylene.

In a lifetime study by skin painting in female NMRI mice, benzo[*a*]pyrene and benzo[*b*]fluoranthene were carcinogenic, benzo[*j*]fluoranthene was weakly carcinogenic, and benzo[*k*]fluoranthene, indeno[1,2,3-*cd*]pyrene, and coronene had no carcinogenic effect (Habs et al., 1980). The order of potency was thus

benzo[*a*]pyrene >> benzo[*b*]fluoranthene > benzo[*j*]fluoranthene > benzo[*k*]-fluoranthene, coronene, indeno[1,2,3-*cd*]pyrene.

Initiation-promotion: Ten doses of PAH at a total dose of 0.25 mg/mouse were applied every second day to the backs of Swiss Millerton mice, which were then promoted with 2.5% croton oil in acetone (Hoffmann & Wynder, 1966). The relative tumour-inducing activity was: benzo[*a*]pyrene > dibenzo[*a,h*]pyrene > dibenzo[*a,l*]pyrene > dibenzo[*a,i*]pyrene > dibenzo[*a,e*]-pyrene > indeno[1,2,3-*cd*]pyrene > benzo[*ghi*]perylene.

In an assay in CD-1 mice, 30 µg of several PAH were applied in 10 doses over 20 days to the shaven backs of groups of 20 mice. Ten days after completion of the initiation, promotion was begun by thrice weekly application of 12-*O*-tetradecanoylphorbol 13-acetate in 0.1 ml acetone. The skin tumours induced were predominantly squamous-cell papillomas. After 20 weeks (10 weeks for benzo[*a*]pyrene), the percentages of skin tumour-bearing animals were 85% with benzo[*a*]pyrene, 45% with benzo[*b*]fluoranthene, 30% with benzo[*j*]fluoranthene, and 5% with benzo[*k*]fluoranthene. The vehicle controls had no tumours (La Voie et al., 1982b).

Sencar mice were treated with the -[*a,e*]-, -[*a,h*]-, -[*a,i*]-, and -[*a,l*]- isomers of dibenzopyrene with TPA as a promoter, anthanthrene as the negative control and with vehicle and sham controls. Dibenzo[*a,e*]pyrene was a very weak tumour initiator and dibenzo[*a,h*]pyrene and dibenzo[*a,l*]pyrene were tumorigenic; dibenzo[*a,l*]pyrene was highly toxic and an extremely potent carcinogen (Cavalieri et al., 1989). In a further investigation of dibenzo[*a,l*]-pyrene, with benzo[*a*]pyrene as the positive control, 7,12-dimethylbenz[*a*]-anthracene, recognized as the most potent carcinogenic PAH, was also tested at 4, 20, and 100 nmol of the PAH in the same regime as above. The tumorigenic activity of dibenzo[*a,l*]pyrene in mouse skin was inversely proportional to the dose, indicating that toxicity interferes with the initiation of tumours. When the effects of equimolar concentrations were compared, benzo[*a*]pyrene was a much weaker tumour initiator than dibenzo[*a,l*]pyrene (Cavalieri et al., 1991). The order of potency was thus dibenzo[*a,l*]pyrene > dibenzo[*a,i*]pyrene > dibenzo[*a,h*]pyrene > benzo[*a*]pyrene > dibenzo[*a,e*]pyrene.

The relationship between the *Ah* locus and the induction of subcutaneous fibrosarcomas was studied after administration of dibenz[*a,c*]anthracene and dibenz[*a,h*]anthracene to B6, D2, and B6D2F1 mice. The doses and results are shown in Table 92. Dibenz[*a,c*]anthracene was a weak tumour inducer in all groups tested. Dibenz[*a,h*]anthracene was a fairly potent inducer of subcutaneous tumours in B6 and B6D2F1 mice, but not in D2 mice. These results, with those of back-crossing experiments, demonstrate a strict correlation between the tumorigenicity of dibenz[*a,h*]anthracene and expression of the Ah^b allele (Kouri et al., 1983).

Table 92. Subcutaneous fibrosarcomas induced by a dose of 300 µg per animal of isomers of dibenzanthracene in different mouse strains

Dibenzanthracene isomer	Strain	Tumour incidence	Carcinogenic index
a,c	B6	1/30	1.1
	D2	0/30	0
	B6D2F$_1$	1/30	1.2
a,h	B6	14/30	24
	D2	0/30	0
	B6D2F$_1$	33/60	30
	B6D2F$_1$ x D2 back-crosses (*Ah^b/Ah^d* phenotype)	38/53	29
	B6D2F$_1$ x D2 back-crosses (*Ah^d/Ah^d* phenotype)	0/33	0

From Kouri et al. (1983)

(b) Other routes

Intraperitoneal injection in newborn mice: The tumorigenic activity of the nonalternant PAH benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, and benzo[*k*]fluoranthene and of indeno[1,2,3-*cd*]pyrene and benzo[*a*]pyrene were evaluated by injecting a total of 0.5, 1.1, 2.1, 2.1, or 0.5 µmol of each compound, respectively, in dimethyl sulfoxide in aliquots of 5, 10, or 20 µl on days 1, 8, and 15 after birth to CD-1 mice (La Voie et al., 1987). Direct comparison was not possible owing to the differences in the total amount injected, but both benzo[*b*]fluoranthene and benzo[*j*]fluoranthene had significant tumorigenic activity, whereas neither benzo[*k*]fluoranthene nor indeno[1,2,3-*cd*]pyrene was tumorigenic under these conditions. There were problems with the solubility of indeno[1,2,3-*cd*]pyrene. The order of potency was thus benzo[*a*]pyrene > benzo[*b*]fluoranthene = benzo[*j*]fluoranthene > benzo[*k*]-fluoranthene, indeno[1,2,3-*cd*]pyrene.

Intramammary injection: 7,12-Dimethylbenz[*a*]anthracene and dibenzopyrene isomers were tested with benzo[*a*]pyrene by intramammary injection. Dibenzo[*a,e*]pyrene was inactive, but dibenzo[*a,l*]pyrene was much more carcinogenic than 7,12-dimethylbenz[*a*]anthracene. At these doses, benzo[*a*]pyrene had only marginal activity. Dibenzo[*a,l*]pyrene thus appears to have the highest mammary cancer potency of all PAH so far tested (Cavalieri et al., 1989).

Lung implantation: The relative potencies of PAH to induce epidermoid carcinomas and pleomorphic sarcomas after intrapulmonary injection, with

benzo[*a*]pyrene as the reference substance, were dibenz[*a,h*]anthracene, 1.91 > benzo[*a*]pyrene, 1.00 > anthanthrene, 0.19 > benzo[*b*]fluoranthene, 0.11 > indeno[1,2,3-*cd*]pyrene, 0.08 > chrysene, 0.03 = benzo[*k*]fluoranthene, 0.03 = benzo[*j*]fluoranthene, 0.03 > phenanthrene, 0.001. Benzo[*ghi*]perylene and benzo[*e*]pyrene had no tumour-inducing effect (Deutsch-Wenzel et al., 1983; Wenzel-Hartung et al., 1990).

Subcutaneous injection: Dose–response curves for benzo[*a*]pyrene and dibenz[*a,h*]anthracene were established after a single subcutaneous injection of PAH in tricaprylin into the right axilla of male C3H mice; 99% of the tumours detected were spindle-cell sarcomas. The responses of vehicle controls were not reported. Under the conditions in this experiment, dibenz[*a,h*]-anthracene was estimated to be 4.5 times more potent than benzo[*a*]pyrene (Bryan & Shimkin, 1943).

Eight of 10 male and six of 10 female C57 black mice had injection-site tumours 60–80 weeks after 10 weekly subcutaneous injections of 1 mg benz[*a*]anthracene, whereas 20/20 males and 17/20 females had tumours after 1 mg dibenz[*a,h*]anthracene (Boyland & Sims, 1967).

7.7.2.2 *Further evidence*

(*a*) *Sebaceous gland assay*

Application of carcinogenic PAH to mouse skin leads to the destruction of sebaceous glands, hyperplasia, hyperkeratosis, and even ulceration (Bock, 1964). An assay of these glands has been used to screen the tumorigenic potential of PAH. Acute topical application of benzo[*a*]pyrene, benz[*a*]-anthracene, or dibenz[*a,h*]anthracene was reported to suppress glandular activity (Bock & Mund, 1958). The order of potency was benzo[*a*]pyrene = dibenz[*a,h*]anthracene > benz[*a*]anthracene. In another study, the order of potency was benzo[*a*]pyrene > benzo[*b*]fluoranthene = benzo[*j*]fluoranthene = benzo[*k*]fluoranthene = indeno[1,2,3-*cd*]pyrene (Habs et al. 1980).

(*b*) *DNA adduct formation*

In a ^{32}P-postlabelling test for covalent binding of PAH to DNA in mouse skin *in vivo* after a single topical application, the relative ability to induce DNA adducts was benzo[*a*]pyrene > benz[*a*]anthracene = dibenz[*a,h*]anthracene = benzo[*ghi*]perylene (Reddy et al., 1984). DNA adducts were not induced by pyrene. In a similar study, the relative ability of PAH to bind covalently to DNA was benzo[*b*]fluoranthene > benzo[*j*]fluoranthene > benzo[*k*]fluoranthene > indeno[1,2,3-*cd*]pyrene (Weyand et al., 1987). In a study *in vitro*, the relative ability for covalent binding of PAH to DNA was reported to be benzo[*a*]pyrene > dibenz[*a,h*]anthracene > benz[*a*]anthracene > pyrene > phenanthrene (Grover & Sims, 1968).

7.7.3 *PAH in complex mixtures*

In a PAH-rich emission mixture prepared by burning tar pitch with coal, the benzo[*a*]pyrene content was about 90 μg/m³, two to three times higher than the concentration measured in old coal plants. The tumour incidence in rats exposed for 16 h/day on five days per week for 22 months with a subsequent eight-month exposure to clean air was 18%; the mortality rate was not increased in comparison with controls exposed to clean air. The lung tumour incidences in mice exposed to the same atmosphere for 10, 12, or 24 months were 86, 70, and 79%, respectively, with 3.5, 12.5, and 32% in concurrent controls. An additive or even potentiating carcinogenic effect with other respiratory-tract carcinogens was demonstrated. In contrast to a group exposed concurrently to diesel exhaust, the coal-tar pitch did not cause particle overload in the lung or impair lung clearance (Heinrich et al., 1986a,b).

Female Wistar rats were exposed by inhalation to 1.1 (groups 1 and 2) or 2.6 mg/m³ (groups 3 and 4) of an aerosol of a PAH-rich hard coal-tar pitch condensate containing 20 or 50 μg/m³ benzo[*a*]pyrene (among other PAH), for 17 h per day on five days per week for 10 (groups 1 and 3) and 20 months (groups 2 and 4) and then to clean air for 20 or 10 months. The aerosol contained benz[*a*]anthracene and chrysene at concentrations similar to that of benzo[*a*]pyrene. Increased mortality was observed due to the development of large, multiple tumours in the lungs and not to toxic effects. The lung tumour rates were 4, 33, 39, and 97% in groups 1, 2, 3, and 4, respectively. Other groups exposed simultaneously to 2 or 6 mg/m³ carbon black, which might serve as a PAH carrier, showed an additional increase in tumour rates, i.e. 89 and 72% in comparison with 39% in group 3. A group exposed only to carbon black had a tumour rate of 18%. The authors therefore concluded that there was a more than additive carcinogenic effect after 10 months of exposure. A 'PAH depot' effect may be involved, in which the residence time of the PAH is prolonged due to attachment to the inert carbon black particles, with an extended period of elution of adsorbed PAH. Furthermore, the irritating, inflammatory, and cell proliferation effects of carbon black enhance the probability of genotoxic effects in the lungs (Heinrich, 1989; Heinrich et al., 1994a). Most of the lung tumours observed after exposure to tar-pitch aerosol with or without carbon black were classified as squamous-cell carcinomas (Heinrich et al., 1994b).

The preliminary findings of a study on coal gasification tars have been reported. Coal-tar is a complex mixture containing over 1000 compounds, of which at least 30 are PAH, including benzo[*a*]pyrene. In a two-year bioassay for carcinogenicity, female B6C3F$_1$ mice were fed up to 100 ppm benzo[*a*]pyrene or up to 1% coal-tar. Forestomach tumours were observed in mice fed benzo[*a*]pyrene, the incidence increasing sharply at doses between 5 and 25 ppm. Forestomach tumours were also seen in mice fed coal-tar, with a clear increase at 0.3%; the incidence was approximately the same at 0.3 and 0.6% but

declined at 1.0%, due to mortality from small intestinal adenocarcinomas, which were not observed at doses below 0.6%. Steady-state DNA adduct levels were examined in the forestomachs and small intestines of mice fed benzo[*a*]pyrene or coal-tar for four weeks. Those fed benzo[a]pyrene had one major adduct, which also accounted for 10–25% of the adducts in the forestomachs of mice fed coal-tar. A linear dose-response relationship was observed between the dose of benzo[*a*]pyrene and the adduct levels in the forestomach. The adduct levels in the forestomachs of mice fed coal-tar increased in a relatively linear manner at doses above 0.3%. Total adducts and benzo[*a*]pyrene-adduct levels in the small intestine increased up to the 0.6% dose of coal-tar and then decreased (Culp et al., 1996).

A two-stage study of carcinogenicity in female CD-1 mice was performed to assess the risk deriving from coal-tar formulations used for human therapeutic purposes, e.g. against dandruff or psoriasis (see also section 8.2.3). Mice were treated epicutaneously five times per week with 50 mg of a 1.5% coal-tar ointment for two weeks ('initiation'), followed by 'promotion' with 50 mg of a 0.1% dithranol cream, used against psoriasis, three times per week for 40 weeks. A single dose of 50 µg benzo[*a*]pyrene was given as an initiator as a positive control. After 40 weeks of promoter treatment, 4/27 tumours were observed in the mice treated with coal-tar and 14/28 in those given benzo[*a*]pyrene; when coal-tar or dithranol was given alone, no tumours were observed (Phillips & Alldrick, 1994).

7.7.4 *Transplacental carcinogenicity*

7.7.4.1 *Benzo[a]pyrene*

Clastogenic responses to benzo[*a*]pyrene and AHH inducibility were measured in 11-day-old embryos of genetically different mice (C57 and DBA mated *inter se* and mixed) after transplacental treatment with 150 mg/kg bw by gavage 15 h before sacrifice. The rate of chromosomal aberrations was not correlated quantitatively with AHH activity. It was concluded that not only activation and detoxification of benzo[*a*]pyrene in maternal tissue but also other genetically controlled processes, such as repair and transformation of primary DNA lesions into true DNA discontinuities, are involved (Adler et al., 1989).

Benzo[*a*]pyrene can cross the placenta in mice and rats (Shendrikova & Aleksandrov, 1974; Shendrikova et al., 1974; Takahashi, 1974; Baranova et al., 1976), but the concentration of ^{14}C-benzo[*a*]pyrene was one to two orders of magnitude lower in mouse embryonic than maternal tissues after oral administration (Neubert & Tapken, 1988). Intraperitoneal administration of benzo[*a*]pyrene to Ha/ICR mice during the last half of pregnancy increased the incidences of pulmonary adenomas and skin papillomas in progeny nursed by

foster mothers, excluding uptake of PAH via the milk (Bulay, 1970; Bulay & Wattenberg, 1971). A similar result was observed in A and C57Bl mice; furthermore, lung tumours were induced in Ha/ICR strain mice and liver tumours in the offspring of all three strains (Nikonova, 1977). Transplacental carcinogenesis has also been reported in rabbits (Beniashvili, 1978).

Benzo[*a*]pyrene was given subcutaneously to strain A and C57Bl mice at 4 or 6 mg per animal once or twice on days 18 and 19 of gestation. The progeny of dams given a single dose of 4 mg had a significantly increased number of adenomas; the 6-mg dose induced 77% lung tumours in A mice and 12% in controls. The maximum dose, 12 mg, induced 32% liver tumours in male C57Bl mice and 9% in females, with 1% in controls (Nikonova, 1977).

7.7.4.2 Pyrene

The tumour incidence in the offspring of strain A mice was not increased by two subcutaneous injections of 6 mg pyrene on days 18 and 19 of pregnancy (Nikonova, 1977).

7.8 Special studies

Adverse effects of PAH unrelated to cancer have also been seen. Proliferating tissues such as bone marrow, lymphoid organs, gonads, and intestinal epithelium are affected, but the major target organs seemed to be those of the haematopoietic and lymphoid systems.

7.8.1 Phototoxicity

Photodynamic compounds can generate superoxide anion radicals in the presence of near ultraviolet light. In the absence of oxygen, they act as photoreducing agents. The main effect is epidermal damage.

7.8.1.1 Anthracene

Increased dermal sensitivity to ultraviolet irradiation was seen in hairless mice after pretreatment with anthracene, but photocarcinogenesis was not significantly increased (Forbes et al., 1976; see also Table 92). In guinea-pigs treated six times on the dorsal skin with anthracene and then irradiated with ultraviolet light, a photoirritant reaction was observed that reached a maximum after a few hours but had faded by 24 h (Lovell & Sanders, 1992).

Petroleum can photosensitize human skin to sunlight, resulting in erythema and pigmentation. Petroleum also enhanced the immunomodulatory effects of ultraviolet radiation on mammalian skin, with depletion of antigen-presenting cells which play a critical role by presenting antigen to thymus-derived lymphocytes. In tests of PAH that are present at relatively high concentrations in crude oils, anthracene but not phenanthrene or benzo[*a*]pyrene induced

photosensitization of mouse skin *in vivo* and *in vitro*. Exposure of skin sections to anthracene at 5 µg/ml, equivalent to 125 ng/mouse, reduced the numbers of antigen-presenting cells and of Thy-1-positive dendritic cells (Burnham & Rahman, 1992).

7.8.1.2 *Benzo[a]pyrene*

Benzo[*a*]pyrene in the presence of near-ultraviolet radiation (290–400 nm) had phototoxic effects, observed as haemolysis of human erythrocytes and inactivation of *Escherichia coli* (Kagan et al., 1989). A significant shortening of tumour-free survival was seen in in Balb/c mice exposed to ultraviolet irradiation before dermal treatment with benzo[*a*]pyrene. Ultraviolet irradiation had a systemic effect, enhancing subsequent dose-dependent tumour induction by benzo[*a*]pyrene (Gensler, 1988).

7.8.1.3 *Pyrene*

In six guinea-pigs given 5 µmol–5 mmol of pyrene dissolved in ethanol, a strong phototoxic reaction was observed 20 h after ultraviolet A irradiation at 1×10^3 J/m^2 (320–400 nm) (Kochevar et al., 1982).

When mast cells were isolated from Sprague-Dawley rats, incubated with 25 µmol/litre pyrene, and irradiated with ultraviolet B radiation (280–320 nm) at 60 kJ/m^2, they released 80% of their serotonin (Gendimenico & Kochevar, 1984).

7.8.1.4 *Comparisons of individual PAH*

The phototoxic effects of benzo[*a*]pyrene, benz[*a*]anthracene, indeno[1,2,3-*cd*]pyrene, fluoranthene, and perylene were compared by treating human fibroblasts with these PAH and then irradiating them with ultraviolet light (< 400 nm). A good correlation was found between the phototoxic effects and known carcinogenic potency: benzo[*a*]pyrene and indeno[1,2,3-*cd*]pyrene were highly toxic, benz[*a*]anthracene was distinctly toxic, fluoranthene slightly toxic, and perylene not cytotoxic (Bauer et al., 1985).

7.8.2 *Immunotoxicity*

7.8.2.1 *Benzo[a]pyrene*

(*a*) *Intraperitoneal and intratracheal injection*

Mice injected intraperitoneally with a single dose of 50, 100, or 200 mg/kg bw benzo[*a*]pyrene showed reduced thymic and splenic weights on day 5, and the splenocyte antibody-forming response of immunized mice was reduced by 60–90%. At the higher concentrations, lymphocyte proliferation

was decreased significantly. Benzo[*a*]pyrene altered both humoral and cellular immunity; B cells were more susceptible than T cells (Xue et al., 1991).

In a study of the effect of accumulation of benzo[*a*]pyrene in lymphoid organs on humoral immunity, B6C3F$_1$ mice were given intratracheal instillations of 0.4, 4, or 40 mg/kg bw daily for seven days and immunized with sheep erythrocytes one day later; then, the number of antigen-specific, antibody-forming cells was measured. The spleen showed a decrease, but lung-associated lymph nodes showed either decreased (up to 60%) or increased (up to 100%) numbers, depending on whether sheep erythrocytes were given intratracheally or intraperitoneally (Schnizlein et al., 1987).

(*b*) *Dermal exposure*

After dermal exposure of female B6C3F$_1$ mice to 0, 5, 20, or 40 mg/kg bw per day for 14 days, dose-dependent suppression of the antibody-forming cell response to sheep erythrocytes was seen both *in vivo* and *in vitro*, the number at 40 mg/kg bw being about 30% of the control value. The immunosuppression was similar when benzo[*a*]pyrene was administered at a high dose subcutaneously in corn oil. The antibody-forming cell response recovered by about day 60 after dermal exposure, while no recovery was seen after subcutaneous injection (Parrott et al., 1989).

Female B6C3F1 mice were exposed to benzo[*a*]pyrene at 0, 0.625, 2.5, 5, 20, or 40 mg/kg bw per day for 28 days and were injected intravenously with sheep erythrocytes on days 11 and 25. An antigen-specific enzyme-linked immunosorbent assay to sheep erythrocyte membranes was used to measure the primary immunoglobulin M response on day 15 and the secondary immunoglobulin G response on day 30. Significant suppression of the primary immunoglobulin M response was observed at doses of 5 mg/kg bw per day and more, but the serum titres of animals treated with 0.625 or 2.5 mg/kg bw per day did not differ from those of vehicle controls. The secondary immunoglobulin G response was significantly decreased at all doses (Deal, 1995).

(*c*) *Comparisons of benzo[a]pyrene and benzo[e]pyrene*

The immunotoxic effects of benzo[*a*]pyrene and benzo[*e*]pyrene have been compared in several studies. In most, benzo[*a*]pyrene was clearly immunosuppressive, whereas benzo[*e*]pyrene was inactive.

In vivo: After activation of splenic lymphocytes, mice were given single intraperitoneal injections of benzo[*a*]pyrene and benzo[*e*]pyrene at 2.5, 10, or 50 mg/kg bw for 24 or 48 h before sacrifice. Mononuclear cell populations were then assayed for AHH activity, blastogenesis, antigen-specific cell-mediated cytotoxicity, and the percentage of macrophages. Neither PAH had a significant effect on blastogenesis. Benzo[*a*]pyrene suppressed cell-mediated cytotoxicity

of T cells by 40–80%, while benzo[*e*]pyrene had no effect. These results suggested that mitogen-activated, AHH-induced splenic lymphocytes metabolize benzo[*a*]-pyrene to immunocytotoxic metabolites. T cells are probably activated by early stimulation of T suppressor cells accompanied by an increase in T suppressor cell factors (Wojdani & Alfred, 1984).

B6C3F$_1$ mice were given 10 daily subcutaneous injections of benzo[*a*]pyrene and benzo[*e*]pyrene at doses of 5, 20, or 40 mg/kg bw over a 14-day period. Three to four days after the last dose, immune function was evaluated. Benzo[*a*]pyrene reduced the number of immunoglobulin M and G antibody plaque-forming cells in response to sheep erythrocytes and lipopolysaccharide, and the TNP-Ficoll plaque-forming cell response was depressed by 77%. These changes indicate altered differentiation and antibody production in mature B cells. No change in the plaque-forming cell response was observed after exposure to benzo[*e*]pyrene (Dean, J.H. et al., 1983).

Mice were given subcutaneous injections of benzo[*a*]pyrene or benzo[*e*]pyrene at 5, 10, or 40 mg/kg bw daily for 11 days, followed by an injection of sheep erythrocytes. Benzo[*a*]pyrene suppressed the antibody response to DNP-Ficoll and sheep erythrocytes but not to lipopolysaccharide; benzo[*e*]pyrene was not immunosuppressive (White & Holsapple, 1984).

In vitro: When benzo[*a*]pyrene was added to cultured mouse spleen cells *in vitro*, a metabolism-activating system was not required to produce immunosuppression. Furthermore, addition of S9 did not increase the degree of immunosuppression produced by benzo[*a*]pyrene (White & Holsapple, 1984).

In several antibody generating systems *in vitro*, both benzo[*a*]pyrene and benzo[*e*]pyrene caused a significant, dose-dependent suppression of the T-dependent and polyclonal antibody responses. A similar result was found *in vitro* after a 14-day exposure of mice to 40 mg/kg bw benzo[*a*]pyrene *in vivo*, with 98% suppression of T-cell-dependent antibody. Benzo[*a*]pyrene-induced suppression is multicellular, and the greatest sensitivity is found early in the immune response. Since benzo[*e*]pyrene induced immunosuppression only at high concentrations, the preparation may have been contaminated with benzo[*a*]pyrene (Blanton et al., 1986).

Benzo[*a*]pyrene and seven of its metabolites were evaluated for their ability to suppress the antibody-forming cell response to sheep erythrocytes *in vitro*. Direct addition of benzo[*a*]pyrene or its 7,8-diol to splenocyte cultures induced similar dose-dependent suppression of the antibody response to the T-dependent antigen, sheep erythrocytes. In contrast, exposure to the 4,5-diol, 9,10-diol, 6,12-dione, 3-hydroxy, and 4,5-epoxide metabolites resulted in decreased antibody responses only with high concentrations; these were associated with decreased viability of the cultures. In addition, co-incubation with the cytochrome P450 inhibitor α-naphthoflavone attenuated the suppressive effects of benzo[*a*]pyrene and benzo[*a*]pyrene 7,8-diol (Kawabata & White, 1987).

The anti-sheep erythrocyte plaque-forming cell and mixed lymphocyte responses were inhibited in murine splenic lymphocytes treated with benzo[*a*]pyrene at concentrations ranging from 10^{-4} to 10^{-8} mol/litre, with maximal depression at 10^{-5} mol/litre. Benzo[*e*]pyrene at the same concentrations did not suppress these responses (Urso et al., 1986).

7.8.2.2 *Dibenz[a,h]anthracene*

The immunosuppressive effects of dibenz[*a,h*]anthracene and 3-methylcholanthrene were studied in AHH-inducible (C57Bl/6) and non-inducible (DBA/2N) mice after intraperitoneal injection of 25, 50, or 100 mg/kg bw five days before challenge with sheep erythrocytes or after administration by gavage of 10–200 mg/kg bw per day for four days. Immunosuppression occurred in both strains but was more pronounced in the C57Bl mice after intraperitoneal injection. The DBA mice were more susceptible to 3-methylcholanthrene given by gavage. The authors suggest that PAH are rapidly metabolized and excreted after gavage in AHH-inducible mice, whereas in non-inducible mice they are absorbed and distributed to the target organs. AHH inducibility may thus play an important role in the immunosuppressive activity of PAH (Lubet et al., 1984a,b; see also section 7.5.2.1).

7.8.2.3 *Fluoranthene*

Murine bone-marrow stromal cells were used as a matrix for the growth and limited development of precursor B cells *in vitro*, thus mimicking B lymphopoiesis *in vivo*. Fluoranthene acutely suppressed B lymphopoiesis, and the precursor B cell populations exposed to 50 µg/ml fluoranthene disappeared within two weeks. Lymphotoxicity was mediated by fluoranthene-induced programmed cell death (apoptosis): 5 µg/ml reduced precursor B cell recoveries by >95% within one or two weeks. Lower doses altered the dynamics of B cell lymphopoiesis, leading to accumulation of precursor B cells (Hinoshita et al., 1992).

7.8.2.4 *Naphthalene*

Naphthalene did not adversely affect the immune response in CD-1 mice of each sex given up to 133 mg/kg bw per day orally for three months. No alteration was seen in the lymphocyte proliferative response to the T-cell mitogens concanavalin A and phytohaemagglutinin, in the delayed hypersensitivity response to sheep erythrocytess, or in the popliteal lymph node response. Furthermore, bone-marrow function was not altered (Shopp et al., 1984).

Naphthalene is known to induce pulmonary and renal toxicity which is mediated by its reactive, electrophilic metabolites 1,2-naphthalene oxide, 1-naphthol, and 1,4-naphthoquinone. The immune response of mice was,

however, unaffected by a 90-day oral treatment with up to 25% of the LD_{50} value. Furthermore, the number of antibody-forming cells in splenocyte cultures was not affected by concentrations of naphthalene up to 200 µmol/litre; however, 200 µmol/litre of 1-naphthol and 7–20 µmol/litre of 1,4-naphthoquinone suppressed the antibody-forming cell response and decreased cell viability. Splenic microsomes were unable to metabolize naphthalene, whereas liver microsomes generated 1,2-naphthalene diol and 1-naphthol. It was concluded that diffusion of liver metabolites to the spleen is insufficient to induce immunotoxicity (Kawabata & White, 1990).

7.8.2.5 Comparisons of individual PAH

The effects of exposure to each of 10 PAH on the immunoglobulin M antibody response to sheep erythrocytes were examined in $B6C3F_1$ mice, with a further investigation of the relationship of the *Ah* gene complex to the immunosuppression. Mice were given subcutaneous injections over 14 days, and splenic antibody-forming cells were evaluated after immunization with sheep erythrocytes. Significant decreases of 55–91% were observed after treatment with dibenz[*a,c*]anthracene, dibenz[*a,h*]anthracene, benz[*a*]-anthracene, and benzo[*a*]pyrene, but no significant effects were observed with anthracene, benzo[*e*]pyrene, perylene, or chrysene. Generally, the structure–activity relationship for immunosuppression was correlated strongly with that for carcinogenicity. Non-inducible DBA/2 mice had greater PAH-induced immunosuppression than AHH-inducible $B6C3F_1$ mice (White et al., 1985).

7.8.2.6 Exposure in utero

Oral administration of benzo[*a*]pyrene at 2 mg/kg bw to Wistar rats on day 19 of gestation induced a relative decrease in the number of thymic glucocorticoid receptors in males and a relative increase in females at six weeks of age. In animals treated at six weeks of age, no change in the number or affinity of steroid receptors was seen in males, but there was a 40% decrease in females. The investigators concluded that benzo[*a*]pyrene binds to pre-encoded hormone receptors and interferes with their maturation. When benzo[*a*]pyrene was given on day 15 of gestation, a 30% decrease in the number of receptors was seen in the offspring (Csaba et al., 1991; Csaba & Inczefi-Gonda, 1992).

Strong suppression of the anti-sheep erythrocytes plaque-forming cell response, the mixed lymphocyte response, and the graft-versus-host response *in vivo* were seen in the progeny of female mice that had been treated intraperitoneally with benzo[*a*]pyrene at a dose of 150 mg/kg bw during mid-gestation (11–13 days). Immunodeficiency was seen within one week after birth and persisted for 18 months. Benzo[*a*]pyrene induced marked disorientation of T cells up to four weeks postnatally. Disruption of T-cell differentiation during ontogenesis was suggested, implying decreased resistance to the

development of neoplasia (Urso & Gengozian, 1980; Urso & Johnson, 1987). A two- to fourfold reduction in maternal leukocytes was seen within five days of an intraperitoneal administration of 150 µg benzo[*a*]pyrene on day 12 of gestation, which persisted to day 10 *post partum* and was attributed to lymphocyte depletion. Thus, benzo[*a*]pyrene exacerbated the depression of leukocytes seen during pregnancy. After intraperitoneal injection, it can reach the lymphocytes, where metabolic activation may take place. Investigation of the maternal T-cell population, both during pregnancy and *post partum*, showed exacerbated decreases in the number of thymocytes present in the thymus. In the spleen, the number of thymocytes positive for Lyt I antigens was also depressed, whereas the number of Lyt 2+ cells was enhanced, reaching levels > 700 times those of controls. These results demonstrate disruption of the maternal T-cell repertoire (Urso et al., 1988; Urso & Johnson, 1988).

7.8.2.7 *Mechanisms of the immunotoxicity of PAH*

Although the mechanism(s) by which PAH adversely affect the immune system has not been defined, several have been proposed (Ladics & White, 1996). The mechanism that is most consistent with that accepted for the carcinogenesis of PAH is that their immunotoxicity is due not to the parent compound but to the reactive metabolites formed. These include the 7,8-diol-9,10-epoxide for benzo[*a*]pyrene (Kawabata & White, 1987) and the 3,4-diol-1,2-epoxide for the synthetic compound 7,12-dimethylbenz[*a*]anthracene (Ladics et al., 1991). Several investigators have suggested that the mechanism of action of PAH results from their ability to enter the cell membrane and disrupt transduction of transmembrane signals and/or alter the conformation of receptors (Pallardy et al., 1992; Thurmond et al., 1989). It has also been suggested that PAH alter the immune response as a result of their ability to induce inappropriate alterations in the levels of various cytokines, such as interleukins 1 and 2 (Lyte et al., 1987; Meyers et al., 1988; Pallardy et al., 1989), and this mechanism is under active investigation. While interaction with the Ah receptor has been suggested as a possible mechanism of action of PAH, the contradictory results from immunotoxicological studies with genetically different strains of mice (high and low AHH responders) must be resolved. Studies primarily conducted *in vitro* support the hypothesis that the immunomodulatory effects of PAH are the result of alterations in calcium mobilization (Burchiel et al., 1991).

7,12-Dimethylbenz[*a*]anthracene has been extensively studied as a prototypic immunotoxicant. Exposure to this compound has been reported to decrease natural killer cell activity (Kimber et al., 1986), to decrease resistance to challenge with tumour cells (Dean et al., 1986), and to increase susceptibility to chemically induced tumours (Elmets et al., 1988). *In vitro*, 7,12-dimethylbenz[*a*]anthracene has been shown to suppress cytotoxic T-cell

activity, possibly by inducing defects in antigen recognition (House et al., 1989) and/or in cytokine production (House et al., 1988). Oral administration of a cumulative dose of 14 mg/kg bw of 7,12-dimethylbenz[*a*]anthracene over 14 days suppressed proliferative responses by about 50% in splenic lymphocytes and by more than 70% in gut-associated lymphocytes (Burchiel et al., 1990).

7,12-Dimethylbenz[*a*]anthracene-mediated immunosuppression was shown to persist for at least eight weeks (Ward et al., 1986; Burchiel et al., 1988). This compound also decreased resistance to bacterial (Ward et al., 1984) and viral (Selgrade et al., 1988) infections. Although no studies have been carried out on humans *in vivo*, the IC_{50} for inhibition of the response of human tonsillar lymphocytes in culture to foreign tissue antigens was 10–40 µmol/litre; however, antibody secretion was affected only at a concentration of 100 µmol/litre (Wood & Holsapple, 1993).

7.8.3 *Hepatotoxicity*

7.8.3.1 *Benzo[a]pyrene*

Non-responsive strains of mice (C57Bl/6, C3H/HeN, and Balb/cAnN) had increased relative liver weights after they were fed for 180 days on a diet containing benzo[*a*]pyrene resulting in an intake of 120 mg/kg bw per day (Robinson et al., 1975).

7.8.3.2 *Comparisons of individual PAH*

The induction of several enzymes has been correlated with cancer promotion. Wistar rats treated orally with 100 mg/kg bw per day of benzo[*a*]pyrene, benz[*a*]anthracene, anthracene, chrysene, or phenanthrene for four days showed induction of cytosolic aldehyde dehydrogenase activity. Benzo[*a*]pyrene and benz[*a*]anthracene were much more effective than the other substances, increasing liver weights by 27 and 19%, respectively (Törrönen et al., 1981).

The extent of liver regeneration that determines the ability to induce a proliferative response was investigated in partially hepatectomized rats fed diets containing various PAH for 10 days. Doses of 51.4 mg/kg bw per day acenaphthene or 180 mg/kg bw per day fluorene induced a significant increase in the rate of liver regeneration, but 15.4 mg/kg bw per day acenaphthene, 51.4 mg/kg bw per day benzo[*a*]pyrene, or 514 mg/kg bw per day pyrene, anthracene, or phenanthracene had no effect (Gershbein, 1975).

7.8.4 *Renal toxicity*

Rats given 50–150 mg/kg bw benzo[*a*]pyrene orally over four days showed moderate induction of renal microsomal carboxylesterase activity; however, administration of 100 mg/kg bw per day anthracene or phenanthrene had no effect (Nousiainen et al., 1984).

7.8.5 Ocular toxicity of naphthalene

The development of cataracts in rats, mice, and rabbits after application of naphthalene is a toxic peculiarity of this compound. Attempts have been made to clarify the metabolic processes responsible for the formation of insoluble precipitates in the eye.

Cataracts developed in the eyes of rabbits within a few days of repeated oral administrations of 0.5–1 g/kg bw per day. Oral administration was more effective than other modes of application (Pike, 1944). The oxidation products of naphthalene may reach the eye via the bloodstream, where 1,2-naphthoquinone is formed which can react with proteins and other cell components to form insoluble precipitates with a characteristic brown colour (Van Heyningen & Pirie, 1967).

C57Bl/6 mice, which are susceptible to induction of cytochromes P450, were given naphthalene by intraperitoneal administration at 500–2000 mg/kg bw. Cataracts were induced in a dose-dependent manner within 8 h. The effect was reduced by pretreatment with P450 inhibitors and antioxidants and was increased by pretreatment with P450 inducers or glutathione depletors. Cataracts were also induced in a dose-dependent manner by intraperitoneal injection of 1,2- or 1,4-naphthoquinone at 5–250 mg/kg bw (molar potency about 10-fold higher). DBA/2 mice, which are resistant to induction of cytochromes P450, did not develop cataracts. The authors concluded that P450-dependent bioactivation was necessary to form reactive intermediates, which are assumed to be naphthoquinone or a free-radical derivative (Wells et al., 1989).

When a lens cell line from transgenic mice was exposed to the 1,2-dihydrodiol and 1,2-naphthoquinone metabolites of naphthalene, the dihydrodiol did not appear to be toxic but the naphthoquinone induced depletion of glutathione levels. Detoxification of naphthoquinone by the enzyme quinone oxidoreductase prevents formation of a semiquinone radical by two-electron reduction (Russell et al., 1991).

All rats of various strains given naphthalene orally developed cataracts. The authors proposed that naphthalene dihydrodiol is produced in the liver, reaches the aqueous humour, and penetrates the lens, where it is metabolized to naphthoquinone. Feeding of 1-naphthol did not induce opacification (Xu et al., 1992).

7.8.6 Percutaneous absorption

As dermal penetration is one of the major routes of entry after occupational exposure, in-vivo and in-vitro models have been developed to assess it.

In experiments *in vitro* on the dermal penetration of benzo[*a*]pyrene and pyrene in guinea-pigs, pyrene was absorbed mainly by passive diffusion, but benzo[*a*]pyrene was biotransformed during absorption, and the 7,8,9,10-tetrol metabolite of the putative ultimate carcinogen was detected in the receptor fluid of diffusion cells (Ng et al., 1992).

In skin preparations from mice, rats, rabbits, guinea-pigs, marmosets, and humans treated with benzo[*a*]pyrene, both the parent compound and a full spectrum of metabolites were detected, depending on metabolic viability, i.e. previously frozen skin preparations could not metabolize benzo[*a*]pyrene (Kao et al., 1985).

^{14}C-Benzo[*a*]pyrene was administered to the nuchal area of mice at doses of 1.25–125 μg/cm^2, and the animals were sacrificed seven days later. Benzo[*a*]pyrene disappeared rapidly from the application site, at a rate of 6% within 1 h and 40% within 24 h; after seven days, 7% remained at the original site. Most of the benzo[*a*]pyrene was excreted via the hepatobiliary system and found in the faeces, with 35% after 24 h, 58% after 48 h, and 80% after seven days. Only 10% of the radiolabel was detected in urine. Uptake was saturated at doses > 15 μg/cm^2, implying an enhanced risk for tumour induction in the skin epithelium (Sanders et al., 1986).

The binding of benzo[*a*]pyrene to DNA and protein in mouse skin was 15–20 times greater when acetone was used as the vehicle than with a low-viscosity oil (Ingram & Phillips, 1993). While acetone solutions of ^{14}C-benzo-[*a*]pyrene readily penetrated skin from human cadavers, a significantly smaller amount moved from soil into skin. No partitioning of benzo[*a*]pyrene from human skin to plasma was observed. An experiment with rhesus monkeys *in vivo* also showed significantly less absorption from soil (Wester et al., 1990).

7.8.7 *Other studies*

7.8.7.1 *Benzo[k]fluoranthene*

Intraperitoneal injections of benzo[*k*]fluoranthene, a widespread PAH, to rats for three days induced maximal cytochrome P450/448 activity. Liver microsomes were then prepared, and the metabolic profile of benzo[*k*]-fluoranthene was analysed by gas chromatography-mass spectrometry. *trans*-5,6-, 8,9-, and mainly 10,11-dihydrodiols were the primary metabolites in noninduced rats. Pretreatment with PAH resulted in the generation of 5,6 and 8,9 isomers as the main metabolites, due to induction of monooxygenases; secondary metabolism to triols and tetrols was also induced. The putative ultimate carcinogen, 3,4-diol-1,2-epoxy benz[*a*]anthracene, was detected after pretreatment of liver microsomes with benzo[*k*]fluoranthene. The authors concluded that benzo[*k*]fluoranthene is a relevant component of environmental pollution and enhanced the carcinogenic risk of benz[*a*]anthracene (Schmoldt et al., 1981; Jacob et al., 1981a).

7.8.7.2 *Benzo[a]pyrene*

Rats were fed a diet containing 400 mg/kg benzo[*a*]pyrene with or without 2 g/kg of β-carotene. Benzo[*a*]pyrene had no effect on serum retinol levels, but

vitamin A levels were decreased in liver and small intestine at two weeks, with a 30% decline by four weeks. A similar effect was not observed in rats fed β-carotene simultaneously. AHH can be induced by β-carotene, but this may not be the mechanism by which tumours are prevented (Edes et al., 1991).

The role of benzo[*a*]pyrene in the induction of arteriosclerosis was investigated by studying its effects on bovine arterial smooth muscle cells *in vitro*. The number of cells was unchanged by exposure, but the secretion of newly synthesized collagen was decreased. Total cellular DNA was decreased and collagen secretion increased when the cells were preincubated with platelet factors rather than a serum-free medium (Stavenow & Pessah-Rasmussen, 1988).

Microsomes from rat aorta transformed benzo[*a*]pyrene into various carcinogenic and toxic metabolites after induction with 3-methylcholanthrene. Thus, carcinogenic metabolites of PAH deriving from cigarette smoke tars could cause endothelial injury, contributing to the role of cigarette smoking in arteriosclerosis (Thirman et al., 1994).

7.8.7.3 *Phenanthrene*

In a study of the oxidation of phenanthrene by liver microsomes from rats with or without pretreatment with inducers, microsomes from untreated rats produced only *trans*-9,10-diol phenanthrene, but pretreatment with various PAH also led to oxidation at the 1,2 and 3,4 positions. Considerable amounts of the proximal carcinogen 1,2-diol phenanthrene were detected, but the concentration of the ultimate carcinogen 1,2-diol-3,4-epoxide was very low. These results are in accordance with the questionable carcinogenic potency of phenanthrene (Jacob et al., 1982a; see Table 91).

7.8.7.4 *Comparisons of individual PAH*

Activation of platelets by calcium ionophore A23187 can mobilize intracellular stores of calcium ion and stimulate thromboxane biosynthesis, which can be measured as thromboxane B2 synthesis. Benz[*a*]anthracene, chrysene, benzo[*a*]pyrene, and benzo[*ghi*]perylene inhibited thromboxane B2 production in A-23187-induced, washed platelets from rabbits, while anthracene and pyrene appeared to stimulate thromboxane B2 synthesis. Fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, and benzo[*e*]pyrene had little or no effect on activation (Yamazaki et al., 1990).

Benzo[*a*]pyrene, benzo[*k*]fluoranthene, benzo[*b*]fluoranthene, chrysene, benz[*a*]anthracene, pyrene, phenanthrene, and fluoranthene were toxic to human hepatoma cell cultures (HepG2), as measured with neutral red, whereas fluorene, anthracene, acenaphthene, and acenaphthylene were not (Babich et al., 1988).

7.9 Toxicity of metabolites

Derivatives of parent PAH have been tested for mutagenicity and carcinogenicity in a number of experiments in order to assign the effects to definite metabolites or to rank the potency of known metabolites of a parent compound. Some studies addressed the steric factors that determine mutagenic or carcinogenic effects, such as the diastereomers of epoxides and the role of the methyl group in 5- and 6-methylchrysene. These studies are summarized in Tables 93 and 94, although a few of the studies are reported in detail below.

7.9.1 Benzo[a]pyrene

Benzo[*a*]pyrene is metabolized to about 20 primary and secondary oxidized metabolites and to a variety of conjugates. Several metabolites can induce mutations, transform cells, and bind to cellular macromolecules, but the 7,8-diol-9,10-epoxides are presently considered to be the major ultimate carcinogens (DePierre & Ernster, 1978; Pelkonen & Nebert, 1982).

Cytochrome P450-dependent metabolizing activity is low in skin. When ^{14}C-benzo[*a*]pyrene was incubated with arachidonic acid and cytosol prepared from rat, mouse, or human epidermis, benzo[*a*]pyrene 1,6-, 3,6-, and 6,12-quinones and other metabolites were formed. These metabolic reactions were inhibited by selective inhibitors of lipoxygenase, demonstrating that human and rodent skin can metabolize benzo[*a*]pyrene through an arachidonic acid-dependent lipoxygenase pathway (Agarwal & Mukhtar, 1991). In cultures of normal human melanocytes, benzo[*a*]pyrene diols and small amounts of quinones and phenols were detected in the fraction extractable in organic solvents, and glucuronide and sulfate conjugates in the water-soluble fraction (Agarwal et al., 1991).

The (+)*anti*-7,8-dihydrodiol-9,10-epoxide of benzo[*a*]pyrene reacts with DNA to yield almost exclusively the deoxyguanosine adduct, in which the epoxide function has reacted with the amino group at C2 of the guanine base. In mouse skin, this one adduct accounts for about 97% of the total binding of benzo[*a*]pyrene to DNA (Jeffrey et al., 1976).

Benzo[*a*]pyrene 7,8-dihydroxy-9,10-epoxide and two other metabolites (which were detected by trapping with exogenous DNA as described by Ginsberg & Atherholt, 1989) were present in all serum samples from mice of three strains and Sprague-Dawley rats after intraperitoneal injection of 50–200 mg/kg bw benzo[*a*]pyrene. It was concluded that transport can occur via the systemic circulation (Garg et al., 1991).

Benzo[*a*]pyrene or its 4,5-oxide or 7,8-diol-9,10-epoxy metabolite was administered directly into Swiss mouse embryos at 0.1–4 µg per embryo on days 10, 12, and 14 of gestation. The 7,8-diol-9,10-epoxy metabolite was the most potent embryotoxic and teratogenic compound in fetuses examined on

Table 93. Mutagenicity of metabolites of polycyclic aromatic hydrocarbons

Compound	Metabolite	Test system	Results	References
Benzo[*b*]fluoranthene	9,10-Diol	Mouse, dermal	Positive	Amin et al. (1991a)
Benzo[*j*]fluoranthene	9,10-Diol	Mouse, dermal	Positive	LaVoie et al. (1980; 1982b)
Benzo[*k*]fluoranthene	9,10-Diol		Positive	LaVoie et al. (1980)
Benzo[*a*]pyrene	7,8-Diol-9,10-epoxide (*syn* + *anti*)	Unscheduled DNA binding	Positive	Gill et al. (1991)
		DNA synthesis	Positive	
	7,8-Diol-9,10-epoxide (2 stereoisomers)	Hamster embryo cells, transformation	Positive	Mager et al. (1977)
	1-Hydroxy	Reverse mutation, *S. typhimurium* TA1538, TA98, TA100	Positive (±S9)	Schoeny et al. (1985)
	2-Hydroxy		Negative (S9)	
	3-Hydroxy		Positive (+S9)	
	4-Hydroxy		Negative (±S9)	
	6-Hydroxy		Positive (–S9)	
	7-Hydroxy		Positive (+S9)	
	9-Hydroxy		Positive (+S9)	
	10-Hydroxy		Negative (±S9)	
	12-Hydroxy		Positive (±S9)	
	1,6-Quinone		Negative (–S9)	
	3,6-Quinone		Positive (+S9)	
	4,5-Quinone		Negative (±S9)	
	6,12-Quinone		Negative (±S9)	
	trans-4,5-Diol		Negative (±S9)	
	cis-4,5-Diol		Positive (+S9)	
	trans-7,8-Diol		Negative (±S9)	
	trans-9,10-Diol		Negative (±S9)	
	4,5-Epoxide		Positive (±S9)	

Table 93 (contd)

Compound	Metabolite	Test system	Results	References
Benzo[a]pyrene (contd)	1-Hydroxy	Forward mutation, *S. typhimurium* TM677	Positive (+S9)	Schoeny et al. (1985)
	2-Hydroxy		Negative (±S9)	
	3-Hydroxy		Negative (±S9)	
	4-Hydroxy		Negative (±S9)	
	6-Hydroxy		Negative (±S9)	
	7-Hydroxy		Positive (±S9)	
	9-Hydroxy		Negative (±S9)	
	10-Hydroxy		Negative (±S9)	
	12-Hydroxy		Negative (±S9)	
	1,6-Quinone		Negative (±S9)	
	3,6-Quinone		Negative (±S9)	
	4,5-Quinone		Negative (±S9)	
	6,12-Quinone		Negative (±S9)	
	trans-4,5-Diol		Positive (+S9)	
	cis-4,5-Diol		Positive (+S9)	
	trans-7,8-Diol		Positive (+S9)	
	trans-9,10-Diol		Negative (±S9)	
	cis-7,8-Diol		Positive (+S9)	
	4,5-Epoxide		Positive (±S9)	
Benzo[*e*]pyrene	9,10-Diol-11,12-epoxide	Bacterial and mammalian cells	Weakly positive	Wood et al. (1980)
Chrysene	1,2-Diol	Bacterial and mammalian cells	Positive	Wood et al. (1977)
	1,2-Diol-3,4-epoxide			
	5,6-Oxide	*S. cerevisiae*, D4-RDII	Positive	Siebert et al. (1981)

Table 93 (contd)

Compound	Metabolite	Test system	Results	References
Cyclopenta[*cd*]pyrene	3,4-Diol	Bacterial and mammalian cells, mutagenicity and transformation	Positive	Gold et al. (1980)
	trans-3,4-Epoxide	Calf thymus DNA, DNA binding	Positive	Beach et al. (1993)
Fluoranthene	2,3-Diol	Bacterial cells	Positive	LaVoie et al. (1982a)
	2,3-Diol-1,10b-epoxide	Bacterial cells:	Positive	Rastetter et al. (1982)
	1,10b-Diol-2,3-epoxide	*S. typhimurium* TM677	Weakly positive or negative	
Dibenz[*a,h*]anthracene	3,4-Diol	Bacterial cells	Most mutagenic compound among 3 dihydrodiols	Wood et al. (1978)
	5,6-Epoxide transformation	Hamster embryo cells,	Positive	Huberman et al. (1972); Marquardt et al. (1972)
Dibenz[*a,h*]pyrene	1,2-Diol	Bacterial cells	Positive	Wood et al. (1981)
Dibenz[*a,l*]pyrene	3,4-Diol	Bacterial cells	Positive	Wood et al. (1981)
5-Methylcholanthrene	1,2-Diol and 7,8-diol	Bacterial cells	Positive	Hecht et al. (1978)
	5-Hydroxy	*S. typhimurium* TA100	Weakly positive	Amin et al. (1979)
1-Methylphenanthrene	1,4-Diol 5,6-Diol	Bacterial cells	Positive	LaVoie et al. (1981c)
Phenanthrene	1,2-Diol-3,4-epoxide	Bacterial and mammalian cells	Positive	Wood et al. (1979)
	9,10-Oxide	*S. cerevisiae* D4-RDII	Positive	Siebert et al. (1981)

S9, 9000 x *g* microsomal fraction of liver

Table 94. Carcinogenicity of metabolites of polycyclic aromatic hydrocarbons

Compound	Metabolite	Species, route of administration	Type of investigation, duration, dose	Results	References
Benz[*a*]anthracene	3,4-Diol and 3,4-diol-1,2-epoxide	Mouse, intraperitoneal	Carcinogenicity, 26 weeks	3*R*,4*R*-Diol: 71% with tumours; other enantiomer not tumorigenic; 3,4-diol-1,2-epoxide: 100% with tumours; other enantiomer: 42%; control: 13%	Wislocki et al. (1979)
Benzo[*b*]fluoranthene	9,10-Diol	Mouse, dermal	Initiation	Positive	Amin et al. (1991a)
Benzo[*j*]fluoranthene	9,10-Diol	Mouse, dermal	Initiation	Positive	LaVoie et al. (1980, 1982b)
	4,5-Diol, 9,10-Diol 2,3-Diol	Mouse, dermal	Initiation	2,3-Diol: 5–11%; 4,5-diol: 78–100%; 9,10-diol: 60%; control: 10%	Rice et al. (1987)
Benzo[*c*]phenanthrene	1,2-Diol 3,4-Diol 5,6-Diol 1,2-Epoxy-3,4-diol	Mouse, dermal	Initiation	1,2-Diol: 3–4% with tumours; 3,4-diol: 28–47% with tumours; 5,6-diol: 3–7% with tumours; 1,2-epoxy-3,4-diol: 80–90% with tumours; control: no tumours	Levin et al. (1980)
	3,4-Diol-1,2-epoxide	Mouse, dermal	Initiation	95–100% with tumours; control: 10%	Amin et al. (1993)
	anti-3,4-Diol-1,2-epoxide	Rat, intramammary	Carcinogenicity 1 x 12.2 µmol	100 % with mammary tumours; vehicle control: 3%	Hecht et al. (1994)

Table 94 (contd)

Compound	Metabolite	Species, route of administration	Type of investigation, duration, dose	Results	References
Benzo[*a*]pyrene	7,8-Diol	Mouse,dermal	Carcinogenicity, 43 µg every 2 weeks, 60 weeks	100% with skin tumours	Conney (1982)
	anti-7,8-Diol-9,10-epoxide	Rat, intramammary	Carcinogenicity, 1 x 12.2 µmol	47% with mammary tumours; vehicle control 3%	Hecht et al. (1994)
Benzo[*e*]pyrene	4,5-Diol 9,10-Diol	Mouse, dermal	Initiation	No significant effect et al. (1980,	Buening et al. (1980); Slaga
	1981) 9,10-Diol 9,10-Diol-11,12-epoxide	Mouse, newborn, intraperitoneal	Carcinogenicity	No induction of pulmonary tumours; significant induction of hepatic tumours	Buening et al. (1980); Chang et al. (1981)
Chrysene	1,2-Diol	Mouse, dermal	Initiation 0.4, 1.25, and 4 µmol/ animal, 1 x	39, 60, and 79% with tumours; control: 7%	Levin et al. (1978)
	1,2-Diol 3,4-Diol	Mouse,dermal	Initiation	1,2-Diol: positive 3,4-Diol: negative	Slaga et al. (1980); Chang

Table 94 (contd)

Compound	Metabolite	Species, route of administration	Type of investigation, duration, dose	Results	References
Chrysene (contd)	1,2-diol, 1,2-diol-3,4-epoxide	Mouse, newborn	Carcinogenicity	Induction of pulmonary adenomas	Buening et al. (1979); Chang et al. (1983)
Dibenz[*a,h*]anthracene	3,4-Diol	Mouse, dermal	Carcinogenicity	Induction of skin tumours; induction of pulmonary tumours in newborn mice	Buening et al. (1979); Slaga et al. (1980), 1981)
	5,6-Epoxide	Mouse, dermal	Initiation	Poorly active	Van Duuren et al. (1967)
Dibenzo[*a,h*]pyrene	1,2-Diol	Mouse, dermal, intraperitoneal	Initiation	Positive	Wood et al. (1981)
Dibenzo[*a,i*]pyrene	3,4-Diol	Mouse, dermal, intraperitoneal	Initiation	Positive	Wood et al. (1981)
Indeno[1,2,3-*cd*]pyrene	1,2-Diol, 1,2-Oxide 8-Hydroxy	Mouse, dermal	Initiation	1,2-Diol: 80%, 1,2-oxide, 80%; 8-hydroxy, 30% tumour-bearing animals	Rice et al. (1986)

Table 94 (contd)

Compound	Metabolite	Species, route of administration	Type of investigation, duration, dose	Results	References
5-Methylcholanthrene	1,2-Diol 7,8-Diol 9,10-Diol	Mouse, dermal	Initiation, 3 µg/animal, 10 x in 20 days	1,2-Diol: 19/20 papillomas, 7/20 carcinomas; 7,8-diol:10/20 papillomas, no carcinomas; 9,10-diol: no papillomas, no carcinomas	Hecht et al. (1980)
	5-Hydroxy	Mouse, dermal	Initiation	45–95% with skin tumours; solvent control: 5%	Amin et al. (1981)
	1,2-Diol-3,4-epoxide	Mouse, newborn	Carcinogenicity	Induction of pulmonary tumours	Amin et al. (1991b)
Phenanthrene	1,2-Diol, 3,4-Diol 9,10-Diol	Mouse, dermal	Initiation	Very weakly positive	Wood et al. (1979)
	1,2-Diol-3,4-epoxide	Mouse, newborn	Carcinogenicity	No induction of pulmonary tumours	Buening et al. (1979)

day 18, causing 85% embryolethality and 100% malformations. Benzo[*a*]pyrene and the other metabolite did not increase the incidence of malformations significantly (Barbieri et al., 1986).

