IPCS

INTERNATIONAL PROGRAMME ON CHEMICAL SAFETY

UNEP

Environmental Health Criteria 205

Polybrominated Dibenzo-*p*-dioxins and Dibenzofurans

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 205

POLYBROMINATED DIBENZO-*p*-DIOXINS AND DIBENZOFURANS

First draft prepared by Dr C. Melber and Dr J. Kielhorn, Fraunhofer Institute of Toxicology and Aerosol Research, Hanover, Germany

Published under the joint sponsorship of the United Nations Environment Programme, the International Labour Organisation, and the World Health Organization, and produced within the framework of the Inter-Organization Programme for the Sound Management of Chemicals.

World Health Organization
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.

WHO Library Cataloguing in Publication Data

Polybrominated dibenzo-*p*-dioxins and dibenzofurans.

(Environmental health criteria ; 205)

1.Dioxins 2.Benzofurans
3.Environmental exposure 4.Occupational exposure
I.International Programme on Chemical Safety II.Series

ISBN 92 4 157205 1 (NLM Classification: QD 405)
ISSN 0250-863X

PRINTED IN FINLAND
98/12247 – VAMMALA – 5000

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ENVIRONMENTAL HEALTH CRITERIA FOR POLYBROMINATED DIBENZO-*p*-DIOXINS AND DIBENZOFURANS

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. + 41 22 – 9799111, fax no. + 41 22 – 7973460, E-mail irptc@unep.ch).

* * *

This publication was made possible by grant number 5 U01 ES02617-15 from the National Institute of Environmental Health Sciences, National Institutes of Health, USA, and by financial support from the European Commission.

Environmental Health Criteria

P R E A M B L E

Objectives

In 1973 the WHO Environmental Health Criteria Programme was initiated 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 and since that time an ever-increasing number of assessments of chemicals and of physical effects have been produced. In addition, many EHC monographs have been devoted to evaluating toxicological methodology, e.g., for genetic, neurotoxic, teratogenic and nephrotoxic effects. Other publications have been concerned with epidemiological guidelines, evaluation of short-term tests for carcinogens, biomarkers, effects on the elderly and so forth.

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

The original impetus for the Programme came from World Health Assembly resolutions and the recommendations of the 1972 UN 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 UN Conference on Environment and Development and the subsequent establishment of the Intergovernmental Forum on Chemical Safety with the priorities for action in the six programme areas of Chapter 19, Agenda 21, all lend further weight to the need for EHC assessments of the risks of chemicals.

Scope

The criteria monographs are intended to provide critical reviews on the effect on human health and the environment of chemicals and of combinations of chemicals and physical and biological agents. As such, they include and review studies that are of direct relevance for the evaluation. However, they do not describe *every* study carried out. Worldwide data are used and are quoted from original studies, not from abstracts or reviews. Both published and unpublished reports are considered and it is incumbent on the authors to assess all the articles cited in the references. Preference is always given to published data. Unpublished data are only used when relevant published data are absent or when they are pivotal to the risk assessment. A detailed policy statement is available that describes the procedures used for unpublished proprietary data so that this information can be used in the evaluation without compromising its confidential nature (WHO (1990) Revised Guidelines for the Preparation of Environmental Health Criteria Monographs. PCS/90.69, Geneva, World Health Organization).

In the evaluation of human health risks, sound human data, whenever available, are preferred to animal data. Animal and *in vitro* studies provide support and are used mainly to supply evidence missing from human studies. It is mandatory that research on human subjects is 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

standard setting. These 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 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., IARC, JECFA, JMPR

Selection of chemicals

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

If an EHC monograph is proposed for a chemical not on the priority list, the IPCS Secretariat consults with the Cooperating Organizations and all the Participating Institutions before embarking on the preparation of the monograph.

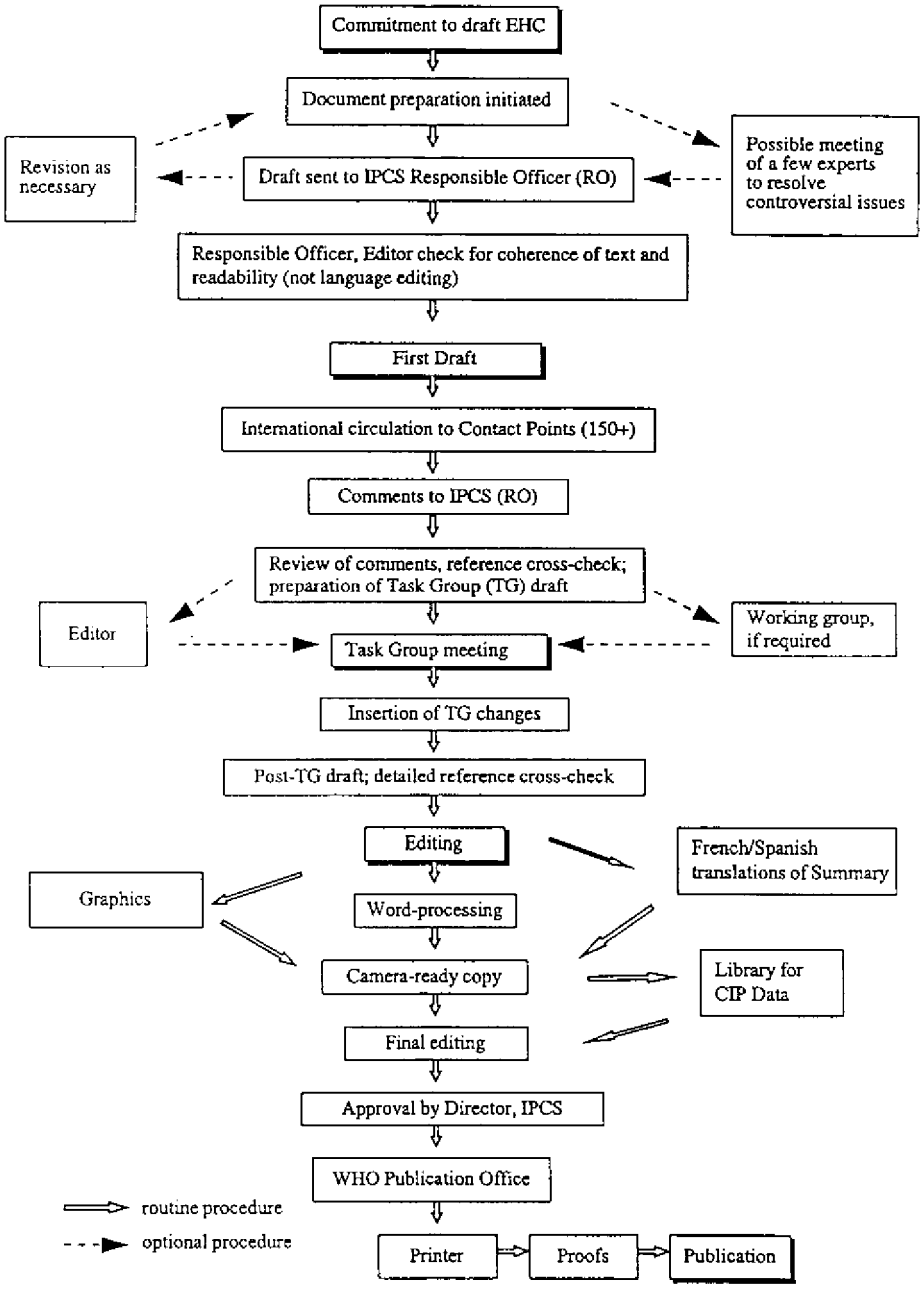
Procedures

The order of procedures that result in the publication of an EHC monograph is shown in the 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 layout and language. The first draft, prepared by consultants or, more usually, staff from 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 document acceptable as a first draft, it is distributed, in its unedited form, to well over 150 EHC contact points throughout the world who are asked to comment on its completeness and accuracy and, where necessary, 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 some four months are allowed before the comments are considered by the RO and author(s). A second draft incorporating comments received and approved by the Director, IPCS, is then distributed to Task Group members, who carry out the 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 health and environmental risks from exposure to the chemical. A summary and recommendations for further research and improved safety aspects are also required. 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.

EHC PREPARATION FLOW CHART



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

All individuals who as authors, consultants or advisers participate in the preparation of the EHC monograph 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 conflict of interest statement. Such a 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 then goes for language editing, reference checking, and preparation of camera-ready copy. 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 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 for health or environmental effects of the agent because of greater exposure; an appreciable time period 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.

WHO TASK GROUP ON ENVIRONMENTAL HEALTH CRITERIA FOR POLYBROMINATED DIBENZO-*p*-DIOXINS AND DIBENZOFURANS

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ENVIRONMENTAL HEALTH CRITERIA FOR POLYBROMINATED DIBENZO-*p*-DIOXINS AND DIBENZOFURANS

A WHO Task Group on Environmental Health Criteria for Polybrominated Dibenzo-*p*-dioxins and Dibenzofurans met at the Fraunhofer Institute for Toxicology and Aerosol Research, Hanover, Germany from 11 to 15 November 1996. Professor U. Heinrich opened the meeting and welcomed the participants on behalf of the host institute. Dr H. Galal-Gorchev, IPCS, welcomed the participants on behalf of the Director, IPCS, and the three IPCS cooperating organizations (UNEP/ILO/WHO). The Task Group reviewed and revised the draft criteria monograph and made an evaluation of the risks for human health and the environment from exposure to polybrominated dibenzo-*p*-dioxins and dibenzofurans.

Dr J. Kielhorn and Dr C. Melber, Fraunhofer Institute for Toxicology and Aerosol Research, Hanover, Germany, prepared the first draft of this monograph. They also prepared the second draft, incorporating comments received following the circulation of the first draft to the IPCS Contact Points for Environmental Health Criteria monographs.

Dr H. Galal-Gorchev, IPCS Central Unit, was responsible for the overall scientific content and Ms M. Sheffer, Scientific Editor, Ottawa, Canada, for the linguistic editing.

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

ABBREVIATIONS

ABS	acrylonitrile-butadiene-styrene
Ah	aromatic hydrocarbon
AHH	arylhydrocarbon hydroxylase
BB	bromobiphenyl
BDE	bromodiphenyl ether
CAS	Chemical Abstracts Service
CYP	cytochrome P-450
DBB/decaBB	decabromobiphenyl
DBDE/decaBDE	decabromodiphenyl ether
DD/DF	dibenzo-*p*-dioxin/dibenzofuran
DiBDD	dibromodibenzo-*p*-dioxin
DiBDF	dibromodibenzofuran
DiXDF	mixed dihalogenated dibenzofuran
EC_{50}	median effective concentration
ED_{50}	median effective dose
EI-SIM-MS	electron impact–selective ion monitoring–mass spectrometry
EPA	Environmental Protection Agency (USA)
EROD	ethoxyresorufin-*O*-deethylase
GC	gas chromatography
HexaBB	hexabromobiphenyl
HIPS	high-impact polystyrene
HpBDD/heptaBDD	heptabromodibenzo-*p*-dioxin
HpBDF/heptaBDF	heptabromodibenzofuran
HPLC	high-performance liquid chromatography
HRGC	high-resolution gas chromatography
HRMS	high-resolution mass spectrometry
HxBDD/hexaBDD	hexabromodibenzo-*p*-dioxin
HxBDF/hexaBDF	hexabromodibenzofuran
HxCDD/hexaCDD	hexachlorodibenzo-*p*-dioxin
I-TEF	international toxicity equivalency factor
I-TEQ	international toxic equivalent
LD_{50}	median lethal dose
LOAEL	lowest-observed-adverse-effect level
LOEL	lowest-observed-effect level
MI-IR	matrix isolation infrared spectrometry
MoBDD/monoBDD	monobromodibenzo-*p*-dioxin
MoBDF/monoBDF	monobromodibenzofuran

MS	mass spectrometry
n	sample size
NCI	negative ion chemical ionization
n.d.	not detected
NOAEL	no-observed-adverse-effect level
NOEL	no-observed-effect level
n.sp.	not specified
OBDE/octaBDE	octabromodiphenyl ether
OcBDD/octaBDD	octabromodibenzo-*p*-dioxin
OcBDF/octaBDF	octabromodibenzofuran
OCDD/OcCDD/octaCDD	octachlorodibenzo-*p*-dioxin
PAH	polycyclic aromatic hydrocarbon
PBB	polybrominated biphenyl
PBDD	polybrominated dibenzo-*p*-dioxin
PBDE	polybrominated diphenyl ether
PBDF	polybrominated dibenzofuran
PBT	polybutylene terephthalate
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDE	polychlorinated diphenyl ether
PCDF	polychlorinated dibenzofuran
PeBDD/pentaBDD	pentabromodibenzo-*p*-dioxin
PeBDE/pentaBDE	pentabromodiphenyl ether
PeBDF/pentaBDF	pentabromodibenzofuran
PeCDF/pentaCDF	pentachlorodibenzofuran
PeHDD	pentahalogenated dibenzo-*p*-dioxin
PeHDF	pentahalogenated dibenzofuran
PHDD	polyhalogenated dibenzo-*p*-dioxin (used as collective term including PCDD, PBDD, PXDD)
PHDF	polyhalogenated dibenzofuran (used as collective term including PCDF, PBDF, PXDF)
PVC	polyvinyl chloride
PXDD	mixed (brominated/chlorinated) halogenated dibenzo-*p*-dioxin
PXDF	mixed (brominated/chlorinated) halogenated dibenzofuran
RI	retention index
RIA	radioimmunoassay
RMM	relative molecular mass
SD	standard deviation

T_3	triiodothyronine
T_4	thyroxin
TBBPA	tetrabromobisphenol A
TBCDD	2,3-dibromo-7,8-dichlorodibenzo-*p*-dioxin
TBDD/2,3,7,8-TeBDD	2,3,7,8-tetrabromodibenzo-*p*-dioxin
TBDF/2,3,7,8-TeBDF	2,3,7,8-tetrabromodibenzofuran
TBPI	bis-tetrabromo-phthalimide ethylene
TCDD/2,3,7,8-TeCDD	2,3,7,8-tetrachlorodibenzo-*p*-dioxin
TCDF/2,3,7,8-TeCDF	2,3,7,8-tetrachlorodibenzofuran
TeBDD/tetraBDD	tetrabromodibenzo-*p*-dioxin
TeBDF/tetraBDF	tetrabromodibenzofuran
TEF	toxicity equivalency factor
TeHDD	tetrahalogenated dibenzo-*p*-dioxin
TEQ	toxic equivalent
TeXDD/tetraXDD	mixed tetrahalogenated dibenzo-*p*-dioxin
THDF	2,3,7,8-tetrahalogenated dibenzofuran
TrBDD/triBDD	tribromodibenzo-*p*-dioxin
TrBDF/triBDF	tribromodibenzofuran
TrHDD/triHDD	trihalogenated dibenzo-*p*-dioxin
TV	television
TxDD	2,3,7,8-substituted mixed tetrahalogenated dibenzo-*p*-dioxin
UV	ultraviolet
WHO	World Health Organization

1. SUMMARY

1.1 Identity, physical and chemical properties, and analytical methods

Polybrominated dibenzo-*p*-dioxins (PBDDs) and polybrominated dibenzofurans (PBDFs) are almost planar tricyclic aromatic compounds. Theoretically, 75 PBDDs and 135 PBDFs are possible. In addition, a large number of mixed halogenated congeners — 1550 brominated/chlorinated dibenzo-*p*-dioxins (PXDDs) and 3050 brominated/chlorinated dibenzofurans (PXDFs) — are theoretically possible. Because of the complexity of the analytical procedures and paucity of analytical reference standards, it has been possible to characterize and determine only a small number of these compounds. The most toxic congeners are those substituted at positions 2, 3, 7, and 8. There are 7 2,3,7,8-substituted PBDDs and 10 2,3,7,8-substituted PBDFs, as well as 337 possible 2,3,7,8-substituted PXDDs and 647 possible 2,3,7,8-substituted PXDFs.

PBDDs/PBDFs have higher molecular weights than their chlorinated analogues, high melting points, low vapour pressures, and low water solubilities. They are generally soluble in fats, oils, and organic solvents. There are very few experimental data on the physical and chemical properties of PBDDs/PBDFs.

Photolysis occurs at a more rapid rate for PBDDs/PBDFs than for polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). PBDDs/PBDFs are thermostable. The temperatures of formation and destruction of PBDDs/PBDFs depend on several conditions, including the presence or absence of oxygen, polymers, and flame retardant additives, such as antimony trioxide (Sb_2O_3).

In the presence of excess chlorine, bromine is substituted by chlorine to give PXDDs/PXDFs.

Because of the toxic nature of these compounds and their photolytic properties, care must be taken during sampling and analysis. Highly sensitive, selective, and specific analytical methods (gas

chromatography/mass spectrometry, or GC/MS) are required because of the large number of PBDD/PBDF congeners. Sampling procedures are identical for all polyhalogenated dibenzo-*p*-dioxins (PHDDs) and polyhalogenated dibenzofurans (PHDFs), but separation and determination of PBDDs/PBDFs (and PXDDs/PXDFs) differ slightly from those of their chlorinated analogues. PBDDs/PBDFs have higher molecular weights and longer GC retention times than the chlorinated analogues, as well as different MS isotopic cluster patterns and interference compounds. Exact identification of specific brominated congeners is very limited owing to the small number of reference standards currently available. For the same reason, determination of mixed halogenated congeners is almost impossible.

1.2 Formation and sources of human and environmental exposure

PBDDs/PBDFs are not known to occur naturally. They are not intentionally produced (except for scientific purposes) but are generated as undesired by-products in various processes. They can be formed by chemical, photochemical, or thermal reactions from precursors and by so-called *de novo* synthesis.

PBDDs/PBDFs have been found as contaminants in brominated organic chemicals (e.g. bromophenols) and, in particular, in flame retardants, such as polybrominated diphenyl ethers (PBDEs), decabromobiphenyl (decaBB or DBB), 1,2-bis(tribromophenoxy)ethane, tetrabromobisphenol A (TBBPA), and others. They have been detected in distillation residues of some bromophenols and bromoanilines and in wastes from chemical laboratories.

PBDFs and, to a lesser extent, PBDDs have been detected as photochemical degradation products of brominated organic chemicals, such as PBDEs and bromophenols.

Laboratory thermolysis experiments showed the formation of PBDDs/PBDFs from bromophenols, PBDEs, polybrominated biphenyls (PBBs), and other brominated flame retardants (pure or in a polymer matrix). There was a broad range of yields, from zero to maximum values (reached from PBDEs) in the g/kg range. Generally,

PBDFs were much more abundant than PBDDs. The optimum PBDF formation temperature of a series of pure flame retardants was in the range of 600–900 °C. The presence of polymers or synergists (e.g. Sb_2O_3) resulted in a decrease in the optimum formation temperature (down to 400 °C). In addition to temperature and the presence of polymer matrix or synergists, several other factors, such as metals, metal oxides, water, oxygen, and the type of combustion apparatus used, influenced the yield and pattern of PBDDs/PBDFs. In ternary mixtures of PBDE, polymer matrix, and Sb_2O_3, tetrabromodibenzofurans (tetraBDFs or TeBDFs) were frequently the most abundant homologue group. 2,3,7,8-Substituted PBDDs/PBDFs (tetra to hepta) were found at varying concentrations; for example, 2,3,7,8-TeBDF was found at up to 2000 mg/kg in pyrolysates of polymers containing octabromodiphenyl ether (octaBDE or OBDE).

In the manufacture of plastics, elevated temperatures (150–300 °C) occur during several processes. Studies of the exhaust streams from machines processing polymers — such as acrylonitrile-butadiene-styrene (ABS) and polybutylene terephthalate (PBT) — containing different types of brominated flame retardants showed that PBDDs/PBDFs (di to octa) can be formed at these temperatures. OBDE and decabromodiphenyl ether (decaBDE or DBDE) produced the highest amounts of PBDDs/PBDFs, the major portion consisting of PBDFs. Levels observed with TBBPA or bis-tetrabromophthalimide ethylene (TBPI) were several orders of magnitude lower. No PBDDs/PBDFs were detected during processing of ABS flame-retarded by brominated styrene or 1,2-bis(tribromophenoxy)ethane. 2,3,7,8-Substituted congeners were not determined (processing of DBDE), were detected at trace levels (processing of OBDE), or were not detected (processing of TBBPA and TBPI).

Various plastic materials at several processing stages were analysed for PBDDs/PBDFs. These included (granulated) resins and moulded parts whose flame retardant additives were known as well as samples from commercial electrical appliances (television sets, printers, computers) whose flame retardant additives were unknown. The highest levels of PBDDs/PBDFs were found in materials flame-retarded with PBDEs and were in the range of several thousand µg/kg, thus exceeding the levels of other flame retardant/polymer systems by

orders of magnitude. Factors influencing the extent of formation are temperature and the duration of such processes as blending, extrusion, and moulding. Again PBDFs dominated, with some exceptions, over PBDDs, with the highly brominated (>tetra) derivatives being prevalent. Peak concentrations were seen with pentabromodibenzofurans (pentaBDFs or PeBDFs) and hexabromodibenzofurans (hexaBDFs or HxBDFs). The latter reached levels as high as 3000 µg/kg in casing parts. Printed circuit boards contained tetra- and pentaBDFs at maximum concentrations of 1300 and 1400 µg/kg, respectively. Total PBDF (mono to hexa) concentrations were in the range of 3.6–3430 µg/kg. 2,3,7,8-Substituted PBDDs/PBDFs were not determined, were not detectable, or were present at relatively low concentrations. Maximum concentrations of 2,3,7,8-substituted PBDFs (tetra to hexa) in casings or printed circuit boards ranged from 11 µg/kg (tetra) to 203 µg/kg (hexa).

Experiments to determine whether PBDFs were released from television sets or similar appliances during use showed air levels ranging from not detected to 1800 pg total PBDFs (tetra to hexa) per appliance.

Burning of products containing brominated compounds caused emission of PBDDs/PBDFs. In experimental fire tests simulating real fire conditions with electrical appliances such as television sets, printers, computer terminals, and their casings, high PBDF (mono to hexa) concentrations were detected in the combustion residues (thousands of mg/kg), in smoke condensate (hundreds of µg/m^2), and in smoke (up to 1700 µg/m^3). PBDD concentrations amounted to about 3% of the detected levels of PBDDs/PBDFs. The 2,3,7,8-substituted isomer was mostly below 3% of the total tetraBDFs. 2,3,7,8-Substituted penta- and hexaBDFs yielded between 1 and 16% of the corresponding totals. Burning of test vehicles resulted in PBDF (mono to octa) concentrations of up to 4.3 µg/kg in the fire residues.

During real fire accidents in private residences (television sets involved), offices (computers involved), and other buildings, concentrations measured were in most cases below the values found in the model experiments described above, but the qualitative composition of the samples was similar. PBDFs were found in almost all samples,

but PBDDs were not always detected; if present, their concentrations were low. The PBDF concentrations in combustion residues were mainly in the µg/kg range (low to high), but single maximum values (sum of mono to hexa) of up to 107 mg/kg were also observed. The PBDF (mono to hexa) area contaminant concentrations in close vicinity to the fire site ranged between 0.1 and 13 µg/m^2 in most cases. Additionally, relevant concentrations of PXDDs/PXDFs could be detected. The proportion of 2,3,7,8-substituted PBDDs/PBDFs was relatively low in most of the samples examined. For example, maximum proportions of 3, 10, or 18% of the corresponding totals of tetra-, penta-, or hexaBDFs, respectively, were reported from fire accidents with television sets. Soot samples collected after a fire in a computer room contained 2,3,7,8-substituted tetra- and pentabromo-dibenzo-*p*-dioxins (tetra/pentaBDDs or TeBDD/PeBDD) and tetra- and pentaBDFs, with a maximum concentration of 48 µg/kg for 2,3,7,8-TeBDF (TBDF).

PXDDs were detected in ash from a wood-fired boiler. However, the sort of wood (treated or untreated) was not specified. No data were available on the incineration of other fuels, such as coal, peat, or fuel oil.

The presence of PBDDs/PBDFs and/or PXDDs/PXDFs has been reported in fly ash and/or flue gas of municipal, hospital, or hazardous waste incinerators. The majority of these compounds are probably produced in the incinerator itself, by formation from precursors at high temperatures in the flame or by *de novo* synthesis at low temperatures in the post-combustion zone of the incinerator. The formation of PXDDs/PXDFs is explained by the extensive bromine–chlorine exchange reactions (with chlorine donors in waste) observed under several test conditions. The quantities of PBDDs/PBDFs and PXDDs/PXDFs measured in fly ash of incinerators were in the range of ng/kg to µg/kg. In most cases, the concentrations of dibenzo-*p*-dioxins exceeded those of dibenzofurans, and PXDDs/PXDFs were more abundant than PBDDs/PBDFs. Of 2,3,7,8-substituted congeners, a mixed tetrahalogenated dibenzo-*p*-dioxin (tetraXDD or TeXDD) (Br_2Cl_2DD) was found.

Analyses of waste samples from some disposal sites showed the presence of PBDDs/PBDFs and PXDDs/PXDFs at concentrations of several hundred to several thousand ng/kg dry weight. The concentration of dibenzo-*p*-dioxins (up to 580 ng/kg) was below that of dibenzofurans (up to 4230 ng/kg). Generally, the homologue profile was dominated by the lower halogenated (mono to tetra) derivatives. Chemical laboratory waste contained PBDDs/PBDFs, with a peak concentration of 15 500 ng/kg for hexaBDFs.

PBDDs/PBDFs were present in plastic materials (with or without metals) of several recycling stages. The samples originated mainly from office machines, printed circuit boards, and other electronic scrap. In some cases, the sum concentration of eight selected PBDD/PBDF congeners having the 2,3,7,8-substitution was as high as 65 μg/kg. Metal reclamation was also found to be a source of PBDDs and/or PXDDs/PXDFs. PBDDs/PBDFs have also been detected in textile industries where brominated flame retardants have been used. PBDFs were found in the exhaust air, in the textiles before and after processing, and in the chimney depositions.

PBDDs/PBDFs and PXDDs/PXDFs (along with PCDDs/PCDFs) have been detected in emissions of motors using leaded petrol, in emissions of motors using unleaded petrol with and without catalytic converters, and in emissions of diesel engines. Because of the brominated and chlorinated scavengers (dibromo- and dichloroethane) used in leaded petrol, the highest levels of PHDDs/PHDFs (several thousand ng/m^3) were found with this type of petrol. Unleaded petrol produced much lower emissions of PHDDs/PHDFs (approximately two orders of magnitude lower). A further reduction was seen after catalytic gas cleaning. The values for diesel engines were somewhat higher than those found with the Otto motors (spark ignition engines) run on unleaded petrol. In exhaust gases from combustion of leaded petrol, PBDDs/PBDFs were more abundant than PXDDs/PXDFs and PCDDs/PCDFs. Generally, the concentrations of dibenzofurans exceeded those of dibenzo-*p*-dioxins, and there was a dominance of lower substituted homologues (mono to tri). Similar patterns were seen in residues adhering to mufflers.

1.3 Environmental transport, distribution, and transformation

There are very few data available on the environmental transport and distribution of PBDDs/PBDFs. Generally, their physicochemical properties suggest similarities to PCDDs/PCDFs. Therefore, if released to the environment, they may be preferably distributed into carbon- or fat-rich compartments, as with PCDDs/PCDFs.

Airborne PBDDs/PBDFs were found to be transported in both the particulate and vapour phase, the partitioning ratio depending on the degree of bromination.

No experimental data are available on the movement of PBDDs/PBDFs in water or soil. For PBDFs (tri to penta), adsorption to sediment was reported. Owing to the low water solubility of PBDDs/PBDFs, leaching through the soil may be limited but may be increased in the presence of organic solvents or humic acids.

There are no experimental data on processes for the transport and distribution of PBDDs/PBDFs between environmental media and biota or within biota. Based on the similar high octanol/water partition coefficients calculated for selected PCDDs/PCDFs, PBDDs/PBDFs, and PXDDs/PXDFs, a bioavailability comparable to that of PCDDs/PCDFs is expected.

Photolysis of PBDDs/PBDFs and PXDDs/PXDFs was studied in organic solvents and on quartz surfaces in the laboratory, as well as in soil and on soot (and dust) particles under outdoor conditions. The slowest photolytic reactions were observed under the latter, more environmentally relevant, conditions. Reductive debromination was found to be a major photochemical pathway. The rate of decomposition of different congeners depended on their bromine substitution pattern. Generally, higher brominated congeners and those with lateral bromines had shorter half-lives. Calculated half-lives were in the order of minutes (use of direct sunlight or ultraviolet [UV] light and quartz vials), hours (use of solid films or soot or dust particles and sunlight), or hundreds to thousands of hours (use of soil and sunlight). For example, the estimated sunlight-induced half-lives for 2,3,7,8-TeBDD

(TBDD) were 0.8 min (in organic solution) or 32 h (dispersed as solid films). A half-life of 3–6 months was estimated for tetraBDD isomers in surface soil. Compared with PCDDs/PCDFs, the brominated counterparts were photochemically less stable. PXDDs/PXDFs preferentially lost their bromine atoms during photolysis and therefore were transformed into PCDDs/PCDFs, which had longer photolytic half-lives. Such a transformation of PXDDs/PXDFs to PCDDs/PCDFs also occurs during incineration processes.

PBDDs/PBDFs seem to be poorly degradable by microorganisms.

The presence of PBDDs/PBDFs in animals and in humans, as seen in a few studies, is indicative of their accumulation potential. 2,3,7,8-TeBDD accumulated in rats during subchronic administration. Bioaccumulation, bioconcentration, or biomagnification factors for PBDDs/PBDFs or PXDDs/PXDFs are not available.

1.4 Environmental levels and human exposure

To date, in contrast to PCDDs/PCDFs, PBDDs/PBDFs have not been frequently included in monitoring programmes. The few studies performed indicate a limited occurrence.

In ambient air, PBDFs were found more frequently than PBDDs. Only lower brominated PBDDs (mono to tetra) were detected at concentrations ranging from not detected (n.d.) to about 0.85 pg/m^3 for monobromodibenzo-*p*-dioxins (monoBDDs or MoBDDs) in a motorway tunnel and an underground garage. Of PBDFs, mono- to hexabrominated homologues were found, their concentrations ranging from n.d. to 74 pg/m^3. The concentrations (mean values) of total PBDDs/PBDFs (tri to hexa) measured, for example, in Germany in a motorway tunnel, in a city, and in a suburban area amounted to 23 pg/m^3, 2 pg/m^3, and 0.59 pg/m^3, respectively; 2,3,7,8-TeBDD was not detected, and the maximum concentrations of 2,3,7,8-TeBDF and 1,2,3,7,8-PeBDF were 0.28 pg/m^3 and 0.08 pg/m^3, respectively. PXDFs were identified in traffic-related air samples at concentrations up to 41 pg/m^3 (Cl_jBr_iDFs). Outdoor dust samples (mainly from motorways) also showed a predominance of PBDFs/PXDFs (maxima

of several thousand ng/kg) over PBDDs/PXDDs (maxima of up to some hundred ng/kg).

Indoor air samples taken from rooms equipped with a number of operating electronic appliances (television and/or computer monitors) showed the presence of PBDFs (tetra to hepta) at total concentrations ranging from 0.23 to 1.27 pg/m^3. PBDDs were not detected. Dust samples collected in computer rooms yielded total PBDF levels of 2.4–5.5 µg/kg dust. In contrast to air, the homologue pattern in dust was dominated by hexaBDFs and heptabromodibenzofurans (heptaBDFs or HpBDFs). Only in dust samples were low concentrations of tetraBDDs (up to 1 µg/kg) and of 2,3,7,8-substituted tetra- and pentaBDFs (up to 0.07 µg/kg) detectable. PBDF concentrations in the one sample of house dust were lower by a factor of 10. The sum concentration of PBDDs/PBDFs equalled that of PCDDs/PCDFs in dust from computer rooms but was lower than that of PCDDs/PCDFs in house dust. Dust from an underground garage contained lower halogenated PBDFs (mono and di) and PXDFs (di to tetra), with a maximum concentration of 4.3 µg/kg for mixed dihalogenated dibenzofurans (DiXDFs).

No data are available on levels of PBDDs/PBDFs in water samples.

In river and marine sediment samples from an industrialized zone, tetraBDDs (up to 0.006 µg/kg dry weight) and tetra- to hexaBDFs (sum up to 0.37 µg/kg dry weight) were detected. Sediment from road drainage contained PBDFs (sum of mono to tri: 2.5 µg/kg; sum of tetra to hepta: 0.3 µg/kg) and PXDFs (sum of di and tri: 1.85 µg/kg), but no PBDDs.

Similarly, soil samples taken near a motorway contained monobromodibenzofurans (monoBDFs or MoBDFs) and dibromodibenzofurans (DiBDFs) (sum: 1.3 µg/kg), tetra- and pentaBDFs (sum: 0.02 µg/kg), and PXDFs (sum: 1 µg/kg), but no PBDDs. Soil samples taken from an incineration field and near a metal reclamation factory gave total PBDF concentrations of up to 100 µg/kg, but no PBDDs were detected. In a series of sewage sludge samples from municipal wastewater treatment plants, total PBDF concentrations ranged from

n.d. to 3 μg/kg. In one case, traces of tetraBDDs and 2,3,7,8-TeBDF were detected. A biocompost sample was nearly free of PBDDs/PBDFs (tetraBDFs: <0.003 μg/kg).

There are no quantitative data on levels of PBDDs/PBDFs in food.

In grass and pine needle samples collected near motorways, lower halogenated PBDFs/PXDFs (mono to tetra) and traces of PBDDs/PXDDs (mono to tri) were detectable.

No PBDDs/PBDFs were found in the few wildlife samples tested.

In cow's milk collected at dairy farms in the vicinity of a municipal waste incinerator, tribromodibenzofurans (triBDFs or TrBDFs), a tetraBDF, and a pentaBDF (not having the 2,3,7,8-substitution pattern) were tentatively identified.

PBDDs/PBDFs have not been detected in the few tested samples of human adipose tissues or milk samples from the general public.

Contamination by PBDDs/PBDFs is possible at a variety of workplaces involved in the production, processing, use, or disposal of certain flame retardants or their products, especially where processes involve elevated temperatures. The magnitude of worker exposure depends not only on the compounds involved but also on the quality of the air and ventilation conditions. There are only limited workplace monitoring data from plastic producing or processing facilities, from offices/studios with large numbers of electrical appliances continuously in use, and from recycling workplaces (including secondary copper plants). Generally, PBDFs were more abundant than PBDDs, and PBDF air concentrations were highest at workplaces where DBDE-containing polymers were produced. In many samples, 2,3,7,8-substituted PBDFs/PBDDs were detectable. PBDD/PBDF contamination was also found at the work area under the fume hood of a chemical laboratory. Monitoring data at waste incineration facilities are lacking.

1.5 Kinetics and metabolism

Most of the studies refer to 2,3,7,8-TeBDD and, to a lesser extent, 1,2,7,8-TeBDF. Half-life calculations have included some additional congeners.

2,3,7,8-TeBDD was absorbed in rats after oral, intratracheal, and dermal administration, the percent absorption varying with route and dose. Single doses of 1 nmol 2,3,7,8-TeBDD/kg body weight led to an absorption of 80% (oral and intratracheal routes) or 12% (dermal route) of the administered dose. The dermal absorption of 1 nmol 1,2,7,8-TeBDF/kg body weight was about 29%. Oral absorption of 2,3,7,8-TeBDD appeared to be comparable to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TeCDD or TCDD). However, dermal absorption of 2,3,7,8-TeBDD was about one-third that of an equimolar dose of 2,3,7,8-TeCDD.

2,3,7,8-TeBDD or 1,2,7,8-TeBDF administered to rats, by any route, was distributed throughout the entire body, with major deposits found in liver and adipose tissue, followed by skin and muscle. For example, 3 days after single oral doses of 2,3,7,8-TeBDD (1 nmol/kg body weight), the portions in these tissues amounted to 20%, 20%, 11%, and 4%, respectively, whereas thymus and adrenals contained 0.03% and 0.4%, respectively, of the administered dose. The partitioning of 2,3,7,8-TeBDD between liver and adipose tissue of rats was found to be influenced by dose, route of exposure, and time post-dosing. The ratios of liver : fat concentrations measured under different conditions ranged from 0.2 to 6.5 (range for single doses of 2,3,7,8-TeBDD in rats). No experimental data were available on the transfer of PBDDs/PBDFs to offspring.

TetraBDD/BDF metabolites were detected in bile and faeces from rats. They were mainly formed by aromatic hydroxylation and hydrolytic debromination. The rate of metabolism (indirectly determined as the rate of biliary excretion) differed between 2,3,7,8-TeBDD (about 7%) and 1,2,7,8-TeBDF (about 50%). Three days after an intravenous dose of 2,3,7,8-TeBDD (1 nmol/kg body weight), 14% of the administered dose was found as metabolites in the faeces of rats.

Elimination and excretion of 2,3,7,8-TeBDD were studied in rats using oral, intravenous, intratracheal, and dermal routes of administration. In all studies, the major route of elimination was through the faeces, the eliminated radioactivity ranging from 2% (dermal route) to 42% (oral route) of the administered dose (1 nmol [^{3}H]2,3,7,8-TeBDD/kg body weight) in faeces samples, and from 0.2 to 1% in urine samples. Similarly, in studies with 1,2,7,8-TeBDF in rats, excretion was mainly through the faeces, only 2–3% of the intravenous, oral, or dermal doses being excreted in urine. During the first days following oral doses, unabsorbed material and biliary excretion appeared to be the major sources of eliminated compound in faeces. The portions of parent 2,3,7,8-TeBDD found in faeces of rats after administration of 1 nmol 2,3,7,8-TeBDD/kg body weight were 53% (oral route), 43% (intratracheal route), and 10–20% (intravenous route). A few days after oral application of 2,3,7,8-TeBDD (1 nmol/kg body weight), about 20% of the dose administered was eliminated as parent compound.

Data on retention and turnover are available for some PBDDs/PBDFs. The relative body burden of 2,3,7,8-TeBDD (and other congeners) in rats depends on the route of exposure and on the dose administered, reflecting differences in absorption. Half-lives were calculated for several PBDDs/PXDDs and PBDFs in various tissues and faeces of rats. They ranged between 1 day (1,2,7,8-TeBDF from body) and 99 days (2,3,4,7,8-PeBDF from liver). The estimated half-lives of 17, 18, and 58 days for 2,3,7,8-TeBDD in liver, faeces, and adipose tissue, respectively, were similar to those reported for 2,3,7,8-TeCDD in liver and faeces, but higher (by a factor of >2) than those reported for 2,3,7,8-TeCDD in adipose tissue. Despite differences in early retention, half-lives of 2,3,7,8-TeBDF and 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8-TeCDF or TCDF) in liver were comparable.

As with PCDDs/PCDFs, half-lives calculated for humans are much longer than those for rats. There are estimations of 3–11 years (mean: 5.9 years) for 2,3,7,8-TeBDD and of 1–2 years (mean: 1.5 years) for 2,3,7,8-TeBDF. The persistence of these compounds in humans was also seen in the case of a chemist who had synthesized 2,3,7,8-TeBDD and 2,3,7,8-TeCDD in 1956. Thirty-five years after

exposure, markedly elevated levels of 2,3,7,8-TeBDD were found in his blood.

1.6 Effects on laboratory mammals and *in vitro* test systems

Most studies were concerned with the toxicity of 2,3,7,8-TeBDD, but some information was also available on other PBDDs/PBDFs and PXDDs/PXDFs.

2,3,7,8-TeBDD caused typical 2,3,7,8-TeCDD-like effects, including wasting syndrome, thymus atrophy, and liver toxicity. Additionally, liver damage described as peliosis hepatis, which has not been reported after exposure of rats to 2,3,7,8-TeCDD, was observed. The pattern of lesions (lethality, histopathology, liver and thymus weights) found in guinea-pigs after a single exposure and in rats after short-term exposure to 2,3,7,8-TeBDF was similar to that of 2,3,7,8-TeCDF.

2,3,7,8-TeBDD interacts with the endocrine system. In rats, dose-related changes in circulating thyroid hormones and impairment of spermatogenic activity have been observed.

The oral LD_{50} (28-day observation period) of 2,3,7,8-TeBDD in Wistar rats was about 100 µg/kg body weight for females and about 300 µg/kg body weight for males. Oral LD_{50} values for 2,3,7,8-TeCDD obtained from other studies ranged between 22 and >3000 µg/kg body weight. Equimolar doses of 2,3,7,8-TeBDF and 2,3,7,8-TeCDF resulted in comparable mortality rates in guinea-pigs. For example, 100% mortality was seen after treatment with both 2,3,7,8-TeBDF (0.03 µmol/kg body weight, 15.8 µg/kg body weight) and 2,3,7,8-TeCDF (0.03 µmol/kg body weight, 10 µg/kg body weight). Pre-peliotic lesions and changes in thyroid hormones were seen in rats after a single dose of 100 µg 2,3,7,8-TeBDD/kg body weight.

In Wistar rats administered 2,3,7,8-TeBDD orally for 13 weeks, evidence for decreased spermatogenic activity, defective and necrotic spermatocytes, signs of severe peliosis hepatis, and changes in circulating thyroid hormones and organ weights were observed. The

no-observed-adverse-effect level (NOAEL) was 0.01 µg/kg body weight per day.

2,3,7,8-TeBDF administered orally to Sprague-Dawley rats for 4 weeks caused dose-dependent growth retardation and histopathological changes in liver and thymus. The NOAEL was 1 µg/kg body weight per day.

Developmental toxicity of some 2,3,7,8-substituted PBDDs/PBDFs occurred in mice at subcutaneous and oral doses that produced no maternal toxicity and no fetal mortality. The lowest-observed-effect levels (LOELs) (in µg/kg body weight) for hydronephrosis and cleft palate after a single oral exposure of pregnant mice were, respectively, as follows: 3 and 48 for 2,3,7,8-TeBDD, 25 and 200 for 2,3,7,8-TeBDF, 400 and 2400 for 2,3,4,7,8-PeBDF, and 500 and 3000–4000 for 1,2,3,7,8-PeBDF. Compared on a molar basis, 2,3,7,8-TeBDD and 2,3,7,8-TeCDD were almost equipotent in induction of hydronephrosis. Compared on a weight basis, generally the brominated isomers were slightly less potent than the chlorinated ones in induction of hydronephrosis and cleft palate. However, 2,3,7,8-TeBDF was more active than 2,3,7,8-TeCDF.

No information was found on the mutagenicity of PBDDs/PBDFs or related end-points.

No long-term toxicity and carcinogenicity studies with PBDDs/PBDFs were available. 2,3,7,8-TeBDD tested positive in a cell transformation assay using murine peritoneal macrophages. However, the transforming potency of 2,3,7,8-TeBDD was seven times less than that of 2,3,7,8-TeCDD. Later, tumours developed in nude mice after subcutaneous injection of the resulting established cell lines.