7.9.2 *5-Methylchrysene*

Investigations of the reaction of 5-methylchrysene diol epoxide enantiomers with DNA bases and in *S. typhimurium* showed the importance of both the absolute configuration and the position of the methyl group. The 5-methylchrysene 1*R*,2*S*-diol-3*S*,4*R*-epoxide, with the methyl group and epoxide ring in the same bay region, were the most reactive (Melikian et al., 1988).

7.9.3 *1-Methylphenanthrene*

Incubation of rat liver preparations with 1-methylphenanthrene gave rise to the 3,4- and 5,6-dihydrodiols of 1-hydroxymethylphenanthrene, 1-methylphenanthrene, 1-hydroxymethyl-phenanthrene, and unidentified derivatives as metabolites; the dihydrodiols were mutagenic in the presence of an exogenous metabolizing system (LaVoie et al., 1981b).

7.10 Mechanisms of carcinogenicity

7.10.1 *History*

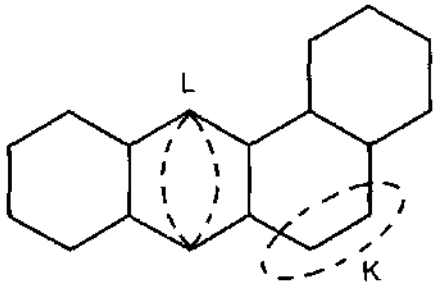
In the early 1940s, theoretical chemistry was used to predict the chemical reactivity of PAH. Pullman (1945, 1947) introduced the terms K- and L-region to describe the reactivity of PAH based on Hückel molecular orbital calculations (see Figure 9). Later, the term 'bay region' was introduced for molecular substructures that contribute to the formation of some ultimate carcinogens. Discrete values of complex delocalization energies at the K- and L-regions of a PAH were correlated with its carcinogenic potency. At the time, however, there was only a limited database on metabolic processes and little experimental confirmation, and the K-region theory was later found to be incompatible with the results of experimental work.

7.10.2 *Current theories*

Miller & Miller (1977) proposed the theory of reactive electrophiles in chemical carcinogenesis. According to this theory, PAH are activated by microsomal enzymes to proximate and finally ultimate carcinogens, which are characterized by an electrophilic centre that can react with nucleophilic sites on macromolecules such as DNA, RNA, and protein.

After the discovery that diol epoxides are metabolites of PAH (Sims & Grover, 1974), a theoretical model was presented by Jerina et al. (1976), who found that synthesized arene oxides were mutagenic without metabolic

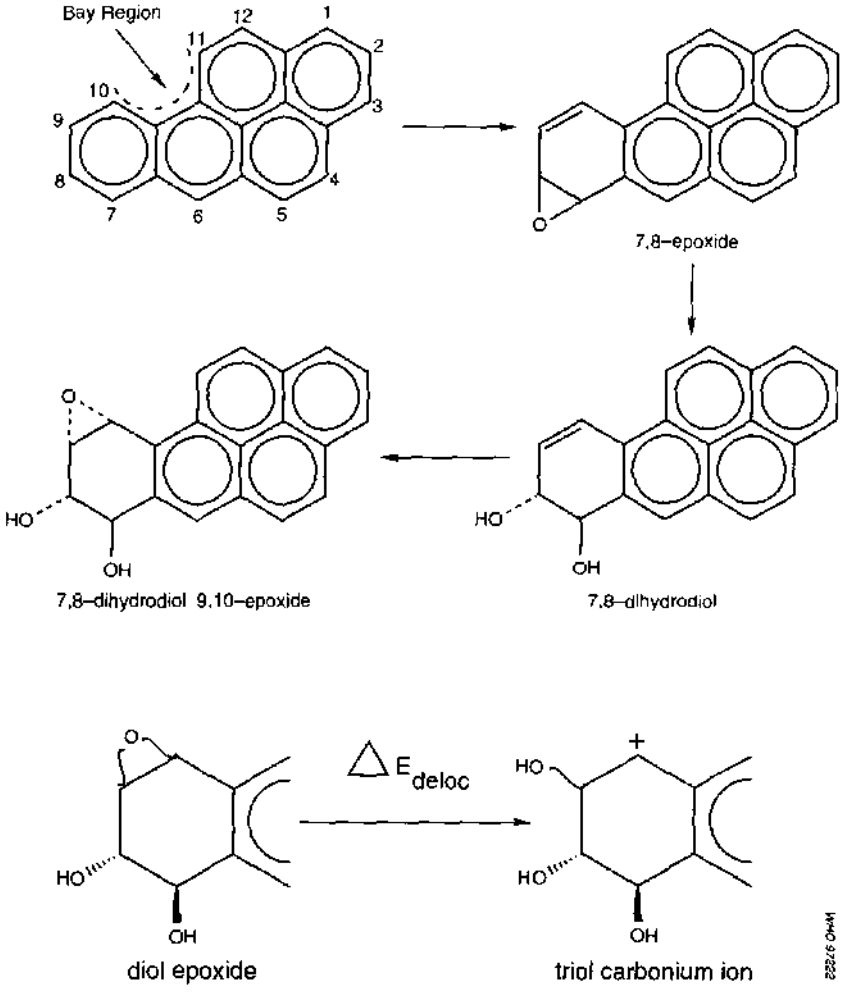
Figure 9. Benz[*a*]anthracene with regions of low bond localization energy (K region) and low para localization energy (L region)



activation. The bay-region theory states two prerequisites for carcinogenic potency: The epoxide group of an ultimate metabolite must be part of a bay region (see Figure 10), and the hydroxy groups of the diol epoxide are preferentially located in the 'pre-bay region'. The presence of the epoxide group on a saturated benzene ring in a bay region facilitates ring opening, i.e. the delocalization energy forming the carbonium ion is higher. This is important for reactions with DNA via a carbonium ion, which is an alkylating agent. For example, the metabolic pathway of benzo[*a*]pyrene is hypothesized to start with a 7,8 oxidation followed by hydrolysis to 7,8-dihydrodiol, and terminated by 9,10-oxidation, yielding the ultimate carcinogen 7,8-dihydrodiol-9,10-epoxide. Calculation of the carbonium ion delocalization energies by the pertubational molecular orbital method results in a rough correlation with experimentally determined carcinogenic potency.

The number of PAH tested for carcinogenicity in experimental animals doubled between the time that Jerina et al. (1976) published their work and 1980, to 50 compounds. A calculation of the carbonium ion delocalization energies at that time (Qianhuan, 1980) revealed a deviation from the energy-carcinogenicity correlation for compounds that had not been investigated by Jerina et al. To avoid the shortcomings of the bay-region theory, Qianhuan took into account the data on all PAH that had been tested for carcinogenicity and postulated the di-region theory, a bifunctional electrophilic theory based on the assumption that formation of two carbonium ions on the same PAH is responsible for carcinogenic activity. A quantitative equation involving the delocalization energies of the twin active regions was deduced. Principally, all of the already defined key regions of PAH (M, E, K, and L; see Figure 11) were used but with different implications. In this theory, the metabolic activation of PAH is dependent on two factors, a geometric and an energy factor. The angular ring, the subangular ring, and an active K region play decisive roles in carcinogenic potency, and two adequately active, adjacent regions are required.

Figure 10. The bay region dihydrodiol epoxide route of metabolism of benzo[a]pyrene



From Jerina et al. (1976)

PAH are proposed to exert carcinogenicity mainly by DNA complementary cross-linking. Qualitative and quantitative data have been presented on the mechanism of formation of PAH–DNA adducts in the radical cation theory. PAH with relatively low ionization potential, which are the most potent carcinogens, are activated via cytochrome P450 by one-electron oxidation (radical cation), whereas PAH with relatively high ionization potential are activated by mono-oxygenation (bay-region diol epoxide). In experiments in rat liver microsomes in which potential DNA adducts were synthesized and used as standards, four depurination products (one-electron oxidation) and one stable product (diol epoxide pathway) of benzo[*a*]pyrene were detected (Cavalieri & Rogan, 1985; Cavalieri et al., 1993; Rogan et al., 1993).

In a review of the evidence for the four mechanisms of PAH carcinogenesis, namely, the diol epoxide mechanism, the radical-cation mechanism, the quinone mechanism, and the benzylic oxidation mechanism, Harvey (1996) concluded that current research provided evidence for all four.

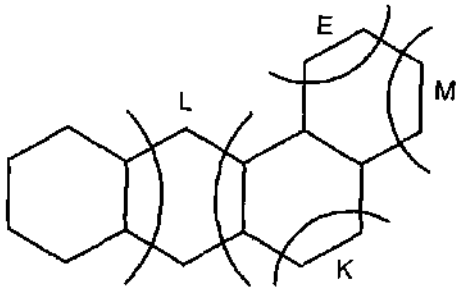
7.10.3 Theories under discussion

A molecular geometrical model has been proposed in which the carcinogenic potency of PAH is predicted from the centre(s) of highest chemical or biochemical reactivity, with the hypothesized introduction of a methyl group into the PAH. A good correlation was found between the predicted carcinogenicity of a series of 50 unsubstituted PAH and the results found in rats and mice. Bioalkylation is suggested to be catalysed by cytosolic methyltransferase with *S*-adenosyl-L-methionine. In an experiment to confirm the model, rats were given subcutaneous injections of benz[*a*]anthracene, and 24 h later the tissue at the application site was homogenized and the assumed metabolites analysed by high-performance liquid chromatography, gas chromatography, and mass spectrometry. The bioalkylation product 7-methylbenz[*a*]anthracene was identified. Six noncarcinogenic hydrocarbons did not yield alkylated metabolites in this experimental approach. The authors concluded that bioalkylation, preferably in the meso-anthranic centres of high reactivity, is a structural prerequisite of carcinogenicity (Flesher & Myers, 1990, 1991; see also section 6.6.2).

The binding of 1-hydroxymethylpyrene to DNA after intraperitoneal injection to rats was similar to the adduct pattern of its active metabolites 1-hydroxymethylpyrene sulfate and 1-chloromethylpyrene with isolated DNA, suggesting secondary activation of hydroxymethyl-PAH sulfates to chloromethyl-PAH (Monnerjahn et al., 1993).

Transformation of PAH via their proximate carcinogens (e.g. benzo[*a*]-pyrene 7,8-diol) to the ultimate carcinogens (e.g. benzo[*a*]pyrene 7,8-diol-9,10-epoxide) is reported to be mediated by cytochrome P450 enzymes; however, two pathways unrelated to P450 have also been discussed. Peroxidase

Figure 11. Key regions for the carcinogenicity of polycyclic aromatic hydrocarbons



M region, metabolic activity region; E region, electrophilic activity region; L region, region of detoxification; K region, amphibolous region

enzymes can transform benzo[*a*]pyrene 7,8-diol to benzo[*a*]pyrene 7,8-diol-9,10-epoxide, but the process requires the presence of superoxide anions, hydrogen peroxides, and hydroxyl radicals produced by polymorphonuclear cells. This reaction was demonstrated in mouse skin after topical treatment with 12-*O*-tetradecanoylphorbol 13-acetate. The catalytic activity of myeloperoxi-dase enhances the reactivity of oxygen species. This alternative mechanism may be important for human exposure to PAH because simultaneous chronic inflammation (e.g. due to smoking) often leads to increased numbers of inflam-matory cells (Marnett et al., 1978; Kensler et al., 1987; Ji & Marnett, 1992).

Epoxidation of benzo[*a*]pyrene 7,8-diol has also been reported to be mediated by lipoxygenase (Hughes et al., 1989; Agarwal & Muhktar, 1991; see also section 7.9).

7.10.3.1 *Acenaphthene and acenaphthylene*

B6C3F$_1$ mice were given a single intraperitoneal injection of 300 mg/kg bw acenaphthene or acenaphthylene. Acenaphthylene caused a > 80-fold induction of hepatic microsomal methoxyresofurin O-deethylase activity, dependent on the *Cyp1a2* gene, which codes for an enzyme that catalyses the oxidative metabolism of diverse substrates; acenaphthene increased the activity by > 20-fold. The tricyclic PAH acenaphthene, acenaphthylene, anthracene, fluorene, and phenanthrene were not competitive inhibitors at the mouse hepatic cytosolic Ah receptor when tested together with ^{3}H-labelled 1,2,7,8-tetrachlorodibenzo-*para*-dioxin or benzo[*a*]pyrene. The authors suggested an association between the relatively nontoxic behaviour of the tricyclic PAH and the observed *Ah* receptor-independent induction of hepatic *Cyp1a2* expression (Chaloupka et al., 1994).

7.10.3.2 Anthracene

A single intraperitoneal injection of 300 mg/kg bw anthracene to $B6C3F_1$ mice caused a > 10-fold induction of hepatic microsomal methoxyresofurin *O*-deethylase (Chaloupka et al., 1994).

7.10.3.3 Benzo[a]pyrene

The toxic effects of benzo[*a*]pyrene in mice vary according to their genetic constitution (see also section 7.5). The crucial point appears to be the *Ah* locus, which determines the inducibility of AHH. For example, administration of benzo[*a*]pyrene at 120 mg/kg bw per day in the diet induced aplastic anaemia and death in nonresponsive AKR/N mice (Ah^d/Ah^d type) within four weeks, with hypocellular bone marrow, myeloid precursors, and promegakaryocytes; responsive AKR/N mice (Ah^b/Ah^b type), however, were still healthy after six months. In contrast, when benzo[*a*]pyrene was given intraperitoneally at 500 mg/kg bw per day, responsive mice survived for a significantly shorter time than nonresponsive mice (Robinson et al., 1975). In order to explain these differences, Nebert et al. (1977) proposed that the gastrointestinal tract and liver of responsive mice have a greater capacity to detoxify an orally administered dose; however, if benzo[*a*]pyrene reaches their bone marrow and other distal tissues, metabolism there leads to increased formation of toxic metabolites.

Mice with high-affinity Ah receptors showed no myelotoxicity after administration of 120 mg/kg bw per day benzo[*a*]pyrene in the diet for six months, but non-responsive mice at the same dose died within three weeks due to myelotoxic effects (Legraverend et al., 1983).

After two oral administrations of 10 or 100 mg/kg bw benzo[*a*]pyrene, the numbers of sister chromatid exchanges and DNA adducts were significantly higher in AHH-non-inducible DBA/2 mice than in inducible C57Bl/6 mice (Wielgosz et al., 1991).

Because PAH produce tumours at the site of administration, it was suggested that they do not require metabolic activation; however, it was shown later that activation occurs in the target tissue. The arylkylating agent, for example the 7,8-dihydrodiol-9,10-epoxide of benzo[*a*]pyrene, reacts with DNA to yield almost exclusively the deoxyguanosine adduct. Methyl groups reaching into the bay region can enhance carcinogenic potency by steric effects (Dipple et al., 1990).

Topical application of the prostaglandin synthetase inhibitor indomethacin after administration of benzo[*a*]pyrene to mice delayed the onset and reduced the size of the skin tumours. It was assumed that prostaglandin-induced suppression of cellular cutaneous immunity plays a role in carcinogenesis, as indomethacin can partially restore cutaneous immunity (Andrews et al., 1991).

7.10.3.4 *Benz[a]anthracene*

Benz[*a*]anthracene was not tumorigenic after intravenous or intramuscular injection in rats. The methyl substituent was shown to be of great importance in the carcinogenicity of this compound, as derivatives were highly carcinogenic when they possessed two or three methyl groups in any combination at position 6, 7, 8, or 12 (Pataki & Huggins, 1969).

7.10.3.5 *Benzo[c]phenanthrene*

Benzo[*c*]phenanthrene is unique among the PAH in that it has no bay region as such but has a 'fjord' region between positions 1 and 12 (see Figure 12). The synthesized metabolite 3,4-dihydrodiol (but not the 1,2 or 5,6 derivative) was as mutagenic in the presence of liver microsomes as the parent compound (Croisy-Delcey et al., 1979).

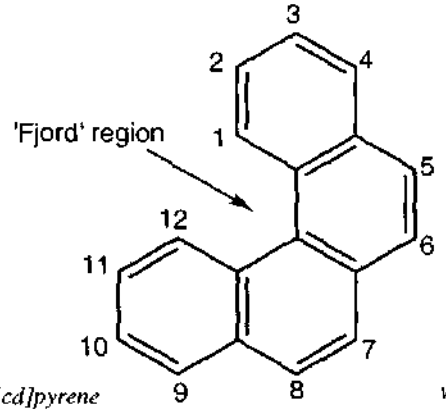
The four fjord diol epoxides of benzo[*c*]phenanthrene are very active tumour initiators. In studies of their reactions with DNA, each diol epoxide became bound covalently to DNA and showed a unique product distribution, with either a preference for reaction with deoxyadenosine residues or a more even distribution between deoxyguanosine and deoxyadenosine residues. A remarkable feature is the efficiency of covalent binding to DNA relative to DNA-catalysed hydrolysis. The authors reported a strong association between reactivity with adenine in DNA and tumour-initiating activity. Interaction with DNA can lead directly to activation of the *ras* protooncogene and to tumour initiation (Dipple et al., 1987).

In cultures of cells from embryos of Sencar mice, Syrian hamsters, and Wistar rats, >74% of all benzo[*c*]phenanthrene–deoxyribonucleoside adducts resulted from the 1*R*,2*S*-epoxy-3*S*,4*R*; deoxyadenosine and deoxyguanosine adducts were formed at a ratio of 3:1. The absolute configuration of the major metabolite and preference for adenosine residues have been found to be typical for other potent carcinogens (Pruess-Schwartz et al., 1987).

7.10.3.6 *Chrysene*

The metabolism of the weak carcinogen chrysene has been investigated in the presence and absence of other xenobiotics. The putative ultimate carcinogenic form of chrysene is the 1,2-dihydroxy-3,4-epoxy-1,2,3,4-tetrahydro metabolite, formed by the inducible cytochrome P450 system. In rats treated with benzo[*a*]pyrene, benzo[*b*]fluoranthene, or benzo[*j*]fluoranthene, 1,2- and 3,4-oxidation were highly induced, and the 1,2,3-triol metabolite was produced, which is a derivative of the 1,2-dihydrodiol-3,4-epoxide. In the absence of induction, chrysene may not be metabolized to the ultimate carcinogen (Jacob et al., 1982b).

Figure 12. 'Fjord' region of benzo[*c*]phenanthrene



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7.10.3.7 Cyclopenta[cd]pyrene

For most PAH, an initial epoxidation step catalysed by cytochrome P450-dependent mono-oxygenases is followed by a second epoxidation; however, cyclopenta[*cd*]pyrene has no bay region (Figure 13) and has been suggested to be activated by a single epoxidation at the cyclopentene double bond, which may be possible in systems that generate peroxyl radicals. Reed et al. (1988) found that peroxyl radicals could activate the mutagenic potential of cyclopenta[*cd*]pyrene. This compound is a member of a subclass of PAH that have a non-aromatic double bond, which may form the centre for conversion to an ultimate mutagen.

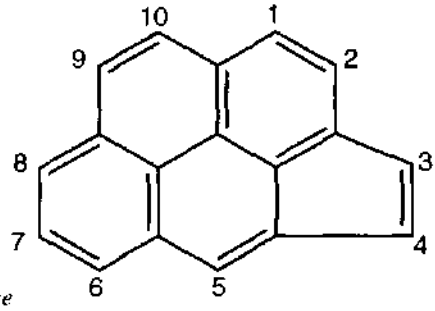
7.10.3.8 Fluorene

A single intraperitoneal injection of 300 mg/kg bw fluorene to B6C3F$_1$ mice caused a greater than fivefold induction of hepatic microsomal methoxyresofurin *O*-deethylase activity (Chaloupka et al., 1994; see also section 7.10.3.1).

7.10.3.9 Indeno[1,2,3-cd]pyrene

The 1,2-diol of indeno[1,2,3-*cd*]pyrene and its epoxide precursor, 1,2-oxide were found to have similar carcinogenic potency in an initiation assay. This result is remarkable, because K-region dihydrodiols such as the 1,2-diol are generally considered to be detoxification products formed by hydrolysis of K-region oxides. Further metabolic activation of the 1,2-diol via epoxidation in the 7-10 area was proposed because 8- and 9-hydroxy indeno[1,2,3-*cd*]pyrene had been detected as metabolites. If substantiated, this would be a unique activation mechanism for PAH (Rice et al., 1990).

Figure 13. Structural formula of cyclopenta[*cd*]pyrene



7.10.3.10 5-Methylchrysene

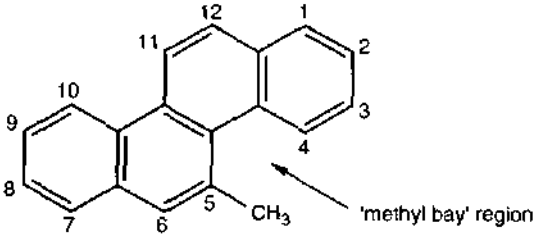
The 5-methyl compound was the most tumorigenic of the methylchrysenes, probably owing to the presence of the methyl group in the same bay region as the epoxide ring (Hecht et al., 1987). Specific dihydrodiol epoxides of 5-methylchrysene are formed from their precursor dihydrodiols after topical application of 5-methylchrysene to mouse epidermis or injection into newborn mice. (±)-*trans*-1,2-Dihydroxy-*anti*-3,4-epoxy-1,2,3,4-tetrahydro-5-methylchrysene was found to be the ultimate carcinogen (Hecht et al., 1985).

5-Methylchrysene 1,2-diol and the 1,2-diol-3,4-epoxide are major proximate and ultimate carcinogens; the corresponding 6-methylchrysene-1,2-diol is also a major metabolite but is much less tumorigenic, perhaps because of the different activity of the corresponding 3,4-epoxides. The formation of epoxide-type adducts from 6-methylchrysene was only 5% of that observed for 5-methylchrysene (Amin et al., 1985b). In an investigation of the stereoselectivity of the metabolic activation of 5- and 6-methylchrysene in mouse skin *in vivo* and in rat and mouse liver *in vitro*, using the resolved enantiomers as reference compounds, the *R,R*-enantiomers predominated (> 90 %), and 5-methylchrysene-1*R*,2*R*-diol was the most tumorigenic compound in an initiation test (Amin et al., 1987).

5-Methylchrysene is uniquely tumorigenic among the monomethylchrysene isomers. Its activity is due mainly to the highly tumorigenic diol epoxide, which has a methyl group and an epoxide ring in the same bay region. Of the isomers, only 5-methylchrysene can form this type of 'methyl bay region diol epoxide' (Figure 14). Substitutions that inhibit its formation lead to a decrease in tumorigenicity (Amin et al., 1990).

The tumorigenicity of racemic *anti*-1,2-diol-3,4-epoxides of chrysene, 5-methyl-, 5-ethyl-, and 5-propylchrysene was determined in newborn mice. Only the 5-methyl compound was highly tumorigenic, demonstrating the importance of molecular shape for tumorigenicity. A methyl group in the same bay region as the epoxide ring leads to exceptional activity, and this may occur a consequence of DNA adduct conformation (Amin et al., 1991b).

Figure 14. Structural formula of 5-methylchrysene



In a study of the binding of ^{3}H-labelled *anti*-5- and *anti*-6-methylchrysene-1,2-diol-3,4-epoxide to DNA in liver and lung of newborn mice after intraperitoneal administration on day 1 of life and sacrifice after 24 h, 1.1 pmol/mg DNA were found with the benzo[*a*]pyrene analogue, 0.5 pmol/mg DNA with the 5-methyl compound, and < 0.01 pmol/mg DNA with the 6-methyl compound, consistent with their known tumorigenic activities. When the parent compounds were tested in the same protocol, however, little radiolabel became associated with DNA adducts. Hence, it can be concluded that the dihydrodiols are the carcinogens in newborn mice (Melikian et al., 1991).

The metabolism of 5-methylchrysene has also been investigated in mouse epidermis *in vivo*. The diol precursors of 1,2-dihydroxy-3,4-epoxy 5-methylchrysene and 7,8-dihydroxy-9,10-epoxy 5-methylchrysene were present in equivalent quantities at every time, and the ratio of DNA adducts with the two precursors was constant over time (Melikian et al., 1983).

In a study of the correlation of the 5-methylchrysene–DNA adduct profile in lung tissue with the spectrum of mutations in the K-*ras* protooncogene of lung tumours, up to 200 mg/kg bw 5-methylchrysene were administered to A/J mice and the lungs were analysed for DNA adducts one to three days later. After a latent period of eight months, 90% all lung tumours had mutations of K-*ras*. *N*2-Deoxyguanosine was detected as a possible promutagenic adduct (You et al., 1994).

7.10.3.11 1-Methylphenanthrene

An investigation of an extensive series of alkylated phenanthrenes suggests that the presence of a methyl substituent at, or adjacent to, the K region (9,10 position) and an unsubstituted angular ring adjacent to a free peri position are the prerequisites for mutagenic activity in *S. typhimurium*. Substitution at the peri position was associated with lack of mutagenicity. A non-K-region dihydrodiol derived from 1-methylphenanthrene was a potent proximate mutagenic metabolite (LaVoie et al., 1983b).

7.10.3.12 Naphthalene

Severe bronchiolar epithelial-cell necrosis was reported in mice after intraperitoneal injection of naphthalene. Lung, liver, and kidney macromolecules were shown by a radiolabelling technique to be the main targets. Maximal binding of naphthalene was found 2–4 h after application, and a threshold was found at 200–400 mg/kg bw, corresponding to glutathione depletion. Covalent binding was highest in tissues with high cytochrome P450 mono-oxygenase activity, i.e. lung, liver, and kidney (Warren et al., 1982).

7.10.3.13 Phenanthrene

A single dose of 300 mg/kg of phenanthrene administered intraperitoneally to $B6C3F_1$ mice caused > 20-fold induction of hepatic microsomal methoxyresofurin *O*-deethylase activity (Chaloupka et al., 1994; see also section 7.10.3.1).

7.10.3.14 Investigations of groups of PAH

In rats, expression of the *Cyp1A1* gene is closely associated with the inducibility of AHH, an enzyme important for bioactivation of PAH. *Cyp1A1* is regulated by several factors, including the Ah receptor and the cytosolic 4S PAH-binding protein. The role of the latter protein was investigated with benzo[*a*]pyrene and benzo[*e*]pyrene in H4-II-E rat hepatoma cells. Both induced gene expression, as measured by ethoxyresorufin O-deethylase activity, but benzo[*a*]pyrene was about 25 times more potent than benzo[*e*]pyrene. Benzo[*a*]pyrene binds to both the Ah receptor and the 4*S* protein, and benzo[*e*]pyrene only to the protein (Houser et al., 1992).

A series of PAH were investigated in a novel short-term test for the detection of carcinogens, the initiator tRNA acceptance assay. Positive responses, i.e. > 15% stimulation, were induced by chrysene, benzo[*c*]-phenanthrene, dibenz[*a,h*]anthracene, benzo[*a*]pyrene, and dibenzo[*a,i*]pyrene; and negative responses were induced by naphthalene, anthracene, phenanthrene, pyrene, benz[*a*]anthracene, benzo[*e*]pyrene, perylene, and coronene. All of the potent carcinogens were active in this assay (Hradec et al., 1990).

In a study of the non-carcinogenic PAH anthracene, fluorene, and naphthalene and several carcinogenic amine derivatives in rats, naphthalene failed to induce induce the cytochrome P450-dependent mixed-function oxidases, whereas anthracene was a weak and fluorene an effective inducer. A relationship was found between carcinogenicity and the ability to induce hepatic P450 activity. It was assumed that fluorene is not carcinogenic because it cannot form mutagenic intermediates (Ayrton et al., 1990).

In a study of the induction of various PAH of the mono-oxygenase isoenzymes in mouse liver microsomes, the cytochrome P448 and P450 groups were classified by using specific inhibitors in studies of 7-ethoxycoumarin activity. According to the pattern of enzymes induced, the following groups were distinguished: (i) P448 type, including dibenz[*a*,*h*]anthracene and benzo[*k*]fluoranthene; (ii) mixed type, including pyrene, benzo[*j*]fluoranthene, and benzo[*e*]pyrene, in which two inhibitors acted on the enzyme reaction; and (iii) special P448 type consisting of indeno[1,2,3-*cd*]pyrene, in which one inhibitor stimulated the reaction. The PAH investigated were not a homogeneous group of selective P448 inducers. No correlation was found with mutagenic potency (Kemena et al., 1988).

8. EFFECTS ON HUMANS

Appraisal

There is little information on human exposure to single, pure polycyclic aromatic hydrocarbons (PAH). That which is available includes reports of accidental exposure to naphthalene and some data from defined short-term studies of volunteers. All other reports are of exposure to mixtures of PAH, which also contained other potentially carcinogenic chemicals, in occupational and environmental situations. Information on the health effects of these mixtures is confined to their carcinogenic potential, for which there is evidence from a number of epidemiological studies, especially for lung cancer and, in some cases, cancers of the skin and of the urinary bladder. Since single PAH and PAH mixtures are known to be carcinogenic in experimental animals, it is plausible to attribute the enhanced cancer risks seen in humans predominately to the PAH. In addition, the results of epidemiological studies are important in risk assessment. Therefore, in contrast to the preceding sections, the results of studies of PAH mixtures are also presented.

Many workplaces have atmospheres with heavy loads of PAH. The cohorts affected are gas workers, those exposed at coke ovens, wood impregnation workers, people working at waste incinerators, workers in bus garages, workers in nickel and copper refineries and in aluminium smelters, asphalt workers, and chimney sweeps. Evaluation of the immunocompetence of coke-oven workers indicated decreased serum immunoglobulin levels and decreased immune function.

Biomarkers used to assess exposure to PAH include hydroxyphenanthrenes and 1-hydroxypyrene in urine and DNA adducts in peripheral blood lymphocytes.

8.1 Exposure of the general population

8.1.1 Naphthalene

Naphthalene is often used in houses as an insect repellant, mainly against moths, and many incidents of poisoning have been reported. Acute haemolytic anaemia is a typical systemic effect of oral, dermal, or inhalation exposure. The lethal oral doses determined in cases of accidental poisoning are 5–15 g for adults and 2 g within two days for a six-year old child. Repeated exposure to naphthalene fumes or dust has led to corneal ulceration, lenticular opacities, and cataracts (Sandmeyer, 1981). Some case reports are described in more detail below.

8.1.1.1 *Poisoning incidents*

(a) Oral exposure

Between 1949 and 1959, 10 cases of the oral poisoning by naphthalene in children were documented in the United States. The amounts were usually not specified but were in the order of grams. Some of the children developed haemolytic anaemia (Anziulewicz et al., 1959).

The symptoms that developed after naphthalene intake included nausea, vomiting, and convulsions after one to several days, often followed by diarrhoea. Other symptoms were disturbances of consciousness, lethargy, ataxia, and, in severe cases, coma and hemiplegia. Haemolytic anaemia occurred concomitantly, with plasma haemoglobin contents of up to 40%, often followed by haemoglobinuria. Mild to severe jaundice can also occur; in one fatal case of poisoning, patchy liver necrosis was reported. Treatment consists of blood transfusions and additional alkalization of the urine; following this treatment, rapid recovery, without persistent damage, was observed (Konar et al., 1939). In five cases of acute haemolytic anaemia in children of about two years of age who had eaten moth balls consisting of pure naphthalene, there was complete recovery within one to four weeks after transfusion (Zuelzer & Apt, 1949; Mackell et al., 1951).

Tests *in vitro* revealed that it was not naphthalene itself but its metabolites α-naphthoquinone and α-naphthol that cause a decrease in reduced glutathione in erythrocytes. Whole blood from patients contained erythrocytes with defective glutathione metabolism. This defect is observed in about 15% of black males and 2% of black females (Zinkham & Childs, 1958).

(b) Dermal exposure

The effects of skin contact in sensitive individuals range from irritation to severe dermatitis after exposure to quite small amounts of naphthalene, such as wearing clothes that had been treated with moth balls. Workers exposed to naphthalene may develop dermatitis on their hands, arms, legs, and abdomen (Gerarde, 1960). Cases of haemolytic anaemia have been reported in babies who absorbed naphthalene from nappies that had been stored with moth balls (Anziulewicz et al., 1959).

(c) Inhalation

Haemolytic anaemia was also observed in babies who had inhaled naphthalene from moth ball-treated wool blankets (Valaes et al., 1963). The case of a man with exfoliative dermatitis was reported, which resolved after all contact with naphthalene was eliminated (Fanburg, 1940).

8.1.1.2 *Controlled studies*

When the forearm skin of three volunteers was treated with anthracene in a 2% benzene solution (dose not specified) and irradiated with a monochromator (340–380 nm), urticarial reactions were seen, with burning and erythema lasting for several days (Crow et al., 1961).

Twelve healthy white men, 12–26 years old, with fair complexions were treated dermally with anthracene in an ethanolic acetone solution at 25 μg/cm^2 and received ultraviolet irradiation 2 h later. Specific skin reactions such as transient erythema, delayed erythema, and whealing were seen. The effect was related both to the anthracene and the amount of radiation energy. Controls who received no anthracene treatment but the same irradiation showed no sign of erythema (Kaidbey & Nonaka, 1984).

Regressive verrucae were reported after up to 120 dermal applications of 1% benzo[*a*]pyrene to human skin over four months. The reversible and benign changes were thought to be neoplastic proliferations, but a group that did not receive benzene was not evaluated (Cottini & Mazzone, 1939). Similar epidermal changes and nucleolar enlargement were reported in volunteers painted daily for four consecutive days on 1-cm^2 areas of the upper back (Rhoads et al., 1954).

8.1.2 *Mixtures of PAH*

8.1.2.1 *PAH in unvented coal combustion in homes*

Interdisciplinary studies were conducted to investigate exposure to PAH and the high lung cancer rates in a rural county, Xuan Wei, located in Yunnan Province, China (Mumford et al., 1987). Mortality from lung cancer in this county is five times the Chinese national average, especially among the women, who have the highest rate in China. Three communes had a mortality rate that was 24 times the national rate: 126 per 100 000 for women and 118 per 100 000 for men during 1973–79. An unusual observation in Xuan Wei is the similarity of the lung cancer rates in men, most of whom are smokers, and women, most of whom are not (< 0.1% smoke). The mortality rate from lung cancer was correlated with domestic use of 'smoky' coal (medium-volatile bituminous coal with low sulfur and high ash) for cooking and heating, but not with use of wood or smokeless coal. Monitoring of air during cooking inside the homes showed that women were exposed to extremely high levels of PAH, with a mean benzo[*a*]pyrene concentration of 14.7 μg/m^3, comparable to the levels to which coke-oven workers are exposed. They were also exposed ≤ 24 mg/m^3 of submicron particles containing up to 82% of the organic matter. The major organic components of smoky coal emissions are the three- to five-ring alkylated PAH, which contributed 43% of the organic mass of the particles

and 61% of the total mutagenicity in assays in *S. typhimurium*. The four-ring PAH were the most tumorigenic (Chuang et al., 1992). Organic extracts of particles from coal smoke were more potent in initiating tumours in Sencar mouse skin than those from wood and smokeless coal combustion and were complete carcinogens (Mumford et al., 1990). Xuan Wei residents exposed to smoky coal emissions had significantly more 9-hydroxy benzo[*a*]pyrene in their urine, and cells obtained by bronchial alveolar lavage had more DNA adducts (detected by ^{32}P-postlabelling) than those of controls (Lewtas et al., 1993). High ratios of the concentrations of methylated PAH and parent PAH (9.8:1 for women and 5.8:1 for men) in urine samples from Xuan Wei residents confirmed that they were exposed to high concentrations of alkylated PAH. Thus, alkylated PAH may play an important role in the etiology of lung cancer in Xuan Wei (Mumford et al., 1995).

1-Hydroxypyrene was used as a urinary biomarker to monitor the exposure of urban populations to PAH originating from coal burning. A good correlation was found between the concentrations of pyrene and benzo[*a*]pyrene in ambient in air and 1-hydroxypyrene in urine (Zhao et al., 1990; see also section 8.3.2).

8.1.2.2 *PAH in cigarette smoke*

A large volume of literature exists on the effects of tobacco smoke on human lungs (see IARC, 1986). On the basis of more than 100 prospective and retrospective studies in more than 15 countries, cigarette smoke has been shown to be by far the most important single factor contributing to the development of lung cancer. Other types of cancer caused by cigarette smoking include cancers of the oral cavities, larynx, pharynx, oesophagus, bladder, renal pelvis, renal adenocarcinoma, and pancreas.

Levels of 11 ng per cigarette benzo[*a*]pyrene were found in mainstream smoke and 103 ng per cigarette in sidestream smoke; the corresponding values were 6.8 and 7.6 ng per cigarette for benzo[*e*]pyrene, 20 and 497 ng per cigarette for chrysene and triphenylene, and 13 and 204 ng per cigarette for benz[*a*]anthracene (Grimmer et al., 1987). In sidestream smoke, PAH with four or more rings were responsible for 83% of the total carcinogenic activity (Grimmer et al., 1988c).

8.1.2.3 *PAH in coal-tar shampoo*

Of eight commercially available coal-tar shampoos, that with the highest PAH content (100 times that of the others), containing, e.g. 285 mg/kg pyrene and 56 mg/kg benzo[*a*]pyrene, was chosen for testing in 11 healthy people. A dose of 20 g shampoo was used in the evening (see section 8.2.3), and the internal dose of PAH was assessed as urinary 1-hydroxypyrene. One day after exposure, the internal dose was 10 times higher than the background level,

similar to that measured in coke-oven workers (van Schooten et al., 1994). The potential carcinogenicity of coal-tar shampoo formulations has been studied (see section 7.7). It has been suggested that modest therapeutic doses of agents containing coal-tar and dithranol are tumorigenic after combined application and that their use should be reviewed (Phillips & Alldrick, 1994).

8.2 Occupational exposure

No studies on occupational exposure to single PAH were available, as in general, industrial workers using or producing coal or coal products are exposed to mixtures of PAH (see section 5.3). Table 95 lists workplaces in which there is exposure to PAH and the types of employees exposed. Epidemiological studies have been conducted on workers exposed at coke ovens in coal coking and coal gasification, at asphalt works, at foundries, and at aluminium smelters and to diesel exhaust. Details of the most recent and most important cohort and case–control studies are given in Tables 96 and 97 (for reviews of these studies, see also IARC 1983, 1984a,b, 1985).

Levels of exposure to single PAH in these occupations have been reported (see section 5.3). In the following text, levels of exposure to 'total PAH' are also given, as reported by the authors; however, 'total PAH' represents only the sum of a limited number of compounds that have been quantified, i.e. the selection of the investigator, and such measurements cannot be compared for the purpose of evaluating levels or risks of pollution.

All benzo[*a*]pyrene concentrations are reported for comparative purposes, as it is the only PAH that has been determined in almost all investigations because of its well-known carcinogenicity. It is commonly used as an indicator of the level of particle-bound PAH, and particularly of carcinogenic ones, but PAH profiles may vary according to source.

The first attribution of a PAH-related cancer to an occupational exposure was that of Pott in 1775, who described the susceptibility of English chimney sweeps to scrotal cancer (Pott, 1775); a second was published by Butlin in 1892. Easily avoidable dermal cancers are seldom seen today, owing to better personal hygiene and better working conditions, but the number of respiratory cancers is still significantly higher in occupational cohorts than in the general population. In a study of Swedish chimney sweeps who were exposed to ≤ 9 µg/m^3 benzo[*a*]pyrene, significantly increased rates of lung tumours were observed, with a standardized mortality rate (SMR) of 2.06 (Table 96; Gustavsson et al., 1988); however, chimney sweeps were also exposed to arsenic, chromium, cadmium, nickel, sulfur dioxide, carbon monoxide, organic solvents, and asbestos.

Most important for an evaluation of the possible risk for cancer due to exposure to PAH are studies of workers exposed at coke ovens in coke plants or in coal-gasification processes, where the PAH concentrations are considerable,

Table 95. Occupations in which there is exposure to polycyclic aromatic hydrocarbons

High exposure
— coke ovens
— coal gasification plants
— chimney sweeping
— petroleum refineries (mainly exposed to naphthalene and its methyl derivatives)
— impregnation of wood with creosotes (mainly exposed to naphthalene, phenanthrene, and fluorene)
— handling of creosote-impregnated wood (e.g. railroad and utility workers, carpenters, mainly exposed to naphthalene, phenanthrene, and fluorene)
Medium exposure
— asphalt and pavement work
— roofing
— aluminium production
— graphite electrode production (e.g. anode production for the aluminium industry)
— founding (processing of e.g. steel and other alloys, from coal additives in moulding sand)
— smokehouses (processing of meat and fish)
Low exposure
— mechanics, bus garage workers, and machinists (from diesel and spark-ignition engine exhaust gases)
— mining (from diesel engine exhaust gases)
— use of lubricating and cutting oils (e.g. in steel production)
— cooking

with levels of 1 mg/m^3 total PAH and 300 µg/m^3 benzo[*a*]pyrene (Lindstedt & Sollenberg, 1982; Swaen et al., 1991; see also section 5.3). The concentrations to which workers are exposed are not available in most epidemiological studies, however, and coke-oven workers may be exposed to several other carcinogenic compounds, such as 2-naphthylamine, arsenic, and benzene.

A significantly increased risk for lung cancer (SMR, 1.95) was found among a cohort of over 5000 workers who were heavily exposed at coke ovens in coke plants and were followed-up for over 30 years. Although no data were available on smoking habits, the observed effect is not likely to be due to smoking since unexposed steel workers in a comparison group were assumed

Table 96. Epidemiological studies of lung cancer in cohorts exposed to polycyclic aromatic hydrocarbons

Group, no., workplace, study period	Comparison group, no. workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths No.	Deaths SMR, RR (95% CI)	Dose–response, remarks	Other tumour sites and diseases[a] Type	Other tumour sites and diseases[a] SMR, RR	Reference
Coal coking								
5321, coke oven 1952–82,	10 497 non-oven, steel industry	Coal-tar pitch volatiles:[b] 3.2 mg/m³ (topside), 2.0 (top-side parttime), 0.88 (side); no information on smoking habits concentration	255	1.95 (1.59–2.33)	Risk decreased with period follow-up; findings consistent across racial categories; strong correlation with duration of exposure and exposure	All causes, all cancers, prostate	1.08 1.34 1.57	Costantino et al. (1995); Rockette & Redmond (1985); Redmond (1983)
5639 coke plant, 1945–84	National population (Netherlands)	No information on smoking habits	62	1.29 (0.99–1.66)	Strong correlation with exposure concentration; risks increased in comparison with workers at a nitrogen fixation plant	Total mortality, respiratory disease, liver	1.19 1.66 3.08	Swaen et al. (1991)
536, coke plant 1963–87	National population (France)	Information on smoking habits	25	2.38***	Coke-oven workers: 1.75 (2 cases); near oven workers: 2.52** (8 cases); unexposed workers: 2.28* (6 cases) Risk increased (not significant) for workers in oldest plant	All causes, all cancers, cardiovascular disease	1.41 1.33 1.33	Chau et al. (1993)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
6767, coke plant	National population (Scotland, England, Wales)	Some information on smoking habits	167	1.17*	–	–	–	Hurley et al. (1983)
subcohorts:								
1617, coke oven, 1966–78	–	–	34	0.94	No clear correlation with duration of exposure; problems in classification of exposure (risk increase with > 5 years and > 10 years exposure)	–	–	
1158, coke oven, 1967–80	–	–	32	1.05	Lung cancer risk increased in younger workers; no clear correlation with duration of exposure and exposure concentration; problems in classification of exposure	–	–	
Coal gasification								
2449 + 1176 (2 cohorts), 1953–65	National population (England and Wales)	2-Naphthylamine, 2 µg/m³; some information on smoking habits	99 23	3.82[b] 2.72	RR given for groups with regular exposure; also increased risk in group with intermittent exposure; correlation with exposure concentration and duration of exposure	Bladder cancer	2.35	Doll et al. (1972)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Coal gasification (contd)								
724, 1953–80	(a) 3792, same plant, not at coke-oven (b) 681, office and administration (a) local population (Hamburg)	Median of 8 measurements: total dust, max. 264 mg/m³, BaP, 28 µg/m³, max. 89; some information on smoking habits	68	3.53**[b]	No correlation with duration of exposure; 88% of workers with > 10 years' exposure	All cancers, urinary system cancers	1.98 4.35	Manz et al. (1983)
295, 1966–86	Local worker population	BaP top of ovens: 1964: 4.3 µg/m³ (0.007–33) 1965: 0.52 µg/m³ (0.021–1.29); no differences in smoking habits between exposed and controls	4	0.82 (0.22–2.11)	Owing to incomplete employment registers, only workers with short and recent exposure selected; no correlation with duration of exposure	All causes	1.27	Gustavsson & Reuterwall (1990); Gustavsson (1989); Gustavsson et al. (1987)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Asphalting								
679, paving, 1959–86	National population (Denmark)	Fume condensate; flooring: 0.5–260 mg/m³; median,19.7 , manual road paving: 4.3–3.4 mg/m³; total PAH: median, 183 µg/m³; BaP, 4 µg/m³ ; some information on smoking habits	25	2.90 (1.88–4.29)	SMR for lung cancer given only for workers aged 40–89 years; overall mortality excess primarily in younger age groups	All causes, all cancers Workers aged 40–89 years: liver cirrhosis	1.63 2.25 4.67	Hansen (1989); Hansen et al., 1991c, 1992; Wong et al. (1992)
2572, paving, 1971–79 to 1985	National population (Sweden)	–	7 8	1.10 1.24 (0.53–2.34)	Follow-up too short	All causes, stomach cancer	0.69 2.01	Engholm et al. (1991); Partanen & Boffetta (1994)
704, roofing, 1971–79 to 1985	National population (Sweden)	–	3 4	2.79 (0.99–9.31)	Follow-up too short	All causes	0.91	Engholm et al. (1991); Partanen & Boffetta (1994)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Asphalting (contd)								
Paving, roofing, others	–	–	332	1.21 (1.08–1.30)	Meta-analysis of 11 studies	Stomach, bladder, skin, non-melanoma, leukaemia	1.33 1.38 1.74 1.41	Partanen & Boffetta (1994)
Paving	–	–	167	0.87 (0.74–1.01)	Meta-analysis of 3 studies	Skin, non-melanoma	2.18	
Roofing	–	–	118	1.96 (1.46–2.11)	Meta-analysis of 4 studies	Stomach	1.71	
Creosote impregnation								
922 populations	National (Sweden and Norway)	–	13	0.79 (0.42–1.35)	–	Skin, non-melanoma	2.37	Karlehagen et al. (1992)
Tar distillation								
76, 1962–72	449, rubber industry; national population (Germany)	PAH: 29 µg/m³, BaP: 4 µg/m³; some information on smoking habits	4		Total rate of tumours increased (RR 3 and 3.4); 3 tumours of the digestive system latency, 10.1–17.5 years	–	–	Schunk (1979)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Foundries								
2990 (2651white males; 339 black males), steel, iron, non-ferrous, 1961–71	General population (USA)	No information on smoking habits	224 white males, 39 black males	1.44** white males; 76** black males		Respiratory disease: white males black males Pneumoconiosis: black males Respiratory tuberculosis: white males All cancers: white males black males	1.10 1.24 5.76 2.32 1.10 1.24	Egan-Baum et al. (1981)
439, steel foundry, 1967–77	Regional population (Toronto)	Data from 1977; total dust, 1.76–5.11 mg/m^3; respirable dust, 0.69–2.65 mg/m^3; no cristobalite or tridymite; coal-tar pitch volatiles, 0.19–0.43 mg/m^3; BaP, 0.049–0.152 µg/m^3;	21	2.50**	Correlation with duration of exposure; no correlation between latency and exposure concentration	–	–	Gibson et al. (1977)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Foundries (contd)								
1718, steel or ferro-chromium production, 31 years	General population (France)	Some information on smoking habits	11	2.04 (1.02–3.64)	No clear correlation with duration of exposure or latency	–	–	Moulin et al. (1990)
4227, steel plant, two cohorts, 1968–84	General population (France)	–	17	0.88 (0.51–1.40)	–	Liver cirrhosis	1.74	Moulin et al. (1993)
210, subcohort: ferroalloy workshop	–	–	2	0.68 (0.08–2.45)	–	–	–	
477, subcohort: steel foundry	–	–	11	2.29 (1.14–4.09)	No clear correlation with duration of exposure or latency; significantly increased for > 30 years since first employment	–	–	

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Foundries (contd)								
10 491, steel foundry, 20 years, 1946–65 to 1985	General population (England and Wales)	–	441	1.47***	SMRs increased for all foundry occupations (fettling shop, pattern, machine, maintenance, inspection); some correlation with latency; no significant correlation with duration of exposure	Stomach cancer	1.37	Sorahan & Cooke (1989)
6494, production of ferrosilicon ferromanganese, 6 plants, 1953–82	General population (Norway)	2 plants, around furnaces:PAH, 3–49 µg/m³; anode paste plant: 2–20 µg/m³; total dust: 10–30 mg/m³; manganese: [3] 0.5–2 mg/m Some information on smoking habits	77	0.99 (0.78–1.24) SIR	Except for 1 plant, furnace workers showed no increase in rate of lung cancer (8 cases); increase in anode paste plant workers (3 cases); no correlation with duration of exposure	–	–	Kjuus et al. (1986)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Foundries (contd)								
5579, iron, steel, 1967–69 non-ferrous foundries, 1972–74 (part to 1980)	(a) National population (Denmark), (b) Economically active population, (c) Skilled and unskilled manual workers	–	74	(b) 1.17	Information on employment available only for 1967–69 and 1972–74; misclassification possible; some correlation with duration of exposure	Malignant respiratory tumours, respiratory disease excluding silicosis, all causes	2.05 1.54 1.11	Sherson & Iversen (1986)
8147, iron foundry, 1950–85	National population (USA) Local population	No information on smoking habits	72	1.23 white males (0.96–1.54) 67 1.32 non-white males (1.02–1.67)	No correlation with duration of exposure; smoking may be responsible for lung cancer	–	–	Andjelkovich et al. (1990)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Aluminium production								
22 010, 15 plants, 1946–77, Söderberg prebake	General population (USA)	No information on smoking habits	272	0.964	Highest SMR > 25 years of exposure in Söderberg proccess: SMR 2.0 based on 5 deaths; some correlation with duration of exposure and latency	All causes, all malignant tumours	85.6 88.6	Rockette & Arena (1983)
6455, 11 plants, Söderberg prebake, 1950–76	General population (France)	Some information on smoking habits	37	1.14 (0.85–1.48)	Electrolysis workers: SMR, 1.36 (4 deaths); no correlation with duration of exposure or latency	–	–	Mur et al. (1987)
4213, 1 plant, Söderberg, 1954–85	General population (British Columbia)	Coal-tar pitch volatiles low: < 0,2 mg/m^3; medium: 0.2–1; high: > 1; information on smoking habits	32	0.93 (0.68–1.25)	For > 20 benzene-soluble material of coal-tar pitch volatiles years: SIR 1.43; correlated with duration of exposure; some correlation with latency; adjustment for smoking: no change in lung cancer risk	Brain, bladder (SIR)	2.17 1.69	Spinelli et al. (1991); Rønneberg & Langmark (1992)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Aluminium production (contd)								
5406, 3 plants, Söderberg, prebake, 1954–85	(a) Regional population (Quebec) (b) Local population	No information on smoking habits	101	1.43*	SMRs for workers exposed to tar; SMR significantly increased compared with local population and never-exposed workers Significant correlation with duration of exposure; correlation with latency; risk decreased in later periods (1974-77)	All cancers, respiratory disease, pneumonia and bronchitis, oesophageal and gastric tumours	1.23 1.65 1.99 1.53	Gibbs (1985); Gibbs & Horowitz (1979)
5485, Söderberg, 1950–51; 1977	–	–	12	1.69	SMR increased in relation to local population	–	–	Gibbs (1985); Gibbs & Horowitz (1979)
694 Soderberg, prebake 1962–91	General population (Norway)	Some information on smoking habits	19	1.16 (0.70–1.81) (SIR)	Workers with at least 3 years of employment; some correlation with duration of exposure and latency	–	–	Rønneberg & Andersen (1995)

Table 96 (contd)

Group, no., workplace, study period	Comparison group, no., workplace	Exposure concentration, exposure to other chemicals, smoking habits	Deaths		Dose–response, remarks	Other tumour sites and diseases[a]		Reference
			No.	SMR, RR (95% CI)		Type	SMR, RR	
Other workplaces								
2219, 11 carbon plants, 1974–83	General white population (USA)	–	29	0.85 (0.57–1.21)	–	All causes, circulatory disease, respiratory disease	0.67 0.60 0.51	Teta et al. (1987)
176, waste incinerator, 1951–85	Regional population	Total dust: 5-100 mg/m^3; some information on smoking habits		1.97 (1.21–2.75)	–	–	–	Gustavsson & Reuterwall (1990); Gustavsson (1989)
Nickel/copper smelter and refinery; special subcohorts exposed to tar	Regional population (Ontario)	Laboratory fume generation experiments showed high concentrations of PAH (asphalt > tar > mastic); no information on smoking habits	50	1.47 (1.02–1.81)	Lung tumours correlated with duration of exposure only if exposed to PAH	–	–	Verma et al. (1992)

OR, odds ratio; RR, relative risk; SMR, standard mortality ratio; SIR, standard incidence ratio; BaP, benzo[*a*]pyrene
SMR or RR for whole study population is given; results for special subcohorts, subdivided by e.g. level or duration of exposure or latency, are given under remarks, unless otherwise stated

* Statistically significant at $p < 0.01$; ** statistically significant at $p < 0.01$; *** statistically significant at $p < 0.001$

[a] Only statistically significant ($p < 0.05$) increased or decreased rates are shown; unless otherwise stated, refers to mortality

[b] Benzene-soluble fraction of total particulate matter

Table 97. Case–control studies of cancer types possibly associated with exposure to polycyclic aromatic hydrocarbons (PAH)

Cases, no., tumour type, study group	Exposure: PAH analysed, exposure concentration	Controls: no., matching	Odds ratio (95% CI)	Remarks	Reference
Lung tumours					
NR	Exhaust from coke plant	Patients with cancer of prostate, breast, or brain	1.41	2 miles from coke oven; some correlation with distance from coke oven, highest at 2 and 3 miles distance; effect may also be due to occupational exposure	Lyon et al. (1981)
Asphalt workers, paving, roofing, others,	–	–	1.12 (0.93–1.34)	Meta-analysis of 5 studies	Partanen & Boffetta (1994)
113 cohort of foundry workers	Iron foundry versus steel or non-ferrous foundry	249 all other deaths in cohort of foundry workers except non-malignant respiratory disease and other cancers	2.36 1.19	OR for workers aged 42–64; of workers aged > 65,only 65% successfully traced	Egan-Baum et al. (1981)
12, cohort of steel workers	Occupation in stainless-steel versus ferro-chromium	58 from cohort of steel foundry workers	4.51		Moulin et al. (1990)

Table 97 (contd)

Cases, no., tumour type, study group	Exposure: PAH analysed, exposure concentration	Controls: no., matching	Odds ratio (95% CI)	Remarks	Reference
Lung tumours (contd)					
(a) 51, (b) 47, cohort of iron foundry	BaP: 0.1–15 µg/m^3; small differences between workplaces	From cohort of iron foundry workers (a) 153, matched by age and intensity of exposure, no cancer cases (b) 47, also same time of entry into foundry (c) 27, cases compared with expected contribution of exposure of foundry workers	–	More cases exposed to high PAH concentrations than controls, not statistically significant; some information on smoking habits	Tola et al. (1979)
74, cohort of aluminium production workers, Söderberg and prebake process	Exposure to coal-tar pitch volatiles; measurements of benzene-soluble material; estimated concentrations for different job categories: 0–3.5 mg/m^3	1138, manual workers at same plant, not matched, similar distribution of birth years	2.00 (1.33–2.75)	Figure given for exposure for 20–41 years at Söderberg pots, no data for exposure to benzene-soluble material in general; strong correlation with exposure concentration, duration of exposure and latency; adjustment for smoking: no difference in risk	Armstrong et al. (1994)

Table 97 (contd)

Cases, no., tumour type, study group	Exposure: PAH analysed, exposure concentration	Controls: no., matching	Odds ratio (95% CI)	Remarks	Reference
Lung tumours (contd)					
45 butchers and slaughter-house workers, 11 years	Combustion products, only low, intermittent exposure to PAH	(a) 99 all butchers and slaughter-house workers dying from malignant disease (b) 100 random sample of all deceased butchers and slaughter-house workers (a) and (b) except tumours related to chemical exposure	0.84 (0.40–1.77)	Smoking habits available	Gustavsson (1989)
Skin tumours					
376 patients with skin cancer	Occupational exposure to PAH	752 general population; 752 patients from hospitals randomly sampled	1.14	No correlation with duration of exposure; no increased risk for other exposures related to PAH, i.e. tar, pitch, soot, coke, bituminous mass; smoking habits not available	Kubasiewicz et al. (1991)

Table 97 (contd)

Cases, no., tumour type, study group	Exposure: PAH analysed, exposure concentration	Controls: no., matching	Odds ratio (95% CI)	Remarks	Reference
Renal-cell carcinoma					
1982–87, hospitals in Montreal, Canada	Occupational exposure to hydrocarbons	64 patients with haematuria, 61 controls and cases not treated with haemodialysis (predisposes for renal cancer)	2.54 (0.96–6.99) 0.29 (0.16–.74)	Exposure to coal, tar, or pitch RR for exposure to coal increased from 10 to 24 years of age; strong correlation with duration of exposure and exposure concentration; no correlation with latency (no case or control with latency < 20 years); smoking habits: no significant difference between cases and controls; history of smoking > 20 cigarettes per day associated with tendency to higher stage disease	Sharpe et al. (1989)
Urinary bladder cancer					
138/(66+ 69), 1970-88 aluminium plant, Söderberg	Occupational exposure to benzene-soluble material or BaP	414, from cohort	2.63 (1.29–5.37), adjusted for smoking	Data consistent with former study; strong correlation with exposure concentration and duration of exposure highest risk: Söderberg potroom workers: 5.15; 1930–54: BaP, max, 51.5 μg/m³; benzene-soluble material, max 10 mg/m³; 1985–89: BaP, max, 3.1 μg/m³; benzene-soluble material, max, 0.36 mg/m³	Tremblay et al. (1995); Theriault et al. (1984)

Table 97 (contd)

Cases, no., tumour type, study group	Exposure: PAH analysed, exposure concentration	Controls: no., matching	Odds ratio (95% CI)	Remarks	Reference
Urinary bladder cancer (contd)					
General population, 1979–82	Contact with diesel or traffic fumes	From population	2.61 (0.70–12.5)	All figures given for 8–28 years employed	Risch et al. (1988)
	Aluminium smelting		1.69 (1.24–2.31)		
	Contact with tars, asphalt		3.11 (1.19–9.68)		

OR, odds ratio; RR, relative risk; SMR, standard mortality ratio; SIR, standard incidence ratio; NR, not reported; BaP, benzo[*a*]pyrene
SMR or RR for whole study population is given; results for special subcohorts, subdivided by e.g. level or duration of exposure, latency, are given under remarks, unless otherwise stated
[a] Only statistically significant increased or decreased rates, unless otherwise stated, mortality
* statistically significant at $p < 0.05$
** statistically significant at $p < 0.01$
*** statistically significant at $p < 0.001$

to have similar smoking habits. In addition, a high correlation was seen between the risk for respiratory cancer and the concentration and duration of exposure. The authors noted however, that the rates of respiratory cancer decreased during the follow-up period, suggesting that implementation of emission controls and occupational exposure limits has been beneficial (Costantino et al., 1995). The increased risk found in this study was also seen in other studies, including some on coal gasification (Doll et al., 1972; Manz et al., 1983; Swaen et al., 1991). Other studies, however, and especially those involving small cohorts, did not show increased rates for lung cancer among coke-oven workers (Hurley et al., 1983; Gustavsson & Reuterwall, 1990; Chau et al., 1993).