A series of several PBDDs and PXDDs (tetra and penta) given intraperitoneally to immature male Wistar rats caused body weight losses 14 days after injection. On the basis of molar ED_{50} values, the most toxic compounds tested were 2,3,7,8-TeBDD, 2-Br_1-3,7,8-Cl_3-DD, and 2,3-Br_2-7,8-Cl_2-DD (TBCDD), which are substituted only in the four lateral positions. The relative potencies of the other PBDDs examined followed the order 2,3,7,8- > 1,2,3,7,8- > 1,2,4,7,8- >

1,3,7,8-DD. In other experiments, there were only slight differences in the ED_{50} values (on a molar basis) for body weight loss, thymic atrophy, and hepatic enzyme induction between 2,3,7,8-TeCDD and 2,3,7,8-TeBDD.

Thymic atrophy and other signs of immunotoxicity (e.g. haematological parameters, alterations of certain lymphocyte subpopulations) were seen with several PBDDs/PXDDs and 2,3,7,8-TeBDF in the rat and with 2,3,7,8-TeBDD and TBCDD in the marmoset monkey (*Callithrix jacchus*). It was concluded that, on a molar basis, the potency of 2,3,7,8-TeBDD was comparable to that of 2,3,7,8-TeCDD in rats and monkeys. For example, a significant effect on certain lymphocyte subpopulations in monkeys was found after a single subcutaneous dose of 30 ng 2,3,7,8-TeBDD/kg body weight versus 10 ng 2,3,7,8-TeCDD/kg body weight. Effects on immunotoxicity after perinatal exposure to PBDDs/PBDFs have not been investigated.

After subchronic dosing of either 2,3,7,8-TeBDD or 2,3,7,8-TeCDD by oral gavage in mice, there was a dose-dependent increase in total hepatic porphyrins.

After single oral doses of 2,3,7,8-TeBDD and 2,3,7,8-TeCDD, reductions in concentration and total amount of vitamin A were observed in the liver of rats, with 2,3,7,8-TeBDD being slightly less potent than 2,3,7,8-TeCDD (on a molar basis).

2,3,7,8-TeBDD and 2,3,7,8-TeBDF produced hyperkeratosis in the rabbit ear assay at a dose of 100 µg/rabbit, but not at 10 µg/rabbit. A no-observed-effect level (NOEL) for 2,3,7,8-TeCDD was 0.01 µg/rabbit.

Several tetra- (Br_1Cl_3DDs, Br_2Cl_2DDs) and penta- (Br_1Cl_4DD) halogenated congeners with 2,3,7,8-substitution were found to have an antiestrogenic potency similar to that of 2,3,7,8-TeCDD, as examined in cultures of human breast cancer cells.

In rats, 2,3,7-tribromodibenzo-*p*-dioxin (2,3,7-triBDD/TrBDD) depressed the disappearance of ouabain from plasma, its excretion into bile, and bile flow to a slightly lesser extent than 2,3,7,8-TeCDD.

PBDDs/PBDFs and PXDDs/PXDFs are potent inducers of certain cytochrome P-450 (CYP)-dependent microsomal enzymes. ED_{50} values of 0.8–1 nmol/kg body weight for CYP1A1 induction and about 0.2 nmol/kg body weight for CYP1A2 induction in rat liver were estimated after single oral doses of 2,3,7,8-TeBDD. CYP1A1 induction (arylhydrocarbon hydroxylase [AHH] and/or ethoxyresorufin-*O*-deethylase [EROD] induction) was observed in a variety of species and tissues *in vivo* and in rat cell cultures *in vitro*. A lot of different congeners were found to be active, as well as pyrolysates from certain flame retardants. Generally, enzyme induction proceeded dose-dependently at non-toxic concentrations, started soon after exposure, and was long-lasting. It was measurable at exposures as low as the pmol range. The induction potency varied over several orders of magnitude for different congeners, depending on their chemical structure. The most potent inducers were TCDD, TBDD, and TBCDD. Compared (on a molar basis) with their chlorinated analogues, the PBDDs and PXDDs had more or less similar potency. In contrast to TCDD, whose relative induction potency was independent of the tissue examined, TBDD was five times more potent at inducing EROD activity in the liver than in skin and lung following subchronic exposure of mice. The ranking order for induction of EROD activity in marmoset monkeys was TCDD > 2,3,4,7,8-pentachlorodibenzofuran (2,3,4,7,8-pentaCDF/PeCDF) > 2,3,4,7,8-PeBDF when enzyme activities were compared with the hepatic concentrations. *In vitro* tests with rat cell cultures resulted in similar molar EC_{50} values of AHH and EROD induction potencies between corresponding PXDFs and PCDFs.

PBDDs/PBDFs are believed to share a common mechanism of action with PCDDs/PCDFs and other related halogenated aromatic hydrocarbons (Ah). Binding to the cytosolic Ah receptor, which plays a central role in mediating 2,3,7,8-TeCDD-like toxicity, was confirmed for several PBDDs and PXDDs/PXDFs. Their receptor-binding affinities varied by several orders of magnitude but were comparable to those of their chlorinated analogues.

1.7 Effects on humans

There are no data on the exposure of humans to PBDDs/PBDFs or on their effects on the health of the general population.

Two cases of acute health problems due to 2,3,7,8-TeBDD/TeCDD exposure have been reported, with symptoms including chloracne.

In another study, male personnel of a chemical plant with documented exposure to PBDDs/PBDFs originating from the use of brominated flame retardants (OBDE and DBDE) were subjected to immunological and additional clinical laboratory tests. Although there were indications of minor changes in immunological parameters, the overall evaluation of their health status did not reveal an impact of 2,3,7,8-TeBDD/TeBDF body burden on the immune system.

There are no reports on cancer mortality caused by PBDDs/PBDFs.

1.8 Effects on other organisms in the laboratory and field

There is only limited information on the effects of PBDDs/PBDFs on microorganisms, plants, or invertebrate or vertebrate wildlife species.

Using the rainbow trout (*Oncorhynchus mykiss*) sac fry early life stage mortality bioassay, a series of PBDD/PBDF congeners were tested and found to be active. This bioassay also demonstrated that for both PBDDs and PBDFs, there was a decreased potency with increased bromine substitution. Both 2,3,7,8-TeBDD and 2,3,7,8-TeBDF were more potent than their chlorinated analogues.

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

2.1 Identity

PHDDs/PHDFs are almost planar tricyclic aromatic compounds. There are eight positions on both the dibenzo-*p*-dioxin and the dibenzofuran molecules where halogen substitution can occur. The positions are numbered as shown in Fig. 1 for PBDDs and PBDFs.

Each series consists of eight homologous groups (groups of congeners having the same number of bromine atoms), and in each group there are different numbers of isomers (see Table 1). Theoretically, 75 PBDDs and 135 PBDFs are possible, as well as a large number of mixed halogenated congeners — 1550 PXDDs and 3050 PXDFs

Table 1. Number of isomers, elemental composition, and molecular weight for PBDDs/PBDFs

Compound	Number of isomers		Elemental composition	Molecular weight
	Total	2,3,7,8-Substituted		
MoBDD	2	–	$C_{12}H_7O_2Br$	263.1
DiBDD	10	–	$C_{12}H_6O_2Br_2$	342.0
TrBDD	14	–	$C_{12}H_5O_2Br_3$	420.9
TeBDD	22	1	$C_{12}H_4O_2Br_4$	499.8
PeBDD	14	1	$C_{12}H_3O_2Br_5$	578.7
HxBDD	10	3	$C_{12}H_2O_2Br_6$	657.6
HpBDD	2	1	$C_{12}HO_2Br_7$	736.5
OcBDD	1	1	$C_{12}O_2Br_8$	815.4
MoBDF	4	–	$C_{12}H_7OBr$	247.1
DiBDF	16	–	$C_{12}H_6OBr_2$	326.0
TrBDF	28	–	$C_{12}H_5OBr_3$	404.9
TeBDF	38	1	$C_{12}H_4OBr_4$	483.8
PeBDF	28	2	$C_{12}H_3OBr_5$	562.7
HxBDF	16	4	$C_{12}H_2OBr_6$	641.6
HpBDF	4	2	$C_{12}HOBr_7$	720.5
OcBDF	1	1	$C_{12}OBr_8$	799.4

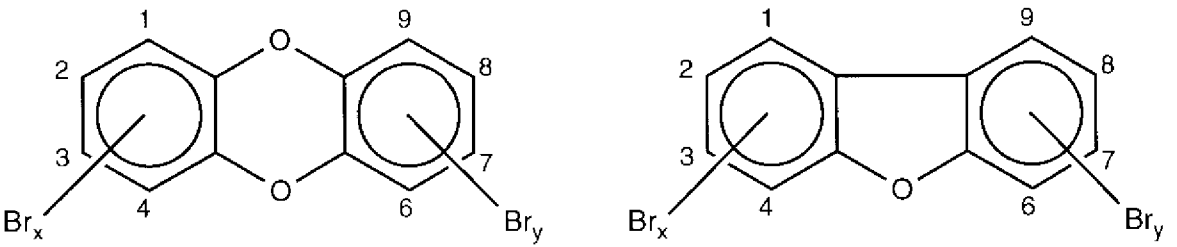

Fig. 1 Structural formulae of PBDDs and PBDFs

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(Buser, 1987a). There are 7 2,3,7,8-substituted PBDDs and 10 2,3,7,8-substituted PBDFs (see Table 2), as well as 337 possible 2,3,7,8-substituted PXDDs and 647 possible 2,3,7,8-substituted PXDFs (Ballschmiter & Bacher, 1996). PCDDs/PCDFs are discussed in a separate Environmental Health Criteria monograph (WHO, 1989).

Table 2. PBDDs/PBDFs brominated at the 2,3,7,8-positions

PBDD congener[a]	PBDF congener[a]
2,3,7,8-TeBDD*	2,3,7,8-TeBDF*
1,2,3,7,8-PeBDD*	1,2,3,7,8-PeBDF*
	2,3,4,7,8-PeBDF*
1,2,3,4,7,8-HxBDD*	1,2,3,4,7,8-HxBDF
1,2,3,6,7,8-HxBDD*	1,2,3,6,7,8-HxBDF
1,2,3,7,8,9-HxBDD*	1,2,3,7,8,9-HxBDF
	2,3,4,6,7,8-HxBDF
1,2,3,4,6,7,8-HpBDD	1,2,3,4,6,7,8-HpBDF
	1,2,3,4,7,8,9-HpBDF
OcBDD	OcBDF

[a] The congeners marked with an asterisk (*) are cited in the German Dioxin Directive (1994) (see Appendix I).

Because of the complexity of the analytical procedures (see section 2.4), it has been possible to characterize only a small number of PBDDs/PBDFs and PXDDs/PXDFs. Tables 3 and 4 show the Chemical Abstracts Service (CAS) numbers that have been allocated to some of these compounds.

2.2 Physical and chemical properties

2.2.1 *Appearance, melting and boiling points, water solubility, vapour pressure, octanol/water partition coefficient, and sorption coefficient*

Experimental data on the physical and chemical properties of PBDDs/PBDFs are scarce (see Table 5). In many cases, only predicted

Table 3. CAS numbers for some PBDDs/PBDFs

PBDD congener[a]	CAS number	PBDF congener[a]	CAS number
$\underline{Br_1DD}$	103456-34-4	$\underline{Br_1DF}$	103456-35-5
1-Br_1DD	105908-71-2	2-Br_1DF	86-76-0
2-Br_1DD	105906-36-3		
$\underline{Br_2DD}$	103456-37-7	$\underline{Br_2DF}$	103456-40-2
1,6-Br_2DD	91371-14-1	2,7-Br_2DF	65489-80-7
2,7-Br_2DD	39073-07-9	2,8-Br_2DF	10016-52-1
2,8-Br_2DD	105836-96-2		
$\underline{Br_3DD}$	103456-38-8	$\underline{Br_3DF}$	103456-41-3
		1,2,8-Br_3DF	84761-81-9
		2,3,8-Br_3DF	84761-82-0
$\underline{Br_4DD}$	103456-39-9	$\underline{Br_4DF}$	106340-44-7
1,2,3,4-Br_4DD	104549-41-9	1,2,7,8-Br_4DF	84761-80-8
2,3,7,8-Br_4DD	50585-41-6	2,3,7,8-Br_4DF	67733-57-7
$\underline{Br_5DD}$	103456-36-6	$\underline{Br_5DF}$	68795-14-2
1,2,3,7,8-Br_5DD	109333-34-8	1,2,3,7,8-Br_5DF	107555-93-1
		2,3,4,6,7-Br_5DF	124388-77-8
		2,3,4,7,8-Br_5DF	131166-92-2
$\underline{Br_6DD}$	103456-42-4	$\underline{Br_6DF}$	103456-33-3
1,2,3,4,7,8-Br_6DD	110999-44-5	1,2,3,4,6,7-Br_6DF	124388-78-9
1,2,3,6,7,8-Br_6DD	110999-45-6	1,2,3,6,7,8-Br_6DF	107555-94-2
1,2,3,7,8,9-Br_6DD	110999-46-7		
$\underline{Br_7DD}$	103456-43-5	$\underline{Br_7DF}$	62994-32-5
		1,2,3,4,6,7,8-Br_7DF	107555-95-3
$\underline{Br_8DD}$	2170-45-8	$\underline{Br_8DF}$	103582-29-2

[a] The homologue groups are underlined.

values are available. It should be noted that for PCDDs, experimental data are often lower than the calculated values (Shiu et al., 1988; Fiedler & Schramm, 1990). This is also to be expected for the brominated and for the mixed halogenated compounds (Fiedler & Schramm, 1990). Measured values for the aqueous solubility of PCDDs decrease dramatically with increase in chlorine substitution and temperature (Shiu et al., 1988).

Table 4. CAS numbers for some PXDDs/PXDFs

PXDD congener[a]	CAS number	PXDF congener[a]	CAS number
Br_1Cl_1DD	109007-09-02	Br_1Cl_1DF	109264-70-2
Br_1Cl_2DD	107227-59-8	Br_1Cl_2DF	107227-60-1
Br_1Cl_3DD	107227-75-8	Br_1Cl_3DF	107227-56-5
		8-Br_1-2,3,4-Cl_3DF	n.g.[b]
Br_1Cl_4DD	109264-61-1	Br_1Cl_4DF	109302-36-5
		1-Br_1-2,3,7,8-Cl_4DF	104549-43-1
		4-Br_1-2,3,7,8-Cl_4DF	115656-08-1
Br_1Cl_5DD	109264-65-5	Br_1Cl_5DF	107103-81-1
Br_1Cl_6DD	109264-67-7	Br_1Cl_6DF	107207-47-6
Br_1Cl_7DD	109264-69-9	Br_1Cl_7DF	109302-40-1
Br_2Cl_1DD	107227-58-7	Br_2Cl_1DF	107227-57-6
Br_2Cl_2DD	107227-74-7	Br_2Cl_2DF	107227-55-4
Br_2Cl_3DD	109031-99-4	Br_2Cl_3DF	107227-53-2
Br_2Cl_4DD	109264-62-2	Br_2Cl_4DF	107207-48-7
Br_2Cl_5DD	109264-66-6	Br_2Cl_5DF	107207-45-4
Br_2Cl_6DD	109264-68-8	Br_2Cl_6DF	109302-39-8
Br_3Cl_1DD	n.g.[b]	Br_3Cl_1DF	107227-54-3
Br_3Cl_2DD	n.g.[b]	Br_3Cl_2DF	107227-52-1
Br_3Cl_3DD	n.g.[b]	Br_3Cl_3DF	107207-46-5
Br_3Cl_4DD	n.g.[b]	Br_3Cl_4DF	107207-42-1
Br_3Cl_5DD	n.g.[b]	Br_3Cl_5DF	n.g.[b]
Br_4Cl_1DD	n.g.[b]	Br_4Cl_1DF	107227-51-0
Br_4Cl_2DD	n.g.[b]	Br_4Cl_2DF	107207-44-3
1,2,3,4-Br_4-7,8-Cl_2DD	134974-39-3		
Br_4Cl_3DD	n.g.[b]	Br_4Cl_3DF	107207-41-0
Br_4Cl_4DD	n.g.[b]	Br_4Cl_4DF	n.g.[b]
1,2,3,4-Br_4-6,7,8,9-Cl_4DD	124728-12-7		
Br_5Cl_1DD	n.g.[b]	Br_5Cl_1DF	107207-49-8
		other Br_5Cl_1DF	n.g.[b]
Br_6Cl_1DD	107207-38-8	Br_6Cl_1DF	n.g.[b]
Br_6Cl_2DD	n.g.[b]	Br_6Cl_2DF	107207-36-3
1,2,4,6,7,9-Br_6-3,8-Cl_2DD	2170-44-7		
Br_7Cl_1DD	n.g.[b]	Br_7Cl_1DF	107207-37-4

[a] The homologue groups are underlined.
[b] n.g. = CAS numbers not found (probably not yet allocated).

PBDDs/PBDFs have higher molecular weights than their chlorinated analogues, high melting points, and low water solubilities, but they are generally soluble in fats, oils, and organic solvents (see Table 5). PBDDs/PBDFs have, like their chlorinated analogues, very low vapour pressures, and at ambient temperatures they are mostly found bound to particles. For the lower substituted compounds, PBDDs/PBDFs have higher calculated pK_{ow} values than the chlorinated congeners (Fiedler & Schramm, 1990) and are therefore more lipophilic.

2.2.2 *Stability of PBDDs/PBDFs*

2.2.2.1 *Photolysis*

In the presence of laboratory light or sunlight, photolysis occurs at a more rapid rate for PBDDs/PBDFs than for PCDDs/PCDFs (Buser, 1988; Chatkittikunwong & Creaser, 1994a; for details, see section 4.2.1). This should be taken into consideration when analyses of these compounds are carried out (see sections 2.4.1 and 4.2.1).

Photolysis on quartz surfaces under sunlight is a much slower process than photolysis in organic solvents (Buser, 1988). PBDDs/PBDFs adsorbed on incinerator soot particles remained relatively stable and degraded only slowly during a 6-h period (Lutes et al., 1990, 1992a,b). Studies of PBDDs in soil showed that for the same congeners, the half-lives in this matrix are four times longer than in solution (Chatkittikunwong & Creaser, 1994a).

Under conditions of ambient temperature and protection from light, there is no appreciable (>1%) degradation of crystalline PBDDs/PBDFs and no significant change (0.6%, with the exception of octaBDD [9.7%]) in standard solution (solvent: *n*-nonane) concentrations over a period of 3 years (Re et al., 1995).

2.2.2.2 *Thermolytic degradation of PBDDs/PBDFs*

As discussed in chapter 3, the temperature of formation and destruction of PBDDs/PBDFs depends on several conditions, such as residence time, the presence/absence of oxygen, polymers, and

Table 5. Physical and chemical properties of some PBDDs/PBDFs

Compound	Appearance	Melting point (°C) (observed)	Boiling point (°C) (predicted)	Water solubility [log *S*] (mol/litre) (predicted)	Vapour pressure [log *P*] (Pa at 25 °C) (predicted)	Octanol/water partition coefficient [log K_{ow}] (predicted)	Sorption coefficient [log K_{oc}] (mol/litre) (predicted)
PBDDs							
1-MoBDD	white needles	104–106[e]	338.2[b]		3.5 x 10^{-3}[b]		
2-MoBDD	n.g.[c]	93–94.5[a] (90–92)[a]	338.2[b]	-6.12[d]	4.0 x 10^{-3}[b]	5.62[d]	4.39[d]
1,6-DiBDD		207[e]	375[b]		1.5 x 10^{-4}[b]		
2,3-DiBDD	n.g.[c]	157.2–158[f]	375[b]	-6.90[d]	1.6 x 10^{-4}[b]	6.25[d]	4.74[d]
2,7-DiBDD		174–176[a] 193–194[e]	375[b]		1.5 x 10^{-4}[b]		
2,8-DiBDD		149.5–151[a] (145–150)[a]	375[b]		1.7 x 10^{-4}[b]		
3,7-DiBDD				-7.24[d]		6.53[d]	4.89[d]
				-7.99[d]		7.14[d]	5.22[d]
1,2,3,4-TeBDD					6 x 10^{-7}[g]		
2,3,7,8-TeBDD	white granules	334–336[a,f]	438.3[b]	-8.72[d]	6.4 x 10^{-7}[b]	7.74[d] 6.50[h] 7.73[i]	5.54[d]
1,2,3,7,8-PeBDD				-9.45[d]		8.32[d]	5.87[d]

Table 5 (contd).

1,2,3,4,6,7,8-HpBDD				-10.89[d]		9.50[d]	6.50[d]
OcBDD		376[i]	523.2[b]	-11.69[d]	4.1 x 10 [11b] 9.3 x 10 [16g]	10.08[d]	6.82[d]
PBDFs							
monoBDF					2.89–3.26[k]		
2-MoBDF				-5.42[d]		5.05[d]	4.08[d]
diBDF					4.35–4.46[k]	5.58–6.09[k]	
2,7-DiBDF				6.25[d]		5.95[d]	4.47[d]
triBDF					5.36–5.47[k]	6.49–6.79[k]	
1,2,8-TrBDF + 2,3,8-TrBDF	colourless prisms[i]	144–148[i]					
2,3,7-TrBDF				-7.26[d]		6.55[d]	4.90[d]
tetraBDF					6.35–6.41[k]	7.72–8.72[k]	
1,2,7,8-TeBDF	colourless needles[i]	240.5–242[i]				6.20[h]	
2,3,7,8-TeBDF	colourless needles[i]	301–302[i]		-7.99[d]		7.14[d] 5.98[h]	5.22[d]
2,3,4,6-TeBDF				-7.99[d]		7.14[d]	5.22[d]
pentaBDF					7.25–7.45[k]		
1,2,3,7,8-PeBDF						7.04[h] 7.56[i]	
2,3,4,7,8-PeBDF				-8.71[d]		7.73[d]	5.54[d]
hexaBDF					8.34[k]		

Table 5 (contd).

Compound	Appearance	Melting point (°C) (observed)	Boiling point (°C) (predicted)	Water solubility [log *S*] (mol/litre) (predicted)	Vapour pressure [log *P*] (Pa at 25 °C) (predicted)	Octanol/water partition coefficient [log K_{ow}] (predicted)	Sorption coefficient [log K_{oc}] (mol/litre) (predicted)
2,3,4,6,7,8-HxBDF				-9.43[d]		8.31[d]	5.86[d]
1,2,3,4,6,7,8-HpBDF					9×10^{-11}[g]		

[a] From Gilman & Dietrich (1957). Melting points in parentheses are values from other sources reported by these authors.
[b] From Rordorf (1987).
[c] n.g. = not given.
[d] Predicted; from Fiedler & Schramm (1990). Sorption coefficient [log K_{oc}] = distribution coefficient between compound adsorbed to soil organic carbon and the compound in solution.
[e] From Tomita et al. (1959)
[f] From Kende & Wade (1973).
[g] From Rordorf et al. (1990).
[h] From Jackson et al. (1993), estimated from measured reverse-phase high-performance liquid chromatography (HPLC) retention times.
[i] From Jackson et al. (1993), calculated.
[j] From Denivelle et al. (1960).
[k] From Watanabe & Tatsukawa (1990).
[l] From Tashiro & Yoshiya (1982).

additives such as Sb_2O_3, as well as the efficiency of the apparatus used for the thermal degradation. In laboratory experiments on the thermolysis of polybrominated flame retardants (see section 3.4; Table 11), the PBDDs/PBDFs formed were destroyed at 800 °C in an air atmosphere after a 2.0-second residence time (Striebich et al., 1991). PBDDs/PBDFs formed at 600 °C from the thermolysis of plastics containing DBDE or PBDE were no longer detectable at 800 °C (Lahaniatis et al., 1991). However, Thoma et al. (1987b) found that PBDDs/PBDFs are still formed at 900 °C. There is thus no definitive information on the temperature needed to destroy PBDDs/PBDFs.

2.2.3 *Chemical reactions*

Aromatic carbon–bromine bonds are generally weaker than similar carbon–chlorine bonds, and, consequently, bromine can be substituted more easily. In general, the reductive substitution of halogens in aromatic structures becomes easier as the halogen atoms' size increases (Wania & Lenoir, 1990).

In the presence of excess chlorine, bromine can be substituted by chlorine to give PXDDs/PXDFs — for example, under conditions such as those present in municipal incinerators (Wilken et al., 1990; Luijk et al., 1992a).

Wania & Lenoir (1990) investigated the effect of heating 1,2,3,4-TeBDD (20 µg) in the presence of copper (1 g) at 100, 120, 150, or 210 °C for a duration of 30 seconds to 1 h. With increasing heating time, the spectrum of PBDD shifted from tetraBDD to lower brominated congeners, and the sum of the quantities decreased. The reaction rate increased with increasing temperature. At 210 °C for 30 min, all PBDDs had disappeared, but the dibenzo-*p*-dioxin ring structure remained intact.

In a further experiment, it was shown that the presence of water (10 or 100 µg/litre) considerably increased the yield of debrominated products.

On heating monoBDD and 1 g copper to 150 °C, the debrominated product dibenzo-*p*-dioxin and dimers of this compound were

identified. On heating hexaBDD and octaBDD to 150 °C in the presence of copper, it was found that appreciable quantities of the original compounds could still be detected after 1 h. The reaction was considerably slower than in the comparable experiment with tetraBDD.

The debromination reactions proceed faster than the respective dechlorination reactions with PCDDs (Hagenmaier et al., 1987).

2.3 Conversion factors

At 25 °C and 101.3 kPa, conversion factors for converting airborne concentrations from ppm to mg/m^3 for a particular PBDD/PBDF congener can be calculated from the relative molecular mass (RMM):

1 ppm = RMM/24.45 mg/m^3
1 mg/m^3 = 24.45/RMM ppm

For example, for monoBDF, 1 ppm = 247.1/24.45 = 10.1 mg/m^3. Similarly, 1 mg/m^3 = 0.099 ppm.

2.4 Analytical methods

2.4.1 General aspects

Some PBDD/PBDF congeners are highly toxic (see chapter 7). Using the principles of Good Laboratory Practice, great precautions should be taken in handling the samples. Additionally, precautions must be taken owing to the photochemical instability of the brominated and mixed brominated/chlorinated congeners (see also sections 2.2.2.1 and 4.2.1). The use of amber-coloured glassware and filters on lamps and windows is mandatory.

Sampling, sample treatment (extraction and clean-up), and analysis for PBDDs/PBDFs and PXDDs/PXDFs follow largely the methods and techniques currently used for PCDDs/PCDFs (Donnelly et al., 1989a,b, 1990; US EPA, 1990, 1992 [Methods 1613 and 8290]; Maier et al., 1994; Ballschmiter & Bacher, 1996). The large number of isomers in some homologous groups (see Table 1) makes the

separation and quantification of individual congeners difficult. Using highly selective, specific, and sensitive analytical methods, 2,3,7,8-substituted PBDDs/PBDFs can be detected, although co-elution with other isomers cannot be excluded.

Accurate identification of specific congeners is limited by the small number of reference standards available. The large number of PXDDs/PXDFs (see section 2.1) makes it impossible to identify and quantify individual congeners. Homologue groups, however, can be analysed semi-quantitatively. Major steps in the analytical procedures are as follows:

- spiking of the homologous sample with labelled standards
- use of matrix-specific extraction procedures (pretreatment of the sample before extraction where necessary)
- clean-up by column chromatography, liquid–liquid extraction, HPLC
- concentration of the eluate (addition of a high-boiling solvent as a keeper where necessary); addition of a recovery standard
- analysis by GC/MS.

Many of the analytical methods for PCDDs/PCDFs have been validated in the past decade in interlaboratory studies organized, among others, by: the World Health Organization (WHO) for biological samples (WHO/EURO, 1989, 1991; Stephens et al., 1992; WHO/ECEH, 1996); the European Community Bureau of Reference for environmental samples, including fly ash (Maier et al., 1994), and for milk powder (Schimmel et al., 1994; Tuinstra et al., 1996); and the European Committee for Standardization for emissions by stationary sources (Bröker, 1996). To avoid systematic errors in the individual steps from sampling to analysis, recoveries must be controlled by the addition of appropriate stable isotope labelled standards before extraction of the sample, clean-up, and final quantification.

2.4.2 *Sampling and extraction*

The sampling procedures recommended for PCDDs/PCDFs (WHO, 1989 and citations in second paragraph of section 2.4.1) also apply for the brominated congeners.

2.4.2.1 *Ambient air, airborne dust, automobile exhaust, flue gas, and products of thermolysis*

Experience from PCDD/PCDF analysis has shown critical or weak points in current gas sampling (ambient air, indoor air, exhaust gas) techniques. Requirements are as follows:

- representativeness of samples; special attention must be given to isokinetic sampling of particles in emission samples
- stability of the sample on the sampling medium during the sampling period; the filter should be kept below 120 °C and protected from light
- recovery of the analytes from the sampling train (as checked by appropriate spiked standards)
- use of clean equipment to avoid contamination of the sample (as checked by appropriate blanks)
- complete trapping of gas and particle phases (aerosols) to avoid sample losses.

Quartz fibre filters with polyurethane foam plugs have been used to sample ambient air up to 1000 m^3 (Wagel et al., 1989; Päpke et al., 1990; Harless et al., 1992; Watanabe et al., 1992). (Note: There may be interferences from brominated organic aromatic flame retardants in polyurethane foam.)

Haglund et al. (1988) described a method to collect both the particulate phase (using a Teflon-coated filter) and the gas phase (cryotechnique) in vehicle exhaust.

Hutzinger et al. (1990) used the so-called Grimmer apparatus to sample automobile exhaust: the sampling train consists of a large glass condenser and a non-impregnated fibreglass filter. Typically, 50 m^3 of automobile exhaust were taken per sample; the temperature at the muffler outlet was kept below 50 °C. The experiments were carried out as stationary motor tests. The total emissions of the motor were sucked through the sampling train by a pressure-controlled blower.

Emissions from a laboratory furnace experiment were collected by a sampling train, including a high-efficiency quartz fibre filter (to

collect organic-laden particulate material) and an XAD-2 resin (to adsorb semivolatile organic compounds) (Riggs et al., 1992).

Thermolytic products have been collected as condensate in a quartz-wool-filled condenser tube (Neupert et al., 1989b).

2.4.2.2 *Water and aqueous samples*

Analysis of water samples should follow a different approach. If the samples are free of particles, a normal liquid–liquid extraction is sufficient. If, however, the samples contain particles, both the particles and the water phase should be extracted separately — the solids by methods recommended for solids, the water phase as described above.

2.4.2.3 *Environmental samples: soil, sediment, and sewage sludge*

For environmental samples, problems arise in obtaining a representative sample. For soil sampling, a method was described by Fortunati et al. (1994).

Prior to the extraction, appropriate measures should be taken to ensure that PHDDs/PHDFs in the sample material are fully accessible to the extraction solvent. In a number of applications, this includes a digestion of the sample (solids) and/or the complete removal of water (wet solid samples) prior to extraction clean-up; a chemical destruction of non-persistent chemicals can be useful by incubating the sample in neat sulfuric acid (H_2SO_4). PHDDs/PHDFs are shown to be stable. Treatment with (strong) bases should be avoided, as PHDDs/PHDFs may degrade.

For the study of sewage samples, Hagenmaier et al. (1992) dried, powdered, and extracted the samples with toluene for 18 h. After concentration, the extracts were treated with concentrated sulfuric acid.

Proven digestion and water removal methods are treatment with hydrochloric acid (10% HCl) and Dean Stark collector (US EPA, 1990; Rappe et al., 1996), respectively. Sediments should be treated with copper powder to eliminate sulfur (Kjeller et al., 1993).

2.4.2.4 *Flame retardants, polymers, fly ash samples, dust, soot, and fire residues*

In general, the analysis of plastics is performed by dissolving the polymer in a suitable solvent. Non-dissolvable plastics should be powdered and Soxhlet-extracted.

Dibromomethane was used to dissolve samples of PBDE (Tondeur et al., 1990). Ranken et al. (1994) noted that this solvent must first be specially purified before use to remove the colour, which caused quantitative interferences in the mass spectrometer. TBBPA can be dissolved in methanol (Tondeur et al., 1990; Ranken et al., 1994).

PBT resins (extruded beads/powder) were extracted best with 1,1,1-trichloroethane/phenol followed by water partitioning of phenol; powdered high-impact polystyrene (HIPS) samples by toluene/reflux; and powdered ABS samples by dichloromethane (Donnelly et al., 1989a).

Kieper (1996) used toluene for Soxhlet extraction from samples of flame-retarded polymers: DBDE (with polystyrene/polystyrene-butadiene), 1,2-bis(tribromophenoxy)ethane (with polystyrene), TBBPA-carbonate oligomer (with PBT), dibromostyrene, and tribromostyrene (both with polyamide 66).

Samples of burnt plastic, PBT material, ash/slag, and soil were Soxhlet-extracted with dichloromethane (Neupert & Pump, 1992). Clausen et al. (1987) used Soxhlet extraction with hexane. ABS was extracted under reflux with methylene chloride (Donnelly et al., 1990).

Dry fly ash was treated with 10% HCl, dried, and neutralized. After further drying, the sample was Soxhlet-extracted with toluene (Hosseinpour et al., 1989). Similar procedures were used by Tong et al. (1991) and Huang et al. (1992a,b).

PBDDs/PBDFs from dust samples and smoke condensate were Soxhlet-extracted with toluene (UBA, 1992; Funcke et al., 1995).

Samples from fire residues were ground and extracted with toluene; wipe samples of soot were extracted with hexane (Harms et al., 1995).

2.4.2.5 *Biological matrices: human milk, blood/plasma, tissues, and fish samples*

For biological samples, most appropriate extraction methods are those giving the highest yields (or recovery) for the lipids in the sample (i.e. milk, blood, tissue).

Neupert et al. (1989a,b) quantified PBDDs/PBDFs in rat liver, adipose tissues, and faeces. After homogenization with sodium sulfate, extraction was performed on a multiple-layer column using dichloromethane/hexane.

Fish samples were ground with sodium sulfate and homogenized. Methanol and sodium oxalate were added to milk samples (De Jong et al., 1992). Diethyl ether and hexane were used to extract PHDDs/PHDFs from the fat fraction of the milk and fish samples.

For PBDD/PBDF determination, samples of human adipose tissue were homogenized, extracted with dichloromethane, dried with sodium sulfate, and solvent-exchanged into hexane (Cramer et al., 1990a). This method was also used by Zober et al. (1992).

Fat removal can be performed utilizing a semipermeable membrane technique (Bergqvist et al., 1993), which enables larger amounts of fat (sample size up to 200 g) to be eliminated from the sample matrix (>95%) and improved detection limits.

Lyophilization has been used successfully in the analysis of TBDD in biological matrices such as rat livers or marmoset monkey tissues (Schulz-Schalge et al., 1991a,b; Schulz et al., 1993; Nagao et al., 1995/96).

2.4.3 *Sample clean-up*

Sample clean-up is carried out to remove those materials that might otherwise interfere with the analysis. A variety of liquid chromatography separations have been used, including silica, florisil,

alumina, and various combinations of these columns. Usually an acid/base wash followed by alumina column chromatography is used to remove the bulk of interferences, and carbon column chromatography is used to remove residual interferences (Donnelly et al., 1986, 1987, 1989b).

Where PBDEs are likely contaminants, a modification of separation techniques is necessary. Alumina columns are ineffective in separating PBDFs from PBDEs. Carbon columns were found to be more effective, but the higher brominated PBDFs could be removed from the column only by back-flushing with an aromatic solvent (Donnelly et al., 1987; Hileman et al., 1989). Bonilla et al. (1990) introduced an HPLC step to the clean-up procedure. The sample was passed through an AX21 carbon column. The column was washed in the forward direction with dichloromethane/cyclohexane and dichloromethane/methanol/benzene and back-flushed with toluene. This procedure decreased the PBDE concentration in the final sample by six orders of magnitude.

Depending on the aim of the analysis (general surveying or specific search and quantification of 2,3,7,8-substituted congeners), a number of ^{13}C standards are required to be added at several stages during sampling and analysis.

In the WHO interlaboratory calibration study for the analysis of PCDDs/PCDFs in human blood and milk, the basic clean-up/separation methods used by some laboratories were activated carbon as the primary PCDD/PCDF isolation step followed by alumina; the remaining laboratories used other procedures, mostly H_2SO_4 followed by alumina. Nearly all methods used a step involving some type of carbon chromatography (Stephens et al., 1992).

2.4.4 Separation

GC is used for the separation of PBDDs/PBDFs. PBDDs/PBDFs have much higher retention times and elution temperatures (30–40 °C higher) than their chlorinated analogues (Buser, 1991). The higher brominated congeners have extremely long retention times, so non-polar (SE 54), medium-length (up to 25-m) columns are generally

used. Such a column is suitable for separating the PBDD and PBDF homologues. Elution temperatures on a 25-m SE 54 high-resolution gas chromatography (HRGC) column range from 184–188 °C for monoBDDs/BDFs to 260–273 °C for pentaBDDs/BDFs. Hexa-, hepta-, and octa-homologues elute during the isothermal phase at 280 °C (Buser, 1986a, 1991). Cross-linked columns allow higher temperatures, reducing analysis times (Hutzinger et al., 1990). Table 6 gives retention indices (RIs) of some PBDDs/PBDFs as well as PBDEs, which are possible contaminants (Donnelly et al., 1987). A 30-m DB-5 or DB-5MS fused capillary column has been found to be quite useful for the GC/MS analysis of tetra- through hepta-substituted PBDDs/PBDFs (Ranken et al., 1994).

Table 6. Retention indices (RIs) of PBDDs, PBDFs, and PBDEs[a,b]

Congener	RI
PBDDs	
2-MoBDD	1868
2,8-DiBDD	2174
1,3,7-TrBDD	2423
2,3,7-TrBDD	2475
2,3,7,8-TeBDD	2800
1,2,7,8-TeBDD	2811
1,2,4,7,8-PeBDD	3072
1,2,3,7,8-PeBDD	3145
1,2,3,4,7,8-HxBDD	3412
1,2,3,6,7,8-HxBDD	3475
1,2,3,7,8,9-HxBDD	3798
1,2,3,4,6,7,8-HpBDD	3763
OcBDD	4219
PBDFs	
2-MoBDF	1834
2,8-DiBDF	2133
1,2,8-TrBDF	2416
2,3,8-TrBDF	2433
1,2,7,8-TeBDF	2740
2,3,7,8-TeBDF	2791
1,2,3,7,8-PeBDF	3103

Table 6 (contd).

Congener	RI
1,2,3,6,7,8-HxBDF	3479
1,2,3,4,6,7,8-HpBDF	3806
OcBDF	4231
PBDEs	
HexaBDE	2888
HexaBDE	3004
HexaBDE	3015
HexaBDE	3030
HexaBDE	3051
HexaBDE	3095
HexaBDE	3286
HexaBDE	3314
HexaBDE	3369
HexaBDE	3411
OctaBDE	3525
OctaBDE	3577
OctaBDE	3601
OctaBDE	3627
OctaBDE	3654
OctaBDE	3737
OctaBDE	3786
NonaBDE	3951
NonaBDE	4003
DecaBDE	4310

[a] From Donnelly et al. (1987).
[b] Chromatographic conditions: 30 m x 0.32 mm DB-5 GC column; He carrier gas at ca. 7 psi head pressure; temperature programmed from 10 min at 170–320 °C at 8 °C/min.

The elution of PCDDs/PCDFs and PBDDs/PBDFs occurs in the order of the molecular weights. PXDDs/PXDFs elute between the corresponding chloro- and bromo-analogues (Buser, 1987a). However, mixed congeners containing bromine elute earlier than expected on a molecular weight basis, relative to the chloro-compounds (e.g. $BrCl_5$

before Cl_7) (Buser, 1991). (For isomer-specific analysis, columns of different polarity should be used.)

Owing to the lack of PBDD/PBDF (and PXDD/PXDF) standards, it has not been possible to identify all congeners. Instead, a combination of MS and GC RI identification has to be used for the analysis of 2,3,7,8-substituted PBDDs/PBDFs, PCDDs/PCDFs, and PXDDs/PXDFs. An RI model has been developed to predict the GC retention times for 1700 of these compounds (Donnelly & Sovocool, 1991; Donnelly et al., 1991a,b).

2.4.5 *Detection, quantification, and confirmation of PBDDs/PBDFs by MS techniques*

Detection, quantification, and confirmation are usually performed by MS, as only this technique shows sufficient selectivity to distinguish PBDDs/PBDFs from other halogenated compounds (e.g. PBDEs) that are present in the sample. MS allows the determination of the number and type of halogens present from characteristic isotope distribution patterns, but it does not give any information about which isomer is present (Buser, 1991). (Among the MS methods, high-resolution mass spectrometry [HRMS] is preferred owing to higher selectivity, and the tandem MS or negative ion chemical ionization [NCI] techniques are a useful screening method because of the diagnostic Br [79/81] fragment.)

Donnelly et al. (1987) and Sovocool et al. (1987) developed and refined US Environmental Protection Agency (EPA) Method 8280 to measure PBDFs/PBDDs by GC/MS. Significant features of the mass spectra reported by these authors include the sequential losses of $Br^{\bullet}$, $COBr^{\bullet}$, ($Br^{\bullet}$+$COBr^{\bullet}$), and ($2Br^{\bullet}$+$COBr^{\bullet}$). PBDEs may interfere with the determination of PBDFs (and PBDDs), as the $(M\text{-}2Br)^{+\bullet}$ fragment has the same *m/z* composition cluster as that of a PBDF with two fewer bromines; additional fragmentation mimics the PBDF containing two fewer bromines (Donnelly et al., 1987). This potential for co-elution (see Tables 6 and 7) must be considered in the evaluation of reports of PBDF (and PBDD) formation where PBDE interference is likely.

Table 7. Molecular ions (M^+, ^{79}Br isotope) of PBDDs, PBDFs, and PBDEs showing possible interference during monitoring and determination[a]

Compound	Brominated congeners									
	mono-	di-	tri-	tetra-	penta-	hexa-	hepta-	octa-	nona-	deca-
PBDFs	246	324	402	480	556	636	714	792		
PBDEs	248	326	404	482	560	638	716	794	872	950
PBDDs	262	340	418	496	574	652	730	808		

[a] From Buser (1986a) and Donnelly et al. (1987).

Confirmation criteria for the detection and quantification of PBDDs/PBDFs have been proposed (Donnelly et al., 1987):

- The retention time/RI must be correct for that analyte (standards are needed).
- Recovery of the "surrogate" standard should be in the 40–120% range.
- All *m/z* monitored for a given analyte must maximize simultaneously ± 1 second, with a signal to noise ratio greater than or equal to 2.5 for each. The $M^{+\cdot}$ cluster is relatively intense for all congeners. For confirmation, two additional ions (*m/z*) should be monitored in electron impact–selective ion monitoring–mass spectrometry (EI-SIM-MS).
- The ratio between the two ions of the $M^{+\cdot}$ cluster must be within 20% (relative) of the theoretical.
- When monitoring for PBDFs, the absence of PBDE should be demonstrated (see Table 7).

The identification and determination of positional isomers are very complex. Complementary methods based on other instrumental techniques such as GC/matrix isolation infrared spectrometry (MI-IR) have been developed to allow the unambiguous identification of each individual compound at high concentrations. For 2,3,7,8-TeBDD and 2,3,7,8-TeBDF, the most intense matrix isolation infrared band is given by frequencies 1478 and 1434, respectively (Wurrey et al., 1989; Childers et al., 1992). Childers et al. (1992) gave additional frequencies for some PXDDs/PXDFs.

2.4.6 *The need for analysis of 2,3,7,8-substituted congeners*

Table 2 (section 2.1) gives the PBDD/PBDF congeners substituted with bromine in the 2,3,7,8-positions. As these are the most toxic congeners, in some investigations only these congeners are determined.

The German Dioxin Directive (1994) established limitations on the concentrations of certain 2,3,7,8-substituted PBDDs/PBDFs in products to be placed on the market (see Appendix I). In 1987, the US EPA issued a Test Rule requiring manufacturers and importers of certain halogen-containing chemicals to analyse their products for 2,3,7,8-substituted PHDDs/PHDFs (US EPA, 1987) (see Appendix I).

Owing to these actions, activities in PBDD/PBDF analysis have improved and 2,3,7,8-substituted standards have been synthesized; in 1995, 12 of the 17 2,3,7,8-substituted isotopically labelled and 11 of the 17 2,3,7,8-substituted native PBDDs/PBDFs were available.

2.4.7 *Interfering substances*

For a general discussion on the problem of interfering substances, see Buser (1991). Substances possibly interfering with PBDF determinations include PBDEs, as discussed above. This is of particular importance in the analysis of substances containing this flame retardant, their thermolytic products, and environmental samples where this flame retardant is implicated. PBBs can be separated from PBDFs/PBDDs and are not a source of cross-contamination (Donnelly et al., 1987).

2.4.8 *Standards*

As mentioned in section 2.4.6, a number of PBDD/PBDF standards have been made commercially available in recent years, in particular the 2,3,7,8-substituted congeners. The following is therefore more of historical than of practical interest.

For use as standards, samples of mono- through octaBDDs (Munslow et al., 1987) and mono- through octaBDFs (Sovocool et al., 1987), as well as PXDDs/PXDFs (Donnelly et al., 1987, 1989b), can

be synthesized by the electrophilic bromination of dibenzo-*p*-dioxin and dibenzofuran. Special reaction conditions optimize the selectivity and the corresponding yield. Extended reaction times result in a higher degree of bromination; elevated temperature and increasing amount of iron/iron (III) chloride ($FeCl_3$) catalyst can be used to accelerate the reaction (Hutzinger et al., 1989).

Donnelly et al. (1991a) prepared over 100 PHDDs using self-condensation of halogenated phenols, coupling of halogenated catechols with halobenzenes or halonitrobenzenes, and electrophilic halogenation.

Ramalingam et al. (1986) synthesized PBDDs from bromocatechol and polybromonitrobenzene in the presence of anhydrous potassium carbonate (K_2CO_3) in acetone. For PXDDs, bromocatechol or chlorocatechol was refluxed with polychloronitrobenzene or polybromonitrobenzene in the presence of anhydrous K_2CO_3 in acetone.

Mixed halogenated compounds can be prepared by the bromination of PCDDs and PCDFs, by the chlorination of PBDDs and PBDFs, or by halogen exchange (Buser, 1987a).

Chatkittikunwong & Creaser (1994b) described the synthesis of PBDDs/PBDFs and PXDDs/PXDFs by electrophilic halogenation of substituted precursors with iron (III) halides (in the absence of halogen).

Jay & Stieglitz (1996) prepared PBDFs and PCDFs from reaction of copper (II) bromide ($CuBr_2$) or copper (II) chloride, dihydrate ($CuCl_2 \cdot 2H_2O$), respectively, with dibenzofuran. For synthesis of PXDFs, the brominated reaction mixture was further reacted with $CuCl_2 \cdot 2H_2O$.

Nestrick et al. (1989) developed a procedure for synthesizing $^{13}C_{12}$-labelled PBDDs/PBDFs from their chlorinated analogues.

3. FORMATION AND SOURCES OF HUMAN AND ENVIRONMENTAL EXPOSURE

PBDDs/PBDFs are not known to occur naturally. They are not intentionally produced but are formed as undesired by-products in various processes. They can be formed by chemical, photochemical, or thermal reactions from precursors or by so-called *de novo* synthesis (from organic materials with bromine). PBDDs/PBDFs have been found as contaminants in brominated organic chemicals. Thermolysis of brominated flame retardants, in particular PBDEs, has been implicated as an obvious source of PBDDs/PBDFs. Heating and burning of products containing such brominated compounds can cause emission of PBDDs/PBDFs. PBDDs/PBDFs have also been detected in traffic-related emissions.

The formation of PXDDs/PXDFs is possible in combustion processes if both bromine and chlorine are present (Buser, 1987a,b), such as in waste incineration, in particular of old computer/business machines, and in motor combustion processes.

3.1 Synthesis and use

PBDDs/PBDFs have no commercial use and are synthesized for research purposes only and as standards for analytical determination (see section 2.4.8). As an exception, a patent was awarded for the use of heptaBDF as a flame retardant (Richtzenhain & Schrage, 1977).

3.2 By-products of brominated organic chemicals (including flame retardants)

Theoretically, some 40 brominated organic chemicals may be contaminated with PBDDs/PBDFs. Such chemicals include flame retardants and fire extinguishers, pesticides (e.g. bromophenols, bromophos, bromoxynil, profonofos), solvents, and chemical intermediates or additives (Esposito et al., 1980; Lee et al., 1986, 1987; Johnson et al., 1989; Bretthauer et al., 1991). Possible PBDD/PBDF formation pathways have been suggested. Chemicals considered as being important are TBBPA and its derivatives, penta- (PeBDE), octa-, and decaBDE, 2,4,6-tribromophenol, and 1,2-bis(tribromophenoxy)-ethane (Johnson et al., 1989).

Analytical data on the occurrence of PBDDs/PBDFs in brominated organic chemicals are scarce. Tables 8 and 9 (concentrations of PBDFs and PBDDs, respectively) include data on PBDEs, PBBs, TBBPA, 1,2-bis(tribromophenoxy)ethane, brominated phenols, brominated anilines, brominated styrenes, and others.

The highest concentrations of PBDFs were found in PBDEs (up to 8000 μg/kg). Maximum PBDF values measured in DBB, TBBPA, bromophenols, and bromoanilines were approximately 115 μg/kg, 64 μg/kg, 31 μg/kg, and 2 μg/kg, respectively. PBDF levels ranging from about 92 to 500 μg/kg were observed in distillation residues of bromophenols and bromoanilines (Table 8). This is of importance, particularly in synthetic and analytical laboratories and laboratory waste disposal (Vogt et al., 1994a,b).

The highest concentration of PBDDs (~8500 μg/kg) was found in 1,2-bis(tribromophenoxy)ethane, followed by 86 μg/kg in 2,4,6-tribromophenol and 8 μg/kg in TBBPA. As seen with PBDFs, PBDDs were strongly enriched in distillation residues of selected bromophenols (Table 9). Additionally, Ritterbusch et al. (1994a) reported the occurrence of PBDFs (mono to penta: 12.1 μg/kg) and PBDDs (mono to penta: 1.1 μg/kg) in solvent wastes of chemical laboratories.

2,3,7,8-Substituted PBDDs/PBDFs were not detected in TBBPA (Thies et al., 1990; Tondeur et al., 1990; Brenner & Knies, 1993a,b; Ranken et al., 1994), TBPI (Brenner & Knies, 1994), or 2,4,6-tribromophenol (Tondeur et al., 1990; Vogt et al., 1994a). A sample of commercial decaBDE was found to contain 1,2,3,7,8-PeBDF (1.6 μg/kg) and 1,2,3,4,7,8-HxBDF (37 μg/kg). Other 2,3,7,8-substituted tetra- to hexaBDDs/BDFs did not exceed 0.1–5.1 μg/kg. All concentrations given were maximum values, because co-elution was possible (UBA, 1992). 2,3,7,8-Substituted PBDDs/PBDFs were not detected in multiple samples (n = 21; three companies) of commercial decaBDE at target limits of quantitation (according to the US EPA Test Rule; see Appendix I) ranging from 0.1 to 1.0 μg/kg, from 0.5 to 5 μg/kg, from 2.5 to 25 μg/kg, and from 100 to 1000 μg/kg for tetra-, penta-, hexa-, and hepta-substitution, respectively. 1,2,3,4,6,7,8-HpBDF was found in all samples at concentrations of 56–300 μg/kg, which were well

Table 8. Concentrations of PBDFs found in brominated organic chemicals

Chemical	Concentrations of PBDFs (µg/kg)[a]									Reference
	Sum	MoBDFs	DiBDFs	TrBDFs	TeBDFs	PeBDFs	HxBDFs	HpBDFs	OcBDF	
PBDE (tetra- to hexaBDE) (commercial)	8000	–	–	–	2000	4000	2000	–	–	Hileman et al. (1989)
PBDE (hexa- to nonaBDE) (commercial)	>4000	–	–	–	n.d. (~200)	2000–4000	2000–4000	present[b]	–	Hileman et al. (1989)
DBDE (commercial)	–	–	–	–	n.d. (~200)	n.d. (~200)	200	present[b,c]	–	Hileman et al. (1989)
DBDE (commercial)	286.3	–	–	–	–	–	2.3	250	34	Donnelly et al.
(concentrate)	6880	–	–	–	23	107	3470	2700	580	(1989a)
DBDE (commercial)	2037	–	0.04	<0.9	0.15	<0.01	<0.2	1842	195	Brenner & Knies (1990)
DBDE (commercial)	79.3	0.4	0.3	0.3	0.8	10.5	67.0	–	–	UBA (1992)
DBDE (commercial)	n.d.	n.d. (0.08)	n.d. (0.08)	n.d. (0.08)	n.d. (0.8)	n.d. (1.5)	n.d. (1.8)	–	–	Kieper (1996)
DBB (commercial)	115	99.9	9.0	5.7	<1	<5	<5	<10	<10	Atochem (1990)
TBBPA + derivatives (technical grade)	63.6	–	n.d.	n.d.	n.d.	1.0	12.2	31.5	18.9	Thoma et al. (1986b); Dumler et al. (1990c)
TBBPA + derivatives (commercial)	>3	2	1	n.d. (<0.5)	n.d. (<1)	n.d. (<2)	<14	–	–	Thies et al. (1990)

Table 8 (contd).

Chemical	Concentrations of PBDFs (µg/kg)[a]									Reference
	Sum	MoBDFs	DiBDFs	TrBDFs	TeBDFs	PeBDFs	HxBDFs	HpBDFs	OcBDF	
TBBPA + derivatives (BC 52) (commercial)	n.d. (0.001–0.4)	–	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	Brenner & Knies (1993a,b)
TBBPA-oligocarbonate	1.46	n.d. (0.01)	n.d. (0.01)	n.d. (0.01)	0.07	0.33	1.06	–	–	Kieper (1996)
TBPI (Saytex BT 93)	0.21	–	0	0	0.21	0	0	0	–	Brenner & Knies (1994)
Hexabromocyclo dodecane (technical)	50	–	<10	<10	20	30	<10	<10	–	Brenner (1993)
4-Bromophenol										Ritterbusch et
(crude)	1.56	0.08	0.72	0.54	0.22	–	–	–	–	al. (1994a);
(distilled)	1.19	0.44	0.75	n.d.	n.d.	–	–	–	–	Vogt et al.
(distillation residue)	378.54	63.47	230.46	69.17	15.44	–	–	–	–	(1994a)
(commercial)	0.37	0.06	0.31	n.d.	n.d.	–	–	–	–	
2,4-Dibromophenol										Ritterbusch et
(crude)	3.30	0.16	1.43	1.44	0.12	–	–	–	–	al. (1994a);
(distilled)	0.85	0.36	0.49	n.d.	n.d.	–	–	–	–	Vogt et al.
(distillation residue)	498.99	62.47	353.51	68.91	4.3	–	–	–	–	(1994a)
2,4,6-Tribromo phenol										Thoma et al.
(technical grade)	31.4	–	2.2	16.2	12.0	1.0	n.d.	n.d.	n.d.	(1986b);
(crude)	4.6	0.79	2.66	1.19	n.d.	n.d.	n.d.	n.d.	–	Dumler et al. (1990c); Vogt et al. (1994a)

Table 8 (contd).

Pentabromophenol (analytical grade)	n.d.	–	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	Thoma et al. (1986b); Dumler et al. (1990c)
Tetrabromophthalic anhydride (analytical grade)	n.d.	–	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	Thoma et al. (1986b); Dumler et al. (1990c)
2,4,6-Tribromo aniline										Vogt et al.
(crude)	1.88	0.24	0.28	n.d.	0.25	1.01	–	–	–	(1994b)
(recrystallized)	0.47	n.d.	0.47	n.d.	n.d.	n.d.	–	–	–	
(distillation residue)	92.35	n.d.	3.15	8.40	15.90	64.90	–	–	–	
2,6-Dibromo-4-nitroaniline (crude)	0.90	0.33	0.57	n.d.	n.d.	n.d.	–	–	–	Vogt et al. (1994b)
1,2-Bis(tribromo-phenoxy)ethane	n.d.	n.d. (0.5)	n.d. (0.5)	n.d. (0.5)	n.d. (0.5)	n.d. (1.0)	n.d. (4.0)	–	–	Kieper (1996)
Polytribromostyrene	16.74	n.d. (0.06)	4.03	1.2	n.d. (0.06)	0.36	3.29	3.01	4.85	Kieper (1996)
Polydibromostyrene	24.09	0.09	0.26	0.21	n.d. (0.03)	n.d. (0.04)	2.33	2.93	18.27	Kieper (1996)

[a] – = no information; n.d. = not detected (detection limits in parentheses, if specified).
[b] Not quantifiable because of lack of standards.
[c] Major component.

Table 9. Concentrations of PBDDs found in brominated organic chemicals

Chemical	Concentrations of PBDDs (µg/kg)[a]									Reference
	Sum	MoBDDs	DiBDDs	TrBDDs	TeBDDs	PeBDDs	HxBDDs	HpBDDs	OcBDD	
DBDE (commercial)	0.4	–	–	–	0.05	0.35	–	–	–	Brenner & Knies (1990)
DBDE (commercial)	n.d. (0.1–<5.1)	n.d. (<0.1)	n.d. (<0.1)	n.d. (<0.1)	n.d. (<0.2)	n.d. (<0.7)	n.d. (<5.1)	–	–	UBA (1992)
DBDE (commercial)	n.d.	n.d. (0.03)	n.d. (0.03)	n.d. (0.03)	n.d. (0.03)	n.d. (0.1)	n.d. (0.35)	–	–	Kieper (1996)
TBBPA + derivatives (technical grade)	n.d.	–	–	n.d.	n.d.	–	–	–	–	Thoma et al. (1986b); Dumler et al. (1990c)
TBBPA + derivatives (commercial)	8	n.d. (<0.5)	n.d. (<0.5)	n.d. (<0.5)	1	2	5	–	–	Thies et al. (1990)
TBBPA + derivatives (BC 52) (commercial)	0.006	–	n.d. (0.001)	n.d.	0.006	n.d.	n.d.	n.d.	n.d. (0.4)	Brenner & Knies (1993a,b)
TBBPA-oligocarbonate	n.d.	n.d. (0.01)	n.d. (0.01)	n.d. (0.01)	n.d. (0.01)	n.d. (0.02)	n.d. (0.06)	–	–	Kieper (1996)
Hexabromocyclo dodecane (technical)	–	–	<10	<10	<10	<10	<10	<10	–	Brenner (1993)
4-Bromophenol										Ritterbusch et
(crude)	0.40	0.04	0.15	0.21	n.d.	–	–	–	–	al. (1994a);
(distilled)	0.14	0.07	0.07	n.d.	n.d.	–	–	–	–	Vogt et al.
(distillation residue)	39.0	14.41	12.84	11.75	n.d.	–	–	–	–	(1994a)
(commercial)	n.d.	n.d.	n.d.	n.d.	n.d.	–	–	–	–	

Table 9 (contd).

2,4-Dibromophenol										Vogt et al.
(crude)	0.16	0.03	0.13	n.d.	n.d.	–	–	–	–	(1994a);
(distilled)	0.08	0.04	0.04	n.d.	n.d.	–	–	–	–	Ritterbusch et
(distillation residue)	18.75	1.37	3.13	10.76	3.49	–	–	–	–	al. (1994a)
2,4,6-Tribromo phenol										Thoma et al.
(technical grade)	85.5	–	–	1.5	84.0	–	–	–	–	(1986b); Dumler
(crude)	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	–	et al. (1990c);
										Vogt et al.
										(1994a)
Pentabromophenol	n.d.	–	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	Thoma et al.
(analytical grade)										(1986b); Dumler
										et al. (1990c)
Tetrabromophthalic	n.d.	–	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	Thoma et al.
anhydride (analytical										(1986b); Dumler
grade)										et al. (1990c)
2,4,6-Tribromo aniline										Vogt et al.
(crude)	n.d.	–	–	–	n.d.	–	–	–	–	(1994b)
(recrystallized)	n.d.	–	–	–	n.d.	–	–	–	–	
(distillation residue)	5.45	–	–	–	5.45	–	–	–	–	
2,6-Dibromo-4-	n.d.	–	–	–	n.d.	–	–	–	–	Vogt et al.
nitroaniline (crude)										(1994b)

Table 9 (contd).

Chemical	Concentrations of PBDDs (µg/kg)[a]									Reference
	Sum	MoBDDs	DiBDDs	TrBDDs	TeBDDs	PeBDDs	HxBDDs	HpBDDs	OcBDD	
1,2-Bis(tribromo-phenoxy)ethane	8455	n.d. (1.0)	n.d. (1.0)	107	8348	n.d. (2.0)	n.d. (5.0)	--	--	Kieper (1996)
Polytribromostyrene	5.63	1.78	3.85	n.d. (0.02)	n.d. (0.02)	n.d. (0.03)	n.d. (0.11)	n.d. (0.19)	<0.38	Kieper (1996)
Polydibromostyrene	n.d.	n.d. (0.02)	n.d. (0.02)	n.d. (0.02)	n.d. (0.02)	n.d. (0.04)	n.d. (0.08)	n.d. (0.12)	<0.17	Kieper (1996)

[a] -- = no information; n.d. = not detected (detection limits in parentheses, if specified).

below the target limit of quantitation of 1000 µg/kg for this congener (Ranken et al., 1994). Another study (Kieper, 1996) analysed five different flame retardants (DBDE, 1,2-bis(tribromophenoxy)ethane, TBBPA-carbonate oligomer, di- and tribromostyrene) for a total of eight 2,3,7,8-substituted PBDDs/PBDFs (tetra to hexa). Contents, if any, were below the respective detection limits. If detection limits were included in the evaluation (as a concentration of half the detection limit), the sum concentrations would range from 0.2 to 18.5 µg/kg. (A problem may arise in evaluating the sum of PBDDs/PBDFs if the analytical limit of detection is rather high in some of the congeners. From a scientific point of view, values below the limit of quantification should not be used. However, for administrative purposes, sometimes a value of one-half the detection limit is assumed and included in the sum of PBDDs/PBDFs. A considerable difference may occur between a scientific and this "administrative" approach. It is highly recommended that the approach for the calculation be stated if values that have not been measured are included in the summation.)

2,3,7,8-TeBDD was absent in bromophenols, whereas 2,3,7,8-TeBDF was identified in crude 4-bromophenol and 2,4-dibromophenol at levels of 0.12 and 0.15 µg/kg, respectively, and in their distillation residues at 8.3 and 9.8 µg/kg, respectively (Vogt et al., 1994a). Both 2,3,7,8-substituted TeBDF and TeBDD were present (at 40 and 10 ng/kg, respectively) in solvent wastes of chemical laboratories (Ritterbusch et al., 1994a).

3.3 Formation from the photochemical degradation of brominated organic chemicals

The formation of PBDDs/PBDFs was observed under laboratory conditions after irradiation of PBDEs (Watanabe & Tatsukawa, 1987) or of bromophenols (Ritterbusch et al., 1994a) and is also believed to occur after outdoor exposure of PBDEs (Birla & Kamens, 1994).

The major photoproducts of the flame retardant DBDE (technical grade) irradiated in hexane solution by UV light and sunlight were lower brominated PBDEs and mono- to hexa-substituted PBDFs. PBDDs were not detected. Based on the kinetics of the reactions, the formation of PBDFs occurred secondarily from debrominated PBDEs

as photoproducts of DBDE, but not directly from DBDE. UV irradiation of DBDE for 16 h gave about a 20% yield of total PBDFs, with tetraBDFs, but not 2,3,7,8-TeBDF, being the main components (Watanabe & Tatsukawa, 1987).

The concentrations of PBDDs/PBDFs (mono to penta) in several bromophenol samples were drastically increased, up to three orders of magnitude, after UV irradiation for 60 min. Apparently, the rate of photochemical PBDD/PBDF formation was greater than the rate of degradation (see section 4.2.1). An exception was 2,3,7,8-TeBDF, which disappeared after irradiation. Both bromophenols themselves as well as their contaminants (PBDEs) may act as PBDD/PBDF precursors (Ritterbusch et al., 1994a). The highest levels were found with the mono- to tetrabrominated homologues. The total concentrations of dibenzofurans exceeded those of dibenzo-*p*-dioxins: 315.7 µg/kg versus 14.2 µg/kg after irradiation of crude 4-bromophenol, and 1750.7 µg/kg versus 301.0 µg/kg after irradiation of crude 2,4-dibromophenol (Ritterbusch et al., 1994a).

The atmospheric stability of PBDDs/PBDFs that resulted after combustion of polyurethane foam (containing PBDEs) at a range of temperatures was examined under sunlight conditions. The formation of PBDFs, primarily tetra- and pentaBDFs (isomers not examined), in the presence of sunlight was seen with the products of low-temperature (400–470 °C) combustion. The formation of the PBDF compounds in this case was thought to be the result of the photolysis of unburned PBDEs (Birla & Kamens, 1994).

For photochemical transformations of higher brominated PBDDs/PBDFs to lower brominated congeners, see section 4.2.1.

3.4 Formation from the laboratory thermolysis of bromine-containing flame retardants

The potential of typical brominated flame retardants to form PBDDs/PBDFs was examined under various conditions in a series of

laboratory thermolysis[a] experiments (see also Tables 10–12). The flame retardants were thermally treated either alone (Buser et al., 1978; Thoma & Hutzinger, 1987a,b; Dumler et al., 1989a, 1990b; Zacharewski et al., 1989) or blended with polymer matrices (Dumler et al., 1990b,c; Riggs et al., 1990; Lahaniatis et al., 1991; Lorenz & Bahadir, 1993). PBDFs were found in most of the samples, but both the concentration and the degree of bromination varied greatly. PBDDs were detected to a lesser extent. Owing to the different conditions used, it is difficult to compare these studies quantitatively — except for results from the same experimental series.

The largest yields of PBDDs/PBDFs were obtained from PBDEs (especially in combination with polymers) and from bromophenols, both reaching values in the g/kg range. In contrast to PBDEs, the bromophenols that are flame retardants of only limited use (BMU, 1989) were not tested in polymer matrices. About an order of magnitude lower yields of PBDDs/PBDFs were observed upon thermolysis of certain PBBs. Again, lower but significant amounts of PBDDs/PBDFs in the mg/kg range were generated by TBBPA and 1,2-bis-(tribromophenoxy)ethane. Formation of PBDDs/PBDFs from all other flame retardants tested was very low or undetectable (Tables 10 and 11). Among the PBDEs, the yield of PBDDs/PBDFs in thermolytic residues decreased from pentaBDE to octaBDE to decaBDE (e.g. Buser, 1986a; Thoma et al., 1987a; Luijk et al., 1991).

The optimum PBDF formation temperatures of flame retardants thermally treated alone were found to be in the range of 600–900 °C. For example, bromophenols and TBBPA showed PBDF formation maxima at 800 °C, pentaBDE at 700–800 °C, and decaBDE at 800–900 °C, whereas PBBs (hexabromobiphenyl, or hexaBB) had no clear peak concentration between 700 °C and 900 °C on pyrolysis in quartz tubes (Thoma et al., 1986a, 1987a). Under other experimental conditions, decaBDE (alone) produced maxima at 600 °C (Dumler et al., 1990a) or 700 °C (Dumler et al., 1989a,c, 1990b,c). When decaBDE was burned in a polymer matrix, the PBDF formation maximum was shifted to lower temperatures (Thoma et al., 1987a;

[a] For definitions of terms referring to thermal treatment, see Appendix II.

Table 10. Survey on the generation of PBDFs and PBDDs during thermolysis of bromoorganic flame retardants

Flame retardant	Conditions of thermolysis[a]	PBDFs[b]	PBDDs[b]	Concentrations (mg/kg)[b,c]	Reference
PBDEs					
Technical PBDE	quartz minivials				Buser (1986a)
	in air 510 °C	mono to penta	mono to tetra	∑ PBDDs/PBDFs: 5000–10 000	
	in air 630 °C	mono to hexa	mono to penta	∑ PBDDs/PBDFs: 100 000	
Technical PBDE (Bromkal 70-5 DE, 70 DE, and G1)	quartz tubes				
	700–900 °C	mono to penta	mono to tetra	TeBDFs: up to 330 400 (700 °C) TeBDDs: up to 15 400 (700 °C)	Thoma et al. (1987a)
	800 °C	mono to penta	mono to tetra	∑ PBDFs/PBDDs: up to 610 393	Zacharewski et al. (1988)
Technical PeBDE (Bromkal 70-5 DE)	various types of ovens (DIN, BIS, VCI)				Dumler et al. (1987);
	600 °C	di to penta	di to tetra	TeBDFs: up to 87 827 TeBDDs: up to 12 374	Hutzinger et al. (1989)
Technical PeBDE	quartz minivials				Luijk et al. (1990, 1991)
	500 °C	tetra, penta	tetra, penta	∑ PBDFs/PBDDs: 12 000	
	600 °C	tetra to hexa	tetra to hexa	∑ PBDFs/PBDDs: 270 000	
Technical PeBDE (Bromkal 70-5 DE)	pyrojector, absence of oxygen (helium) 700–900 °C	di to tetra	none	small amounts	Thoma & Hutzinger (1987b, 1989)
Technical OBDE	quartz minivials				Buser (1986a)
	630 °C	tetra to hepta	tri to hepta	∑ PBDDs/PBDFs: 50 000	
Technical OBDE	quartz minivials				Luijk et al. (1990, 1991)
	600 °C	tetra to hexa	tetra to hexa	∑ PBDDs/PBDFs: 56 000	

Table 10 (contd).

Technical DBDE	quartz minivials 630 °C	tetra to hepta	tetra to octa	∑ PBDDs/PBDFs: 10 000–20 000	Buser (1986a)
Technical DBDE (FR 300 BA)	quartz tubes 700 °C	tetra to octa	hepta, octa	OcBDD/BDF: 2690	Thoma et al. (1987a)
	800 °C	tetra, hexa to octa	hepta, octa	OcBDD/BDF: 9230	
	900 °C	penta to octa	hepta, octa	OcBDD/BDF:13 413	
	800 °C	tetra, hexa to octa	hepta, octa	∑ PBDDs/BDFs: 10 935	Zacharewski et al. (1988)
Technical DBDE	VCI oven 400–1000 °C	hepta, octa	hepta, octa	n.sp.	Klusmeier et al. (1988)
Technical DBDE	VCI oven 300–800 °C	mono to hepta	tetra, hexa, hepta	∑ PBDFs: up to 7222 (700 °C) ∑ PBDDs: up to 588 (800 °C)	Dumler et al. (1989c); Hutzinger (1990)
Technical DBDE	DIN oven 400 °C	hexa to octa	hexa to octa	∑ PBDDs/PBDFs: 470/364	Dumler (1989); Dumler et al. (1990a)
	600 °C	hexa to octa	hexa to octa	∑ PBDDs/PBDFs: 2756/447	
	800 °C	tri to hepta	tri to octa	∑ PBDDs/PBDFs: 1114/690	
Technical DBDE	quartz minivials 600 °C	tetra to hexa	tetra to hexa	∑ PBDDs/PBDFs: 1700	Luijk et al. (1990, 1991)
Technical DBDE (Fr 300 BA)	pyrojector, absence of oxygen 700 °C	hepta, octa	none	n.sp.	Thoma & Hutzinger (1987b, 1989)

Table 10 (contd).

Flame retardant	Conditions of thermolysis[a]	PBDFs[b]	PBDDs[b]	Concentrations (mg/kg)[b,c]	Reference
Two technical PBDE mixtures (Br_3–Br_{10})	high-temperature flow reactor in nitrogen 650 °C in air 625 °C	 di to tetra tri, tetra	 none di to tetra	 ∑ PBDFs: 900 ∑ PBDFs: 600 ∑ PBDDs: 900	Striebich et al. (1990, 1991)
PBBs					
Technical hexaBB (FireMaster® FF-1)	glass tubes (open) in air 380–400 °C (sealed) in nitrogen 380–400 °C	 tetra, penta traces (tetra)	 n.a. n.a.	 TeBDFs/PeBDFs: 40/4	O'Keefe (1978)
Technical hexaBB (FireMaster® BP-6)	quartz tubes 700, 800, 900 °C 800 °C	 di to hepta tri to hepta	 none none	 TeBDFs: up to 1523 ∑ PBDFs: 2070	Thoma et al. (1987a) Zacharewski et al. (1988)
Technical hexaBB (FireMaster® BP-6)	pyrojector, absence of oxygen 600–900 °C	 none	 none		Thoma & Hutzinger (1987b, 1989)
Technical decaBB (Adine 0102)	glass tubes (loosely plugged) 800 °C	 none	 none	 –	Atochem (1987)

Table 10 (contd).

Bromophenols					
2-Bromophenol	3 different types of ovens 600 °C	mono to tri	mono, di	∑ PBDFs: up to 215 425 ∑ PBDDs: up to 60 634	Dumler et al. (1987); Hutzinger et al. (1989)
2,4,6-Tribromophenol	quartz tubes 700, 800, 900 °C	di to penta	di to hexa	TeBDDs: up to 896 000 TeBDFs: up to 8950	Thoma et al. (1986a)
2,4,6-Tribromophenol	3 different types of ovens 600 °C	tri to penta	di to penta	∑ PBDFs: up to 8820 ∑ PBDDs: up to 880 503	Dumler et al. (1987); Hutzinger et al. (1989)
2,4,6-Tribromophenol	pyrojector, absence of oxygen 600–900 °C	none	di to penta	n.sp.	Thoma & Hutzinger (1987b, 1989)
2,4,6-Tribromophenol	high-temperature flow reactor in nitrogen 625 °C in air 500 °C	n. a. none	none tetra	n.sp.	Striebich et al. (1990, 1991); Dellinger et al. (1993)
2,4,6-Tribromophenol	high-temperature flow reactor 300–800 °C	n.sp.	tri, tetra	1,3,6,8- and 1,3,7,9-TeBDD: 310 000 and 250 000 (500 °C)	Sidhu et al. (1995)

Table 10 (contd).

Flame retardant	Conditions of thermolysis[a]	PBDFs[b]	PBDDs[b]	Concentrations (mg/kg)[b,c]	Reference
Pentabromophenol	quartz tubes 700, 800, 900 °C	penta to hepta	penta to octa	∑ PBDFs: up to 7042 ∑ PBDDs: up to 7508	Thoma et al. (1986a)
Pentabromophenol	3 different types of ovens 600 °C	tri, tetra	tetra, hepta	TeBDFs: up to 3307 TeBDDs: up to 3567	Dumler et al. (1987); Hutzinger et al. (1989)
Pentabromophenol	pyrojector, absence of oxygen 700 °C	none	hepta, octa	small amounts	Thoma & Hutzinger (1987b, 1989)
Others					
TBBPA	quartz tubes 700, 800, 900 °C	mono to tetra	mono to tetra	∑ PBDDs/PBDFs: up to 1150/498	Thoma et al. (1986a)
TBBPA	BIS oven 240 °C	di	di	low levels	Thies et al. (1990)
Hexabromocyclo-dodecane	quartz tubes 700 °C	tri to hexa	tri, tetra	∑ PBDFs: 0.25 ∑ PBDDs: 0.05	Brenner (1993)
1,2-Bis(tribromo-phenoxy)ethane	high-temperature flow reactor in nitrogen 475 °C in air 450 °C	none none	none tetra	n.sp.	Striebich et al. (1990, 1991)

Table 10 (contd)

Tetrabromophthalic anhydride	quartz tubes 700, 800, 900 °C	none	none		Thoma et al. (1986a)
2,4,6-Tribromoaniline	sealed tubes 640 °C	tetra	tetra	n.sp.	Alsabbagh et al. (1992)
N-(tribromophenyl)-maleimide	sealed tubes 630 °C	tetra	tetra	n.sp.	Alsabbagh et al. (1992)

[a] For definitions and descriptions of apparatuses used for thermolysis experiments, see Merz et al. (1986) or Appendix II.
[b] n.a. = not analysed, n.sp. = not specified.
[c] Owing to the different conditions used, different studies should not be compared quantitatively.

Table 11. Survey on the generation of PBDFs and PBDDs during thermolysis of bromoorganic flame retardants in polymer matrices

Flame retardant	Polymer (additive)	Conditions of thermolysis[a]	Maximum yields (mg/kg) of PBDFs (sum of homologue groups detected)[b]	PBDDs present[b]		Reference
				Yes	No	
PBDEs						
PentaBDE (Bromkal 70-5 DE)	polystyrene	quartz tube 700–900 °C	420 000 (Br_1–Br_5)	–	–	Thoma et al. (1987a)
	polyethylene	quartz tube 700–900 °C	200 000 (Br_1–Br_5)	–	–	
PentaBDE	polyurethane	3 different ovens (VCI, BIS, DIN) 600–800 °C	~50 000[c,d]	x		Dumler et al. (1989b)
PentaBDE	polyurethane	DIN oven 300–800 °C	42 000 (Br_1–Br_6)[c]	x		Hutzinger (1990)
PentaBDE	polyurethane foam	ignition vessel 670–780 °C	n.sp.	x		Birla & Kamens (1994)
PentaBDE	laminate (SiO_2)	BIS oven 600 °C	2000 (Br_1–Br_5)	x		Lenoir et al. (1994)
	laminate (TiO_2)	BIS oven 600 °C	2.6 (Br_1–Br_3)		x	
OctaBDE	ABS (Sb_2O_3)	3 different ovens (VCI, BIS, DIN) 600–800 °C	>100 000[c,d]	x		Dumler et al. (1989b)
OctaBDE	ABS (Sb_2O_3)	DIN oven 300–800 °C	280 000 (Br_1–Br_7)[c]	x		Hutzinger (1990)

Table 11 (contd).

OctaBDE	ABS (Sb_2O_3)	DIN oven 600 °C	9000 (Br_3–Br_5)	x		Neupert et al. (1989b)
DecaBDE (FR 300 BA)	polystyrene	quartz tube 700–900 °C	7000 (Br_1–Br_7)	–	–	Thoma et al. (1987a)
	polyethylene	quartz tube 700–900 °C	170 000 (Br_1–Br_8)	–	–	
DecaBDE	polystyrene (Sb_2O_3)	3 different ovens (VCI, BIS, DIN) 600–800 °C	>100 000[c,d]	x		Dumler et al. (1989b)
DecaBDE	polystyrene (Sb_2O_3)	DIN oven 300–800 °C	228 000 (Br_1–Br_8)[c]		x	Hutzinger (1990)
DecaBDE	polypropylene	DIN oven 400–800 °C	255 000 (Br_1–Br_7)[c]		x	Dumler (1989)
DecaBDE	polypropylene	closed glass vials 600 °C	393 000 (Br_1–Br_5)[c]		x	Dumler (1989)
DecaBDE	polypropylene (Sb_2O_3)	3 different ovens (VCI, BIS, DIN) 600–800 °C	>100 000[c,d]	x		Dumler et al. (1989b)
DecaBDE	polypropylene (Sb_2O_3)	DIN oven 300–800 °C	290 000 (Br_1–Br_8)[c]	x		Hutzinger (1990)
DecaBDE	HIPS (Sb_2O_3)	mass burning rate apparatus (21% O_2) 500–800 °C	1600 (Br_1–Br_8)		x	Pinkerton et al. (1989)
DecaBDE	HIPS (Sb_2O_3)	quartz tube reactor in nitrogen 275–825 °C in air 500–700 °C	4300 (Br_2–Br_8)[e] 710 (Br_2–Br_6)[e]		x x	Luijk et al. (1990, 1991)
DecaBDE	PBT	quartz tube apparatus 400–800 °C	none		x	Lahaniatis et al. (1989)

Table 11 (contd)

Flame retardant	Polymer (additive)	Conditions of thermolysis[a]	Maximum yields (mg/kg) of PBDFs (sum of homologue groups detected)[b]	PBDDs present[b]		Reference
				Yes	No	
DecaBDE	PBT	VCI oven 400–600 °C	n.sp. (presence of Br_1–Br_6)		x	Sovocool et al. (1990)
DecaBDE	PBT (Sb_2O_3)	quartz tube apparatus 400–800 °C	13 100 (Br_1–Br_6)	–	–	Clausen et al. (1987); Lahaniatis et al. (1989)
DecaBDE	PBT (Sb_2O_3)	VCI oven 300–800 °C	160 000 (Br_1–Br_7)[c]	–	–	Dumler et al. (1989c)
	PBT (Sb_2O_3)	DIN oven 300–800 °C	228 000 (Br_1–Br_6)[c]		x	Hutzinger (1990)
	PBT (Sb_2O_3)	BIS oven 400–1000 °C	13 800 (Br_1–Br_5)[e]	x		Zier et al. (1990)
	PBT (Sb_2O_3)	BIS oven in nitrogen + H_2O 600 °C	29 500 (Br_1–Br_7)	x		Lenoir et al. (1994)
	epoxide resin (Sb_2O_3)	quartz tube apparatus 400–800 °C	none		x	Clausen et al. (1987)
	plastic sheets	quartz tube apparatus 600 °C	n.sp. (presence of various PBDFs)	x		Lahaniatis et al. (1989)

Table 11 (contd).

PBBs						
HexaBB (FM BP-6)	polystyrene	quartz tube 700–900 °C	8900 (Br_1–Br_4)	–	–	Thoma et al. (1987a)
	polyethylene	quartz tube 700–900 °C	43 000 (Br_1–Br_4)	–	–	
DecaBB	PBT	quartz tube reactor in nitrogen + 10% O_2 400–700 °C	100 (Br_3–Br_8)[e]	x		Luijk & Govers (1992)
Others						
TBBPA	PBT	3 different ovens (VCI, BIS, DIN) 600–800 °C	<100[d,e]	x		Dumler et al. (1989b)
	PBT (Sb_2O_3)	VCI oven 800 °C	41 (Br_1, Br_2)[e]	x		Hutzinger (1990)
	PBT (Sb_2O_3)	BIS oven 600 °C	0.11 (Br_1–Br_4)[e]	x		Thies et al. (1990)
	epoxide laminate	3 different ovens (VCI, BIS, DIN) 600–800 °C	<50[d,e]	x		Dumler et al. (1989b)
	epoxide laminate	DIN oven 600–800 °C	10 (Br_1–Br_3)[e]	x		Dumler (1989); Hutzinger (1990)
	epoxide laminate	VCI oven 800 °C	23 (Br_1–Br_3)[e]	x		Hutzinger (1990)

Table 11 (contd).

Flame retardant	Polymer (additive)	Conditions of thermolysis[a]	Maximum yields (mg/kg) of PBDFs (sum of homologue groups detected)[b]	PBDDs present[b] Yes	PBDDs present[b] No	Reference
	epoxide laminate (Cu)	3 different ovens (VCI, BIS, DIN) 600–800 °C	<100[d,e]	x		Dumler et al. (1989b)
	epoxide laminate	VCI oven 800 °C	40 (Br_1–Br_2)[e]	x		Hutzinger (1990)
	polycarbonate	3 different ovens (VCI, BIS, DIN) 600–800 °C	<10[d,e]	x		Dumler et al. (1989b)
	polycarbonate	DIN oven 600 °C	8.9 (Br_1–Br_3)[e]		x	Hutzinger (1990)
TBBPA	polycarbonate	BIS oven 600 °C	5.5 (Br_1–Br_6)[e]	x		Thies et al. (1990)
	ABS	BIS oven 600 °C	0.4 (Br_2–Br_4)[e]	x		Thies et al. (1990)
	ABS	quartz tube reactor in nitrogen + 10% O_2 400–700 °C	3.1 (Br_1–Br_5)[e]	x		Luijk & Govers (1992)
	ABS (Sb_2O_3)	VCI oven 800 °C	5.4 (Br_1, Br_2)[e]		x	Hutzinger (1990)
		BIS and DIN ovens 600 °C	2.1 (Br_1–Br_4)[e]	x		Thies et al. (1990)

Table 11 (contd).

1,2-Bis(tribromophenoxy) ethane	ABS (Sb_2O_3)	3 different ovens (VCI, BIS, DIN)				
		600–800 °C	<1000[c,d]	x		Dumler et al. (1989b)
	ABS (Sb_2O_3)	DIN oven				Hutzinger
		600 °C	500 (Br_1–Br_4)[c]	x		(1990)
Tetrabromophthalic anhydride	polyurethane	3 different ovens (VCI, BIS, DIN)				Dumler et al. (1989b)
		600–800 °C	<100[c,d]		x	
	polyurethane	VCI oven				Hutzinger
		800 °C	44 (Br_1)[c]		x	(1990)
		DIN oven				
		800 °C	0.4 (Br_4)[c]		x	
Hexabromocyclo dodecane	polystyrene	3 different ovens (VCI, BIS, DIN)				
		800 °C	10[c,d]		x	Dumler et al. (1989b)
		DIN oven				
		800 °C	5.5 (Br_2–Br_4)[c]		x	Hutzinger (1990)
Hexabromocyclo dodecane	polystyrene	quartz tube apparatus				Brenner
		700 °C	0.11 (Br_4–Br_6)	x		(1993)
	polystyrene insulation foam	quartz tube apparatus				Brenner
		700 °C	0.38 (Br_2–Br_8)	x		(1993)
Polybrominated polystyrene	polyester	3 different ovens (VCI, BIS, DIN)				Dumler et al. (1989b)
		600–800 °C	<100[d,e]	x		Hutzinger (1990)
		DIN oven				
		600 °C	36 (Br_1–Br_4)[e]		x	

Table 11 (contd).