Coke-oven workers in China who were exposed to PAH were reported to have decreased titres of immunoglobulins M, G, and A in their serum and decreased immune function (Lei, 1993).

Several epidemiological studies have been performed on the potential risks of handling asphalt (for a review, see Partanen & Boffetta, 1994; see also Table 96). The job descriptions included: roofer, waterproofer, highway maintenance worker, production of hot-lay asphalt, slater, grader, paver, surfacer, and mastic asphalt worker. Their exposure to PAH depended on the type of asphalt involved. When bitumen is used as the binder, the PAH content is relatively low, with about 200 $\mu g/m^3$ total PAH and up to about 5 $\mu g/m^3$ benzo[*a*]pyrene (Hansen, 1989; see also section 5.3); bitumen binders contain primarily asphaltenes, straight and branched aliphatic hydrocarbons, naphthene aromatics, and resins. Most of the PAH are removed by vacuum distillation, but they may be formed during cracking operations or be reintroduced in the flux used in blended or fluxed bitumens. Coal-tars and coal-tar pitches have been used as binders, especially in the past, and may contain substantial amounts of PAH (Partanen & Boffetta, 1994). These workers may also be exposed to several other substances, such as silica, limestone, and asbestos (Chiazze et al., 1991; Partanen & Boffetta, 1994). In the meta-analysis of Partanen & Boffetta (1994), increased risks for lung tumours were seen for both pavers and roofers; tumours of the stomach, bladder, and skin and leukaemia were also observed. The excess risks were more pronounced for roofers than for pavers. It is not clear however, if these effects are due to exposure to PAH, as it could not be determined whether the carcinogenicity was due to exposure to bitumen or to tar fumes.

Workers are exposed dermally to very high concentrations of PAH when impregnating wood with creosote, as shown by measuring biomarkers, especially the excretion of 1-hydroxypyrene in urine (see section 8.3.2). In a cohort study on 922 creosote-exposed workers, a significant increase in the risk for skin cancer (SIR, 2.37) and increased risks for lip cancer and malignant lymphoma were observed. Since the work involves some time outdoors, it cannot be ruled out that exposure to sunlight contributed to the risks for cancers of the skin and lip (Karlehagen et al., 1992).

The PAH concentrations in iron, steel, and other ferroalloy foundries reach levels of 50 μg/m^3 and that of benzo[*a*]pyrene about 10 μg/m^3 (Verma et al., 1992; see also section 5.3). Workers in these plants also have nearly ubiquitous exposure to silica sand. Silicosis and other chronic respiratory abnormalities have been reported for decades to be the major health problems of foundry workers. These workers are also exposed to asbestos, used for heat protection and insulation around furnaces, to benzene, toluene, formaldehyde, iron, and, depending on the kind of metal or alloy, to metals such as lead, chromium, manganese, nickel, copper, cadmium, and zinc. Increased mortality from lung cancer has been observed consistently in many studies of foundry workers (Palmer & Scott, 1981; Andjelkovich et al., 1990). When the results of all the relevant studies are combined, the SMR is 1.43 (Andjelkovich et al., 1990); however, those authors argue that the elevated risk is due to smoking, because the SMRs did not increase with time since first employment in a foundry or with the length of employment in a foundry. Furthermore, the incidence of emphysema, which can also be attributed to smoking, is increased in epidemiological studies of foundry environments.

Information on the possible risks for cancer due to exposure to PAH can also be obtained from studies of workers in aluminium plants. Two types of anode are used: the continuous Söderberg anode and the prebaked anode. Both are manufactured from coal-tar pitch and coke, and coal-tar pitch volatiles evaporate from them during baking. The exposure is heavier in potrooms where the Söderberg process is used, because the anodes are baked continuously. With use of prebaked anodes, exposure to PAH may occur in the carbon area where the anodes are prebaked. During Söderberg electrolysis, workers may be exposed to 0.3–3.5 mg/m^3 of benzene-soluble organic material and 3–35 μg/m^3 benzo[*a*]pyrene; workers in the carbon area are exposed to 0.4–1.2 mg/m^3 benzene-soluble organic material and 0.4–1.2 μg/m^3 benzo[*a*]pyrene. Similar concentrations have been detected in other parts of such plants (Rønneberg & Langmark, 1992). These workers are also exposed to fluorides, carbon dioxide, sulfur dioxide, magnetic fields, hot work environments, and, in some cases, asbestos.

In a large case–control study, an increased risk for lung cancer was found with exposure in a Söderberg potroom, and a significant correlation was seen between the increased risk and the duration and concentration of exposure and latency. Adjustment for smoking did not alter the correlation (Armstrong et al., 1994). Similar increases were detected in cohort studies (Rockette & Arena, 1983; Mur et al., 1987). The studies differed with regard to the magnitude of the risk and the presence of a dose–response relationship (see also reviews by Abramson et al., 1989; Rønneberg & Langmark, 1992). In some studies, however, few cases were available (Rockette & Arena, 1983; Mur et al., 1987; Rønneberg & Andersen, 1995).

Work in potrooms where Söderberg electrolytic cells were used was also associated with an increased risk for urinary bladder cancer (Spinelli et al.,

1991; Rønneberg & Langmark, 1992; Tremblay et al., 1995). This risk may be due to exposure not only to PAH but also to aromatic amines, which have been detected in the potrooms (Tremblay et al., 1995).

Asthma-like symptoms, lung function abnormalities, and chronic bronchitis have also been detected in workers in aluminium plants (reviewed by Abramson et al., 1989; Kongerud et al., 1994), but the quality of the studies in which these effects were shown is variable. These diseases are believed to be due not to PAH but rather to exposure to alumina, cryolite, carbon, fluorides, and sulfur dioxide; however, the causative agents have yet to be defined.

Increased risks for lung cancer were found in several studies of workers exposed to diesel exhausts (WHO, 1996). In comparison with the occupations described above, the concentrations of PAH to which these workers are exposed are usually relatively low. The benzo[*a*]pyrene concentrations in automobile repair shops and garages reach about 70 ng/m^3 (Waller, 1981; Lindstedt & Sollenberg, 1982; Waller et al., 1985), and truck drivers are exposed to less than 10 ng/m^3 (Guillemin et al., 1992).

The finding of an increased risk for lung cancer must always be interpreted in relation to the influence of tobacco smoking. The tobacco smoking habits were seldom known for all of the persons studied, but there are several reasons for concluding that the increased risks for lung cancer are not due solely to tobacco smoking:

The general population is often used as a reference group, but their lung cancer rate is usually lower than that of the working population because fewer members of the general population are tobacco smokers (Gibson et al., 1977; Hansen et al., 1989; Andjelkovich et al., 1990; Moulin et al., 1993). Calculations presented by several authors indicate that differences in tobacco smoking habits can contribute only about 20% of the excess risk for lung cancer (Hurley et al., 1983; Maclaren & Hurley, 1987; Gustavsson et al., 1988; Andjelkovich et al., 1990; Moulin et al., 1993). Greater contributions are improbable, since other illnesses usually linked to tobacco smoking have not been observed to occur more frequently than in controls.

Another reason for excluding tobacco smoking as the major reason for the increased lung tumour rates is that increased risks also have been found in studies in which the controls were workers with other occupations, and not the general population. In this case, it can be assumed that the tobacco smoking habits are similar, and it is probable that the increased risks are due to the exposure conditions and not to tobacco smoking (Costantino et al., 1995). In addition, several studies show a strong correlation between the risk for respiratory cancer and the concentration and duration of exposure and latency. Finally, in studies in which the information on tobacco smoking habits was good and adjustments could be made for the influence of tobacco smoking (Spinelli et al., 1991; Armstrong et al., 1994), tobacco smoking had no effect on the cancer risk. As no information is available about the comparative cancer

risks of non-smokers and smokers, however, the increased risk for lung cancer may always represent a combined risk due to tobacco smoking and the exposure conditions.

Although workers are always exposed to several substances, it seems plausible to attribute the increased lung cancer risks at least partially to PAH. The increased risk is seen for workers in several occupations which have exposure to PAH in common. Although other carcinogenic chemicals were present, they differed with each occupation. Airborne high-molecular-mass PAH, which are considered to be the most carcinogenic, are adsorbed mainly onto particulate matter (see section 4.1.2), and it was often difficult to distinguish the toxicological effects caused by particles from those caused by the PAH themselves.

8.3 Biomarkers of exposure to PAH

Several methods have been developed to assess internal exposure to PAH after exposure in the environment and in workplaces, which can be used to evaluate the adequacy of protective regulations. In most studies, metabolites of PAH were measured in urine, 1-hydroxypyrene being widely used.

The genotoxic effects of PAH have been determined in tests for mutagenicity in urine and faeces, micronucleus formation, chromosomal aberration and sister chromatid exchange in peripheral blood lymphocytes, adducts of benzo[*a*]pyrene with DNA in peripheral lymphocytes, and other tissues and with proteins such as albumin; and antibodies to DNA adducts. In addition, oncogene expression and immunostaining of differentiation antigens on lung cell surfaces have been measured as biological indexes of the risk for lung cancer.

8.3.1 Urinary metabolites in general

The metabolites measured in urine and faeces include urinary thioethers (Burgaz et al., 1992), 1-naphthol (Bieniek, 1994; Hansen et al., 1994, 1995), β-naphthylamine (Hansen et al., 1994), hydroxy phenanthrenes (Martin et al., 1989; Adlkofer et al., 1990; Grimmer et al., 1994; Mannschreck et al., 1996), and 1-hydroxypyrene.

No difference in thioether excretion in urine was observed between controls and coke-oven workers or workers in coke and graphite-electrode-producing plants. It was concluded that the determination of thioethers in urine is of little value, since smoking is a strong confounding factor (Ferreira et al., 1994a,b; Reuterwall et al., 1991).

A good correlation was found with the excretion of hydroxylated phenanthrenes and 1-hydroxypyrene in urine (Mannschreck et al., 1996).

When phenanthrene, pyrene, and benzo[*a*]pyrene metabolites are determined simultaneously, precursors of carcinogens are also measured, thus providing an estimate of individual risk (Grimmer et al., 1993).

8.3.2 *1-Hydroxypyrene*

1-Hydroxypyrene, a metabolite of pyrene, was introduced as a biomarker of exposure to PAH by Jongeneelen et al. (1986) and has since been widely used. Its advantages are that pyrene is present in all PAH mixtures at relatively high concentrations (2–10%), and in certain environments the pyrene content of the total PAH is fairly constant (Zhao et al., 1990; Buchet et al., 1992; Jongeneelen, 1994). In studies at different workplaces, a strong correlation was found between the pyrene concentrations in air and those of benzo[*a*]pyrene, other selected PAH, and total PAH (Jongeneelen et al., 1990; Tolos et al., 1990; Zhao et al., 1990; Van Rooij et al., 1992; Ferreira et al., 1994a,b; Jongeneelen, 1994; Elovaara et al., 1995; Levin et al., 1995; Ovrebø et al., 1995; Quinlan et al., 1995a). Pyrene is metabolized predominantly to 1-hydroxypyrene (Grimmer et al., 1993, 1994; Levin et al., 1995), which can be measured easily. In contrast to other PAH metabolites, which are excreted mainly in faeces, 1-hydroxypyrene is excreted in urine.

8.3.2.1 *Method of determination*

The method of determination consists of enzymatic hydrolysis of conjugated 1-hydroxypyrene in urine samples, followed by solid-phase extraction and high-performance liquid chromatographic separation with fluorescence detection (Jongeneelen et al., 1986). An enzyme-linked immunosorbent assay has also been used (Santella et al., 1994).

In most publications, the concentrations of metabolites in urine are adjusted to that of creatinine in order to compensate for variations in urine flow rates; however, it must be borne in mind that creatinine excretion fluctuates widely because of internal and external factors. Therefore, correction of the concentrations of chemicals in urine in this way does not necessarily improve the correlation with exposure (Levin et al., 1995). For comparisons, a mean urinary creatinine concentration of 13 mmol/litre can be assumed, giving the relationship:

1 µmol 1-hydroxypyrene per mol creatinine
= 1.93 µg/g creatinine
= ~ 3 ng/ml urine.

Since most authors give 1-hydroxypyrene concentrations in µmol/mol creatinine, that unit is used in the following text and tables.

8.3.2.2 *Concentrations*

(a) General population

The concentrations of 1-hydroxypyrene in urine from the general population exposed to PAH are compiled in Table 98. The background concentrations of 1-hydroxypyrene in different countries range from 0.06 to 0.23 µmol/mol creatinine. No difference related to age or sex was seen (Zhao et al., 1992), and ethanol consumption did not influence 1-hydroxypyrene concentrations (Van Rooij et al., 1994a).

For nonsmokers, food accounted for 99% of the total daily pyrene intake (Van Rooij et al., 1994a). Five volunteers who ate low-PAH meals and high-PAH meals showed 100- to 250-fold increases in benzo[*a*]pyrene dose, accompanied by a four- to 12-fold increase in 1-hydroxypyrene excretion in urine (Buckley & Lioy, 1992). A 10- to 80-fold increase was detected in one subject who ate 9 oz (250 g) of charcoal-grilled beef. In this study of 10 subjects, an eightfold interindividual variation in urinary excretion of 1-hydroxypyrene was found after one day. As this variation was not appreciably altered after adjustment of the 1-hydroxypyrene concentration by urinary creatinine concentration or individual body weight, it was assumed to be due to individual differences in the rate of absorption, metabolism, or excretion of pyrene (Kang et al., 1995).

The intake of pyrene from cigarette smoking (12 nmol/day) is about the same as the dietary intake from normal food (9.4 nmol/day) (Van Rooij et al., 1994a). Tobacco smokers who are not otherwise exposed to PAH have about twice the level of 1-hydroxypyrene in their urine as nonsmokers (Jongeneelen et al., 1990; Sherson et al., 1992; Van Rooij et al., 1994a; Levin et al., 1995), although no significant difference was found in some studies (Jongeneelen et al., 1988b; Zhao et al., 1992; Ny et al., 1993).

Urine samples from schoolchildren living along arterial roads in Tokyo had 1.1–1.6 times more urinary 1-hydroxypyrene than those from children in a less polluted suburban area (Kanoh et al., 1993). Much higher levels were detected in persons living in highly industrialized regions in Poland, due to emissions from coke-oven plants. The concentrations are a maximum of twice as high in winter as in summer (Jongeneelen, 1994; Ovrebø et al., 1995). 1-Hydroxypyrene levels of up to 1.5 µmol/mol creatinine have been detected in towns in China (Zhao et al., 1990, 1992).

Very high exposure to PAH occurs during application of coal-tar ointments or shampoos by patients with eczema or psoriasis. The mean 1-hydroxypyrene concentrations reached 500 µmol/mol creatinine, and the maximum value was 5000 µmol/mol creatinine (Santella et al., 1994).

Table 98. Concentrations of 1-hydroxypyrene in urine (μmol/mol creatinine) in the general population

Type of exposure or population investigated[a]	No. of subjects	1-Hydroxypyrene		Reference
		Median or mean[b]	Range	
Non-smokers				
General population, southern Germany	28	0.06[c]	< 0.02–0.17	Göen et al. (1995)
General population Southern Germany	49	< 0.02[c]	0.02–0.28	Göen et al. (1995)
Students and university personnel, Netherlands	24	0.23[d]	–	Jongeneelen et al. (1986)
University staff, Netherlands	52	0.26	–	Jongeneelen et al. (1988b)
University staff, Netherlands	39	0.12[e]	0.04–029	Van Rooij et al. (1994a)
Students and university staff, Turkey	15	0.24[e]	–	Burgaz et al. (1992)
Smokers				
General population, Germany	21	0.12[c]	< 0.02–0.68	Göen et al. (1995)
General population, Germany	20	0.14[c]	< 0.02–0.3	Göen et al. (1995)
Students and university staff, Netherlands	22	0.27[d]	–	Jongeneelen et al. (1986)
Students and university staff, Netherlands	38	0.28	–	Jongeneelen et al. (1988b)
Students and university staff, Netherlands	37	0.25[e]	0.10–0.79	Van Rooij et al. (1994a)
Students and university staff, Turkey	14	0.33[e]	–	Burgaz et al. (1992)
Smoking status not specified				
Persons in urban and rural areas Estonia	27	0.20[d]	–	Elovaara et al. (1995)
Persons from rural areas, Biala Podlaska, Poland	81	–	0.19–0.27	Ovrebrø et al. (1995)
Students and university staff, Montreal, Canada	18 N 3 S	0.077[d]	0.002–0.57	Viau et al. (1993)

Table 98 (contd)

Type of exposure or population investigated[a]	No. of subjects	1-Hydroxypyrene		Reference
		Median or mean[b]	Range	
Smoking status not specified (contd)				
Healthy volunteers, Columbia, MO,USA	36 N 17 S	0.14[e]	0.02-0.98	Santella et al. (1994)
Polluted ambient air				
School children, Tokyo, Japan, M+F, N[f]	NR	–	~ 0.4–0.6	Kanoh et al. (1993)
Urban population Beijing, China	74 N 84 S	0.68[e] 0.76[e]	–	Zhao et al. (1992)
Urban population, Shenyang, China, F	13 N	1.55[e]	–	Zhao et al. (1990)
Urban population, Taiyuan, China, F	17 N	1.20[e]	–	Zhao et al. (1990)
Urban population, Beijing, China, F	15 N	0.67[e]	–	Zhao et al. (1990)
Urban population, Beijing, China, F (girls)	15 N	0.72[e]	–	Zhao et al. (1990)
Urban population, Bytom, Silesia, Poland (boys)	72	0.66[e]	–	Jongeneelen (1994)
Urban population, Bytom, Silesia, Poland (girls)	76	0.59[e]	–	Jongeneelen (1994)
Gliwice, Silesia, Poland[g]	30	–	0.84–1.54	Ovrebrø et al. (1995)
Other exposures				
Windsurfers sailing on polluted water, Ketelmeer, Netherlands	6	0.32[e]	0.16–0.81	Jongeneelen (1994)
Therapeutic treatment with coal-tar				
Patients with eczema	5	–	~50–500	Jongeneelen et al. (1986)
Patients with psoriasis	4	–	13.2–811[c]	Clonfero et al. (1989)
Patients with psoriasis	53	547[e]	10–5160	Santella et al. (1994)
Patient with psoriasis	1	3.45	–	Viau et al. (1995)

Table 98 (contd)

Type of exposure or population investigated[a]	No. of subjects	1-Hydroxypyrene		Reference
		Median or mean[b]	Range	
Food				
Eating 9 oz [250 g] char-broiled beef	10	~ 0.5[e,n]	~ 0.15-1.2	Kang et al. (1995)

N, non-smokers; S, smokers; M, males; F, females; NR, not reported
[a] Unless otherwise stated, male persons were investigated; in some cases, insufficient characterization of exposure given
[b] 1-Hydroxypyrene concentration in urine (μmol/mol creatinine), range, and, if available, median concentration; otherwise, geometric or arithmetic means
[c] Calculated from 1-hydroxypyrene concentrations given in original reference as μg/g creatinine.
[d] Geometric mean
[e] Arithmetic mean
[f] Benzo[*a*]pyrene measured at 0.0006–0.0024 μg/m^3 by stationary sampling
[g] Benzo[*a*]pyrene measured at 0.009–0.041 μg/m^3 by stationary sampling
[n] Calculated from 1-hydroxypyrene concentrations given in original reference as ng/ml urine or pmol/ml urine

(b) Workplaces

1-Hydroxypyrene concentrations have been measured in the urine of persons at various workplaces (Table 99); the urine of concurrent controls was examined in most investigations. Unexposed workers at the same plant, such as administrative workers, had slightly higher 1-hydroxypyrene concentrations than the general population (Zhao et al., 1990; Buchet et al., 1992; Ny et al., 1993; Levin et al., 1995).

The highest 1-hydroxypyrene excretion, up to 90 μmol/mol creatinine, was found in urine from workers impregnating wood with creosote, although the PAH levels in the air were quite low. The high exposure can be attributed to significant dermal uptake, which is several times higher than that by inhalation (see below).

Other workplaces where there is heavy exposure are coke ovens, coal-liquefaction plants, aluminium plants, and plants producing carbon or graphite electrodes. The concentrations of 1-hydroxypyrene in the urine of workers at these sites were 1–10 μmol/mol creatinine.

The manufacture and handling of bitumens did not result in a significant increase in urinary excretion of 1-hydroxypyrene (Jongeneelen et al., 1988a,b;

Table 99. Concentration of 1-hydroxypyrene in urine (μmol/mol creatinine) at industrial workplaces without and with exposure to polycyclic aromatic hydrocarbons

Type of exposure, population investigated[a]	Benzo[*a*]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Controls					
Unexposed workers in petrochemical industry	–	120	0.11	< 0.05–1.08	Boogaard and van Sittert (1994, 1995)
Office workers at graphite electrod- producing plant	–	9N 6S	0.33[d] 0.36[d]	–	Buchet et al. (1992)
Unexposed workers in coke and graphite electrode-producing plants	–	137	0.26[e,f]	0.01–1.04	Ferreira et al. (1994a,b)
Workers in administration of municipal waste incineration	–	13N 8S	N: 0.05[f] S: 0.09[f]	< 0.05–0.12 < 0.05–0.67	Göen et al. (1995)
Workers in water supply plants, cotton manufacture, garbage recycling	–	119	0.008	–	Hansen et al. (1994)
Workers in various environments	–	121	0.012[e]	–	Hansen et al. (1995)

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Controls (contd)					
Unspecified control group	–	52N 38S	0.26 0.28	–	Jongeneelen et al. (1988b)
Workers in shipping yards at hot rolling mill	–	14N 28S	0.17[d] 0.51[d]	–	Jongeneelen et al. (1990)
Industry, Netherlands	–	28	0.51	–	
Coke-oven office workers					Levin et al. (1995)
Before renovation		8 N	0.18[g]	–	
After renovation	–	8 N	0.09[g]	–	
Construction workers	–	34 N	0.4[g]	–	Levin et al. (1995)
Water supply workers	–	26 N 42 S	N: 0 S: 0	0–0.010 0–0.022	Omland et al. (1994)
Unexposed workers	–	10N 10S	N: 0.05[d,f] S: 0.21[d,l]	< 0.03–0.12 0.03–1.2	Schaller et al. (1993)
Workers in water purification plants, Denmark	–	20N 26S	N: 0.16[e] S: 0.26[e]	0.1–0.22 0.18–0.34	Sherson et al. (1992)

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Controls (contd)					
Guards in aluminium plant	-	9	0.31[d]	–	Ny et al. (1993)
Maintanance work in blast furnace	–	48	0.61[e,f]	–	Van Hummelen et al. (1993)
Office workers in steel plant	–	10	0.51[e]	–	Zhao et al. (1990)
Coke ovens					
Work at topside oven	0.8–32 5.9[d]	3 N 3 S	N: 5.7[d,f] S: 6.1[d,f]	–	Buchet et al. (1992)
Work at benchside	–	4N 6S	N: 1.2[d,f] S: 0.75[d,f]	–	
Various occupations near ovens	0.39–13	93 N 68 S	N: 2.7[f] S: 3.5[f]	0.25–16 0.29–29	Cenni et al. (1993)
Subgroup topside	2.18	21	6.64[f]	0.29–29	
Subgroup of larry-car operators	–	12	7.76[f]	3.5–16	

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Coke ovens (contd)					
Workers with various tasks	–	41 N 65 S	0.94[e] 1.53[e]	–	Clonfero et al. (1995)
Workers at two coke-oven plants	–	54	0.78[e,f]	0.01–48	Ferreira et al. (1994a,b)
Work at two top side ovens	–	19 9	3.3[d] 2.7[d]	0.8–7.5 1.3–6.5	Jongeneelen et al. (1990)
Other work: coke side, push side, maintenance	–	25	1.9[d]	0.6–4.1	
Coke oven Before renovation, various occupations	0.9–37 4	10	4.7[g]	0.3–30[g]	Levin et al. (1995)
After renovation	0.2–6.8 0.7	10	1.3[g]	0.3–5.7[g]	
Coke workers at 3 plants	0.72–1.5	66	2.45–13.48	–	Ovrebrø et al. (1995)

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[a]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Coke ovens (contd)					
Oven workers		33 7 N 26 S	0.39[e,f]	–	van Hummelen et al. (1993)
Subgroup at top side	–	7	0.85[e,f]	–	
Personnel at coke side, push side, top side, miscellaneous	–	13	2.64	0.54–11	Van Rooij et al. (1994b)
Workers at top of ovens	–	15 S	4.34[e]	–	Zhao et al. (1990)
Workers at top or side of ovens	–	12 S	2.87[e]	–	
Coal liquefaction					
Engineers (control room and plant operations)	–	5	8.53	–	Quinlan et al. (1995b)
Technicians (plant maintenance)	–	5	3.74	–	

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene ($\mu g/m^3$)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Petrochemical industry					
Various operations	–	–	–	0.25–0.68	Boogaard & van Sittert (1994, 1995)
Workers inspecting furnace, replacing burners in boilers, manufacturing rubber grades	–	–	–	max. 1.56	
Subgroup: production of needle coke from ethylene cracker residue:	< 0.01–< 0.22	1			
Maintenance		3	1.02	0.16–5.51	
Operation	< 0.17	12	1.13	0.22–13.2	
Coal-tar distillation operators, cleaner	–	4	–	3.7–11.8	Jongeneelen et al. (1986)
Creosote impregnation					
Wood impregnation plant	–	19	1.6[d,e]	0.18–10	Viau et al. (1993)
Wood impregnation plant	0.01–0.05 (0.012[d])	6	97[e]	–	Elovaara et al. (1995)
Wood impregnation plant	–	1	20–90	–	Jongeneelen et al. (1988b)

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Aluminium production					
Söderberg type anodes:					
Potroom workers, respiratory protection 50% of time	1.9–36 2.8	9	0.4–3.6[g] 2.1[g]	–	Levin et al. (1995)
Potroom workers	–	–	–	–	Ny et al. (1993)
Electricians, technicians, engineers, laboratory workers, industrial hygiene personnel	–	5	0.69[d]	–	
Foremen, technicians, tappers, crucible cleaner	0.88[d]	4	2.6[d]	–	
Crane operators, all-rounders, electrician	7.9[d]	8	14[d]	–	
Potmen	2.2[d]	9	31[d]	–	
Electrode men	37[d]	4	40	–	

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[a]pyrene ($\mu g/m^3$)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Aluminium production (contd)					
Prebake type anodes:					
Electrode production compartment	–	20		0.17–27	Van Rooij et al. (1992)
Paste plant	1.3	8	3.0[d]	1.6–7.4	
Bake oven	0.3	5	4.4[d]	0.98–13	
Pot refining department	1.2	7	6.0[d]	1.9–12	
Anode bake area	–	28	–	0.55-3.6	Tolos et al. (1990)
Anodes from liquid pitch and coke	–	17	–	2.1–37[f]	Schaller et al. (1993)
Anodes not specified	–	28	4.2[f]	0.05–65	Göen et al. (1995)
Production of electrodes					
Graphite electrodes	–	15	3.2[f]	–	van Hummelen et al. (1993)
Graphite electrodes					Buchet et al. (1992)
End-product conditioning	0.002–0.4	3N	N: 0.55[f]	–	
	0.03[d]	7S	S: 0.55[f]		
Second thermal treatment of electrodes	0.002–0.5	8N	N: 0.57[f]	–	
	0.03[d]	17S	S: 0.79[f]		

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[a]pyrene ($\mu g/m^3$)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Production of electrodes (contd)					
Graphite electrodes (contd)					Buchet et al. (1992)
Progressive heating of raw electrodes	0.002–1.9	2N	N: 2.53[f]	–	
	0.04[d]	4S	S: 3.13[f]		
Maintenance and repair	0.002–7.5	15N	N: 1.21[f]	–	
	0.21[d]	2S	S: 3.76[f]		
Grinding and mixing of raw components	0.57–25	5N	N: 2.98[f]	–	
	5.4[d]	9S	S: 2.83[f]		
Electrode impregnation	0.83–73	3N	N:4.14[f]	–	
	6.2[d]	5S	S: 4.96[f]		
Graphite electrodes	–	93	1.7[e,f]	0.03–20	Ferreira et al. (1994a,b)
Carbon electrodes I	–	6	5.8[f]	3.7–43	Göen et al. (1995)
Carbon electrodes II	–	3	12.7[f]	9.4–15	
Carbon electrodes III	–	14	8.4[f]	1.1–65	
Graphite electrodes					Mannschreck et al. (1996)
Crushing	0.09[e]	2	5.0[e,f]	0.6–9.4	
Baking	1.1–12[e]	30	12[f]	0.9–170	
Graphitization	0.01–0.11[e]	24	0.9[f]	0.1–3.3	
Impregnation	0.44-1.1[e]	9	11[f]	3.2–42	
Conditioning	0.01[e]	2	1.2[e,f]	0.9–1.5	

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Carbon black					
Plants manufacturing carbon black	–	5	–	0.32–0.35	Gardiner et al. (1992)
Newspaper printing ink	–	1N 1S	0.47 0.67	–	Jongeneelen et al. (1988b)
Road paving					
Bitumen and coal-tar binders	–	43	–	0.9–2.8	Jongeneelen et al. (1988a)
Bitumen and coal-tar binders	–	28	–	0.9–3.2	Jongeneelen et al. (1988b)
Bitumen binder	–	18 N 21 S	N: 0.53[e] S: 0.67[e]	–	Burgaz et al. (1992)
Bitumen binder	–	3	0.6[e]	–	Jongeneelen et al. (1988b)
Bitumen binder	< 0.05	57	0.7[g]	–	Levin et al. (1995)
Impregnation of road stones with coal-tar	–	38	4.26[i]	0.62–22	Göen et al. (1995)

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene ($\mu g/m^3$)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Foundries					
Iron foundry, melting, machine moulding, casting, sand preparations; high concentration in casting and moulding	0.02	25 N 45 S	N: 0.022 S: 0.027	0.006–0.075 0.006–0.16	Omland et al. (1994)
Iron foundry	< 0.002 0.005–0.012 > 0.012	14 14 18	2.7[e] 1.8[e] 3.6[e]	0.3–6.3 0.3–4.2 0.5–9.7	Santella et al. (1993)
Iron foundry I	0.00 0–0.39	19 14	0.013 0.017	–	Hansen et al. (1994)
Iron foundry II	0.00 0–0.039 > 0.039	13 24 18	0.031 0.022 0.027	–	
Iron foundry	– 20 S	16 N S: 0.42[e]	N: 0.11[e] 0.025–0.59	0.09–0.13	Sherson et al. (1992)
Steel plant	–	12 S	1.34[e]	–	Zhao et al. (1990)

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[a]pyrene ($\mu g/m^3$)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Diesel exhaust					
Railway tunnel under construction	< 0.000–0.04	5N 8S	N: 0.08[f] S: 0.18[f]	0.04–0.31 0.08–0.38	Cenni et al. (1993)
Gate-keepers of harbour terminal for containers	–	3N 4S	N: 0.47[e] S: 0.67[e]	–	Jongeneelen et al. (1988b)
Waste incinerations					
Municipal waste incineration	–	53	0.08[f]	< 0.05–0.41	Göen et al. (1995)
Industrial waste incineration	–	43	0.06[f]	< 0.05–0.47	
Garbage incineration plant	–	35N 17S	N. 0.12[d,f] S: 0.22[d,f]	< 0.05–0.41 0.07–0.41	Schaller et al. (1993)
Miscellaneous workplaces					
Glass manufacture	–	10	0.85	0.2–3.8	Göen et al. (1995)
Lubricating oils in earthenware factories	–	7N	0.32[f]	0.12–0.77	Cenni et al. (1993)
Clean-up of soil of a dump contaminated with coal-tars derived from pyrolysis of used tyres after major fire	–	29	0.11[d]	0.01–0.75	Viau et al. (1993)

Table 99 (contd)

Type of exposure, population investigated[a]	Benzo[*a*]pyrene (μg/m³)[b]	No. of subjects	1-Hydroxypyrene		Reference
			Median or mean[c]	Range	
Miscellaneous workplaces (contd)					
Chimney sweeping	–	27	0.05–1.4	–	Göen et al. (1995)
		0.36[f]			
Meat smoking	–	13	< 0.05–0.57	–	
		0.21[f]			
Fire fighting	0.03–0.7	S	–	0.65–1.0	
		N		0.51–0.6	

N, non-smokers; S, smokers; M, males; F, females

[a] Unless otherwise stated, male persons were investigated; in some cases, insufficient characterization of exposure given

[b] Benzo[*a*]pyrene, stationary or personal sampling

[c] 1-Hydroxypyrene concentration in urine (μmol/mol creatinine), range, and, if available, median concentrations; otherwise, geometric or arithmetic means. Maximum concentrations are given, in post shift samples, in some studies from the end of the week. Figures are given for the whole study population and on subgroups with high exposure.

[d] Geometric mean

[e] Arithmetic mean

[f] Calculated from 1-hydroxypyrene concentrations given in original publication as μg/g creatinine or μg/litre.

[g] Calculated from 1-hydroxypyrene concentrations, given in the original publication as ng/ml urine

Knecht & Woitowitz, 1990; Burgaz et al., 1992), suggesting that the main source of exposure to PAH during paving is the coal-tar used as a binder and not the bitumen itself (Knecht & Woitowitz, 1990).

In some studies of occupationally exposed individuals, the difference in urinary 1-hydroxypyrene concentration between smokers and nonsmokers was greater than expected, suggesting a more than additive effect of exposure and smoking on the body burden (Jongeneelen et al., 1990; Sherson et al., 1992; Ovrebø et al., 1994; Clonfero et al., 1995; Ovrebø et al., 1995; van Schooten et al., 1995). It was hypothesized that the induced P450 enzymes in smokers result in faster biotransformation and less efficient ciliary clearance of particles in the upper airways (Van Rooij et al., 1994a).

The 1-hydroxypyrene concentrations in urine correlated in most cases with the PAH concentrations in air (Buchet et al., 1992; Levin et al., 1995; Mannschreck et al., 1996). The weak correlation between the levels of pyrene in air and 1-hydroxypyrene concentrations in urine was attributed to extensive dermal uptake of the PAH (Van Rooij et al., 1992, 1993a,b; Ovrebø et al., 1995). The 1-hydroxypyrene concentrations in urine correlated quite well with exposure of the skin, monitored by analysing absorbent pads attached to skin sites during shifts (Van Rooij et al., 1992, 1993a,b).

Significant dermal uptake, representing up to 95% of the total, was concluded from the results of several studies of workers exposed at coke ovens, in coal-liquefaction plants, in the petrochemical industry, in aluminium reduction plants, in a graphite electrode plant, in a needle-coke plant, during road paving, and while impregnating wood with creosote oil (Jongeneelen et al., 1990; Van Rooij et al., 1992, 1993a,b; Boogaard & van Sittert, 1994; Ferreira et al., 1994a,b; Van Rooij et al., 1994b; Boogaard & van Sittert, 1995; Elovaara et al., 1995; Quinlan et al., 1995a,b,c). For example, workers impregnating wood with creosote had an average, estimated dermal uptake that was 15 times higher than the estimated respiratory uptake (Van Rooij et al., 1993b).

Use of dermal protection in the form of impermeable polyvinyl chloride suits led to a substantial decrease in the urinary concentrations of 1-hydroxypyrene (Boogard & van Sittert, 1994, 1995). Frequent changes of work clothes and underclothes reduced 1-hydroxypyrene excretion by 37–55% (Van Rooij et al., 1994b; Quinlan et al., 1995c).

8.3.2.3 *Time course of elimination*

The excretion of 1-hydroxypyrene increased significantly between the beginning and end of a shift and from one day to another during one week. Decreases were observed between two shifts, but the high values did not drop to the preshift level of the day before (Jongeneelen et al., 1988b, 1990; Tolos et al., 1990; Buchet et al., 1992; Van Rooij et al., 1992; van Hummelen et al., 1993; Van Rooij et al., 1993b; Ornland et al., 1994; Van Rooij et al., 1994b;

Elovaara et al., 1995; Quinlan et al., 1995a,b; van Schooten et al., 1995). After an exposure-free weekend, the 1-hydroxypyrene concentrations in the urine of heavily exposed workers did not drop to control levels (Jongeneelen et al., 1988b, 1990; Viau et al., 1993; Elovaara et al., 1995; Quinlan et al., 1995b). The baseline values in exposed workers are slightly higher than those in unexposed controls (Jongeneelen et al., 1988a, 1990; Tolos et al., 1990; Quinlan et al., 1995a,b).

Elimination of 1-hydroxypyrene is biphasic, a moderately rapid phase being followed by a second, much slower elimination (Jongeneelen et al., 1988b, 1990; Viau et al., 1995). The half-lives of the first phase have been determined at various workplaces and for non-occupationally exposed persons after inhalation, dermal, and oral exposure. Regardless of the route of exposure, they range from 4.4 to 48 h, most values being about 16 h (Jongeneelen et al., 1988b, 1990; Buchet et al. 1992; Buckley & Lioy, 1992; Schaller et al., 1993; Boogard & van Sittert, 1994; Quinlan et al., 1995a,b; Viau & Vyskocil, 1995; Viau et al., 1995). In one study, a half-life of 16 days was given for the slower phase (Jongeneelen et al. 1988b). This slow elimination suggests that pyrene accumulates in a secondary compartment, most probably adipose tissue, from which it is released only slowly (Jongeneelen et al., 1990).

8.3.2.4 Suitability as a biomarker

When 1-hydroxypyrene was used as a biomarker for exposure to PAH, the oral, dermal, and inhalation routes were all shown to be important. Furthermore, low levels of exposure to PAH can be determined. A great advantage is that the determination of urinary 1-hydroxypyrene is easy and rapid and thus well suited for use in large-scale epidemiological studies.

Comparison of different work environments may, however, be difficult, because the proportion of pyrene in the total PAH or in comparison with benzo[*a*]pyrene may vary (Jongeneelen et al., 1990; Buchet et al., 1992; van Rooij et al., 1993a; Boogard & van Sittert, 1994; Hansen et al., 1994). For example, the creosote oil used in a wood impregnation plant contained about 3.4% pyrene and less than 0.0004% benzo[*a*]pyrene. Levels of 2–10% pyrene and 0.4–0.6% benzo[*a*]pyrene are found in coal-tar, which is the main PAH contaminant in the coke industry, in the primary aluminium industry, and during road paving with tar. Polluted ambient air contains about 6.5% benzo[*a*]pyrene and 1.8–2.7% pyrene (IARC, 1985; Zhao et al., 1990).

It is not currently possible to assess the risk presented by exposure to PAH on the basis of urinary 1-hydroxypyrene concentrations, as epidemiological studies have not demonstrated a relationship with long-term effects. An indirect dose-response relationship between urinary 1-hydroxypyrene level and the relative risk for lung cancer has, however, been estimated for coke-oven workers: 2.3 µmol 1-hydroxypyrene per mol creatinine was estimated to be

equal to a relative risk for lung cancer of approximately 1.3 (Jongeneelen, 1992). Because of the varying composition of PAH mixtures, this risk estimation cannot be used for other workplaces or ambient air, where a correction factor may be necessary.

8.3.3 *Mutagenicity in urine*

The mutagenicity of urine from persons exposed to PAH has been assayed in a number of studies by Ames' test with *Salmonella typhimurium* TA98 or TA100, with and without metabolic activation. In most of these studies, several urine samples from both control and exposed subjects could not be assayed because of the toxicity of the urine (Heussner et al., 1985; Jongeneelen et al., 1986; Clonfero et al., 1989, 1990; Ferreira et al., 1994b; Santella et al., 1994; Clonfero et al., 1995).

Although tobacco smoke was mutagenic in the presence of metabolic activation, no increase in mutagenic activity was found in most studies of workers exposed in occupations such as coking (Reuterwall et al., 1991; Ferreira et al., 1994a,b), coal-tar distillation (Jongeneelen et al., 1986), work in Söderberg potrooms of aluminium plants (Krøkje et al., 1988), in anode plants (Clonfero et al., 1984; Venier et al., 1985; Krøkje et al., 1988; Clonfero et al., 1990), and in a graphite electrode plant (Ferreira et al., 1994a). Only the heavy exposure of patients with psoriasis to coal-tar applications (Clonfero et al., 1989, 1990; Santella et al., 1994) and of workers at coke ovens (Mielzynska & Snit, 1992; Clonfero et al., 1995) and in a carbon plant (Heussner et al., 1985) resulted in mutagenic urine. Ames' test therefore appears not to be sensitive enough to detect the presence of urinary mutagens due to occupational exposure to low levels of PAH (Becher et al., 1984; Clonfero et al., 1989, 1990).

Expectorate from workers in a coke plant and in Söderberg potrooms in an aluminium plant showed significantly increased mutagenicity in Ames' test with *S. typhimurium* TA98 and TA100 in the presence of metabolic activation (Krøkje et al., 1988; Krøkje, 1989).

8.3.4 *Genotoxicity in lymphocytes*

Genotoxic effects in lymphocytes have been proposed as markers for exposure to PAH. In studies of iron foundry workers with relatively low exposure to PAH, elevated frequencies of mutation at the *hprt* locus in lymphocytes correlated approximately with the levels of DNA adducts (Perera et al., 1993, 1994). In one study of coke-oven workers, significant differences from controls were found in the number of single-strand DNA breaks; however, there was no difference between tobacco smokers and nonsmokers (Salagovic et al., 1995).

No increases in the rates of micronuclei, chromosomal aberrations, or sister chromatid exchange were detected in workers at coke ovens (Reuterwall et al.,

1991), a carbon plant (Heussner et al., 1985), an aluminium plant (Becher et al., 1984), or a graphite electrode plant (van Hummelen et al., 1993) or in chimney sweeps (Carstensen et al., 1993), although in most cases significant effects of smoking could be detected. In one study in which an increase was found, there was no difference between tobacco smokers and nonsmokers (Bender et al., 1988; Salagovic et al., 1995). Environmental pollution in Silesia was associated with significant increases in the frequencies of sister chromatid exchange and chromosomal aberration in peripheral blood cells, independently of smoking (Perera et al., 1992).

8.3.5 *DNA adducts*

DNA adducts with reactive metabolites (mainly diol epoxides) of benzo[*a*]pyrene and other PAH have been identified in numerous studies (see Section 6). For example, cigarette smokers have higher levels of adducts with PAH in their lungs than nonsmokers, and there is a linear relationship between adduct levels and daily or lifetime cigarette consumption (Phillips et al., 1988).

As binding of electrophilic PAH metabolites to DNA is thought to be a key step in the initiation of cancer, measurement of DNA adducts could be an indicator of exposure to PAH and also of cancer risk. As a surrogate for lung tissue, which is an important target organ for PAH in humans, the more easily accessible nucleated blood cells and blood proteins (haemoglobin, albumin) have been investigated.

8.3.5.1 *Method of determination*

The methods for measuring DNA adducts include immunoassays with polyclonal and monoclonal antibodies (enzyme-linked immunosorbent assay [ELISA] and ultrasensitive enzymatic radioimmunoassay), ^{32}P-postlabelling, and synchronous fluorescence spectrophotometry. Direct comparisons of adduct levels determined by different techniques may be misleading, however, because different end-points are measured. For example, polyclonal and monoclonal antisera recognize not only the benzo[*a*]pyrene diol epoxide adducts against which they are raised, but also benz[*a*]anthracene, chrysene, benzo[*k*]fluoranthene, and dibenz[*a,c*]anthracene, which also form N^2 guanine adducts. The ^{32}P-postlabelling assay is even less specific, as it may detect several aromatic and hydrophobic adducts (Dell'Omo & Lauwerys, 1993).

The detection limits for the three methods are one adduct per 10^7–10^8 nucleotides for the immunological methods (Dell'Omo & Lauwerys, 1993) and synchronous fluorescence spectrometry (Dell'Omo & Lauwerys, 1993; Rojas et al., 1995) and up to one adduct per 10^{10} nucleotides in the ^{32}P-postlabelling assay (Beach & Gupta, 1992; Dell'Omo & Lauwerys, 1993; Ovrebø et al., 1994). DNA adducts may be overlooked with ^{32}P-postlabelling, because of incomplete nuclease P_1 digestion, resistance to ^{32}P-labelling, dephosphorylation

of certain adducts, or co-migration with normal nucleotides (Herbert et al., 1990b; Beach & Gupta, 1992; Krick et al., 1993; Segerbäck & Vodicka, 1993; Pavanello & Levis, 1994). This method is being improved (Segerbäck & Vodicka, 1993; Szyfter et al., 1994).

The results obtained when the different methods were applied in parallel were usually similar, but the magnitude of the effect differed (Ovrebø et al., 1990, 1992; Pavanello & Levis, 1994; Perera et al., 1994). For example, in one study of psoriasis patients treated with coal-tar, 20–100 times higher levels were found with ELISA than with the ^{32}P-postlabelling method (Pavanello & Levis, 1994). In another study, the ^{32}P-postlabelling method was more sensitive than the ELISA (Krick et al., 1993). Considerable differences were found in DNA adduct levels in interlaboratory comparisons (Hemminki et al., 1990a; Beach & Gupta, 1992; Krick et al., 1993; Phillips & Castegnaro, 1993).

In the descriptions below, the DNA adduct levels are expressed as number of adducts per 10^8 nucleotides or fmol/μg DNA; 33.2 fmol/μg DNA corresponds to one adduct per 10^8 nucleotides. Since the levels in background samples and also in samples from exposed subjects are sometimes below the limit of detection, the number of positive samples is often given as well (Herbert et al., 1990a,b; Dell'Omo & Lauwerys, 1993).

8.3.5.2 *Concentrations*

In general, exposures that lead to the excretion of high concentrations of 1-hydroxypyrene in urine also lead to elevated DNA adduct levels. Table 100 gives the DNA adduct levels derived from studies in which air concentrations and DNA adduct levels were measured in parallel. Although the concentrations of PAH that occur under different exposure conditions differ by orders of magnitude (see section 5.3), the differences in DNA adduct levels are quite small, in contrast to the results of experiments on excretion of 1-hydroxypyrene. Table 101 summarizes the results of an investigation of workers in an aluminium reduction plant where the two methods were applied (van Schooten et al., 1995).

In all populations studied, there is substantial interindividual variation in PAH–DNA adduct levels, after exposure by inhalation or orally, which is greater than that described for 1-hydroxypyrene excretion in urine (Hemminki et al., 1990a,b; Santella et al., 1993; Szyfter et al., 1994; Rojas et al., 1995). In one study, about 50-fold interindividual variations were reported among controls and about 100-fold variations among coke-oven workers (Rojas et al., 1995). The variations are probably due to differences in the induction of AHH activity in lymphocytes and in the resulting detoxification of carcinogenic PAH, the ability to repair DNA lesions, and the turnover of damaged cells (Dell'Omo & Lauwerys, 1993; Szyfter et al., 1994; Kang et al., 1995; Rojas et al., 1995). These interindividual variations result in a wide overlap in the ranges of values between exposed and unexposed subjects in all studies.