Flame retardant	Polymer (additive)	Conditions of thermolysis[a]	Maximum yields (mg/kg) of PBDFs (sum of homologue groups detected)[b]	PBDDs present[b] Yes	PBDDs present[b] No	Reference
Polytribromostyrene	PBT (Sb_2O_3)	quartz tube 400–800 °C	none		x	Clausen et al. (1987)
Dibromopropyldian	polypropylene (Sb_2O_3)	3 different ovens (VCI, BIS, DIN) 600–800 °C	<100[c,d]		x	Dumler et al. (1989b)
	polypropylene (Sb_2O_3)	DIN oven 600 °C	28 (Br_2, Br_3)[c]		x	Hutzinger (1990)
1,2-Bis(tetrabromo phthalimido)ethane	ABS (Sb_2O_3)	3 different ovens (VCI, BIS, DIN) 600–800 °C	<500[c,d]	x		Dumler et al. (1989b)
	ABS (Sb_2O_3)	DIN oven 800 °C	118 (Br_1,Br_2)[c]	x		Hutzinger (1990)
	PBT (Sb_2O_3)	quartz tube 400–800 °C	none		x	Clausen et al. (1987)
Tetrabromobenz imidazolone	PBT	quartz tube 400–800 °C	none		x	Clausen et al. (1987)
Polypentabromo benzylacrylate	PBT	quartz tube 400–800 °C	none		x	Clausen et al. (1987)

[a] For definitions and descriptions of apparatuses used for thermolysis experiments, see Merz et al. (1986) or Appendix II.
[b] – = no information; n.sp. = not specified.
[c] Related to flame retardant.
[d] Figures only.
[e] Related to blend.

Dumler et al., 1990b). This effect was not observed with pentaBDE (Thoma et al., 1987a; Dumler, 1989; Hutzinger, 1990). Plastic/flame retardant mixtures showed maxima of PBDF formation at 600 °C (decaBDE/PBT: Dumler et al., 1990b; decaBB/PBT: Luijk & Govers, 1992), 700 °C (pentaBDE/polyurethane: Dumler, 1989; Hutzinger, 1990; TBBPA/ABS: Luijk & Govers, 1992), or 800 °C (TBBPA/epoxide laminate: Dumler, 1989; Hutzinger, 1990). The presence of Sb_2O_3 in the polymer matrices resulted in a further decrease of the optimum formation temperature (down to 400 °C) of PBDFs from octa- or decaBDE (Clausen et al., 1987; Bieniek et al., 1989; Dumler, 1989; Dumler et al., 1989a,c, 1990a,b; Hutzinger, 1990; Zier et al., 1990; Luijk et al., 1991). An example referring to decaBDE is given in Table 12.

The polymer matrix and the synergistic action of Sb_2O_3 influenced both the optimum temperature range and the yield of PBDFs (e.g. with hexaBB and decaBDE). Additionally, thermolysis in a polymer matrix changed the ratio of PBDF congeners to lower brominated compounds (e.g. Thoma et al., 1987a; Dumler et al., 1990b; see also Table 12). Frequently, tetraBDFs were the most abundant homologue group in these ternary mixtures (e.g. Clausen et al., 1987; Bieniek et al., 1989; Dumler, 1989; Hutzinger, 1990; Luijk et al., 1991).

Formation of PBDDs from decaBDE, octaBDE, decaBB, and TBBPA showed a similar dependence on temperature and/or matrix as seen with PBDFs, but the yields were much lower (Dumler et al., 1990b; Hutzinger, 1990; Luijk & Govers, 1992). Thermolysis of pentaBDEs, bromophenols, 1,2-bis(tribromophenoxy)ethane, and some samples of TBBPA resulted in an increase in the relative proportion of PBDDs (e.g. Dumler, 1989; Hutzinger, 1990; see also Table 10). Some smaller differences in the temperature profiles between PBDDs and PBDFs can also occur (e.g. thermolysis of TBBPA/laminate: formation maximum of PBDFs at 800 °C, of PBDDs at 400 °C and 700 °C; Dumler, 1989).

The influence of various metals (tin, iron, zinc, copper) and metal oxides (oxides of zinc, copper, iron, and antimony; silica, SiO_2, and titanium dioxide, TiO_2) on yield and pattern of PBDFs/PBDDs was

Table 12. Yields of PBDFs from combustion of DBDE, alone and in a polypropylene matrix[a,b]

PBDFs	PBDF yield (mg/kg) from combustion of DBDE alone			PBDF yield (mg/kg) from combustion of DBDE in a polypropylene matrix[c]		
	400 °C	600 °C	800 °C	400 °C	600 °C	800 °C
MonoBDFs	–	–	–	14 432	10 676	4192
DiBDFs	–	–	–	26 462	14 845	4850
TriBDFs	–	–	11	39 997	24 036	8354
TetraBDFs	–	–	28	107 517	49 677	29 147
PentaBDFs	–	–	35	37 419	18 458	6867
HexaBDFs	96	447	81	24 432	5465	948
HeptaBDFs	204	1449	959	4762	1033	353
OctaBDF	107	860	–	–	–	–
Total	407	2756	1114	255 021	124 190	54 711

[a] Adapted from Dumler et al. (1990a,b).
[b] Combustion of commercial DBDE samples (pure or mixture of polypropylene/12.5% DBDE/7.5% Sb_2O_3) in DIN oven.
[c] Yield related to DBDE.

studied in thermolysis of decaBDE or pentaBDE in polymer matrices (Lenoir et al., 1994). During thermolysis of decaBDE in PBT (500 °C, BIS oven), all metals and the oxides of copper (I), iron (III), and antimony (III) caused an increase in PBDD concentrations. PBDF concentrations were enhanced by oxides of iron and antimony. Oxides of zinc and copper (II) strongly reduced yields of PBDDs/PBDFs. During thermolysis of pentaBDE in laminate (600 °C, BIS oven), SiO_2 was found to be nearly inert, whereas addition of TiO_2 (3%) led to a significant reduction in PBDD/PBDF levels.

Water also plays an important role, as seen in thermolysis experiments with decaBDE in PBT/Sb_2O_3 performed in a nitrogen atmosphere with and without water at 600 °C (Zier et al., 1990; Lenoir et al., 1994). Water increased the PBDF, PBDD, and 2,3,7,8-TeBDF concentrations by a factor of 7.5, 2.8, and 10, respectively.

Other factors influencing yields and pattern of PBDDs/PBDFs formed in thermolysis of flame retardants are oxygen (O'Keefe, 1978; Thoma & Hutzinger, 1987a,b; Bieniek et al., 1989; Luijk & Govers, 1992), air flow rates (Klusmeier et al., 1988), types of combustion apparatuses (types frequently used are described in Appendix II), and residence time. Thermolysis of a flame-retarded resin (glass fibre reinforced PBT/decaBDE/Sb_2O_3) under simulated municipal waste incineration conditions confirmed the formation of PBDDs/PBDFs (Riggs et al., 1990).

2,3,7,8-TeBDF was identified after pyrolysis of a technical-grade flame retardant (alone) consisting primarily of 2,2',4,4',5,5'-hexaBB (Buser et al., 1978; O'Keefe, 1978). Thies et al. (1990) found 2,3,7,8-substituted congeners from experiments where TBBPA (alone) was pyrolysed. 2,3,7,8-Substituted congeners were also found after pyrolysis of plastics containing decaBB (Luijk & Govers, 1992), PBDEs (Dumler, 1989; Lahaniatis et al., 1989, 1991; Dumler et al., 1990c; Hutzinger, 1990; Zier et al., 1990), and TBBPA (Thies et al., 1990; Lorenz & Bahadir, 1993). Maximum concentrations of 2,3,7,8-TeBDF of up to 2000 mg/kg of flame retardant were found for pyrolysed polymers containing octaBDE (Dumler, 1989). The highest values of 1,2,3,7,8-PeBDF measured following thermolysis of plastics containing decaBDE, decaBB, or TBBPA were 63 mg/kg of plastic blend

(Zier et al., 1990), 1.2 mg/kg (Luijk & Govers, 1992), and 50 µg/kg (Thies et al., 1990), respectively. Maximum levels of 2,3,7,8-substituted tetra- and pentaBDDs were below 1 mg/kg of blend. For example, 2,3,7,8-TeBDD was formed during thermolysis of a decaBDE/epoxide resin at 0.8 mg/kg (Lahaniatis et al., 1991). Because of a lack of reference substances, the higher brominated PBDDs/PBDFs with the 2,3,7,8-substitution pattern were not quantified. (However, a tentative value of 21 µg/kg for 1,2,3,4,6,7,8-HpBDF was found by Brenner [1993] after thermolysis of an insulating board consisting of polystyrene foam blended with hexabromocyclododecane.)

Pyrolysis of PBBs (hexaBB) and PBDEs (PeBDE) at 800 °C in the presence of polyvinyl chloride (PVC) did not result in the formation of PBDFs/PBDDs but led to mixed brominated/chlorinated biphenyls or diphenyl ethers and to fully chlorinated compounds. Apparently, the substitution reactions were favoured over ring-closing reactions (Thoma et al., 1987b). Moreover, such bromine–chlorine exchange reactions were also observed during pyrolysis (at 800 °C and 900 °C) of PBDDs/PBDFs, leading to PXDDs/PXDFs and PCDDs/PCDFs. The chlorine source could be either organic or inorganic (Thoma et al., 1987b,c,d, 1989) (see also section 3.9.1).

Possible pathways for the thermolytic formation of PBDDs/PBDFs from flame retardants have been discussed by several authors (Buser, 1986a; Bieniek et al., 1989; Dumler, 1989; Hutzinger, 1990; Lahaniatis et al., 1991; Luijk et al., 1991; Luijk & Govers, 1992).

Other related polycyclic compounds identified after thermolysis of brominated flame retardants included bromomethyldibenzofurans (Sovocool et al., 1990; Lenoir et al., 1994), brominated benzo[b]-naphtho[2,3-d]furans (Lenoir, 1994), brominated phenazines (Alsabbagh et al., 1992), and hexabromonaphthalene (O'Keefe, 1978).

Detailed descriptions of thermolysis experiments involving PBDEs and TBBPA are given in the Environmental Health Criteria monographs for these compounds (WHO, 1994b, 1995).

3.5 Formation during production of plastic materials and presence in consumer products containing flame retardants

Brominated flame retardants are or were routinely added, at levels up to 20%, to a number of commercial products, such as plastics, textiles, carpets, and other materials (Buser, 1986a; WHO, 1994a,b, 1995). They also have applications in a variety of industries, such as the electronic, electrical engineering, building, and transport industries (BMU, 1989; Troitzsch, 1990).

PBDDs/PBDFs, as contaminants of certain flame retardants (section 3.2), would be transferred to the flame-retarded products and, under thermolytic stress (section 3.4), could additionally be formed from these chemicals during manufacturing processes.

3.5.1 *Formation during production processes*

PBDD/PBDF levels were monitored during typical processes (extrusion, injection moulding, etc.) used in the production and processing of flame-retarded polymers (Donnelly et al., 1989a; Bonilla et al., 1990; Brenner & Knies, 1990, 1992, 1993a,b, 1994; Thies et al., 1990). Extruder temperatures of 150–300 °C were noted (Donnelly et al., 1989a; Brenner & Knies, 1990). The polymers used were ABS and PBT, and the flame retardants examined included OBDE, DBDE, TBBPA-carbonate oligomer (BC 52), TBPI, brominated styrene, and 1,2-bis(tribromophenoxy)ethane. Results from different studies (monitoring exhaust streams) are compiled in Table 13. It can be seen that OBDE and DBDE produced the highest amounts of PBDDs/PBDFs, the major portion consisting of PBDFs. PBDF concentrations of about 73 000 ng/m^3 air or 7.7 ng/g extruded resin were measured during extrusion of resins containing DBDE, and 1850 ng/g extruded resin in the case of OBDE. The levels observed with TBBPA and TBPI were lower by several orders of magnitude. No PBDDs/PBDFs were formed from brominated styrene or brominated phenoxyethane. Homologue groups present included monoBDFs through octaBDF (Table 13). 2,3,7,8-Substituted congeners were not determined with DBDE (Brenner & Knies, 1990) and were not detected with TBBPA-carbonate oligomer and TBPI, at detection limits ranging from 1 to

Table 13. Formation of PBDDs/PBDFs during manufacturing processes (production of flame-retarded polymers in the chemical industry)

Process	Sample	Concentrations (ng/m³)[a]			Reference
		PBDDs	PBDFs	PBDF homologue groups	
Manufacture of PBT/glass fibre resin blended with DBDE/Sb_2O_3	Exhaust stream from extruder head	1	72 904	DiBDFs: 322; TrBDFs: 705; TeBDFs: 980; PeBDFs: 3910; HxBDFs: 22 162; HpBDFs: 39 550; OcBDF: 5275	Brenner & Knies (1990)
Manufacture of PBT/glass fibre resin blended with TBBPA (BC 52)/Sb_2O_3	Exhaust stream from extruder head	1.1	1.3 (0.87)[b]	DiBDFs: 0.42 (0.16); TrBDFs: 0.48 (0.31); TeBDFs: 0.24 (0.06); PeBDFs: 0.04 (0.33); HxBDFs: 0.15 (0.01)[b]	Brenner & Knies (1993a,b)
	Exhaust stream from granulator	0.2	0.7 (1.01)[c]	DiBDFs: 0.23 (0.2); TrBDFs: 0.29 (0.49); TeBDFs: 0.17 (0.08); PeBDFs: 0.02 (0.07); HxBDFs: 0 (0.24)[b]	
	Exhaust stream from injection moulding machine	n.d. (0.001–0.2)	0.08	DiBDFs: 0.004; TrBDFs: 0.012; TeBDFs: 0.014; PeBDFs: 0.013; HxBDFs: 0.039	
	Exhaust stream from storage hood	n.d. (0.001–0.2)	0.006	DiBDFs: 0.004; TeBDFs: 0.002	
Manufacture of PBT/glass fibre resin blended with TBPI	Exhaust stream from extruder head	0.05	0.78	DiBDFs: 0.18; TrBDFs: 0.34; TeBDFs: 0.18; PeBDFs: 0.03; HxBDFs: 0.06	Brenner & Knies (1994)
Processing of ABS/TBBPA material	Off-gas near a compounding machine	6	213	MoBDFs: 13; DiBDFs: 200	Thies et al. (1990)

Table 13 (contd).

Process	Sample	Concentrations (ng/g of extruded resin)			Reference
		PBDDs	PBDFs	PBDF homologue groups	
Processing of PBT/DBDE + Sb_2O_3 resin	Fumes generated during extrusion	n.a.	7.69	TeBDFs: 1.35; PeBDFs: 2.47; HxBDFs: 1.67; HpBDFs: 2.20	Donnelly et al. (1989a)
Processing of	Fumes generated during extrusion				Bonilla et al. (1990)
• ABS/brominated styrene terpolymer resin		n.d.	n.d.	–	
• ABS/1,2-bis(tribromo-phenoxy)ethane resin		n.d.	n.d.	–	
• ABS/TBBPA resin		0.006	0.020	n.sp.	
• ABS/OBDE resin		0.54	1850	n.sp.	

[a] n.a. = not analysed; n.d. = not detected (detection limits in parentheses, if specified); n.sp. = not specified.
[b] Values in parentheses were obtained when PBT was blended with BC 52 (TBBPA)-PBT-batch (~50% BC 52) instead of BC 52 powder.

58 pg/m³ (Brenner & Knies, 1992, 1993a,b, 1994). However, PBDFs with 2,3,7,8-substitution (0.012 ng/g extruded resin; but co-elution was possible) were found for OBDE just above the detection limit (Bonilla et al., 1990).

The actual worker exposure to PBDDs/PBDFs depends on the ventilation and exhaust conditions around the machines. In some studies, parallel measurements of the workplace atmosphere were undertaken, results of which are discussed in section 5.3 (Brenner & Knies, 1990, 1993a,b; Thies et al., 1990).

3.5.2 Presence in resins and polymer products

PBDDs/PBDFs were determined in various plastic materials at several processing stages (BMU, 1989; Donnelly et al., 1989a; Bonilla et al., 1990; Brenner & Knies, 1990, 1992, 1993a,b, 1994; Hutzinger, 1990; McAllister et al., 1990; Thies et al., 1990; Luijk et al., 1992c; UBA, 1992; Lorenz & Bahadir, 1993; Meyer et al., 1993; Kieper, 1996). These examinations included granulated resins, moulded parts whose flame retardant additives were known, and parts of commercial electrical appliances (television sets, printers, computers) whose flame retardants were unknown (see also Tables 14 and 15). Polymers used were ABS, HIPS, polystyrene, polyamide, PBT, polypropylene, or polyurethane in combination with about 5–20% PBDEs, TBBPA, bromopolystyrenes, TBPI, or other compounds as flame retardants (see Table 14). The highest levels of PBDDs/PBDFs were found in materials flame-retarded with PBDEs and were in the range of several thousand µg/kg, thus exceeding the levels in any other systems by orders of magnitude. The range of PBDF concentrations found is given in Table 14. PBDFs were, with few exceptions, the predominant components. The contamination of electrical appliances having an unknown flame retardant equipment is shown in Table 15.

Generally, the concentrations of PBDDs/PBDFs were higher in the typical resin/PBDE products than in the flame retardants alone (see also section 3.2). For comparisons, it should be noted that the concentrations shown in Table 14 were calculated on the basis of the weight of the resins. They would be higher if calculated on the basis of the weight of flame retardant in the resins. Values obtained using the latter

Table 14. Concentrations of PBDFs in several flame-retarded plastic materials

Resin/flame retardant	Concentrations (µg/kg)[a]						Reference
	Sum (homologue groups)	TetraBDFs	PentaBDFs	HexaBDFs	HeptaBDFs	OctaBDF	
ABS/OBDE							
• Granulate	–	2100	24 000	50 000	3900	1700	BMU (1989)
• Normal extrusion[b] (n = 17–22)	–	2.8–3.6	870–1800	2100–2380	500–780	26–64	Donnelly et al. (1989a)
• Abusive extrusion[b] (n = 1–2)	–	150–170	29 000–34 000	8200–10 000	500–920	19	Donnelly et al. (1989a)
• Pre-extrusion resin	38 300 (Br_4–Br_7)	–	–	–	–	–	Bonilla et al. (1990)
• Post-extrusion resin	84 500 (Br_4–Br_7)	–	–	–	–	–	Bonilla et al. (1990)
• Normal moulding[c]	–	3	1100	<135 000	–	–	McAllister et al. (1990)
• Abusive moulding[c]	–	170	<14 000	<118 000	–	–	McAllister et al. (1990)
ABS/TBBPA							
• Pre-extrusion resin	1090 (Br_4–Br_7)	–	–	–	–	–	Bonilla et al. (1990)
• Post-extrusion resin	n.d.	–	–	–	–	–	Bonilla et al. (1990)
• Commercial polymer	–	n.d. (<2)	n.d. (<3)	n.d. (<20)	–	–	Thies et al. (1990)

Table 14 (contd).

Resin/flame retardant	Concentrations (μg/kg)[a]						Reference
	Sum (homologue groups)	TetraBDFs	PentaBDFs	HexaBDFs	HeptaBDFs	OctaBDF	
ABS/brominated styrene terpolymer							
• Pre-extrusion resin	37.5 (Br_4–Br_7)	–	–	–	–	–	Bonilla et al. (1990)
• Post-extrusion resin	84.0 (Br_4–Br_7)	–	–	–	–	–	Bonilla et al. (1990)
ABS/1,2-bis(tribromophenoxy)ethane							
• Pre-extrusion resin	44.5 (Br_4–Br_7)	–	–	–	–	–	Bonilla et al. (1990)
• Post-extrusion resin	16.2 (Br_4–Br_7)	–	–	–	–	–	Bonilla et al. (1990)
HIPS/DBDE							
• Normal extrusion[f]	–	–	4.5	950	720	150	Donnelly et al. (1989a)
• Abusive extrusion[g]	–	2.3	22.6	107	78	0.5	Donnelly et al. (1989a)
• Extreme extrusion[g]	–	0.01	8.6	200	2100	3200	Donnelly et al. (1989a)
• Base resin	–	10	40	<5300	–	–	McAllister et al. (1990)

Table 14 (contd).

• Normal moulding[c]	–	10	50	<14 300	–	–	McAllister et al. (1990)
• Abusive moulding[c]	–	10	60	<5500	–	–	McAllister et al. (1990)
• Extreme moulding[c]	–	20	200	<34 100	–	–	McAllister et al. (1990)
• Pre-extrusion resin	–	–	–	–	~1500	~4500	Luijk et al. (1992c)
• Post-extrusion resin (4 cycles at 275 °C)	–	–	–	–	~9000	~45 000	Luijk et al. (1992c)
Polystyrene/DBDE							
• Compound	194	2.7	14.6	174	–	–	UBA (1992)
• 2 casing parts manufactured from the above compound	640; 1313 (Br_7–Br_8)	54; 39	147; 106	1092; 409	–	–	UBA (1992)
Polystyrenebutadiene/ DBDE							
• Compound	n.d.	n.d. (0.2)	n.d. (1.0)	n.d. (50)	–	–	Kieper (1996)
Polystyrene 1,2-bis-(tribromophenoxy)-ethane							
• Compound	n.d.	n.d. (0.7)	n.d. (2.5)	n.d. (9.7)	–	–	Kieper (1996)

Table 14 (contd).

Resin/flame retardant	Concentrations (μg/kg)[a]						Reference
	Sum (homologue groups)	TetraBDFs	PentaBDFs	HexaBDFs	HeptaBDFs	OctaBDF	
Polyamide/ polytribromostyrene							
• Compound	15.3 (Br_1–Br_8)	n.d. (0.2)	0.28	1.81	3.43	6.21	Kieper (1996)
Polyamide/ polydibromostyrene							
• Compound	4.18 (Br_1–Br_8)	0.64	0.38	0.37	0.46	2.15	Kieper (1996)
PBT/DBDE							
• Granulate (n = 7)	–	6–501	20–920	59–65 000	66–136 000	n.d.–2600	BMU (1989)
• Normal extrusion[f] (n = 17–22)	–	1–26	18–130	71–1600	180–3800	410–4100	Donnelly et al. (1989a)
• Abusive extrusion[f] (n = 5)	–	76–240	13 000–43 000	69 000–180 000	48 000–94 000	1200–11 000	Donnelly et al. (1989a)
• Extreme extrusion[f] (n = 3)	–	1020–2590	68 200–82 800	272 000–708 000	72 500–108 000	–	Donnelly et al. (1989a)
• Blend	–	6.2	27	151	~560	~280	Brenner & Knies (1990)
• Base resin	–	3	20	110	–	–	McAllister et al. (1990)
• Normal moulding[g]	–	3	2	13	–	–	McAllister et al. (1990)

Table 14 (contd).

• Abusive moulding[g]	–	30	>7800	>16 100	–	–	McAllister et al. (1990)
• Extreme moulding[g]	–	1000	>54 000	>7000	–	–	McAllister et al. (1990)
PBT/TBBPA							
• Commercial polymer	–	n.d. (<0.2)	n.d. (<0.1)	n.d. (<1)	–	–	Thies et al. (1990)
• Extruder granulate (n = 3)	–	n.d.	n.d.	0.4–0.8	0.6–3.5	–	Brenner & Knies (1990)
• Moulded test articles (n = 2)	–	0.17–0.2	n.d.–0.06	1.5–2.2	1.9–3.8	–	Brenner & Knies (1993a,b)
• Compound	8.41 (Br_1–Br_6)	0.14	2.13	6.14	–	–	Kieper (1996)
PBT/bromopolystyrene							
• Granulate (n = 2)	–	n.d.–5	2–10	34–130	11–460	–	BMU (1989)
PBT/bis-tetrabromophthalimide							
• Granulate	–	–	5	35	31	–	BMU (1989)
PBT/TBPI							
• Polymer	–	0.57	0.07	0.02	3.4	–	Brenner & Knies (1994)
• Granulate (n = 2)	–	up to 0.8	0	0	0	–	Brenner & Knies (1994)
• Moulded test article	–	0	0	0	0	–	Brenner & Knies (1994)

Table 14 (contd).

Resin/flame retardant	Concentrations (µg/kg)[a]						Reference
	Sum (homologue groups)	TetraBDFs	PentaBDFs	HexaBDFs	HeptaBDFs	OctaBDF	
Polypropylene/DBDE							
• Granulate	–	53	191	10 000	1370	2600	BMU (1989)
Polyurethane/PeBDE							
• Granulate	–	18 000	57 000	44 000	–	–	BMU (1989)

[a] – = not mentioned; n.d. = not detected (detection limit in parentheses, if specified); n.sp. = not specified.
[b] Normal/abusive extrusion conditions: 227 °C/246 °C; 1 min/10 min cycle.
[c] Normal/abusive moulding conditions: 225 °C/245 °C; 1 min/10 min cycle.
[d] Normal/abusive/extreme extrusion conditions: 216–218 °C/238–243 °C/266–271 °C; 30 second/5 min/7 min cycle.
[e] Normal/abusive/extreme moulding conditions: 215–220 °C/235–245 °C/265–270 °C; 30 second/5 min/7 min cycle.
[f] Normal/abusive/extreme extrusion conditions: 250–254 °C/254 °C/254 °C; 23 second/5 min/10 min cycle.
[g] Normal/abusive/extreme moulding conditions: 255 °C/255 °C/255 °C; 23 second/5 min/10 min cycle.

Table 15. PBDF/PBDD concentrations found in plastics from commercial electrical appliances with unknown polymer/flame retardant system[a]

PBDFs/PBDDs	Concentrations[b] (µg/kg) in plastics for	
	casings (n = 8)	printed circuit boards (n = 6)
MonoBDFs	n.d.–0.5	n.d.–19.8
DiBDFs	n.d.–3.1	n.d.–149
TriBDFs	n.d.–13.1	0.2–441
TetraBDFs	n.d.–48.9	0.6–1264
2,3,7,8-TeBDF[c]	<0.1–1.2	<0.1–11.1
PentaBDFs	n.d.–1126	n.d.–1372
1,2,3,7,8-PeBDF[c]	<0.1–16.4	<0.1–24
2,3,4,7,8-PeBDF[c]	<9.1–31.5	<0.1–6.5
HexaBDFs	n.d.–2952	n.d.–185
1,2,3,4,7,8-HxBDF[c]	<0.7–203	<1.5–9.9
Total PBDFs	n.d.–4125	3.6–3430
Total PBDDs	n.d.–113.6[d]	1.9–1974[e]

[a] Adapted from UBA (1992).
[b] n.d. = not detected. Detection limits: <0.1 µg/kg for mono- to triBDFs/BDDs; <0.1–<0.3 µg/kg for tetra- and pentaBDFs/BDDs; <0.7–<2.1 µg/kg for hexaBDFs; and <1.0–<15.5 µg/kg for hexaBDDs.
[c] Maximum values, because co-elution could not be excluded.
[d] Seven of eight samples = n.d.
[e] Five of six samples = 1.9–13.9 µg/kg.

calculation were presented by Hutzinger (1990), who found PBDF concentrations ranging from 112 to 1888 mg/kg in polymers (polypropylene, PBT, ABS, or polyurethane) containing PBDEs as flame retardants.

Frequently, additional processing resulted in a further increase in total PBDFs (see Table 14). For example, the sum of mono- to hexaBDF levels measured in casing parts was higher by a factor of about 5 compared with that found in the corresponding polymer–flame retardant blend (polystyrene/DBDE) actually used in manufacturing (Table 14; UBA, 1992). Factors influencing the extent of formation of PBDFs are temperature and the duration of such processes as blending, extrusion, and moulding (Table 14; Donnelly et al., 1989a; McAllister et al., 1990).

Within the PBDF homologue groups, the highly brominated (>tetra) derivatives were generally prevalent. Frequently, peak concentrations were seen with penta- and hexaBDFs (see Tables 14 and 15). In casing parts, hexaBDFs reached levels as high as 2950 µg/kg; in printed circuit boards, maximum concentrations (>1000 µg/kg) were seen with tetra- and pentaBDFs (Table 15; UBA, 1992). Concentrations of mono-, di-, and triBDFs were in the low µg/kg range, <30 µg/kg for polymers and casings (BMU, 1989; Brenner & Knies, 1990; Thies et al., 1990; UBA, 1992) and <450 µg/kg for printed circuit boards (UBA, 1992).

2,3,7,8-Substituted PBDFs were not analysed (Brenner & Knies, 1990), were not detectable (Thies et al., 1990; Brenner & Knies, 1993a,b, 1994; Kieper, 1996), or were detected in some formulations at low concentrations (Donnelly et al., 1989a; Bonilla et al., 1990; McAllister et al., 1990; UBA, 1992; Lorenz & Bahadir, 1993; Meyer et al., 1993; Kieper, 1996). A moulded part consisting of ABS/PBDE contained 2,3,7,8-TeBDF at 2 µg/kg (Meyer et al., 1993). Plastics for casings and printed circuit boards of electrical appliances (n = 14) were found to be contaminated with tetra-, penta-, and hexaBDFs having the 2,3,7,8-substitution pattern (Table 15; UBA, 1992). The maximum concentrations of 2,3,7,8-TeBDF were <4% of the total tetraBDFs in the case of casings and <17% in the case of printed circuit boards. Higher percentages were found for the two pentaBDFs and for 1,2,3,4,7,8-HxBDF, usually <7–22%, but exceptionally reaching as high as 75% of the respective homologue groups (UBA, 1992). As co-elution cannot be excluded, all concentrations of 2,3,7,8-substituted congeners may be overestimated (UBA, 1992; Meyer et al., 1993; Kieper, 1996).

PBDDs were not routinely detected in the samples examined (Donnelly et al., 1989a; Brenner & Knies, 1990, 1992, 1993a,b; Hutzinger, 1990; Thies et al., 1990; Kieper, 1996). If present, their maximum concentrations in several thermoplastic resins ranged from 0.006 to 4500 µg/kg (Bonilla et al., 1990; McAllister et al., 1990; Lorenz & Bahadir, 1993; Meyer et al., 1993; Kieper, 1996) and from 1.9 to 1974 µg/kg in plastics taken from electrical appliances (UBA, 1992). Whereas some samples contained PBDDs, mainly tetraBDDs, only trace amounts of PBDFs were present in these samples (Lorenz

& Bahadir, 1993; Kieper, 1996). With plastics from electrical appliances, one out of eight casing parts and all six printed circuit boards examined gave positive results for PBDDs (UBA, 1992). The PBDD concentration measured in the one positive casing part was 114 µg/kg (consisting only of hexaBDDs). In five out of seven positive samples, the percentage of PBDDs was low, amounting to a maximum of 2.7% of the total PBDD/PBDF content. In the remaining two samples (showing concentrations of about 3 and 2000 µg/kg, the latter being mainly tetraBDDs), the percentages of PBDDs were 46.3 and 99.7% of the total PBDD/PBDF levels (UBA, 1992; see also Table 15). Altogether, the PBDD homologue distribution pattern was somewhat irregular and included mono- to hexaBDDs (UBA, 1992).

Although 2,3,7,8-substituted penta- and hexaBDDs were present (up to 25 µg/kg) in a few samples (McAllister et al., 1990; UBA, 1992; Meyer et al., 1993), 2,3,7,8-TeBDD could not be detected at detection limits mostly below 0.3 µg/kg (Bonilla et al., 1990; McAllister et al., 1990; UBA, 1992; Lorenz & Bahadir, 1993; Meyer et al., 1993; Kieper, 1996).

3.6 Emissions from flame-retarded consumer products

Some experiments were performed to clarify whether or not PBDFs were released from television sets, computers, or similar appliances (Bruckmann et al., 1990; Ranken et al., 1990; Thies et al., 1990; UBA, 1992). Positive (Bruckmann et al., 1990; Thies et al., 1990; UBA, 1992) and negative (Ranken et al., 1990; UBA, 1992) results were obtained.

None of three (Ranken et al., 1990) and three of four (UBA, 1992) units (television sets, computer monitors) tested in experimental chambers were found to release PBDFs during 3 × 8 h of operation (Ranken et al., 1990) or continuously for 72 h (UBA, 1992).

In the experiments conducted by UBA (1992), maximum total emissions of PBDFs (tetra through hexa) were 1800 pg/unit tested, with tetraBDFs being most prevalent. 2,3,7,8-TeBDF was detected in one appliance (see Table 16). Concentrations of PBDEs concomitantly measured ranged between <1.4 and 890 ng/unit. No PBDFs (detection

Table 16. Emissions of PBDFs from television sets and computer monitors as measured in a test chamber[a,b]

PBDFs	Emissions (pg/appliance)[c]			
	Television sets (n = 2)		Computer monitors (n = 2)	
			Colour	Monochrome
TetraBDFs[d]	320	n.d.	1045	605
2,3,7,8-TeBDF[e]	n.d. (3)	n.d. (5)	15	n.d. (3)
PentaBDFs	n.d.	n.d.	330	165
1,2,3,7,8-PeBDF[e]	n.d. (7)	n.d. (8)	<5	n.d. (5)
2,3,4,7,8-PeBDF[e]	n.d. (7)	n.d. (8)	<5	n.d. (5)
HexaBDFs	n.d. (10)	n.d. (10)	424	80
HeptaBDFs	n.d. (10)	n.d. (10)	n.d. (15)	n.d. (10)
OctaBDF	n.a.	n.a.	n.a.	n.a.
Sum tetra- to hexaBDFs[f]	320	n.d.	1799	850

[a] Adapted from UBA (1992).
[b] Volume of test chamber: 1.17 m^3; sampling time: continuously, 72 *h*; sucking speed: 1.5 m^3/h.
[c] n.d. = not detected (detection limits in parentheses, where specified); n.a. = not analysed.
[d] Emissions correspond to about 3, 10, and 6 pg/m^3, respectively.
[e] Maximum value, as co-elution with internal standard cannot be excluded.
[f] Emissions less than the detection/quantification limit are set as 50% of the limit when calculating the sum of homologues.

limit: 3–10 pg/printer) and only low levels of PBDEs were observed with the three printers tested.

In another study (Bruckmann et al., 1990), air samples were collected in a closed room (27 m^3) at different distances from a new television set operating for 3 days for approximately 17 h/day. Total concentrations of PBDFs (tri through penta) were found to be 155 pg/m^3 air at a distance of 0.15 m above the television set and 28 pg/m^3 air in the centre of the room (2.2 m distant from the television set). According to a calculation of UBA (1992), the concentrations of tetraBDFs (11 pg/m^3), pentaBDFs (0.5 pg/m^3), and hexaBDFs (<0.1 pg/m^3) measured above the television set correspond to 732, 33, and <7 pg emitted per television set, respectively. For comparison, ambient air concentrations were 0.16 pg/m^3 (tetraBDFs) and <0.05 pg/m^3

(penta- plus hexaBDFs). Bruckmann et al. (1990) did not analyse for any 2,3,7,8-substituted congeners.

Thies et al. (1990) detected tetraBDFs and pentaBDFs at concentrations of 3 and 8 pg/m^3, respectively, in an air sample (sampling volume about 500 m^3) taken 15 cm above a television cabinet being installed and kept at 46 °C in an office room (45 m^3). No PBDFs could be detected in an air sample collected at a distance of 2.2 m from the television set.

The presence of PBDFs in "non-experimental" rooms equipped with monitors and other appliances is discussed in sections 5.1.1 and 5.3.1.

3.7 Presence in fire residues, smoke condensates, and gases after fires

3.7.1 Experimental fires

Experimental fire tests simulating real fire conditions were performed with electrical appliances such as television sets, printers, computer terminals, and their casings (Fabarius et al., 1990; Hamm & Theisen, 1992; UBA, 1992). High PBDF concentrations can be produced under these conditions (see Table 17). The total PBDD/PBDF concentrations in combustion residues reached values of between 1 and 1930 mg/kg (up to 0.2%) for electrical appliances and between 8000 and 9000 mg/kg (almost 1%) for the casing parts (see Table 17). PBDD/PBDF levels determined in some appliances before burning were in the range n.d.–4.2 mg/kg (Hamm & Theisen, 1992; UBA, 1992). Analysis of smoke condensate from the fire test room gave area contaminant concentrations ranging from 6 to 1610 μg/m^2 (Table 17), most values being in the range of 100–400 μg/m^2 (Hamm & Theisen, 1992; UBA, 1992). The smoke samples collected during the fire tests contained 0.8–1700 μg PBDDs/PBDFs/m^3 (Table 17).

With some exceptions — where penta- or decaBDE was specified — the flame retardants being included in the test materials were unknown (Fabarius et al., 1990; Hamm & Theisen, 1992; UBA, 1992).

Table 17. PBDF/PBDD concentrations in samples from fire tests with electrical appliances

Fire object (*n*)	Fire site	Homologue groups analysed/abundant	Total PBDF/PBDD concentrations[a,b]			Reference
			Combustion residue (mg/kg)	Smoke condensate (µg/m²)	Smoke/gas (µg/m³)	
Television set (2)	barrel	Br_3–Br_6/Br_3, Br_4	79.27 (2.0)	n.a.	0.825 (0.02)	Fabarius et al. (1990)
Casing parts of electrical appliances (2)	room	Br_1–Br_6/Br_4–Br_5	7750–8700 (n.d.)	177–1610 (n.d.)	n.a.	Hamm & Theisen (1992); UBA (1992)
Electrical appliances (6)	room	Br_1–Br_6/Br_4–Br_6	1–1930 (n.d. –0.7)	6–323 (n.d.–1.2)	11–1700 (n.d.–3.9)	Hamm & Theisen (1992); UBA (1992)

[a] PBDD concentration in parentheses.
[b] n.a. = not analysed, n.d. = not detected.

As seen in Table 17, the total PBDD concentrations were low in combustion residues (n.d.–2.0 mg/kg), smoke condensate (n.d.–1.2 μg/m²), and smoke (n.d.–3.9 μg/m³), with a maximum of 3% of the corresponding total PBDD/PBDF concentrations (Fabarius et al., 1990; Hamm & Theisen, 1992; UBA, 1992).

In one study (Fabarius et al., 1990), the most abundant homologue groups were the tri- and tetraBDDs/BDFs. Other studies (Hamm & Theisen, 1992; UBA, 1992) showed a homologue distribution pattern dominated by tetra- through hexaBDDs/BDFs.

The proportion of 2,3,7,8-TeBDF was mostly under 3% of the total of tetraBDFs. 2,3,7,8-Substituted congeners of penta- and hexaBDFs yielded between 1% and a maximum of 16% of the corresponding totals. For example, concentrations measured in fire residues ($n = 8$) ranged from 0.005 to 18 mg/kg for 2,3,7,8-TeBDF, from 0.005 to 116 mg/kg for the two 2,3,7,8-substituted PeBDFs, and from <0.014 to 567 mg/kg for 1,2,3,4,7,8-HxBDF (UBA, 1992). PBDDs with 2,3,7,8-substitution were not detected; however, their detection limits were relatively high (0.001–1.9 mg/kg in combustion residues) (UBA, 1992).

During preliminary experiments intended to improve sampling techniques, test vehicles (one car, one subway wagon) were burnt in a traffic tunnel (Wichmann et al., 1993). Noticeable amounts of PBDFs (Br_1–Br_6) and PCDDs (Cl_4–Cl_8) were released at both fires. The measurements performed gave values in the high ng/m³ range (graphics given only). Fire residue samples from different materials (e.g. paint, floor coverings, cables, etc.) taken from the burnt-out vehicles (one car: $n = 4$, one subway wagon: $n = 5$) showed concentrations of up to 4000 ng/kg for mono- to triBDFs and up to 250 ng/kg for tetra- to octaBDFs. PBDD concentrations were lower, having peak concentrations of 120 ng/kg for mono- to triBDDs and of <22 ng/kg for tetra- to octaBDDs (Zelinski et al., 1994).

(It is very difficult to extrapolate values from gases, smoke, or flue gases [ng/m³] to areal contamination [ng/m²]. The indoor and outdoor dispersion vary greatly and are quite difficult to predict.

Furthermore, human exposure and resulting body burdens cannot be predicted.)

3.7.2 Accidental fires

Analyses of fire residues, smoke condensates, gases, and firemen's trousers confirmed the expected release of PBDDs/PBDFs during real fire accidents (Buser, 1986b; Bruckmann et al., 1990; Fabarius et al., 1990; Hamm & Theisen, 1992; Neupert & Pump, 1992; UBA, 1992; Zelinski et al., 1993, 1994; Schacht et al., 1995; see also Table 18). With some exceptions (bowling hall, stockhouse, computer room), all fire cases examined occurred in private residences with television sets being involved. Concentrations measured were mostly below the values found in the model experiments described above, but the qualitative compositions of the samples were similar (see section 3.7.1).

PBDFs dominated clearly over PBDDs. In contrast to PBDFs, which were found in almost all samples, PBDDs were not regularly detected; if present, their concentrations were low. For example, PBDDs could be identified in three of nine television fire incidents. In these three incidents, only four of nine samples contained PBDDs, the maximum concentrations being less than 1.5% of the sum of PBDF/PBDD (mono through hexa) levels (UBA, 1992). Likewise, the sum of PBDD concentrations in two soot samples collected after a fire in a computer room were in the range of 0.1–30 µg/kg (Schacht et al., 1995).

As seen in Table 18, PBDFs covered a wide range of concentrations. The PBDF levels of combustion residues were mainly in the µg/kg range, but two maximum values of 107 mg/kg (Br_1–Br_6; UBA, 1992) and 17 mg/kg (Br_1–Br_8; Zelinski et al., 1993) were also observed (see Table 18). The PBDF (Br_1–Br_6) area contaminant concentrations (caused by smoke condensates) in close vicinity to the fire site ranged between 0.1 and 13.1 µg/m^2 in most cases (UBA, 1992). In the adjoining areas, the levels measured were usually lower by a factor of 2–34. Interestingly, there were no systematic correlations between PBDF concentrations found in combustion residues (fire site) and those found in wipe samples (area contamination). It is

Table 18. Release of PBDFs during accidental fires in private residences or other buildings

PBDFs	Building[a] (*n*)	Major fire object[b] (*n*)	PBDF concentrations[b]				Reference
			Fire residues (µg/kg)	Smoke condensate[c] (ng/m²)		Gases (ng/m³)	
				A	B		
MonoBDFs	house (3)	TV set (1)	n.d.–0.1	8.1–16.9	n.d.–2.2	n.a.	UBA (1992)
	house (1)	TV set (1)	17	n.d.	n.d.	n.a.	UBA (1992)
	flat (5)	TV set (1)	n.d.	n.d.–9.3	n.d.–14.7	n.a.	UBA (1992)
DiBDFs	residence (4–5)	n.sp.	0.1–2.3	n.a.	n.a.	<0.1	Fabarius et al. (1990)
	house (3)	TV set (1)	n.d.–2.7	43.7–146	n.d.–36	n.a.	UBA (1992)
	house (1)	TV set (1)	595	2	15	n.a.	UBA (1992)
	flat (5)	TV set (1)	n.d.–0.7	n.d.–75.8	n.d.–94	n.a.	UBA (1992)
TriBDFs	residence (4–5)	n.sp.	0.06–7.31	n.a.	n.a.	<0.1–2.0	Fabarius et al. (1990)
	house (3)	TV set (1)	0.2–17.9	122–1131	24–110	n.a.	UBA (1992)
	house (1)	TV set (1)	3491	272	85	n.a.	UBA (1992)
	flat (5)	TV set (1)	n.d.–3.4	n.d.–756	n.d.–587	n.a.	UBA (1992)
TetraBDFs	stock house (1)	n.sp.	n.sp.	4.0; 32	8.9; 123	n.a.	Bruckmann et al. (1990)
	residence (4–5)	n.sp.	0.12–8.6	n.a.	n.a.	<0.1–1.5	Fabarius et al. (1990)
	house (3)	TV set (1)	0.4–93.2	532–4396	64–216	n.a.	UBA (1992)

Table 18 (contd).