Table 100. DNA–polycyclic aromatic hydrocarbon adduct levels[a] in various situations of exposure

Population investigated, type of emission	Benzo[a]pyrene (μg/m³)	Method of detection	Exposed		Controls		Reference
			No. of subjects	No. of DNA adducts/10⁸ nucleotides	No. of subjects	No. of DNA adducts/10⁸ nucleotides	
Polluted ambient air							
Industrialized area, Silesia, Poland	0.015–0.057	^{32}P-Postlabelling	15	13	13	2.3	Hemminki et al. (1990a)
Industrialized area Silesia, Poland		^{32}P-Postlabelling	19	14	15	4.8	Motykiewicz (1995)
Winter inversion, Teplice, Czech Republic	0.002–0.008	^{32}P-Postlabelling	29	2.6–6.8	–	–	Binková et al. (1995)
Burning of smoky coal at home, with and without chimneys, China	19	Immunoassay	18	7.7	18	5.2	Mumford et al. (1993)
Coke ovens							
Door maintenance	2.3–6.5	Immunoassay	11	5.8[b]	–	–	Assennato (1993a)
Work topside	7.3	Fluorescence spectrophotometry immunoassay	13	ND–73[b]	–	–	Haugen et al. (1986)

Table 100 (contd)

Population investigated, type of emission	Benzo[a]pyrene (μg/m³)	Method of detection	Exposed		Controls		Reference
			No. of subjects	No. of DNA adducts/10⁸ nucleotides	No. of subjects	No. of DNA adducts/10⁸ nucleotides	
Coke ovens (contd)							
Battery work	0.54–90	^{32}P-Postlabelling	31	15	13	2.3	Hemminki et al. (1990a)
Working at high and low traffic density areas	0.001–0.009	^{32}P-Postlabelling	31	2.2	22	2.2	Yang et al. (1996)
Aluminium production							
Aluminum plant, prebake anode process		^{32}P-Postlabelling					Van Schooten et al. (1995)
Anode factory	1.5		–	26	–	–	
Pot relining	1.1			47	–	–	
Electrode past plant	0.9	^{32}P-Postlabelling	34	11	14	10	Ovrebø et al. (1994)
Foundries							
Foundry	0.02	–	–	7.4	–	–	Perera et al. (1994)
Iron foundry	< 0.005– 0.06 ELISA	Competitive	67	4.4-9.6	–	–	Santella et al. (1993)

Table 100 (contd)

Population investigated, type of emission	Benzo[*a*]pyrene (μg/m³)	Method of detection	Exposed		Controls		Reference
			No. of subjects	No. of DNA adducts/10^8 nucleotides	No. of subjects	No. of DNA adducts/10^8 nucleotides	
Foundries (contd)							
Iron foundry	< 0.005– 0.06	^{32}P-Postlabelling	67	1.9-2.5	–	–	Perera et al. (1994)
Iron foundry	< 0.05–> 0.2	Fluorescent ELISA	35	0.8-21	10	2.2	Perera et al. (1988)
Iron foundry	< 0.05	^{32}P-Postlabelling	19	7.3	4	4	Szyfter et al. (1994)
	0.005–0.2	–	63	19	–	–	
	> 0.2	–	6	29	–	–	

ND, not detected; ELISA, enzyme-linked immunosorbent assay

[a] Median or mean values; ranges are from means or medians of several measurements of groups of exposed persons

[b] In original publication given as fmol/μg DNA. Number of adducts per 10^8 nucleotides = fmol/μg DNA x 33.2

Table 101. Comparison of methods for measuring exposure to polycyclic aromatic hydrocarbons in an aluminium plant

Exposure	Benzo[*a*]-pyrene ($\mu g/m^3$)[a]	Pyrene ($\mu g/m^3$)[a]	1-Hydroxypyrene in urine (μmol/mol creatinine)[b] ± SD	DNA adducts in leukocytes (adducts/ 10^8 nucleotides) ± SD
Bake oven	0.35	1.5	3.65 ± 2.11	30.1 ± 42.1
Anode factory	1.51	5.6	3.25 ± 1.89	26.2 ± 15.0
Pot relining	1.05	32.3	6.20 ± 8.44	47.3 ± 39.1
Electrolysis	0.03	0.12	0.48 ± 0.27	12.8 ± 10.0
Foundry	0.02	0.04	0.47 ± 0.20	7.4 ± 9.6

From van Schooten et al. (1995)
[a] Geometric mean
[b] Arithmetic mean

Significant correlations were found in most studies between the estimated or measured exposure to PAH and adduct levels (Herbert et al., 1990a,b; Ovrebø et al., 1990, 1992; Assennato et al., 1993a; Perera et al., 1994; Szyfter et al., 1994; van Schooten et al., 1995), but no such correlation was found in others (Herbert et al., 1990a,b; Kriek et al., 1993; Mumford et al., 1993, Ovrebø et al., 1994; Schoket et al., 1995).

As shown with 1-hydroxypyrene concentrations in urine, the DNA adduct concentrations in certain workers may correlate better with dermal exposure than with PAH concentrations in air (Herbert et al., 1990a,b).

(a) General population

The levels of DNA adduct in control subjects range from 0.2 to about 10 adducts per 10^8 nucleotides in leukocytes (Dell'Omo & Lauwerys, 1993). DNA adducts were also found in 43% of placentas and in 27% of liver samples and 42% of lung specimens from 15 spontaneously aborted human fetuses. As there was only 60% concordance between placenta and fetal lung or liver with regard to the presence of detectable adducts, DNA adducts in the placenta are not a good indicator of adduct formation in fetal organs. Although several of the mothers were smokers, none of the fetal samples containing DNA adducts were from women who smoked during pregnancy, indicating that smoking is unlikely to have caused adduct formation (Hatch et al., 1990).

Conflicting results have been obtained concerning the effects of tobacco (cigarette) smoking on DNA adduct levels in peripheral blood cells. Most investigations of human peripheral lymphocytes have found no remarkable effect of smoking in control or exposed persons, in contrast to those of lung and

bronchial tissue (Hemminki et al., 1990a; Herbert et al., 1990a,b; Dell'Omo & Lauwerys, 1993; Binková et al., 1995; van Schooten et al., 1995; Yang et al., 1996). Some publications, however, report maximal differences in DNA adduct levels of about threefold between tobacco smokers and nonsmokers among controls and exposed individuals (Savela & Hemminki, 1991; Kriek et al., 1993; Santella et al., 1993; Rojas et al., 1995; van Schooten et al., 1995). Granulocytes from smokers and nonsmokers showed no difference in DNA adduct levels, but threefold increases were observed in T lymphocytes, which have a much longer life than granulocytes (Savela & Hemminki, 1991).

A synergistic effect of tobacco smoking and occupational exposure to PAH was reported in two studies (Rojas et al., 1995; van Schooten et al., 1995). It was hypothesized that tobacco smoke induces AHH in lymphocytes, resulting in increased formation of benzo[*a*]pyrene–DNA adducts in smokers. The large interindividual differences may be due to the presence of both non-inducible and highly inducible variants in human lymphocytes (Rojas et al., 1995).

Elevated DNA adduct levels have been detected in the general populations of industrialized areas in Poland (Silesia) and the Czech Republic (Teplice) (Hemminki et al., 1990a,b; Perera et al., 1992; Binková et al., 1995), with levels up to 13 (Hemminki et al., 1990a; Motykiewicz, 1995) and 5 adducts per 10^8 nucleotides (Binková et al., 1995). Levels of 8 adducts per 10^8 nucleotides were found in leukocytes from women exposed to high PAH concentrations from burning smoky coal in China. Although DNA adducts were also detected in placenta, no dose–response relationships were found between exposure to benzo[*a*]pyrene and placental DNA adduct level or the percentage of samples with detectable DNA adducts (Mumford et al., 1993).

The consumption of charcoal-grilled foods leads to elevated DNA adduct levels (Rothman et al., 1993; Kang et al., 1995), and such food may represent a major dietary source of PAH for some populations. Eating charcoal-grilled beef resulted in a 1.9–3.8-fold increase above the individual baseline adduct levels in four of 10 subjects (Kang et al., 1995).

Psoriatic patients undergoing coal-tar treatment had a DNA adduct level of about 8 per 10^8 nucleotides (Pavanello & Levis, 1994; Santella et al., 1995), and levels up to 13 per 10^8 were found in skin biopsy samples obtained from patients who had received treatment with coal-tar ointment. There was no correlation of the adduct levels after different treatments. No information was given on controls (Phillips et al., 1990).

(b) Occupational exposure

Workers exposed to PAH had elevated mean levels of adducts and a higher percentage of positive samples (measured concentrations above the detection limit) than controls. Elevated DNA adduct levels have been detected in leukocytes from workers exposed in coke-oven plants (Assennato et al., 1993a,b; Dell'Omo & Lauwerys, 1993; Harris et al., 1985; Haugen et al., 1986;

Hemminki et al., 1990a,b; Ovrebø et al., 1992; Kriek et al., 1993a,b; Rojas et al., 1995), aluminium manufacture (Dell'Omo & Lauwerys, 1993; Kriek et al., 1993a,b; Schoket et al., 1995; van Schooten et al., 1995), and foundries (Perera et al., 1988; Dell'Omo & Lauwerys, 1993; Santella, 1993; Santella et al., 1993; Perera et al., 1994) and among firefighters (Dell'Omo & Lauwerys, 1993) and roofers (Herbert et al., 1990a,b; Dell'Omo & Lauwerys, 1993).

In cases of high exposure, for example at coke ovens, 5–70 adducts per 10^8 nucleotides have been measured. Significant correlations with exposure concentrations have been found, although the level was no more than threefold greater than in controls (Hemminki et al., 1990a,b; Ovrebø et al., 1990; Assennato et al., 1993a).

8.3.5.3 *Suitability as a biomarker*

DNA adduct levels in the lung may not a reliable indicator of human cancer risk, although Phillips et al. (1988) found a correlation between cigarette smoking and DNA adducts in the human lung. Weston et al. (1993) observed no correlation between lung DNA adduct levels and a measure of recent tobacco smoking, serum cotinine. Tissue samples taken from different portions of the same lung showed variations in DNA adduct levels

The use of lymphocytes as a surrogate for lung cells has also been questioned, because no correlation has been found between PAH–DNA adduct levels in human lung and leukocytes (van Schooten et al., 1992). In studies on rats exposed to coke-oven emissions, the DNA adduct levels were lower in leukocytes than in lung tissues; in addition, several types of adducts observed in lung tissue were not present in leukocytes (Binková et al., 1994). Granulocytes, which form the majority of peripheral leukocytes, have a relatively short life, < 24 h, in contrast to lung cells; therefore, adducts are probably lost within a few days (Savela & Hemminki, 1991; Dell'Omo & Lauwerys, 1993). This can be avoided by using the subfraction of T lymphocytes which have a half-life of several years. The adduct levels in T lymphocytes were three times higher than in granulocytes (Savela & Hemminki, 1991).

DNA adducts are much less sensitive for assessing exposure than excretion of 1-hydroxypyrene in urine. Additionally, because of the large interindividual differences in control and exposed groups, adduct levels can be compared only on a group basis. Thus, PAH–DNA adducts can be used as a qualitative biomarker of exposure to combustion emissions but to only a limited extent as a quantitative marker. This method may, however, allow identification of subjects who are highly susceptible to the DNA-damaging properties of PAH and are therefore predisposed to lung cancer. This was seen in one investigation of lung cancer patients with a family history of lung cancer. Monocytes from these patients treated *in vitro* with PAH showed a slight but significant enhancement of formation of benzo[*a*]pyrene–DNA adducts in comparison

with controls (Nowak et al., 1992). It is not yet known whether metabolism in leukocytes is identical to that in lung cells.

8.3.6 *Antibodies to DNA adducts*

Antibodies to DNA adducts in leukocytes of exposed workers have also been found (Harris et al., 1985; Haugen et al., 1986; Vähäkangas et al., 1992; Santella et al., 1995). In a study of coal-tar-treated patients, elevated levels were also found in controls (Santella et al., 1995). Since the initial antigenic stimulus could have occurred several years previously, antibodies to benzo[*a*]pyrene–DNA adducts are considered general indicators of past exposure to PAH.

8.3.7 *Protein adducts*

Because genotoxic compounds can bind to haemoglobin and serum protein, the assessment of PAH–blood protein adducts has also been considered as a possible marker of exposure to PAH or even as a surrogate for the evaluation of adduct concentrations at the level of the target organs. This approach has several advantages. Relatively large amounts of haemoglobin and albumin can be obtained easily from a small volume of human blood. As the lifetime of haemoglobin in humans is about 120 days and that of albumin 20–24 days, exposure days and weeks previously can be measured. Albumin is synthesized in the liver, where PAH are metabolized. Therefore, reactive metabolites may easily gain access to the proteins. Finally, there may be lower interindividual variation, because there is no repair, as in the case with DNA (Dell'Omo & Lauwerys, 1993).

The benzo[*a*]pyrene–albumin adduct concentrations were similar in foundry workers and controls, both smokers and nonsmokers (Omland et al., 1994), and in patients with psoriasis treated with coal-tar (Santella et al., 1995). Minor effects were detected in foundry workers and roofers (Lee et al., 1991). No pronounced differences in haemoglobin adduct levels were detected in workers in steel foundries and in one graphite electrode producing plant (Ferreira et al., 1994a,b). In another study, however, significantly increased benzo[*a*]pyrene binding was detected in serum proteins from smoking and non-smoking foundry workers (Sherson et al., 1990).

As protein adducts have been used in relatively few studies, no conclusion can be drawn about the usefulness of this biomarker.

8.3.8 *Activity of cytochrome P450*

Increased mRNA levels of the *CYP1A1* gene, which belongs to the P4501A1 family responsible for the metabolism of PAH, including

benzo[*a*]pyrene, have been proposed as biomarkers for exposure to PAH (Cosma et al., 1992; Van den Heuvel et al., 1993; see also Section 6). Although the basal levels were not increased, 3-methylcholanthrene caused greater induction in lymphocytes from railroad workers exposed to creosote in cell culture (Cosma et al., 1992).

The activity of the cytochrome P450 CYP 1A2 was also determined by measuring caffeine metabolites in urine. There were significant differences between tobacco smokers and nonsmokers, but there was no difference between nonsmoking and smoking foundry workers and the respective controls (Sherson et al., 1992).

8.3.9 *Differentiation antigens on the surface of lung cells*

Lung epithelial cells from sputum have been tested for antigens that indicate neoplastic transformation (Assennato et al., 1993b). One of 23 coke-oven workers who had differentiation antigens on the cell surface was a smoker who had severe airways obstruction and moderate dysplasia of bronchial epithelium cells.

8.3.10 *Oncogene proteins*

Since oncogene activation may be an early step in the carcinogenic process, its detection may be a useful marker for identifying individuals at risk for the development of malignancy. Plasma levels of *ras* oncogene-related p21 proteins were elevated in a sample of male residents of the highly industrialized Silesian region of Poland. They also had elevated DNA adduct levels in their leukocytes (Perera et al., 1992). Three of 18 foundry workers screened for the oncogene proteins sis, fes ß-TGF, int-1, myb, src, myc, mos, and ras in serum had elevated levels of ras and fes; however, two were smokers. No unexposed individuals had abnormal serum oncogene protein expression. The levels of ras and fes proteins were also increased in the serum of lung cancer patients (Brandt-Rauf et al., 1990).

9. EFFECTS ON OTHER ORGANISMS IN THE LABORATORY AND THE FIELD

Appraisal

Data on toxicity are available mainly for naphthalene, phenanthrene, and fluoranthene and are scarce for other polycyclic aromatic hydrocarbons (PAH). Both metabolism and photooxidation can alter the toxicity of PAH in the environment, photooxidation tending to increase toxicity.

At low concentrations, PAH can stimulate the growth of microorganisms and algae. The highest values for the no-observed-effect concentration (NOEC) were determined for naphthalene (3100 µg/litre in Anabaena flos-aquae*) and acenaphthene (> 4600 µg/litre in* A. flos-aquae*). The NOEC values for fluorene, phenanthrene, fluoranthene, and pyrene were about one order of magnitude lower. Benz[a]anthracene and chrysene were the most toxic towards algae (NOEC, < 10 µg/litre). The effective concentration that caused a 50% change (EC_{50}) was between 5 µg/litre for benzo[a]pyrene and 54 000 µg/litre for fluoranthene.*

For invertebrates like crustaceans, insects, molluscs, polychaetes, and echinoderms, naphthalene is least toxic, with a 48-h median lethal concentration (LC_{50}) of 700–23 000 µg/litre. For three-ring PAH, the LC_{50} values ranged between < 1 and 3000 µg/litre. Anthracene may be more toxic than the other three-ring PAH, with 24-h LC_{50} values between < 1 and 260 µg/litre. The values for most four- and five-ring PAH are between 0.7 and 1800 µg/litre and 0.4–120 µg/litre, respectively. For six-ring PAH, only one LC_{50} is available: 0.2 µg/litre for benzo[ghi]perylene. The values for the NOEC are slightly lower than those for the LC_{50}. The lowest NOEC reported for benzo[a]pyrene is 0.14 µg/litre.

Naphthalene, acenaphthene, and fluorene are the least toxic for vertebrates like fish and amphibians, with 96-h LC_{50} values of 110 to > 10 000 µg/litre; for phenanthrene and fluoranthene, the LC_{50} values are 30–4000 µg/litre. Most of the 96-h LC_{50} values for anthracene were between 2.8 and 360 µg/litre. For four- and five-ring PAH, the LC_{50} values were 0.7–26 µg/litre. The lowest NOEC reported for benzo[a]pyrene in fish was 2.4 µg/litre. In tests with sediment-dwelling organisms, LC_{50} values of 3–15 mg/kg dry weight were determined for fluoranthene.

In earthworms, an LC_{50} value for growth was 170–200 mg/kg dry weight for fluorene, and an LC_{50} value for reproduction was 150 mg/kg dry weight for phenanthrene. At 180 mg/kg dry weight, chrysene, benzo[k]fluoranthene, and benzo[a]pyrene did not affect reproduction by the earthworm Folsoma candida.

9.1 Laboratory experiments

Data on the toxicity of individual PAH to members of different taxonomic groups are presented in Tables 102–104; data on the toxicity of PAH metabolites are not included. The use of solvents in laboratory tests for toxicity often results in unrealistically high concentrations of PAH, which exceed their maximum water solubility. Only data from tests with concentrations that did not exceed 10 times the estimated solubility were used.

9.1.1 Microorganisms

9.1.1.1 Water

The effects of several three-, four-, and five-ring, unsubstituted PAH on the growth of the bacterium *Escherichia coli* were studied at concentrations of 10^{-5}, 10^{-6}, and 10^{-7} mol/litre in the growth medium (Hass & Applegate, 1975). Benz[*a*]anthracene, dibenz[*a*,*h*]anthracene, and benzo[*a*]pyrene promoted bacterial growth; pyrene and phenanthrene had slight promoting effects at low concentrations (10^{-7} and 10^{-6}, respectively) and inhibitory effects at higher concentrations, whereas anthracene and chrysene inhibited bacterial growth at all concentrations.

The effects of dissolved PAH on the growth rate, lag time before initiation of growth, and the number of cells at the end of the log growth phase, measured as maximum light absorbance, were determined in two species of marine bacteria, *Serratia marinorubra* and *Vibrio parahaemolyticus*. Most of the PAH tested increased the lag time and decreased the growth rate and cell yield; pyrene, however, increased the growth rate. The extent of inhibition of growth was a function of both the concentration of PAH and their inherent toxic properties, which decreased with solubility. Thus, the toxicity of naphthalene at a concentration of 13 mg/litre was similar to that of benzo[*a*]pyrene at 5 µg/litre (Calder & Lader, 1976).

PAH at concentrations of 5–20 µg/litre stimulated the growth of the freshwater algae *Chlorella vulgaris*, *Scenedesmus obliquus*, and *Ankistrodesmus orauntic* (Gräf & Nowak, 1966) and of the marine dinoflagellate *Gyrodinium* sp. (Ishio et al., 1977). The growth of sporelings of the marine algae *Antithamnium plumula* was progressively inhibited after exposure to 10–300 µg/litre of benz[*a*]anthracene.

The growth of the blue–green alga *A. flos-aquae* was inhibited by 16–50% in comparison with controls after exposure to 5-29 µg/litre benz[*a*]anthracene for 14 days. In the same study, 14 days' exposure to fluoranthene at concentrations of 38-1100 µg/litre inhibited growth by 19-65%. The NOEC values ranged from 1 µg/litre for benzo[*a*]pyrene to > 4600 for acenaphthene (Bastian & Toetz, 1982). Other data for cyanobacteria are listed in Table 102.

Table 102. Results of tests for the toxicity of polycyclic aromatic hydrocarbons (PAH) towards algae and plants

Compound, species	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
Aromatic two-ring PAH						
Acenaphthene						
Anabaena flos-aquae	A S Continuous light no solvent 250 x 10^3 cells/ml	2 h	NOEC	> 4600	Nitrogen fixation, acetylene reduction activity	Bastian & Toetz (1985)
Selenastrum capricornutum		96 h	EC_{50}	520	Cell number	US Environmental Protection Agency (1978a)[a]
Fluoranthene						
Anabaena flos-aquae	A S Continuous light, no solvent 250 x 10^3 cells/ml	2 h	NOEC	260	Nitrogen fixation	Bastian & Toetz (1985)
Dunaliella bioculata	N S Continuous light, solvent ,methanol 250 x 10^3 cells/ml	72 h	EC_{50}	15 500	Growth rate	Heldal et al. (1984)
Naphthalene						
Anabaena flos-aquae	A S Continuous light, 250 x 10^3 cells/ml	< 14 d	NOEC	3100	Growth	Bastian & Toetz (1985)

Table 102 (contd)

Compound, species	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
Aromatic two-ring PAH						
Naphthalene (contd)						
Chlamydomonas angulosa		24 h	LC_{61}	34 400[b]		US Environmental Protection Agency (1986d)
Chlorella vulgaris		48 h	LC_{61}	33 000[b]	Cell number	Kauss & Hutchinson (1975)
Champia parvula	N R Solvent ,TEG	11-14 d	MATC	< 695	Female growth Number cystocarps	Thursby et al. (1985)
Nitzchia palea	A S No solvent	4 h	EC_{50}	2820	Assimilation rate	Millemann et al. (1984)
Selenastrum capricornutum	A S Solvent, methanol	4 h	EC_{50}	2960	Assimilation rate	Millemann et al. (1984)
Aromatic three-ring PAH						
Anthracene						
Chlamydomonas angulosa (log phase)	N S 5×10^4 cells/ml	3 h	EC_{50}	239[b]	Photosynthesis inhibition	Hutchinson et al. (1980)
Chlorella vulgaris (log phase)	N S 20×10^4 cells/ml	3 h	EC_{50}	535[b]	Photosynthesis inhibition	Hutchinson et al. (1980)
Selenastrum capricornutum	N R Solvent, acetone[c] (8 h) nitrile UV-A cont.[c] 1×10^5 cell/ml	28 h	EC_{50} EC_{10}	3.9-37 1.5-7.8	Growth rate Growth rate	Gala & Giesy (1992)

Table 102 (contd)

Compound, species	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
Aromatic three-ring PAH (contd)						
Anthracene (contd)						
Selenastrum capricornutum	Cool-white light		EC_{30}	40 000[d]	Growth	US Environmental Protection Agency (1987b)
Selenastrum capricornutum	Gold fluorescent light		NOEC	8000[d]	Growth	US Environmental Protection Agency (1987b)
Fluoranthene						
Anabaena flos-aquae	A S Continuous light, no solvent	2 h	LOEC	230	Nitrogen fixation	Bastian & Toetz (1982)
Scenedesmus subspicatus	N S Solvent, acetone	96 h	EC_{10}	1.6	Growth	Kördel et al. (1981)
		96 h	EC_{50}	12		
Selenastrum capricornutum		96 h	EC_{50}	54 400[d]	Cell number	US Environmental Protection Agency (1978b)
Selenastrum capricornutum		96 h	EC_{50}	54 600[d]	Chlorophyll *a*	US Environmental Protection Agency (1978b)

Table 102 (contd)

Compound, species	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
Aromatic three-ring PAH (contd)						
Fluoranthene (contd)						
Skeletonema costatum		96 h	EC_{50}	45 000[d]	Chlorophyl *a*	US Environmental Protection Agency (1978b)
Skeletonema costatum		96 h	EC_{50}	45 600[d]	Cell number	US Environmental Protection Agency (1978b)
Phenanthrene						
Anabaena flos-aquae	A S Continuous light, no solvent, 250 x 10^3 cells/ml	2 h	NOEC	130	Nitrogen fixation	Bastian & Toetz (1985)
Nitzschia palea	A S No solvent	4 h	EC_{50}	870	Assimilation rate	Millemann et al. (1984)
Selenastrum capricornutum	A S No solvent	4 h	EC_{50}	940	Inhibition of photosynthesis	Millemann et al. (1984)
Aromatic four-ring PAH						
Benz[a]anthracene						
Anabaena flos-aquae	A S Continuous light	< 14 d	NOEC	1	Growth	Bastian & Toetz (1982)
Anabaena flos-aquae	A S Continuous light, no solvent 250 x10^3 cells/ml	2 h	NOEC	19[b]	Acetylene reduction	Bastian & Toetz (1985)

Table 102 (contd)

Compound, species	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
Aromatic four-ring PAH (contd)						
Benz[a]anthracene (contd)						
Anabaena flos-aquae	A S Continuous light	< 14 d	EC 16-48%	18-29[b]	Growth	Bastian & Toetz (1982)
Antithamnium plumula		–	EC 17%	10	Cell production level in algal medium	Boney & Corner (1962)
Chrysene						
Anabaena flos-aquae	A S Continuous light, no solvent, 250 x 10^3 cells/ml	2 h	NOEC	5[b]	Nitrogen fixation	Bastian & Toetz (1985)
Anabaena flos-aquae	A S Continuous light	14 d	EC_{35}	62-96% saturation	Growth	Bastian & Toetz (1982)
Pyrene						
Anabaena flos-aquae	A S Continuous light	< 14 d	NOEC	< 100	Growth	Bastian & Toetz (1982)
Chlamydomonas angulosa (log phase)	N S 5 x 10^4 cells/ml	3 h	EC_{50}	202[b]	Inhibition of photosynthesis	Hutchinson et al. (1980)
Chlorella vulgaris (log phase)	N S 20 x 10^4 cell/ml	3 h	EC_{50}	332[b]	Inhibition of photosynthesis	Hutchinson et al. (1980)

Table 102 (contd)

Compound, species	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
Aromatic five-ring PAH						
Benzo[a]pyrene						
Anabaena flos-aquae	N S 500 x 10^3 cells/ml	72 h	EC_{50}	> 4000[d]	Growth	Schoeny et al. (1988)
Antithamnium plumula		96 h	EC 49%	10[b]	Cell production	Boney & Corner (1962)
		96 h	EC 54%	100[d]	increase	
Chlamydomonas reinhardtii	N S 500 x 10^3 cells/ml	72 h	EC_{50}	> 4000[d]	Growth	Schoeny et al. (1988)
Euglena gracilis	N S 500 x 10^3 cells/ml	72 h	EC_{50}	> 4000[d]	Growth	Schoeny et al. (1988)
Porphyra tenera		80-320 min	EC	1000[d]	Cell size decrease	Boney (1974)
Poteriochromonas malhamensis	N S 500 x 10^3 cells/ml	72 h	EC_{50}	> 4000[d]	Growth	Schoeny et al. (1988)
Scenedesmus acutus	N S 500 x 10^3 cells/ml	72 h	EC_{50}	5[b]	Growth	Schoeny et al. (1988)
Scenedesmus quadracanda	N S Solvent, acetone	20 h	EC	300[d]	Chlorophyll content	Zachleder et al. (1983)
Scenedesmus	N S Solvent, acetone	20 h	EC	300[d]	Biomass	Zachleder et al. (1983)

Table 102 (contd)

Compound, species	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
Aromatic five-ring PAH (contd)						
Perylene						
Scenedesmus subspicatus	N S Solvent acetone 96 h	96 h EC_{50}	EC_{10} > 0.32	0.0066	Growth	Kördel et al. (1981)

A, analysed concentration; N, nominal concentration; S, static system; F, flow-through system; R (0.5 d), system with renewal (each half day)

NOEC, no-observed-effect concentration; EC, effect concentration; LC, lethal concentration; MATC, maximum acceptable toxicant concentration; LOEC, lowest-observed-effect concentration; TEG, triethylene glycol

[a] From Cairns & Nebeker (1982)

[b] Concentration higher than solubility but not exceeding it by 10 times

[c] Explicitly mentioned that organisms were tested for phototoxicity of test substance either by sunlight or artificial UV radiation

[d] Concentration 10 times higher than the solubility

Naphthalene had no detectable effect on the rate of respiration of yeast at concentrations up to its maximum solubility (31 mg/litre) (Haubenstricker et al., 1990).

9.1.1.2 Soil

Few data are available on the toxicity of PAH to microbial communities in the soil. Application of 500 mg/kg dry weight naphthalene did not reduce soil microbial respiration or nitrogen mineralization, and 90% was degraded within 10–20 days (Kirchmann et al., 1991).

9.1.2 Aquatic organisms

9.1.2.1 Plants

For many PAH, the dose that is toxic for algae exceeds the maximum solubility in water (Table 102). Benz[*a*]anthracene (four-ring) and benzo[*a*]pyrene (five-ring) are considered to be the most toxic PAH, with EC_{50} values of 29 µg/litre and 5–15 µg/litre, respectively. The EC_{50} values for three-ring PAH are 240–940 µg/litre. Naphthalene and fluorene (two-ring) are considered to be the least toxic, with EC_{50} values of 2800–15 000 µg/litre. The ranges of EC_{50} values are 200–330 for four-ring compounds and 5–> 4000 µg/litre for five-ring PAH.

9.1.2.2 Invertebrates

Data on the toxicity of two- to six-ring PAH are available for invertebrates such as crustaceans, insects, molluscs, polychaetes, and echinoderms (Table 103). The data include the results of both phototoxicity and non-phototoxicity tests, which are poorly comparable because of the dynamic character of the ultraviolet radiation-induced oxidation products and the often short exposure time.

Phenanthrene, fluorene, and triphenylene did not cause photoinduced toxicity to *Daphnia* and did not absorb light. Benzo[*a*]fluorene, benzo[*k*]-fluorene, and chrysene were considered to be moderately toxic to *Daphnia*, whereas anthracene, fluoranthene, pyrene, benz[*a*]anthracene, benzo[*k*]-fluoranthene, benzo[*a*]pyrene, benzo[*e*]pyrene, perylene, dibenz[*a*]anthracene, and benzo[*ghi*]perylene were very toxic and absorbed more energy. Toxicity was also correlated with the phosphorescence lifetime and energetic state of the molecule. The authors stated that the photodynamic response was due to the PAH assimilated into organisms rather than to external photoproducts. Phototoxicity occurs as a result of energy transfer from activated (triplet) PAH to ground-state oxygen. Singlet oxygen is a reactive chemical that can denature biomolecules within aquatic organisms. The authors estimated that

Table 103. Results of tests for the toxicity of polycyclic aromatic hydrocarbons (PAH) towards invertebrates

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
PROTOZOANS							
Aromatic two-ring PAH							
Acenaphthylene							
Tetrahymena puryformis			24 h	EC_{50} growth	6300[a]	Population	Yoshioka et al. (1986)
Aromatic three-ring PAH							
Anthracene							
Paramecium aurelia	10 000 cells/litre	N S Dark	1.5 h	NOEC	1000[b]	Mortality	Joshi & Misra (1986)
		0.75 h sun[c]	0.75 h	EC 100%	100[a]	Mortality	
Paramecium caudatum			1 h	LC_{90}	1000[b]		US Environmental Protection Agency (1987b)
Aromatic five-ring PAH							
Benzo[a]pyrene							
Gyrodinium sp.	Log phase	N S Solvent ,acetone	12 d	EC 20%	5[a]	Cell division period increase	Ishio et al. (1977)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
INSECTS							
Aromatic two-ring PAH							
Fluoranthene							
Chironomus riparius	Larvae	N S l/d = 16/8	48 h	EC_{50}	2350	Reproduction	Finger et al.
		A IF	30 d	NOEC	142	Emergence	(1985)
Hexagenia bilineata		N S	96 h	LC_{50}	5800		
Naphthalene							
Chironomus attenuatus	4th instar	A F Solvent, ethanol	Chronic	LOEC	500	Physiology	Darville & Wilhm (1984)
				LC_{50}	1300		
Chironomus tentans	Larvae	A S Solvent, methanol	48 h	LC_{50}	2,810		Millemann et al. (1984)
Somatochlora cingulata	Nymph	– S 21 °C	96 h	LC_{50}	1000-2500		Correa & Coler (1983)
Tanytarsus dissimilas	Life cycle	A F Solvent, ethanol	Chronic	LOEC	500	Physiology	Darville & Wilhm (1984)
				LC_{50}	1300		
Anthracene							
Aedes aegypti	3rd-4th instar	N S Solvent DMSO 6 h sun, 18 h dark[c]	1 d	LC_{50}	< 1.0		Borovsky et al. (1987)
Aedes aegypti	< 8 h old	N S 1 h UV[c]	12 h	LC_{50}	150[a]		Kagan et al. (1985)
Aedes taeniorrhynchus	3rd-4th instar	N S Solvent, DMSO 6 h sun, 18 h dark[c]	1 d	LC_{50}	260[a]		Borovsky et al. (1987)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
INSECTS (contd)							
Aromatic three-ring PAH (contd)							
Anthracene *(contd)*							
Culex quinquefasciatus	3rd-4th instar	N S Solvent, DMSO 6 h sun, 18 h dark[c]	1 d	LC_{50}	37		Borovsky et al. (1987)
Culex sp.			24 h	LC_{50}	26.8		US Environmental Protection Agency (1987b)
Fluoranthene							
Aedes aegypti	3rd-4th instar	N S Solvent, DMSO 6 h sun, 18 h dark[c]	24 h	LC_{50}	10		Borovsky et al. (1987)
Aedes aegypti	< 8-h old	N S 1 h sun[c]	12 h	LC_{50}	12		Kagan et al. (1985)
Aedes taeniorrhynchus	3-4 instar	N S Solvent, DMSO 6 h sun, 18 h dark[c]	24 h	LC_{50}	48		Borovsky et al. (1987)
Culex quinquefasciatus	3rd-4th instar	N S Solvent, DMSO 6 h sun, 18 h dark[c]	24 h	LC_{50}	45		Borovsky et al. (1987)
Phenanthrene							
Chironomus tentans	Larvae	A S Solvent, methanol	48 h	LC_{50}	490		Millemann et al. (1984)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
INSECTS (contd)							
Aromatic four- ring PAH							
Pyrene							
Aedes aegypti	1st instar	N S 1 h UV[c]	11 d	EC 18%	0.9	Adult emergence	Kagan & Kagan (1986)
			7 d	NOEC	0.9		
		N S Dark	11 d	NOEC	30		
	< 8-h old	N S 1 h UV[c]	12 h	LC_{50}	20		Kagan et al. (1985)
Aedes aegypti	3rd–4th instar	N S Solvent, DMSO 6 h sun, 8 h dark[c]	24 h	LC_{50}	35		Borovsky et al. (1987)
Aedes aeniorrhynchus	3rd–4th instar	N S Solvent, DMSO 6 h sun, 8 h dark[c]	24 h	LC_{50}	60		Borovsky et al. (1987)
Culex quinquefasciatus	3rd–4th instar	N S Solvent, DMSO 6 h sun, 8 h dark[c]	24 h	LC_{50}	37		Borovsky et al. (1987)
Aromatic five-ring PAH							
Benzo[a]pyrene							
Aedes aegypti	1st instar	N S 30 min UV[c]	11 d	NOEC	0.14	Adult emergence	Kagan & Kagan (1986)
	1st instar	N S Dark	11 d	NOEC	0.9		
	4th instar	N S Dark	7 d	NOEC	6700[b]		
	4th instar	N S 30 min UV[c]	7 d	NOEC	30[a]		
	4th instar	N S 30 min UV[c]	7 d	LC_{50}	120[b]		

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
POLYCHAETES							
Aromatic two- ring PAH							
Fluorene							
Neanthes arenacoendata	Immature adult	A S Solvent, acetone artificial seawater	96 h	LC_{50}	1000		Rossi & Neff (1978)
Naphthalene							
Neanthes arenacoendata	Immature adult	A S Solvent, acetone artificial seawater	96 h	LC_{50}	3800		Rossi & Neff (1978)
Aromatic three-ring PAH							
Fluoranthene							
Neanthes arenacoendata	Immature adult	A S Solvent, acetone artificial seawater	96 h	LC_{50}	500[a]		Rossi & Neff (1978)
1-Methylphenanthrene							
Neanthes arenacoendata	Immature adult	A S Solvent, acetone artificial seawater	96 h	LC_{50}	300[b]		Rossi & Neff (1978)
Phenanthrene							
Neanthes arenacoendata	Immature adult	A S Solvent, acetone artificial seawater	96 h	LC_{50}	600		Rossi & Neff (1978)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
POLYCHAETES (contd)							
Aromatic four-ring PAH							
Benz[a]anthracene							
Nereis virens		A S Sediment	6 d	NOEC	14.4 mg/kg	Oxygen consumption	McElroy (1985)
				NOEC	14.4 mg/kg	Ammonia extraction	
				NOEC	14.4 mg/kg	Microsomal AHH activity	
Chrysene							
Neanthes arenacoendata	Immature adult	A S Solvent, acetone artificial seawater	96 h	NOEC	> 1000[b]	Mortality	Rossi & Neff (1978)
Aromatic five-ring PAH							
Benzo[a]pyrene							
Neanthes arenaceodentata	Adult	A S Solvent, acetone	96 h	NOEC	> 1000[b]	Mortality	Rossi & Neff (1978)
Dibenz[a,h]anthracene							
Neanthes arenacoendata	Immature adult	A S Solvent, acetone artificial seawater	96 h	NOEC	> 1000[b]	Mortality	Rossi & Neff (1978)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
MOLLUSCS							
Aromatic two-ring PAH							
Acenaphthene							
Aplexa hypnorum	Adult	A F Solvent, isopropanol 9	6 h	LC_{50}	> 2040		Holcombe et al. (1983)
Fluorene							
Mudalia potosensis		N S	96 h	LC_{50}	5600[a]		Finger et al. (1985)
Naphthalene							
Physa gyrina	Adult	A S No solvent	48 h	LC_{50}	5020		Millemann et al. (1984)
Aromatic three-ring PAH							
Fluoranthene							
Mytilus edulis	40–50-mm shell	A S Solvent, acetone	9 d	EC_{50}	80	Feeding rate	Donkin et al. (1989)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
MOLLUSCS (contd)							
Aromatic five-ring PAH							
Benzo[a]pyrene							
Mercenaria mercenaria		A F Columns	1-15 weeks	EC	< 0.001	Haemocyte lysozome concentration increase	Anderson et al. (1981)
			8 weeks	EC	< 0.001	Bacterial clearance	
Mytilus californianus			24 h	EC 20%	10 000[b]	Oxygen consumption	Sabourin & Tullis (1981)
				EC 10%	1000[b]	Oxygen consumption	
ECHINODERMS							
Aromatic five-ring PAH							
Benzo[a]pyrene							
Strongylocentrotus purpuratus	Gametes	S N Solvent, ethanol	30-45 min	NOEC	0.5	Cytological abnormality	Hose (1985)
				LOEC	0.5	Mitoses/embryo	
				NOEC	50[b]	Fertilization success	

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
ECHINODERMS (contd)							
Aromatic five-ring PAH (contd)							
Benzo[a]pyrene (contd)							
Psammechinus miliaris	Eggs (fertilized 30 min)	N S Artificial seawater	100 min	NOEC	2000[b]	Development	Bresch et al. (1972)
CRUSTACEANS							
Aromatic two-ring PAH							
Acenaphthene							
Daphnia magna		N S Solvent	48 h	LC_{50}	41 000[b]		LeBlanc (1980)
				NOEC	600	Mortality	LeBlanc (1980)
Fluorene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:1 d[c]	2 d	NOEC	17.0	Mortality	Newsted & Giesy (1987)
Daphnia magna		A IF l/d = 16/8	21 d	NOEC	62.5	Reproduction	Finger et al. (1985)
			21 d	LOEC	125	Reproduction	
		N S 270 mg.l $CaCO_3$ Solvent, acetone	49 h	EC_{50}	430	Immobilization	
Daphnia pulex		N S Solvent, acetone	48 h	EC_{50}	212	Immobilization	Smith et al. (1988)
Gammarus pseudolimnaeus		N S	96 h	LC_{50}	600		Finger et al. (1985)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic two-ring PAH (contd)							
Naphthalene							
Daphnia magna	< 24 h	N S Solvent	48 h	NOEC	600	Mortality	LeBlanc (1980)
Daphnia magna	Adult	N Solvent, ethanol	4 h	LOEC	1000	Behaviour	Whitman & Miller (1982)
Daphnia magna		A S No solvent	48 h	LC_{50}	2160		Millemann et al. (1984)
Daphnia magna		N S	48 h	EC_{50}	4700	Immobilization	Smith et al. (1988)
Daphnia magna	< 24 h	S	48 h	LC_{50}	4100		Crider et al.(1982)
Daphnia magna		N S Solvent	48 h	LC_{50}	8600		LeBlanc (1980)
Daphnia magna	4–6 d	N S No solvent	48 h	LC_{50}	16 000		Bobra et al. (1983)
Daphnia magna		N S Solvent, acetone + triton-X-100	48 h	LC_{50}	22 600		Eastmond et al. (1984)
Daphnia pulex	24-h old	A R Filtered crystals	Chronic	NOEC	330	Increased lifespan & reproduction	Geiger & Buikema (1982)
			Chronic	LOEC	600	Growth	
Daphnia pulex	1.9–2.1 mm	N S No solvent	96 h	LC_{50}	1000		Trucco et al. (1983)
Daphnia pulex		N S	48 h	LC_{50}	3400		Geiger & Buikema (1981)
Daphnia pulex		N S Solvent, acetone	48 h	EC_{50}	4660	Immobility	Smith et al. (1988)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic two-ring PAH (contd)							
Naphthalene (contd)							
Elasmopus sp.	Adult	– S 22 °C (closed bottles)	24 h	LC_{50}	5000		Lee & Nicol (1978)
Gammarus minus		A S	48 h	LC_{50}	3930		Millemann et al. (1984)
Hemigrapsus nudus		A F Seawater	8 d	LC_{50}	2800	Mortality and locomotion	Gharrett & Rice (1987)
		4 h water/8 h air	18 d	EC_{50}	700		
		A F Seawater	8 d	LC_{50}	2200	Mortality and locomotion	
		8 h water/4 h air	18 d	EC_{50}	2000		
		A F Seawater	8 d	LC_{50}	1100	Mortality and locomotion	
		12 h water/0 h air	18 d	EC_{50}	800		
Neomysis americana		A F Artificial seawater, 25 °C	96 h	LC_{50}	850		Smith & Hargreaves (1983)
		A F Artificial seawater, 15°C	96 h	LC_{50}	1280		
Palaemonetes penaeus	–	– – –	24 h	LC_{50}	2500		Anderson et al. (1974)
Pandalus goniurus	-	- S 12 °C	96 h	LC_{50}	970		Korn et al. (1979)
		S 8 °C	96 h	LC_{50}	1020		
		S 4 °C	96 h	LC_{50}	2200		

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic two-ring PAH (contd)							
Naphthalene (contd)							
Parhyale hawaiensis	Adult	– S 22 °C	24 h	LC_{50}	6000		Lee & Nicol (1978)
Aromatic three-ring PAH							
Anthracene							
Artemia salina	1 d	N S 1 h UV[c]	3 h	LC_{50}	20		Kagan et al. (1985)
Artemia salina		N S Dark[c]	24 h	LC_{50}	> 50		Abernethy et al. (1986)
Daphnia magna		– – UV-A=0	21 d	NOEC	2.2	Population growth	Foran et al. (1991)
		UV-A=31[c]	21 d	NOEC	2.2		
		UV-A=60[c]	21 d	NOEC	2.2		
		UV-A=117[c]	21 d	NOEC	1.9		
Daphnia magna	Adult	N S 1 h UV[c]	2 h	LC_{50}	20		Kagan et al. (1985)
Daphnia magna		A R (0.5d) No solvent –UV:1 d; +UV:0.21 d[c]	1.21 d	LC_{50}	15		Newsted & Giesy (1987)
Daphnia magna	4-6 d	N S Dark	48 h	LC_{50}	35.6		Abernethy et al. (1986)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic three-ring PAH (contd)							
Anthracene (contd)							
Daphnia magna	4–6 d	N S No solvent	48 h	LC_{50}	3030[b]		Bobra et al. (1983)
Daphnia magna	< 24 h	S	48 h	LC_{0}	< 500[b]		Eastmond et al. (1984)
Daphnia pulex		A S 0.25 h sun, 1 d dark[c]	24.25 h	EC_{50}	1.2	Immobility	Alfred & Giesy (1985)
		A S 0.17 h sun, 1 d dark[c]	24.17 h	EC_{100}	9.6		
		A S 0.75 h filtered sun, 1 d dark[c]	24.75 h	NOEC EC 75%	12.7 26.4		
Daphnia pulex		N S Solvent, acetone	48 h	EC_{50}	754[b]	Immobility	Smith et al. (1988)
Benzo[a]fluorene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.96 d[c]	1.96 d	LC_{50}	4.8		Newsted & Giesy (1987)
Benzo[b]fluorene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.93 d[c]	1.93 d	LC_{50}	2.2		Newsted & Giesy (1987)
Fluoranthene							
Artemia salina		N S 1 h sun[c]	3 h	LC_{50}	40		Kagan et al. (1985)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic three-ring PAH (contd)							
Fluoranthene (contd)							
Daphnia magna	Adult	N S 1 h sun[c]	2 h	LC_{50}	4		Kagan et al. (1985)
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.45 d[c]	1.45 d	LC_{50}	9.0		Newsted & Giesy (1987)
Daphnia magna		N S	48 h	LC_{50}	325 000[b]		US Environmental Protection Agency (1978b)
Daphnia magna	< 24 h	N S Solvent	48 h	LC_{50} NOEC	320 000[b] < 8800[b]	Mortality	LeBlanc (1980)
Mysidopsis bahia			96 h	LC_{50}	40		US Environmental Protection Agency (1978b)
Phenanthrene							
Artemia salina		N S Dark	24 h	LC_{50}	677		Abernethy et al. (1986)
Daphnia magna		A S	48 h	LC_{50}	700		Millemann et al. (1984)
Daphnia magna		N S Solvent, acetone + triton-X-100	48 h	LC_{50}	840		Eastmond et al. (1984)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic three-ring PAH (contd)							
Phenanthrene (contd)							
Daphnia magna	4–6 d	N S No solvent	48 h	LC_{50}	1160		Bobra et al. (1983)
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:1 d[c]	2 d	NOEC	40.1	Immobility	Newsted & Giesy (1987)
Daphnia magna		A F Glass column	21 d	LC_{50}	130		Hooftman & Evers-de Ruiter (1992a)
			21 d	EC_{50}	50	Reproduction	
			21 d	NOEC	21	Reproduction	
			21 d	NOEC	66	Mortality	
			21 d	NOEC	38	Growth, condition, behaviour	
		A R Glass column	21 d	EC_{50}	180	Reproduction	
			21 d	NOEC	100	Reproduction	
Daphnia magna	4–6 d	N S Dark	48 h	LC_{50}	207		Abernethy et al. (1986)
Daphnia pulex	1.9–2.1 mm	N S No solvent	96 h	LC_{50}	100		Trucco et al. (1983)
Daphnia pulex		N S Solvent, acetone	48 h	EC_{50}	350	Immobility	Smith et al. (1988)
Daphnia pulex		N S	48 h	EC_{50}	734	Immobility	Passino & Smith (1987)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic three-ring PAH (contd)							
Phenanthrene (contd)							
Daphnia pulex			48 h	LC_{50}	960-1280		Geiger & Buikema (1982)
	24 h	A R (1 d) No solvent	~ 50 d	NOEC	110	Reproduction, growth	
Daphnia pulex		N R (2-3 d) 97% A.I. Solvent, acetone	21 d	$LC_{33-73\%}$ LOEC	130 60	Reproduction, growth	Savino & Tanabe (1989)
Gammarus minus		A S	48 h	LC_{50}	460		Millemann et al. (1984)
Aromatic four- ring PAH							
Benz[a]anthracene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.52 d[c]	1.52 d	LC_{50}	1.8		Newsted & Giesy (1987)
Daphnia pulex	1.9–2.1 mm	N S No solvent photo period: 12 h	96 h	LC_{50}	10		Trucco et al. (1983)
Chrysene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:1 d[c]	2 d	LC_{50}	0.7		Newsted & Giesy (1987)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic four- ring PAH (contd)							
Chrysene (contd)							
Daphnia magna	Juvenile + adult	N S Solvent, acetone l/d = 16 h/8 h	2 d	NOEC	288[b]	Mortality	Eastmond et al. (1984)
Pyrene							
Artemia salina	1 d	N S 1 h UV[c]	3 h	LC_{50}	8		Kagan et al. (1985)
Artemia salina		N S Dark	24 h	LC_{50}	> 99		Abernethy et al. (1986)
Daphnia magna	Adult	N S 1 h UV[c]	2 h	LC_{50}	4		Kagan et al. (1985)
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.14 d[c]	1.14 d	LC_{50}	5.7		Newsted & Giesy (1987)
Daphnia magna	4–6 d old	N S Dark	48 h	LC_{50}	91		Abernethy et al. (1986)
Daphnia magna	4–6 d old	N S no solvent	48 h	LC_{50}	1820[b]		Bobra et al. (1983)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic four- ring PAH (contd)							
Triphenylene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:1 d[c]	2 d	NOEC	1.7	Mortality	Newsted & Giesy (1987)
Aromatic five-ring PAH							
Benzo[a]pyrene							
Artemia salina	Eggs		48 h	NOEC	10 000[b]	Viability	Kuwabara et al. (1980)
Calanus heigolandicus	Adult		–	LC	4	Mortality	Lee et al. (1972)
Calanus heigolandicus			7 d	EC	50[b]	AHH activity	Walters et al. (1979)
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.19 d[c]	1.19 d	LC_{50}	1.5	Stimulation	Newsted & Giesy (1987)
Daphnia pulex	1.9–2.1 mm	N S No solvent photo period: 12 h	96 h	LC_{50}	5[a]		Trucco et al. (1983)
Benzo[e]pyrene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.64 d[c]	1.64 d	LC_{50}	0.7		Newsted & Giesy (1987)
Benzo[k]fluoranthene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.54 d[c]	1.54 d	LC_{50}	1.4[a]		Newsted & Giesy (1987)

Table 103 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
CRUSTACEANS (contd)							
Aromatic five- ring PAH (contd)							
Dibenz[a,h]anthracene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.13 d[c]	1.13 d	LC_{50}	0.4[a]		Newsted & Giesy (1987)
Perylene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.76 d[c]	1.76 d	LC_{50}	0.6[a]		Newsted & Giesy (1987)
Aromatic six-ring PAH							
Benzo[ghi]perylene							
Daphnia magna		A R (0.5 d) No solvent –UV:1 d; +UV:0.58 d[c]	1.58 d	LC_{50}	0.2		Newsted & Giesy (1987)

A, analysed concentration; N, nominal concentration; S, static system; F, flow-through system; IF, intermittent flow; R(0.5 d), system with renewal (each half day); S\F, first period in a static system followed by a period in a flow-through system; l/d, light/dark; +UV, with UV radiation; –UV, without UV radiation; DMSO, dimethyl sulfoxide; EC, exposure concentration; NOEC, no-observed-effect concentration; LC, lethal concentration; LOEC, lowest-observed-effect concentration

[a] Concentration higher than solubility but not exceeding it by 10 times
[b] Concentration 10 times higher than the solubility
[c] Explicitly mentioned that organisms were tested for phototoxicity of test substance either in sunlight or artificial UV radiation
[d] From Cairns & Nebeker (1982)

the internal concentration of PAH that caused the death of 50% of the *Daphnia* was 77 nmol/g wet weight in continuous light. No toxic effects were seen in the absence of light. The phototoxicity of anthracene was confirmed by experiments with *Daphnia pulex* (Newsted & Giesy, 1987).

PAH-polluted elutriates derived from polluted sediments were highly toxic to *Daphnia magna* when combined with either sunlight or 354-nm near-ultraviolet radiation, whereas none of the elutriates was toxic in the absence of light (Davenport & Spacie, 1991).

A 50% decrease in feeding rate was reported in the mussel *Mytilus edulis* after nine days' exposure to 80 µg/litre fluoranthene (Donkin et al., 1989). When *Mercenaria mercenaria* clams were exposed in flow-through tanks, in which seawater was pumped through sand columns adsorbing 50 mg benzo[*a*]pyrene, the concentrations in the water were generally below the detection limit (< 0.001 µg/litre), whereas the tissue concentrations were 2–4 µg/kg (0.15 µg/kg in control clams). This resulted in an increased intrahaemocytic lysozyme concentration and significantly impaired ability to clear bacteria. Thus, resistance to bacterial infection is decreased by PAH (Anderson et al., 1981).

The 96-h LC_{50} values in the marine polychaete *Neanthes arenaceodentate* were 3800 µg/litre for the two-ring PAH naphthalene and 1000 µg/litre for fluorene, 600 µg/litre for phenanthrene, and 300 µg/litre for 1-methylphenanthrene (three-ring). None of the four- and five-ring PAH were toxic up to the highest concentration tested, 1000 µg/litre, except fluoranthene, which had a 96-h LC_{50} of 500 µg/litre (Rossi & Neff, 1978).

9.1.2.3 Vertebrates

Data on toxicity to vertebrates like fish and amphibians are available for two- to five-ring PAH (Table 104). Most of the data are derived from phototoxicity tests.

As discussed in Section 4, fish can metabolize PAH into intermediates that may have teratogenic, mutagenic, or carcinogenic properties and are associated with hepatic tumours in free-living fish. In addition, certain PAH can cause physiological changes that affect the growth, reproduction, swimming performance, and respiration of fish. The effect of environmental carcinogens on fish populations depends on the exposure received at each susceptible life stage, the ability at each stage to absorb and metabolize the carcinogen and repair the ensuing damage, and the consequences of tumour induction in vital organs at each life stage. Additional factors that influence carcinogenicity are the stage of organism development, the route of exposure, genetic variation, and cytochrome P450 mixed-function oxygenase activity (Bailey et al., 1989).

Several PAH can produce cancer-like growths and are teratogenic and mutagenic to fish. In *Oncorhynchus mykiss* (former name for *Salmo gairdneri*),

Table 104. Results of tests for the toxicity of polycyclic aromatic hydrocarbons (PAH) towards vertebrates

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH							
Aromatic two- ring PAH							
Acenaphthene							
Cyprinodon variegatus	Juvenile	N S	96 h	LC_{50}	2200		Heitmuller et al. (1981)
Ictalurus punctatus		A F solvent	96 h	LC_{50}	1720		Holcombe et al. (1983)
Lepomis macrochirus	0.32–1.2 g	N S solvent	96 h	LC_{50}	1700		Buccafusco et al. (1981)
Pimephales promelas	Juvenile (32 d)	A F solvent	96 h	LC_{50}	1600		Holcombe et al. (1983)
Pimephales promelas	Embryo–juvenile	A F l/d = 16/8 h	32 d	NOEC	509	Survival	Cairns & Nebeker (1982)
		Glass column	96 h	LC_{50}	608		
		Solvent, DMF					
Oncorhynchus mykiss	Juvenile	A F Solvent,	96 h	LC_{50}	670		Holcombe et al. (1983)
		isopropanol	48 h	LC_{50}	1130		
Salmo trutta	Juvenile	A F Solvent,	69 h	LC_{50}	580		Holcombe et al. (1983)
		isopropanol	48 h	LC_{50}	650		
Acenaphthylene							
Oryzias latipes			48 h	LC_{50}	11 000		Yoshioka et al. (1986)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic two- ring PAH (contd)							
Fluorene							
Lepomis macrochirus		A F	30 d	NOEC	19	Predating prey	Finger et al.
				NOEC	42	Growth	(1985)
				NOEC	49	Mortality	
		N S	96 h	LC_{50}	910		
Oncorhynchus mykiss		N S	96 h	LC_{50}	820		
Pimephales promelas		N S	96 h	LC_{50}	> 100 000[a]		
Naphthalene							
Abramis bramana	1 year	N R	96 h	LC_{50}	10 000		Frumin et al. (1992)
Fundulus heteroclitus			30 d	NOEC	1600		US Environmental Protection Agency (1986d)
Micropterus salmoides	Eggs and larvae (to 4 d posthatch	A F No solvent	7 d	NOEC	28	Survival	Black et al. (1983)
				LC_{6-36}	28–239		
	Embryo–larva (to 4 d posthatch)	A F No solvent	7 d	LC_{50}	510		
Micropterus salmoides	Eggs–larvae	A F No solvent	7 d	LC_{50}	680		Millemann et al. (1984)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic two- ring PAH (contd)							
Naphthalene (contd)							
Oncorhynchus gorbuscha	Fry	- S 4–12 °C	96 h	LC_{50}	1370–1240		Korn et al. (1979)
Oncorhynchus kisutch	1 g	A F	96 h	LC_{50}	770	Parasitized	Moles (1980)
Oncorhynchus kisutch	1 g	A F	40 d	EC	670–1400	Growth	Moles et al. (1981)
Oncorhynchus kisutch	1 g	A F	96 h	LC_{50}	2100		Moles (1980)
Oncorhynchus kisutch	1 g	A F	96 h	LC_{50}	3220	Unparasitized	Moles (1980)
Oncorhynchus mykiss	Eggs–larvae (to 4 d post-hatch)	A F No solvent	23 d 27 d	NOEC NOEC LC_{50}	15 15 110	Hatching Survival	Black et al. (1983)
Oncorhynchus mykiss	Eggs–larvae	A F No solvent	27 d	NOEC	120	Survival, teratogenity	Millemann et al. (1984)
Oncorhynchus mykiss	Adult	A F	96 h	LC_{50}	1600		DeGraeve et al. (1982)
Oncorhynchus mykiss	13–21 d	N S Solvent, acetone	96 h	LC_{50}	4500		Edsall (1991)
Oncorhynchus mykiss		A F	96 h	LC_{50}	2300		DeGraeve et al. (1980)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic two- ring PAH (contd)							
Naphthalene (contd)							
Pimephales promelas	Embryo–larva	A F	30 d	NOEC	450	Growth,	DeGraeve et al.
				EC	850	hatching	(1982)
			Sub-chronic	LOEC	850	Reproduction	
Pimephales promelas	Juvenile	A S No solvent	96 h	LC_{50}	1990		Millemann et al. (1984)
Pimephales promelas		A F	96 h	LC_{50}	4900	Reproduction	DeGraeve et al. (1980)
Pimephales promelas		A F	96 h	LC_{50}	6140		Geiger et al. (1985)
Pimephales promelas		A F	96 h	LC_{50}	6080		Holcombe et al. (1984)
Pimephales promelas	Adult	A F	96 h	LC_{50}	7900		DeGraeve et al. (1982)
Pimephales promelas		A F	96 h	LC_{50}	8900		DeGraeve et al. (1980)
Tilapia oreochromis		N R	96 h	LC_{50}	22 400		Frumin et al. (1992)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic three- ring PAH							
Anthracene							
Lepomis macrochirus	1–1.5 g	N F 24 h UV, 0 h dark[b]	200 h	NOEC	1.2	Mortality	Oris & Giesy (1986)
Lepomis sp.	Juvenile (2–3 cm)	A F Low UV intensity[b]	96 h	LC_{50}	2.78		Oris & Giesy (1985)
Lepomis macrochirus	1–1.5 g	N F 24 h UV[b]	125 h	LC_{50}	4.5		Oris & Giesy (1986)
			96 h	LC_{50}	4.5		
Lepomis macrochirus			96 h	LC_{50}	11.9		US Environmental Protection Agency (1987b)
Lepomis sp.	Juvenile (2–3 cm)	A F High UV intensity[b]	96 h	LC_{50}	11.9		Oris & Giesy (1985)
Lepomis macrochirus		A F Shadow[b], microcosm + sediment	144 h	LC_{0}	12.7		Bowling et al. (1983)
		2 h sun[b]	25 h	LC_{50}	12.7		
		12 h sun[b]	72 h	LC_{100}	12.7		
Lepomis macrochirus	1–1.5 g	N F 12 h UV, 12 h dark[b]	200 h	NOEC	15	Mortality	Oris & Giesy (1986)
Lepomis sp.	Juvenile (2–3 cm)	A F Medium UV intensity[b]	96 h	LC_{50}	18.2		Oris & Giesy (1985)
		Low UV intensity[b]	96 h	LC_{50}	26.5		

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic three- ring PAH (contd)							
Anthracene (contd)							
Lepomis macrochirus	1–1.5 g	N F 6 h UV, 18 h dark[b]	96 h	LC_{50}	46		Oris & Giesy (1986)
Pimephales promelas		A R (0.5 d) No solvent; dark: 1 d; +UV:16 h[b]	1.6 d	LC_{50}	5.4		Oris & Giesy (1987)
		Dark	4 d	NOEC	5.4	Mortality	
Pimephales promelas	Adult	A F No sun	9 weeks	LOEC	6.6	Egg production	Hall & Oris (1991)
	F_1	A R No sun		NOEC	6.6	Deformities	
		No sun		NOEC	≥12	Survival and	
		Sun[b]		LOEC	12	hatching	
Pimephales promelas	0.8 g	N S 0.5 h sun[b]	24 h	LC_{50}	360[c]	Hatcing	Kagan et al. (1985)
Fluoranthene							
Brachydanio rerio	Eggs–larvae	A F Yellow light	41 d	NOEC	4.8	Growth	Hooftman & Evers-de Ruiter (1992b)
		Solvent, TBA	41 d	NOEC	48	Mortality	
Cyprinodon variegatus	Juvenile	N S	96 h	LC_0	560 000[a]		Heitmuller et al. (1981)
Lepomis macrochirus		N S	96 h	LC_{50}	3980[a]		US Environmental Protection Agency (1978b)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic three- ring PAH (contd)							
Fluoranthene (contd)							
Lepomis macrochirus	0.32–1.2 g	N S Solvent	96 h	LC_{50}	4000[a]		Buccafusco et al. (1981)
Pimephales promelas	0.8 g	N S 0.5 h surf[b]	24 h	LC_{50}	200		Kagan et al. (1985)
Phenanthrene							
Brachydanio rerio	Eggs–larvae (2 d)	A R Yellow light solvent, TBA	28 d	NOEC	28	Growth	Hooftman & Evers-de Ruiter (1992d)
			28 d	LOEC	49		
Gambusia affinis			24 h	LC_{50}	150		Neff (1979)
Micropterus salmoides	Eggs–larvae	A F	7 d	LC_{50}	180		Black et al. (1983)
Micropterus salmoides	Eggs–larvae	A F No solvent	7 d	LC_{50}	250		Millemann et al. (1984)
Oncorhynchus mykiss	Eggs–larvae	A F No solvent	23 d	NOEC	4	Hatching	Black et al. (1983)
			27 d	NOEC	4	Survival	
Oncorhynchus mykiss	Eggs–larvae	A F No solvent	27 d	LC_{50}	30		Millemann et al. (1984)
Oncorhynchus mykiss	Eggs–larvae	A F	7 d	LC_{50}	40		Black et al. (1983)
Oncorhynchus mykiss		N S Solvent, acetone	96 h	LC_{50}	3200[c]		Edsall (1991)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic three- ring PAH (contd)							
Phenanthrene (contd)							
Pimephales promelas		A R (0.5 d) No solvent; dark: 1 d; +UV:4 d[b]	5 d	NOEC	10	Mortality	Oris & Giesy (1987)
		Dark	5 d	NOEC	10		
Aromatic four- ring PAH							
Benz[a]anthracene							
Pimephales promelas		A R (0.5 d) No solvent; dark: 1 d; +UV:4 d[b]	3.7 d	LC_{50}	1.8		Oris & Giesy (1987)
		Dark	5 d	NOEC	1.8	Mortality	
Poecilia formosa		F Injection	9 months	EC	23[c]	Thyroid morphology	Woodhead et al. (1982)
Benzo[k]fluoranthene							
Brachydanio rerio	Eggs–larvae	N F Yellow light solvent, TBA	42 d	LC_{50}	0.68		Hooftman & Evers-de Ruiter (1992c)
			42 d	NOEC	0.23	Growth	
			42 d	NOEC	0.40	Mortality	
Poecilia formosa		F Injection	9 months	EC	23[c]	Spleen morphology	Woodhead et al. (1982)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic four-ring PAH (contd)							
Chrysene							
Brachydanio rerio	Eggs–larvae	A R (2 d) Yellow light, glass column	28 d	NOEC	> 0.9	Growth, mortality, hatching	Hooftman & Evers-de Ruiter (1992b)
Pyrene							
Pimephales promelas		A R (0.5 d) No solvent; dark: 1 d; +UV:0.13 d[b]	1.13 d	LC_{50}	25.6		Oris & Giesy (1987)
		Dark	4 d	NOEC	25.6	Mortality	
Pimephales promelas		N S Solvent, DMSO	24 h	NOEC	≥ 10 000[c]	Mortality	Kagan et al. (1987)
Aromatic five-ring PAH							
Benzo[a]pyrene							
Brachydanio rerio	Eggs–larvae	A R (2 d) Yellow light, glass column	28 d	NOEC	> 4	Growth, mortality, hatching	Hooftman & Evers-de Ruiter (1992b)
Fundulus grandis		F Injection	7 d	EC 18%	30 mg/kg bw	Liver weight increase	Melius et al. (1980)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic five- ring PAH (contd)							
Benzo[a]pyrene (contd)							
Hippoglossoides elassodon	Adult	Food	5 h	EC 21%	8 mg/kg	Hatching success	Hose et al. (1981)
Ictalurus punctatus	Juvenile		7–10 months	EC	1.0	Skeletal structure, pigmentation	Martin (1980)
Leuresthes tenuis	Embryo	A S Solvent, acetone	14 d	NOEC	7.0[c]	Hatching success	Winkler et al. (1983)
			14 d	EC 12%	7.0[c]	Larval morphology	
Leuresthes tenius	Larvae		-	EC	100[a]	Growth	Puffer et al. (1979)
				EC	100[a]	Development	
Micropogonias undulatus	Adult 45–60 g	Oral	30 d	NOEC	5.7 mg/kg (fresh wt)	Behaviour, growth, respiration, locomotion	Thomas (1988)
				EC 34%	5.7 mg/kg (fresh wt)	Ovarian growth, hormone level	

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic five- ring PAH (contd)							
Benzo[a]pyrene (contd)							
Oryzias latipes	6–10 d	A R 0.5 mg/litre DMF	2 x 0.25[d]	NOEC	47+11[a,c]	Neoplastic lesions	Hawkins et al. (1988, 1990)
			2 x 0.25[e]	NOEC	47+11[a,c]		
			2 x 0.25[f]	NOEC	< 47+11[a,c]		
Oncorhynchus kisutch	Embryos 1 d after fertilization	N S\F0.1 mmol/litre DMSO	24 h[g]	NOEC	10 000[a]	Emergence	Ostrander et al. (1988)
	1 week before hatching			NOEC	10 000[a]	Emergence	
	1 d after fertilization			NOEC	< 10 000[a]	Orientation	
	1 week before hatching			NOEC	< 10 000[a]	Orientation	
Oncorhynchus mykiss	Eggs	Injection		EC	4.5 mg/egg	Carcinogenicity	Black et al. (1988)
Oncorhynchus mykiss	Embryo–larva	A R Column	36 d	NOEC	2.4[c]	Morphology	Hannah et al. (1982)
			36 d	LOEC	6.7[c]	Morphology	
Parophyrus vetulus	Eggs–larvae	A S Solvent, ethanol	5 d	NOEC	> 2.1	Development	Hose et al. (1982)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic five- ring PAH (contd)							
Benzo[a]pyrene (contd)							
Pimephales promelas		A R (0.5 d) No solvent; dark: 1 d; +UV:40 h[b]	2.7 d	LC_{50}	5.6[c]		Oris & Giesy (1987)
		Dark[b]	4 d	NOEC	5.6	Mortality	
Poecilia reticulata	6–10 d	A S 0.5 mg/litre DMF; dark[b]	2 x 0.25[e]	NOEC	32+8	Neoplastic lesions	Hawkins et al. (1988, 1990)
			2 x 0.25[f]	NOEC	32+8		
Poeciliopsis lucida		A S Solvent, acetone	1 d/6 months	LC 70%	3750[a]	Lethal effects	Goddard et al. (1987)
Poeciliopsis lucida				NOEC	1250[a]	Lethal effects	
Psettichtys melanostictus	Embryo	A F Solvent, ethanol	5 d	EC 50%	0.1	Hatching success	Hose et al. (1982)
			7 d	EC	0.1	Development	
Benzo[e]pyrene							
Pimephales promelas		A R (0.5 d) No solvent; dark: 1 d; +UV:4 d[b]	5 d	NOEC	2.9	Mortality	Oris & Giesy (1987)
		Dark	5 d	NOEC	2.9	Mortality	

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
FISH (contd)							
Aromatic five- ring PAH (contd)							
Dibenz[a,h]anthracene							
Pimephales promelas	Larvae	A R (0.5 d) No solvent; dark: 1 d; +UV:4 d[b]	5 d	NOEC	0.15	Mortality	Oris & Giesy (1987)
		Dark	5 d	NOEC	0.15	Mortality	
Perylene							
Pimephales promelas	Larvae	A R (0.5 d) No solvent; dark: 1 d; +UV:4 d[b]	5 d	NOEC	1.7[c]	Mortality	Oris & Giesy (1987)
		Dark	5 d	NOEC	1.7[c]	Mortality	
Aromatic six- ring PAH							
Benzo[ghi]perylene							
Brachydanio rerio	Eggs–larvae	A R (2 d) Yellow light, glass column	28 d	NOEC	> 0.16	Growth, mortality, hatching	Hooftman & Evers-de Ruiter (1992d)
Pimephales promelas	Larvae	A R (0.5 d) No solvent; dark: 1 d; +UV:4 d[b]	5 d	NOEC	0.15	Mortality	Oris & Giesy (1987)
		Dark	5 d	NOEC	0.15	Mortality	

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
AMPHIBIANS							
Aromatic two- ring PAH							
Naphthalene							
Xenopus laevis	Larvae (3 weeks)	F Fluorescent light	96 h 6 h	LC_{50} EC_{50}	2100 1700–2300	Absence of swimming	Edmisten & Bantle (1982)
Aromatic three-ring PAH							
Anthracene							
Rana pipiens	Embryo	Sun[b]	24 h	LC_{50}	110[c]		Kagan et al. (1985)
Rana pipiens			5 h	LC_{50}	25		US Environmental Protection Agency (1987b)
Fluoranthene							
Rana pipiens		N S 1 h sun[b]	24 h	LC_{50}	90		Kagan et al. (1985)

Table 104 (contd)

PAH, species	Life stage	Test conditions	Duration	Effect	Concentration (µg/litre)	End-point	Reference
AMPHIBIANS (contd)							
Aromatic four- ring PAH							
Pyrene							
Rana pipiens	Embryo	Sun[b]	24 h	LC_{50}	140[c]		Kagan et al. (1985)

A, analysed concentration; N, nominal concentration; S, static system; F, flow-through system; IF, intermittent flow; R (0.5 d), system with renewal (each half day); S\F, first period in a static system, second in a flow-through system; l/d, light/dark; UV, with UV radiation; –UV, without UV radiation; TBA, tertiary butyl alcohol; DMF, dimethylformamide; DMSO, dimethyl sulfoxide; LC, lethal concentration; NOEC, no-observed-effect concentration; EC, effect concentration; LOEC, lowest-observed-effect concentration

[a] Concentration 10 times higher than solubility

[b] Explicitly mentioned that organisms were tested for phototoxicity of test substance either by sunlight or artificial UV radiation

[c] Concentration higher than solubility but not exceeding it by 10 times

[d] Exposure followed by 168 d depuration

[e] Exposure was followed by 252 d depuration

[f] Exposure followed by 365 d depuration

[g] Exposure followed by 62 d depuration

the liver was the primary target after exposure to benzo[*a*]pyrene in the diet and by intraperitoneal injection. Administration by the latter route, while not directly relevant to environmental exposure, also produced a fibrosarcoma and a stomach papilloma in one individual, along with tumours, indicating that the route of exposure is of some importance in fish (Hendricks et al., 1985).

Painting of benzo[*a*]pyrene and 3-methylcholanthrene onto the skin of three freshwater fish twice a week for three to six months caused epitheliomas in *Gasterosteus aculetus* (stickleback) and *Rhodeus amarus* (bitterling) but not in *Cyprinus carpio* (carp). In the same study, 10 injections of 10 mg benzo[*a*]pyrene in glycerol to *G. aculetus* produced injection-site necrosis but no tumours (Ermer, 1970).

The phototoxic potential of PAH in fish is influenced by the amount of ultraviolet radiation and light absorbed. Adult fathead minnows (*Pimephales promelas*) were exposed in the absence of artificial ultraviolet radiation to 0.6 or 12 µg/litre anthracene for six weeks, the dose being increased to 20 µg/litre for three weeks in two groups. Eggs were collected daily, placed in clean water, and exposed or unexposed to ultraviolet radiation, until 96 h after hatching. All fish showed impaired egg production. The lethal concentration in eggs was estimated to be 19 µg/g (Hall & Oris, 1991).

Exposure of the spot *Leiostomus xanthurus* to PAH-contaminated sediment derived from a river station resulted in death, with fin erosion, ulceration of the lateral body surface, and several lesions of internal organs. The total concentration of the 21 PAH analysed in the sediment was 21 000 mg/kg dry weight. The 28-day LC_{50} was estimated to be 3.2% of contaminated sediment (Roberts et al., 1989).

9.1.2.4 Sediment-dwelling organisms

The 10-day LC_{50} values for fluoranthene in sediment were 11 mg/kg dry weight for the marine amphipod *Eohaustorius estuarius*, 5.1 mg/kg for the marine amphipod *Rhepoxynius abronius*, and 15 mg/kg for the freshwater amphipod *Hyalella azteca*. The sensitivity of *E. estuarius* was not influenced by salinity (DeWitt et al., 1989).

The toxicity of a sediment containing 46% fines and 0.9% total organic carbon and artificially supplemented with solutions of fluoranthene, phenanthrene, benz[*a*]anthracene, benzo[*a*]pyrene, 2,6-dimethylnaphthalene, 1-methylnaphthalene, and 2-methylnaphthalene was tested in the amphipod *Rhepoxynius abronius*. Significant mortality occurred among amphipods exposed for 10 days to nominal concentrations of 15, 10, 10, 5, 0.5, 0.5, and 0.5 mg/kg dry weight of the seven PAH, respectively, whereas at levels five times lower no toxic effects were observed (Plesha et al., 1988).

The ability of an amphipod species to metabolize xenobiotic compounds to reactive intermediates appears to influence its sensitivity to chemical

contaminants. Contaminated sediments are generally more toxic to *R. abronius* than *E. washingtonianus* (Plesha et al., 1988). *R. abronius* and *E. washingtonianus* exposed to sediment-associated, radiolabelled benzo[*a*]pyrene accumulated similar concentrations of radiolabel in their tissues after seven days of exposure, but the proportions of metabolites and the amount of radiolabel bound to cellular macromolecules were greater in *R. abronius* than in *E. washingtonianus*. One explanation for the greater sensitivity of *R. abronius* might therefore be a greater tendency to form reactive metabolites that are more acutely toxic than the parent compound (Reichert et al., 1985).

The toxicity of fluoranthene in sediment to the marine amphipods *R. abronius* and *Corophium spinicorne* was determined by equilibrium partitioning. Toxicity was determined in well-sorted, fine sands containing organic carbon at 0.18, 0.31, or 0.48% of dry weight. The epibenthic tube-dwelling *C. spinicorne* was less sensitive than the free-burrowing *R. abronius*, possibly because of different routes of exposure. The 10-day LC_{50} values for *R. abronius* were 3.4, 6.5, and 10.7 mg/kg dry weight in sediments with increasing organic carbon. A 10-day LC_{50} value for *C. spinicorne* could be determined only in the sediment with the lowest organic carbon level, at 5.1 mg/kg dry weight. An LC_{50} of 24 µg/litre was calculated on the basis of equilibrium partitioning. In another experiment, a 10-day LC_{50} of 4.2 mg/kg dry weight was determined for *R. abronius* in sediment with 0.26% organic carbon (Swartz et al., 1988).

Diporeia sp. were exposed to a sediment artificially contaminated with a mixture of labelled phenanthrene and pyrene and nine unlabelled PAH at total concentrations of 21, 41, 120, and 330 µmol/kg dry sediment. The amphipods avoided the sediment containing the highest dose during the first six days; the estimated LC_{50} at day 26 was estimated to be 600 mmol/kg dry sediment. Deaths occurred after 26 days' exposure to the two highest concentrations. The concentration of PAH required to elicit 38% mortality at day 19 was 2.9 mmol/kg organism. The authors concluded that PAH probably have a non-polar narcotic mode of action and suggested that their effect is additive, which is supported by the observation that exposure of *Diporeia* for 31 days to pyrene resulted in an LD_{50} of 5.8 mmol/kg organism (Landrum et al., 1991).

9.1.2.5 *Toxicity of combinations of PAH*

In the concentration addition model, additive effects were found for phenanthrene, anthracene, naphthalene, and acenaphthene in *Daphnia magna* (Muñoz & Tarazona, 1993). Although the toxic effects of combined PAH to *Brachydanio rerio* were also found to be additive, the concentrations tested were close to the maximum solubility of the PAH (Hooftman et al., 1993).

9.1.3 Terrestrial organisms

9.1.3.1 Plants

The effects of anthracene on emergence were tested in three native Australian plant species, heath banksia (*Banksia ericifolia*), she-oak (*Casuarina distyla*), and yellow bloodwood (*Eucalyptus eximia*), and in three crop species, oat (*Avena sativa*), cucumber (*Cucumis sativus*), and soya bean (*Glycine max*). *A. sativa* and *C. sativus* were sensitive, with EC_{50} values of 30 and 720 mg/kg, respectively; the the other plants were not sensitive up to the highest concentration tested (1000 mg/kg dry weight) (Mitchell et al., 1988).

9.1.3.2 Invertebrates

Porcellio scaber and *Oniscus asellus* showed little difference in their sensitivity to benzo[*a*]pyrene. The growth of both species was affected after exposure for nine weeks to 100 and 316 mg/kg dry weight. The only difference in response was that the lipid pool was reduced in *O. asellus* and the protein pool was reduced in *P. scaber*. No effects were observed at 32 mg/kg dry weight (Van Brummelen & Stuijfzand, 1993).

The LC_{50} values for fluorene in four earthworm species, *Allolobophora tuberculata, Eisenia foetida, Eudrilus eugenia*, and *Perionyx excavatus*, in an artificial soil were 210, 17, 200, and 170 mg/kg dry weight, respectively, after two weeks' exposure in soil. In the contact test, the LC_{50} values were 120, 170, 47, and 78 $\mu g/m^2$, respectively (Neuhauser et al., 1986). Fluorene at 750 mg/kg dry weight significantly reduced the reproduction of *E. foetida*, but no deaths occurred (Neuhauser & Callahan, 1990).

Benzo[*a*]pyrene was incorporated in the food of the wood louse *Porcellio scaber*, and respiration, growth, and food consumption were measured for four weeks. Consumption was not affected by concentrations up to 125 mg/kg, but at the highest dose, male isopods showed significantly greater assimilation (34%) than controls (26%) due to an active mechanism, which was not observed in females. Growth varied considerably between individuals. At the highest concentrations, however, the growth efficiency of males was significantly decreased. At 25 mg/kg, no significant effects were found (Van Straalen & Verweij, 1991).

The toxicity of PAH to the earthworm *E. foetida* and the springtail *Folsoma candida* was studied in a standard soil. As phenanthrene at a concentration of 1000 mg/kg dry weight appeared to be removed almost completely from the soil after 14 days' exposure, the soil was renewed regularly. The 28-day LC_{50} of phenanthrene in *F. candida* was 150 mg/kg dry weight, the EC_{50} was 120 mg/kg, and the NOEC for reproduction was 75 mg/kg dry weight. The 21-day EC_{50} for reproduction of *E. foetida* was 240 mg/kg dry weight. No effect on the survival or reproduction of *F. candida* was seen after 28 days' exposure

to chrysene, benzo[*k*]fluoranthene, or benzo[*a*]pyrene at 180 mg/kg dry weight, and no effect on the reproduction or survival of *E. foetida* was seen after 14 days' exposure to chrysene at 1000 mg/kg dry weight (Bowmer et al., 1993).

Ingestion of naphthalene, anthracene, benz[*a*]anthracene, pyrene, or benzo[*a*]pyrene at 1000 mg/kg food per day for 18 days caused significant increases in mortality among *Acheta domesticus* crickets. Naphthalene caused 50% mortality within 12 days, and anthracene, benz[*a*]anthracene, pyrene, and benzo[*a*]pyrene caused 39, 26, 23, and 32% mortality, respectively, after 18 days (Walton, 1980).

9.1.3.3 *Vertebrates*

Benzo[*a*]pyrene and chrysene dissolved in oil and covering less than 10% of the surface of duck eggs reduced hatching and had teratogenic and embryotoxic effects (Hoffmann & Gay, 1981). The 72-h LD_{50} values for chick embryos were 14 µg/kg egg for benzo[*k*]fluoranthen, 39 µg/kg for dibenz[*a*,*h*]anthracene, and 79 µg/kg for benz[*a*]anthracene (Brunström et al., 1991). The LD_{50} values for acenaphthene, anthracene, phenanthrene, and fluorene in red-winged blackbirds were > 100 mg/kg (Schafer et al., 1983).

9.2 Field observations

9.2.1 Microorganisms

9.2.1.1 *Water*

The effects on a benthic community were determined in the sediment of a stream in which a gradient of PAH concentrations was found, with total PAH contents at four sites of 0, 3100, 39 000, and 49 000 mg/kg dry weight. Detrital accumulation and the redox potentials increased with PAH level. Removal of fungi at the polluted sites was probably the major factor in detrital accumulation, and a reduction in bacterial biomass was thought to be the primary cause of the increased redox potentials (Catallo & Gambrell, 1987).

Addition of 1000 mg/kg dry weight naphthalene to anaerobic salt marsh sediments resulted in significant inhibition of methanogenesis, sulfate reduction, and evolution of carbon dioxide. Phenanthrene at the same concentration had no significant effect on these activities, whereas the same dose of naphthalene inhibited methanogenesis and then stimulated it relative to controls; however, the sulfate reduction was sustained. Carbon dioxide evolution was reduced in only one of the three experiments (Kiene & Capone, 1984).

9.2.1.2 *Soil*

No data were available

9.2.2 Aquatic organisms

9.2.2.1 Plants

No data were available

9.2.2.2 Invertebrates

In the study of Catallo & Gambrell (1987), described in section 9.2.1.1, the population densities of nematodes, oligochaetes, and other benthic invertebrates were significantly decreased at the site with a total PAH content of 3100 mg/kg and were eradicated at the other two sites.

Ecosystem responses were tested in small, multispecies, aquatic systems (Leffler microcosm) exposed once to fluorene dissolved in acetone at a concentration of 0.12, 0.50, 2, 5, or 10 mg/litre. The estimated half-life was 2.1 days. The LOEL for respiration in the dark (R_{ni}) and the ratio of net productivity:R_{ni} was 0.12 mg/litre in all four communities, suggesting that the responses of these microcosms were not completely independent of their source communities. At 5 and 10 mg/litre, the zooplankton populations were almost eliminated (Stay et al., 1988).

9.2.2.3 Vertebrates

PAH metabolites were found in English sole (*Parophrys vetulus*) with hepatic tumours collected from Puget Sound, Washington, USA (Malins, 1982; Malins et al., 1984). Detectable levels of similar organic free radicals were found only when microsomes were incubated with the PAH fraction of extracts of sediment from this area and not after incubation with the alkane or aromatic polychlorinated biphenyl fractions (Collier et al., 1992).

A consistent, statistically significant association was found between the prevalence of hepatic tumours in free-living *P. vetulus* and the levels of PAH in bottom sediment from sites where the fish were captured, in a series of studies conducted over seven years in Puget Sound. The strongest relationships were found for four categories of hepatic lesion. Other contaminants such as trace metals, polychlorinated biphenyls, pesticides, and chlorinated butadienes were measured in all studies, but the strongest associations with liver lesions were found with PAH. The concentrations of total PAH in the sediment ranged from 0.005 mg/kg dry weight at an uncontaminated site to 540 mg/kg at the most polluted location (Landahl et al., 1990).

9.2.3 *Terrestrial organisms*

9.2.3.1 *Plants*

No data were available.

9.2.3.2 *Invertebrates*

Application of naphthalene at 200 g/m^2 to four soils resulted in a significant reduction in soil arthropods such as *Collembola*. At 10 g/m^2, the densities of most arthropods increased (Best et al., 1978).

9.2.3.3 *Vertebrates*

No data were available.

10. EVALUATION OF RISKS TO HUMAN HEALTH AND EFFECTS ON THE ENVIRONMENT

10.1 Human health

10.1.1 Exposure

Polycyclic aromatic hydrocarbons (PAH) are released into the environment as a result of incomplete combustion of organic materials, especially during industrial processes, incineration of refuse, and fossil fuel combustion. They are released mainly into the atmosphere, adsorbed onto particulate matter, which is deposited in the aquatic and terrestrial environments. Direct contamination may also occur from, e.g. creosote-preserved wood and deposition of contaminated refuse such as sewage sludge and fly ash.

Owing to their long-range transport, PAH, and particularly those with high molecular masses, are ubiquitous in the environment. The levels in ambient air vary considerably, ranging from several picograms per cubic metre to < 1 $\mu g/m^3$, although phenanthrene has been found at levels of several micrograms per cubic metre.

10.1.1.1 General population

Humans are exposed to various complex mixtures of PAH in the air, food, water, and soil. The main sources of human exposure are emissions from the combustion of coal, diesel, petrol, kerosene, wood, biomass, and synthetic chemicals such as plastics. PAH account for a significant portion of the carcinogenicity of some mixtures, such as coal-tar soot, but not of others such as cigarette smoke, diesel emissions, and urban aerosol. The levels of selected PAH of toxicological significance in various environmental media are given in Tables 105 and 106.

Pollution of indoor air by PAH is due mainly to tobacco smoking, residential heating, and PAH from outdoor ambient air (Table 107). Extremely high values (e.g. 15 000 ng/m^3 benzo[*a*]pyrene) were found in unvented rooms with open fireplaces, especially those in which soft coal was used for cooking and heating. High concentrations have also been reported in wood-heated saunas.

The predominant sources of PAH pollution in urban areas are motor vehicle traffic (both petrol- and diesel-fuelled) and residential heating, especially with wood, coal, and biomass. The concentrations are up to one order of magnitude higher in winter than in summer. Such differences may limit the validity of sampling campaigns performed during only part of the year for the purpose of estimating mean human exposure in urban areas.

Table 105. Reported levels of selected polycyclic aromatic hydrocarbons (PAH) in various media

Compound	Ambient air (ng/m³)	Drinking-water (ng/litre)	Surface water (ng/litre)[a]	Soil (µg/kg)	Soil near industrial sources (mg/kg)	Sediment (µg/kg)[b]
Acenaphthene[c]	0.06–370	0.02–14	0.08–1200	1–21	1–5100	0.04–3800
Anthracene	0.004–61	0.5–9.7	0.01–930	0.2–70	0.2–140	0.06–27 000
Benzo[*a*]pyrene	0.002–780	0.04–2.0	0.03–910	0.8–3200	0.8–38	0.004–110 000
Chrysene[d]	0.01–260	No data	10–1100	2.1–2700	2.1–1200	0.04–21 000
Dibenzo[*a*,*l*]pyrene	0.05–1.5	No data	No data	No data	No data	No data
Fluoranthene	0.03–810	0.58–3400	0.4–6400	0.3–3700	0.3–340	0.1–610 000
Fluorene	0.02–420	0.008–21	0.33–2500	1–14	1–8600	0.5–6500
Naphthalene[c]	0.03–940	0.38–8.8	0.4–2100	3–60	3–5.2	0.7–44 000
Phenanthrene	0.002–1800	2.2–90	0.24–5700	17–1700	17–20 000	0.06–65 000
Pyrene	0.002–540	0.3–40	0.12–3100	0.1–4500	0.1–1600	0.1–410 000

Only detected values are given, owing to the variability of limits of detection; data obtained from studies of road tunnels were excluded even though short-term exposure to such high levels may contribute significantly to overall daily exposure.

[a] Concentration may exceed solubility in water owing to presence of particulates in sample

[b] Highest values usually determined in harbour sediments

[c] Probably underestimated because of shortcomings in sampling and analytical procedures; data were not provided from laboratories that found high values for three- to six-ring PAH.

[d] Most measurements performed with gas chromatography, so that actual levels are overestimates due to analytical interference by triphenylene.

Table 106. Reported levels (µg/kg) of selected polycyclic aromatic hydrocarbons (PAH) in food

Compound	Meat and meat products	Fish and seafood[a]	Dairy products	Oil, fats, and margarine	Vegetables and fruit	Cereals
Acenaphthene	No data	0.9–500	No data	0.02–0.45	No data	0.6–0.7
Anthracene	0.9–31	0.05–240	No data	0.02–460	0.09–0.4[b]	0.5–1.3
Benzo[*a*]pyrene	0.01–42 (130[c])	0.003–290	0.08–1.3	0.02–140	0.05–6.2[b]	0.1–0.8
Chrysene[d]	0.15–0.6	0.03–210	1.3–1.5	0.1–120	0.5–69[b]	0.77
Dibenzo[*a,l*]pyrene	No data	No data	No data	No data	No data	No data
Fluoranthene	0.48–100	0.1–1800	0.01–4.2 (8.0[e])	0.02–460	0.93–120[b]	0.3–28
Fluorene	No data	0.2–370	No data	0.02–200	No data	1.3–2.7
Naphthalene[c]	No data	0.8–210	No data	No data	No data	No data
Phenanthrene	3–64	0.1–2700	0.56–0.72	0.09–1400	0.47–17[b]	9.9–29
Pyrene	0.55–63	0.03–1500	0.04–2.7 (4.8[e])	0.02–330	0.83–70[b]	0.22–21

[a] Data from industrially polluted areas included as food items from those areas enter the market

[b] Values detected in vegetables grown on contaminated soil are excluded

[c] Exceptionally high values found in processed foods, but PAH not determined in these studies

[d] Most measurements performed with gas chromatography, so that actual levels are overestimates due to analytical interference by triphenylene.

[e] Found in infant food

Table 107. Ranges of indoor concentrations of selected polycyclic aromatic hydrocarbons

Compound	Concentration (ng/m^3)
Acenaphthene	2.5–1650
Anthracene	1–410
Fluoranthene	5–270
Naphthalene	300–2300
Pyrene	3.6–32
Benzo[*a*]pyrene	0.04–370[a]
Phenanthrene	3–550
Chrysene	0.6–110

[a] Levels up to 14 700 ng/m^3 found in Chinese houses with open fires

No conclusion can be drawn about the relative PAH emissions from petrol-fuelled engines (without catalytic converters) and diesel-fuelled engines, given the limited number of studies in which emissions were compared under the same conditions of sampling and analysis.

Drinking-water generally contains low levels of individual PAH, up to some hundreds of nanograms per litre, depending on the compound. The levels of PAH in beverages, including alcoholic drinks, are usually < 0.01 µg/kg.

10.1.1.2 Occupational exposure

The concentrations in air to which workers are exposed depend on the type of industry. The levels in coke ovens, where the highest exposure may occur, are up to several hundred micrograms per cubic metre. Table 108 shows some levels of occupational exposure.

10.1.2 Toxic effects

10.1.2.1 Bioavailability

Owing to the high lipophilicity of this class of compounds, their bioavailability after ingestion and inhalation must be considered to be significant. Dermal adsorption appears to depend on the PAH being studied and the species evaluated: 3% of an applied dose of benzo[*a*]pyrene was absorbed by human skin and 10% by mouse skin within 24 h.

10.1.2.2 Acute toxicity

Values for the median lethal dose (LD_{50}) indicate that PAH have moderate to low acute toxicity. For example, the oral LD_{50} of benzo[*a*]pyrene is

Table 108. Ranges of occupational exposure to selected polycyclic aromatic hydrocarbons

Compound	Concentration (μg/m³)
Acenaphthene	0.44 in oil refining to 135 in aluminium smelting
Anthracene	0.028 in oil refining to 405 in coke ovens
Fluoranthene	0.085 in oil refining to 191 in coke ovens
Naphthalene	0.22 in roofing to 2900 in food smoke-houses
Pyrene	0.11 in oil refining to 333 in coke ovens
Benzo[*a*]pyrene	0.09 in chimney sweeping to 137 in coke ovens
Phenanthrene	0.085 in road paving to 1167 in coke ovens
Chrysene[a]	0.085 in oil refining to 191 in coke ovens

[a] Most measurements performed by gas chromatography, so that the actual levels may be overestimates due to analytical interference by triphenylene

> 1600 mg/kg in mice, and the oral LD_{50} of naphthalene is 350–700 mg/kg bw in mice but 500–9000 mg/kg bw in rats.

Case reports have shown that exposure to naphthalene results in haemolytic anaemia, and lethal doses of 2–3 g for children and 5–25 g for adults have been reported. There appears to be no cause for concern about any acute toxicity of occupational exposure or exposure of the general population, with the exception of accidental ingestion.

10.1.2.3 Irritation and allergic sensitization

Anthracene, benzo[*a*]pyrene, and naphthalene are primary irritants. Anthracene and benzo[*a*]pyrene were reported to be sensitizers, whereas phenanthrene did not induce contact sensitivity.

10.1.2.4 Medium-term toxicity

Oral administration resulted in no-observed-adverse-effect levels of 175 mg/kg bw per day acanaphthene for hepatotoxicity; 125 mg/kg bw per day fluoranthene for nephropathy, increased relative liver weights, and haematological and clinical effects; 125 mg/kg bw per day fluorene for altered haematological parameters; and 75 mg/kg bw per day pyrene for nephropathy. Anthanthrene at 1000 mg/kg bw per day had no effect.

The daily uptake of PAH by humans is estimated to be 3.7 μg, corresponding to about 0.05 μg/kg bw per day (70 kg bw). Human exposure is thus six orders of magnitude lower than the concentrations administered in studies in mice. There is thus no cause for concern about any medium-term toxic effects in humans for the PAH tested so far.

10.1.2.5 *Carcinogenicity*

The carcinogenicity of individual PAH and PAH-containing mixtures in experimental animals has been well studied. Virtually no data exist on the carcinogenicity of individual PAH in humans, although a limited database on the carcinogenicity of PAH-containing mixtures is available: these have been shown to increase the incidence of cancer in exposed human populations. The finding that a number of individual PAH are carcinogenic to experimental animals indicates that they are potentially carcinogenic to humans. PAH can produce tumours both at the site of contact and distantly, and the carcinogenic potency of PAH may vary with the route of exposure.

(a) Experimental models

Benzo[*a*]pyrene is the best-studied PAH. It has been tested in multiple species and by various routes, including orally, by inhalation, and by skin painting for dermal carcinogenesis. It has been shown to be carcinogenic by all routes tested in a number of animal species. Those species in which no tumours were found are suspected to have been tested at inadequate doses or observed for an insufficient portion of their life span.

Other PAH have been assayed for dermal carcinogenicity as either complete carcinogens or as initiators in initiation-promotion models. Assays used commonly include tests for lung adenomas in newborn mice treated by intraperitoneal, intrapulmonary, or subcutaneous injection. There are insufficient data to determine whether other PAH are carcinogenic.

(b) Epidemiology

Numerous epidemiological studies have been reported of groups of workers exposed to environments that contain mixtures of PAH, all of which also contained chemicals other than PAH. Cases of respiratory diseases such as pneumoconiosis, respiratory tuberculosis, pneumonia, and bronchitis and diseases of the circulatory system were reported in these studies, but these effects are not considered to be specific to PAH because of simultaneous exposure to agents that cause similar effects. Thus, exposure in iron and steel foundries entails exposure not only to PAH but also to other potentially carcinogenic materials such as nickel, chromium, silica, soot, asbestos, and benzene. The working environment of aluminium smelters is unlikely to include nickel or chromium but includes alumina, aluminium fluoride, and aromatic amines. If each of these materials occurred in a separate location in a factory, classical epidemiological techniques would have little difficulty in identifying the agent responsible for a statistically significant cancer excess. This is not the situation, and, while epidemiological studies have produced convincing evidence that cancers occur in workers exposed to PAH, the

attribution of exposure to PAH as the cause of these excesses can only be based on information from animal models.

Studies of workers exposed to mixtures of PAH indicate that the lung is the target organ after inhalation. Confounding by cigarette smoking cannot explain the effects observed. Studies of workers at gas and coke ovens, at aluminium smelters, in iron and steel foundries, and with bitumen and asphalt consistently show excess risks for lung cancer. Coke-oven workers are probably exposed to the highest concentrations, and studies of these populations have provided evidence of dose-response effects. In a comparison of the mortality of 5321 coke-oven workers with that of 10 497 steel workers at the same plants, a monotonic positive trend was shown in the relationship between the risk for lung cancer and the estimated level of cumulative exposure to coal-tar pitch volatiles, the benzene-soluble fraction of particulate matter. With a risk of unity for the unexposed group, the risk increased from 1.2 for those with exposure of 1–49 mg/m^3 x months to 3.1 for the group with the heaviest exposure of ≥ 650 mg/m^3 x months. Analysis of the relative risks and the numbers of deaths from lung cancer on which they were based resulted in the conclusion that 124 deaths occurred among these coke-oven workers over a period of 30 years that can be attributed to exposure to coal-tar pitch volatiles, i.e. 2.3% of the cohort. Earlier findings from this study were used by others to estimate a unit risk coefficient of 8.7×10^{-2} for exposure to benzo[*a*]pyrene, i.e. the absolute lifetime risk of lung cancer from a working lifetime exposure to 1 µg/m^3 of benzo[*a*]pyrene. Given the large number of cancers that occurred in this cohort, this risk coefficient is probably the best estimate currently available. It should be recognized, however, that the reports on which this estimate is based gave relatively little information on exposure levels, no data on time trends in the level of exposure, and no data on benzo[*a*]pyrene levels in the participating plants.

There is also good evidence that urinary bladder cancer has occurred in cohorts of aluminium smelters and gas and coke workers, although the overall findings are not as consistent as those for lung cancer. In a study of aluminium smelters, a positive trend was observed between the risk for urinary bladder cancer and the estimated level of cumulative exposure to benzene-soluble matter. PAH cannot be assumed to be responsible for this trend, however, because the known bladder carcinogen 2-naphthylamine and other aromatic amino and nitro compounds were present in the working environment. Similarly, the excess of urinary bladder cancer in gas workers is more likely to be a consequence of exposure to 2- and 1-naphthylamines.

PAH are almost certainly one of the carcinogenic agents responsible for lung cancers in cigarette smokers, although the role of PAH in the many other diseases caused by cigarette smoking, including nonmalignant diseases of the respiratory system, is unknown. Studies on the rates of mortality from lung cancer in relation to indoor burning of coal or wood in open fires for cooking

and heating add to the body of evidence that links PAH and the risk for lung cancer. Workers exposed to diesel or petrol fumes had relatively low exposure to PAH, and studies of these exposure are not likely to assist in quantification of the risks for cancer associated with exposure to PAH.

Quantitative risk assessments were not made for PAH, either individually or as mixtures; however, Appendix I gives some comparative features of three approaches to quantitative risk assessment that have been used and which have been at least partly validated.

10.1.2.6 Reproductive toxicity

(a) Developmental studies

There are no studies in humans. Embryotoxic effects have been described in experimental animals exposed to PAH such as benz[*a*]anthracene, benzo[*a*]pyrene, and naphthalene. In mice treated intraperitoneally with benzo[*a*]pyrene, increased numbers of stillborn and resorbed fetuses, decreased fetal weight, and increased incidences of congenital anomalies were seen at a minimum dose of 50 mg/kg bw per day. In mice given benzo[*a*]pyrene in the diet, malformations were found after administration of 120 mg/kg bw per day on days 2–10 of gestation. In another study in mice fed the compound during most of the period of gestation, no treatment-related embryotoxic effects were found after doses up to 133 mg/kg bw per day.

In mice, 1 mg benzo[*a*]pyrene per gram of food would result in consumption of 5 g/day. In humans, the total median intake from food, air, water, and soil for a 70-kg person would be 0.05 µg/kg bw per day. In view of the inter- and intraspecies differences, there is at present no reason for concern about effects on development.

(b) Fertility

There are no studies in humans. In mice fed diets containing benzo[*a*]pyrene at doses up to 133 mg/kg bw per day, no treatment-related effects on fertility were seen, again indicating no concern about effects of PAH on fertility in humans.

10.1.2.7 Immunotoxicity

At least one immunotoxic PAH will probably be present in any mixture of PAH. Thus, when such mixtures are evaluated for their potential effects on human health, the immune system should be considered a primary target organ. Benzo[*a*]pyrene caused immunosuppression in mice after dermal application for 28 days at doses as low as 625 µg/kg bw. In a study in which the immune status of coke-oven workers was evaluated, suppressed immune status was

found, as indicated by decreased serum antibody levels and functional impairment.

The mechanisms of immunosuppressive action that have been proposed include formation of active metabolites (including diol epoxides), alterations in cytokine levels, disruption of membrane fluidity, interaction with the Ah receptor, and alterations in calcium ion flux.

10.1.2.8 Genotoxicity

PAH have repeatedly been shown to have genotoxic effects both in *in vivo* in rodents and *in vitro* in mammalian (including human) cell lines and prokaryotes. Some PAH, however, appear not to be genotoxic. Most of the unsubstituted PAH categorized as genotoxic are not genotoxic *per se* but require metabolism to intermediates which react with DNA to form DNA adducts and induce genotoxic damage. Genotoxic events are postulated to be a required step in the carcinogenic process.

10.2 Environment

10.2.1 Environmental levels and fate

Concentrations of up to 100 µg/kg of individual PAH have been detected in soil, although higher concentrations of pyrene, phenanthrene, chrysene, and benzo[*a*]pyrene have been found. The concentrations in soil near industrial sources of PAH are up to three orders of magnitude higher.

Concentrations of up to 6 µg/litre have been reported for individual PAH in surface water, including polluted rivers. High concentrations have also been reported in sediment, which acts as a sink for PAH. The concentrations in sediment from rivers, lakes, and seas are generally < 30 mg/kg dry weight. Concentrations up to 655 mg/kg dry weight have been reported in sediment from harbours. The individual PAH compounds detected in environmental samples vary according to their source and any degradative processes. Phenanthrene is the PAH found in highest concentrations in aquatic samples. Those that occur at the highest concentrations in sediment include phenanthrene, fluoranthene, pyrene, benz[*a*]anthracene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, and indeno[1,2,3-*cd*]pyrene.

PAH are sparingly soluble in water and therefore have affinity for sediment, soil, and biota. They may be removed from the environment by biodegradation or photodegradation. The rates of degradation vary and generally decrease with increasing numbers of aromatic rings. PAH are inherently biodegradable, and low-molecular-mass compounds can be completely degraded by acclimated microorganisms. In surface waters with low numbers of unacclimated microorganisms, PAH can persist for longer periods of time.

10.2.2 Ecotoxic effects

10.2.2.1 Terrestrial organisms

Few data are available on the effects of PAH on terrestrial organisms, and none are available on plants, wild mammals, or birds. The values for the concentration causing 50% lethality (LC_{50}) reported for earthworm species are 150 mg/kg dry weight for phenanthrene and 170–210 mg/kg for fluorene. The no-observed-effect level for the survival and reproduction of earthworm species was 180 mg/kg dry soil for chrysene, benzo[*k*]fluoranthene, and benzo[*a*]pyrene. PAH in soil are unlikely to exert toxic effects on terrestrial invertebrates, except when the soil is highly contaminated.

10.2.2.2 Aquatic organisms

The toxicity of naphthalene, phenanthrene, and fluoranthene on aquatic organisms has been well studied in the laboratory, but that of other PAH has not. The toxicity of PAH to aquatic organisms is affected by metabolism and photo-oxidation, and they are generally more toxic in the presence of ultraviolet light. Naphthalene is the least toxic to invertebrates, with 48-h LC_{50} values of 700–23 000 µg/litre. The LC_{50} values for three-ring PAH range from < 1 to 3000 µg/litre, anthracene being more toxic than the others, with 24-h LC_{50} values of < 1 to 260 µg/litre. Four-, five-, and six-ring PAH have 48-h LC_{50} values of 0.2–1800 µg/litre, their toxicity increasing with molecular mass. The 96-h values for acute toxicity in fish were 110 to > 10 000 µg/litre for naphthalene, 30–4000 µg/litre for three-ring PAH (those for anthracene being 2.8–360 µg/litre), and 0.7–26 µg/litre for four- and five-ring PAH.

The no-observed-effect and lowest-observed-effect levels for invertebrates were 300–1000 µg/litre for naphthalene, 2–600 µg/litre for three-ring PAH, 5–290 µg/litre for four-ring PAH, and 0.1–50 µg/litre for five-ring PAH. The long-term treatment doses resulting in toxicity in fish were 15–1600 µg/litre for naphthalene, 1.2–510 µg/litre for three-ring PAH, 0.9–26 µg/litre for four-ring PAH, and 0.15–7 µg/litre for five-ring PAH. Higher no-observed-effect concentrations have been reported, but in studies in which the PAH were present at more than than 10 times their maximum aqueous solubility; these have therefore been ignored. As the concentrations of PAH reported in surface water are usually in the range of nanograms per litre, they are unlikely to exert adverse effects on aquatic organisms, except in cases of heavy exposure, as with creosote.

The LC_{50} values reported for sediment-dwelling organisms were 3.4–15 mg/kg dry sediment for fluoranthene, 10 mg/kg for phenanthrene, 10 mg/kg for benz[*a*]anthracene, and 5 mg/kg for benzo[*a*]pyrene. Since sediment acts as a sink for PAH, sediment-dwelling organisms may be adversely affected.

PAH may induce neoplastic effects in aquatic organisms. Tumour development has been reported in fish exposed to benzo[*a*]pyrene and 3-methyl-cholanthrene after oral, dermal, or intraperitoneal administration. Hepatic tumours have been found in wild fish living in water with sediment containing PAH at a concentration of 250 mg/kg. Although the levels of PAH in sediment are generally an order of magnitude lower than this value, the possibility that tumours might be formed at lower concentrations cannot be excluded. The ecological significance of the carcinogenic effects of PAH in fish has not been assessed.

11. RECOMMENDATIONS FOR THE PROTECTION OF HUMAN HEALTH AND THE ENVIRONMENT

11.1 General recommendations

- International agreement on analytical procedures and interlaboratory quality control studies is strongly recommended. Sampling strategies and analytical procedures should be optimized and standardized before surveys of exposure to polycyclic aromatic hydrocarbons (PAH) are undertaken.
- Emissions and effluents of PAH from both point and diffuse sources should be monitored and inventories compiled.
- Concentrations of individual PAH should be given rather than 'total PAH'. When PAH are designated as 'not detected', the relevant limits of detection should be given.
- PAH emissions and effluents should be reduced by:
  — filtration and scrubbing of industrial emissions,
  — treatment of effluents, and
  — use of catalytic converters and particle traps on motor vehicles.

11.2 Protection of human health

- Owing to their proven immunotoxic effects, coal-tar shampoos should be used for anti-dandruff therapy only if no other treatment is available.
- In view of the proven immunotoxic and carcinogenic effects of PAH in coke-oven workers, exposure to PAH in occupational settings should be eliminated or minimized by reducing emissions to the extent possible or, when they cannot be sufficiently reduced, by providing effective personal protection.
- Public education about the sources and health effects of exposure to PAH should be improved.
- Use of unvented indoor fires, as in many developing countries, should be discouraged, and they should be replaced by more efficient, well-vented combustion devices.
- The risk of exposure to PAH from passive smoking should be stressed and measures taken to avoid it.
- Urban air pollution should be monitored all year round and not only seasonally.

11.3 Recommendations for further research

11.3.1 General

— Investigate the suitability of benzo[*a*]pyrene as an indicator of the effects of PAH on human health and the environment and examine the use of other PAH as surrogates.

11.3.2 Protection of human health

— More data should be collected on the human body burden of PAH and on biomarkers for these compounds.
— The reproductive effects of PAH should be studied further.
— More studies on dermal absorption are required.
— The contribution of the high-molecular-mass PAH to the overall carcinogenic potential of PAH should be studied.

11.3.3 Environmental protection

— The toxic effects of PAH to plants and earthworms should be studied.
— The body burdens and possible toxic effects of PAH in wild mammals and birds should be investigated, as most of these species can metabolize PAH.
— The extent to which higher-molecular-mass PAH are absorbed by sediments and serve as a sink and the effects of disturbing sediment, e.g. by dredging, on aquatic organisms should be investigated.
— The environmental significance of the tumours that have been found in fish exposed to PAH must be addressed.
— Reliable data should be collected on environmentally relevant PAH like dibenzo[*a*,*l*]pyrene, particularly with regard to noncarcinogenic end-points such as effects on the immune system.

11.3.4 Risk assessment

— Quantitative estimates of the risks presented by PAH should be compared using various approaches and exposure scenarios, both for human health and ecological protection.
— The risk estimates obtained by the various approaches to risk assessment based on data on human exposure should be compared.
— Risk assessment procedures that allow integrated assessment of the risks due to inhalation and to oral and dermal exposure should be developed and validated.