PBDFs	Building[a] (*n*)	Major fire object[b] (*n*)	PBDF concentrations[b]				Reference
			Fire residues (µg/kg)	Smoke condensate[c] (ng/m²)		Gases (ng/m³)	
				A	B		
	house (1)	TV set (1)	16 063	2392	225	n.a.	UBA (1992)
	flat (5)	TV set (1)	n.d.–6.6	n.d.–2432	n.d.–1505	n.a.	UBA (1992)
	computer room (1)	equipment (2 soot samples)	13.6–2700	n.a.	n.a.	n.a.	Schacht et al. (1995)
PentaBDFs	stock house (1)	n.sp.	n.sp.	0.8; 4.4	9.2; 78	n.a.	Bruckmann et al. (1990)
	residence (4–5)	n.sp.	<0.05–5.7	n.a.	n.a.	<0.01	Fabarius et al. (1990)
	house (3)	TV set (1)	n.d.–64	1597–3919	142–232	n.a.	UBA (1992)
	house (1)	TV set (1)	64 932	4263	401	n.a.	UBA (1992)
	flat (5)	TV set (1)	0.1–6.5	n.d.–3671	n.d.–1670	n.a.	UBA (1992)
	computer room (1)	equipment (2 soot samples)	15–2100	n.a.	n.a.	n.a.	Schacht et al. (1995)
HexaBDFs	stock house (1)	n.sp.	n.sp.	4.5; 27	15; 52	n.a.	Bruckmann et al. (1990)
	residence (4–5)	n.sp.	<0.05–2.34	n.a.	n.a.	<0.1	Fabarius et al. (1990)
	house (3)	TV set (1)	n.d.–102	2210–4840	69–981	n.a.	UBA (1992)
	house (1)	TV set (1)	21 740	544	121	n.a.	UBA (1992)
	flat (5)	TV set (1)	n.d.–7.7	n.d.–3574	n.d.–1504	n.a.	UBA (1992)

Table 18 (contd).

	computer room (1)	equipment (2 soot samples)	0.8–711	n.a.	n.a.	n.a.	Schacht et al. (1995)
HeptaBDFs	stock house (1)	n.sp.	n.sp.	0.25; 1.2	7; 13	n.a.	Bruckmann et al. (1990)
	residence (4–5)	n.sp.	<0.05–0.87	n.a.	n.a.	<0.1	Fabarius et al. (1990)
	computer room (1)	equipment (2 soot samples)	n.d.–6.2	n.a.	n.a.	n.a.	Schacht et al. (1995)
OctaBDF	stock house (1)	n.sp.	n.sp.	1.1; 5.6	0.5; 8.5	n.a.	Bruckmann et al. (1990)
	residence (4–5)	n.sp.	<0.05	n.a.	n.a.	<0.1	Fabarius et al. (1990)
Sum MoBDFs–TrBDFs	flat (1)	TV set (1)	1894	n.a.	n.a.	n.a.	Zelinski et al. (1993)
		other things (11)	1.2–203	n.a.	n.a.	n.a.	
	flat (1)	window frame, etc. (4)	2.3–67.8	n.a.	n.a.	n.a.	Zelinski et al. (1993)
	flat (1)	wallpaper above TV, etc. (5)	n.d.–20.7	n.a.	n.a.	n.a.	Zelinski et al. (1993)
Sum MoBDFs–HxBDFs	house/flat (8)	TV set (1)	0.5–235	n.d.–13 054	n.d.–5374	n.a.	UBA (1992)
	house (1)	TV set (1)	106 838	7473	847	n.a.	UBA (1992)
Sum DiBDFs–OcBDF	residence (4–5)	n.sp.	0.3–27.3	n.a.	n.a.	<0.1–3.5	Fabarius et al. (1990)
Sum TeBDFs–HpBDFs	computer room (1)	equipment (2 soot samples)	29–5600	n.a.	n.a.	n.a.	Schacht et al. (1995)

Table 18 (contd).

PBDFs	Building[a] (*n*)	Major fire object[b] (*n*)	PBDF concentrations[b]				Reference
			Fire residues (µg/kg)	Smoke condensate[c] (ng/m^2)		Gases (ng/m^3)	
				A	B		
Sum TeBDFs–OcBDF	stock house (1)	n.sp. (6)	n.d.–1.4	n.sp.	n.sp.	n.a.	Bruckmann et al. (1990)
	flat (1)	TV set (1)	14 910	n.a.	n.a.	n.a.	Zelinski et al. (1993)
		other things (11)	0.4–63.6	n.a.	n.a.	n.a.	
	flat (1)	window frame, etc. (4)	1.0–14.4	n.a.	n.a.	n.a.	Zelinski et al. (1993)
	flat (1)	wallpaper above TV, etc. (5)	2.7–169	n.a.	n.a.	n.a.	Zelinski et al. (1993)

[a] Sampling time after fire: 0–7 days (if specified).
[b] n.a. = not analysed; n.d. = not detectable; n.sp. = not specified.
[c] Samples taken close to fire site (A) or distant from fire site (B).

thus difficult to predict the area contamination from residue analysis (Hamm & Theisen, 1992; UBA, 1992). Zelinski et al. (1993) found that not only the distance from the source but also the surface characteristics of the objects influence the PBDF content of the samples. The maximum gaseous emissions of PBDFs (Br_2–Br_8) measured in other cases amounted to 3.5 ng/m^3 (Fabarius et al., 1990). One study provided tentative information on contamination of a fireman's trousers after fighting a fire. PBDFs (Br_2–Br_8) were found at a concentration of 2.01 µg/kg (Fabarius et al., 1990).

The distribution pattern of the PBDF homologue groups was, in most cases, dominated by tetra- through hexabrominated homologues (see Table 18). On the other hand, peak concentrations were found for the di- through tetraBDFs (Zelinski et al., 1993). The following order resulted from ranking the dibenzofuran concentrations detected in fire residues from a computer room: TeBDFs > PeBDFs > HxBDFs > HpBDFs (Schacht et al., 1995).

The proportion of 2,3,7,8-substituted isomers was low in the samples examined (Fabarius et al., 1990; UBA, 1992; Zelinski et al., 1993). Single maximum proportions of 3, 10, or 18% of the corresponding totals of tetra-, penta-, or hexaBDFs, respectively, were reported from fire accidents with television sets (UBA, 1992; see also Table 19). Neupert & Pump (1992) reported on the occurrence of 2,3,7,8-substituted tetra- and pentaBDFs in residue samples collected after a fire in the store of a plastics production plant. Maximum concentrations found were 0.4, 1, and 2 µg/kg for 2,3,7,8-TeBDF, 1,2,3,7,8-PeBDF, and 2,3,4,7,8-PeBDF, respectively. 2,3,7,8-TeBDD, which usually has detection limits of 0.1 µg/kg (residue samples) or 0.8 ng/m^2 (wipe samples), was identified at 0.3 µg/kg in one residue sample of a television set (UBA, 1992). In four out of five samples from the warehouse fire, 2,3,7,8-TeBDD was detected at concentrations of 0.3–0.5 µg/kg (Neupert & Pump, 1992). Soot samples ($n = 2$) collected after a fire in a computer room (Schacht et al., 1995) contained 2,3,7,8-TeBDD (n.d./0.6 µg/kg), 1,2,3,7,8-PeBDD (n.d./<0.04 µg/kg), 2,3,7,8-TeBDF (0.03/48.3 µg/kg), 1,2,3,7,8-PeBDF (0.2/15 µg/kg), and 2,3,4,7,8-PeBDF (0.4/14 µg/kg).

Table 19. 2,3,7,8-Substituted PBDFs found in residue and wipe samples after accidental fires of television sets in private residences[a]

Congeners	Residence[b] (*n*)	PBDF concentrations[c]		
		Fire residues (µg/kg)	Smoke condensate[d,e] (ng/m²)	
			A	B
2,3,7,8-TeBDF	house (3)	<0.1–1.0	17–40.9	1.8–6.8
	house (1)	264	n.r.	1.8
	flat (5)	<0.1–0.1	<0.2–22	<0.1–12.2
1,2,3,7,8-PeBDF	house (3)	<0.1–3.1	22.5–143	<2.5–6.1
	house (1)	1863	n.r.	<1.3
	flat (5)	<0.1–0.1	<0.2–80.7	<0.3–38.2
2,3,4,7,8-PeBDF	house (3)	<0.1–1.8	17.7–87.7	<0.4–<2.9
	house (1)	848	n.r.	<1.3
	flat (5)	<0.1–<0.3	<0.4–31.5	<0.3–15.9
1,2,3,4,7,8-HxBDF	house (3)	<0.3–17.7	107–851	<6–<17.5
	house (1)	1932	n.r.	<8.5
	flat (5)	<0.5–<8.0	<8.5–173	<4.5–69.5

[a] Adapted from UBA (1992).
[b] Sampling time after fire: 0.5–4 days.
[c] Maximum values, as co-elution cannot be excluded.
[d] Samples taken close to fire site (A) or distant from fire site (B).
[e] n.r. = not recorded.

Analyses of a wipe sample close to the fire site (television) showed the presence of PXDFs (tri through octa; up to three bromines per molecule). The total concentration was 3240 ng/m², or about half the amount found for PBDFs in this sample (UBA, 1992). Small quantities of PXDFs were also detected in a soot sample from a fire of a bowling hall in which plastic, wood, and other materials burnt (Buser, 1986b). Significant concentrations of PXDFs (about 30–200 µg/kg) occurred in fire residues from a department store (Wilken & Schanne, 1994).

PCDDs/PCDFs were concomitantly present in many samples, but mostly at lower total concentrations than PBDDs/PBDFs (Bruckmann et al., 1990; Fabarius et al., 1990; UBA, 1992; Wichmann et al., 1992a,b; Wilken & Schanne, 1994; Schacht et al., 1995). Exceptions

were gas samples and firemen's trousers containing a higher proportion of PCDDs/PCDFs (Fabarius et al., 1990). The estimated PBDD/PBDF concentrations in the soot sample from a fire in a bowling hall were also lower than those of PCDDs/PCDFs (Buser, 1986b). Other contaminants additionally identified in fire residues were polycyclic aromatic hydrocarbons (PAHs) (Hamm & Theisen, 1992), chlorinated and brominated benzenes, bromophenols, polychlorinated biphenyls (PCBs), PBBs, PBDEs, and TBBPA (Buser, 1986b; Fabarius et al., 1990; Zelinski et al., 1993).

3.8 Formation from incineration of fuels

PXDDs (Br_1Cl_5DDs) and hexachlorodibenzo-*p*-dioxins (hexaCDDs) were detected in ash from a wood-fired boiler (nature of "wood" not specified) at concentrations of 55 µg/kg and 418 µg/kg, respectively (Harless et al., 1989).

No data were available on incineration of coal, peat, or fuel oil in power plants.

3.9 Formation during waste disposal and treatment

3.9.1 *Incineration*

The formation of PCDDs/PCDFs in fly ash of waste incinerators was recognized in the 1970s (Olie et al., 1977); the additional presence of PXDDs/PXDFs was first reported in 1986 (Schäfer & Ballschmiter, 1986). Since then, a number of studies (Nakano et al., 1987; Öberg et al., 1987; Schwind et al., 1988, 1989; Sovocool et al., 1988, 1989; Harless et al., 1989; Hosseinpour et al., 1989; Huang et al., 1991, 1992a,b; Tong et al., 1991; Funcke & Hemminghaus, 1993; Hartenstein, 1993; Chatkittikunwong & Creaser, 1994c; Takasuga et al., 1994) have documented the presence of PBDDs/PBDFs and/or PXDDs/PXDFs in fly ash and/or flue gas of municipal, clinical, or hazardous waste incinerators (see also Table 20).

The amount of PHDDs/PHDFs formed critically depends on the combustion conditions and on the extent to which the combustion can be controlled. In the past, it was customary in many industrial

Table 20. Detection of PBDDs/PBDFs and PXDDs/PXDFs in fly ash or flue gas from waste incinerators

Waste (country)[a]	Sample (*n*)	Homologue groups detected[a]				Concentrations	Reference
		PBDDs	PBDFs	PXDDs	PXDFs		
Municipal (Germany)	fly ash (1)	n.a.	TeBDFs	$Br_1Cl_{3-7}DDs$ $Br_2Cl_{2-6}DDs$	$Br_1Cl_{3-7}DFs$ $Br_2Cl_{2-3}DFs$ Br_3Cl_1DFs	TeBDFs: 16 ng/kg $\sum$ PXDDs: 5535 ng/kg $\sum$ PXDFs: 3157 ng/kg	Schwind et al. (1988, 1989); Hosseinpour et al. (1989)
Municipal (USA)	fly ash (1)	n.a.	n.a.	$Br_1Cl_{3-7}DDs$	$Br_1Cl_{3-7}DFs$	$\sum$ PXDDs: 108 µg/kg[b] $\sum$ PXDFs: 9.8 µg/kg[b]	Sovocool et al. (1988, 1989)
Municipal (USA)	fly ash (1)	n.sp.	n.sp.	Br_1Cl_5DDs	n.sp.	Br_1Cl_5DDs: 31 µg/kg	Harless et al. (1989)
Municipal (USA)	fly ash (1)	n.a.	n.a.	$Br_1Cl_{3-7}DDs$	$Br_1Cl_{3-7}DFs$	$\sum$ PXDDs: 56 µg/kg $\sum$ PXDFs: 47 µg/kg	Tong et al. (1991)
Municipal (n.sp.)	fly ash (3)	n.a.	n.a.	$Br_1Cl_{3-7}DDs$	n.a.	$\sum$ PXDDs: 0.5–163 µg/kg	Huang et al. (1992a)
Municipal (USA)	fly ash (3)	n.a.	n.a.	$Br_2Cl_{2-6}DDs$	$Br_2Cl_{2-6}DFs$	$\sum$ PXDDs: 772–2602 ng/kg $\sum$ PXDFs: 334–1513 ng/kg	Huang et al. (1992b)
Municipal (Japan)	fly ash (1)	n.a.	n.a.	Br_2Cl_2DDs	n.d.	Br_2Cl_2DDs: 0.4 ng/kg	Huang et al. (1992b)
Municipal (Canada)	fly ash (1)	n.a.	n.a.	Br_2Cl_2DDs	$Br_2Cl_{2-6}DFs$	$\sum$ PXDDs: 1704 ng/kg $\sum$ PXDFs: 1335 ng/kg	Huang et al. (1992b)
Municipal (United Kingdom)	fly ash (3)	Di-, TrBDDs	Mo-, DiBDFs	$Br_1Cl_{1,4,5}DDs$ $Br_2Cl_{1-2}DDs$	$Br_1Cl_{1,3,4}DFs$ $Br_2Cl_{1-3}DFs$	$\sum$ PBDDs: 145–436 ng/kg $\sum$ PBDFs: 12–325 ng/kg $\sum$ PXDDs: 406–1005 ng/kg $\sum$ PXDFs: 1347–2922 ng/kg	Chatkittikunwong & Creaser (1994c)

Table 20 (contd).

Clinical (United Kingdom)	fly ash (1)	n.d. (Di, TrBDDs)	MoBDFs	$Br_1Cl_{1,4,5}DDs$ $Br_2Cl_{1-2}DDs$	$Br_1Cl_{1,3}DFs$ Br_2Cl_1DFs	MoBDFs: 77 ng/kg $\sum$ PXDDs: 705 ng/kg $\sum$ PXDFs: 427 ng/kg	Chatkittikunwong & Creaser (1994c)
Hazardous (Sweden)	flue gas (6)	n.d. (TeBDDs)	n.a.	Br_1Cl_3DDs	Br_1Cl_3DFs	Br_1Cl_3DDs: n.d.–1.3 ng/m^3 Br_1Cl_3DFs: n.d.–4.5 ng/m^3	Öberg et al. (1987)
Hazardous (Germany)	flue gas (2)	n.sp.	n.sp.	n.a.	n.a.	$\sum$ PBDDs: 0.76–0.82 ng/m^3 $\sum$ PBDFs: 0.76–0.82 ng/m^3	Hartenstein (1993)

[a] n.a. = not analysed; n.d. = not detected; n.sp. = not specified.
[b] Estimated concentration.

countries — and may still be usual in many developing countries — to burn the waste in landfills by open fires, with incomplete combustion forming toxic by-products. Such conditions may be especially favourable for the formation of PHDDs/PHDFs. When the problem was recognized, the technology of waste incineration was greatly improved, and it has at present reached a high degree of sophistication. An important reduction can be achieved by optimizing the burning conditions, by increasing the temperature, residence time, and turbulence. Energy can be recovered by boilers and can be converted to heat or electricity. A quick quench to temperatures below 250 °C has been found to minimize the formation of PCDDs/PCDFs.

The flue gas emissions can be controlled by scrubbing the gases with dry, semi-dry, or wet technologies. Addition of lime and charcoal before filtering in a baghouse has been successful. Further reduction can be achieved by the use of catalysts, which destroy the remaining PCDDs/PCDFs. This indicates that control of environmental hazards may be achieved by appropriate but, of course, more expensive measures. The improved technology should not exclude initiatives for waste minimization and the development of new technologies for recycling of plastics and wastes.

Several possibilities for the origin of PHDDs/PHDFs exist. Some PHDDs/PHDFs may be introduced in trace amounts by the feedstock and may resist combustion. Far larger amounts can be produced in the incinerator itself, by formation from precursors at high temperatures in the flame (see section 3.4 and, for example, Sidhu et al., 1995) or by *de novo* synthesis at low temperatures in the post-combustion zone of the incinerator through gas–solid interactions on fly ash. The latter hypothesis has been proved by several studies (Stieglitz et al., 1989; Stieglitz & Vogg, 1990; Heinbuch & Stieglitz, 1992, 1993; Luijk et al., 1992b, 1994). The formation of PXDDs/PXDFs is explained by the extensive bromine–chlorine exchange reactions observed under several test conditions (Thoma et al., 1987b,c, 1989; Zier et al., 1991; Luijk et al., 1992a, 1994). It is assumed that because of the large quantities of chlorine donors in waste, these reactions ultimately result in the formation of completely chlorinated compounds. On the other hand, irreversible bromination of PCDDs/PCDFs may occur as the fly

ash moves from hotter to cooler regions of the incinerator (Sovocool et al., 1989; Huang et al., 1992b).

There are some reports on the consequences of an increase in bromine input during test operations in incinerators. In a large-scale experiment at the municipal waste incinerator at Bielefeld-Herford (Germany), material containing 4.8% pentaBDE was added to the normal fuel (Lahl et al., 1991). The fly ash from the electrostatic precipitator was analysed for PCDDs/PCDFs and PXDDs/PXDFs as well as for inorganic bromine. The bromine content in the samples ranged from 0.37 to 0.59 mg Br^-/kg. Of the mixed PXDD/PXDF congeners, only monobromopolychlorinated PXDDs/PXDFs (Br_1Cl_xDDs/Br_1Cl_xDFs) could be detected in the five fly ash samples. The concentrations of PXDDs/PXDFs (Br_1Cl_2 to Br_1Cl_7) ranged from 1.547 to 10.163 µg/kg. Interestingly, the concentrations of the purely chlorinated compounds (Cl_4–Cl_8DDs/Cl_4–Cl_8DFs) were much higher (up to 406.17 µg/kg) than normally detected (50–150 µg/kg). Although the highest concentrations of all three parameters analysed (Br^-, PXDDs/PXDFs, and PCDDs/PCDFs) were found in the same sample, a quantitative relationship could not be established. Whereas concentrations of PCDDs were higher than concentrations of PCDFs in all samples, concentrations of PXDFs were higher than concentrations of PXDDs (Lahl et al., 1991). An increase in concentrations of monobromopolychlorinated PXDDs/PXDFs in the crude gas was also found after addition of tetrabromomethane to the furnace of a municipal waste incinerator (Wilken et al., 1990). Wanke et al. (1996) studied the influence of additional bromine input (up to sixfold) into municipal solid waste incinerators at a pilot plant (nominal throughput: 200 kg/h) at combustion temperatures of 850 or 950 °C. Extruded polystyrene foams and rigid polyurethane foams (2–4% Br by weight) were introduced as bromine source. In the raw gases of the polyurethane foam combustion, no increase in PCDDs/PCDFs was detected when compared with "normal" fuel. The concentrations of PBDDs/PBDFs were very low in all experiments. Both test series, the experiments with extruded polystyrene and rigid polyurethane foams, showed elevated levels of PXDDs/PXDFs. As was reported from the Bielefeld-Herford incinerator, of the PXDDs/PXDFs, those congeners containing only one bromine were the most abundant; dibrominated species could hardly be detected. More than two bromine atoms could

not be identified in any sample. Similarly, the PXDDs/PXDFs contributed only 20–30% of the total sum of all halogenated (PCDDs/PCDFs + PXDDs/PXDFs) dibenzo-*p*-dioxins and dibenzofurans. This finding was not found earlier by Hartenstein (1993), who reported much higher concentrations of PBDDs/PBDFs than of PCDDs/PCDFs in flue gas samples of a hazardous waste incinerator. The results of Wanke et al. (1996) showed that, at least for the PXDFs, there is a correlation between the content of bromide in the fly ash (up to 5% by weight) and the concentrations of PXDFs. Above 5% Br^-, no further increase in concentrations of PXDFs could be determined; in other words, saturation was obtained. For the PBDDs, such a correlation could not be established, as the concentrations of the PBDDs were too low and the standard deviation too large.

Once the PHDDs/PHDFs have been formed, they partition between stack gas (gas phase) and fly ash (solid phase) (Schramm et al., 1990).

The quantities of PBDDs/PBDFs and PXDDs/PXDFs measured in fly ash of incinerators were in the range of ng/kg to µg/kg (see Table 20). The few flue gas measurements gave values in the low ng/m^3 range (see Table 20).

In most cases, the concentrations of dibenzo-*p*-dioxins exceeded those of dibenzofurans (see Table 20). PXDDs/PXDFs were more abundant than PBDDs/PBDFs (Chatkittikunwong & Creaser, 1994c; see also Table 20) but were present in fly ash or chimney residues at lower levels than their fully chlorinated counterparts, reaching 1–20% of the levels of PCDDs/PCDFs (Schäfer & Ballschmiter, 1986; Sovocool et al., 1988, 1989; Harless et al., 1989; Tong et al., 1991; Chatkittikunwong & Creaser, 1994c). For example, the total PCDD/PCDF levels ranged from 37 to 62 µg/kg, and the total PBDD/PBDF plus PXDD/PXDF levels amounted to 1.2–3.5 µg/kg in fly ash from municipal (n = 3) and clinical (n = 1) incinerators (Chatkittikunwong & Creaser, 1994c). However, flue gas of a hazardous waste incinerator contained more PBDDs/PBDFs than PCDDs/PCDFs (1.6 versus 0.05 ng/m^3) (Hartenstein, 1993). No information was provided on the ratio of chlorinated to brominated compounds in the feedstock.

The main homologues of PXDDs/PXDFs that were detected consisted of mono- and dibromopolychlorinated PXDDs/PXDFs (see Table 20). Generally, the highest levels were found for Br_1Cl_4, Br_1Cl_5, or Br_1Cl_6 congeners (Schwind et al., 1988, 1989; Sovocool et al., 1988, 1989; Hosseinpour et al., 1989; Tong et al., 1991; Chatkittikunwong & Creaser, 1994c).

The isomer distribution patterns of PXDDs/PXDFs were similar to those found for PCDDs/PCDFs and similar among different samples, indicating common mechanisms of formation, regardless of the incinerator conditions and nature of the feedstock (Harless et al., 1989; Huang et al., 1992b; Chatkittikunwong & Creaser, 1994c).

Of the 2,3,7,8-substituted congeners, 2,3-Br_2-7,8-Cl_2DD was found in several fly ash samples at concentrations ranging from 4 to 12 ng/kg (maximum values because the degree of co-elution was not known) (Huang et al., 1992b). In another test series, 2,3,7,8-TeBDD and 2,3,7,8-TeBDF could not be detected (detection limits: 16 and 8 ng/kg, respectively) (Chatkittikunwong & Creaser, 1994c).

There are several methods to minimize the emissions of dibenzo-*p*-dioxins and dibenzofurans from incinerators. They are mostly described for PCDDs/PCDFs, but they may also be valid for PBDDs/PBDFs and PXDDs/PXDFs (Boyd & Mortland, 1985; Hagenmaier et al., 1987a,b; Vogg et al., 1987; Vogg, 1989; Wania & Lenoir, 1990; Acharya et al., 1991; Spahl et al., 1993; Gullett et al., 1994; Schreiber, 1994; van de Plassche et al., 1994; Vehlow, 1995).

Flue gas monitored after flue gas cleaning in the stack of a Swedish municipal solid waste incinerator operating at a high combustion efficiency did not contain certain PBDDs/PBDFs and PXDDs/PXDFs (Öberg & Bergström, 1990). The detection limits for the compounds examined were 0.4 ng/m^3 (tetraBDFs/BDDs, pentaBDDs) and 0.03 ng/m^3 (Br_1Cl_3DDs/DFs, Br_2Cl_2DDs).

3.9.2 Disposal

Disposal sites (dumps, landfills) are expected to be an important source of brominated dibenzo-*p*-dioxins/dibenzofurans (Sovocool et

al., 1988; Donnelly et al., 1990; Öberg & Bergström, 1990) because they receive plastic waste, municipal incinerator fly ash, automotive fluff (ground-up waste residue from junked cars, which remains after the bulk metals have been reclaimed), etc. and can also be subject to occasional fires.

A detailed investigation of waste samples from three German disposal sites confirmed the occurrence of PBDDs/PBDFs and PXDDs/PXDFs along with PCDDs/PCDFs (Dawidowsky, 1993). The sum of the concentrations of PBDDs/PBDFs and PXDDs/PXDFs was in the range of several hundred to thousands of ng/kg dry weight (see Table 21). The concentration of dibenzo-*p*-dioxins was low (PBDDs/PXDDs: 6–580 ng/kg) in relation to the concentration of dibenzofurans (PBDFs/PXDFs: 217–4229 ng/kg). PBDFs may be prevalent over PBDDs (maximum concentrations >3000 ng/kg versus >300 ng/kg) because they originate from PBDEs, which were found at high concentrations in the same samples (sum of mono- to decaBDEs: 4400–17 500 ng/kg dry weight, $n = 6$).

The homologue profile was dominated by lower halogenated derivatives (up to Br_4/X_4; Table 21). This pattern contrasted to that of PCDDs/PCDFs, which had peak concentrations of higher chlorinated homologues (Dawidowsky, 1993). These waste samples showed high total concentrations of PCDDs/PCDFs, with PCDDs being the most abundant components. PCDD values ranged from 3000 to 9000 ng/kg ($n = 5$) or to nearly 30 000 ng/kg ($n = 6$); PCDFs reached levels of 2000–5600 ng/kg ($n = 6$) (Dawidowsky, 1993).

Another study determined PBDDs/PBDFs (and PCDDs/PCDFs) in several waste samples of an analytical "dioxin" laboratory (Ritterbusch et al., 1994b). The sum of PBDD/PBDF concentrations in waste oil samples ($n = 3$) from the GC/MS system ranged from <150 to 14 000 ng/kg (sum of PCDDs/PCDFs: <200–13 600 ng/kg). Other laboratory waste samples ($n = 4$) also contained PBDDs/PBDFs (mono to hexa), with a peak concentration of 15 500 ng/kg for hexaBDFs.

Table 21. Occurrence of PBDDs/PBDFs and PXDDs/PXDFs in waste samples from three disposal sites[a]

Compounds	Concentrations (ng/kg dry weight)[b]					
	Disposal site A		Disposal site B		Disposal site C	
	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6
Dibenzo-*p*-dioxins						
PBDDs	9 (Br_1, Br_3)	24 (Br_1, Br_3)	155 (Br_2)	313 (Br_1, Br_2)	2 (Br_1, Br_3)	1 (Br_3)
PXDDs	43 (X_2, X_4)	33 (X_2, X_4)	29 (X_3, X_4)	267 (X_3–X_4)	5 (X_4)	5 (X_4)
Total	52	57	184	580	7	6
Dibenzofurans						
PBDFs	1672 (Br_1–Br_4)	2675 (Br_1–Br_3)	386 (Br_1–Br_4)	3262 (Br_1–Br_4)	170 (Br_1–Br_3)	285 (Br_1–Br_4)
PXDFs	1261 (X_2–X_4)	1554 (X_2–X_4)	682 (X_2–X_4)	539 (X_2–X_4)	47 (X_2–X_4)	9 (X_4)
Total	2933	4229	1068	3801	217	294

[a] Adapted from Dawidowsky (1993).
[b] Homologue groups quantified are given in parentheses.

3.9.3 Recycling

3.9.3.1 Plastics

Granulated parts of office machine casings, which were reprocessed once or several times, were analysed for the eight 2,3,7,8-substituted PBDDs/PBDFs of the 1994 German Dioxin Directive (Meyer et al., 1993). The polymer was ABS flame-retarded with either PBDE or TBBPA, as well as mixed electronic waste with unknown flame retardants. Materials flame-retarded with PBDE yielded the highest concentrations of these eight PBDDs/PBDFs (16–65 µg/kg), depending on processing. Mixed electronic waste was contaminated by lower concentrations (4.9–26 µg/kg). Concentrations of the PBDDs/PBDFs in TBBPA flame-retarded material were lower still (n.d.–2.4 µg/kg).

Recycling of printed circuits containing TBBPA flame retardant can lead to the formation of PBDDs/PBDFs (Lorenz & Bahadir, 1993). Whereas untreated basic recovered material contained total concentrations of PBDDs/PBDFs (mono to octa) of 0.22 µg/kg, shredded material was contaminated with PBDDs/PBDFs at levels ranging from 0.03 µg/kg (metal fraction) to 1.13 µg/kg (mixed and plastic fraction) following treatments with hammer mill, impact grinder, separation, and granulation processes. The main components were monoBDFs (0.05–0.32 µg/kg), diBDFs (0.23 µg/kg), and tetraBDDs (0.03–0.73 µg/kg). 2,3,7,8-TeBDD/TeBDF were not detected (detection limits: 0.01–0.05 µg/kg). The contamination was probably due to thermal decomposition of TBBPA.

Scrap of electronic devices (printed circuit boards with electronic components) and other flame-retarded plastics were subjected to various recycling activities (Dumler-Gradl et al., 1995). After mechanical processing (hammer mill) of TBBPA-containing plastics, no PBDDs/PBDFs could be detected. After (laboratory-scale) pyrolysis and solvolysis procedures, especially of chopped printed circuit boards, high amounts of PBDDs/PBDFs were found in the condensed (a) or extracted (b) material (e.g. in (a), sums of tetra-, penta-, hexa-, and heptaBDFs were 7035, 5470, 213, and 31 µg/g, respectively; 2,3,7,8-TeBDF: 29 µg/kg; 2,3,4,7,8-PeBDF: 24 µg/kg; or in (b), sums

of tetra-, penta-, and hexaBDDs were 0.23, 0.20, and 1.98 µg/kg, respectively; 2,3,7,8-TeBDD: 0.06 µg/kg; sums of tetra-, penta-, hexa-, and heptaBDFs were 230, 309, 97.5, and 0.9 µg/kg, respectively; 2,3,7,8-TeBDF: 5 µg/kg; 2,3,4,7,8-PeBDF: 12.6 µg/kg). Only small amounts of mono- and diBDFs were detected at (pilot-scale) pyrolysis of printed circuit boards and mixed flame-retarded plastics.

Plastic material recovered from cables and subsequently burned contained PBDDs/PBDFs (mono to hexa) at a concentration of 36.5 µg/kg (Lorenz, 1994).

3.9.3.2 *Metals*

The formation of PCDDs/PCDFs (WHO, 1989) and other organochlorine compounds (Sinkkonen et al., 1994) during metal reclamation is well known. However, the occurrence of the brominated analogues is documented in only a few cases. PBDDs/PBDFs (tri to hexa) were analysed in ash samples ($n = 2$) from a metal reclamation factory in southern Taiwan (Watanabe et al., 1993), and PXDDs/PXDFs were identified in ash samples ($n = 2$) from a secondary copper furnace in the USA (Harless et al., 1989).

Whereas PBDDs were not detected (detection limits: <0.25–<1.0 µg/kg) in the Taiwanese study, PBDFs reached total concentrations of 15–45 µg/kg (composed of triBDFs: 5–9 µg/kg, tetraBDFs: 4–15 µg/kg, pentaBDFs: 3–9 µg/kg, and hexaBDFs: 3–12 µg/kg). PXDDs/PXDFs were also observed, but not quantified, in these samples. The ratio of PBDFs to PCDFs was about 1 : 100. PBDEs were also present at somewhat lower levels than PCDFs (Watanabe et al., 1993).

The ash from a secondary copper furnace contained Br_1Cl_6DDs (1–34 µg/kg) and Br_1Cl_6DFs (17 µg/kg). These concentrations were approximately one order of magnitude lower than those of PCDDs (27–411 µg/kg) and PCDFs (89–173 µg/kg) (Harless et al., 1989).

3.10 Presence in automotive exhaust

The combustion processes occurring in motors of automobiles can lead to the formation of PBDDs/PBDFs and PXDDs/PXDFs

(Buser, 1987a,b,c). Simulation experiments using iso-octane (plus additives) as a defined fuel (Ballschmiter et al., 1990; Bacher et al., 1991) and single measurements using commercial petrol (Haglund et al., 1988) gave positive results. The same was true for a joint project between three German universities, which was initiated to provide more and representative data on the emission of PHDDs/PHDFs from motor vehicles under realistic conditions (Hagenmaier et al., 1990a; Hutzinger et al., 1990; Weberruß, 1990; Schwind et al., 1991; Dawidowsky, 1993). In this study, 46 exhaust samples were taken from Otto (spark-ignition or Otto-cycle engine) and diesel motors running with different commercial fuels. The experiments were carried out mostly as stationary motor tests.

PHDDs/PHDFs were detected in emissions of motors running on leaded petrol and on unleaded petrol with and without catalytic converters (Haglund et al., 1988; Hagenmaier et al., 1990a; Hutzinger et al., 1990; Dawidowsky, 1993) and in emissions of diesel engines (Hagenmaier et al., 1990a; Hutzinger et al., 1990; Dawidowsky, 1993) (see also Table 22). Because of the brominated and chlorinated scavengers (dibromo- and dichloroethane) used in leaded petrol, the highest levels of PHDDs/PHDFs were found with this type of petrol. The emissions reached several thousand ng/m^3 (e.g. >6000 ng/m^3 in exhaust air or 90 000 ng/litre fuel used). Unleaded petrol, which contained only trace amounts of halogenated compounds, caused much lower emissions of PHDDs/PHDFs (approximately two orders of magnitude lower). A further reduction to below 5% of the emissions from non-halogenated petrol was achieved in experiments using catalytic cleaning of the exhaust. The values for diesel engines were a little higher than those found with the Otto motors run on unleaded petrol and equipped with a catalytic converter. In the case of diesel combustion, the sources of the halogens needed for the PHDD/PHDF formation could not be clearly identified (e.g. Hagenmaier et al., 1990a). The negative results of Haglund et al. (1988) obtained for a heavy-duty diesel truck are thought to be due to their higher detection limits compared with the later studies (Dawidowsky, 1993) showing positive results for both diesel cars (n = 8) and trucks (n = 2). Differences in results involving diesel fuel combustion can also be due in part to differences in diesel fuel compositions (seasonal, manufacture).

Table 22. Emissions of PHDDs/PHDFs from automobile combustion engines under various motor/fuel conditions[a,b]

PHDDs/PHDFs	Mean emissions (ng/litre fuel)[c]							
	Leaded petrol (n = 4)		Unleaded petrol (n = 6)		Unleaded petrol with catalytic converter (n = 3)		Diesel fuel (n = 8)	
Dibenzo-*p*-dioxins								
PBDDs	1576	(Br_1–Br_4)	18	(Br_1–Br_4)	0.8	(Br_1–Br_3)	1.9	(Br_1–Br_3)
PXDDs	742	(up to X_5)	4	(up to X_4)	0.6	(up to X_4)	0.4	(up to X_3)
PCDDs	606	(Cl_1–Cl_8)	42	(Cl_1–Cl_8)	0.9	(Cl_1–Cl_8)	3.4	(Cl_1–Cl_8)
Total PHDDs	2924		64		2.3		5.7	
Dibenzofurans								
PBDFs	45 428	(Br_1–Br_6)	364	(Br_1–Br_5)	22.6	(Br_1–Br_5)	136	(Br_1–Br_4)
PXDFs	22 418	(up to X_5)	101	(up to X_5)	6.9	(up to X_5)	24	(up to X_4)
PCDFs	21 698	(Cl_1–Cl_8)	502	(Cl_1–Cl_8)	7.9	(Cl_1–Cl_8)	167	(Cl_1–Cl_8)
Total PHDFs	89 544		967		37.4		327	

[a] Adapted from Dawidowsky (1993).

[b] The exhaust samples were obtained from Otto motors run on leaded petrol and unleaded petrol with and without catalytic converters and from diesel engines. Br/Cl content of fuels: leaded petrol, 76/70 mg/kg; unleaded petrol and diesel, <1 mg/kg.

[c] Homologue groups present are given in parentheses.

A considerable portion of the PHDDs/PHDFs consisted of PBDDs/PBDFs and PXDDs/PXDFs (Dawidowsky, 1993; see also Table 22). In exhaust gases from combustion of leaded petrol, they were more abundant than PCDDs/PCDFs: PBDDs/PBDFs > PXDDs/PXDFs ≈ PCDDs/PCDFs.

Generally, in all studies performed, the concentrations of dibenzofurans exceeded those of the dibenzo-*p*-dioxins (see also Table 22). There was a dominance of lower substituted homologues (mono to tri), as can be seen, for example, from the homologue profile of PBDFs and PXDFs shown in Table 23.

Table 23. Homologue distribution pattern of PBDFs and PXDFs detected in exhaust samples (n = 4) from vehicle motors run on leaded petrol[a]

Homologue groups	Mean emissions (ng/litre fuel)
PBDFs	
MonoBDFs	27 501
DiBDFs	17 496
TriBDFs	354
TetraBDFs	70
PentaBDFs	6.6
HexaBDFs	0.2
HeptaBDFs	n.d.[b]
PXDFs	
Br_1Cl_1DFs	18 941
Br_1Cl_2DFs	377
Br_2Cl_1DFs	2649
Br_1Cl_3DFs	67
Br_2Cl_2DFs	174
Br_3Cl_1DFs	138
Br_3Cl_2DFs	19.5
Br_4Cl_1DFs	7.4

[a] Adapted from Dawidowsky (1993).
[b] n.d. = not detected.

Brominated and mixed brominated/chlorinated congeners with 2,3,7,8-substitution were not determined. However, they may be present at trace amounts as well as their chlorinated counterparts (Marklund et al., 1987, 1990; Bingham et al., 1989; Hagenmaier et al., 1990a; Bacher et al., 1991).

PHDDs/PHDFs were detected not only in exhaust samples but also in residues adhering to mufflers. An absorption in the mufflers was observed for PBDDs/PBDFs and PXDFs (Ballschmiter et al., 1990) and for PCDDs/PCDFs (Ballschmiter et al., 1990; Marklund et al., 1990). The two samples tested by Ballschmiter et al. (1990) showed some correlations to the exhaust samples (dominance of lower brominated homologues, prevalence of PBDFs over PBDDs and of PBDFs over PCDFs).

Results from traffic-related environmental samples are discussed in chapter 5.

The increasing use of lead-free petrol will reduce the input of PHDDs/PHDFs into the environment from this source (cf. Hagenmeier, 1994).

3.11 Formation during textile processing

Different textile processes (resin finish on the basis of magnesium chloride [$MgCl_2$], flame-proof finishes on the basis of Sb_2O_3/hexabromocyclododecane, ammonium bromide [NH_4Br], and PVC) were tested for the occurrence and formation of PBDDs/PBDFs (Sedlak et al., 1996). The eight 2,3,7,8-substituted congeners of the 1994 German Dioxin Directive (see Appendix I) were determined in the exhaust air, the textiles before and after processing, and the chimney depositions. Exhaust air concentrations were between 29 and 102 pg/m^3. Concentrations in textiles before processing were 1.35–5.65 ng/kg and after processing 1.80–41.0 ng/kg. The only significant increase was observed for the PVC process (5.65 versus 41.0 ng/kg). Chimney depositions showed concentrations of 92.3–6618 ng/kg. Only traces of PXDDs/PXDFs were detected in the textiles, but chimney depositions contained up to 17 µg/kg.

4. ENVIRONMENTAL TRANSPORT, DISTRIBUTION, AND TRANSFORMATION

4.1 Transport and distribution between media

Because of their physicochemical properties (see chapter 2), PBDDs/PBDFs are expected to be preferably distributed into carbon- or fat-rich compartments.

4.1.1 Air

Airborne PBDDs/PBDFs released from various sources (see chapter 3) are transported in both the particulate and vapour phase. In traffic-related air samples, the lower halogenated PBDDs/PBDFs (Br_1, Br_2), PCDDs/PCDFs (Cl_1 through Cl_3), and PXDDs/PXDFs (X_2) were detected predominantly in the gaseous phase (Ballschmiter et al., 1990). In contrast, Lutes et al. (1992a) found that tetra- and pentaBDDs/BDFs generated by combustion of polyurethane foam containing PBDEs partitioned primarily to the particulate phase. The ratio of concentrations between particulate phase and particulate plus vapour phase was 0.95–0.99. Monitoring of ambient city air (Harless et al., 1992) for PBDDs/PBDFs (tetra through hexa), PCDDs/PCDFs (tetra through octa), and PXDDs/PXDFs (tetra: Br_1Cl_3) revealed that most of the penta- and hexa- and about 60% of the tetraBDDs/BDFs were associated with the particulate phase. Most of the hepta- and octaCDDs/CDFs were also particle-bound, whereas the lower chlorinated congeners, including the mixed tetrahalogenated compounds, were distributed to the gaseous phase.

During atmospheric transport, photochemical transformation can occur (see section 4.2.1). There are no data on deposition of PBDDs/PBDFs to soil, vegetation, or water.

4.1.2 Water and sediments

Few data are available on the movement of PBDDs/PBDFs through water and associated sediment.

Watanabe (1988) determined adsorption coefficients on sediment of several halogenated organic compounds, including PBDFs. Adsorption coefficients (log K_d; where K_d = [µg/g sediment]/[µg/ml water]) were 4.51, 4.62, and 4.79 for triBDFs, tetraBDFs, and pentaBDFs, respectively.

4.1.3 Soil

To date, no experimental data on the behaviour of PBDDs/PBDFs in soil are available. Mobility of PBDDs/PBDFs in soil is assumed to be governed by their low water solubility (see chapter 2) and their strong adsorption to particulate matter. Their mobility depends on soil type (organic matter content, dissolved humic acids, pH, etc.), weather conditions, and congeners studied. In general, mobility is expected to be rather low; however, in special cases, such as at waste disposal sites where organic solvents are concomitantly present, significant leaching could occur. As shown for PCDDs (Webster et al., 1986), the presence of dissolved humic substances in water can also enhance the solubility of such compounds.