- Comparative risk assessment methods for evaluating immunotoxicity and other noncancer risks associated with exposure to PAH should be improved.
- The use of the results of alternative tests, such as those for genotoxicity and other short-term effects, in assessing risks due to PAH should be evaluated.
- Human exposure to alkylated PAH in a variety of situations should be investigated further and data acquired on the mutagenicity and experimental carcinogenicity of these compounds.

12. PREVIOUS EVALUATIONS BY INTERNATIONAL BODIES

12.1 International Agency for Research on Cancer

Polycyclic aromatic hydrocarbons (PAH) have been evaluated by a number of working groups convened by IARC. The evaluations made between 1973 and 1983 and summarized in Supplement 7 to the *IARC Monographs* (IARC, 1987) are shown in Table 109.

12.2 WHO Water Quality Guidelines

PAH have been assessed in the *WHO Guidelines for Drinking-water Quality* (WHO, 1984, 1996). A quantitative risk assessment was conducted using the two-stage birth-death mutation model. The resulting guidelines for benzo[*a*]pyrene in drinking-water corresponding to excess lifetime risks for gastric cancer of 10^{-4}, 10^{-5}, and 10^{-6} are 7, 0.7, and 0.07 µg/litre.

The data are insufficient to derive guidelines for other PAH, but the following recommendations are made for the group:

- Because of the close association between PAH and suspended solids, treatment to achieve the recommended level of turbidity, when necessary, will ensure that the PAH levels are reduced to a minimum.
- Water should not be contaminated with PAH during water treatment or distribution. The use of coal-tar-based and similar materials for pipe linings and coatings on storage tanks should therefore be discontinued. It is recognized that it may be impracticable to remove coal-tar linings from existing pipes, and research is needed on methods of minimizing the leaching of PAH from such materials.
- In monitoring levels of PAH, the use of specific compounds as indicators for the group as a whole is recommended. The choice of indicator compound will vary in each situation. PAH should be monitored regularly in order to determine background levels, against which any changes can be assessed and remedial action taken, if necessary.
- When drinking-water is known to have been contaminated by PAH, the specific compounds present and the source of the contamination should be identified, as the carcinogenic potential of PAH varies.

12.3 FAO/WHO Joint Expert Committee on Food Additives

Benzo[*a*]pyrene was assessed at a meeting of this Committee in June 1990 (WHO, 1991). The Committee concluded that, for the purpose of evaluation, the most significant toxicological effect of benzo[*a*]pyrene is its carcinogenicity. It was recognized that benzo[*a*]pyrene is only one member of a class of more than 100 compounds and that they should be considered as a class.

Table 109. Degree of evidence for carcinogenicity in humans and in experimental animals and overall evaluations of carcinogenicity to humans for agents evaluated in *IARC Monographs*

Compound	Degree of evidence for carcinogenicity		Overall evaluation[a]	*IARC Monographs* volume (year)
	Human	Animal		
Anthanthrene	ND	L	3	32 (1983)
Anthracene	ND	I	3	32 (1983)
Benz[*a*]anthracene	ND	S	2A	3 (1973); 32 (1983)
Benzo[*b*]fluoranthene	ND	S	2B	3 (1973); 32 (1983)
Benzo[*j*]fluoranthene	ND	S	2B	3 (1973); 32 (1983)
Benzo[*k*]fluoranthene	ND	S	2B	32 (1983)
Benzo[*ghi*]fluoranthene	ND	I	3	32 (1983)
Benzo[*a*]fluorene	ND	I	3	32 (1983)
Benzo[*b*]fluorene	ND	I	3	32 (1983)
Benzo[*ghi*]perylene	ND	I	3	32 (1983)
Benzo[*c*]phenanthrene	ND	I	3	32 (1983)
Benzo[*a*]pyrene	ND	S	2A	3 (1973); 32 (1983)
Benzo[*e*]pyrene	ND	I	3	3 (1973); 32 (1983)
Chrysene	ND	L	3	3 (1973); 32 (1983)
Coronene	ND	I	3	32 (1983)
Cyclopenta[cd]pyrene	ND	L	3	32 (1983)
Dibenz[*a*,*h*]anthracene	ND	S	2A	3 (1973); 32 (1983)
Dibenzo[*a*,*e*]pyrene	ND	S	2B	3 (1973); 32 (1983)
Dibenzo[*a*,*h*]pyrene	ND	S	2B	3 (1973); 32 (1983)
Dibenzo[*a*,*i*]pyrene	ND	S	2B	3 (1973); 32 (1983)
Dibenzo[*a*,*l*]pyrene	ND	S	2B	3 (1973); 32 (1983)
Fluoranthene	ND	I	3	32 (1983)
Fluorene	ND	I	3	32 (1983)
Indeno[1,2,3-*cd*]pyrene	ND	S	2B	3 (1973); 32 (1983)
5-Methylchrysene	ND	S	2B	32 (1983)
1-Methylphenanthrene	ND	I	3	32 (1983)
Perylene	ND	I	3	32 (1983)
Phenanthrene	ND	I	3	32 (1983)
Pyrene	ND	I	3	32 (1983)
Triphenylene	ND	I	3	32 (1983)

Adapted from IARC (1987)

ND, no adequate data; I, inadequate evidence; L, limited evidence; S, sufficient evidence

[a] Group 1, the compound is carcinogenic to human; Group 2A, the compound is probably carcinogenic to humans; Group 2B, the compound is possibly carcinogenic to humans; Group 3, the compound is not classifiable as to its carcinogenicity to humans

The Committee was unable to establish a tolerable intake for benzo[*a*]pyrene on the basis of the available data. Nevertheless, the large difference between the estimated human intake of benzo[*a*]pyrene and the doses that induce tumours in animals suggests that any effects on human health are likely to be small. Despite this, the considerable uncertainties in risk estimation require that efforts be made to minimize human exposure to benzo[*a*]pyrene as far as is practicable.

The Committee acknowledged the complexity of reducing exposure to benzo[*a*]pyrene and other PAH. Furthermore, it noted that exposure to benzo[*a*]pyrene constitutes only a fraction of consumers' exposure to these compounds and that some other members of this class, not evaluated at the meeting, have toxicological properties similar to those of benzo[*a*]pyrene and may thus contribute to the overall carcinogenic risk. In this regard, strategies to minimize exposure to benzo[*a*]pyrene would also be effective in reducing overall exposure to PAH. These include practices that consumers can effect, such as washing fruits and vegetables thoroughly to remove any surface contamination, trimming excess fat prior to barbecuing meats to minimize 'flare-ups', and cooking in a fashion that prevents contact of food with flames. Measures that can be taken by the food industry include use of indirect heating for drying foods, switching to non-coal-fired roasters (e.g. for roasting coffee beans), using protective coverings (e.g. cellulose casing) when smoking foods conventionally, and ensuring compliance with the limits for PAH in food additives specified by national and international bodies. The Committee urged application of these measures in order to minimize contamination of food with PAH, including benzo[*a*]pyrene.

12.4 WHO Regional Office for Europe Air Quality Guidelines

PAH have been assessed as atmospheric pollutants by the WHO Regional Office for Europe (WHO, 1987). No guideline was set, but the group concluded that no safe level of PAH could be recommended, owing to their carcinogenicity. There is no known threshold for the induction of cancer by benzo[*a*]pyrene, the most thoroughly studied PAH, nor is there an ambient mixture of PAH that does not contain benzo[*a*]pyrene and other substances for which there is sufficient evidence of carcinogenicity in animals.

A number of estimates have been made of the risk presented by PAH, based primarily on studies in which benzo[*a*]pyrene was used as the index compound. The US Environmental Protection Agency (1984d) proposed an upper-bound lifetime cancer risk of 62 per 100 000 exposed people per microgram of benzene-soluble coke-oven emission per cubic metre of ambient air. Assuming a 0.71% content of benzo[*a*]pyrene in these emissions, it can be estimated that nine out of 100 000 people exposed to 1 ng/m^3 benzo[*a*]pyrene over a lifetime would be at risk of developing cancer.

APPENDIX I

SOME APPROACHES TO RISK ASSESSMENT FOR POLYCYCLIC AROMATIC HYDROCARBONS

I.1 Introduction

Various environmental and technical problems hinder assessment of the risk posed by mixtures containing polycyclic aromatic hydrocarbons (PAH), particularly for carcinogenicity. Ambient environments contain a wide range of PAH, some of which are highly carcinogenic, while others are probably not (Table A1.1). Both anthropogenic and natural sources may contribute to the ambient levels of these compounds, such as in the air over industrial towns (see Section 3).

As PAH undergo transformation in the environment (see Section 4), it cannot be taken for granted that the composition of ambient mixtures is similar to that of the source. No data are available to assess the potency of individual PAH in humans, but the carcinogenicity of several mixtures containing PAH has been estimated in epidemiological studies (Albert et al., 1983). Mixtures of a similar type, such as from coke ovens, may not always present the same risk, however, even when the fuel used and the operating conditions are similar. Furthermore, PAH are often not the only contributors to the carcinogenic risk presented by a given mixture: other chemicals and particulate matter (Heinrich, 1995) may also contribute. The basis for quantification and the way in which the quantity of a mixture is expressed are also important. Should the levels be expressed in terms of the total mass of extractable material or as a surrogate? If the concentration of a chosen surrogate is used as an indicator of the quantity of the mixture and its toxicity, the surrogate selected must be predictive of the toxicity of the mixture.

Estimating the risk conferred by exposure to PAH has further problems. As humans are exposed to mixtures of PAH and other compounds and not to pure PAH, experimental data must be used to estimate the risk for exposure to individual PAH and the result extrapolated to the low doses to which humans are exposed. Such extrapolation is problematic, because species may differ in the enzymes that activate PAH (Michel et al., 1995) and in their susceptibility to the tumorigenic effects of PAH; these differences may otherwise be a simple reflection of differences in weight, surface area, basal metabolic rate, or respiratory volume. The degree to which species differences affect extrapolated human risks is unknown.

Another set of problems stems from the large number of PAH that are typically found in a complex mixture. Only a small percentage of the environmentally relevant PAH has been investigated for carcinogenicity in

experimental animals, and the toxicity of most of the PAH in complex mixtures remains unknown. About a dozen PAH of known toxicity are monitored in most programmes, and these contribute only a small proportion of the risk represented by PAH fractions extracted from complex mixtures (Thorslund & Farrar, 1990a). It is therefore likely that summing the risks posed by individual PAH of known toxicity does not accurately reflect the contribution of all the PAH in a mixture.

Table A1.1. Evaluations of the carcinogenicity of some polycyclic aromatic hydrocarbons

Compound	IARC (1987)[a]	US Environmental Protection Agency (1993)[a]	Task Group[b]
Acenaphthene		D	Questionable
Acenaphthylene		D[c]	
Anthanthrene			Positive
Anthracene	3	D[c]	Negative
Benz[*a*]anthracene	2A	B2[c]	Positive
Benzo[*b*]fluoranthene	2B	B2[c]	Positive
Benzo[*j*]fluoranthene	2B	B2	Positive
Benzo[*ghi*]fluoranthene			(Negative)
Benzo[*k*]fluoranthene	2B	B2[c]	Positive
Benzo[*a*]fluorene			Questionable
Benzo[*b*]fluorene			Questionable
Benzo[*ghi*]perylene	3	D[c]	Negative
Benzo[*c*]phenanthrene			(Positive)
Benzo[*a*]pyrene	2A	B2[c]	Positive
Benzo[*e*]pyrene	3	C	Questionable
Chrysene	3	B2[c]	Positive
Coronene			Questionable
Cyclopenta[*cd*]pyrene		B2	Positive
Dibenz[*a,h*]anthracene		B2[c]	Positive
Dibenzo[*a,e*]pyrene		B2	Positive
Dibenzo[*a,h*]pyrene		B2	Positive
Dibenzo[*a,i*]pyrene		B2	Positive
Dibenzo[*a,l*]pyrene		B2	Positive
Dibenzo[*e,l*]pyrene		D	
Dibenzo[*a,e*]fluoranthene		B2	
Dibenzo[*a,h*]fluoranthene		B2	
Dibenzo[*a,i*]fluoranthene		B2	
Dibenzo[*a,l*]fluoranthene		B2	
Fluoranthene	3	D[c]	(Positive)
Fluorene	3	D[c]	Negative

Table A1.1 (contd)

Compound	IARC (1987)[a]	US Environmental Protection Agency (1993)[a]	Task Group[b]
ndeno[1,2,3-*cd*]pyrene		B2[c]	Positive
5-Methylchrysene			Positive
1-Methylphenanthrene			(Negative)
Naphthalene		D[c]/C	Questionable
Perylene			(Negative)
Phenanthrene	3	D[c]	Questionable
Pyrene		D[c]	Questionable
Triphenylene			(Negative)

[a] Based on the results of studies in humans and experimental animals
[b] Based on the results of studies in experimental animals
[c] Consensus position of the US Environmental Protection Agency; others have been presented at scientific meetigs (Schoeny et al., 1994; McClure & Schoeny, 1995).

I.2 Approaches to risk assessment

Three of the most popular approaches for assessing dose–response relationships for PAH are presented, with their strengths and weaknesses. These approaches are toxicity equivalence factors, comparative potency, and use of benzo[*a*]pyrene as a surrogate.

I.2.1 Toxicity equivalence factors and related approaches

Several approaches to quantification may be considered for assessing the risks posed by mixtures of agents. When there are sufficient data, they can be used as a basis for quantitative estimates of risk. Relatively few mixtures containing PAH have been tested under conditions that are acceptable for risk assessment. For some processes, e.g. coal coking, the data on human exposure are sufficiently complete to allow quantitative estimates of risk; however, changes in the parameters of the combustion process, such as temperature and amount of oxygen feedstock, may result in variations in the types, amounts, and physical status of PAH in the mixture. These variations may be sufficient to alter the risk posed by the mixture. One way of resolving the uncertainty inherent in differences in the composition of mixtures is to base quantitative estimates on considerations of individual components; this alternative is explored below.

I.2.1.1 *Principle*

The approaches are based on an assumption of additive risk, which leads, in principle, to an estimate of the risk associated with identified PAH. In practice, the risks attributable to individual PAH are summed, or the risk posed by individual PAH is expressed relative to that for benzo[*a*]pyrene, and then the levels of these equivalents are summed. The latter process is the toxicity equivalence factor approach.

The first step is to estimate the potency of a single PAH, which serves as a standard against which the potency of other compounds is later derived. In practice, this compound is usually benzo[*a*]pyrene. Since no data from studies in humans are available that are suitable for assessing the potency of individual PAH, the potency of benzo[*a*]pyrene in humans is estimated from the results of studies in animal models. The uncertainty associated with this extrapolation is discussed above.

The second step is to estimate the potency of the PAH relative to that of benzo[*a*]pyrene, in order to obtain a benzo[*a*]pyrene equivalent. The estimate is based on the relative potencies of benzo[*a*]pyrene and other PAH in experimental animals. The key assumption is that the relative potency of two PAH in an animal model is the same or similar to that of the same compounds in humans. This comparative potency approach has been used in relation to chlorinated dibenzo-*para*-dioxins and dibenzofurans (US Environmental Protection Agency, 1987) and for PAH and PAH-rich mixtures (Albert et al., 1983; Clement Associates, 1988). Further evidence supports the assumption made in this approach (Albert et al., 1983; Lewtas, 1985a,b; Nesnow, 1990).

The third step involves summation of risks, which can be done either by summing the benzo[*a*]pyrene equivalents and multiplying by the potency of benzo[*a*]pyrene or by estimating the potency of each PAH in humans (risk for cancer) and then adding them. The underlying assumption is that the individual estimates of risk are additive. Although there may be interactions between PAH, the risks appear to be approximately additive, especially at low, environmentally relevant doses (Krewski et al., 1989; Nesnow et al., 1995). In the absence of information to the contrary, additivity is assumed.

The toxicity equivalence factor approach is feasible if at least two pieces of information are available:

— the amount of each PAH in the mixture, or the amount of each 'major bioactive PAH', and
— a quantitative estimate of the risk associated with each identified PAH.

For the first requirement, good data on the composition of mixtures of PAH can be generated with existing analytical techniques (see Section 2), although such analyses can be resource-intensive. Quantitative risk estimates

for individual PAH are generally not available, as the data are insufficient. The deficiencies include the following:

— the data relate to exposures that are not typically used in deriving quantitative estimates of risk after oral or inhalation exposure,
— the study populations were inappropriately small,
— the studies involved only one dose,
— dose–response relationships were not reported, and/or
— different PAH were tested in different studies with different designs.

Environmental mixtures contain many more PAH than can be monitored feasibly or practically. Furthermore, the toxicity of most environmentally relevant PAH has not yet been quantified. As the toxicity equivalence factor approach assumes additivity of the risks posed by PAH in a mixture, the question remains of the extent to which the risk of one PAH is representative of the risk of all of those in the mixture.

1.2.1.2 Development and validation

1.2.1.2.1 Derivation of the potency of benzo[a]pyrene

Quantitative risk estimation is problematic for even the best-studied PAH, namely benzo[*a*]pyrene. The risk for carcinogenicity is best estimated on the basis of data from long-term, usually lifetime, assays in which several doses are tested in large groups of animals. The data on benzo[*a*]pyrene are less than optimal: few lifetime assays have been conducted with exposure other than dermal and generally few doses were tested.

Krewski et al. (1989) calculated the probability of occurrence of a tumour at a specified time after continuous exposure to a mixture of PAH. The dose of the PAH mixture was described as a benzo[*a*]pyrene equivalent dose d as follows:

$$d = \sum R_i d_i + d_a \qquad (1)$$

where R_i is the relative carcinogenicity of the i^{th} PAH in comparison with that of benzo[*a*]pyrene, d_i is the dose of the i^{th} PAH, and d_a is the dose of benzo[*a*]pyrene.

The tumour probability, $P(d)$, was calculated from a two-stage birth–death mutation model. For the mixture as a whole, $P(d)$ is given by the equation:

$$P(d) = 1\text{-}\exp(-A(1 + bd)^2) \qquad (2)$$

where the unknown values A and b were estimated from bioassays with benzo[*a*]pyrene as $A = 0.00616$ and $b = 3.52\ \mu g^{-1}$ for lifetime exposure.

The US Environmental Protection Agency (1993) made a quantitative risk estimate for oral exposure to benzo[*a*]pyrene that consisted of a range of values, from 4.5 to 11.7 per (mg/kg)/day, with a geometric mean of 7.3 per (mg/kg)/day. These estimates were obtained by using three methods of determining an upper bound on a linear low-dose term from data on the incidence of gastrointestinal tumours in mice exposed to benzo[*a*]pyrene in the diet (Neal & Rigdon, 1967). The models used were a form of the two-stage Moolgavkar, Venzon, and Knudson model (Moolgavkar & Venzon, 1979; Moolgavkar & Knudson, 1981) and a Weibull-type model (Rees & Hattis, 1994). The total numbers of tumours in male and female rats exposed in the diet (Brune et al., 1981) were used in a linearized multistage procedure to derive an upper bound on the low-dose term (slope factor), shown in Table AI.2.

The potency of benzo[*a*]pyrene in humans exposed by inhalation has been assessed on the basis of extrapolations from the results for rodents, sometimes exposed other than by respiration. Since the sensitivity to PAH is likely to differ with the route of exposure, assessments made on the basis of exposure by inhalation are preferable; some of these are shown in Table AI.3.

The commonest method of extrapolation is based on the relative surface areas or body weights of experimental animals and humans, assuming that these are good predictors of the relative potency of PAH in two species. While such scaling factors have been validated for other compounds (Chappell, 1989) and have been used in the case of PAH (Thorslund & Farrar, 1990b; Collins et al., 1991; Collins & Alexeeff, 1993), this relationship may not hold for PAH. Muller et al. (1995a,b, 1996) compared the actual and extrapolated doses required to produce a tumorigenic effect of a given magnitude in a particular rodent species, on the basis of the assumption that if extrapolations based on surface area and/or body weight hold between rodents to humans, the same extrapolation should hold even more closely for closely related rodents. The extrapolated dose was estimated from that required to induce a similar response in another rodent species under matching experimental conditions and by extrapolating to the target rodent species. Examples of the analysis are shown in Tables AI.4 and AI.5. The extrapolated and actual doses differed by much as two orders of magnitude, even though mice and rats are closely related, of similar sizes, and with similar diets, and, furthermore, laboratory rodents are placed in similar habitats. It would be expected that extrapolation from one rodent to another is more justified than extrapolation from rodents to humans, but these analyses indicate that extrapolation from rodents to humans may lead to even larger errors. Although some of the discrepancies may be due to methodological problems, much of the difference is due to the low predictive value of the two extrapolations. Extrapolations based on surface area or body weight further imply that species differences in the metabolism of the parent compound to the primary carcinogen are functions

Table A1.2. Slope factors for humans based on the results of studies in which benzo[*a*]pyrene was fed to rodents in the diet

Study	Slope factor (per [mg/kg]/day)	Comments
Neal & Rigdon (1967)	5.9	Two-stage, conditional upper bound
	9.0	Two-stage, slope from 10% response
	4.5	Weibull-type model
Brune et al. (1981)	11.7	Linearized multistage procedure applied to oesophageal, laryngeal, and forestomach tumours in males and females

From US Environmental Protection Agency (1992b, 1993)

of body weight or surface area. This assumption is not supported by the available pharmacokinetic data for PAH (Michel et al., 1995).

Some risk assessments are based on extrapolation of the results of studies in which the compound was deposited directly onto or in close proximity to the tissues where tumours were later observed. Since the dose delivered to these tissues is direct and not a reflection of body weight or surface, it may well be argued that extrapolations based on body weight or surface area are not appropriate.

It is likely, therefore, that the estimates of the potency of PAH in humans based on such extrapolations would lead to substantial errors. It is therefore preferable to use other approaches, like the relative potency approach, in order to estimate human risk.

1.2.1.2.2 Derivation of relative potencies of PAH other than benzo[a]pyrene

Some quantitative risk estimates for mixtures of PAH are based on the assumption that all PAH (or all carcinogenic PAH) have the same potency as benzo[*a*]pyrene and that the carcinogenic effect of the mixture can be estimated by summing the effects of each PAH. Some PAH are less carcinogenic in animal models than benzo[*a*]pyrene, and a few are more active. In order to provide more reasonable estimates of the carcinogenicity of PAH mixtures, schemes have been devised that are similar to the toxicity equivalence factor approaches of the US Environmental Protection Agency and NATO for evaluating chlorinated dibenzodioxins and dibenzofurans, which are based on a general or specific hypothesis of relative potency.

In a provisional quantitative risk assessment of PAH, the US Environmental Protection Agency (1993) used benzo[*a*]pyrene as a standard and derived the relative potencies of individual PAH in increments of order of magnitude by comparison. Only the results of carcinogenicity bioassays were considered,

Table AI.3. Estimated carcinogenic potency of benzo[*a*]pyrene in humans exposed by inhalation, on the basis of extrapolations from the results of studies in experimental animals

Risk (ng/m^3)	Species	Route of exposure	Assumptions	Data source	UCL/ MLE	Assessor
1.7×10^{-6a}	Hamster	Inhalation on salt particulate	Risk proportional to (body weight)$^{2/3}$ and inhalation rate = 0.037 m^3/d	Thyssen et al. (1981)	UCL	US Environmental Protection Agency (1984a); Collins & Alexeeff (1993)
1.1×10^{-5}	Hamster	Inhalation on salt particulate	Risk proportional to (body weight)$^{2/3}$ and inhalation rate = 0.063 m^3/d	Thyseen et al. (1981)	UCL	Collins & Alexeeff (1993)
4.7×10^{-6}	Hamster	Intratracheal instillation with ferric oxide	Risk proportional to (body weight)$^{2/3}$	Saffiotti et al. (1972)	UCL	Collins & Alexeeff (1993)
4.4×10^{-6}	Hamster	Intratracheal instillation with ferric oxide	Risk proportional to (body weight)$^{2/3}$	Feron et al. (1973)	UCL	Collins & Alexeeff (1993)
2.0×10^{-6}	Rat	Inhalation of coal-tar/pitch condensation aerosol[b]	Risk equal to that of humans and hamsters at same air level of benzo[*a*]pyrene, corrected for rat lifetime (2 years)	Heinrich et al. (1994c)	UCL	Heinrich et al. (1994c)

Table AI.3 (contd)

Risk (ng/m^3)	Species	Route of exposure	Assumptions	Data source	UCL/ MLE	Assessor
7.0×10^{-9}	Hamster		Risk proprotional to body weight	Thyssen et al. (1981)	MLE	Clement Associates (1990)
3.6×10^{-8}	Hamster		Risk proprotional to body weight$^{3/4}$	Thyssen et al. (1981)	MLE	Clement Associates (1990)
6.2×10^{-8}	Hamster		Risk proprotional to body weight$^{2/3}$	Thyssen et al. (1981)	MLE	Clement Associates (1990)
3.9×10^{-9}	Hamster		Risk equal to that of humans and hamsters at same air level of benzo[*a*]pyrene	Thyssen et al. (1981)	MLE	Clement Associates (1990)

UCL, upper confidence limit; MLE, maximum likelihood estimate

[a] Recalculated from 6.11 mg/kg bw per day using the assumptions of the US Environmental Protection Agency for humans: 70 kg, 20 m^3/d

[b] Other active polycyclic aromatic hydrocarbons present in the administered aerosol

Table AI.4. Comparison of actual and extrapolated doses of 3-methylcholanthrene and benzo[*a*]pyrene required to obtain observed tumour incidence; topical administration

Species	Compound	Actual dose (mg)		Extrapolated dose (mg)		Extrapolated/ actal dose	Reference
		Surface area	Body weight	Surface area	Body weight		
Rat	3-Methylcholanthrene	58	9.0	20	0.16	0.35	Cavalieri et al. (1978)[a]; Zackheim (1964)
Hamster	3-Methylcholanthrene	5.0[b]	0.048	0.074	0.0096	0.015	Cavalieri et al. (1978)[a]; Bernfeld & Homburger (1983)
Hamster	Benzo[*a*]pyrene	5.0[b]	0.019	0.028	0.0038	0.056	Cavalieri et al. (1978)[a]; Bernfeld & Homburger (1983)

[a] Data for mouse

[b] No tumours induced at dose tested; assumed 1% response rate

Table AI.5. Comparison of actual and extrapolated doses of 3-methylcholanthrene and benzo[*a*]pyrene required to obtain observed tumour incidence; respiratory exposure

Species	Compound	Actual dose (mg)		Extrapolated dose (mg)		Extrapolated/ actual dose	Reference
		Surface area	Body weight	Surface area	Body weight		
Rat	3-Methylcholanthrene	0.50[a]	7.0[a]	15[a]	14[a]	30[a]	Nettesheim & Hammons (1971)[b]; Hirano et al. (1974)
Hamster	3-Methylcholanthrene	0.7[a]	4.7[a]	7.1	6.7[a]	10[a]	Nettesheim & Hammons (1971)[b]; Hammond et al. (1987)
Rat	Benzo[*a*]pyrene	15	31	33	2.1	2.2	Furst et al. (1979)[b]; Ishinishi et al. (1976)
Hamster	Benzo[*a*]pyrene	30	33	222	1.1	0.73	Furst et al. (1979)[b]; Saffiotti et al. (1972)

[a] Nettesheim & Hammons (1971) used intratracheal instillation, while Hirano et al. (1974) used intrapulmonary pellets to administer 3-methylcholanthrene to rats. Hammond et al. (1987) applied intrabronchial pellets to hamsters. The fact that instillation is considered to be less effective in inducing tumours than implantation is considered to be the likely explanation for the apparent discrepancy between the actual and the extrapolated doses.

[b] Data for mouse

and these were limited to those in which benzo[*a*]pyrene and other PAH were assayed by the same protocol and within the same time frame. The studies involved various routes of exposure, including skin painting, intraperitoneal and subcutaneous injection, and lung implantation (see section 7.7.2). Maximum likelihood estimates from a two-stage model were used for comparison, and ranges of estimates were presented. These values for the results of complete carcinogenesis assays in mouse skin are shown in Table AI.6. The US Environmental Protection Agency (1993) considered that the data on PAH did not meet all of the requirements for application of the toxicity equivalence factor approach and recommended that the values be applied only to carcinogenicity and not to other end-points. In order to differentiate between these values and a toxicity equivalence factor meant for use in evaluating all types of toxicity, the relative potencies were designated 'estimated orders of potential potency'. The US Environmental Protection Agency (1993) further recommended that these values not be used for evaluating inhaled PAH mixtures, for the following reasons:

— The US Environmental Protection Agency currently has no consensus value for an inhalation unit risk.
— There is no basis for assuming that the relative order of potency for PAH is the same after oral and inhalation exposure.
— The co-carcinogenic potential of particulate carriers in the lung has not been sufficiently elucidated.

Improvements to the estimated orders of potential potency have been published (McClure & Schoeny, 1995). As the two-stage model requires estimation of several parameters in the absence of data, other models were investigated. It was shown that the model used had little effect on the values when applied consistently; however, the assay type and data set used could alter the values by orders of magnitude. It was proposed, therefore, that data from all available appropriate assays should be modelled and that a central tendency estimate, rounded to powers of 10, would give a more realistic value. The list of PAH for which estimated orders of potential potency were estimated was expanded from that of US Environmental Protection Agency (1993) to include several more that could be considered probable human carcinogens (Table AI.7).

In a preliminary validation exercise based on published data on PAH-containing mixtures, McClure & Schoeny (1995) used the values in Table AI.7 to estimate the carcinogenicity of two synthetic mixtures of PAH (Pfeiffer, 1973), both as a sum of their components and as whole mixtures. A good correlation was found between the two measurements of potency. These results indicate that the components of a defined mixture have additive risks.

The additivity of the potency of mixtures of five PAH was investigated in the mouse lung adenoma model (Nesnow et al., 1995). Different combinations of PAH with different biological and chemical properties were tested. Some interaction was found in which the potency of the compounds was more or less

Table AI.6. Estimated orders of potency of selected polycyclic aromatic hydrocarbons in mouse skin carcinogenesis

Compound	Relative potency[a]		Reference
Benzo[*a*]pyrene	1.0	1.0	
Benz[*a*]anthracene	0.145	0.1	Bingham & Falk (1969)
Benzo[*b*]fluoranthene	0.167	0.1	Habs et al. (1980)
Benzo[*k*]fluoranthene	0.020	0.01	Habs et al. (1980)
Chrysene	0.004	0.001	Wynder & Hoffmann (1959)
Dibenz[*a,h*]anthracene	1.11	1.0	Wynder & Hoffmann (1959)
Indeno[1,2,3-*cd*]pyrene	0.055[b]	0.1	Habs et al. (1980); Hoffmann & Wynder (1966)

[a] Model was $P(d) = 1-\exp[-A(1 + bd)^2]$ for all except indeno[1,2,3-*cd*]-pyrene. Actual figures (left) and rounded to order of 10 (right)

[b] Simple mean of relative potencies (0.021 and 0.089), the latter being derived with the one-hit model

Table AI.7. Estimated order of carcinogenic potency for 13 polycyclic aromatic hydrocarbons in Group B2 as compared with benzo[*a*]pyrene

Compound	Estimated order of potency	No. of estimates
Benzo[*a*]pyrene	1.0	
Benz[*a*]anthracene	0.1	4
Benzo[*b*]fluoranthene	0.1	8
Benzo[*j*]fluoranthene	0.1	7
Benzo[*k*]fluoranthene	0.1	7
Chrysene	0.1	5
Cyclopenta[*cd*]pyrene	0.1	4
Dibenz[*a,h*]anthracene	1.0	3
Dibenzo[*a,e*]fluoranthene[a]	1.0	3
Dibenzo[*a,e*]pyrene	1.0	3
Dibenzo[*a,h*]pyrene	1.0	2
Dibenzo[*a,i*]pyrene	0.1	3
Dibenzo[*a,l*]pyrene	100	2
Indeno[1,2,3-*cd*]pyrene	0.1	4

Adapted from McClure & Schoeny (1995)

[a] Not evaluated by the Task Group

than additive. The differences were no more than twofold and, therefore, would not be large enough to alter significantly the outcome of risk assessments, in which uncertainty of an order of magnitude is not seen as excessive.

Complex environmental mixtures differ from defined synthetic mixtures in that they contain not only PAH of known carcinogenicity but also hundreds of PAH and other potentially carcinogenic non-PAH compounds for which carcinogenicity has not been established. The risk attributed to a PAH for which data on exposure and carcinogenic potency exist would be similar to that for the entire mixture only if the PAH for which the potency is unknown contributed little or nothing to the potency of the mixture as a whole. McClure & Schoeny (1995) reported that the carcinogenic activity of a coal liquefaction material (Mahlum et al., 1984) was similar to that estimated by adding benzo[*a*]pyrene equivalents derived from estimated orders of potential potency for several measured PAH. Similar estimates for coal flue gas, petrol-engine exhaust, and diesel exhaust, however, resulted in underestimates of two to three orders of magnitude of the risks presented by the PAH fractions when the results of studies by either lung implantation or dermal application in rodents were used for the calculation (Grimmer et al., 1984; Thorslund & Farrar, 1990a; see Table AI.8).

Other strategies for risk assessment based on the toxicity equivalence factor approach for individual PAH have been published. The resulting estimates are compared in Table AI.9. Krewski et al. (1989) analysed the published values and derived a new set based largely on estimates from a two-stage model (Clement Associates, 1988). Applications of their toxicity equivalence factors to the results of bioassays wth PAH mixtures (Pfeiffer, 1977; Schmähl et al., 1977) indicated that their values would be unlikely to underestimate the carcinogenic risk posed by whole mixtures.

A set of toxicity equivalence factors was derived for the 17 PAH commonly measured at hazardous waste sites, and a new list (see column 2 of Table AI.9) was calculated on the basis of older work and the primary literature (Nisbet & LaGoy, 1992). The values tend to overestimate the carcinogenic risks of mixtures.

The relative potency values in 14 publications were used to classify PAH into categories of high, medium, low, and very low risk (column 3 of Table AI.9), and 'environmental assessment levels' were calculated on the basis of the highest potency for a PAH, rounded to an order of magnitude, relative to that of benzo[*a*]pyrene (Malcom & Dobson, 1994).

Kalberlah et al. (1995) and others adopted the approach of the US Environmental Protection Agency Office of Pesticides, Pollution Prevention and Toxic Substances to determine the relative potencies of PAH. A panel of five experts made independent reviews of the existing data on about 150 PAH and scored them as having high, moderate, marginal, or slight potential carcinogenicity. The panel considered data from studies of skin painting in mice, the induction of lung and liver adenomas in newborn mice, mammary

Table AI.8. Comparison of the carcinogenic potency of the polycyclic aromatic hydrocarbon (PAH) fraction of PAH-rich mixtures with the integrated potency of the eight PAH found in the fraction

Method of estimating risk	Potency relative to that of benzo[*a*]pyrene		
	Flue gas	Diesel-engine exhaust	Petrol-engine exhaust
Fraction of mixture containing PAH with ≥ three rings	0.38	0.28	0.67
Sum of risk for eight PAH	0.0011	0.0018	0.0007
Fraction/sum	340	150	950

From Thorslund & Farrar (1990a)

tumours in rats, studies by oral administration, studies of genotoxicity, and structure–activity relationships. Their evaluation, converted to powers of 10 to represent levels of concern, is presented in column 4 of Table AI.9.

Columns 5 and 6 of Table AI.9 show the values of the US Environmental Protection Agency (1993) and McClure & Schoeny (1995), discussed previously. The results of the six toxicity equivalence factor approaches show a reasonable degree of agreement for PAH that are generally considered to be carcinogenic. In all of them, dibenz[*a*,*h*]anthracene appears to be equipotent or somewhat more potent than benzo[*a*]pyrene; and, in most, the benzofluoranthenes and benzanthracene were about 10% as potent as benzo[*a*]pyrene. The greatest variation in the estimated relative potency is observed for chrysene, although all agree that chrysene is not as potent a carcinogen as benzo[*a*]pyrene.

I.2.1.3 Application

Application of the toxicity equivalence factor approach to assessing the risk posed by dibenzodioxins and dibenzofurans involves the following steps:

(1) analytical determination of the agents in the environmental sample;
(2) multiplication of the concentrations of congeners in the sample by the toxicity equivalence factors to express the concentration in terms of the standard agent (e.g. benzo[*a*]pyrene) equivalents;
(3) summation of the products in step (2) to obtain the equivalents of the standard agent in the sample;
(4) determination of human exposure to the mixture in question, expressed in terms of standard chemical equivalents; and
(5) combination of the exposure derived in step 4 with information on the toxicity (here, carcinogenic potency) of the standard chemical in order to estimate the risks associated with exposure to the mixture.

Table AI.9. Relative potencies of indicator polycyclic aromatic hydrocarbons

Compound	[1]	[2]	[3]	[4]	[5]	[6]	[7]
1-Methylphenanthrene			0.001				
Acenaphthene		0.001	0.001	0.001	0		
Acenaphthylene		0.001	0.001	0.01			
Anthanthrene	0.320						0.28
Anthracene		0.01	0.01	0.01			
Benz[*a*]anthracene	0.145	0.1	0.1	0.1	0.1	0.1	0.014
Benzo[*a*]pyrene	1.0	1.0	1.0	1.0	1.0	1.0	1.0
Benzo[*b*]fluoranthene	0.141	0.1	0.1	0.1	0.1	0.1	0.11
Benzo[*e*]pyrene	0.004		0.01				0
Benzo[*ghi*]perylene	0.022	0.01	0.01	0.01			0.012
Benzo[*j*]fluoranthene				0.1		0.1	0.045
Benzo[*k*]fluoranthene	0.061	0.1	0.1	0.1	0.01	0.1	0.037
Chrysene	0.0044	0.01	0.01	0.01	0.001	0.1	0.026
Coronene			0.001				
Cyclopenta[*cd*]pyrene	0.023		0.1			0.1	0.012
Dibenzo[*a,e*]pyrene						1.0	
Dibenz[*a,c*]anthracene[a]			0.1				
Dibenz[*a,h*]anthracene	1.11	5	1.0	1.0	1.0	1.0	0.89
Dibenzo[*a,l*]pyrene						100	100
Dibenzo[*a,e*]fluoranthene[a]							1.0
Dibenzo[*a,h*]pyrene						1.0	1.2
Dibenzo[*a,i*]pyrene						0.1	
Fluoranthene		0.001	0.001	0.01			
Fluorene		0.001	0.001	0			
Indeno[1,2,3-*cd*]pyrene	0.232	0.1	0.1	0.1	0.1	0.1	0.067
Naphthalene		0.001	0.001				
Perylene			0.001				
Phenanthrene		0.001	0.001	0			0.00064
Pyrene	0.81	0.001	0.001	0.001			0

[1] Krewski et al. (1989); [2] Nisbet & LaGoy (1992); [3] Malcolm & Dobson (1994); [4] Kalberlah et al. (1995); [5] US Environmental Protection Agency (1993); [6] McClure & Schoeny (1995); [7] Muller et al. (1995a,b, 1996)

[a] Not evaluated by the Task Group

These steps were followed for PAH, using one or more of the toxicity equivalence factors given in Table AI.9 and benzo[*a*]pyrene as the standard.

1.2.2 Comparative potency approach

1.2.2.1 Principle

The comparative potency approach is used to estimate the potency of the PAH in mixtures without having to identify or quantify the individual compounds. The carcinogenic potency of an unknown mixture in humans is estimated from the potency of the mixture in a bioassay and from the potency of another mixture(s) in the same bioassay and in humans. It is assumed that the relationship (ratio) between the potency of a mixture in a bioassay and human cancer risk is constant for different (PAH-rich) mixtures. The relationship is expressed in equation (3):

$$\begin{aligned}&\text{Human risk carcinogen1 / Bioassay potency carcinogen1}\\&= \text{Human risk carcinogen2 / Bioassay potency carcinogen2}\\&= k\end{aligned} \qquad (3)$$

The carcinogenic risk to humans due to exposure to a mixture can readily be derived by a rearrangement of terms.

The potency in the bioassay and the risk to humans are expressed in terms of the mass of extractable organic compounds contained in the mixture. Although this method is intended to predict human risks due to PAH, in practice it estimates the risk due to all organic compounds present. This discrepancy may not be significant when estimating the carcinogenic risk of mixtures rich in polycyclic organic matter, such as coal-tar, but may be important when estimating the risk of exposure to cigarette smoke or ambient air, in which PAH do not necessarily play a major role.

1.2.2.2 Development and validation

The comparative potency approach was initially proposed as part of an approach to assessing the carcinogenic risk of PAH in diesel emissions (Albert et al., 1983; Lewtas, 1985a,b; Nesnow, 1990). For source mixtures such as coal-tar, coke-oven emissions, and diesel and petrol emissions tested both for skin tumorigenicity in mice (Nesnow et al., 1982a,b) and in short-term bioassays (Lewtas, 1985a), there was generally good agreement (Lewtas, 1985a). Furthermore, there appears to be a good correlation between the potency of mixtures in bioassays and in humans, although this correlation is based on limited epidemiological data of good quality (Nesnow, 1990; Lewtas, 1993). Thus, risk assessments of relatively high quality are currently available only for cigarette smoke, coke-oven emissions, and coal-tar. Although the concept of comparative potency has been extensively validated, some outstanding issues remain, which are discussed below.

I.2.2.3 *Key implicit and explicit assumptions*

The comparative approach assumes that several distinct sources contribute PAH to the environment at a given location in an ambient environment. For example, in an industrial city in winter, the sources of PAH may include emissions from steel manufacture, from cars, lorries, and transport from other locations, and from home heating. Each source is assumed to make a specific contribution to the overall risk for lung cancer due to PAH. The proportion of the total risk attributable to each source depends on its potency (risk per unit mass of organic compounds) and the overall contribution to the mass of organic compounds in the ambient air. The total risk can be expressed as follows:

$$\begin{aligned}\text{Total risk} = {} & (\text{unit risk source1} \times \text{mass of organic compounds source1}) + \\ & (\text{unit risk source2} \times \text{mass of organic compounds source2}) + \\ & (\text{unit risk source3} \times \text{mass of organic compounds source3}) \ldots\end{aligned} \quad (4)$$

The unit risks for some sources are listed in Table AI.10.

Implicit in this approach are the assumptions that the composition of the organic compounds emitted from each source (such as diesel engines) is constant and that the potency (unit risk) of a given source mixture is constant. If the potency and composition of the organic compounds within mixtures from the same source vary widely, the standard mixture may not accurately represent other mixtures from a similar source. There is some evidence, however, that mixtures from similar sources have substantially different compositions. Thus, the benzo[*a*]pyrene content of the emissions from four diesel engines varied over a 600-fold range (Nesnow et al., 1982a,b; see Table AI.11). Significant differences in the potency of the four mixtures were also seen. Nevertheless, the potency of the mixtures appears to correlate reasonably well with their benzo[*a*]pyrene content (see Table AI.11), so that the comparative potency approach may be viable in principle but may not be appropriate for expressing the potency of the mixtures in terms of the mass of the organic content. A possible alternative is to express the potency in terms of the level of benzo[*a*]pyrene present in the mixture. If this solution is used, the comparative potency approach becomes essentially the benzo[*a*]pyrene surrogate approach discussed in section I.2.3.

The levels of PAH in ambient air may be influenced by multiple sources, and some PAH may be transformed in the environment. In order to estimate the carcinogenic risk due to exposure to an ambient mixture, the contribution of individual sources to the ambient air levels must be estimated, because contributing sources differ in their carcinogenicity. Making reliable estimates of the contribution of individual sources to ambient air levels is still a difficult, non-routine process.

Table AI.10. Potency of some source mixtures expressed as average dose causing 50% papilloma incidence in male and female Sencar mice

Source mixture	Organic compounds (mg)
Coke-oven main	0.14
Coke-oven topside	0.36
Roofing tar	2.0
Nissan diesel emission	1.6
Volkswagen Rabbit diesel emission	Tumour incidence did not reach 50% with tested doses

From Nesnow et al. (1982a,b)

Table AI.11. Benzo[*a*]pyrene content of organic fraction extracted from diesel-engine emissions

Source	Benzo[*a*]pyrene (ng/mg organic fraction)	% Mice with tumours/ ng benzo[*a*]pyrene[a]
Nissan	1200	0.024
Oldsmobile	2.0	NS
Caterpillar	2.0	NS
Volkswagen Rabbit	26	0.11

From Nesnow et al. (1982a,b); NS, no significant response observed over the range of doses tested

[a] Calculated by Muller et al. (1995a,b, 1996)

I.2.2.4 Application

In order to use the comparative potency approach, the carcinogenicity of the major source mixtures that contribute to a given ambient environment must be established. The potency of a number of the mixtures has been estimated (see Table AI.10), and the potency of source mixtures that affect air has been expressed in terms of risk per mass of organic compounds per cubic metre. The levels of each source mixture must then be estimated for a given ambient environment. The total risk is calculated as shown in equation (4).

I.2.3 Benzo[*a*]pyrene as a surrogate for the PAH fraction of complex mixtures

I.2.3.1 Principle

The third approach assumes that the risk due to the PAH component of complex mixtures and the levels of individual PAH in the mixtures are

proportional to those of benzo[*a*]pyrene in the mixture and vary proportionately. Using this approach, the risk due to the PAH component of mixtures can be estimated as the product of the environmental levels of benzo[*a*]pyrene and the estimate of the risk attributable to mixtures per unit amount of benzo[*a*]pyrene.

In general, the approach does not predict the potency of an ambient complex mixture as a whole but merely its PAH component. There is no reason to believe that benzo[*a*]pyrene is a good indicator of chlorinated compounds such as dioxins and dibenzofurans or volatile organic compounds such as benzene and 1,3-butadiene, which may be present in some ambient complex mixtures. The contribution of non-PAH to the overall risk of exposure to complex mixtures must thus be assessed separately.

I.2.3.2 Development and validation

Benzo[*a*]pyrene was initially favoured as an indicator of all urban pollution, and in a number of assessments based primarily (or entirely) on studies of the general population exposed to ambient air benzo[*a*]pyrene was used as an index of exposure to a wider mixture of materials (Nisbet et al., 1985). There is evidence, however, that benzo[*a*]pyrene cannot serve as an indicator of the toxicity of whole mixtures. Various factors may influence the relative contents of PAH and other contaminants of ambient air. For instance, the relative proportions of benzo[*a*]pyrene and other PAH in ambient urban air has been declining over the years, while the levels of volatile organic compounds and others have been rising. In some mixtures, such as cigarette smoke condensate, PAH probably play only a minor role in overall toxicity. Pott & Heinrich (1992) showed that mixtures containing large amounts of carcinogenic compounds other than PAH, such as cigarette smoke, are much more potent at a given level of benzo[*a*]pyrene than mixtures that owe much of their carcinogenicity to PAH, such as coke-oven emissions. Nisbet et al. (1985) argued convincingly that benzo[*a*]pyrene cannot serve as a general indicator of all pollutants in the ambient air, although it may be a suitable indicator for the carcinogenic risk posed by four- to seven-ring, unsubstituted PAH in the mixture.

Muller et al. (1995a,b, 1996) examined the PAH profiles of a wide range of mixtures from many sources and found that they were generally similar (see also section I.2.3.3). Furthermore, those mixtures rich in PAH and in which PAH are likely to contribute a significant proportion of the risk of the mixture are very similar in potency expressed per unit amount of benzo[*a*]pyrene. This observation is consistent with the notion that the PAH components of these mixtures are approximately equipotent. (Establishment of the potency of mixtures is discussed in section I.2.3.4.)

These findings do not imply that all mixtures are similar. Differences in the PAH profiles of the same mixture have been analysed in order to establish

markers for mixtures from a particular source (Gordon & Bryan, 1973; Greenberg et al., 1981; Vogt et al., 1987). The levels of some substituted PAH are clearly not related to the levels of benzo[*a*]pyrene in the mixture (Albert et al., 1983). The differences in the profile of four- to seven-ring, unsubstituted PAH in various mixtures are probably too small to alter the estimated risks of the PAH component of the mixtures significantly.

I.2.3.3 PAH profiles of complex mixtures

Information on the levels of PAH in various source mixtures is provided in Section 5. The levels of PAH in environmental mixtures used in the following analysis were derived mainly from monitoring in Canada (Muller et al., 1995a,b, 1996). About 100 mixtures were classified into different types, such as diesel emissions, coke-oven emissions, ambient air particulate, soils, and sediments, to facilitate the analysis, and the profiles of the 15 PAH most commonly tested in mixtures were compared. The results are expressed as the ratios of the levels of each PAH relative to benzo[*a*]pyrene; the geometric mean, the upper and lower 95% confidence limits, and the confidence range were calculated for each PAH ratio for a mixture type. The confidence range was determined by dividing the upper confidence limit by the lower confidence limit and used as a measure of the range of the means of the relative levels a given PAH will assume about 95 times out of 100.

The PAH profiles for petrol exhaust emissions are provided as an example in Table AI.12. It can be seen that the confidence range for each PAH is less than 5.0, and many are less than 2.0, indicating that the levels of these PAH, relative to benzo[*a*]pyrene, are very stable and vary little among the sources. They also indicate that any variation among samples is probably not large enough to alter the estimated risk. The low variation also means that the level of benzo[*a*]pyrene is a good predictor of the levels of the other PAH that may be present in a given mixture from petrol engines.

The confidence ranges for all types of combustion mixture and all of the PAH considered are presented in Table AI.13. The confidence ranges were < 50 for more than 90% of all entries and ≤ 6 for 50% of all entries. Given the degree of uncertainty usually associated with risk assessment, the uncertainty presented by the variation in PAH profile is relatively small. In addition, while some compounds in a given mixture may be found at higher levels than expected, those of other compounds may be lower than expected, and there may therefore be little difference between the estimates of risk based on chemical analysis and those based on the predicted composition of a mixture. The ambient mixtures appear to be less variable than the combustion emission mixtures. For example, Table AI.14 shows that the confidence ranges for most types of ambient air are similar. The corresponding levels of PAH relative to benzo[*a*]pyrene are shown in Table AI.15.

Table AI.12. Confidence ranges for petrol engine exhaust (relative to benzo[*a*]pyrene

Compound	Mean	95% confidence		
		Lower limit	Upper limit	Interval
Anthracene	7.0	9.3	5.3	1.8
Phenanthrene	25	38	17	2.3
Fluoranthene	7.3	14	4.0	3.5
Pyrene	9.5	19	4.7	4.0
Benz[*a*]anthracene	0.81			
Perylene	0.27	0.60	0.12	4.8
Benzo[*e*]pyrene	1.1	1.4	0.79	1.8
Benzo[*ghi*]perylene	2.6	3.5	2.0	1.7
Dibenz[*a,h*]anthracene	0.072			
Coronene	2.0	2.7	1.4	1.9
Indeno[1,2,3-*cd*]pyrene	0.80	1.1	0.60	1.8
Anthanthrene	0.38	0.55	0.27	2.1
Chrysene and triphenylene	3.0	4.6	1.9	2.4
Benzofluoranthenes	1.1	1.5	0.85	1.7

From Muller (1995a,b, 1996), based on data from Hoffmann & Wynder (1962), Grimmer & Hildebrandt (1975), Grimmer & Böhnke (1978), Alsberg et al. (1985), and Hagemann et al. (1982)

Samples of ambient air were collected in 1982–86 at point sources in Hamilton, Ontario, Canada, on days when the wind was blowing from nearby steel-mill operations 50% or more of the time. Mobile sources and home heating also contributed, but the urban levels of PAH were much lower on days when the wind was not blowing from the direction of the steel mills. The average level of benzo[*a*]pyrene was about 1.8 ng/m^3. Samples of ambient air from mobile sources were collected in Toronto, Ontario, during the summer near the intersection of two busy multi-lane highways in 1988–92. The average level of benzo[*a*]pyrene was about 0.17 ng /m^3. The highway intersection is surrounded by residential areas, and the samples of ambient air associated with home heating were collected in the same location as the mobile sources by the same collection and analytical protocol, but in winter. The average level of benzo[*a*]pyrene was 0.41 ng/m^3. Home heating and mobile emisions are considered to be the main sources of PAH. In samples of ambient air collected on Wallpole Island, Ontario, a rural location with little traffic and no industrial source, the average levels of benzo[*a*]pyrene was about 0.093 ng/m^3 (T. Dann, personal communication).

Table AI.16 presents the average PAH profiles of the combustion emissions, ambient air particulates, soils, and sediments and the average profile of the four types of mixture. The confidence ranges indicate that the four mixtures had fairly similar PAH profiles.

Table AI.13. Confidence ranges for polycyclic aromatic hydrocarbons in various combustion mixtures (relative to benzo[*a*]pyrene

Compound	Combustion mixture									
	Coke ovens	Coal-tar	Coal-fired power plants	Coal stoves	Open burning and fireplaces	Wood stoves	Diesel emissions	Petrol emissions	Roofing asphalt	Paving asphalt
Anthracene	3.2	440	830			7.6		1.8		
Phenanthrene	1.3	32	27			2.7		2.3		
Fluoranthene	2.7	4.5	3.2	43	4.9	1.4	6.1	3.5	5.7	5.2
Pyrene	2.6	280	37	18	5.6	1.5	5.3	4.0	19	2.7
Benz[*a*]anthracene	19		2.8	32	160	2.0	9.0		130	8.5
Perylene	2.3	27		23	8.2	31		4.8	4.6	5.9
Benzo[*e*]pyrene	1.3	5.5	3.3	35	1.6	1.5	2.2	1.8		
Benzo[*ghi*]perylene	8.8			12	32	2.7	8.9	1.7	5.1	2.8
Dibenz[*a,h*]anthracene				310	3.2		2.4			
Coronene	7.8			7.1	730		3.2	1.9		
Indeno[1,2,3-*cd*]pyrene				7.4	1.9	7.0	1.8			
Anthanthrene	8.7							2.1		
Chrysene and triphenylene	2.1	12	430	42	25	4.3	7.3	2.4	23	6.1
Benzofluoranthenes	5.7		2.0	28	7.1	5.0	3.4	1.7	7.7	120
Incidence of confidence range > 50	0	2	2	1	2	0	0	0	1	1

Table AI.14. Confidence ranges for particulates extracted from ambient air (relative to benzo[*a*]pyrene)

Compound	Point source	Near mobile source	Home heating	Transport	Geometric mean
Anthracene	2.8	5.7	6.7	2.0	20
Phenanthrene	2.3	1.7	2.6	1.4	13
Fluoranthene	2.2	1.5	1.7	1.4	8.1
Pyrene	2.4	1.4	1.7	1.4	6.7
Benz[*a*]anthracene	2.0	1.4	1.5	1.2	2.3
Perylene	2.7	1.3	1.2	1.9	1.7
Benzo[*e*]pyrene	2.8	1.3	1.6	1.1	1.3
Benzo[*ghi*]perylene	2.5	1.5	1.6	1.2	2.4
Indeno[1,2,3-*cd*]pyrene	1.3	1.4	1.8	1.2	1.2
Anthanthrene	2.0	3.4	1.8	41	1.9
Chrysene and triphenylene	2.1	1.3	2.0	1.3	1.4
Benzofluoranthenes	2.5	1.3	1.9	1.3	1.7

Table AI.15. Mean profiles of polycyclic aromatic hydrocarbons in ambient air (relative to benzo[*a*]pyrene)

Compound	Point source	Near mobile source	Home heating	Transport	Geometric mean
Anthracene	5.5	7.6	1.0	1.8	2.9
Phenanthrene	38	200	39	43	60
Fluoranthene	14	48	12	13	18
Pyrene	9.3	28	11	7.1	12
Benz[*a*]anthracene	1.4	0.82	1.0	0.78	0.97
Perylene	0.33	0.25	0.22	0.24	0.26
Benzo[*e*]pyrene	1.5	1.3	1.6	1.4	1.4
Benzo[*ghi*]perylene	1.4	1.5	2.4	1.3	1.6
Indeno[1,2,3-*cd*]pyrene	1.5	1.3	1.5	1.4	1.4
Anthanthrene	0.19	0.15	0.13	0.20	0.17
Chrysene and triphenylene	3.0	2.7	3.5	2.9	3.0
Benzofluoranthenes	3.6	2.9	3.6	4.4	3.6

Table AI.16. Average profiles for combustion-derived mixtures, ambient air, soil, and sediment (relative to benzo[*a*]pyrene)

Compound	Source mixtures	Ambient air	Soil	Sediment	Geometric mean	Confidence range
Anthracene	3.9	2.9	0.85	0.47	1.6	24
Phenanthrene	18	60	2.8	3.6	4.3	23
Fluoranthene	4.9	18	2.5	3.2	3.8	1.8
Pyrene	4.5	12	3.1	2.4	2.8	2.9
Benz[*a*]anthracene	1.8	0.97	1.4	1.4	1.2	2.6
Perylene	0.51	0.26	0.34	1.4	0.45	17
Benzo[*e*]pyrene	1.0	1.4		1.4	1.1	7.4
Benzo[*ghi*]perylene	0.98	1.6	1.4	0.96	1.0	1.4
Dibenz[*a,h*]anthracene	0.35		0.45	0.30	0.28	4.2
Coronene	0.35			0.45	0.34	3.9
Indeno[1,2,3-*cd*]pyrene	0.51	1.4	0.95	1.2	0.86	4.7
Anthanthrene	0.49	0.17		0.19		0.31
Chrysene and triphenylene	2.3	3.0	1.4	1.2	2.0	3.3
Benzofluoranthenes	1.6	3.6	1.7	2.4	2.5	11

On the basis of this analysis, Muller et al. (1995a,b, 1996) concluded that a wide variety of mixtures have fairly similar profiles of commonly assayed PAH. They assumed that the PAH fraction of all sources of environmental mixtures have profiles similar to the average, as shown in the sixth column of Table AI.16. This conclusion does not include substituted PAH, as there is strong evidence that different mixtures contain different levels of substituted PAH. It does not imply that there are no real differences due to the source of the mixture, the type of fuel, and the pyrolysis conditions that produced it. Furthermore, aerial transport of PAH, degradation in sunlight or by soil microorganisms, and other factors may alter the PAH profile. These factors are, however, unlikely to generate large enough differences in the PAH profiles of mixtures to significantly alter the estimate of risk for a given mixture.

I.2.3.4 Potency of complex mixtures

If benzo[*a*]pyrene is a suitable indicator of the carcinogenic potency of the PAH in a mixture, then the potency of a mixture expressed as the tumour incidence per nanogram of its benzo[*a*]pyrene content should be numerically similar for all mixtures in which PAH are expected to be the major cause of tumorigenic effects. Nesnow et al. (1982b) tested a number of PAH-rich mixtures in a tumour-initiation assay in mouse skin. The different types of mixture were roughly equipotent when the potency was expressed in terms of benzo[*a*]pyrene content (Table AI.17).

I.2.3.5 Key implicit and explicit assumptions

The approach assumes that the levels of individual PAH relative to benzo[*a*]pyrene are relatively stable from mixture to mixture. It also assumes that the risk attributable to PAH in any given mixture is proportional to the risk due to benzo[*a*]pyrene. In other words, the level of benzo[*a*]pyrene is sufficient to estimate the risk of the PAH fraction in a mixture.

Differences in the composition of mixtures from different sources have been used to estimate the contribution of those sources to the ambient levels of PAH (see Gordon & Bryan, 1973; Greenberg et al., 1981; Vogt et al., 1987). Other authors have reported transformation of PAH in the environment. Since some compounds are more photosensitive than others, the proportion of PAH in ambient air will change over time as a result of the different transformation rates of different compounds (Van Cauwenberghe, 1985). Muller et al. (1995a,b, 1996) examined a wide variety of mixtures from different combustion sources and ambient mixtures and concluded that the differences in the profiles of four- to seven-ring unsubstituted PAH relative to the benzo[*a*]pyrene content are not large enough to affect the risk posed by the mixtures significantly. The PAH profile of a tested mixture may deviate from the average profile by

Table A1.17. Potency of various mixtures in an assay for tumour initiation

Source mixture	Incidence per ng benzo[*a*]pyrene
Coke-oven main	9.4×10^{-2}
Coke-oven topside	7.6×10^{-2}
Smoky coal	7.8×10^{-2}
Smokeless coal	2.8×10^{-2}
Roofing tar	1.1×10^{-2}
Wood smoke	5.8×10^{-2}
Diesel engine exhaust	2.5×10^{-2}
Petrol engine exhaust	5.6×10^{-2}
Max/min	8.6

Calculated from data of Nesnow et al. (1982b)

about an order of magnitude (up or down). Since the levels of some PAH may be above and those of others below the expected levels, these differences would tend to cancel each other out, leading to an error of much less than one order of magnitude. Such small differences are below the resolution of the risk assessment process.

I.2.3.6 Application

The first step is to estimate the carcinogenic risk due to exposure to the PAH present in a typical mixture, and this estimate is used for all subsequent assessments. WHO (1987) estimated that the risk for lung cancer due to lifelong exposure to PAH in mixtures by inhalation was 8.7×10^{-5}/ng benzo[*a*]pyrene per m^3. This estimate was based on the assessment of the risk for lung cancer of coke-oven workers conducted by the US Environmental Protection Agency (1984d), which generated an upper bound risk estimate expressed in terms of benzene-extractable material. WHO (1987) converted the US Environmental Protection Agency estimate into benzo[*a*]pyrene levels by assuming that the benzene extract contained 0.71% benzo[*a*]pyrene. Sloof et al. (1989) in the Netherlands estimated the risk for lung cancer to be 1.0×10^{-4}/ng benzo[*a*]pyrene per m^3 on the basis of the estimate of WHO and the assessments of Pike (1983) and Tuomisto & Jantunen (1987). That of Pike (1983) was based on the mortality from lung cancer of gas workers, and that of Tuomisto & Jantunen (1987) was based on the exposure of Chinese women to smoky coal smoke. Muller et al. (1995a,b, 1996) proposed 2.3×10^{-5}/ng benzo[*a*]pyrene per m^3 as the risk for lung cancer from lifelong exposure to PAH in ambient mixtures on the basis of the study of the US Environmental Protection Agency (1984d) on coke-oven workers and assuming that

benzo[*a*]pyrene represents 1.7 ng/µg of benzene-extractable material from coke-oven emissions. Rather than the upper bound, Muller et al. used a maximum likelihood estimate, calculated from the US Environmental Protection Agency study, of 3.9×10^{-5}/µg of benzene-extractable per m^3.

The next step is to estimate the environmental levels of benzo[*a*]pyrene. In a simplified situation, in which the population is exposed to a fixed level of environmental benzo[*a*]pyrene, the lifetime cancer risk is estimated as the product of the potency of a typical mixture (expressed as risk per nanogram of benzo[*a*]pyrene per cubic metre in the case of air) and the level of benzo[*a*]pyrene (expressed as ng/m^3) in the environment. For example, using the risk estimate proposed by Muller et al. (1995a,b, 1996) and assuming that a population is exposed over a lifetime to benzo[*a*]pyrene at 0.5 ng/m^3, the risk of the population is about 1.2×10^{-5}. In other words, about one person in 100 000 would be expected to develop lung cancer in his or her lifetime as a result of exposure to PAH in air.

I.3 Comparison of the three procedures

Each approach has its advantages and disadvantages (Table AI.18):

I.3.1 Individual PAH approach

Main advantages:

- Clearly defined chemical species are assessed.
- A good body of scientific literature is available to evaluate it.
- Not affected by variability in the composition of mixtures
- Relatively easy to apply in ambient environments affected by many sources
- Regulatory experience exists.

Main disadvantages:

- May underestimate risk due to all PAH by considering only a few compounds
- Depends on extrapolation from animal models to humans
- Resource-intensive, as monitoring and analysis are required

I.3.2 Comparative potency approach

Main advantages:

- The risk of whole mixtures, rather than only a few components, is estimated.
- A good body of scientific literature is available to evaluate it.
- Takes advantage of existing data on human carcinogenicity

Table AI.18. Features and properties of three approaches to risk assessment for mixtures containing polycyclic aromatic hydrocarbons (PAH)

Property or feature	Individual PAH approach	Comparative potency approach	Benzo[*a*]pyrene (BaP) surrogate approach
Portion of mixture for which cancer risk is estimated	Selected PAH in complex mixtures account for only portion of risk of PAH fraction	Entire complex mixture	Unsubstituted PAH component of complex mixtures
PAH included in assessment	Relatively few PAH out of hundreds in environment	All PAH	Most PAH, except PAH for which levels do not correlate well with those of BaP, e.g. substituted PAH
Other toxicants included	None	Toxicants present in source mixture	Some, e.g. those present in coke-oven emissions
Assumption of additivity of components of mixture	Yes: good evidence that risks of ill health due to exposure to PAH are approximately additive; little known about additivity of risks due to PAH and other compounds	No: assumption not required	No: assumption not required

Table AI.18 (contd)

Property or feature	Individual PAH approach	Comparative potency approach	Benzo[*a*]pyrene (BaP) surrogate approach
Incorporates directly available data on human cancer risk	No: requires extrapolation from animal models to estimate potency of all PAH	Yes: requires data from animal models to estimate potency of some mixtures	Yes: potency derived from data on potency of coke-oven emissions in humans
Assumption that mixtures from similar sources are about equipotent	No: not applicable	Yes: assumes mixtures from similar sources are about equipotent when expressed in terms of mass of organic extractable material; basis for assumption equivocal	Yes: assumes mixtures from similar sources are about equipotent when expressed in terms of mass of BaP; assumption supported by available data
Assumption that mixtures from different sources are about equipotent	No: assumption not required	No: assumption not required	Yes: evidence from studies of animal models supports the assumption; human data are equivocal and inadequate to validate the approach
Assumption that environmental transformation processes do not change the PAH profile enough to affect the overall cancer risk significantly	No: assumption not required	Yes: evidence that PAH profile relative to BaP does not vary enough to affect the estimated risk significantly	Yes: evidence that PAH profile relative to BaP does not vary enough to affect estimated risk significantly

Table AI.18 (contd)

Property or feature	Individual PAH approach approach	Comparative potency	Benzo[*a*]pyrene (BaP) surrogate approach
Assumption that the PAH profile of various mixtures is roughly comparable	No: assumption not required	No: assumption not required	Yes: good evidence to support this assumption
Suitable for ambient environments affected by multiple sources	Yes	Requires apportioning of ambient mixture to multiple sources	Yes
Monitoring requirements	Selected PAH	Organic extractable matter (this information is not usually reported and methods not standardized)	BaP
Regulatory use	Yes	No	Yes

- Simple and requires inexpensive monitoring

Main disadvantages:

- Does not define the contribution of PAH to estimated overall risk.
- Difficult to use for assessing speciated components of a mixture.
- Risk estimates require estimates of the contributions of individual sources to the levels of organic compounds in the ambient environment.
- The assumption that mixtures from the same source are associated with similar risks may not be supported by the available data.
- The levels of compounds extractable in organic solvents are not usually reported, and the analytical methods are not standardized.

I.3.3 Benzo[*a*]pyrene surrogate approach

Main advantages:

- Can be used to estimate risk of entire PAH component of a mixture
- Simple and based on a few testable assumptions
- Well supported by the available data
- Relatively easy and inexpensive to apply for regulatory purposes
- Regulatory experience exists.

Main disadvantages:

- May result in overestimate of the risk of PAH within a mixture
- Some PAH, such as substituted ones, are not well represented by benzo[*a*]pyrene and must be considered separately.

APPENDIX II

SOME LIMIT VALUES

Regulatory decisions about chemicals taken in a country can be fully understood only within the framework of the legislation of that country. Furthermore, the regulations and guidelines of all countries are subject to change and should always be verified with the appropriate regulatory authorities before application.

II.1 Exposure of the consumer

The concentrations of some components of polycyclic aromatic hydrocarbons (PAH), especially benzo[*a*]pyrene, in air, water, and food and the use of PAH-containing technical products are regulated by law in many countries. The available limit values are listed in Table AII.1.

II.2 Occupational exposure

Regulations for limits in the air at different workplaces are compiled in Table AII.2. Only values for individual substances are given. For some occupations, e.g. roofers and asphalt workers, limit values are not given for individual compounds but for the mixture of organic vapours released, e.g. bitumen fumes, coal-pitch, and coal-tar volatiles, that are soluble in benzene or hexane. These limit values were not taken into account.

II.3 Classification

Only classifications based on a toxicological end-point are given here. Those relevant to exposure in the workplace are shown in Table AII.3. Especially in industrialized countries, classifications also exist for industrial emissions into air, water, and soil. As these are special regulations, which differ from country to country, they are not included.

Some classifications refer to technical mixtures with a high PAH content:

II.3.1 European Union

- A maximum of 50 mg/kg benzo[*a*]pyrene will render coal-tar-derived products carcinogenic (Directive 94/69/EC: European Economic Community, 1994a).
- For lubricant base oils analysed by the legally defined method, the cut-off to define carcinogenicity is 3% of the extract containing mainly PAH, equivalent to 0.5–1 mg/kg benzo[*a*]pyrene (CONCAWE, 1994).

Table AII.1. Limit values for consumer exposure to individual polycyclic aromatic hydrocarbons (PAH) in various countries

Country, year	Compound	Limit value	Reference
Ambient air			
Italy			EEC (1994)
01.01.1996 to 31.12.1998	Benzo[*a*]pyrene	2.5 ng/m^3	
From 01.01.1999	Benzo[*a*]pyrene	1 ng/m^3	
Former USSR, 1985	Benzo[*a*]pyrene	1 ng/m^3	Khesina (1994); UNEP (1994)
Ambient water			
USA, 1984	Sum of benzo[*a*]pyrene, benz[*a*]anthracene, benzofluoranthenes, chrysene, fluoranthene, indeno[1,2,3-*cd*]pyrene, anthracene, pyrene, dibenz[*a*]anthracene	0.2 µg/litre	Slooff et al. (1989)
Former USSR, 1990	Benzo[*a*]pyrene	0.005 µg/litre	UNEP (1994)
EEC, 1980	Sum of fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene	1.2 µg/litre	Slooff et al. (1989)
Drinking-water			
WHO guideline, 1995	Benzo[*a*]pyrene	0.7 µg/litre	WHO (1996)
EEC, 1980 (adopted by most Member States and numerous other European countries)	Sum of fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene	0.2 µg/litre	EEC (1980)

Table All.1 (contd)

Country, year	Compound	Limit value	Reference
Drinking-water (contd)			
Former Czechoslovakia, 1991	Sum of PAH expressed as fluoranthene Benzo[*a*]pyrene	40 µg/litre 0.01 µg/litre	UNEP (1994)
Canada, 1991	Benzo[*a*]pyrene	0.01 µg/litre	UNEP (1994)
Netherlands,1989	Sum of fluoranthene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene	0.1 µg/litre	Slooff et al. (1989)
Soil			
USSR, 1985	Benzo[*a*]pyrene	0.02 mg/kg	UNEP (1994)
Food			
EEC, 1991; use of flavourings in food	Benzo[*a*]pyrene	0.03 µg/kg	European Economic Community (1991)

Table AII.1 (contd)

Country, year	Compound	Limit value	Reference
Food (contd)			
Germany, 1988: meat and meat products	Benzo[*a*]pyrene	1 µg/kg	EEC (1988)
Italy, 1988: food and beverages	Benzo[*a*]pyrene	0.03 µg/kg	Anon. (1988)
Other			
EEC, 1994: tar-oil products for wood preservation	Benzo[*a*]pyrene	50 mg/kg	Appendix I, No. 32 of guideline 94/60/EG dated 20.12.1994
Germany, 1994: products for wood preservation	Benzo[*a*]pyrene	5 mg/kg	Appendix to §1 of German Chemicals Prohibition Directive (1994)
Former Czechoslovakia, 1991	Benzo[*a*]pyrene	Prohibited in cosmetics	UNEP (1994)

Table All.2. Limit values for individual polycyclic aromatic hydrocarbons at various workplaces

Country, year	Workplace or emission source	Compound	Limit value	Reference
Finland, 1989	NR skin absorption	Benzo[*a*]pyrene	10 µg/m³ (TWA)	UNEP (1994)
France, 1988	Production of carbon electrodes	Benzo[*a*]pyrene	0.15 µg/m³	Lafontaine et al. (1990b)
Germany, 1989	Cokeries, oven area	Benzo[*a*]pyrene	5 µg/m³	German Federal Department for Worker Safety (1989)
	Other workplaces	Benzo[*a*]pyrene	2 µg/m³	German Federal Department for Worker Safety (1989)
Sweden,1993	NR	Benzo[*a*]pyrene	2 µg/m³ (LLV) 20 µg/m³ (STV);	Swedish National Board of Occupational Safety & Health (1994)
Former USSR, 1990	NR	Benzo[*a*]pyrene	0.15 µg/m³ (MAC)	UNEP (1994)
Argentina, 1991	NR	Naphthalene	50 mg/m³ (TWA; MPC) 75 mg/m³ (STEL; MPC)	UNEP (1994)
Bulgaria, 1985	NR	Naphthalene	20 mg/m³ (MPC)	UNEP (1994)
Canada, 1991	NR	Naphthalene	50 mg/m³ (TWA; TLV) 75 mg/m³ (STEL; TLV)	UNEP (1994)
Germany, 1993	NR	Naphthalene	50 mg/m³ (MAK)	American Conference of Governmental Industrial Hygienists (1995)
Hungary, 1985	NR	Naphthalene	20 mg/m³ (TWA: MAC) 100 mg/m³ (STEL; MAC)	UNEP (1994)

Table AII.2 (contd)

Country, year	Workplace or emission source	Compound	Limit value	Reference
Italy, 1991	NR	Naphthalene	50 μg/m³	EEC (1991)
Mexico, 1991	NR	Naphthalene	50 mg/m³ (TWA; MXL) 75 mg/m³ (STEL; MXL)	UNEP (1994)
Poland, 1985	NR	Naphthalene	20 mg/m³ (TWA; MPC)	UNEP (1994)
Romania, 1985	NR	Naphthalene	30 mg/m³ (TWA; MPC) 40 mg/m³ (MPC)	UNEP (1994)
Sweden, 1991	NR; skin absorption	Naphthalene	0.2 mg/m³ (TWA; HLV) 0.6 mg/m³ (STEL; HLV)	UNEP (1994)
Switzerland, 1987	NR	Naphthalene	50 mg/m³ (TWA; MAK)	UNEP (1994)
United Kingdom, 1992	NR	Naphthalene	50 mg/m³ (TWA; OES) 75 mg/m³ (STEL; OES)	UNEP (1994)
USA, 1993	NR	Naphthalene	52 mg/m³ (TWA) 79 mg/m³ (STEL)	American Conference of Governmental Industrial Hygienists (1995)
Former USSR, 1993	NR	Naphthalene	20 mg/m³	American Conference of Governmental Industrial Hygienists (1995)
Former Yugoslavia, 1985	NR	Naphthalene	50 mg/m³ (TWA; MAC)	UNEP (1994)
USA, 1993	Cokeries, oven area	Phenylene	0.1 mg/m³ (TWA)	American Conference of Governmental Industrial Hygienists (1995)

Table AII.2 (contd)

Country, year	Workplace or emission source	Compound	Limit value	Reference
Former USSR, 1989	NR	Phenylene	0.8 mg/m^3 (MAC)	UNEP (1994)
USA, 1993	NR	Pyrene	0.1 mg/m^3 (TWA)	American Conference of Governmental Industrial Hygienists (1995)
Former USSR, 1989	NR	Pyrene	0.03 mg/m^3 (MAC)	UNEP (1994)

NR, not reported; HLV, hygienic limit value; LLV, level limit value; MAC, maximum allowable concentration; MAK, maximum workpace concentration; MPC, maximum permissible concentration; MXL, maximum limit; OES, occupational exposure standard; TWA, time-weighted average; STEL, short-time exposure level; STV, short-term value; TLV, threshold limit value

Table AII.3.Toxicological classifications of polycyclic aromatic hydrocarbons with regard to exposure in the workplace

Compound	ACGIH[a] (TLV)		IARC[b] MAK[d]	EU[c]	German
	TWA (8 h)	STEL (15 min)			
Benz[*a*]anthracene	A2	A2	2A	Carcinogenic, category 2	A2
Benzo[*b*]fluoranthene	A2	A2	2B	Carcinogenic, category 2	A2
Benzo[*j*]fluoranthene			2B	Carcinogenic, category 2	A2
Benzo[*k*]fluoranthene			2B	Carcinogenic, category 2	A2
Benzo[*a*]yrene	A2		2A	Carcinogenic, category 2	A2
Chrysene	A2	A2	3	Carcinogenic, category 2	A2
Dibenz[*a,h*]anthracene			2A	Carcinogenic, category 2	A2
Dibenzo[*a,e*]pyrene		2B			A2
Dibenzo[*a,h*]pyrene		2B			A2
Dibenzo[*a,i*]pyrene		2B			A2
Dibenzo[*a,l*]pyrene		2B			A2
Indeno[1,2,3-*cd*]pyrene		2B			A2
5-Methylchrysene		2B			

[a] American Conference of Governmental Industrial Hygienists (1995)
[b] IARC (1987) see Section 12
[c] EU-Rili (Appendix I)
[d] German Dangerous Chemicals Directive (1995)

- Any substance containing benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*j*]fluoranthene, benzo[*a*]pyrene, or dibenz[*a*,*h*]anthracene at a concentration > 0.1% is regarded as carcinogenic (Annex I of Directive 67/548/EEC). It therefore cannot be supplied to the general public in the European Union (Directive 94/60/EC) but only to professional users (Von Meyerinck, 1995).

11.3.2 USA

- Coal-tar and coal-pitch volatiles, which are mixtures of organic vapours with high PAH levels, are classified as A1, confirmed human carcinogens (American Conference of Governmental Industrial Hygienists, 1995).
- Diesel exhaust is considered to be a suspected human carcinogen (A2), but notice has been given of intended changes (American Conference of Governmental Industrial Hygienists, 1995).

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1. RÉSUMÉ

1.1 Choix des composés pour la monographie

Les hydrocarbures aromatiques polycycliques forment un vaste groupe de composés et c'est par centaines qu'ils peuvent être libérés dans l'environnement lors de la combustion incomplète ou de la pyrolyse des matières organiques, constituant ainsi une source importante d'exposition humaine. L'étude des matrices susceptibles d'être importantes sur le plan écologique tels que les résidus de combustion du charbon, les gaz d'échappement des véhicules à moteur, les huiles lubrifiantes usées et la fumée de tabac, montre que leur activité cancérogène est essentiellement liée à leur teneur en HAP.

Les HAP se présentent presque toujours sous la forme de mélanges. Etant donné que leur composition est complexe et qu'elle dépend du processus qui leur a donné naissance, il n'a pas été possible de passer en revue tous les mélanges susceptibles de contenir des hydrocarbures aromatiques polycycliques. C'est pourquoi on a choisi 33 composés (31 composés originaux et 2 dérivés alkylés) en vue d'une évaluation sur la base des données pertinentes relatives aux points d'aboutissement toxicologiques retenus ou à l'exposition (Tableau 1).Toutefois, étant donné que l'on ne disposait d'études épidémiologiques que pour les mélanges et que ces études sont indispensables pour l'évaluation du risque, les sections 8 et 10 exposent les résultats relatifs à des mélanges d'HAP, contrairement au reste de la monographie.

Nombre d'articles et de mises au point ont été publiées au sujet de la présence, de la distribution et de la transformation des HAP dans l'environnement ainsi que de leurs effets toxicologiques et écotoxicologiques. Sauf indication contraire, seules les références des 10 à 15 dernières années sont prises en compte dans la monographie. En revanche, les mises au point consacrées à des études plus anciennes sont citées à titre de complément d'information.

Tableau 1. Les hydrocarbures aromatiques polycycliques évalués dans cette monographie

Nom commun	Nom CAS	Synonyme[a]	No d'enregistrement CAS
Acenaphthylene	Acenaphthylene		91-20-3
Acenaphthene	Acenaphthylene, 1,2-dihydro-		208-96-8
Anthanthrene	Dibenzo[*def,mno*]chrysene		191-26-4
Anthracene	Anthracene		120-12-7
Benz[*a*]anthracene	Benz[*a*]anthracene	1,2-Benzanthracene, tetraphene	56-55-3
Benzo[*a*]fluorene	11*H*-Benzo[*a*]fluorene	1,2-Benzofluorene	238-84-6
Benzo[*b*]fluorene	11*H*-Benzo[*b*]fluorene	2,3-Benzofluorene	243-17-4
Benzo[*b*]fluoranthene	Benz[*e*]acephenanthrylene	3,4-Benzofluoranthene	205-99-2
Benzo[*ghi*]fluoranthene	Benzo[*ghi*]fluoranthene	2,13-Benzofluoranthene	203-12-3
Benzo[*j*]fluoranthene	Benzo[*j*]fluoranthene	10,11-Benzofluoranthene	205-82-3
Benzo[*k*]fluoranthene	Benzo[*k*]fluoranthene	11,12-Benzofluoranthene	207-08-9
Benzo[*ghi*]perylene	Benzo[*ghi*]perylene	1,12-Benzoperylene	191-24-2
Benzo[*c*]phenanthrene	Benzo[*c*]phenanthrene	3,4-Benzophenanthrene	195-19-7
Benzo[*a*]pyrene	Benzo[*a*]pyrene	3,4-Benzopyrene[b]	50-32-8
Benzo[*e*]pyrene	Benzo[*e*]pyrene	1,2-Benzopyrene	192-97-2
Chrysene	Chrysene	1,2-Benzophenanthrene	218-01-9
Coronene	Coronene	Hexabenzobenzene	191-07-1
Cyclopenta[*cd*]pyrene	Cyclopenta[*cd*]pyrene	Cyclopenteno[*cd*]pyrene	27208-37-3
Dibenz[*a,h*]anthracene	Dibenz[*a,h*]anthracene	1,2:5,6-Dibenzanthracene	53-70-3
Dibenzo[*a,e*]pyrene	Naphtho[1,2,3,4-*def*]chrysene	1,2:4,5-Dibenzopyrene	192-65-4
Dibenzo[*a,h*]pyrene	Dibenzo[*b,def*]chrysene	3,4:8,9-Dibenzopyrene	189-64-0

Tableau 1 (cont.)

Dibenzo[*a,l*]pyrene	Benzo[*rst*]pentaphene	3,4:9,10-Dibenzopyrene	189-55-9
Dibenzo[*a,l*]pyrene	Dibenzo[*def,p*]chrysene	1,2:3,4-Dibenzopyrene	191-30-0
Fluoranthene	Fluoranthene		206-44-0
Fluorene	9*H*-Fluorene		86-73-7
Indeno[1,2,3-*cd*]pyrene	Indeno[1,2,3-*cd*]-pyrene	2,3-*o*-Phenylenpyrene	193-39-5
5-Methylchrysene	Chrysene, 5-methyl-		3697-24-3
1-Methylphenanthrene	Phenanthrene, 1-methyl-		832-69-9
Naphthalene	Naphthalene		91-20-3
Perylene	Perylene	*peri*-Dinaphthalene	198-55-0
Phenanthrene	Phenanthrene		85-01-8
Pyrene	Pyrene	Benzo[*def*]phenanthrene	129-00-0
Triphenylene	Triphenylene	9,10-Benzophenanthrene	217-59-4

Des listes assez complètes de synonymes ont également été publiées par le CIRC (1989) et par Loening & Merritt (1990).

[a] Synonyme commun utilisé dans la littérature

[b] On le trouve également sous le nom de benzo(*def*)chrysène.

1.2 Identité, propriétés physiques et chimiques et méthodes d'analyse

On désigne généralement par hydrocarbures aromatiques polycycliques un vaste ensemble de composés contenant un ou plusieurs noyaux aromatiques condensés et constitués de carbone et d'hydrogène. Ces hydrocarbures sont solides à la température ambiante. Ils ont pour caractéristiques communes d'avoir un point de fusion et un point d'ébullition élevés, une faible tension de vapeur et d'être très peu solubles dans l'eau-d'autant moins que leur masse moléculaire est plus élevée. Ils sont solubles dans de nombreux solvants organiques et très lipophiles. Chimiquement, ils sont relativement inertes. Les réactions intéressantes du point de vue de leur devenir dans l'environnement et des possibilités de pertes au cours des prélèvements d'air sont celles qui comportent une photodécomposition ou dans lesquelles interviennent les oxydes d'azote, l'acide nitrique, les oxydes de soufre, l'acide sulfurique, l'ozone et les radicaux hydroxyles.

L'échantillonnage dans l'air ambiant s'effectue par recueil de matières particulaires sur filtres en fibre de verre, polytétrafluoréthylène, ou fibre de quartz, au moyen d'échantillonneurs de grand volume ou d'échantillonneurs passifs. Comme il y a risque de volatilisation des hydrocarbures de la phase gazeuse, qui pourraient s'évaporer des filtres lors de l'échantillonnage, on a l'habitude de les piéger par adsorption sur mousse de polyuréthane. La variabilité des résultats provient essentiellement de ce processus d'échantillonnage.

Sur les lieux de travail, les prélèvements d'air se font à faible débit; les particules sont recueillies sur des filtres de fibre de verre ou de polytétrafluoréthylène et la phase gazeuse sur résine Amberlite XAD-2. Les collecteurs de gaz de cheminées sont constitués de filtres en fibre de verre ou en quartz placés devant un réfrigérant destiné à retenir le condensant et une cartouche d'adsorbant (en général, de l'XAD-2). Les gaz d'échappement des véhicules à moteur sont recueillis au laboratoire, pendant un cycle de fonctionnement normalisé simulant les conditions réelles. Les émissions sont recueillies telles quelles ou après dilution dans de l'air froid filtré.

De nombreuses techniques d'extraction et de purification ont été décrites. En fonction de la matrice, on peut extraire les HAP au

Soxhlet, par action des ultrasons, par partage liquide-liquide ou, après dissolution et digestion alcaline, au moyen d'un solvant sélectif. On utilise aussi l'extraction par fluide supercritique pour diverses substances solides présentes dans l'environnement. L'efficacité de l'extraction dépend largement du solvant et nombre de solvants utilisés par le passé se sont révélés inappropriés. Après l'extraction, on procède généralement à une purification par chromatographie sur colonne - d'alumine, de gel de silice ou de sephadex LH-20- mais aussi par chromatographie sur couche mince.

La recherche et le dosage s'effectuent classiquement par chromatographie en phase gazeuse avec détection par ionisation de flamme ou encore par chromatographie en phase liquide à haute performance avec détection en UV ou fluorescence, généralement en série. Pour la chromatographie en phase gazeuse, on utilise des colonnes capillaires en silice fondue, avec des polysiloxanes comme phase stationnaire (SE-54 et SE-52); pour la chromatographie en phase liquide, on utilise couramment des colonnes de gel de silice C-18. Pour confirmer l'identité des pics chromatographiques, on couple généralement un spectromètre de masse au chromatographie.

Le choix des HAP à doser dépend de l'objectif de la mesure: étude à visée sanitaire, investigation écotoxicologique ou recherche d'une source de pollution. La recherche et le dosage de divers groupes de composés peuvent être demandés au niveau national ou international.

1.3 Sources d'exposition humaine et environnementale

On sait peu de choses sur la production des hydrocarbures aromatiques polycycliques et sur les processus auxquels ils peuvent être soumis, mais il est probable que ces activités n'entraînent la libération que de petites quantités d'hydrocarbures. Ceux que l'on retrouve dans l'environnement sont principalement utilisés comme intermédiaires dans la préparation du chlorure de polyvinyle et de divers plastifiants (naphtalène), de pigments (acénaphtène, pyrène), de colorants (anthracène, fluoranthène) et de pesticides (phénanthrène).

Les émissions les plus importantes d'hydrocarbures aromatiques polycycliques sont dues à la combustion incomplète de matières

organiques lors de divers processus industriels ou d'autres activités humaines, notamment:

— les diverses opérations effectuées sur la houille, le pétrole brut et le gaz naturel, y compris la cokéfaction, la conversion de la houille, le raffinage du pétrole et la production de noirs de carbone, de créosote, de goudron de houille et de bitume;
— la production d'aluminium, de fer et d'acier dans les divers ateliers et fonderies;
— la génération d'énergie calorifique par les centrales thermiques, le chauffage des habitations et la cuisine;
— le brûlage des déchets;
— la circulation automobile; et
— la fumée de tabac dispersée dans l'environnement.

Les HAP, notamment ceux qui ont une masse moléculaire élevée, s'adsorbent sur les particules de matière une fois qu'ils sont libérés dans l'environnement par la voie atmosphérique. La pénétration dans l'hydrosphère et la géosphère s'effectue ensuite selon un processus de dépôt par voie humide ou par voie sèche. La conservation du bois par traitement au créosote constitue une autre source de pénétration d'HAP dans l'hydrosphère et les dépôts de déchets contaminés, comme par exemple les boues d'égout et les cendres volantes, contribuent à l'introduction de ces composés dans la géosphère. On possède peu de renseignements sur le passage des HAP dans la biosphère. Des hydrocarbures aromatiques polycycliques sont présents à l'état naturel dans la tourbe, le lignite, la houille et le pétrole brut. La plupart de ceux que l'on trouve dans l'anthracite sont fermement liés à la structure carbonée et ne peuvent pas en être extraits par lessivage.

On a pu déterminer le passage d'HAP dans l'environnement en mettant en évidence un profil de concentration caractéristique, mais cela n'a été possible que dans quelques cas. Le benzo*(a)* pyrène est fréquemment utilisé comme indicateur de la présence d'HAP, notamment dans les études un peu anciennes. En général, les chiffres concernant les émissions d'HAP ne sont que des estimations basées sur des données plus ou moins fiables et elles ne donnent qu'une idée approximative de l'exposition.

Les sources d'HAP les plus importantes sont les suivantes:

La cokéfaction de la houille: les émissions atmosphériques d'HAP résultant de la cokéfaction de la houille ont sensiblement diminué en Allemagne au cours des 10 dernières années par suite des améliorations techniques apportées aux installations industrielles, à la fermeture des anciennes usines et au recul de la production de coke. On pense que la situation est à peu près la même dans le reste de l'Europe occidentale, au Japon, et aux Etats-Unis, sans toutefois disposer de données à ce sujet.

La production d'aluminium (notamment en raison de l'utilisation d'anodes spéciales en graphite), *de fer et d'acier* ainsi que les liants utilisés en fonderie dans les moules à sable. On ne possède guère d'informations à ce sujet.

Le chauffage collectif ou individuel: les émissions sont principalement constituées de phénanthrène, de pyrène et de chrysène. Les poêles à bois en émettent 25 à 1000 fois plus que les poêles à charbon et dans les régions où l'on se chauffe surtout au bois, la majeure partie des émissions atmosphériques d'HAP ont ce mode de chauffage pour origine, principalement l'hiver. On considère donc que le chauffage des habitations est une source importante d'hydrocarbures aromatiques polycycliques dans les pays en développement où l'on brûle de la biomasse dans des poêles assez rudimentaires.

La cuisine: la combustion incomplète des matières combustibles peut produire des HAP, de même que le chauffage de l'huile de cuisine et la cuisson des denrées alimentaires elles-mêmes.

La circulation des véhicules à moteur: les principaux hydrocarbures rejetés par les véhicules dotés de moteurs à essence sont le fluoranthène et le pyrène, tandis que le naphtalène et l'acénaphtène prédominent dans les gaz d'échappement des moteurs diesel. Le cyclopenta (*cd*)-pyrène est émis en grande quantité par les moteurs à essence mais il est juste au-dessus de la limite de détection dans les gaz d'échappement des moteurs diesel. Le taux d'émission, qui dépend du composé, du type de véhicule et de l'état de son moteur, de même que des conditions dans lesquelles l'essai se déroule, varie de quelques nanogrammes par kilomètre à > 1000 mg/km. La pose d'un catalyseur

réduit de façon spectaculaire les émissions d'HAP par les moteurs d'automobiles.

Les feux de forêt: Dans les pays où la forêt couvre de vastes étendues de territoire, les feux de forêt peuvent contribuer de façon importante à l'émission d'HAP.

Centrales thermiques à charbon: Les HAP libérés dans l'atmosphère par ces centrales sont principalement des composés bi- et tricycliques. Dans les zones contaminées, la teneur en HAP de l'air ambiant peut être supérieure à celle qui résulte des émission de cheminées.

Incinération des déchets: les émissions d'HAP dans les gaz de cheminée des usines d'incinération sont inférieures à 10 mg/m^3 dans un certain nombre de pays.

1.4 Transport, distribution et transformation dans l'environnement

La destinée des HAP, qu'ils soient seuls ou en mélange, dépend d'un certain nombre de processus de distribution et de transformation. Les processus de distribution les plus importants sont le partage entre l'eau et l'air, entre l'eau et les sédiments et entre l'eau et les organismes vivants.

Comme ces hydrocarbures sont hydrophobes et très peu solubles dans l'eau, il n'ont qu'une très faible affinité pour la phase aqueuse; toutefois, bien qu'ils soient libérés dans l'environnement par la voie atmosphérique, on les retrouve également en concentration importante dans l'hydrosphère, du fait que leur constante de Henry est faible. Etant donné que leur affinité est plus grande pour la phase organique que pour la phase aqueuse, leur coefficient de partage entre les solvants organiques- comme l'octanol- et l'eau est élevé. Ils ont également une forte affinité pour les fractions organiques des sédiments, du sol et des organismes vivants, de sorte qu'ils s'accumulent dans les organismes aquatiques et sédimentaires ainsi que dans leur nourriture. On ne connaît pas avec certitude l'importance relative de leur fixation à partir de la nourriture et de l'eau. Chez la daphnie et les mollusques, il y a corrélation positive entre

l'accumulation d'HAP provenant de l'eau et le coefficient de partage entre l'octanol et l'eau (K_{ow}). Par contre, chez les poissons et les algues qui sont capables de métaboliser ces hydrocarbures, il n'y a pas de corrélation entre la concentration des différents HAP et le K_{ow}.

Le phénomène de bioamplification, c'est-à-dire la concentration d'une substance dans l'organisme d'animaux à chaque niveau trophique successif de la chaîne alimentaire, n'a pas été observé en milieu aquatique et il ne semble pas qu'il puisse se produire, car la plupart des organismes sont tout à fait à même de métaboliser les HAP. Ce sont les organismes qui occupent les niveaux trophiques les plus élevés qui possèdent la plus grande capacité de biotransformation.

La dégradation des HAP s'opère par photodécomposition, par biodégradation microbienne et, chez les organismes supérieurs, par métabolisation.Cette dernière voie de transformation n'a guère d'influence sur la destinée globale des HAP dans l'environnement, mais elle joue néanmoins un rôle biologique important du fait que des métabolites cancérogènes sont susceptibles de se former. Comme les HAP sont chimiquement stables et dépourvus de groupements réactifs, l'hydrolyse n'intervient pas dans leur décomposition. Il n'existe guère d'épreuves classiques pour l'étude de la biodégradation des HAP. Celle-ci s'opère généralement en aérobiose, la vitesse du processus diminuant fortement avec le nombre de cycles aromatiques. En anaérobiose, la dégradation est beaucoup plus lente.

Dans l'air et dans l'eau, les HAP subissent une photo-oxydation en présence de radicaux ou molécules sensibilisateurs comme OH, NO_3 ou O_3. Au laboratoire, le temps de demi-réaction avec les radicaux OH présents dans l'air est d'environ 1 jour; en revanche la constante de vitesse est généralement beaucoup plus faible dans le cas des réactions avec NO_3 et O_3. En principe, l'adsorption des HAP de masse moléculaire élevée sur les particules carbonées devrait stabiliser leur réaction avec les radicaux OH. La réaction des HAP possédant 2 à 4 cycles avec NO_3, réaction qui a lieu principalement en phase gazeuse, conduit à la nitration de ces hydrocarbures, c'est-à-dire à la formation de produits notoirement cancérogènes. Pour certains d'entre eux, la photo-oxydation semble être plus rapide dans l'eau que dans l'air. Les calculs basés sur la physico-chimie et la biodégradabilité de

ces hydrocarbures montrent que ceux qui possèdent quatre cycles aromatiques ou davantage persistent dans l'environnement.

1.5 Concentrations dans l'environnement et exposition humaine

Les HAP sont présent dans tout l'environnement et de nombreuses études ont permis d'en mettre un certain nombre en évidence dans divers compartiments.

1.5.1 Air

La concentration des HAP a tendance à être environ 10 fois plus élevée l'hiver que l'été. En hiver, la source principale d'hydrocarbures aromatiques polycyclique est le chauffage des habitations, alors qu'en été, ce sont les gaz d'échappement des véhicules à moteur qui sont les principaux responsables. Dans l'air de diverses agglomérations, on a mis en évidence, pour un certain nombre d'hydrocarbures aromatiques polycycliques, des concentrations moyennes de 1 à 30 ng/m^3. Dans certains grands centres urbains, où la circulation automobile est très intense et où l'on utilise beaucoup la biomasse pour le chauffage des habitations, comme Calcutta par exemple, on a trouvé des teneurs allant jusqu'à 200 ng/m^3 pour divers HAP. Sous les tunnels routiers, on a mesuré des concentrations de 1 à 50 ng/m^3. Le cyclopenta(*cd*) pyrène et le pyrène étaient présents à des concentrations atteignant 100 ng/m^3. Dans une station de métro, on a mesuré des concentrations allant jusqu'à 20 ng/m^3. A proximité de sources de pollution d'origine industrielle, la concentration moyenne des divers HAP s'étalait de 1 à 10 ng/m^3. La concentration du phénanthrène pouvait atteindre environ 310 ng/m^3.

La concentration de fond des HAP est d'au moins deux ordres de grandeur plus faible qu'à proximité de sources telles que les véhicules à moteur. A titre d'exemple, à 1100 m d'altitude, on a obtenu des valeurs comprises entre 0,004 et 0,03 ng/m^3.

1.5.2 Eaux superficielles et précipitations

On pense que la majeure partie des HAP présents dans l'eau y ont été entraînés par ruissellement ou résultent de retombées atmos-

phériques (petites particules) ou encore de l'abrasion de l'asphalte (grosses particules). Il reste que, pour une étendue d'eau donnée, la majeure partie des HAP n'a pas toujours la même origine. En général, la plupart des échantillons d'eaux superficielles contiennent divers HAP à des concentrations pouvant atteindre 50 ng/litre, mais dans des cours d'eau extrêmement pollués, on a relevé des concentrations allant jusqu'à 6 000 ng/litre. La teneur des eaux souterraine en HAP se situe entre 0,02 et 1,8 ng/litre et dans des échantillons d'eau destinée à la boisson, on a trouvé des valeurs du même ordre de grandeur. Les HAP présents dans l'eau de boisson ont principalement pour origine le revêtement d'asphalte des réservoirs et des canalisations.

Dans l'eau de pluie, on relève des concentrations comprises entre 10 et 200 ng/litre, et des teneurs allant jusqu'à 1000 ng/litre ont été mesurées dans la neige et le brouillard.

1.5.3 Sédiments

La concentration des HAP dans les sédiments est généralement 10 fois plus forte que dans les précipitations.

1.5.4 Sol

Les HAP présents dans le sol proviennent principalement des dépôts atmosphériques, de la carbonisation des végétaux et du dépôt de particules en suspension dans les effluents ou divers types de déchets. L'ampleur de la pollution des sols dépend de facteurs tels que le type de cultures auxquels ils sont soumis, la porosité et la teneur en substances humiques.

A proximité des sources industrielles, on a trouvé, pour les divers HAP, des concentrations dans le sol pouvant atteindre 1 g/kg. Pour les HAP ayant une autre origine, par exemple les gaz d'échappement des véhicules à moteur, la concentration dans le sol va de 2 à 5 mg/kg. Dans les zones non polluées, la teneur du sol en HAP se situe entre 5 et 100 µg/kg.

1.5.5 Denrées alimentaires

Les denrées alimentaires crues ou plus généralement, non transformées, ne contiennent normalement pas de grandes quantités d'HAP, mais il peut s'en former lorsqu'on fait rôtir, griller, frire ou que l'on prépare d'une manière ou d'une autre, ces produits. Les légumes peuvent être contaminés par le dépôt de particules aéroportées ou par le sol sur lequel ils ont poussé. D'après diverses mesures, la concentration d'un certain nombre d'HAP dans la viande, le poisson, les produits laitiers, les légumes, les fruits, les céréales et les produits céréaliers, les pâtisseries et confiseries, les boissons ainsi que les huiles et graisses animales et végétales, se situe entre 0,01 et 10 µg/kg. On a trouvé des concentrations supérieures à 100 µg/kg dans de la viande fumée et pouvant aller jusqu'à 86 µg/kg dans du poisson fumé; dans des céréales fumées, les valeurs relevées atteignaient 160 µg/kg. Dans de l'huile de coco on en a trouvé jusqu'à 460 µg/kg. Dans le lait maternel, les mesures ont donné des valeurs comprises entre 0,003 et 0,03 µg/kg.

1.5.6 Organismes aquatiques

On sait que les organismes marins absorbent et accumulent les HAP présents dans l'eau. Le degré de contamination dépend du développement industriel et urbain ainsi que du trafic maritime dans la zone. Des concentrations allant jusqu'à 7 mg/kg ont été mises en évidence dans des organismes aquatiques vivant à proximité de points de décharge d'effluents industriels et la concentration moyenne d'HAP dans l'organisme d'animaux aquatiques prélevés dans des zones contaminées s'est révélée comprise entre 10 et 500 µg/kg, avec des pointes à 5 mg/kg.

Chez des animaux aquatiques prélevés dans des zones où la pollution par des HAP n'avait pas d'origine précise, on a trouvé des valeurs moyennes de 1 à 100 µg/kg, mais des valeurs supérieures sont possibles, par exemple 1 mg/kg chez des homards du Canada.

1.5.7 Organismes terrestres

Chez des insectes, on a relevé des concentrations comprises entre 730 et 5 500 µg/kg. La teneur en HAP des déjections de lombrics varie

sensiblement selon le lieu: dans une région fortement industrialisée de l'est de l'Allemagne, la teneur en benzo(*a*)pyrène des déjections de lombrics atteignait 2 mg/kg.

1.5.8 Population générale

Les principales sources d'exposition non professionnelle sont les suivantes: air pollué, fumée de feux et foyers non couverts, fumée de tabac dispersée dans l'environnement, denrées alimentaires et eau de boisson contaminées, utilisation de produits contaminés par des HAP. On peut trouver dans l'air à l'intérieur des habitations des HAP qui proviennent du chauffage ou de la présence de fumée de tabac, à des concentrations moyennes de 1 à 100 ng/m^3, avec des pointes à 2300 ng/m^3.

On estime que l'apport d'HAP d'origine alimentaire est de 0,10 à 10 µg par jour et par personne. La quantité totale de benzo (*a*) pyrène absorbée quotidiennement avec l'eau de boisson est estimée à 0,0002 µg par personne. Ce sont les céréales et les produits céréaliers qui contribuent le plus à cet apport car ils sont un constituant majeur de la ration alimentaire totale.

1.5.9 Exposition professionnelle

A proximité d'une batterie de fours à coke, on a relevé des teneurs en benzo (*a*) pyrène allant de <0,1 à 100-200 µg/m^3, avec des pointes à environ 400 µg/m^3. Dans les installations modernes de gazéification de la houille, la concentration des HAP est en général de < 1 µg/m^3 et de 30 µg/m^3 au maximum. Des échantillons prélevés individuellement dans la zone de travail de personnes affectées à divers postes dans des raffineries de pétrole, ont révélé une exposition comprise entre 2,6 et 470 µg/m^3. Dans des échantillons d'air prélevés dans des ateliers de préparation de bitume, on a obtenu une concentration en HAP totaux de 0,004 à 50 µg/m^3. Lors de travaux d'enrobage de chaussées, on a relevé sur le site des concentrations atmosphériques en HAP totaux pouvant atteindre 190 µg/m^3, avec une concentration moyenne de 0,13 µg/m^3. Dans une fonderie d'aluminium, la dosimétrie individuelle indiquait des valeurs de 0,05-9,6 µg/m^3 pour les HAP totaux, mais dans les urines d'ouvriers travaillant dans une unité de production d'aluminium, on n'en a relevé

que de très petites quantités. Dans une fonderie allemande, les échantillons d'air ambiant contenaient des HAP à des concentrations pouvant atteindre 5 µg/m^3. Elles étaient respectivement égales à 3-40 µg/m^3 dans des mines de fer et à 4-530 µg/m^3 dans des mines de cuivre. Dans les vapeurs de cuisson d'une usine de produits alimentaires, la concentration en HAP était comprise entre 0,07 et 26 µg/m^3.

1.6 Cinétique et métabolisme

Les HAP sont absorbés au niveau des poumons, des voies digestives et par la voie percutanée. La vitesse de résorption au niveau pulmonaire dépend de la nature de l'hydrocarbure, de la granulométrie des particules sur lesquelles il est adsorbé et de la composition du substrat adsorbant. Les hydrocarbures adsorbés sur des particules sont éliminés plus lentement des poumons que les hydrocarbures libres. Chez les rongeurs, la résorption est rapide dans les voies digestives, mais les métabolites, excrétés par la voie biliaire, finissent par retourner dans l'intestin. Des études effectuées sur des rongeurs avec des mélanges d'HAP ayant subi un postmarquage au ^{32}P ont montré qu'après absorption par la voie percutanée, les hydrocarbures parvenaient jusqu'aux poumons où ils se liaient à l'ADN. Chez la souris, la vitesse d'absorption percutanée dépend de la nature du composé.

Quelle que soit la voie d'administration, les HAP se répartissent dans tout l'organisme et on en retrouve dans pratiquement tous les organes internes, mais plus particulièrement dans ceux qui sont riches en lipides. Après injection intraveineuse à des rongeurs, les HAP sont rapidement éliminés du courant sanguin, mais ils sont capables de traverser la barrière foeto-placentaire et on en a décelé la présence dans les tissus foetaux.

Les HAP ont un métabolisme complexe. En général, le composé initial est époxydé puis transformé en phénol, diol ou tétrol qui peut lui-même se conjuguer à l'acide sulfurique ou à l'acide glucuronique (sulfuro- ou glucuro-conjugaison) ou encore au glutathion. La plupart du temps, la métabolisation d'un hydrocarbure aromatique polycyclique entraîne sa détoxication, mais certains d'entre eux subissent une activation en composés susceptibles de se lier à l'ADN, principalement des époxydes-diols, qui sont capables d'amorcer un

processus de cancérisation. Les métabolites et leurs conjugués sont excrétés dans les urines et les matières fécales, mais les conjugués excrétés dans la bile peuvent être hydrolysés par les enzymes de la flore intestinale et être ensuite réabsorbés. On peut déduire des données disponibles au sujet de la charge totale de l'organisme humain en HAP, que ces hydrocarbures ne s'accumulent pas dans l'organisme et qu'ils se renouvellent rapidement. Il faut exclure de cette conclusion les HAP qui forment des liaisons covalentes avec certain constituants tissulaires, notamment les acides nucléiques et que les processus de réparation ne permettent pas d'éliminer.

1.7 Effets sur les mammifères de laboratoire et effets *in vitro*

La toxicité aiguë des HAP est faible à modérée. Un HAP bien caractérisé, le naphtalène, a donné des valeurs de la DL_{50} par voie orale ou intraveineuse égales à 100-500 mg/kg pc chez la souris et une DL_{50} moyenne par voie orale de 2700 mg/kg pc chez le rat. Pour les autres HAP, on a obtenu des valeurs similaires. Du naphtalène administré en doses uniques à des souris, des rats et des hamsters, a provoqué l'apparition d'une nécrose des bronchioles.

Des études à court terme ont révélé des anomalies hématologiques, dues à une myélotoxicité dans le cas du benzo(a)pyrène et qui, en ce qui concerne le dibenz (*a,h*) anthracène, consistaient en modifications hémolymphatiques. Dans le cas du naphtalène, on a constaté une anémie. Toutefois, une étude de 7 jours au cours de laquelle on a administré du naphtalène à des souris par voie orale et intrapéritonéale, a révélé l'existence d'une tolérance aux effets de cet hydrocarbure. On n'a que rarement décrit des effets généraux dus à une longue exposition à des HAP, car l'effet toxicologique retenu dans la plupart des études correspondantes était la cancérisation. Aux doses où se déclenche un processus de cancérisation, on observe également des effets toxiques importants.

En étudiant les effets cutanés indésirables des HAP après application sur l'épiderme, on a constaté que des hydrocarbures faiblement ou non cancérogènes comme le pérylène, le benzo(*e*) pyrène, le phénanthrène, le pyrène, l'anthracène, l'acénaphtène, le fluorène et le fluoranthène étaient inactifs, alors que les hydrocarbures

cancérogènes comme le benz (*a*) anthracène, le dibenz (*a,h*)-anthracène et le benzo(*a*)pyrène provoquaient une hyperkératose. Les vapeurs d'anthracène et de naphtalène peuvent causer une légère irritation oculaire. Le benzo(*a*)pyrène a provoqué une hypersensibilité de contact chez des cobayes et des souris. Le benz*(a)*anthracène, le benzo(*a*) pyrène, le dibenz(*a,h*)anthracène et le naphtalène se sont révélés embryotoxiques chez la souris et le benzo(*a*)pyrène a également eu des effets tératogènes et des effets sur la reproduction. On a fait de gros efforts pour tenter de d'élucider les bases génétiques des effets embryotoxiques du benzo(*a*)pyrène. On n'observe de morts foetales et de malformations que si le système des monooxygénases du cytochrome p450 est inductible, chez la mère (avec migration transplacentaire) ou chez l'embryon. On ne peut expliquer tous les effets observés par une prédisposition génétique, mais chez la souris et le lapin, cet hydrocarbure a une activité transplacentaire qui se traduit par des adénomes pulmonaires et des papillomes cutanés dans la descendance des animaux traités. On a également observé une diminution de la fécondité et la destruction des ovocytes.