Other transport mechanisms to be considered include transport via dust particles or volatilization (to air and vegetation), via eroded soil (to surface waters), and via biomass removal, as reported for PCDDs/PCDFs (Young, 1983; WHO, 1989).

The persistence of PCDDs/PCDFs (tetra to octa) in soil environments was found to be high (e.g. Orazio et al., 1992), and movement of 2,3,7,8-TeCDD was primarily associated with liquid carrier contaminants such as petroleum oil (Kapila et al., 1989). 2,3,7,8-TeCDD half-lives in soil were calculated to be as high as 10–12 years (di Domenico et al., 1980; Young, 1983).

4.1.4 Biota

Isolated reports have been published on the presence of PBDDs/PBDFs in animals and plants (see chapter 5). However, no data are available on the transport of PBDDs/PBDFs and their distribution between environmental media and biota. The similar high octanol/water partition coefficients calculated for selected PCDDs/PCDFs,

PBDDs/PBDFs, and PXDDs/PXDFs (Fiedler & Schramm, 1990; see also Table 5 in section 2.2.1) indicate that PBDDs/PBDFs may have a bioavailability qualitatively comparable to that reported for PCDDs/PCDFs (de Wit, 1993; Rappe, 1993). A factor influencing the range of bioavailability both within the homologue groups and between analogues is the molecule size.

There are no data on the transfer of PBDDs/PBDFs to plants via deposition processes or uptake from soil or on their transfer along the terrestrial food-chain. However, it may be expected that the brominated congeners behave qualitatively like their chlorinated analogues.

PCDDs/PCDFs can also enter aquatic biota, primarily via sediment (Fairchild et al., 1992; de Wit, 1993; Fletcher & McKay, 1993; Pruell et al., 1993). PBDDs/PBDFs are expected to have a similar potential (see also chapter 5).

4.2 Environmental transformation

4.2.1 *Photochemical degradation*

Photolysis of PBDDs/PBDFs was studied in organic solvents and on quartz surfaces in the laboratory as well as in soil and on soot particles under outdoor conditions. The slowest photolytic reactions were observed under the latter, more environmentally relevant, conditions. From experiments with octaCDD, it is known that photochemical dechlorination on soil takes place in the axial positions, resulting in 2,3,7,8-TeCDD; in solutions, the lateral chlorines are removed, resulting in 1,4,6,9-TeCDD (Kieatiwong et al., 1990).

Laboratory studies showed that PBDDs/PBDFs and PXDDs/PXDFs degrade in organic solvents after irradiation with sunlight, artificial light, or UV light (Buser, 1988; Neupert et al., 1988; Lahaniatis et al., 1991; Lenoir et al., 1991; Chatkittikunwong & Creaser, 1994a; Ritterbusch et al., 1994a; Watanabe et al., 1994). The major photochemical pathway is a reductive debromination, resulting in the formation of lower brominated congeners (Buser, 1988; Neupert et al., 1988; Lenoir et al., 1991; Chatkittikunwong & Creaser, 1994a; Ritterbusch et al., 1994a) and, finally, in the formation of unsubsti-

tuted dibenzo-*p*-dioxin and dibenzofuran (Buser, 1988). Other products also observed after photolysis were diaryl ethers, which were generated by ring fission of mono- and diBDDs but not of higher brominated PBDDs (Lenoir et al., 1991), and, occasionally, benzyl derivatives. The latter were formed by reaction of photoproducts of 2,3,7,8-TeBDD with the solvent toluene (Neupert et al., 1988). Studies of different PBDD/PBDF congeners indicated that the rate of decomposition depends on the bromine substitution pattern. Generally, higher brominated congeners and those with lateral bromines had shorter half-lives (Buser, 1988; Neupert et al., 1988; Lenoir et al., 1991); in one experiment (Lenoir et al., 1991), however, octaBDD was more stable than a hexaBDD. Two dibenzofurans (2,3,7,8-TeBDF and octaBDF) dissolved in toluene and irradiated by fluorescent light for several days were found to decompose more rapidly than the corresponding dibenzo-*p*-dioxins (Neupert et al., 1988). The rate of photolysis decreased with increasing polarity of the solvent, as tested for 2,3,7,8-TeBDD (Lahaniatis et al., 1991) and 1,2,3,4-TeBDD (Lenoir et al., 1991).

The calculated half-lives (assuming a first-order kinetic scheme) were in the range of minutes (use of direct sunlight or UV light and quartz vials) or of the order of 100–1000 h (use of laboratory daylight or artificial light and glass vials). For example, half-lives of 0.8 and 0.7 min were estimated for TBDD and TBDF, respectively, after 60 min of sunlight irradiation in organic solution in quartz vials (Buser, 1988). The mean half-lives found for PBDDs and PXDDs exposed to laboratory daylight in dodecane solution stored in glass vials were 480 h for tetraBDDs, 150 h for pentaBDDs, and 300–995 h for various PXDDs (tetra- to hexahalogenated congeners with various Br/Cl combinations: Br_{1-3}/Cl_{1-5}) (Chatkittikunwong & Creaser, 1994a).

Compared with the chlorinated analogues, PBDDs/PBDFs had a faster photolytic reaction in iso-octane, with half-lives of 3 min for 1,2,3,4-TeBDD and 380 min for 1,2,3,4-TeCDD (Buser, 1988; Lenoir et al., 1991). The easier loss of bromine than of chlorine from the parent molecule has important consequences for the PXDDs/PXDFs, in that they undergo photolytic degradation to form PCDDs/PCDFs. Consistently short photolytic half-lives (0.5–4 min) were observed for

the mixed mono- and dibromotetrachloroDDs/DFs tested, whereas the resulting tetraCDDs/CDFs were much more stable (Buser, 1988).

The possibility of removing PBDDs/PBDFs from laboratory wastes by UV photolysis was examined by Ritterbusch et al. (1994a,b). This method of decontamination was successful when solutions with a low concentration of other photochemically active species were applied. It was unsuitable for the degradation of the PBDD/PBDF contamination of bromophenols, because the rate of photochemical PBDD/PBDF formation was greater than the rate of degradation. The rate of photolytic degradation was slower in waste solutions than in PBDD/PBDF standard solutions (Ritterbusch et al., 1994a). The degradation of PBDDs/PBDFs occurred faster than that of PCDDs/PCDFs (Ritterbusch et al., 1994b).

Photolysis of PBDDs/PBDFs (Br_4) and PXDDs/PXDFs (X_5, X_6: mono- and dibromo-2,3,7,8-TeCDDs/CDFs) occurred much more slowly on quartz surfaces, under sunlight, than in organic solvents (Buser, 1988). The photolytic half-lives of the tetrabrominated congeners tested were in the range of 30 h; those for the chlorinated analogues were in the range of 65–300 h (see Table 24).

Table 24. Sunlight-induced photolysis of tetrahalogenated dibenzo-*p*-dioxins and dibenzofurans dispersed as solid films[a]

Compound	Estimated half-life[b] (h)
Dibenzo-*p*-dioxins	
1,2,3,4-TeBDD	26
2,3,7,8-TeBDD	32
1,2,3,4-TeCDD	65
2,3,7,8-TeCDD	300
Dibenzofurans	
2,3,7,8-TeBDF	35
2,3,7,8-TeCDF	120

[a] From Buser (1988).
[b] Values derived from few data points obtained after a total exposure time of 10 and 20 h; estimated accuracy of half-lives ± 50%.

These solid-phase experiments appeared to predict the real environmental behaviour of PBDDs/PBDFs far better than the organic solution-phase experiments. Photodegradation studied in soil (Chatkittikunwong & Creaser, 1994a) and on airborne particles (Lutes et al., 1990, 1992a,b) under outdoor conditions was found to be a slow process. Half-lives of PBDDs and PXDDs in soil samples that were placed outdoors over a 3-month period were in the range of 600–4000 h for tri- to hexahalogenated congeners. These half-lives were, on average, four times longer than those estimated for the same congeners in solution in the laboratory. The half-life of tetraBDD isomers in surface soil was estimated to be 3–6 months (Chatkittikunwong & Creaser, 1994a). Other tests conducted in Teflon film chambers under realistic outdoor conditions showed that PBDDs/PBDFs (tetra through hexa) adsorbed on incinerator soot particles remained relatively stable or degraded only slowly during 6 h. No significant decay of the 2,3,7,8-substituted congeners was found (Lutes et al., 1990, 1992a,b). The photodegradation, if occurring, was believed to have a half-life of at least 3 h and probably much longer (Lutes et al., 1992a,b). Similarly, particle-bound emissions of tetraBDDs, tetraBDFs, and pentaBDFs (isomers not determined), produced from high-temperature (670–780 °C) combustion of polyurethane foam containing PBDEs and monitored in outdoor Teflon chambers in the presence of sunlight, were stable over several hours. In contrast, a decay of tetraBDDs was observed after low-temperature (400–470 °C) combustion of the polyurethane foam (Birla & Kamens, 1994). Little disappearance of higher halogenated PXDFs was seen in preliminary tests monitoring halogenated dibenzofurans associated with airborne dust collected on a glass filter and exposed to sunlight (Watanabe et al., 1994).

4.2.2 Microbial degradation

Like other halogenated aromatics, PBDDs/PBDFs seem to be very recalcitrant against microbial degradation. Only a monobrominated dibenzofuran (2-bromodibenzofuran) was tested and found to be degraded by bacteria. Bacteria of the genus *Pseudomonas* isolated from water of the river Rhine or of industrial wastewater treatment plants and cultured in the laboratory on 1,2-dichlorobenzene or 4-

chlorophenol as single-carbon source were able to oxidize 2-bromodibenzofuran (Springer & Rast, 1988).

There are some reports on the degradation of the parent, non-halogenated dibenzo-*p*-dioxin and/or dibenzofuran by microorganisms in soil and by white rot fungi (Cerniglia et al., 1979; Hammel et al., 1986; Bumpus, 1989; Hofmann et al., 1992). Some active bacterial strains were found in several genera — for example, *Beijerinckia* (Klecka & Gibson, 1980), *Brevibacterium* (Strubel et al., 1989, 1991), *Pseudomonas* (Klecka & Gibson, 1979; Foght & Westlake, 1988; Springer & Rast, 1988; Fortnagel et al., 1989a,b, 1990; Harms et al., 1990; Figge et al., 1991), *Sphingomonas* (Wittich et al., 1992; Figge et al., 1993; Happe et al., 1993), and *Staphylococcus* (Monna et al., 1993).

4.3 Bioaccumulation and biomagnification

At present, bioaccumulation, bioconcentration, or biomagnification factors for specific PBDD/PBDF congeners are not available. The presence of PBDDs/PBDFs in animals (section 5.1.6.2) and in humans (section 5.3.2), as seen in a few isolated studies, is indicative of their accumulation potential (see also section 6.5.1). This is to be expected from the lipophilic properties of PBDDs/PBDFs and the high accumulation potential of better-studied related compounds, such as PCDDs/PCDFs (e.g. WHO, 1989; Cook et al., 1991). The extent of accumulation and biomagnification may vary depending on species and congeners tested, as found for PCDDs/PCDFs (e.g. de Wit, 1993; de Wit et al., 1993; Rappe, 1993; Walker & Peterson, 1994).

5. ENVIRONMENTAL LEVELS AND HUMAN EXPOSURE

5.1 Environmental levels

Limited data are available on levels of PBDDs/PBDFs in the environment. Most monitoring data available for PBDDs/PBDFs have been collected near identified sources (e.g. roadways).

5.1.1 Air

5.1.1.1 Ambient air

Sources of PBDDs/PBDFs are complex and may have changed in recent years. Therefore, temporal differences and changes must be taken into account. In some European countries, the use of leaded petrol (and the use of scavengers) has been largely abandoned; in the USA, the use of leaded petrol with scavengers declined even earlier than in Europe. In other countries, this type of petrol may still be in use. Thus, some of the data given may not be representative of the present situation (e.g. in motorway tunnels, urban areas, etc.). Furthermore, the type of waste incineration has drastically changed within the last decade in industrialized countries. These variables have to be kept in mind when evaluating the following compilations and Table 25.

1) Air samples

PBDD/PBDF levels analysed in urban air, more or less close to traffic, and in air collected at industrial sites are compiled in Table 25. The sampling methods used covered particulate-associated and vapour-phase PBDDs/PBDFs. Only low concentrations of PBDDs (mono through tetra) were detected, with maximum concentrations of about 0.85 pg monoBDDs/m^3 in a motorway tunnel and an underground garage. Higher brominated (penta through octa) homologues were not detected or were not analysed (Table 25). No PXDDs were found (Ballschmiter et al., 1990).

PBDFs were found to a greater extent than PBDDs (Table 25). Of the homologue groups, mono- to hexaBDFs could be detected; hepta-

Table 25. Concentrations of PBDDs/PBDFs in ambient air

Congener	Country	Sampling site (*n*)	Year of sampling[a]	Concentration[b] (pg/m^3)	Reference
Dibenzo-*p*-dioxins					
MonoBDDs	Germany	motorway tunnel (1)	n.sp.	0.85	Ballschmiter et al. (1990)
	Germany	air outlet from an underground garage (2)	n.sp.	0.50–0.86	Ballschmiter et al. (1990)
DiBDDs	Germany	motorway tunnel (1)	n.sp.	<0.15	Ballschmiter et al. (1990)
	Germany	air outlet from an underground garage (2)	n.sp.	n.d.–<0.15	Ballschmiter et al. (1990)
TriBDDs	Germany	motorway tunnel in Essen (3)	1990	0.37–0.75[c]	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	0.05–0.09	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d. (0.03)	Päpke et al. (1990)
	Japan	urban area of Osaka (5)	n.sp.	n.d.–0.09	Watanabe et al. (1992)
	Taiwan	recycling resource centre (3)	n.sp.	0.3–0.5	Watanabe et al. (1992)
TetraBDDs	Germany	motorway tunnel in Essen (3)	1990	n.d.–0.18	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d.–0.04	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d.	Päpke et al. (1990)
	Japan	urban area of Osaka (5)	n.sp.	n.d.–0.3	Watanabe et al. (1992)
	Taiwan	recycling resource centre (3)	n.sp.	n.d.–0.2	Watanabe et al. (1992)
	USA	Research Triangle Park, NC (5)	1990/91	n.d.	Harless et al. (1992)
PentaBDDs	Germany	motorway tunnel in Essen (3)	1990	n.d.	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d.	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d.	Päpke et al. (1990)
	Japan	urban area of Osaka (5)	n.sp.	n.d.	Watanabe et al. (1992)
	Taiwan	recycling resource centre (3)	n.sp.	n.d.	Watanabe et al. (1992)
	USA	Research Triangle Park, NC (5)	1990/91	n.d.	Harless et al. (1992)

Table 25 (contd).

HexaBDDs	Germany	motorway tunnel in Essen (3)	1990	n.d. (0.1–0.2)	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d. (0.1–0.4)	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d. (0.2)	Päpke et al. (1990)
	Japan	urban area of Osaka (5)	n.sp.	n.d.	Watanabe et al. (1992)
	Taiwan	recycling resource centre (3)	n.sp.	n.d.	Watanabe et al. (1992)
	USA	Research Triangle Park, NC (5)	1990/91	n.d.	Harless et al. (1992)
HeptaBDDs	Germany	motorway tunnel in Essen (3)	1990	n.d. (0.3–0.5)	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d. (0.5–0.7)	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d. (0.5)	Päpke et al. (1990)
OctaBDD	Germany	motorway tunnel in Essen (3)	1990	n.d. (0.7–0.8)	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d. (1)	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d. (1)	Päpke et al. (1990)
Dibenzofurans					
MonoBDFs	Germany	motorway tunnel (1)	n.sp.	73.72	Ballschmiter et al. (1990)
	Germany	air outlet from an underground garage (2)	n.sp.	37.26–42.12	Ballschmiter et al. (1990)
DiBDFs	Germany	motorway tunnel (1)	n.sp.	28.50	Ballschmiter et al. (1990)
	Germany	air outlet from an underground garage (2)	n.sp.	2.12–6.20	Ballschmiter et al. (1990)
TriBDFs	Germany	motorway tunnel (1)	n.sp.	n.d. (0.1)	Ballschmiter et al. (1990)
	Germany	air outlet from an underground garage (2)	n.sp.	n.d. (0.05–0.1)	Ballschmiter et al. (1990)
	Germany	motorway tunnel in Essen (3)	1990	17–25	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	0.71–2.0	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	0.40–0.82	Päpke et al. (1990)
	Japan	urban area of Osaka (5)	n.sp.	0.3–1.0	Watanabe et al. (1992)

Table 25 (contd).

Congener	Country	Sampling site (*n*)	Year of sampling[a]	Concentration[b] (pg/m^3)	Reference
TetraBDFs	Germany	motorway tunnel (1)	n.sp.	n.d. (0.1)	Ballschmiter et al. (1990)
	Germany	air outlet from an underground garage (2)	n.sp.	n.d. (0.05–0.1)	Ballschmiter et al. (1990)
	Germany	motorway tunnel in Essen (3)	1990	1.50–3.35	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	0.15–0.53	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d.	Päpke et al. (1990)
	Japan	urban area of Osaka (5)	n.sp.	0.2–2.3	Watanabe et al. (1992)
	Taiwan	recycling resource centre (3)	n.sp.	2.1–6.6	Watanabe et al. (1992)
	USA	Research Triangle Park, NC (5)	1990/91	0.13–0.20	Harless et al. (1992)
PentaBDFs	Germany	suburban area of the small city Borken (3)	1990	n.d.	Päpke et al. (1990)
	Japan	urban area of Osaka (5)	n.sp.	0.2–3.7	Watanabe et al. (1992)
	Taiwan	recycling resource centre (3)	n.sp.	1.8–7.7	Watanabe et al. (1992)
	Germany	motorway tunnel (1)	n.sp.	n.d. (0.1)	Ballschmiter et al. (1990)
	Germany	air outlet from an underground garage (2)	n.sp.	n.d. (0.05–0.1)	Ballschmiter et al. (1990)
	Germany	motorway tunnel in Essen (3)	1990	0.32–0.41	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	0.08–0.22	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d.–0.14	Päpke et al. (1990)
	USA	Research Triangle Park, NC (5)	1990/91	n.d.–0.22	Harless et al. (1992)
HexaBDFs	Germany	motorway tunnel in Essen (3)	1990	n.d. (0.1–0.2)	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d. (0.2–0.4)	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d. (0.2)	Päpke et al. (1990)

Table 25 (contd).

	Japan	urban area of Osaka (5)	n.sp.	0.3–5.1	Watanabe et al. (1992)
	Taiwan	recycling resource centre (3)	n.sp.	1.1–3.4	Watanabe et al. (1992)
	USA	Research Triangle Park, NC (5)	1990/91	n.d.–0.30	Harless et al. (1992)
HeptaBDFs	Germany	motorway tunnel in Essen (3)	1990	n.d. (0.4–0.5)	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d. (0.3–0.7)	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d. (0.5)	Päpke et al. (1990)
OctaBDF	Germany	motorway tunnel in Essen (3)	1990	n.d. (0.7–0.8)	Päpke et al. (1990)
	Germany	urban area of Düsseldorf (6)	1990	n.d. (1)	Päpke et al. (1990)
	Germany	suburban area of the small city Borken (3)	1990	n.d. (1)	Päpke et al. (1990)

[a] n.sp. = not specified.
[b] n.d. = not detected (detection limits in parentheses, if specified).
[c] Containing possibly confounding components.

and octaBDFs were not identified. Because of the small database, only trends in the homologue pattern can be seen. It appears that lower brominated homologues (mono through tetra) dominate, particularly in samples related to traffic. The highest concentration was measured for monoBDFs in an air sample from a motorway tunnel and amounted to 74 pg/m^3 (Table 25). However, Harless et al. (1992) found pentaBDFs and hexaBDFs (0.22 and 0.30 pg/m^3, respectively) in addition to tetraBDFs (0.19 pg/m^3 air) only after long-term sampling (7 days; 2660 m^3 air) of ambient air (at Research Triangle Park, NC, USA). Eight tetraBDF, two pentaBDF, and one hexaBDF isomers were detected. Using shorter sampling periods (24 h), only tetraBDFs were detected, at concentrations ranging from 0.13 to 0.20 pg/m^3. The lower concentrations of PBDFs reported from this study would probably be due in part to the absence of scavengers in motor fuel in North America.

The highest concentrations of PBDF homologue groups were found in motorway tunnels (Table 25). No data are available on concentrations of lower brominated homologues (Br_1 through Br_3) in samples from industrial areas. The highest concentrations of tetraBDFs were reported in the vicinity of a resource recycling centre in Taiwan (2.1–6.6 pg/m^3) and in the German motorway tunnels (n.d.–3.4 pg/m^3) (see Table 25).

The sums of total PBDDs/PBDFs (tri to hexa) in the air of a motorway tunnel, of a city, and of a suburban area in Germany were 22.3/0.7 pg/m^3 (means; n = 3), 1.97/0.08 pg/m^3 (means; n = 6), and 0.59 pg/m^3/n.d. (means; n = 3), respectively. The concentrations of 2,3,7,8-TeBDF ranged from n.d. to 0.28 (n = 12) and those of 1,2,3,7,8-PeBDF from n.d. to 0.08 pg/m^3 (n = 12) (Päpke et al., 1990; Hiester, 1992).

PBDFs and their possible precursors, PBDEs, were concomitantly found in air samples from industrial areas of Taiwan and Japan. Tri-, tetra-, penta-, and hexaBDE concentrations ranged from 6 to 34, from 10 to 55, from 5 to 34, and from 6 to 81 pg/m^3, respectively (Watanabe et al., 1992).

Of the PXDFs, dihalogenated dibenzofurans were detected in traffic-related air samples at concentrations of up to 40.8 pg/m^3 (Cl_1Br_1DFs), which is higher than found for the brominated congeners. Concentrations of monoBDFs exceeded the levels of monoCDFs (Ballschmiter et al., 1990).

2) Dust samples

Outdoor dust samples were collected in motorway tunnels in Germany and, as a control, in the eaves of a house in a German rural area remote from the source (traffic). These and samples obtained from the USA (roadside dust) and from Japan (dust of a motorway tunnel) were analysed for PCDDs/PCDFs, PBDDs/PBDFs (Br_1 through Br_4), and PXDDs/PXDFs (up to X_4) (Ballschmiter et al., 1990). The results are summarized in Table 26 (PBDDs/PBDFs) and Table 27 (PXDDs/PXDFs). Whereas the control sample (from eaves) contained no PBDDs (detection limit: 20 ng/kg dust) and only 30 ng monoBDFs/kg dust, the samples taken close to traffic showed a complex pattern of homologues.

Concentrations of PBDDs in the dust samples were low, ranging from n.d. to 690 ng/kg, from n.d. to 960 ng/kg, from n.d. to 170 ng/kg, and from n.d. to 110 ng/kg for mono-, di-, tri-, and tetraBDDs, respectively (Ballschmiter et al., 1990).

PBDFs were present at higher levels than PBDDs and reached maximum values of 8860 ng monoBDFs/kg dust (motorway tunnel in Germany), 22 280 ng diBDFs/kg dust (motorway tunnel in Japan), 5680 ng triBDFs/kg dust (highway in the USA), and 650 ng tetraBDFs/kg dust (highway in the USA). Dust samples (n = 7) taken from motorway tunnels in Germany showed decreasing levels from mono- through tetraBDFs. Concentrations ranged from 330 to 8860 ng/kg, from 300 to 6730 ng/kg, and from n.d. to 920 ng/kg for mono-, di-, and triBDFs, respectively; tetraBDFs were not detected (Table 26).

Airborne dust collected at an urban area in Osaka (Japan) contained PBDFs (tetra to hexa) and PCDDs/PCDFs (tetra to octa) at concentrations ranging from 4.2 to 17 pg/m^3 (n = 7) and from 30 to 250 pg/m^3, respectively. Monobrominated PXDD/PXDF (tetra to octa)

Table 26. Concentrations of PBDDs/PBDFs in outdoor dust samples[a]

Congener	Country	Sampling site (*n*)	Year of sampling[b]	Concentration[c] (ng/kg)
Dibenzo-*p*-dioxins				
MonoBDDs	Germany (Ulm)	motorway tunnel (inside of city) (5)	1988/89	n.d.–180
	Germany	motorway tunnel (outside of city) (2)	1989	n.d.–690
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	≤20
	Japan	motorway tunnel (1)[e]	1978	100
DiBDDs	Germany (Ulm)	motorway tunnel (inside of city) (5)	1988/89	n.d.–120
	Germany	motorway tunnel (outside of city) (2)	1989	n.d.–540
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	n.d.
	Japan	motorway tunnel (1)[e]	1978	960
TriBDDs	Germany	motorway tunnel (7)	1988/89	n.d.
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	n.d.
	Japan	motorway tunnel (1)[e]	1978	170

Table 26 (contd).

TetraBDDs	Germany	motorway tunnel (7)	1988/89	n.d.
	Germany	eaves (1)	n.sp.	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	n.d.
	Japan	motorway tunnel (1)[e]	1978	110
Dibenzofurans				
MonoBDFs	Germany (Ulm)	motorway tunnel (inside of city) (5)	1988/89	1050–2570
	Germany	motorway tunnel (outside of city) (2)	1989	330–8860
	Germany	eaves (rural area) (1)	n.sp.	30
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	510
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	2100
	Japan	motorway tunnel (1)[e]	1978	1370
DiBDFs	Germany (Ulm)	motorway tunnel (inside of city) (5)	1988/89	1560–5030
	Germany	motorway tunnel (outside of city) (2)	1989	300–6730
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	690
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	2700
	Japan	motorway tunnel (1)[e]	1978	22 280

Table 26 (contd).

Congener	Country	Sampling site (*n*)	Year of sampling[b]	Concentration[c] (ng/kg)
TriBDFs	Germany (Ulm)	motorway tunnel (inside of city) (5)	1988/89	75–310
	Germany	motorway tunnel (outside of city) (2)	1989	n.d.–920
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	530
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	5680
	Japan	motorway tunnel (1)[e]	1978	4820
TetraBDFs	Germany	motorway tunnel (7)	1988/89	n.d.
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	650
	Japan	motorway tunnel (1)[e]	1978	310

[a] Adapted from Ballschmiter et al. (1990).
[b] n.sp. = not specified.
[c] n.d. = not detected (detection limit 20 ng/kg).
[d] Reference sample from US National Institute of Standards.
[e] Reference sample from Japanese National Institute for Environmental Studies.

Table 27. Concentrations of PXDDs/PXDFs in outdoor dust from various sources[a]

PXDDs/PXDFs	Country	Sampling site (*n*)	Year of sampling[b]	Concentration[c] (ng/kg)
Dibenzo-*p*-dioxins				
Cl_iBr_jDDs	Germany	motorway tunnel (inside of city) (5)	1988/89	n.d.
	Germany	motorway tunnel (outside of city) (2)	1989	n.d.–170
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA	street (2)[d]	1978 and 1982	n.d.
	Japan	motorway tunnel (1)[e]	1978	n.d.
Dibenzofurans				
X_2DFs				
Cl_iBr_jDFs	Germany	motorway tunnel (inside of city) (5)	1988/89	1150–4340
	Germany	motorway tunnel (outside of city) (2)	1989	300–9600
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	1280
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	4150
	Japan	motorway tunnel (1)[e]	1978	7220
X_3DFs				
Cl_iBr_2DFs	Germany	motorway tunnel (inside of city) (5)	1988/89	180–830
	Germany	motorway tunnel (outside of city) (2)	1989	n.d.–1300
	Germany	eaves (rural area) (1)	n.sp.	n.d.

Table 27 (contd).

PXDDs/PXDFs	Country	Sampling site (*n*)	Year of sampling[b]	Concentration[c] (ng/kg)
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	2270
	Japan	motorway tunnel (1)[e]	1978	1130
Cl_2Br_1DFs	Germany	motorway tunnel (inside of city) (5)	1988/89	n.d.–180
	Germany	motorway tunnel (outside of city) (2)	1989	n.d.–970
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	1160
	Japan	motorway tunnel (1)[e]	1978	7870
X_4DFs				
Cl_1Br_3DFs	Germany	motorway tunnel (7)	1988/89	n.d.
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	n.d.
	Japan	motorway tunnel (1)[e]	1978	n.d.

Table 27 (contd).

Cl_2Br_2DFs	Germany	motorway tunnel (7)	1988/89	n.d.
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	830
	Japan	motorway tunnel (1)[e]	1978	370
Cl_3Br_1DFs	Germany	motorway tunnel (7)	1988/89	n.d.
	Germany	eaves (rural area) (1)	n.sp.	n.d.
	USA (St. Louis, MO)	street (outskirts of the city) (1)[d]	1978	n.d.
	USA (Washington, DC)	highway (inside of city; pooled sample over 12 months) (1)[d]	1982	380
	Japan	motorway tunnel (1)[e]	1978	170

[a] Adapted from Ballschmiter et al. (1990).
[b] n.sp. = not specified.
[c] n.d. = not detected (detection limit 20 ng/kg).
[d] Reference sample from US National Institute of Standards.
[e] Reference sample from Japanese National Institute for Environmental Studies.

concentrations were roughly estimated at one-tenth to one-quarter those of PCDDs/PCDFs in the same samples (Watanabe et al., 1995).

PXDDs (Cl_iBr_jDDs) were identified at a maximum concentration of 170 ng/kg in a motorway tunnel (Table 27). PXDFs were found in dust samples from motorway tunnels, highways, and roads at concentrations ranging from 300 to 9600 ng/kg for X_2DFs, from n.d. to 7870 ng/kg for X_3DFs, and from n.d. to 830 ng/kg for X_4DFs (Table 27).

There are no data available for the 2,3,7,8-substituted PBDDs/PBDFs (Ballschmiter et al., 1990).

Generally, the dust samples from the German motorway tunnels contained higher concentrations of PBDFs than of PCDFs, both consisting of lower halogenated homologues (e.g. 16 510 ng mono- to triBDFs/kg versus 5610 ng mono- to tetraCDFs/kg). The relation between concentrations of PBDDs and PCDDs was varying. Whereas within PBDDs only mono- and dibrominated homologues could be detected (at sum concentrations of up to about 1200 ng/kg), there was a dominance of hepta- and octaCDDs within PCDDs (total concentrations of up to about 1700 ng/kg). Altogether, these patterns may be reflective of contributions from automobile exhaust. The sample from the eaves (remote from traffic, long-term residue) showed lower concentrations of PBDFs (30 ng monoBDFs/kg) than of PCDFs (sum of mono- to hexaCDFs: 735 ng/kg), no PBDDs, and hepta- to octaCDDs (at 180 ng/kg). The lower proportion of brominated compounds in the latter sample may be partly due to an easier photodehalogenation compared with the chlorinated ones (Ballschmiter et al., 1990).

Another sample of roadside dust (n = 1; collected after the ban of halogenated scavengers in petrol) from Germany was analysed for tetra- to heptaBDDs/BDFs and tetra- to octaCDDs (Schacht et al., 1995). PBDDs were not detected, whereas the sum of PBDFs amounted to 52 ng/kg (no 2,3,7,8-substituted congeners). The sum concentration of PCDDs/PCDFs was 581 ng/kg.

5.1.1.2 Indoor air

PBDDs/PBDFs were determined in rooms equipped with a number of operating electronic appliances (computers, monitors, printers; Tables 28, 29, and 30; Schacht et al., 1995), in an underground garage (Table 31), in a private room, and in rooms after accidental fires (see chapter 3).

PBDDs (tetra through octa) were not detected in air samples from rooms equipped with electronic appliances (Chriske et al., 1990), but they were present in dust samples (n = 3) from computer rooms (mainly tetraBDDs: 0.03–1 µg/kg; 2,3,7,8-TeBDD: n.d.–<2 µg/kg; Schacht et al., 1995). Dust samples from an underground garage tested for mono- to tetraBDDs contained no or ouly traces of monoBDDs and no PXDDs (Table 31) (Ballschmiter et al., 1990).

Total concentrations of PBDFs (Br_4 through Br_7) measured in air from "computer"-related rooms ranged from 0.23 to 1.27 pg/m^3, whereas dust samples collected in the same rooms yielded total levels of 2.43–5.48 µg/kg dust (Tables 28 and 30; UBA, 1992). The homologue pattern was also different. In contrast to air (see Table 28), the homologue pattern in dust was dominated by hexa- and heptaBDFs (Table 30; UBA, 1992). In addition, only in dust samples were there indications for the presence of 2,3,7,8-substituted tetra- and pentaBDFs (Table 30; UBA, 1992). Another study (Schacht et al., 1995) found comparable PBDF concentrations (sum PBDFs [Br_4–Br_8]: 3.6–3.8 µg/kg; 2,3,7,8-TeBDF: 0.01–0.07 µg/kg) and profiles in dust samples (n = 3), which were obtained from vacuum cleaning and from the air conditioning system of computer rooms. The PBDF concentrations in house dust (n = 1) were lower by a factor of 10. The sum concentration of PBDDs/PBDFs equalled that of PCDDs/PCDFs in dust from computer rooms (3.6–4.8 µg/kg versus 0.5–4.5 µg/kg) but was lower than that of PCDDs/PCDFs in house dust (0.3 µg/kg versus 34.2 µg/kg) (Schacht et al., 1995). Chemicals of low volatility tend to accumulate on solid surfaces and in house dust (Gebefügi, 1989; Gebefügi & Kreuzig, 1989).

It should be noted that concentrations of PBDEs concomitantly measured in the air and dust samples of the offices were considerably

Table 28. Indoor air concentrations of PBDFs

Congener groups	Type of room	Number of samples	Equipment in room (number per room)	Concentration[a] (pg/m^3)	Reference
TetraBDFs	office	6	display and computer monitors (2–8)	0.21–0.56	Chriske et al. (1990)
	office (police central traffic control)	1	monitors (approximately 50)	0.41	UBA (1992)
	office (direction rooms of a TV studio)	3	display and computer monitors (approximately 50)	<0.1–0.47	UBA (1992)
2,3,7,8-TeBDF	office	4	monitors (approximately 50)	n.d. (0.03–0.08)	UBA (1992)
PentaBDFs	office	6	display and computer monitors (2–8)	0.03–0.61	Chriske et al. (1990)
	office (police central traffic control)	1	monitors (approximately 50)	n.d. (n.sp.)	UBA (1992)
	office (direction rooms of a TV studio)	3	display and computer monitors (approximately 50)	0.1–0.5	UBA (1992)
1,2,3,7,8-PeBDF	office	4	monitors (approximately 50)	n.d. (0.05–0.1)	UBA (1992)
2,3,4,7,8-PeBDF	office	4	monitors (approximately 50)	n.d. (0.05–0.1)	UBA (1992)
HexaBDFs	office	10	monitors (2–8/approximately 50)	n.d.–0.4 (0.1)	Chriske et al. (1990); UBA (1992)
HeptaBDFs	office	10	monitors (2–8/approximately 50)	n.d. (0.1–0.2)	Chriske et al. (1990); UBA (1992)
OctaBDF	office	6	display and computer monitors (2–8)	n.d. (n.sp.)	Chriske et al. (1990)
Sum PBDFs	office	6	display and computer monitors (2–8)	0.23–1.18	Chriske et al. (1990)
	office	4	monitors (approximately 50)	0.25–1.27	UBA (1992)

[a] n.d. = not detected (detection limit in parentheses); n.sp. = not specified.

Table 29. Correlation between number of monitors operating in a room and sum of concentrations of tetra- and pentaBDFs[a]

Number of monitors per room	Sum of concentrations of tetraBDFs and pentaBDFs (pg/m^3)[b]
0 (ambient air)	<0.1
0	0.2–0.39
2	0.42–0.47
4	0.57–0.80
5[c]	0.23
8	1.18

[a] Adapted from Chriske et al. (1990).
[b] 2,3,7,8-Substituted congeners not determined.
[c] Equipment from one manufacturer.

Table 30. Concentrations of PBDFs in indoor dust samples collected in rooms equipped with a number of display and/or computer monitors[a]

PBDFs	Concentration (µg/kg)[b]			
	Sample 1 (Police traffic control office)	Sample 2 (TV studio)	Sample 3 (TV studio)	Sample 4 (TV studio)
TetraBDFs	0.351	0.196	0.265	0.295
2,3,7,8-TeBDF[c]	n.d. (0.001)	n.d. (0.001)	0.002	0.005
PentaBDFs	0.159	0.331	0.691	0.456
1,2,3,7,8-PeBDF[c]	0.004	0.012	0.020	0.015
2,3,4,7,8-PeBDF[c]	0.003	0.004	0.006	0.005
HexaBDFs	1.71	1.44	1.78	0.982
HeptaBDFs	2.41	3.51	0.744	0.693
Sum PBDFs	4.63	5.477	3.48	2.426

[a] Adapted from UBA (1992).
[b] n.d. = not detected (detection limit in parentheses).
[c] Maximum value given; co-elution with internal standard, etc., cannot be excluded.

higher than those of PBDFs. Total PBDE concentrations (Br_4 through Br_{10}) ranged from 97 to 969 pg/m^3 air and from 507 to 2939 µg/kg dust (UBA, 1992).

The PBDF profile (Br_1 through Br_4) found in dust from an underground garage was dominated by the lower brominated homologues (Ballschmiter et al., 1990). Concentrations ranged from 150 to 560 ng/kg dust for mono- and diBDFs in samples from the floor, but they increased at special sampling sites (wall, ventilation motors) to a maximum of 3500 ng diBDFs/kg dust (Table 31).

Table 31. Concentrations of PBDDs/PBDFs and PXDDs/PXDFs in dust samples collected in Germany in 1988/89 from an underground garage[a]

	Concentration[b] (ng/kg)		
	Floor (n = 6)	Wall (n = 1)	Waste air motors (n = 1)
Dibenzo-*p*-dioxins			
MonoBDDs	n.d.	n.d.	40
DiBDDs–tetraBDDs	n.d.	n.d.	n.d.
Cl_xBr_yDDs	n.d.	n.d.	n.d.
Dibenzofurans			
MonoBDFs	150–390	550	2800
DiBDFs	280–560	820	3500
TriBDFs	n.d.	n.d.	500
TetraBDFs	n.d.	n.d.	n.d.
Cl_1Br_1DFs[c]	180–630		4300
Cl_1Br_2DFs[c]	n.d.		790
Cl_2Br_1DFs[c]	n.d.		520
Cl_1Br_3DFs[c]	n.d.		n.d.
Cl_2Br_2DFs[c]	n.d.		n.d.
Cl_3Br_1DFs[c]	n.d.		n.d.

[a] Adapted from Ballschmiter et al. (1990).
[b] n.d. = not detected (detection limit 20 ng/kg).
[c] Concentrations for floor and wall dust samples combined.

PXDFs were analysed for and detected in dust samples from the underground garage (Ballschmiter et al., 1990). Their concentrations ranged from n.d. to 4300 ng/kg dust for di- through tetrahalogenated PXDFs (Table 31).

5.1.2 Water and sediment

No information is available on contamination of water with PBDDs/PBDFs.

River and marine sediment samples ($n = 5$) from Japan monitored for tetra- to hexaBDDs/BDFs contained tetraBDDs (n.d.–0.006 µg/kg dry weight) and tetra- to hexaBDFs at total concentrations of 0.03–0.37 µg/kg dry weight. There were also data on PXDDs/PXDFs (Watanabe et al., 1995).

Ballschmiter et al. (1990) investigated sediment (sludge) collected from a drain that received runoff water from a German motorway crossing. They found no PBDDs (Br_1 through Br_4), but PBDFs and PXDFs were found, their concentrations ranging from 180 to 1690 ng/kg dry sludge (di- and trihalogenated homologues; sum of PBDFs/PXDFs: 2500/1850 ng/kg). Although this was a single sample, it showed, together with the results for PCDDs/PCDFs, a pattern recognized as typical for automobile-derived contamination — that is, predominance of PBDFs over PCDFs, prevalence of dibenzofurans over dibenzo-*p*-dioxins, and increased concentrations of lower halogenated dibenzo-*p*-dioxins and dibenzofurans (Ballschmiter et al., 1990). Similar correlations were observed with samples of soil (section 5.1.3) and grass (section 5.1.6.1) collected near motorways (Ballschmiter et al., 1990).

Sediments recently sampled from road sewers ($n = 2$) in Germany (Schacht et al., 1995) had lower concentrations of PBDFs (sum of tetra to hepta: up to 300 ng/kg) than of PCDFs (sum of tetra to octa: up to 1300 ng/kg).

Analyses of laminated sediment core from the Baltic Proper revealed PBDEs in sediment layers dating from the 1950s and later and made evident a dramatic increase, ranging from 4- to 20-fold, in

PBDEs in the 1980s. Analyses for PBDFs were not performed (Nylund et al., 1992).

5.1.3 Soil, sewage sludge, and biocompost

One soil sample taken near a motorway (depth: 0–2 cm) in Germany contained 0.74 µg monoBDFs/kg, 0.58 µg diBDFs/kg, and 1 µg Cl_1Br_1DFs/kg soil. Other PBDFs or PXDFs (up to Br_4/X_4) were not detected, and no PBDDs (Br_1 through Br_4) were found (detection limit: 0.02 µg/kg). PCDFs (Cl_1 through Cl_5) were present at concentrations ranging from 0.04 to 0.38 µg/kg, and PCDDs (Cl_1 through Cl_8) were detected at 0.1 µg/kg (Ballschmiter et al., 1990). Another soil sample (n = 1; no details given) collected near a motorway in Germany also did not contain PBDDs, and the concentrations of PBDFs were low (sum of tetra- and pentaBDFs: 0.02 µg/kg; 2,3,7,8-TeBDF: <0.001 µg/kg). The sum of PCDDs/PCDFs was 0.3 µg/kg (Schacht et al., 1995).

Watanabe et al. (1992) reported the presence of PBDFs (about 100 µg total PBDFs/kg), together with PBDEs and PCDDs/PCDFs, in a soil sample (depth not given) taken from an incineration field in Taiwan that was contaminated with large amounts of ash from plastic materials. They also found a large number of PXDFs but no PXDDs in the same sample.

Further samples from disposal sites are discussed in chapter 3.

Soil samples (n = 2) collected at a metal reclamation factory area in southern Taiwan had total PBDF concentrations of 48 and 87 µg/kg. The PBDFs included triBDFs (16–30 µg/kg), tetraBDFs (16–34 µg/kg), pentaBDFs (11–18 µg/kg), and hexaBDFs (5 µg/kg). PBDDs were not detected (detection limits ranged from <0.25 to 1 µg/kg) (Watanabe et al., 1993).

Soil samples (n = 3; 0–10 cm depth, if necessary after removal of plants and grass; at distances of >1000–<2000 m, main wind direction) were collected a few months after a fire in a warehouse where bromine-containing plastic pellets were stored (Neupert & Pump, 1992). In one of three sites (samples), 2,3,7,8-substituted PBDDs/

PBDFs were found (detection limit 0.5 ng/kg) at 3.5 ng/kg (sum of five tetra- to hexaBDDs and of three tetra- to pentaBDFs).