On a également procédé à de nombreuses études sur la génotoxicité des HAP et sur leur aptitude à provoquer une transformation cellulaire. La plupart des 33 HAP qui font l'objet de cette monographie sont génotoxiques ou ont des chances de l'être. Les seuls composés pour lesquels on ait obtenu des résultats négatifs dans toutes les épreuves sont l'anthracène, le fluorène et le naphtalène. Dans le cas du phénanthrène et du pyrène, l' irrégularité des résultats ne permet pas de se prononcer avec certitude sur leur génotoxicité.

Les travaux très complets qui ont été consacrés à la cancérogénicité des hydrocarbures aromatiques polycycliques montrent que 26 des 33 composés qui font l'objet de la présente monographie sont effectivement cancérogènes ou soupçonnés de l'être (Tableau 2). Le mieux connu de tous est le benzo(*a*)pyrène, qui a été étudié par toutes les méthodes existantes sur sept espèces. Plus d'une douzaine d'études ont été consacrées à l'anthracène, à l'anthanthrène, au benz(*a*)-anthracène, au chrysène, au dibenz(*a,h*) anthracène, au dibenzo(*a,i*)-pyrène, au 5-méthylchrysène, au phénanthrène et au pyrène. Aux études spéciales sur l'immunotoxicité, la phototoxicité et l'hépatotoxicité des HAP s'ajoutent un certain nombre d'articles sur la toxicité oculaire du naphtalène. L'anthracène, le benzo(*a*)pyrène et un certain

Tableau 2. Résultats des épreuves de génotoxicité et de cancérogénicité effectuées sur les 33 hydrocarbures aromatiques polycycliques étudiés.

Compound	Genotoxicity	Carcinogenicity
Acenaphthene	(?)	(?)
Acenaphthylene	(?)	No studies
Anthracene	-	-
Benz[*a*]anthracene	+	+
Benzo[*a*]fluorene	(?)	(?)
Benzo[*a*]pyrene	+	+
Benzo[*b*]fluoranthene	+	+
Benzo[*b*]fluorene	(?)	(?)
Benzo[*c*]phenanthrene	(+)	+
Benzo[*e*]pyrene	+	?
Benzo[*ghi*]fluoranthene	(+)	(-)
Benzo[*ghi*]perylene	+	-
Benzo[*j*]fluoranthene	+	+
Benzo[*k*]fluoranthene	+	+
Chrysene	+	+
Coronene	(+)	(?)
Cyclopenta[*cd*]pyrene	+	+
Dibenzo[*a,e*]pyrene	+	+
Dibenz[*a,h*]anthracene	+	+
Dibenzo[*a,h*]pyrene	(+)	+
Dibenzo[*a,i*]pyrene	+	+
Dibenzo[*a,l*]pyrene	(+)	+
Fluoranthene	+	(+)
Fluorene	-	-
Indeno[1,2,3-*cd*]pyrene	+	+
1-Methylphenanthrene	+	-
5-Methylchrysene	+	+
Naphthalene	-	?
Perylene	+	-
Phenanthrene	(?)	(?)
Pyrene	(?)	-
Triphenylene	+	-

+, résultat positif; -, résultat négatif; ?, résultat douteux
Parenthèses, résultat tiré d'une petite base de données.

nombre d'autres HAP sont phototoxiques pour la peau des mammifères ou les cultures cellulaires *in vitro*, lorsqu'on les applique sous rayonnement ultraviolet. D'une façon générale, on considère que les HAP ont un effet immunodépresseur. Après administration de benzo*(a)*pyrène à des souris, on a observé une forte immunodépression dans la descendance de ces animaux pendant une période pouvant atteindre 18 mois. On a également constaté un accroissement de la régénération du tissu hépatique et une augmentation du poids du foie. La formation de cataractes sous l'effet du naphtalène chez des souris appartenant à des souches génétiquement différentes, a été attribuée à l'inductibilité du cytochrome P 450.

Dès les années 30, on a proposé des modèles théoriques prenant en compte un grand nombre de résultats expérimentaux, pour tenter de prévoir le pouvoir cancérogène des hydrocarbures aromatiques polycycliques à partir de leur structure moléculaire. Le premier de ces modèles se fondait sur la forte réactivité chimique de certaines doubles liaisons (théorie de la région K). Ultérieurement, on a tenté une approche systématique du problème basée sur la synthèse chimique des divers métabolites possibles et l'étude de leur activité mutagène. Selon cette théorie, dite de la "région en baie", les époxydes adjacents à une région en baie conduisent à la formation d'ions carbonium très stables qui peuvent alkyler les bases nucléiques. Parmi les autres théories, on peut citer la théorie de la "di-région" et celle du "cation radical potentiel".

De nombreux HAP sont cancérogènes pour l'animal et pourraient également l'être pour l'Homme. D'ailleurs, on a montré que l'exposition à divers mélanges contenant des HAP augmentait l'incidence des cancers dans les populations humaines en cause. Ce qui est préoccupant, c'est que les HAP dont l'étude expérimentale a révélé l'activité cancérogène chez l'animal, sont probablement aussi cancérogènes pour l'Homme. Ces composés font apparaître des tumeurs non seulement au point de contact, mais aussi à distance de ce point. Le pouvoir cancérogène peut varier avec la voie d'administration. On a proposé plusieurs méthodes pour évaluer le risque associé à une exposition à ces composés, seuls ou en mélange. Sans se prononcer en faveur de telle ou telle méthode, la monographie indique les données nécessaires, les hypothèses et les conditions de validité etc.

pour trois d'entre elles qui ont été plus ou moins validées en vue d'une évaluation quantitative du risque.

1.8 Effets sur l'Homme

Les HAP sont présents dans l'environnement et sur les lieux de travail dans des conditions d'une telle complexité que pour étudier l'exposition humaine à chacun de ces composés à l'état pur, on s'est limité à des expériences sur des volontaires, sauf dans le cas du naphtalène que l'on utilise comme anti-mite sur les vêtements.

L'application cutanée d'anthracène, de fluoranthène et de phénanthrène provoque des réaction cutanées spécifiques et le benzo(*a*)-pyrène donne naissance à des verrues régressives et réversibles que l'on a classées comme étant de nature néoplasique. On connaît les effets généraux du naphtalène en raison des nombreux cas d'ingestion accidentelle auxquels il a donné lieu, notamment chez des enfants. La dose létale est de 5 000 à 15 000 mg pour un adulte et de 2 000 mg sur deux jours pour un enfant. Après contact cutané ou ingestion, l'intoxication se caractérise par une anémie hémolytique aiguë, qui peut également toucher le foetus par la voie transplacentaire.

On sait que le tabagisme est la cause la plus importante de cancers du poumon et qu'il accroît également l'incidence des tumeurs de la vessie, du bassinet du rein, de la cavité buccale, du pharynx, du larynx et de l'oesophage. On estime en revanche que les HAP présents dans les denrées alimentaires ne jouent pas un rôle important dans l'apparition des cancers chez l'Homme. Dans les zones très industrialisées, on a observé un accroissement de la charge de l'organisme en HAP par suite de la pollution de l'air ambiant. Les malades dont on traite le psoriasis par des applications de goudron, sont également exposés à des HAP.

C'est en 1775 que l'on a, pour la première fois, avancé que l'exposition professionnelle à la suie était une cause de cancer du scrotum. Par la suite, on a remarqué que l'exposition aux goudrons et aux paraffines provoquait des cancers de la peau. Le poumon est désormais la principale localisation des cancers dus aux HAP, les cancers cutanés étant devenus plus rares par suite des progrès de l'hygiène individuelle.

On a effectué des études épidémiologiques sur l'exposition aux HAP chez des ouvriers de fours à coke pendant la cokéfaction et la gazéification de la houille, chez des ouvriers d'ateliers de préparation d'asphalte, de fonderies, de fours à aluminium ou encore chez des travailleurs exposés aux gaz d'échappement de moteurs diesel. On a constaté une augmentation des tumeurs pulmonaires dues aux HAP chez les ouvriers des fours à coke et des ateliers de préparation de l'asphalte, de même que chez ceux qui travaillaient près des cuves de réduction électrolytique de l'alumine par le procédé Söderberg. C'est chez les ouvriers des fours à coke que l'on a mis en évidence le risque le plus élevé, avec un rapport comparatif de mortalité (SMR) de 195.Plusieurs études ont permis d'établir des relations dose-réponse. Dans les usine d'aluminium, on a observé non seulement des cancers de la vessie, mais aussi des symptômes asthmatiformes, des anomalies de la fonction pulmonaire et des bronchites chroniques. Chez les ouvriers des fours à coke, il y avait diminution des taux d'immuno-globulines sériques et dépression des fonctions immunitaires. Par ailleurs, on a signalé l'apparition d'une cataracte après cinq ans d'exposition au naphtalène.

On a mis au point un certain nombre de méthodes pour évaluer l'exposition interne aux HAP. Dans la plupart de ces études, on a procédé au dosage des métabolites urinaires de ces composés: thioéthers, 1-naphtol, β-naphtylamine, hydroxyphénantrènes et 1-hydroxypyrène. Ce dernier composé est largement utilisé comme indicateur biologique d'une exposition à des HAP.

Les effets génotoxiques des HAP ont été étudiés en recherchant la présence de substances mutagènes dans les urines et les matières fécales et celle de micronoyaux, d'aberrations chromosomiques et d'échanges entre chromatides-soeurs dans les lymphocytes du sang périphérique. En outre, on a dosé les adduits du benzo(*a*)pyrène et de l'ADN dans les lymphocytes du sang périphérique et dans d'autres tissus, ainsi que ceux que cet HAP forme avec des protéines comme l'albumine. On a également procédé au dosage des anticorps dirigés contre les adduits de l'ADN.

Plusieurs études se sont donné pour but de rechercher s'il était possible d'utiliser la présence de 1-hydroxypyrène dans l'urine et d'adduits de l'ADN dans les lymphocytes comme marqueurs d'une

exposition à des HAP. Il est plus facile de doser le 1-hydroxypyrène urinaire que les adduits de l'ADN. Ce marqueur a en outre l'avantage d'être moins sujet aux variations interindividuelles et de permettre de repérer des expositions plus faibles. Les deux types de marqueurs ont été utilisés pour évaluer l'exposition humaine dans divers environnements. Ainsi, on a relevé une augmentation de la concentration urinaire de 1-hydroxypyrène sur les divers lieux de travail de cokeries, d'usines de production d'aluminium, d'ateliers d'imprégnation du bois, de fonderies et d'ateliers de préparation d'asphalte. L'exposition la plus forte a été observée chez les ouvriers des fours à coke et chez ceux qui travaillent à l'imprégnation du bois avec du créosote. Chez ces travailleurs, l'exposition est à 95% percutanée, alors que dans le reste de la population, les HAP sont absorbés principalement par la voie alimentaire et par la voie respiratoire lors de la consommation de tabac.

L'estimation du risque associé à une exposition à des HAP repose sur l'évaluation de cette exposition et sur les résultats des études épidémiologiques. En ce qui concerne les ouvriers des fours à coke, on aboutit à un risque relatif de cancer du poumon égal à 15,7. En procédant de la même manière pour la population générale, on trouve que le risque, pour un individu donné, de faire un cancer du poumon au cours de son existence est de 10^{-4} à 10^{-5} par ng de benzo(*a*)pyrène par m^3 d'air. En d'autres termes, il y a environ une personne sur 10 000 ou 100 000 qui va faire un cancer au cours de sa vie par suite de la présence de benzo(*a*)pyrène dans l'air ambiant.

1.9 Effets sur les autres êtres vivants au laboratoire et dans leur milieu naturel

S'il y a absorption simultanée de lumière UV ou de lumière visible, les HAP peuvent provoquer des effets toxiques aigus sur les poissons et sur des invertébrés aquatiques comme les daphnies. La toxicité des HAP peut être modifiée par dégradation et métabolisation. A faible concentration, les HAP peuvent stimuler la croissance des microorganismes et des algues. Le composé le plus toxique pour les algues est le benz(*a*)anthracène, un hydrocarbure tétracyclique. La valeur de la CE_{50} pour ce composé (réduction de 50% des paramètres vitaux) est égale à 1-29 µg/litre. Dans le cas du benzo(*a*)pyrène, un composé pentacyclique, elle est égale à 5-15 µg/litre. Toujours en ce

qui concerne les algues, la CE_{50} pour la plupart des HAP tricycliques est égale à 240-290 µg/litre. Le naphtalène, qui est bicyclique, est le moins toxique, avec une CE_{50} de 2 800-34 000 µg/litre.

Il n'y a pas de différence de sensibilité bien nette entre les divers groupes taxonomiques d'invertébrés tels que les crustacés, les insectes, les mollusques, les polychètes et les échinodermes. Le naphtalène est le moins toxique avec une CL_{50} à 96 h de 100 à 2 300 µg/litre. Pour trois HAP tricycliques, la valeur de ce même paramètre varie de <1 à 3 000 µg/litre. L'anthracène pourrait être plus toxique que les autres HAP tricycliques avec une CL_{50} à 24 h comprise entre <1 et 260 µg/litre. Pour les composés à quatre, cinq et six noyaux aromatiques, la CL_{50} à 96 h est comprise entre 0,2 et 1 200 µg/litre. Des effets toxiques aigus (CL_{50}) ont été observés chez des poissons à des concentrations de 110 à > 10 000 µg/litre de naphtalène, de 30 à 40 000 µg/litre d'HAP tricycliques (anthracène, 2,8-360 µg/litre) et de 0,7 à 26 µg/litre d'HAP tétra- et pentacycliques.

Chez des poissons vivant à l'état sauvage, on a attribué la présence de tumeurs hépatiques à la contamination des sédiments par des HAP à la concentration de 250 mg/kg. On a également provoqué l'apparition de tumeurs chez des poissons par exposition en laboratoire. L'exposition des poissons à certains HAP peut aussi provoquer chez eux des anomalies physiologiques et perturber leur croissance, leur reproduction, leur locomotion et leur respiration.

1. RESUMEN

1.1 Selección de compuestos para esta monografía

Los hidrocarburos aromáticos policíclicos (HAP) constituyen una clase muy amplia de compuestos, y durante la combustión incompleta o la pirolisis de materia orgánica pueden liberarse cientos de sustancias distintas, que son una fuente importante de exposición humana. Los estudios de diversas matrices aplicables al medio ambiente, como los efluentes de la combustión de carbón, los gases de escape de los vehículos de motor, el aceite lubricante de motores usados y el humo del tabaco, han demostrado que los HAP de esas mezclas son los principales responsables de su potencial carcinogénico.

En las mezclas casi siempre hay presentes HAP. Debido a que la composición de tales muestras es compleja y varía con el proceso de formación, no es posible examinar con detalle en la presente monografía todas las mezclas que contienen HAP. Así pues, se seleccionaron 33 compuestos distintos (31 HAP originales y dos derivados alquilo) para evaluarlos tomando como base la disponibilidad de datos pertinentes sobre los efectos finales toxicológicos y/o la exposición (Cuadro 1). Sin embargo, dado que sólo se disponía para las mezclas de estudios epidemiológicos, que son imprescindibles para la evaluación del riesgo, en las Secciones 8 y 10 se presentan los resultados de estudios de mezclas de HAP, en contraposición con el resto de la monografía.

Se han publicado numerosos artículos y reseñas sobre la presencia, distribución y transformación de HAP en el medio ambiente y sobre sus efectos ecotoxicológicos y toxicológicos. Solamente se citan en la presente monografía referencias de los 10-15 últimos años, a menos que no se dispusiera de otra información; en relación con los estudios más antiguos y con otra información se citan reseñas.

1.2 Identidad, propiedades físicas y químicas y métodos analíticos

El término "hidrocarburos aromáticos policíclicos" se refiere en general a una clase muy amplia de compuestos orgánicos que

Cuadro 1. Hidrocarburos aromáticos policíclicos evaluados en esta monografía

Nombre común	Nombre CAS	Sinónimo[a]	Nº de registro CAS
Acenaftileno	Acenaftileno		91-20-3
Acenafteno	Acenaftileno, 1,2-dihidro-		208-96-8
Antantreno	Dibenzo[*def*,*mno*]criseno		191-26-4
Antraceno	Antraceno		120-12-7
Benz[*a*]antraceno	Benz[*a*]antraceno	1,2-Benzantraceno, tetrafeno	56-55-3
Benzo[*a*]fluoreno	11*H*-Benzo[*a*]fluoreno	1,2-Benzofluoreno	238-84-6
Benzo[*b*]fluoreno	11*H*-Benzo[*b*]fluoreno	2,3-Benzofluoreno	243-17-4
Benzo[*b*]fluoranteno	Benz[*e*]acefenantrileno	3,4-Benzofluoranteno	205-99-2
Benzo[*ghi*]fluoranteno	Benzo[*ghi*]fluoranteno	2,13-Benzofluoranteno	203-12-3
Benzo[*j*]fluoranteno	Benzo[*j*]fluoranteno	10,11-Benzofluoranteno	205-82-3
Benzo[*k*]fluoranteno	Benzo[*k*]fluoranteno	11,12-Benzofluoranteno	207-08-9
Benzo[*ghi*]perileno	Benzo[*ghi*]perileno	1,12-Benzoperileno	191-24-2
Benzo[*c*]fenantreno	Benzo[*c*]fenantreno	3,4-Benzofenantreno	195-19-7
Benzo[*a*]pireno	Benzo[*a*]pireno	3,4-Benzopireno[b]	50-32-8
Benzo[*e*]pireno	Benzo[*e*]pireno	1,2-Benzopireno	192-97-2
Criseno	Criseno	1,2-Benzofenantreno	218-01-9
Coroneno	Coroneno	Hexabenzobenceno	191-07-1
Ciclopenta[*cd*]pireno	Ciclopenta[*cd*]pireno	Ciclopenteno[*cd*]pireno	27208-37-3

Cuadro 1 (sigue).

Dibenz[*a,h*]antraceno	Dibenz[*a,h*]antraceno	1,2:5,6-Dibenzantraceno	53-70-3
Dibenzo[*a,e*]pireno	Naftol[1,2:3,4-*def*]criseno	1,2:4,5-Dibenzopireno	192-65-4
Dibenzo[*a,h*]pireno	Dibenzo[*b,def*]criseno	3,4:8,9-Dibenzopireno	189-64-0
Dibenzo[*a,i*]pireno	Benzo[*rst*]pentafeno	3,4:9,10-Dibenzopireno	189-55-9
Dibenzo[*a,l*]pireno	Dibenzo[*def,p*]criseno	1,2:3,4-Dibenzopireno	191-30-0
Fluoranteno	Fluoranteno		206-44-0
Fluoreno	9*H*-Fluoreno		86-73-7
Indeno[1,2,3-*cd*]pireno	Indeno[1,2,3-*cd*]pireno	2,3-*o*-Fenilenpireno	193-39-5
5-Metilcriseno	Criseno, 5-metil-		3697-24-3
1-Metilfenantreno	Fenantreno, 1-metil-		832-69-9
Naftaleno	Naftaleno		91-20-3
Perileno	Perileno	*peri*-Dinaftaleno	198-55-0
Fenantreno	Fenantreno		85-01-8
Pireno	Pireno	Benzo[*def*]fenantreno	129-00-0
Trifenileno	Trifenileno	9,10-Benzofenantreno	217-59-4

Han notificado listas muy amplias de sinónimos el CIIC (1983) y Loening y Merritt (1990).
[a]Sinónimo común que aparece en la bibliografía.
[b]También denominado benzo[del]criseno.

contienen dos o más anillos aromáticos condensados formados por átomos de carbono y de hidrógeno. A temperatura ambiente, los HAP son sólidos. Las características comunes de la clase son puntos de fusión y de ebullición elevados, presión de vapor baja y solubilidad en agua muy baja, que tiende a disminuir con el aumento del peso molecular. Los HAP son solubles en muchos disolventes orgánicos y muy lipófilos. Desde el punto de vista químico son bastante inertes. Las reacciones que tienen interés con respecto a su destino en el medio ambiente y a las posibles fuentes de pérdidas durante el muestreo atmosférico son la fotodescomposición y las reacciones con óxidos de nitrógeno, ácido nítrico, óxidos de azufre, ácido sulfúrico, ozono y radicales hidroxilo.

El aire del ambiente se muestrea recogiendo partículas suspendidas en filtros de fibra de vidrio, de politetrafluoroetileno o de fibra de cuarzo mediante muestreadores de alto volumen o pasivos. Los HAP en fase de vapor, que podrían volatilizarse de los filtros durante el muestreo, se suelen retener por adsorción en espuma de poliuretano. La fase de muestreo es con diferencia la fuente más importante de variabilidad de los resultados.

En el lugar de trabajo se toman muestras con tasas de flujo bajas; se recogen las partículas en filtros de fibra de vidrio o de politetrafluoroetileno y los vapores en resina XAD-2 de amberlita. Los dispositivos para el muestreo de gases de chimenea constan de un filtro de fibra de vidrio o de fibra de cuarzo en la parte frontal de un refrigerador para recoger la materia condensable y un cartucho adsorbente (por lo general XAD-2). Las muestras de gases de escape de los vehículos se toman en condiciones de laboratorio, con ciclos de conducción normalizados que simulan las condiciones en carretera. Las emisiones se recogen sin diluir o bien una vez diluidas con aire frío filtrado.

Se han descrito numerosas técnicas de extracción y purificación. En función de la matriz, los HAP se extraen de las muestras con un aparato de Soxhlet, por ultrasonidos, mediante reparto líquido-líquido o, tras la disolución o la digestión alcalina de la muestra, con un disolvente selectivo. También se ha utilizado la extracción de fluidos supercríticos a partir de diversos sólidos del medio ambiente. La eficacia de la extracción depende en gran medida del disolvente

utilizado, y muchos de los que se utilizaban en el pasado no eran apropiados. Las muestras extraídas se suelen purificar por cromatografía en columna, en particular sobre alúmina, silicagel o Sephadex LH-20, pero también por cromatografía en capa fina.

La identificación y la cuantificación se realizan habitualmente mediante cromatografía de gases con detección por ionización de llama o mediante cromatografía líquida de alta resolución (HPLC) con detección por ultravioleta o fluorescencia, generalmente en serie. En la cromatografía de gases se utilizan columnas capilares de sílice fundido con polisiloxanos (SE-54 y SE-52) como fases estacionarias; en la HPLC se suelen utilizar columnas de sílice-C18. Con frecuencia se acopla un detector de espectrometría de masas a una columna de fase gaseosa a fin de confirmar la identidad de los picos.

La elección de los HAP que se han de determinar depende de la finalidad de la medición, por ejemplo para estudios orientados a la salud o ecotoxicológicos, o bien para investigar las fuentes. Es posible que se exija o se recomiende la realización de pruebas para distintos conjuntos de compuestos a nivel nacional e internacional.

1.3 Fuentes de exposición humana y ambiental

Es poca la información disponible acerca de la producción y elaboración de HAP, pero es probable que sólo se desprendan pequeñas cantidades como resultado directo de esas actividades. Los HAP que se encuentran se utilizan sobre todo como productos intermedios en la producción de cloruro de polivinilo y de agentes plastificantes (naftaleno), pigmentos (acenafteno, pireno), tintes (antraceno, fluoranteno) y plaguicidas (fenantreno).

Las mayores emisiones de HAP se derivan de la combustión incompleta de materia orgánica durante procesos industriales y en otras actividades humanas, en particular:

— elaboración de carbón, de petróleo crudo y de gas natural, incluida la coquificación de carbón, la conversión de carbón, el refinado de petróleo y la producción de negro de humo, de creosota, de alquitrán de hulla y de betún;
— producción de aluminio, de hierro y de acero en fábricas y fundiciones;

— calefacción en centrales de energía y en residencias y cocinado;
— combustión de basuras;
— tráfico de vehículos de motor; y
— humo de tabaco en el medio ambiente.

Los HAP, especialmente los de mayor peso molecular, cuando se incorporan al medio ambiente a través de la atmósfera se adsorben en las partículas en suspensión. La hidrosfera y la geosfera se ven afectadas de manera secundaria por la deposición húmeda y seca. La madera conservada con creosota es otra fuente de liberación de HAP en la hidrosfera, y la deposición de desechos contaminados, como fangos de alcantarillado y cenizas en suspensión, contribuye a las emisiones de HAP en la geosfera. Hay poca información acerca del paso de HAP a la biosfera. Hay HAP presentes naturalmente en la turba, el lignito, el carbón y el petróleo crudo. La mayoría de los HAP de las antracitas están fuertemente unidos a la estructura del carbón y no se pueden lixiviar.

La liberación de HAP en el medio ambiente se ha determinado mediante la identificación de un perfil característico de su concentración, pero esto sólo ha sido posible en un pequeño número de casos. Con frecuencia se ha utilizado el benzo[*a*]pireno como indicador de HAP, especialmente en estudios más antiguos. En general, las emisiones de HAP son solamente estimaciones basadas en datos más o menos fidedignos y apenas dan una idea general de la exposición.

Las fuentes más importantes de HAP son las siguientes:

Coquificación de carbón: Las emisiones de HAP en suspensión en el aire procedentes de la coquificación de carbón en Alemania han disminuido considerablemente en los 10 últimos años gracias a las mejoras técnicas de las instalaciones existentes, al cierre de otras antiguas y a la menor producción de coque. Se supone que la situación es análoga en el resto de Europa occidental, el Japón y los Estados Unidos, pero no se dispone de datos.

Producción de aluminio (principalmente en ánodos especiales de carbón), *de hierro y de acero* y los aglutinantes utilizados en la arena de moldeo de las *fundiciones*: La información disponible es escasa.

Cocinas y calefacción de viviendas: Los principales componentes que se emiten son fenantreno, fluoranteno, pireno y criseno. Las emisiones de los hornillos de leña son 25-1000 veces superiores a las que se producen en los de carbón, y en las zonas donde predomina el uso de leña en las viviendas la mayor proporción de HAP en suspensión puede derivarse de esta fuente, especialmente en invierno. Por consiguiente, se supone que la liberación de HAP en la calefacción de las viviendas es una fuente importante en los países en desarrollo, donde con frecuencia se quema biomasa en hornillos relativamente simples.

Cocinado: Pueden emitirse HAP durante la combustión incompleta de los combustibles, del aceite de cocinar y de los alimentos que se cocinan.

Tráfico de vehículos de motor: Los principales compuestos que se liberan de los vehículos de gasolina son el fluoranteno y el pireno, mientras que en los gases de escape de los vehículos de motor diesel abundan el naftaleno y el acenafteno. Aunque los motores de gasolina emiten una proporción elevada de ciclopenta[*cd*]pireno, su concentración en los gases de escape de los motores diésel está apenas por encima del límite de detección. Las tasas de emisión, que dependen de la sustancia, el tipo de vehículo, el estado de su motor y las condiciones de la prueba, oscilan entre unos pocos nanogramos por kilómetro y > 1000 mg/km. Las emisiones de HAP de los vehículos de motor se reducen enormemente con la instalación de catalizadores.

Incendios forestales: En los países con grandes superficies forestales, los incendios pueden contribuir de manera importante a las emisiones de HAP.

Centrales eléctricas de carbón: Los HAP que se liberan en la atmósfera a partir de dichas centrales son sobre todo compuestos de dos y tres anillos. En las zonas contaminadas, los niveles de HAP en el aire pueden ser más elevados que los de los gases de chimenea.

Incineración de basuras: Las emisiones de HAP en los gases procedentes de este tipo de incineración fueron en varios países < 10 mg/m^3.

1.4 Transporte, distribución y transformación en el medio ambiente

Son varios los factores de distribución y de transformación de los que depende el destino tanto de los HAP por separado como de las mezclas. Los procesos de distribución más importantes son el reparto entre el agua y el aire, entre el agua y los sedimentos y entre el agua y la biota.

Puesto que los HAP son hidrófobos, con escasa solubilidad en el agua, su afinidad por la fase acuática es muy pequeña; sin embargo, a pesar del hecho de que la mayoría de los HAP se liberan en el medio ambiente a través de la atmósfera, también se encuentran concentraciones considerables en la hidrosfera, debido a sus bajas constantes de la ley de Henry. Como la afinidad de los HAP por las fases orgánicas es mayor que la que tienen por el agua, sus coeficientes de reparto entre disolventes orgánicos como el octanol y el agua son elevados. Su afinidad por las fracciones orgánicas de los sedimentos, del suelo y de la biota es también alta, por lo que se acumulan en los organismos del agua y los sedimentos y en sus alimentos. No se conoce con claridad la importancia relativa de su ingesta con los alimentos y el agua. En *Daphnia* y en los moluscos, hay una correlación positiva entre la acumulación de HAP procedentes del agua y el coeficiente de reparto octanol:agua (K_{ow}). Sin embargo, en los peces y las algas capaces de metabolizar los HAP no hay correlación entre las concentraciones internas de distintos HAP y el K_{ow}.

No se ha observado bioamplificación -aumento de la concentración de una sustancia en animales de niveles tróficos sucesivos de las cadenas alimentarias- en sistemas acuáticos y no cabe prever que se produzca, puesto que la mayoría de los organismos tienen un potencial de biotransformación elevado para los HAP. El mayor potencial de biotransformación se observa en los organismos de los niveles tróficos más altos de las cadenas alimentarias.

Los HAP se descomponen por fotodegradación, biodegradación por microorganismos y metabolismo en la biota de niveles más altos. Aunque la última ruta de transformación tiene escasa importancia para el destino global de los HAP en el medio ambiente, es una vía importante para la biota, debido a que pueden formarse metabolitos carcinogénicos. Dado que los HAP son químicamente estables, sin grupos reactivos, la hidrólisis no interviene en su degradación. Hay pocas pruebas normalizadas para la biodegradación de los HAP. En general, se biodegradan en condiciones aerobias, registrando un fuerte aumento la tasa de biodegradación con el número de anillos aromáticos. En condiciones anaerobias la degradación es mucho más lenta.

Los HAP se fotooxidan en el aire y en el agua en presencia de radicales sensibilizantes como OH, NO_3 y O_3. En condiciones de laboratorio, la semivida de la reacción con radicales OH presentes en el aire es de alrededor de un día, mientras que las reacciones con NO_3 y O_3 suelen tener unas constantes de velocidad mucho más bajas. La adsorción de HAP de peso molecular alto en partículas carbonosas en el medio ambiente debería estabilizar la reacción con radicales OH. La reacción, que tiene lugar sobre todo en la fase gaseosa, de HAP de entre dos y cuatro anillos con NO_3 da lugar a la formación de nitro-HAP, que son conocidos como mutágenos. Parece que la fotooxidación de algunos HAP en el agua es más rápida que en el aire. Los cálculos basados en parámetros fisicoquímicos y de degradación indican que los HAP con cuatro o más anillos aromáticos persisten en el medio ambiente.

1.5 Niveles ambientales y exposición humana

Los HAP están omnipresentes en el medio ambiente, habiéndose detectado en numerosos estudios diversos HAP por separado en distintos compartimentos.

1.5.1 Aire

Los niveles de cada uno de los HAP tienden a ser más elevados en invierno que en verano por lo menos en un orden de magnitud. La fuente predominante durante el invierno es la calefacción de las viviendas y la del verano el tráfico urbano de vehículos de motor. Se

detectaron concentraciones medias de distintos HAP de 1-30 ng/m^3 en la atmósfera de diversas zonas urbanas. En grandes ciudades con un tráfico intenso de vehículos de motor y utilización abundante de combustibles a base de biomasa, como Calcuta, se encontraron niveles de hasta 200 ng/m^3 de distintos HAP. En túneles de carreteras se detectaron concentraciones de hasta 1-50 ng/m^3. Había ciclopenta[*cd*]-pireno y pireno en concentraciones de hasta 100 ng/m^3. En una estación de metro se midieron concentraciones de HAP de hasta 20 ng/m^3. En las cercanías de fuentes industriales, las concentraciones medias de los distintos HAP oscilaban entre 1 y 10 ng/m^3. Había fenantreno presente hasta un máximo aproximado de 310 ng/m^3.

Los valores básicos de los HAP son por lo menos uno o dos órdenes de magnitud menores que los obtenidos cerca de fuentes como el tráfico de vehículos de motor. Por ejemplo, los niveles a 1100 m oscilaban entre 0,004 y 0,03 ng/m^3.

1.5.2 Agua superficial y precipitación

La mayor parte de los HAP presentes en el agua proceden al parecer del agua de escorrentía urbana, de la precipitación atmosférica (partículas más pequeñas) y de la abrasión del asfalto (partículas mayores). La principal fuente de HAP, sin embargo, varía para distintas masas de agua. En general, la mayoría de las muestras de agua superficial contienen distintos HAP en concentraciones de hasta 50 ng/litro, pero los ríos muy contaminados tenían concentraciones de hasta 6000 ng/litro. Los niveles de HAP en el agua freática son del orden de 0,02-1,8 ng/litro, y las muestras de agua potable contienen concentraciones del mismo orden de magnitud. Las principales fuentes de los HAP presentes en el agua potable son los depósitos y las tuberías con revestimiento de asfalto.

Los niveles de HAP por separado en el agua de lluvia oscilaban entre 10 y 200 ng/litro, mientras que en la nieve y la niebla se detectaron concentraciones de hasta 1000 ng/litro.

1.5.3 Sedimentos

Las concentraciones de los distintos HAP en los sedimentos eran por lo general un orden de magnitud superiores a las presentes en la precipitación.

1.5.4 Suelo

Las principales fuentes de los HAP presentes en el suelo son la deposición atmosférica, la carbonización de material vegetal y la deposición a partir de aguas residuales y desechos particulados. El grado de contaminación del suelo depende de factores como su cultivo, su porosidad y su contenido de sustancias húmicas.

Cerca de fuentes industriales se han encontrado concentraciones de distintos HAP de hasta 1 g/kg de suelo. La concentración en el suelo a partir de otras fuentes, como los gases de escape de los automóviles, son del orden de 2-5 mg/kg. En zonas no contaminadas, los niveles de HAP eran de 5-100 µg/kg de suelo.

1.5.5 Alimentos

Los alimentos crudos no suelen contener niveles elevados de HAP, pero se forman al elaborarlos, asarlos o cocerlos en horno o freírlos. Las hortalizas pueden contaminarse por la deposición de partículas de la atmósfera o por el crecimiento en suelo contaminado. Las concentraciones de los distintos HAP en la carne, el pescado, los productos lácteos, las frutas y hortalizas, los cereales y sus productos, los dulces, las bebidas y las grasas y los aceites animales y vegetales eran del orden de 0,01-10 µg/kg. Se han detectado concentraciones de más de 100 µg/kg en carne ahumada y de hasta 86 µg/kg en pescado ahumado; los cereales ahumados contenían hasta 160 µg/kg. En el aceite de coco se encontraron concentraciones de hasta 460 µg/kg. Los niveles en la leche materna humana eran de 0,003-0,03 µg/kg.

1.5.6 Organismos acuáticos

Se sabe que los organismos marinos adsorben y acumulan HAP del agua. El grado de contaminación depende del desarrollo industrial y urbano y de los movimientos de transporte marítimo. Se han

detectado concentraciones de HAP de hasta 7 mg/kg en organismos acuáticos que vivían cerca de efluentes industriales, y los niveles medios de HAP en los animales acuáticos muestreados en lugares contaminados fueron de 10-500 µg/kg, aunque también se detectaron niveles de hasta 5 mg/kg.

Los niveles medios de HAP en animales acuáticos muestreados en diversos lugares con fuentes sin especificar de dichos compuestos fueron de 1-100 µg/kg, pero se encontraron concentraciones de hasta 1 mg/kg, por ejemplo en langostas en el Canadá.

1.5.7 Organismos terrestres

Las concentraciones de HAP en insectos oscilaban entre 730 y 5500 µg/kg. El contenido de las heces de las lombrices de tierra depende considerablemente del lugar: las de una región muy industrializada de Alemania oriental contenían concentraciones de benzo[*a*]pireno de hasta 2 mg/kg.

1.5.8 Población general

Las principales fuentes de exposición no profesional son las siguientes: atmósfera contaminada, humo de fuegos abiertos y del cocinado, humo de tabaco en el medio ambiente, alimentos y agua potable contaminados y utilización de productos contaminados. Pueden encontrarse HAP en el aire de espacios cerrados como consecuencia de la calefacción de las viviendas y del humo del tabaco del medio ambiente en concentraciones medias de 1-100 ng/m^3, con un máximo de 2300 ng/m^3.

Se ha estimado que la ingesta de distintos HAP con los alimentos es de 0,10-10 µg/día por persona. La ingesta diaria total de benzo[*a*]-pireno con el agua potable se estimó en 0,0002 µg/persona. Los cereales y productos derivados son los que más contribuyen a la ingesta de HAP con los alimentos, por ser el principal componente de la alimentación total.

1.5.9 *Exposición profesional*

Cerca de una batería de hornos de coque, los niveles de benzo[*a*]-pireno oscilaban entre < 0,1 y 100-200 μg/m³, con un máximo aproximado de 400 μg/m³. En los sistemas modernos de gasificación del carbón, la concentración de HAP suele ser < 1 μg/m³, con un máximo de 30 μg/m³. Las muestras personales tomadas de operadores de equipo de refinerías de petróleo mostraron una exposición a 2,6-470 μg/m³. En muestras de aire tomadas cerca de instalaciones de elaboración de asfalto en refinerías, los niveles totales de HAP fueron de 0,004-50 μg/m³. En las proximidades de obras de pavimentación de carreteras, las concentraciones totales de HAP en muestras personales de aire eran de hasta 190 μg/m³, con un valor medio en las muestras de aire de la zona de 0,13 μg/m³. Los niveles de HAP en las muestras personales de aire tomadas en una fundición de aluminio eran de 0,05-9,6 μg/m³, pero las muestras de orina de los trabajadores de una fábrica de aluminio contenían niveles muy bajos. Las muestras de aire de la zona contenían concentraciones de HAP de hasta 5 μg/m³ en una fundición alemana, 3-40 μg/m³ en minas de hierro μg/m³ y 4-530 μg/m³ en minas de cobre. Las concentraciones de HAP en humos de cocinado en una fábrica de productos alimenticios oscilaban entre 0,07 y 26 μg/m³.

1.6 Cinética y metabolismo

Los HAP se absorben por las vías respiratorias, el aparato digestivo y la piel. La tasa de absorción por los pulmones depende del tipo de HAP, el tamaño de las partículas sobre las que están adsorbidos y la composición del adsorbente. Los HAP adsorbidos sobre partículas se eliminan de los pulmones con mayor lentitud que los hidrocarburos libres. En el aparato digestivo se produce una absorción rápida en los roedores, pero los metabolitos vuelven al intestino mediante la excreción biliar. En estudios de absorción percutánea de mezclas de HAP marcados con ^{32}P en roedores se observó que los componentes de las mezclas llegaban a los pulmones, donde se unían al ADN. La tasa de absorción percutánea en ratones varía en función del compuesto.

Los HAP se distribuyen ampliamente en todo el organismo tras la administración por cualquier vía y se encuentran en casi todos los

órganos internos, particularmente en los ricos en lípidos. Los HAP inyectados por vía intravenosa se eliminan con rapidez de la corriente sanguínea en los roedores, pero pueden atravesar la barrera placentaria y se han detectado en tejidos fetales.

El metabolismo de los HAP es complejo. En general, los compuestos originales se convierten mediante epóxidos intermedios en fenoles, dioles y tetroles, que a su vez pueden conjugarse con los ácidos sulfúrico y glucurónico o con el glutatión. El metabolismo produce en su mayor parte una desintoxicación, pero algunos HAP se activan a especies que se unen al ADN, principalmente diolepóxidos, que pueden inducir tumores. Los metabolitos de los HAP y sus conjugados se excretan en la orina y las heces, pero los conjugados que se excretan en la bilis pueden hidrolizarse por la acción de enzimas de la flora intestinal y reabsorberse. De la información disponible acerca de la carga total en el cuerpo humano cabe deducir que los HAP no persisten en el organismo y que su ciclo metabólico es rápido. De la deducción anterior están excluidos los grupos de HAP que se unen por enlaces covalentes a elementos constitutivos de los tejidos, en particular ácidos nucleicos, y que no se eliminan por reparación.

1.7 Efectos en mamíferos de laboratorio y en sistemas de prueba *in vitro*

La toxicidad aguda de los HAP parece ser de moderada a baja. Un producto bien caracterizado, el naftaleno, mostró valores para la DL_{50} por vía oral e intravenosa de 100-500 mg/kg de peso corporal en ratones y una DL_{50} media por vía oral de 2700 mg/kg de peso corporal en ratas. Los valores para otros HAP son semejantes. Con dosis altas únicas de naftaleno se indujo en ratones, ratas y hámsteres necrosis bronquiolar.

En estudios de corta duración se pusieron de manifiesto efectos hematológicos adversos en forma de mielotoxicidad con el benzo[*a*]pireno, cambios hemolinfáticos con el dibenz[*a,h*]antraceno y anemia con el naftaleno; sin embargo, en un estudio de siete días en el que se administró a ratones naftaleno por vía oral e intraperitoneal se observó tolerancia al efecto de este producto.

Sólo raramente se han descrito efectos sistémicos provocados por un tratamiento prolongado con HAP, porque el efecto final de la mayor parte de los estudios ha sido la carcinogenicidad. Se manifiestan efectos tóxicos importantes a dosis en las cuales se desencadenan también respuestas carcinogénicas

En estudios de los efectos adversos en la piel tras la aplicación cutánea, los productos con carcinogenicidad nula o débil, como el perileno, el benzo[*e*]pireno, el fenantreno, el pireno, el antraceno, el acenaftaleno, el fluoreno y el fluoranteno, fueron inactivos, mientras que los compuestos carcinógenos, como el benz[*a*]antraceno, el dibenz[*a,h*]antraceno y el benzo[*a*]pireno provocaron hiperqueratosis. Los vapores de antraceno y de naftaleno provocaron irritación ocular leve. El benzo[*a*]pireno indujo hipersensibilidad por contacto en cobayas y ratones.

El benz[*a*]antraceno, el benzo[*a*]pireno, el dibenz[*a,h*]antraceno y el naftaleno fueron embriotóxicos para ratones y ratas. El benzo[*a*]-pireno mostró asimismo teratogenicidad y efectos en la reproducción. Se han realizado grandes esfuerzos para aclarar la base genética de los efectos embriotóxicos del benzo[*a*]pireno. Sólo se observan mortalidad fetal y malformaciones si es inducible el sistema citocromo P-450 monooxigenasa, bien en la madre (con permigración placentaria) o bien en el embrión. No todos los efectos observados se pueden explicar por la predisposición genética, pero en ratones y conejos el benzo[*a*]pireno mostró actividad carcinogénica transplacentaria, produciendo adenomas pulmonares y papilomas cutáneos en la descendencia. Se observó asimismo reducción de la fecundidad y destrucción de oocitos.

Se han estudiado también ampliamente los HAP en valoraciones de la genotoxicidad y de la transformación celular; la mayor parte de los 33 HAP comprendidos en la presente monografía son genotóxicos o probablemente genotóxicos. Los únicos compuestos para los cuales se obtuvieron resultados negativos en todas las valoraciones fueron el antraceno, el fluoreno y el naftaleno. Habida cuenta de la falta de uniformidad de los resultados, el fenantreno y el pireno no podrían clasificarse de manera fidedigna como genotóxicos.

En un trabajo amplio sobre la carcinogenicidad de los HAP se ha puesto de manifiesto que 26 de los 33 productos estudiados son, o se sospecha que son, carcinogénicos (Cuadro 2). El compuesto mejor caracterizado es el benzo[*a*]pireno, que se ha estudiado aplicando todos los métodos actuales en siete especies. Los HAP que han sido objeto de 12 estudios o más son el antantreno, el antraceno, el benz[*a*]-antraceno, el criseno, el dibenz[*a,h*]antraceno, el dibenzo[*a,i*]pireno, el 5-metilcriseno, el fenantreno y el pireno. Los estudios especiales de fototoxicidad, inmunotoxicidad y hepatotoxicidad de los HAP se complementan con informes sobre la toxicidad ocular del naftaleno.

Cuadro 2. Resumen de los resultados de las pruebas de genotoxicidad y carcinogenicidad de los 33 hidrocarburos aromáticos policíclicos estudiados

Compuesto	Genotoxicidad	Carcinogenicidad
Acenafteno	(?)	(?)
Acenaftileno	(?)	No hay estudios
Antraceno	-	-
Benz[*a*]antraceno	+	+
Benzo[*a*]fluoreno	(?)	(?)
Benzo[*a*]pireno	+	+
Benzo[*b*]fluoranteno	+	+
Benzo[*b*]fluoreno	(?)	(?)
Benzo[c]fenantreno	(+)	+
Benzo[*e*]pireno	+	?
Benzo[*ghi*]fluoranteno	(+)	(-)
Benzo[*ghi*]perileno	+	-
Benzo[*j*]fluoranteno	+	+
Benzo[*k*]fluoranteno	+	+
Criseno	+	+
Coroneno	(+)	(?)
Ciclopenta[*cd*]pireno	+	+
Dibenzo[*a,e*]pireno	+	+
Dibenz[*a,h*]antraceno	+	+
Dibenzo[*a,h*]pireno	(+)	+
Dibenzo[*a,i*]pireno	+	+

Cuadro 2 (sigue).

Compuesto	Genotoxicidad	Carcinogenicidad
Dibenzo[*a*,*l*]pireno	(+)	+
Fluoranteno	+	(+)
Fluoreno	-	-
Indeno[1,2,3-*cd*]pireno	+	+
1-Metilfenantreno	+	-
5-Metilcriseno	+	+
Naftaleno	-	?
Perileno	+	-
Fenantreno	(?)	(?)
Pireno	(?)	-
Trifenileno	+	-

+, positivo; -, negativo; ?, dudoso
Paréntesis, resultado derivado de un número de datos pequeño

El antraceno, el benzo[*a*]pireno y algunos otros HAP mostraron fototoxicidad en la piel de mamíferos y en cultivos de células *in vitro* cuando se aplicaron con radiación ultravioleta. Se ha notificado que en general estos compuestos tienen efectos inmunosupresores. Tras la administración intraperitoneal de benzo[*a*]pireno a ratones, se manifestó una fuerte supresión de los parámetros inmunitarios en la descendencia durante un período de hasta 18 meses. Se observó asimismo una mayor regeneración hepática y un aumento del peso del hígado. El efecto del naftaleno en la aparición de cataratas en un roedor se ha atribuido a su capacidad de inducción del sistema del citocromo P-450 en estudios realizados con estirpes de ratones genéticamente diferentes.

Los modelos teóricos para pronosticar la actividad carcinogénica de los HAP a partir de sus estructuras, basados en una gran cantidad de trabajos experimentales, se presentaron ya en los años treinta. El primer modelo se basaba en la elevada reactividad química de determinados dobles enlaces (teoría de la región K). Más tarde se aplicó un método más sistemático, basado en la síntesis química de posibles metabolitos y en su actividad mutagénica. Según esta teoría de la región "bay", los epóxidos adyacentes a una región "bay" dan lugar a iones carbonio muy estables. Otros métodos teóricos son la "teoría de la región di" y la "teoría del potencial del radical catiónico".

Muchos de los distintos HAP son carcinógenos para los animales y pueden serlo para el ser humano y se ha observado que la exposición a varias mezclas con HAP aumenta la incidencia de cáncer en poblaciones humanas. Existe la preocupación de que los HAP cuya carcinogenicidad se ha demostrado en animales de experimentación puedan serlo también para el ser humano. Los HAP producen tumores, tanto en el lugar del contacto como en otros lejanos. Su actividad carcinogénica puede variar en función de la vía de exposición. Se han propuesto diversos métodos para evaluar el riesgo asociado a la exposición a estos productos aislados y en mezclas. En la presente monografía no se respalda ningún método; sin embargo, se describen las necesidades de datos, las hipótesis, la aplicabilidad y otras características de tres procesos de evaluación cuantitativa del riesgo que han sido validados en cierta medida.

1.8 Efectos en el ser humano

Habida cuenta de la complejidad del perfil de los HAP en el medio ambiente y en los lugares de trabajo, la exposición humana a productos puros por separado se ha limitado a experimentos científicos con voluntarios, salvo en el caso del naftaleno, que se utiliza como antipolilla para la ropa.

Tras la aplicación cutánea, el antraceno, el fluoranteno y el fenantreno indujeron reacciones específicas en la piel, y el benzo[*a*]-pireno produjo la formación de verrugas regresivas reversibles que se clasificaron como proliferaciones neoplásicas. Los efectos sistémicos del naftaleno se conocen por los numerosos casos de ingesta accidental, particularmente de niños. La dosis letal por vía oral es de 5000-15 000 mg para los adultos y 2000 mg ingeridos durante dos días en los niños. El efecto normal tras la exposición por vía cutánea u oral es una anemia hemolítica aguda que, a través de la placenta, puede afectar también a los fetos.

El humo del tabaco es el factor aislado más importante en la inducción de tumores de pulmón y también de un aumento de la incidencia de tumores de la vejiga urinaria, la pelvis renal, la boca, la faringe, la laringe y el esófago. No se considera que sea elevada la contribución de los HAP en los alimentos a la aparición de tumores humanos. En zonas fuertemente industrializadas se detectó también

una mayor carga corporal de HAP debido a la contaminación del aire. Están expuestos asimismo a estos compuestos los enfermos de psoriasis tratados con alquitrán de hulla.

La exposición profesional al hollín como causa de cáncer de escroto se observó por primera vez en 1775. Más tarde se informó que la exposición a los alquitranes y la parafina inducían cáncer cutáneo. Ahora el cáncer inducido por HAP afecta principalmente al pulmón, mientras que el cáncer cutáneo es más raro gracias a una mayor higiene personal.

Se han realizado estudios epidemiológicos de trabajadores expuestos en hornos de coque durante la coquificación del carbón y su gasificación, en obras de asfaltado, fundiciones e instalaciones de aluminio y a los gases de escape de los motores diésel. Se ha detectado un índice de cáncer de pulmón más elevado debido a su exposición a los HAP en los trabajadores de hornos de coque, los que utilizan asfalto y los de las salas de crisoles de Söderberg de las instalaciones de reducción de aluminio. El riesgo más elevado se observó en los trabajadores de hornos de coque, con una razón de mortalidad normalizada de 195. En varios estudios se hallaron relaciones dosis-respuesta. En las instalaciones de aluminio no sólo se detectó cáncer de la vejiga urinaria, sino también síntomas semejantes al asma, anomalías funcionales de los pulmones y bronquitis crónica. En los trabajadores de hornos de coque se observó asimismo una disminución de la concentración de inmunoglobulina en el suero y una reducción de la función inmunitaria. Se notificó que una exposición durante cinco años al naftaleno había provocado cataratas.

Se han elaborado varios métodos para evaluar la exposición interna a los HAP. En la mayoría de los estudios, se midieron en orina metabolitos de los HAP como los tioéteres urinarios, el 1-naftol, la β-naftilamina, los hidroxifenantrenos y el 1-hidroxipireno. Este último se ha utilizado con frecuencia como índice biológico de exposición.

Se han determinado los efectos genotóxicos de los HAP mediante pruebas de mutagenicidad en orina y heces y de la presencia de micronúcleos, aberraciones cromosómicas e intercambio de cromátidas hermanas en linfocitos de la sangre periférica. Además, se han medido aductos de benzo[*a*]pireno con ADN en linfocitos periféricos y en

otros tejidos y con proteínas como la albúmina, así como anticuerpos de aductos de ADN.

En varios estudios se han investigado el 1-hidroxipireno en orina y los aductos de ADN en linfocitos como marcadores. El primero se puede medir más fácilmente que los segundos, presenta menos variación entre los individuos y es posible detectar niveles de exposición más bajos. Ambos marcadores se han utilizado para evaluar la exposición humana en diversas condiciones. En varios puestos de trabajo de instalaciones de coque, de fabricación de aluminio, de instalaciones de impregnación de la madera, de fundiciones y de obras de asfaltado se observó un aumento de la excreción de 1-hidroxipireno o de aductos de ADN. Las exposiciones más elevadas se detectaron en los trabajadores de hornos de coque y en los de impregnación de la madera con creosota, que absorbían hasta el 95% del total de HAP a través de la piel, a diferencia de la población general, en la que predomina la absorción mediante los alimentos y el humo del tabaco.

Las estimaciones del riesgo relacionado con la exposición a HAP y sus mezclas se basan en estimaciones de la exposición y en los resultados de estudios epidemiológicos. Los datos obtenidos de trabajadores de hornos de coque pusieron de manifiesto un riesgo relativo de cáncer de pulmón de 15,7. Teniendo esto en cuenta, se ha calculado que el riesgo de aparición de cáncer de pulmón en la población general durante toda la vida es de 10^{-4} a 10^{-5} por ng de benzo[*a*]pireno por m^3 de aire. Dicho de otra manera, como consecuencia de la exposición al benzo[*a*]pireno en el aire cabría esperar la aparición de cáncer de pulmón a lo largo de toda la vida en una persona de cada 10 000 a 100 000.

1.9 Efectos en otros organismos en el laboratorio y en el medio ambiente

Los HAP tienen toxicidad aguda para los peces y para *Daphnia magna* en combinación con la absorción de radiación ultravioleta y luz visible. El metabolismo y la degradación alteran la toxicidad de los HAP. A concentraciones bajas, los HAP pueden estimular el crecimiento de microorganismos y algas. Los HAP más tóxicos para las algas son el benz[*a*]antraceno (cuatro anillos), con una CE_{50}

(concentración a la cual determinados parámetros vitales se reducen a la mitad) de 1-29 µg/litro, y el benzo[*a*]pireno (cinco anillos), con una CE_{50} de 5-15 µg/litro. Los valores de la CL_{50} en las algas para la mayor parte de los HAP de tres anillos son de 240-940 µg/litro. El naftaleno (dos anillos) es el menos tóxico, con valores de la CE_{50} de 2800-34 000 µg/litro.

No se han observado diferencias claras de sensibilidad entre distintos grupos taxonómicos de invertebrados, como los crustáceos, los insectos, los moluscos, los poliquetos y los equinodermos. El naftaleno es el menos tóxico, con valores de la CL_{50} en 96 horas de 100-2300 µg/litro. Los valores de la CL_{50} en 96 horas para los HAP de tres anillos oscilan entre <1 y 3000 µg/litro. El antraceno puede ser más tóxico que los otros HAP de tres anillos, con CL_{50} en 96 horas entre <1 y 260 µg/litro. Los valores de la CL_{50} en 96 horas para los HAP de cuatro, cinco y seis anillos son de 0,2-1200 µg/litro. Se observó toxicidad aguda (CL_{50}) en peces a concentraciones de 110 a >10 000 µg/litro de naftaleno, 30-4000 µg/litro de HAP de tres anillos (antraceno, 2,8-360 µg/litro) y 0,7-26 µg/litro para los de cuatro o cinco anillos.

La contaminación de los sedimentos con HAP a concentraciones de 250 mg/kg se asoció a tumores hepáticos en peces vivos libres. Se han inducido asimismo tumores en peces expuestos en el laboratorio. La exposición de los peces a determinados HAP puede producir también cambios fisiológicos y afectar a su crecimiento, reproducción, capacidad natatoria y respiración.

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