A series of sewage sludge samples (n = 13) from municipal wastewater treatment plants in Germany were analysed for PBDDs/PXDDs and PBDFs/PXDFs as well as for other polyhalogenated compounds (Hagenmaier et al., 1992). The analytical data on these sludges, which were destined to be discharged on fields for fertilization, are summarized in Tables 32 and 33. PBDFs were detected in

Table 32. PBDFs detected in sewage sludge samples from municipal wastewater treatment plants in Germany[a]

PBDFs	Number of samples	Concentration[b] (µg/kg)
MonoBDFs	9	0.05–0.67
DiBDFs	8	0.27–1.99
TriBDFs	9	0.07–0.20
TetraBDFs	9	0.03–0.23
PentaBDFs	9	n.d.–0.01
Sum of mono- to pentaBDFs		0.29–3.05

[a] Adapted from Hagenmeier et al. (1992).
[b] n.d. = not detected.

Table 33. Concentrations of PBDFs and other polyhalogenated aromatic compounds in sewage sludge[a]

Compounds	Number of samples	Concentration (µg/kg)			
		Range	Median	Mean	Standard deviation
Mono- to pentaBDFs	13	0.21–3.05	1.11	1.17	0.92
Tri- to heptaBDEs	13	0.49–17.73	8.37	8.58	5.51
PCBs	17	233–3456	674	911	767
Total PCDDs	13	3.27–27.82	9.20	10.71	6.89
Total PCDFs	13	0.18–7.09	0.53	1.07	1.83

[a] Adapted from Hagenmaier et al. (1992).

all samples, their concentrations (sum of mono- to pentaBDFs) reaching a maximum of about 3 µg/kg. Of the homologue groups (Table 32), the diBDFs were predominant, with concentrations ranging from 0.27 to 1.99 µg/kg. PBDDs, PXDDs, and PXDFs were not found in any samples. The ratio of median concentrations for PBDFs and PCDDs/PCDFs was 1 : 9 (Table 33). PBDEs (tri- to heptaBDEs), the possible precursors of PBDFs, were detected in all samples at higher levels than PBDFs (Table 33).

Another study group (Ballschmiter et al., 1990) did not find any PBDDs/PBDFs (mono through tetra) in sewage sludge samples from rural (n = 12) and urban (n = 5) areas (detection limit not given).

Traces of tetraBDDs (0.006 µg/kg) and tetra- to heptaBDFs (sum: 0.32 µg/kg; 2,3,7,8-TeBDF: 0.003 µg/kg) together with PCDDs/PCDFs (sum: 8.34 µg/kg) were found in a single sewage sludge sample (Schacht et al., 1995).

Biocompost (n = 1) containing tetra- to octaCDDs/CDFs (sum: 3.2 µg/kg) showed no detectable PBDDs (detection limit not given); of PBDFs (tetra to hepta), only tetraBDFs (<0.003 µg/kg; no 2,3,7,8-TeBDF) were present (Schacht et al., 1995).

5.1.4 Food and feed

Little information was found in the literature. Market basket surveys are not available.

Crops and other vegetation growing in the vicinity of potential emitters can be contaminated, as shown by an analysis of grass collected near a motorway. The pattern and levels of PBDDs/PBDFs and PXDDs/PXDFs found in plants are described in section 5.1.6.1.

The occurrence of PBDDs/PBDFs in seafood and milk is discussed in section 5.1.6.2.

5.1.5 Other products

For levels of PBDDs/PBDFs in consumer products such as electrical appliances, and emissions from them, see chapter 3.

5.1.6 Terrestrial and aquatic organisms

5.1.6.1 Plants

Grass samples collected in the vicinity of a motorway contained lower brominated PBDFs (mono- through triBDFs), PXDFs (up to X_3), and traces of monoBDDs (Ballschmiter et al., 1990; Table 34). Between the homologue groups, concentrations peaked at mono- and diBDFs and Cl_1Br_1DFs (>2000 ng/kg). This pattern paralleled that found in the soil sample from the same sampling site, but the levels in the grass sample were higher (Table 34).

PBDDs/PBDFs (mono through tetra), PXDDs/PXDFs (up to tetra), and PCDDs/PCDFs (mono to octa) have been analysed in needles from a pine near a highway (Schwind, 1991). Again, concentration peaks were seen with the mono- and diBDFs (see Table 34) and with monoCDFs (4380 ng/kg dry weight). The sum of the PHDFs (mono to octa) was about a factor of 100 higher than the sum of PHDDs (owing to the high concentration of mono- to trihalogenated dibenzofurans [>12 500 ng/kg dry weight]).

5.1.6.2 Animals

1) Wildlife

In a randomly selected pooled sample consisting of shrimp (380 g) and mussels (600 g) as well as fish (cod: 1500 g; plaice: 300 g), a tetraBDD and a tetraBDF could be detected, both non-2,3,7,8-substituted (De Jong et al., 1992). A quantitative analysis was not performed.

Neither PBDDs/PBDFs nor PXDDs/PXDFs were found in one homogenate of muscle from Baltic salmon (*Salmo salar*; Sweden) at detection limits of 0.2–20 ng/kg fresh weight (Wiberg et al., 1992).

Table 34. Concentrations of PBDDs/PBDFs and PXDDs/PXDFs in environmental samples taken near motorways in Germany

Homologue groups	Concentration[a] (ng/kg)		
	Grass[b] (after 1 month of dryness)	Soil[b] (depth: 0–2 cm)	Pine needles[c] (dry weight)
Dibenzo-*p*-dioxins			
MonoBDDs	60	n.d.	28.3
DiBDDs	n.d.	n.d.	25.5
TriBDDs	n.d.	n.d.	5.5
TetraBDDs	n.d.	n.d.	<4
Cl_1Br_1DDs	–	–	+
Cl_1Br_2DDs	–	–	5.6
Cl_2Br_1DDs	–	–	1.7
X_4DDs	–	–	<4[d]
Dibenzofurans			
MonoBDFs	2530	740	5491
DiBDFs	2170	580	1053
TriBDFs	40	n.d.	258
TetraBDFs	n.d.	n.d.	53
Cl_1Br_1DFs	2420	1000	788
Cl_1Br_2DFs	240	n.d.	358
Cl_2Br_1DFs	190	n.d.	116
Cl_2Br_2DFs	n.d.	n.d.	58
Cl_3Br_1DFs	n.d.	n.d.	55
Cl_1Br_3DFs	–	–	74

[a] n.d. = not detected (detection limit 20 ng/kg); – = not analysed; + = detectable, but not quantifiable.
[b] Adapted from Ballschmiter et al. (1990).
[c] Adapted from Schwind (1991).
[d] For each of the three possible combinations (Cl_2Br_2, Cl_3Br_1, Cl_1Br_3).

PBDDs/PBDFs were not detected (detection limits of 2, 3, and 8 ng/kg for tetra-, penta-, and hexaBDDs/BDFs, respectively) in pooled muscle samples of young (n = 15), middle-aged (n = 15), and old (n = 15) carp (*Cyprinus carpio*) collected from the Buffalo River, NY, USA (Loganathan et al., 1995). No PBDDs/PBDFs or PXDDs/PXDFs

(tetra to hexa) were detected (detection limit not specified) in fish (n = 4) captured in 1993 in rivers near Osaka in Japan (Watanabe et al., 1995).

One composite sample of muscle from osprey (*Pandion haliaetus*; Sweden; n = 35; cf. Jansson et al., 1993) was analysed for PBDDs/PBDFs and for PXDDs/PXDFs. The results were negative at detection limits of 0.2–20 ng/kg fresh weight (Wiberg et al., 1992).

PCDDs/PCDFs were present in all samples mentioned above. For example, carp from the Buffalo River (Loganathan et al., 1995) were found to contain noticeable concentrations of total PCDDs (27–146 ng/kg wet weight) and total PCDFs (22–99 ng/kg) along with total PCBs (>2 mg/kg) and total PBDEs (13–23 µg/kg).

2) Farm animals

Cow's milk collected at dairy farms in the deposition area of an "old technology" municipal waste incinerator in the Netherlands was analysed for PBDDs/PBDFs and PXDDs/PXDFs. The pooled (n = 11) milk sample contained compounds that were tentatively identified (but not quantified) as two triBDFs, one tetraBDF, and one pentaBDF, all four not having the 2,3,7,8-substitution pattern (De Jong et al., 1992). The same sample contained high levels of PCDDs/PCDFs.

5.2 General population exposure

5.2.1 Exposure data

There is no quantitative information available on exposure of the general population, special subpopulations, or infants to PBDDs/PBDFs from several sources (see chapter 3 and section 5.1).

5.2.2 Monitoring of human tissues and fluids

Few studies have monitored PBDDs/PBDFs in human tissues or milk.

On behalf of the National Human Adipose Tissue Survey, the US EPA initiated a study in 1987 to analyse 2,3,7,8-substituted tetra- through hexaBDDs/BDFs in adipose tissue of the general population. Eight hundred and sixty-five individual tissue specimens were collected and combined into 48 composite samples referring to the nine US census divisions and three age groups. None of the six targeted PBDDs/PBDFs was detected at average detection limits of approximately 1 ng/kg for 2,3,7,8-TeBDD/TeBDF (range: 0.4–8.9 ng/kg), 10 ng/kg for 1,2,3,7,8-PeBDD/PeBDF and for 1,2,3,4,7,8-HxBDD (range: 0.8–54 ng/kg), and 40 ng/kg for 1,2,3,4,7,8-HxBDF (range: 2.5–120 ng/kg) on a lipid weight basis (Cramer et al., 1990a,b). In all samples analysed, there were indications of the presence of PBDEs (hexa through octa) (Cramer et al., 1990a,b).

Neither PBDDs/PBDFs nor PXDDs/PXDFs were found in two human adipose tissue samples (male: 70 years of age; female: 60 years of age) examined in a German study (Dawidowsky, 1993). The detection limits for tetra-, penta-, and hexa-substituted congeners were 0.2–1.6 ng/kg, 0.9–3.5 ng/kg, and 4.6–14 ng/kg on a lipid weight basis, respectively.

One composite sample of human milk (38 g) from Sweden was examined for PBDDs/PBDFs and PXDDs/PXDFs (Wiberg et al., 1992). Results of this study, which did not give any information on the number of original specimens or sampling strategy, were negative. The detection limits were reported to be in the range of 0.2–20 ng/kg on a lipid weight basis. PCDDs/PCDFs were detectable in this sample. Another investigation from Germany (Dawidowsky, 1993) led to similar results. Although PCDDs/PCDFs were present in three human milk samples, PBDDs/PBDFs and PXDDs/PXDFs were not detectable at detection limits of 0.8–5.5 ng/kg (tetra substitution), 3.2–13 ng/kg (penta substitution), and 13–53 ng/kg (hexa substitution) on a lipid weight basis. At present, the presence of PBDDs/PBDFs in human milk at very low concentrations cannot be ruled out (Somogyi & Beck, 1993).

5.3 Occupational exposure

5.3.1 Workplace monitoring data

Contamination by PBDDs/PBDFs is possible in a variety of workplaces involved in producing, processing, using, or disposing of certain flame retardants or products containing them (see chapter 3), especially where the processes involve elevated temperatures. There are only limited workplace monitoring data.

5.3.1.1 Flame retardant/polymer industry

A set of data is available on workplaces in the chemical industry producing flame-retarded polymers (Brenner & Knies, 1990, 1992, 1993a,b, 1994; Thies et al., 1990; Brenner, 1993; Kieper, 1996). Air samples were taken during extrusion production and moulding of PBT, ABS, polystyrene, or polyamide resins blended with various brominated flame retardants (see Table 35).

PBDF concentrations measured near the extruder and injection moulding machines, in the whole building, in the storage and refilling area, and at other sites are summarized in Table 35. Within the homologue groups measured, a maximum concentration of about 600 ng/m^3 was found for the sum of hexaBDFs. In one experimental series, PBDF concentrations in workplace air were higher by a factor of about 1000 where DBDE was used, compared with TBBPA. The difference was explained by the different properties of DBDE and TBBPA, as well as different exhaust and ventilation conditions (Brenner & Knies, 1993a). In the study involving DBDE/PBT, PBDF concentrations near the extruder workplace and in the air of the whole room were similar, indicating their general distribution in the building. Lower levels were found in the storage and refilling area (Brenner & Knies, 1990).

2,3,7,8-TeBDF was not detected in a lot of samples (detection limits 1–100 pg/m^3 air) (Brenner & Knies, 1990, 1993a, 1994; Thies et al., 1990; Kieper, 1996) but was detectable in some others (Kieper, 1996). Low amounts of penta- and hexaBDFs substituted in the 2,3,7,8-position (0.3–2.6 ng/m^3 air) were tentatively identified at DBDE/PBT workplaces (Brenner & Knies, 1990).

Table 35. Air contamination by PBDFs measured at workplaces where flame-retarded thermoplastic resins are produced and processed

PBDFs	Resin/flame retardant[a]	Number of sampling stations	Air volume[b] (m^3)	Concentration[c] (ng/m^3)	Reference
MonoBDFs	PBT/TBBPA	3	13–18	n.d.–0.26 (<0.006)	Kieper (1996)
	PS-PS-butadiene/DBDE	5	20–30	0.01–0.16	Kieper (1996)
	PS/1,2-bis(tribromophenoxy)ethane	3	17–24	0.012–0.017	Kieper (1996)
	PA/polytribromostyrene	5	19–29	0.017–0.049	Kieper (1996)
	PA/polydibromostyrene	5	6–12	0.026–0.129	Kieper (1996)
DiBDFs	PBT-glass fibre/DBDE	3	30–150	0.2–1.3	Brenner & Knies (1990)
	PBT-glass fibre/TBBPA	2	185–260	n.d.–0.34 (<0.004)	Brenner & Knies (1993a)
	PBT/TBBPA	3	13–18	0.19–1.02	Kieper (1996)
	PBT-glass fibre/TBPI	3–4	185–260	up to 0.14	Brenner & Knies (1994)
	ABS/TBBPA	1	4–5	n.d. (<1)	Thies et al. (1990)
	PS-PS-butadiene/DBDE	5	20–30	0.040–0.223	Kieper (1996)
	PS/1,2-bis(tribromophenoxy)ethane	3	17–24	0.043–0.139	Kieper (1996)
	PA/polytribromostyrene	5	19–29	0.070–0.21	Kieper (1996)
	PA/polydibromostyrene	5	6–12	0.094–0.24	Kieper (1996)
TriBDFs	PBT-glass fibre/DBDE	3	30–150	1.1–13	Brenner & Knies (1990)
	PBT-glass fibre/TBBPA	2	185–260	n.d.–0.11 (<0.012)	Brenner & Knies (1993a)
	PBT/TBBPA	3	13–18	0.065–3.04	Kieper (1996)

Table 35 (contd).

	PBT-glass fibre/TBPI	3–4	185–260	up to 0.18	Brenner & Knies (1994)
	ABS/TBBPA	1	4–5	n.d. (<1)	Thies et al. (1990)
	PS-PS-butadiene/DBDE	5	20–30	0.077–0.484	Kieper (1996)
	PS/1,2-bis(tribromophenoxy)ethane	3	17–24	0.063–0.337	Kieper (1996)
	PA/polytribromostyrene	5	19–29	0.049–0.274	Kieper (1996)
	PA/polydibromostyrene	5	6–12	0.075–0.169	Kieper (1996)
TetraBDFs	PBT-glass fibre/DBDE	3	30–150	5.1–34	Brenner & Knies (1990)
	PBT-glass fibre/TBBPA	2	185–260	0.03–0.05	Brenner & Knies (1993a)
	PBT/TBBPA	3	13–18	0.157–6.92	Kieper (1996)
	PBT-glass fibre/TBPI	3–4	185–260	up to 0.14	Brenner & Knies (1994)
	ABS/TBBPA	1	4–5	n.d. (<1)	Thies et al. (1990)
	PS/hexabromocyclododecane	1	n.sp.	0.02	Brenner (1993)
	PS-PS-butadiene/DBDE	5	20–30	0.173–0.40	Kieper (1996)
	PS/1,2-bis(tribromophenoxy)ethane	3	17–24	0.136–0.30	Kieper (1996)
	PA/polytribromostyrene	5	19–29	0.017–0.43	Kieper (1996)
	PA/polydibromostyrene	5	6–12	0.051–0.15	Kieper (1996)
PentaBDFs	PBT-glass fibre/DBDE	3	30–150	8.6–143	Brenner & Knies (1990)
	PBT-glass fibre/TBBPA	2	185–260	0.07–0.19	Brenner & Knies (1993a)
	PBT/TBBPA	3	13–18	0.11–5.63	Kieper (1996)
	PBT-glass fibre/TBPI	3–4	185–260	up to 0.12	Brenner & Knies (1994)
	ABS/TBBPA	1	4–5	n.d. (<1)	Thies et al. (1990)

Table 35 (contd).

PBDFs	Resin/flame retardant[a]	Number of sampling stations	Air volume[b] (m^3)	Concentration[c] (ng/m^3)	Reference
	PS/hexabromocyclododecane	1	n.sp.	1	Brenner (1993)
	PS-PS-butadiene/DBDE	5	20–30	0.27–0.49	Kieper (1996)
	PS/1,2-bis(tribromophenoxy)ethane	3	17–24	0.16–0.18	Kieper (1996)
	PA/polytribromostyrene	5	19–29	0.04–0.30	Kieper (1996)
	PA/polydibromostyrene	5	6–12	0.035–0.12	Kieper (1996)
HexaBDFs	PBT-glass fibre/DBDE	3	30–150	13–594	Brenner & Knies (1990)
	PBT-glass fibre/TBBPA	2	185–260	0.05–0.26	Brenner & Knies (1993a)
	PBT/TBBPA	3	13–18	0.06–3.61	Kieper (1996)
	PBT-glass fibre/TBPI	3–4	185–260	up to 0.11	Brenner & Knies (1994)
	ABS/TBBPA	1	4–5	n.d. (<1)	Thies et al. (1990)
	PS-PS-butadiene/DBDE	5	20–30	0.81–6.37	Kieper (1996)
	PS/1,2-bis(tribromophenoxy)ethane	3	17–24	0.33–0.73	Kieper (1996)
	PA/polytribromostyrene	5	19–29	0.06–0.29	Kieper (1996)
	PA/polydibromostyrene	5	6–12	0.028–0.099	Kieper (1996)
HeptaBDFs	PBT-glass fibre/DBDE	3	30–150	~88–260	Brenner & Knies (1990)
	PBT-glass fibre/TBBPA	2	185–260	n.d.–<0.04 (<0.013)	Brenner & Knies (1993a)
	PA/polytribromostyrene	5	19–29	0.07–0.30	Kieper (1996)
	PA/polydibromostyrene	5	6–12	0.04–0.07	Kieper (1996)

Table 35 (contd).

OctaBDF	PBT-glass fibre/DBDE	3	30–150	n.d.–~7	Brenner & Knies (1990)
	PBT-glass fibre/TBBPA	2	185–260	n.d.–<0.08 (0.026)	Brenner & Knies (1993a)
	PA/polytribromostyrene	5	19–29	0.03–0.20	Kieper (1996)
	PA/polydibromostyrene	5	6–12	<0.004–0.09	Kieper (1996)

[a] PA = polyamide 66; PS = polystyrene; TBBPA = tetrabromobisphenol A or its derivatives.
[b] n.sp. = not specified.
[c] n.d. = not detected (detection limits in parentheses, if specified).

Concentrations of PBDDs in the DBDE/PBT study were two orders of magnitude below those of PBDFs, with di-, tri-, tetra-, penta-, and hexaBDD concentrations being <0.05, 0.35, 2.04, 8.37, and 17 ng/m^3 air, respectively (Brenner & Knies, 1990). The tentative analysis for 2,3,7,8-substituted PBDDs showed the presence of 2,3,7,8-TeBDD (<0.5 ng/m^3 air), of 1,2,3,7,8-PeBDD (1.3 ng/m^3 air), and of two hexaBDDs (1 and 1.6 ng/m^3) (Brenner & Knies, 1990). In contrast, no PBDDs were found in air samples of the workplace area in a TBBPA study, although low amounts of di- (0.94 ng/m^3), tri- (0.07 ng/m^3), and tetraBDDs (0.08 ng/m^3) were emitted by the extruder equipment. The detection limits of di- through octaBDDs ranged from 0.001 to 0.4 ng/m^3 air. No 2,3,7,8-substituted PBDDs (tetra- through hexaBDDs) were found (detection limits ranged from 0.001 to 0.1 ng/m^3 air) (Brenner & Knies, 1993a). Preliminary tests at the workplace during injection moulding of polystyrene blended with hexabromocyclododecane showed tetraBDD concentrations of 2.5 ng/m^3, consisting of two isomers with no 2,3,7,8-substitution (Brenner, 1993). During extruder experiments with PBT/TBPI (Brenner & Knies, 1994), the sum concentration of PBDDs was in the low pg/m^3 range, and no 2,3,7,8-substituted congeners were detected (detection limits ranged from 1 to 240 pg/m^3; tetra to octa).

Air samples from selected workplaces (operated permanently or periodically) of three plastic processing plants were monitored for mono- to hexaBDFs/BDDs (resin/flame retardant used: polystyrene-polystyrene-butadiene/DBDE; polystyrene/1,2-bis(tribromophenoxy)-ethane; PBT/TBBPA-carbonate oligomer) and for mono- to octaBDFs/BDDs (resin/flame retardant used: polyamide 66/polytribromostyrene; polyamide 66/polydibromostyrene), including eight or more 2,3,7,8-substituted congeners (Kieper, 1996). Depending on workplace and flame retardant used, the sum of PBDF/PBDD concentrations (mono to hexa) ranged from 258 to 77 414 pg/m^3. The highest sum concentrations within workplaces permanently operated (range: 258–10 018 pg/m^3) were found at workplaces processing 1,2-bis(tribromophenoxy)ethane. Although samples from the latter did not contain any 2,3,7,8-substituted congeners, many other samples gave positive results, with maximum sum concentrations of some thousand pg/m^3. The highest concentrations of 2,3,7,8-substituted PBDFs/PBDDs were seen at the TBBPA-carbonate oligomer/PBT workplaces (operated

permanently and periodically). The ranges of concentrations (n = 3) in pg/m³ were as follows:

2,3,7,8-TeBDF:	<4–<165	(detection limit elevated owing to interfering components)
1,2,3,7,8-PeBDF:	2–100	
2,3,4,7,8-PeBDF:	6–313	
1,2,3,4,7,8-HxBDF:	7–445	
2,3,7,8-TeBDD:	<6–<293	(detection limit elevated owing to interfering components)
1,2,3,7,8-PeBDD:	23–1137	
1,2,3,4,7,8-/1,2,3,6,7,8-HxBDD:	25–1161	
1,2,3,7,8,9-HxBDD:	9–578	

Results for 2,3,7,8-substituted heptaBDFs/BDDs were obtained at workplaces processing polyamide flame-retarded by polytribromostyrene (n = 5) or polydibromostyrene (n = 5). The concentrations measured (in pg/m³) ranged in the following manner (n = 10):

1,2,3,4,6,7,8-HpBDF:	26–280
1,2,3,4,7,8,9-HpBDF:	<1–13
1,2,3,4,6,7,8-HpBDD:	<1.4–11

5.3.1.2 *Offices/studios*

Some monitoring results are available for workplaces (offices; television studios) equipped with a number of electrical appliances continually in use, such as display and computer monitors (Chriske et al., 1990; UBA, 1992; see also section 5.1.1). Maximum air concentrations measured were 0.56 pg/m³, 0.61 pg/m³, and 0.4 pg/m³ for tetra-, penta-, and hexaBDFs (see Table 28). Although heptaBDFs could not be detected in air samples (Table 28), they were found in the corresponding dust samples at concentrations ranging from 0.7 to 3.5 µg/kg (Table 30). Similarly, no detectable amounts of tetra- and pentaBDFs substituted in the 2,3,7,8-position were found in air samples (Table 28), but they were found in dust samples, with concentrations ranging from n.d. to 0.005 µg/kg (2,3,7,8-TeBDF) and

from 0.003 to 0.020 µg/kg (two pentaBDFs) (Table 30). PBDDs were not detected (Chriske et al., 1990).

5.3.1.3 *Recycling plants*

Recycling of plastic materials (pure or in combination with other materials, e.g. metals, that can have catalytic effects) may be a source of PHDDs/PHDFs, depending — *inter alia* — on the type of flame retardants blended with them (see chapter 3). Workplace air was monitored in a pilot plant recycling defective printed circuits (Lorenz & Bahadir, 1993). These printed circuits contained copper and TBBPA, a flame retardant with a relatively low potency for generation of PBDDs/PBDFs. Air samples (n = 2) taken near the running shredding systems did not contain PBDFs, although small amounts of mono- and diBDFs were found in the shredded material (0.05–0.32 µg/kg). However, owing to short sampling time and resulting low air volumes sampled (6–7 m^3), the detection limits were only 0.02–0.1 ng/m^3 (mono- through pentaBDFs) and 0.2–0.4 ng/m^3 (hexa- through heptaBDFs). No PBDDs (mono- through octaBDDs) were found in the air samples (detection limits 0.02–2 ng/m^3 for mono- through heptaBDDs), but residues of tetraBDDs (0.03–0.73 µg/kg) were present in samples of the processed waste. Neither 2,3,7,8-TeBDD nor 2,3,7,8-TeBDF was found in any of the samples tested (Lorenz & Bahadir, 1993).

In 1991, air samples (air volume: 20–30 m^3; sampling period: 6–10 h) from three workplace stations in an operating secondary copper plant and ground dust samples (pooled sample from five sites) collected in another secondary copper plant (shut down in 1990) in Germany were monitored for PBDDs/PBDFs (Kieper, 1996). The sum of mono- to hexaBDF concentrations in the workplace air samples ranged from 8 to 190 pg/m^3. 2,3,7,8-TeBDF was present in one sample at 0.4 pg/m^3 but was not detected in either of the other air samples (detection limits 0.1–0.8 pg/m^3). PBDDs (mono to hexa) were not detectable (detection limits 0.1–1.4 pg/m^3). The dust sample contained 21.02 µg mono- to hexaBDFs/kg, with a maximum of 8.4 µg/kg for tetraBDFs. Concentrations of 2,3,7,8-TeBDF, 1,2,3,7,8-PeBDF, 2,3,4,7,8-PeBDF, and 1,2,3,4,7,8-HxBDF were 0.09, 0.10, 0.12, and

0.09 µg/kg (maximum values), respectively. PBDDs (mono to hexa) were not found (detection limits 0.01–0.03 µg/kg).

5.3.1.4 *Other workplaces*

Monitoring data from workplaces in waste incineration facilities and disposal sites are lacking.

PBDF concentrations of up to 100 mg/kg (see section 3.7.2) were found in combustion residues of accidental fires; therefore, firemen and other workers coming in contact with fire fume, dust, and residues could be exposed to these substances.

The work area under the fume hood (waste oil from air pumps) of a laboratory was found to be contaminated with PBDDs/PBDFs and PCDDs/PCDFs (Ritterbusch et al., 1994b). Wipe tests showed a sum concentration of 580 ng/m^2 for PBDDs/PBDFs (mono to hexa) and of 360 ng/m^2 for PCDDs/PCDFs (tetra to octa).

5.3.2 *Monitoring of human tissues and fluids*

A chemist who suffered from acute intoxication after synthesizing several grams of 2,3,7,8-TeBDD and 2,3,7,8-TeCDD in 1956 without using a hood or protective clothing (Schecter & Ryan, 1990, 1991, 1992; Schecter, 1992; see also section 8.2) was examined 35 years after exposure. His blood contained 625 ng TBDD/kg blood lipid (concomitant with 18 ng TCDD/kg blood lipid) (Schecter, 1992; Schecter & Ryan, 1992). For comparison, the average blood lipid concentration in the general population of the USA was reported as 3–5 ng/kg for TCDD and not detectable for TBDD (Schecter et al., 1994a). (Blood analysis of a Japanese student who had developed chloracne [see section 8.2] 1 month after exposure to PCDFs and PBDFs showed no detectable amounts of PCDFs [or PCBs]. PBDFs were not included in this analysis, performed in 1982, about half a year after exposure [Asahi & Urabe, 1987].)

Zober et al. (1992) investigated employees of a chemical plant that had produced thermoplastic resins (PBT) blended with the flame retardants OBDE or DBDE. The potential for exposure to PBDDs/

PBDFs including the homologue groups di- through octaBDDs/BDFs was established by workplace air measurements, as described by Brenner & Knies (1990) (see also section 5.3.1). However, analysis of the blood lipids of the personnel focused mainly on 2,3,7,8-TeBDD/TeBDF. Elevated levels of both congeners were found in the blood samples of potentially exposed workers, although both isomers could not (TBDF) or could hardly (TBDD) be identified in the previous workplace air samples (see section 5.3.1). The concentrations of TBDD in venous blood exceeded those of TBDF (ranges: n.d.–478 ng/kg blood lipid versus n.d.–112 ng/kg blood lipid) among the male study group (see Table 36). As seen in Table 36, there was a correlation between blood levels of TBDF/TBDD measured and job type, working conditions, and working period. Highest median (TBDF/TBDD: 18/91 ng/kg blood lipid) and maximum (TBDF/TBDD: 112/478 ng/kg blood lipid) values were found in the 18 extruder operators who had been first engaged before 1986. The lower levels observed in the 11 operators employed during and after 1986 may be due to the shorter exposure time, changes in production process, or technical improvements, as suggested by the authors of this study. Other long-term employees (n = 5) showed intermediate blood levels ranging from <7 to 26 ng TBDF/kg blood lipid and from 7 to 48 ng TBDD/kg blood lipid. The lowest values were seen in the technical support personnel (see Table 36). Data from a referent group were not provided, but among referents (n = 5) of a preceding study, PBDFs/PBDDs were either not detected or marginally present (Zober et al., 1992).

Table 36. Concentrations of 2,3,7,8-TeBDF and 2,3,7,8-TeBDD in blood of personnel from industry using PBDE[a]

Job type/first year on job	Number	Concentration (ng/kg blood lipid)[b,c]					
		2,3,7,8-TeBDF			2,3,7,8-TeBDD		
		Median	Minimum	Maximum	Median	Minimum	Maximum
Extruder operators	29	8	n.d.	112	40	n.d.	478
1975–1985	18	18	n.d.	112	91	16	478
1986–1988	11	4	n.d.	11	n.d.	n.d.	11
Maintenance mechanics (1975–1983)	3	16	<7	26	17	17	22
Production employees; other areas (1976–1982)	2	7	7	7	28	7	48
Technical support personnel	8	2	n.d.	11	n.d.	n.d.	5

[a] From Zober et al. (1992).
[b] Samples collected in 1990/91.
[c] n.d. = not detected (detection limit not specified).

6. KINETICS AND METABOLISM

6.1 Absorption

6.1.1 *Dibenzo-p-dioxins*

All studies available on absorption of PBDDs refer to absorption of 2,3,7,8-TeBDD in rats (see Table 37). It was absorbed after oral, dermal, and intratracheal administration, the percent absorption varying with route and dose.

After administration of a single dose of 1 nmol/kg body weight, absorption was about 80% by the oral and intratracheal routes, whereas only about 12% was absorbed through the skin. Oral absorption of single doses declined from 80% at lower doses (1–10 nmol/kg body weight) to about 50% at higher doses (500 nmol/kg body weight), thus suggesting non-linear absorption.

Oral and pulmonary absorption of an equimolar dose of 2,3,7,8-TeCDD using identical experimental conditions as with TBDD were 88% and 95%, respectively (Diliberto et al., 1996). Dermal absorption of TBDD was about one-third that of an equimolar dose of TCDD (Jackson et al., 1991; Diliberto et al., 1993). Differences between TBDD and TCDD pulmonary and dermal absorption may be explained by the octanol/water partition coefficients and the size of the halogen substituents.

As enteral absorption for many PHDDs/PHDFs is known to be variable and incomplete (especially demonstrated for the higher chlorinated congeners), subcutaneous application has been used in several of the relevant studies. For the majority of the chlorinated congeners, a significant degree of absorption was reported within a few days (exception: octachlorodibenzo-*p*-dioxin, or OCDD). A 99% absorption rate has been reported by this route for TBDD (600 ng/kg body weight) as well as for TCDD (300 ng/kg body weight) in the rat (Nagao et al., 1995/96).

Table 37. Absorption of 2,3,7,8-TeBDD in rats

Strain (sex)	Route (vehicle)	Dosing regimen	Absorption,[a] time	Method	References
Wistar (female, male) (n = 4)	oral (arachis oil with 5% toluene)	single dose 100 µg/kg body weight (= 0.2 µmol/kg body weight)	80% (male), 48 h 83% (female), 48 h	faeces analysis [TBDD]	Ivens et al. (1992)
Fischer 344 (male) (n = 3–4)	oral (water : ethanol : Emulphor® = 3 : 1 : 1)	single dose 0.5 µg/kg body weight (= 0.001 µmol/kg body weight)	78%, 72 h	faeces and tissue analysis [^{3}H-TBDD]	Diliberto et al. (1990a,b, 1993); Kedderis et al. (1992a)
		5 µg/kg body weight (= 0.01 µmol/kg body weight)	82%, 72 h		
		50 µg/kg body weight (= 0.1 µmol/kg body weight)	60%, 72 h		
		250 µg/kg body weight (= 0.5 µmol/kg body weight)	47%, 72 h		
Fischer 344 (male) (n = 3- 4)	intratracheal (water : ethanol : Emulphor® = 3 : 1 : 1)	single dose 0.5 µg/kg body weight (= 0.001 µmol/kg body weight)	80%, 72 h	faeces and tissue analysis [^{3}H-TBDD]	Diliberto et al. (1991, 1993); Kedderis et al. (1992a)
Fischer 344 (male) (n = 3–4)	dermal (acetone)	single dose 0.5 µg/kg body weight (= 0.001 µmol/kg body weight) (= 0.2 nmol/1.8 cm^2)	12%, 72 h	faeces and tissue analysis [^{3}H-TBDD]	Jackson et al. (1991); Kedderis et al. (1992a); Diliberto et al. (1993)

[a] Values based on concentration of 2,3,7,8-TeBDD [TBDD] or on ^{3}H activity ([^{3}H-TBDD]).

6.1.2 Dibenzofurans

Dermal absorption of 1,2,7,8-TeBDF was examined in male Fischer 344 rats following a single dose of 1 nmol/kg body weight (Kedderis et al., 1994). About 29% of the administered dose was absorbed, quantified on the basis of the amount found in tissues (4%, excluding the skin site) and excreted within 72 h. This dermal absorption was intermediate, compared with that of TBDD (12%: see section 6.1.1), TCDD (41%: Banks & Birnbaum, 1991), and TCDF (48%: Brewster et al., 1989) after single equimolar doses.

6.2 Distribution

6.2.1 Levels in organs and blood

6.2.1.1 Dibenzo-p-dioxins

Almost all studies available on PBDD distribution refer to disposition of 2,3,7,8-TeBDD in the rat. As shown in Table 38, TBDD is distributed throughout the whole body, with major deposits found in liver and adipose tissue, followed by skin and muscle (Kedderis et al., 1991a; Diliberto et al., 1993). Appreciable amounts of [^{3}H]TBDD were also found in adrenals and thymus (Diliberto et al., 1993). Three days after oral exposure to [^{3}H]TBDD, liver and adipose tissue contained more than 65% of the body burden (Diliberto et al., 1993). Distribution of TBDD can be described by a physiologically based pharmacokinetic model consisting of a blood compartment and five tissue compartments: liver, fat, skin, slowly perfused tissues, and richly perfused tissues (Kedderis et al., 1992b, 1993; Buckley, 1995).

The partitioning of TBDD between liver and adipose tissue was studied in Fischer 344 rats exposed to[^{3}H]TBDD (Diliberto et al., 1990a,b, 1991, 1993; Kedderis et al., 1990, 1991a,b, 1992a, 1993; Jackson et al., 1991) and found to be influenced by dose, route of exposure, and time post-dosing (see Table 39 for representative data).

Table 38. Distribution of TBDD-derived radioactivity in Fischer 344 rats 3 days after oral, dermal, or intratracheal administration of 1 nmol [^{3}H]TBDD/kg body weight[a,b]

Tissue	% administered dose[c,d]			% absorbed dose/g tissue[d,e]		
	Oral	Dermal	Intratracheal	Oral	Dermal	Intratracheal
Liver	20.3	2.4	19.5	2.4	2.4	0.3
Adipose tissue	19.6	3.8	24.7	0.8	1.6	1.2
Skin	10.9	1.8	8.3	0.3	0.5	0.3
Muscle	3.5	0.8	3.0	0.04	0.07	0.04
Blood	0.4	0.06	0.2	0.03	0.03	0.01
Thymus	0.03[f]	0.03	0.08	0.2	1.1	0.4
Adrenals	0.4[f]	0.01	0.02	0.5	1.2	0.4
Kidneys	–	0.05	0.1	–	0.2	0.1
Spleen	–	0.01	0.02	–	0.2	0.06
Lungs	–	0.06	0.1	–	0.5	0.2
Heart	–	0.02	0.03	–	0.2	0.06
Testes	–	0.02	0.05	–	0.06	0.03
Brain	–	0.01	0.02	–	0.05	0.02
Stomach	–	0.03	0.1	–	0.3	0.2
Small intestines	–	0.04	0.2	–	0.2	0.1
Large intestines	–	0.04	0.2	–	0.3	0.1

[a] Adapted from Diliberto et al. (1993); oral absorption = 79%; dermal absorption = 12%; intratracheal absorption = 78%.
[b] Mean values; n = 3–4; standard deviation and statistical details omitted.
[c] Percentage of the administered dose normalized to 100% recovery.
[d] – = not analysed.
[e] Percentage adjusted to 100% absorption.
[f] n = 1.

Dose-dependent changes in partition ratios were seen in the intravenous and oral studies (see Table 39). Liver concentrations of TBDD were disproportionately increased at the higher doses compared with the lower dose of 1 nmol/kg body weight (liver : fat concentration ratios: 2.6 and 0.2 by the intravenous route at high and low dose, respectively, and >5 and 2.9 by the oral route). However, whereas liver concentrations of TBDD were disproportionately increased at 10 nmol/kg body weight compared with the 1 nmol/kg body weight oral dose, the increase was related to dose in the 10–100 nmol/kg body

Table 39. Partition of [^{3}H]TBDD-derived radioactivity between liver and adipose tissue of rats[a,b]

Route of exposure	Dose[c] (nmol/kg body weight)	Observation period (days)	TBDD concentration (pmol/g)		Liver : fat concentration ratio
			Liver	Fat	
Intravenous	1	3	8.1	2.4	3.4
	1	56	0.2	1.1	0.2
	100	56	117.2	45.3	2.6
Oral	1	3	4.9	1.7	2.9
	10	3	79.9	13.6	5.9
	100	3	518.3	93.4	5.6
	500	3	2216.3	340.1	6.5
Intratracheal	1	3	4.1	2.1	2.0
Dermal	1	3	0.6	0.4	1.5

[a] Adapted from Kedderis et al. (1992a); Diliberto et al. (1993).
[b] Male Fischer 344 rats; n = 3–4; single doses; vehicle: ethanol : Emulphor® : water = 1 : 1 : 3 (oral, intravenous, intratracheal exposure), acetone (dermal exposure).
[c] 1 nmol/kg body weight corresponds to 0.5 µg/kg body weight.

weight dose range. Factors influencing the dose-dependent nature of TBDD tissue distribution are discussed by Kedderis et al. (1993).

Intravenous, oral, intratracheal, and dermal treatment with 1 nmol [^{3}H]TBDD/kg body weight resulted 3 days later in liver : fat concentration ratios of 3.4, 2.9, 2.0, and 1.5, respectively (see Table 39). The lower ratio observed for the dermal exposure is explained by differences in absorbed dose (low internal exposure; see section 6.1) and dose-related tissue distribution.

Time-dependent changes in the distribution pattern were demonstrated in the intravenous study. Partition ratios (liver : fat) of 3.4 and 0.2, respectively, observed 3 and 56 days after single intravenous exposure, indicated an increased distribution in adipose tissue with increasing time after dosing (Table 39). In addition to redistribution, tissue-specific elimination may also be occurring.

Other studies using non-labelled TBDD (vehicle: arachis oil with 5% toluene) and another rat strain (Wistar) also found higher concentrations of TBDD in the liver than in adipose tissue 2 days after single oral doses (Neupert et al., 1989; Ivens et al., 1992) and after daily oral exposure for 91 days (Ivens et al., 1990, 1993).

Both TCDD and TBDD appeared to be distributed in a similar manner, and differences (e.g. after dermal exposure) can be attributed to the higher lipophilic nature of TBDD (Diliberto et al., 1993; Kedderis et al., 1993). Concerning distribution between adipose tissue and blood, a 2.7-fold higher fat/blood partition coefficient was assumed for TBDD compared with TCDD (Kedderis et al., 1993).

Tissue concentrations and concentration ratios (liver : adipose tissue) have been compared under identical experimental conditions for TBDD (single subcutaneous injection, Wistar rats, 600 ng/kg body weight) and TCDD (300 ng/kg body weight) (Nagao et al., 1995/96). As shown in Table 40, the liver : adipose tissue concentration ratio increases with increasing doses for both congeners. In contrast, the concentration ratios for TCDD/TBDD were rather dose-independent in adipose tissue and also in the liver. Whereas hepatic tissue concentrations were very similar at the doses used for TBDD and TCDD, concentrations in adipose tissue were higher for TBDD over the entire dose range. When increasing the dose 100-fold (from 30 to 3000 ng/kg body weight), hepatic concentrations increased 174 times for TCDD and 256 times for TBDD. In contrast, concentrations in adipose tissue increased only 26 times (TCDD) and 21 times (TBDD).

When the two corresponding chlorinated and brominated 1,2,3,7,8-pentahalogenated dibenzo-*p*-dioxins (PeHDDs) were given as a mixture (2 nmol/kg body weight, each) subcutaneously to Wistar rats, the same tissue distribution (liver : adipose tissue concentration ratio of about 7) was found for both congeners at the maximal tissue concentrations (Golor et al., 1993).

Following administration of 2,3,7-trihalogenated dibenzo-*p*-dioxins (TrHDDs: Cl_3DD, Br_3DD, Cl_2BrDD) to female Wistar rats (single intravenous injections of mixtures containing 3, 10, or 50 µg/kg body weight for each congener), dose- and time-dependent

Table 40. Comparison of tissue concentration and of liver : adipose tissue concentration ratio after a single subcutaneous injection of 2,3,7,8-TeBDD or 2,3,7,8-TeCDD in rats[a,b]

Dose (ng/kg body weight)	Liver tissue			Adipose tissue			Liver : adipose tissue concentration ratio	
	TCDD (ng/g)	TBDD (ng/g)	TCDD : TBDD concentration ratio	TCDD (ng/g)	TBDD (ng/g)	TCDD : TBDD concentration ratio	TCDD	TBDD
30	0.16	0.08	2.1	0.14	0.6	0.2	1.2	0.2
300	3.38	3.60	0.9	0.82	2.7	0.3	4.1	1.4
3000	27.9	20.5	1.4	3.7	12.5	0.3	7.7	1.9
Increase: 30–3000	174 x	256 x		26 x	21 x			

[a] Adapted from Nagao et al. (1995/96).
[b] Female Wistar rats; n = 3 or 6; single subcutaneous doses; vehicle: toluene/DMSO (dimethyl sulfoxide); observation: day 7 after treatment.

changes in tissue concentrations (liver, adipose tissue, thymus) were seen. It is remarkable that concentrations of all three congeners were highest in adipose tissue and lowest in liver (about two orders of magnitude lower a few hours after the injection). Surprisingly high concentrations were found in the thymus, almost an order of magnitude higher than in liver (Golor et al., 1995).

6.2.1.2 *Dibenzofurans*

Disposition studies on dibenzofurans were conducted in rats using [4,6-^{3}H]-1,2,7,8-TeBDF (Kedderis et al., 1994). As with TBDD, the major tissue depots included liver, adipose tissue, and skin (1–72 h after single intravenous, oral, and dermal doses of 1 nmol/kg body weight). Relatively high concentrations of 1,2,7,8-TeBDF were also observed in the adrenal glands. Generally, concentrations in the liver exceeded those in fat, and perirenal fat contained higher amounts than epididymal fat. For example, 72 h following oral administration, there was a liver : fat concentration ratio of about 2. A decline in liver concentrations of 1,2,7,8-TeBDF was seen from 1 to 24 h after intravenous treatment. This was related to metabolic elimination and to a slight accumulation in adipose tissue.

Tissue levels of 1,2,7,8-TeBDF in lungs, small intestine, heart, stomach, spleen, and thymus of Fischer 344 rats 1 h after intravenous dosing were in the range of 0.2–1.3% of the dose/g of tissue. The corresponding results for liver, kidneys, perirenal fat, adrenals, and skin were 4.9, 0.5, 0.1, 5.1 and 0.1%, respectively. The tissue levels of 1,2,7,8-TeBDF 72 h after oral administration were 0.11% (liver), 0.07% (perirenal fat), 0.10% (adrenals), and 0.03% (skin) of the dose/g of tissue (Kedderis et al., 1994).

Seven days after a single subcutaneous dose of 2,3,4,7,8-PeBDF (420 ng/kg body weight) was given to marmoset monkeys (n = 3), a liver : fat concentration ratio of 12.2 was observed (Schulz et al., 1993). A similar high deposition rate in liver was found with the chlorinated analogue, whereas TCDD showed a liver : fat concentration ratio of about 1.

6.2.2 Transfer to offspring

There are no experimental data available on transfer of PBDDs/PBDFs to offspring.

However, transfer of various PCDDs/PCDFs via placenta and/or through milk has been documented in rats, mice, goats, cows (WHO, 1989), marmoset monkeys (Hagenmaier et al., 1990b), and humans (Schecter & Ryan, 1994; Schecter et al., 1994b, 1995, 1996a,b). The bioavailability of PCDDs/PCDFs from breast milk was found to be high (up to >95%) in human infants (Jödicke et al., 1992; McLachlan, 1993; Pluim et al., 1993).

6.3 Metabolic transformation

6.3.1 Dibenzo-p-dioxins

Metabolism of PBDDs has been studied in rats given 2,3,7,8-TeBDD orally or intravenously. No metabolites were found in liver (Kedderis et al., 1991a), but metabolites were detected in bile from rats (male Fischer 344 or female Sprague-Dawley) (Kedderis et al., 1991a; De Jongh et al., 1992, 1993).

Three days after an intravenous dose of 1 nmol [^{3}H]TBDD/kg body weight, faeces of F344 rats (see also section 6.4) contained about 3% of the administered dose as parent compound and about 14% of the dose as metabolites (Kedderis et al., 1991a). About 80–90% of faecal and biliary radioactivity excreted following intravenous dosing was attributed to TBDD metabolites (Kedderis et al., 1991a,b).

If biliary excretion of TBDD-derived radioactivity is considered as an indirect assessment of metabolism, TBDD is relatively slowly metabolized. Within a 5-h period, 6.6% of a radiolabelled intravenous dose (1nmol [^{3}H]TBDD/kg body weight) was excreted in bile of male Fischer 344 rats (Kedderis et al., 1991b).

Studies with pretreated or untreated rats showed that TBDD and TCDD did not induce their own metabolism *in vivo* (Kedderis et al., 1991b, 1992a).

The main metabolites identified (three hydroxybromodibenzo-*p*-dioxins and one dihydroxytetrabromoether) were formed by aromatic hydroxylation and hydrolytic debromination and suggest the metabolic pathway shown in Fig. 2 (De Jongh et al., 1993). Similarities and differences in the metabolic pathways of TBDD and TCDD are discussed by De Jongh et al. (1993).

In summary, several of the metabolites are similar. Quantitatively, the dioxin ring-opening route seems to be favoured somewhat more in TCDD (Poiger & Buser, 1984) than in TBDD metabolism; qualitatively, the absence of a second methoxytribromodibenzodioxin differs from TCDD metabolism.

6.3.2 *Dibenzofurans*

Information on the metabolism of [^{3}H]1,2,7,8-TeBDF is available from a study determining biliary elimination of [^{3}H]1,2,7,8-TeBDF-derived radioactivity (Kedderis et al., 1994). Approximately 50% of the administered dose of [^{3}H]1,2,7,8-TeBDF was excreted in the bile of rats in 8 h. HPLC analysis confirmed the presence of metabolites of 1,2,7,8-TeBDF in the bile (Kedderis et al., 1994). If biliary excretion of PHDDs/PHDFs is used as an indirect measure of metabolism, as is assumed by several authors (Kedderis et al., 1991b; McKinley et al., 1993), this result is indicative of a considerable metabolism of 1,2,7,8-TeBDF, in contrast to 2,3,7,8-TeBDD, which was more slowly metabolized (see above). The differences between both congeners can be explained by their different structures, 2,3,7,8- versus 1,2,7,8-substitution, the latter being more susceptible to metabolism owing to the presence of two adjacent unsubstituted carbon atoms in the 1,2-bromine ring (Kedderis et al., 1994).

6.4 Elimination and excretion

Elimination of PHDDs/PHDFs occurs predominantly after conversion to more polar metabolites in the liver and excretion of these metabolites via the bile. There is apparently no or very little excretion of unchanged congeners with the bile. However, for several of the chlorinated congeners, secretion of the unchanged substances

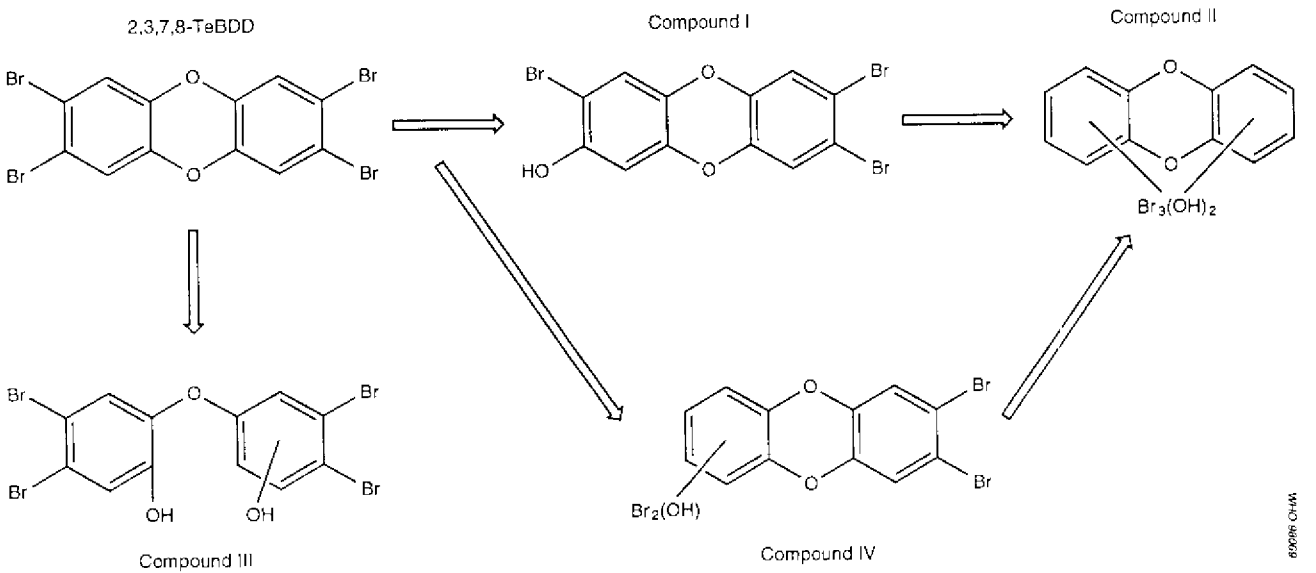

Fig. 2. Proposed biotransformation routes of 2,3,7,8-TeBDD in rat.
(From: De Jongh et al., 1993)

into the intestinal lumen and subsequent excretion via the faeces have been described (Abraham et al., 1989). It can be expected that the lipophilic PBDDs/PBDFs may also be secreted into the intestinal lumen.

6.4.1 Dibenzo-p-dioxins

Elimination was studied in rats with TBDD and with 1,2,3,7,8-PeBDD (faecal excretion was monitored in rats only with TBDD). The animals were exposed to single doses of TBDD or [1,6-^{3}H]TBDD (see Table 41) by the oral (Diliberto et al., 1990a,b, 1993; Ivens et al., 1992), intravenous (Kedderis et al., 1991a), intratracheal (Diliberto et al., 1991, 1993), or dermal (Jackson et al., 1991; Diliberto et al., 1993) route or to subcutaneous doses of 1,2,3,7,8-PeBDD (Golor et al., 1993).

In all studies, the major route of elimination was through the faeces (see Table 41). Eliminated radioactivity (after 2–3 days) in faeces ranged from 2 to 42% of the administered dose of 1 nmol [^{3}H]TBDD/kg body weight and from 0.2 to 1% in urine. Unabsorbed material and biliary excretion appeared to be the major source of eliminated compound in faeces.

Based on the intravenous and oral studies (Table 41), the cumulative elimination of radioactivity in faeces was dose-dependent. Higher doses tended to result in higher elimination rates. The dose-related differences following oral administration were a consequence of differences in amounts eliminated on days 1 and 2 in each group and are likely due to differences in percent absorption (maybe due to the competing processes of uptake versus transit or to limited aqueous solubility of TBDD at high doses). In the 56-day intravenous study, a disproportionately greater elimination of radioactivity at the high (100 nmol/kg body weight) versus the low (1 nmol/kg body weight) dose was observed beginning 3 weeks after treatment (Kedderis et al., 1991a, 1992a).

Faecal elimination curves of the intravenous study were analysed and found to be tri-exponential for the low dose (1 nmol/kg body weight) and bi-exponential for the high dose (100 nmol/kg body

Table 41. Elimination of 2,3,7,8-TeBDD in rats after single radiolabelled and unlabelled doses[a]

Strain (sex)	Route (vehicle)	Dose		Observation period (days)	Test	Elimination[b] (% of administered dose)		Reference
		nmol/kg body weight	µg/kg body weight			Faeces	Urine	
Fischer 344 (male) (n = 3–4)	intravenous (water : ethanol : Emulphor® = 3 : 1 : 1)	1	0.5	1	R	8–10	n.sp.	Kedderis et al. (1991a)
		1	0.5	56		50	4.5	
		100	50	56		70	7.6	
Fischer 344 (male) (n = 3–4)	oral (water : ethanol : Emulphor® =3 : 1 : 1)	1	0.5	3	R	42 ± 2	0.3	Diliberto et al. (1993)
		10	5	3		39 ± 1	0.3	
		100	50	3		58 ± 5	0.2	
		500	250	3		72 ± 5	0.2	
Fischer 344 (male) (n = 3–4)	intratracheal (water : ethanol : Emulphor® = 3 : 1 : 1)	1	0.5	3	R	41 ± 2	1	Diliberto et al. (1993)
Fischer 344 (male) (n = 3–4)	dermal (acetone)	1	0.5	3	R	2	0.2	Diliberto et al. (1993)
Wistar (female, male) (n = 5)	oral (arachis oil with 5% toluene)	200	100	2	U	20 (male) 17 (female)	n.sp.	Ivens et al. (1992)
				3–7		1	n.sp.	

[a] R = administration of [1,6-^{3}H]-2,3,7,8-TeBDD (purity = >98%); elimination refers to eliminated radioactivity. U = administration of unlabelled 2,3,7,8-TeBDD (purity = 98%); elimination refers to recovery of TBDD.

[b] n.sp. = not specified.

weight), with estimated initial and terminal half-lives of <1 and 18 days, respectively (see Table 45 in section 6.5.1).

From the oral study (Table 41), it was shown that 0.3% or less of the administered dose was eliminated in the urine for all dose groups. However, the relative amounts of urinary elimination were found to be higher at the two low doses, consistent with enhanced absorption at these dose levels (Diliberto et al., 1993).

Elimination of radioactivity in faeces over the first 3 days was comparatively high after oral and intratracheal administration but was lower after intravenous and dermal exposure to 1 nmol [^{3}H]TBDD/kg body weight (see Table 41). The faecal excretion of only 2% of the administered dermal dose implies a very low elimination following dermal exposure. However, based on the percentage of the dermally absorbed dose, a value of 17% is obtained, which is comparable to the faecal elimination of an equimolar intravenous dose (Kedderis et al., 1991a; Diliberto et al., 1993).

According to Kedderis et al. (1992b), the large percentage of the dose excreted in urine and faeces after a single intravenous dose of [^{3}H]TBDD within the first few days could be attributed to a rapidly excreted impurity in the radiolabelled TBDD, which is not detectable by conventional radio-HPLC techniques. On the other hand, HPLC in combination with preceding hexane extraction of faeces was reported to distinguish successfully between parent TBDD and TBDD metabolites in analysing TBDD-derived radioactivity in faeces (Kedderis et al., 1991a; Diliberto et al., 1993). Results are compiled in Table 42. Three days after oral, intratracheal, and intravenous administration to rats of 1 nmol [^{3}H]TBDD/kg body weight, approximately 22, 18, and 3%, respectively, of the administered dose could be attributed to the parent compound extracted from faeces. In relation to the total radioactivity measured in faeces over 3 days, the content of parent TBDD ranged from 10 to 67% (see Table 43) after intravenous, intratracheal, and oral exposure (Kedderis et al., 1991a; Diliberto et al., 1993). Possible metabolites account for approximately 18–24% (oral study) and 14% (intravenous study) of the administered dose (Diliberto et al., 1993). Most of the parent TBDD was found in faeces collected at days 1 and 2 (see Table 42 and Ivens et al., 1992 in Table 41), which is

Table 42. Percent administered dose of parent [^{3}H]TBDD recovered in faeces of rats[a,b]

Route	Dose (nmol/kg body weight)	% administered dose excreted in faeces characterized as parent [^{3}H]TBDD[c]			
		Day 1 after dosing	Day 2 after dosing	Day 3 after dosing	Cumulative (days 1–3)
Oral	1	11.7 ± 3.6	7.9 ± 2.1	2.5 ± 1.6	22.2 ± 2.1
	10	6.9 ± 4.9	12.5 ± 3.8	2.0 ± 1.2	21.4 ± 1.8
	100	16.1 ± 9.6	16.7 ± 9.0	2.6 ± 1.6	35.4 ± 1.8[d]
	500	26.4 ± 11.2	18.3 ± 9.7	3.6 ± 3.4	48.3 ± 3.0[d]
Intratracheal	1	12.4 ± 1.7	4.6 ± 0.7	0.6 ± 0.02	17.6[d]
Intravenous[e]	1	1.6 ± 0.3	0.7 ± 0.3	0.5 ± 0.3	2.8[d]

[a] Adapted from Diliberto et al. (1993).
[b] Fischer 344 rats.
[c] Mean ± SD; n = 3 or 4; faecal extraction with hexane followed by HPLC characterization of the extract.
[d] Statistically different from 1 nmol/kg oral dose group ($p < 0.05$).
[e] Kedderis et al. (1991a).

Table 43. Contents of parent [^{3}H]TBDD in faeces of rats[a]

Route	Dose (nmol/kg body weight)	% total radioactivity in faeces characterized as parent [^{3}H]TBDD (cumulative percentages days 1–3)	Reference
Oral	1	53[b]	Diliberto et al. (1993)
	10	55	
	100	60	
	500	67	
Intratracheal	1	43	
Intravenous	1 and 100	10–20	Kedderis et al. (1991a)

[a] Group size: n = 3–4.
[b] Percentage represents the amount of parent TBDD that was excreted via faeces (days 1–3) as a result of unabsorbed TBDD and/or gastrointestinal transluminal excretion of TBDD.

consistent (at least for the oral and intratracheal studies) with the assumption that a portion of the material is not absorbed during passage through the gastrointestinal tract. Excretion of absorbed TBDD is thought to be limited by the rate of metabolism (Kedderis et al., 1991a).

6.4.2 Dibenzofurans

Elimination of 1,2,7,8-TeBDF in rats was primarily via biliary excretion of metabolite(s) (see section 6.3) in the faeces (Kedderis et al., 1994). Following intravenous administration of 1 nmol [^{3}H]-1,2,7,8-TeBDF/kg body weight, 39% of the dose was found in faeces and 55% in intestinal contents after 24 h. After administration of identical oral and dermal doses, 58 and 23%, respectively, of the doses were excreted into the faeces within 72 h. Excretion in the urine was only 2–3% of the intravenous, oral, or dermal dose (Kedderis et al., 1994).

6.5 Retention and turnover

Data on retention and turnover are available for some PBDDs (tri to penta), a tetraXDD, and some PBDFs (tetra, penta) in rats and for 2,3,7,8-TeBDD in humans.

6.5.1 Animal studies

Apparent elimination half-lives are complex for several PHDDs/PHDFs, for various reasons:

- As is well known for TCDD (Abraham et al., 1988), elimination from the main organs (liver and adipose tissue) does not proceed in the expected semi-logarithmic form, but is at least biphasic.
- The elimination half-lives from liver and adipose tissue (and also from some tissues like the thymus) are often not identical. This is due to metabolic conversion in and excretion from the liver and a more or less rapid equilibrium in the adipose tissue (probably dependent on lipid solubility).

- Tissue distribution is dose-dependent. The liver : adipose tissue concentration ratio may change by orders of magnitude when the dose is greatly increased (or decreases during elimination).
- The role of elimination may be different in various rodent strains, and it is dramatically different between various species (e.g. orders of magnitude difference for TCDD between rodents and humans). For this reason, the rate of cumulation to a steady state (for a given dose and dose interval) differs by a factor of about 100 for the rat and humans in the case of TCDD. Although no comparative data (rat versus human) exist for the elimination half-lives for PBDDs/PBDFs, similar differences must be expected.

The relative body burden of radiolabelled TBDD in rats depends on the route of exposure and on the dose administered (see Table 44), reflecting differences in absorption. However, if per cent body burdens are adjusted for the percent absorbed dose, they become more similar after oral, intratracheal, dermal, and intravenous administration of 1 nmol [^{3}H]TBDD/kg body weight (see Table 44). More than half of the body burden was found in the liver and adipose tissue of rats (Diliberto et al., 1993; see also section 6.2).

Distribution patterns of TBDD between the major tissue depots changed with dose and time (see also section 6.2), as seen in single-dose studies (Kedderis et al., 1991a, 1993; Diliberto et al. 1993) and in short-term studies (Ivens et al., 1990). For example, liver : adipose tissue TBDD concentration ratios declined from a maximum of 30 to 0.2 during a period of 56 days after a single intravenous administration of 1 nmol TBDD/kg body weight (Kedderis et al., 1991). At this dose, radioactivity levels in the liver peaked by 7 h and then gradually declined, concomitantly with a slow accumulation in adipose tissue, which reached the maximum concentration by 14 days (Kedderis et al., 1990). Levels of TBDD-derived radioactivity in blood declined rapidly to <2% of the administered dose by day 1 after dosing (Kedderis et al., 1991a). Liver concentrations of TBDD in rats, 1–78 days following a single subcutaneous injection of 600 ng (1.2 nmol)/kg body weight, were highest on day 3 after administration (Nagao et al., 1990c). In a 91-day oral study, the concentration of TBDD increased in the liver and adipose tissue of rats during

Table 44. Body burden of [^{3}H]TBDD-derived radioactivity in rats[a] 3 days after administration of a single dose

Route (vehicle)	Dose (nmol/kg body weight)	% body burden		Reference
		Administered dose	Absorbed dose	
Oral (water : ethanol : Emulphor® = 3 : 1 : 1)	1	58	73	Diliberto et al. (1993)
	10	61	75	
	100	41	67	
	500	28	59	
Intratracheal (water : ethanol : Emulphor® = 3 : 1 : 1)	1	59 ± 2	76 ± 2	Diliberto et al. (1993)
Dermal (acetone)	1	10 ± 1	82 ± 18	Diliberto et al. (1993)
Intravenous (water : ethanol : Emulphor® = 3 : 1 : 1)	1	82 ± 2	–	Diliberto et al. (1993); Kedderis et al. (1991a)

[a] Fischer 344 rats, n = 3–4.

treatment. In the subsequent recovery phase, a biphasic decline in TBDD concentrations in the liver was observed, whereas the concentration of TBDD in adipose tissue remained fairly constant, until a decrease began after a 30-day recovery period (Ivens et al., 1990).

Estimated half-lives for TBDD and other PBDD/PBDF congeners are compiled in Table 45.

Half-lives calculated for TBDD were as high as 58 days in adipose tissue and skin. Shorter half-lives (up to 27 days) were found in blood, muscle, liver, and whole body (Table 45). Compared with TCDD, half-lives of TBDD were similar for liver and whole body but higher (39–58 days versus 17–25 days) for adipose tissue (Kedderis et al., 1991a and references therein; Nagao et al., 1995/96 and references therein).

To compare the kinetics of three pairs of corresponding polychlorinated and polybrominated PHDDs/PHDFs, a mixture of the six substances (2,3,7,8-tetrahalogenated dibenzofuran [THDF], 2,3,4,7,8-pentahalogenated dibenzofuran [2,3,4,7,8-PeHDF], and 1,2,3,7,8-PeHDD) was given subcutaneously (single dose) to Wistar rats (1–2 nmol/kg body weight each). Concentration changes in liver and adipose tissue were monitored over a period of 95 days (Golor et al., 1993). Kinetics of both the chlorinated and brominated 2,3,4,7,8-PeHDF were similar in liver and also in adipose tissue, but levels were more than 10 times higher in the liver. The rate of decline was also very similar for the chlorinated and brominated 1,2,3,7,8-PeHDD in liver and also in adipose tissue, but the profile of the kinetics was different in the two tissues. Besides the level being more than one order of magnitude higher in the liver, there was a rather steady decline in the concentration in the hepatic tissue, while the concentration in adipose tissue increased within the first month (possibly because of redistribution phenomena) and then slowly declined thereafter.

The most remarkable difference between the chlorinated and brominated congeners was found in the case of THDF. The chlorinated congener (TCDF) is known to be rapidly eliminated from liver as well as adipose tissue in the rat. This was also found in these studies, the

Table 45. Biological half-lives of several PBDD/PBDF congeners in rats after single doses

Strain (sex)	Congener[a] (solvent)	Route (observation period)	Dose	Elimination from	Calculated half-life (days) (kinetic phase)	Reference
Dibenzo-*p*-dioxins						
Fischer 344 (female) (n = 3–4)	[^{3}H]TBDD (water : ethanol : Emulphor® = 3 : 1 : 1)	intravenous (56 days)	1 nmol/kg body weight	whole body	0.7 (1st phase) 2.9 (2nd phase) 17.8 (3rd phase)	Kedderis et al. (1991a)
			100 nmol/kg body weight	whole body	0.6 (1st phase) 17.8 (2nd phase)	
			1 nmol/kg body weight	liver	4.5 (1st phase) 16.5 (2nd phase)	
				adipose tissue	57.8	
				skin	2.5 (1st phase) 57.8 (2nd phase)	
				muscle	1.6 (1st phase) 26.7 (2nd phase)	
				blood	18.2	
Wistar (female) (n = 3–10)	TBDD (toluene/DMSO = 1+2; v/v)	subcutaneous (78 days)	60 ng/kg body weight (1.2 nmol/kg body weight)	liver	13.3 (12.0–14.9)[b]	Nagao et al. (1995/96)
				adipose tissue	39.4 (26–82)[b]	
Wistar (female) (n = n sp.)	1,2,3,7,8-PeBDD (toluene/DMSO = 1+2; v/v)	subcutaneous (35–95 days)	2.2 nmol/kg body weight[c]	liver	21 (17–27)[b]	Golor et al. (1993)
				adipose tissue	55 (39–97)[b]	

Table 45 (contd).

Strain (sex)	Congener[a] (solvent)	Route (observation period)	Dose	Elimination from	Calculated half-life (days) (kinetic phase)	Reference
Wistar (female) (n = 3)	2,3,7-TrBDD (<5% toluene in peanut oil/0.9% NaCl, 1+9, v/v)	intravenous 14 days	50 µg/kg body weight[c] (119 nmol/kg body weight)	liver adipose tissue thymus	2 (3rd phase) (47 h) 2–3 (3rd phase) (43 h) 3–4 (3rd phase) (91 h)	Golor et al. (1995)
	2,3-Cl_2,7-Br_1DD (<5% toluene in peanut oil/0.9% NaCl, 1+9, v/v)	intravenous 14 days	50 µg/kg body weight[c] (151 nmol/kg body weight)	liver adipose tissue thymus	3–4 (3rd phase) (72 h) 1.5 (3rd phase) (36 h) 3–4 (3rd phase) (92 h)	
Dibenzofurans						
Wistar (female) (n = n.sp.)	TBDF (toluene/DMSO = 1+2; v/v)	subcutaneous (35–95 days)	1.7 nmol/kg body weight[c]	liver adipose tissue	20 (17–25)[b] 30 (26–36)[b]	Golor et al. (1993)
Wistar (female) (n = n.sp.)	2,3,4,7,8-PeBDF (toluene/DMSO = 1+2; v/v)	subcutaneous (35–95 days)	1.1 nmol/kg body weight[c]	liver adipose tissue	99 (59–302)[b] 80 (49–220)[b]	Golor et al. (1993)

Table 45 (contd).

Strain (sex)	Congener[a] (solvent)	Route (observation period)	Dose	Elimination from	Calculated half-life (days) (kinetic phase)	Reference
Fischer 344 (male) (n = 3–4)	[^{3}H]1,2,7,8-TeBDF (water : ethanol : Emulphor® = 3 : 1 : 1)	intravenous (24 h)	1 nmol/kg body weight	body	1	Kedderis et al. (1994)

[a] n.sp. = not specified.
[b] 95% confidence interval in days.
[c] Given in a mixture together with other brominated and chlorinated PHDD/PHDF congeners.

rate of decline from the liver clearly being biphasic. In contrast, the brominated congener (TBDF) was much more slowly eliminated from both the liver and adipose tissue. While in the liver, the overall elimination rate resembled that of the chlorinated congener at the second elimination phase (about 2 weeks after administration); the elimination rate in the adipose tissue was comparatively slow (after an initial increase in the concentration during the first 2 weeks after administration). Thus, the THDF exhibited the larger kinetic difference between the chlorinated and the brominated forms, the TBDF being much more persistent in the rat. No comparative data are available for this pair of congeners in other species.

6.5.2 Human studies

Some data on retention and turnover in humans are available for 2,3,7,8-TeBDD and 2,3,7,8-TeBDF.

The first report of a PBDD in human tissue is indicative of the very long persistence of these compounds. Thirty-five years after exposure, markedly elevated levels of TBDD (625 pg TBDD/g blood lipid) were found in the blood of a chemist (see also section 5.3) who had synthesized TBDD and TCDD in 1956 (Schecter & Ryan, 1990, 1991, 1992; Schecter, 1992). It was not possible to calculate the actual half-life because of the lack of earlier measurements.

Another study (Zober et al., 1992) provided data for estimating the apparent half-lives of TBDD and TBDF. Employees of a chemical plant who had PBDD/PBDF body burdens resulting from processing brominated flame retardants (OBDE and DBDE) were monitored over a 3-year period from 1989 to 1991 (see also section 5.3). Based on data from three subjects, the following half-lives were calculated:

2,3,7,8-TeBDD: 2.9–10.8 years (mean: 5.9 years)
2,3,7,8-TeBDF: 1.1–1.9 years (mean: 1.5 years)

These half-lives are much longer than those reported in rats (section 6.5.1), but they are consistent with findings on the chlorinated analogues. Estimated half-lives of TCDD in humans ranged between 5 and 11 years (Poiger & Schlatter, 1986; Pirkle et al., 1989; Wolfe et al., 1994).

7. EFFECTS ON LABORATORY MAMMALS AND *IN VITRO* TEST SYSTEMS

7.1 Single exposure

7.1.1 Dibenzo-p-dioxins

The toxicity of PBDDs and PXDDs was studied in rats and mice after single oral (Yang et al., 1983; Ivens-Kohl et al., 1990; Ivens et al., 1992), intraperitoneal (Mason et al., 1987a,b), or subcutaneous (Nagao et al., 1990a) doses.

Single oral doses of 10, 33, 100, or 300 μg 2,3,7,8-TeBDD/kg body weight given to Wistar rats (5/sex per group) caused severe decreases in body weight gain (males) or weight loss (females) and deaths (see Table 46) at 100 and 300 μg/kg body weight during the 28 days of observation. At 33 μg/kg body weight, rats of both sexes showed slight decreases in body weight; at 10 μg/kg body weight, only the weight of females was slightly reduced. Emaciation, piloerection, and poor general health were seen in females of the 100 μg/kg body weight dose group and in most animals of the highest dose group (Ivens et al., 1992). Food and water intake were reduced in rats administered 100 μg TBDD/kg body weight (Ivens-Kohl et al., 1990). Absolute and relative thymus weights were decreased with increasing dose from 10 to 100 μg/kg body weight. At the highest dose of 300 μg/kg body weight, the thymus was not detectable in either male or female rats. The liver-to-body-weight ratio was increased in all dose groups (males: 10–25%, females: 4–15%). A dose-dependent increase in the testes-to-body-weight ratio began at 33 μg/kg body weight. Nearly all animals dying before term showed signs of haemorrhage in the gastrointestinal tract (Ivens et al., 1992).

Histological alterations were consistently seen in the thymus of TBDD-treated rats. Early thymic atrophy (characterized by the phagocytosis of lymphocytes by histiocytes) was observed at doses of 10 and 33 μg TBDD/kg body weight, and severe atrophy was seen at 100 μg TBDD/kg body weight. Lymphocytic depletion was also observed in the spleen at and above 33 μg TBDD/kg body weight and in Peyer's patches of the ileum (folliculi lymphatici aggregati) at

Table 46. Mortality associated with oral administration of PBDDs/PBDFs

PBDD/PBDF (carrier)	Species (strain)	Sex (number per group)	Dose (µg/kg body weight)	Details	Observation period[a]	Mortality (number dead/ number treated)	Time to death[b] (days)	Reference
2,3,7,8-TeBDD (arachis oil with 0.5–5% toluene)	rat (Wistar)	female, male (5)	10, 33	single dose	28 days	no mortality	–	Ivens et al. (1992)
			100	single dose	28 days	female: 3/5	n.r.	
			300	single dose	28 days	female: 5/5	11–19	
						male: 3/5	16–22	
2,3,7,8-TeBDD (arachis oil)	rat (Wistar)	female, male (10)	0.01	daily doses (for 90 days)	90 days	female: 1/10	n.r.	Löser & Ivens (1989); Ivens et al. (1993)
			0.10	daily doses (for 90 days)	90 days	female: 1/10	n.r.	
			1	daily doses (for 90 days)	90 days	female: 1/10 male: 2/10	n.r.	
			3	daily doses (for 90 days)	90 days	female, male: 5/10	n.r.	
			10	daily doses (for 90 days)	90 days	female, male: 10/10 (dead or moribund)	up to 35 days	

Table 46 (contd).

2,3,7,8-TeBDF (corn oil/ acetone)	rat (Sprague-Dawley)	female, male (5)	1, 10, 50	daily doses (5 days/week for 4 weeks)	4 weeks	no mortality	–	Hardy et al. (1990)
			150	daily doses (5 days/week for 4 weeks)	4 weeks	female: 4/5 male: 3/5 (dead or moribund)	18–24	
			500	daily doses (5 days/week for 4 weeks)	4 weeks	female, male: 5/5 (dead or moribund)	17–25	
2,3,7,8-TeBDF (corn oil)	guinea-pig (Hartley)	male (6)	0.47	single dose	30 days	no mortality	–	Moore et al. (1979)
			1.58	single dose	30 days	no mortality	–	
			4.74	single dose	30 days	1/6	26	
			15.84	single dose	30 days	6/6	10–13	

[a] After first dosing.
[b] n.r. = not recorded.

300 µg/kg body weight. Apart from lymphatic tissue, the organ most affected by TBDD treatment was the liver. First signs of dose-dependent hepatotoxicity such as cytoplasmic vacuolation and rarefaction and cellular hypertrophy were seen after a single administration of 10 µg TBDD/kg body weight. The cellular hypertrophy was accompanied by swelling of the nuclei, accentuated nucleoli, and cytoplasmic transformations. Pre-peliotic foci in the liver were seen at doses of 100 and 300 µg TBDD/kg body weight (Ivens et al., 1992).

Haematological investigations (in rats administered single oral doses of 10–300 µg TBDD/kg body weight) revealed a dose-dependent but marginal decline in haemoglobin content and cell number (thrombocytes, leukocytes, erythrocytes). Electrophoresis of serum proteins showed small dose-dependent changes in alpha- and beta-globulins. Thyroid hormone concentrations were reduced dose-dependently in serum (Ivens et al., 1992) (for details, see section 7.8.4).

A series of several PBDDs and PXDDs (tetra through penta) given intraperitoneally to immature male Wistar rats (n = 4) caused body weight losses 14 days after injection (Mason et al., 1987a,b). The most toxic compounds tested were 2,3,7,8-TeBDD, 2-Br_1-3,7,8-Cl_3-DD, and TBCDD, which are substituted only in the four lateral positions. The latter analogue exhibited the highest activity in this series (Mason et al., 1987a). The relative potencies of PBDDs examined followed the order 2,3,7,8- > 1,2,3,7,8- > 1,2,4,7,8- > 1,3,7,8- (Mason et al., 1987a,b). There were slight differences in the ED_{50} values for body weight loss (on a molar basis) between TCDD (Mason et al., 1986) and TBDD (Mason et al., 1987a,b).

Further effects following a single dose of PBDD/PBDF or PXDD/PXDF congeners are described in sections 7.8.1 (thymic atrophy) and 7.8.6 (hepatic enzyme induction).

Significant increases in relative liver weights were observed in male Sprague-Dawley rats given single oral doses of 25 µg 2,3,7-TrBDD/kg body weight (Yang et al., 1983) and in pregnant mice administered single subcutaneous doses of 5–90 µg TBDD/kg body weight (Nagao et al., 1990a).

7.1.2 Dibenzofurans

2,3,7,8-TeBDF given to six guinea-pigs in single oral doses of 0.47, 1.58, 4.74, or 15.84 µg/kg body weight (equivalent on a molar basis to 0.3–10 µg/kg 2,3,7,8-TeCDF) was lethal at 4.74 and 15.84 µg/kg body weight (see also Table 46). No deaths or body weight effects were seen in the lower dose groups. Animals found dead showed at necropsy a marked reduction in the size of the thymus, lack of body fat, and reduction of muscle mass. Histopathological findings in tissues of lethally intoxicated guinea-pigs included a loss of lymphoid cells in the thymic cortex and hyperplasia of epithelial cells in the renal pelvis, ureter, and urinary bladder. In addition, hypocellularity of bone marrow and seminiferous tubules, lymphoid elements in spleen, Peyer's patches, and adrenal haemorrhage were seen. In contrast to other rodents and rabbits, whose primary target organ after dibenzo-*p*-dioxin/dibenzofuran exposure is the liver, there was a lack of liver damage in the guinea-pigs. Mild thymus lymphoid hypoplasia was the only histological alteration in animals of the highest non-lethal dose group that survived the 30-day observation period (Moore et al., 1979). Altogether, the pattern of lesions was similar to those described for guinea-pigs exposed to PCDDs/PCDFs (Moore et al., 1979; Kociba & Schwetz, 1982; WHO, 1989).

7.1.3 Remarks on the lethality of PBDDs/PBDFs

It is characteristic for TCDD and related compounds to have a significant latency period between the time of exposure and the time of death. Further, these compounds are relatively persistent and produce similar effects (e.g. wasting syndrome) regardless of single or short-term exposure as a result of prolonged internal exposure (e.g. McConnell, 1989; WHO, 1989, 1994a). For these reasons, the few mortality data available for PBDDs/PBDFs from single and multiple dosing studies are compiled in this section.

All information refers to 2,3,7,8-TeBDD (Loeser & Ivens, 1989; Pinkerton et al., 1989; Ivens-Kohl et al., 1990; Ivens et al., 1992, 1993) and to 2,3,7,8-TeBDF (Moore et al., 1979; Pinkerton et al., 1989; Hardy et al., 1990).

The oral LD_{50} of TBDD in Wistar rats was about 100 µg/kg body weight for females and 300 µg/kg body weight for males (Ivens-Kohl et al., 1990). After single oral exposure of Sprague-Dawley rats to TBDD and TBDF, the LD_{50} was >500 µg/kg body weight (Brominated Flame Retardants Information Panel, 1987, cited in Pinkerton et al., 1989 and in Hardy et al., 1990, the latter specifying >500 µg/kg for TBDD and >5000 µg/kg for TBDF, *n* = n.sp.). Additional data on mortality are summarized in Table 46.

Although differences in the dosage regimen make a direct comparison impossible, it can be seen from Table 46 that guinea-pigs are more sensitive than rats to the lethal action of TBDF, which is consistent with findings observed with TCDD/TCDF (WHO, 1989). Generally, there are large differences in sensitivity between species and strains. For example, a more than 500-fold difference in acute LD_{50} values between the most TCDD-susceptible (Long-Evans) and the most TCDD-resistant (Han/Wistar) rat strains was reported for TCDD (WHO, 1989; Pohjanvirta et al., 1993, 1994). The oral LD_{50} was reported to range from 22 to >3000 µg TCDD/kg body weight in different rat strains (WHO, 1989).

In a study in guinea-pigs (Moore et al., 1979), equimolar doses of TCDF and TBDF resulted in comparable mortality rates (see Table 46). None (0/6) of the animals died following oral exposure to 1 µg TCDF/kg body weight (equivalent to 1.58 µg TBDF/kg body weight), whereas all six animals died at 10 µg TCDF/kg body weight (equivalent to 15.84 µg TBDF/kg body weight). Similarly, the mean time to death was about 12 days for both chemicals.

7.2 Short-term exposure

7.2.1 *Dibenzo-p-dioxins*

In a 3-month toxicity study (Ivens-Kohl et al., 1989; Löser & Ivens, 1989; Ivens et al., 1993), 2,3,7,8-TeBDD was administered daily by gavage to Wistar rats (10/sex per group) at doses of 0.01, 0.1, 1, 3, or 10 µg/kg body weight. No overt signs of toxicity were seen at 0.01 and 0.1 µg/kg body weight per day. Doses of 3 and 10 µg/kg body weight per day caused a high mortality (see Table 46) and

wasting syndrome. Mean body weight gain, feed intake, and water intake were reduced dose-dependently from 1 µg/kg body weight per day.

Changes in haemoglobin content, packed cell volume, and number of thrombocytes were seen mainly in rats given the 1 and 3 µg TBDD/kg body weight per day doses. The prothrombin time was markedly prolonged at 3 µg/kg body weight per day. Clinical chemistry showed slight increases in plasma alkaline phosphatase, aspartate aminotransferase, and total blood bilirubin in males and females receiving a dose of 1 µg/kg body weight per day. These changes were significant at 3 µg/kg body weight per day. Alanine aminotransferase was increased in females only at 3 µg/kg body weight per day. There was also a decrease in serum triglyceride levels, mainly at 1 and 3 µg/kg body weight per day. Dose-dependent changes in thyroid hormone concentrations in serum were observed in male and female rats. Triiodothyronine (T_3) levels were increased and thyroxin (T_4) levels were decreased at doses of 0.1 µg/kg body weight per day and higher. The effects at 0.1 µg/kg body weight per day, however, were considered to be marginal (see also section 7.8.4). Activities of microsomal enzymes were dose-dependently elevated (see also section 7.8.6). Protein excretion in urine increased in males and females at doses higher than 3 µg/kg body weight per day.

Changes in relative organ weights (increase in liver, lung, kidney; reduction in thymus) were generally observed at doses of 1 µg/kg body weight per day and higher. Relative liver weights were significantly increased at doses of 0.1 µg/kg body weight per day and higher; relative thymus weights were significantly decreased at doses of 0.01 µg/kg body weight per day and higher. Histopathological examination revealed dose-dependent changes, mainly beginning at the 1 µg/kg body weight per day dose. These included severe atrophy of lymphatic tissue in thymus and spleen and liver damage described as peliosis hepatis parenchymatosa (irregular-shaped cavernous and blood-filled spaces in the liver, lack of epithelial lining, blood cysts in the sinusoidal lumen and in Disse's spaces, etc.) (Bannasch et al., 1985). Spermatogenesis in the testes was adversely affected, and defective or necrotic spermatocytes were found in the epididymis (Ivens et al., 1993; see also section 7.5).

The NOAEL in this study was considered to be 0.01 μg TBDD/kg body weight per day.

Compared with TCDD, TBDD elicits a similar spectrum of toxic effects following subchronic exposure but appears to be less active than TCDD. A subchronic NOAEL for TCDD (dosing for only 5 days/week) in rats was reported to be 0.01 μg/kg body weight per day (Kociba et al., 1976). However, peliosis hepatis was not reported to occur after treatment with TCDD (Ivens et al., 1993 and references therein).

7.2.2 Dibenzofurans

2,3,7,8-TeBDF was administered to Sprague-Dawley rats (5/sex per group) at daily oral doses of 1, 10, 50, 150, or 500 μg/kg body weight, 5 days/week for 4 weeks (Fulfs, 1989; Hardy et al., 1990). Most animals in the 150 and 500 μg/kg body weight per day dose groups died or were in a moribund condition between study days 17 and 24. At 150 μg/kg body weight per day, 3 of 10 rats survived through day 28 (see Table 46). Animals from the two highest dose groups had yellowish urine beginning around day 15. Group mean body weight was depressed in a time- and dose-dependent manner. The mean relative thymus weights of males at 150 μg/kg body weight per day and of females at 10 and 50 μg/kg body weight per day were decreased. No significant changes in relative mean liver, adrenal, or spleen weights were detected. In this study, treatment-related histopathological alterations were noted in the liver and thymus of animals in the 50 μg/kg body weight per day dose group and, to a lesser extent, in the 10 μg/kg body weight per day dose group. Liver changes consisted of panlobular hypertrophy of the hepatocytes with associated hepatocyte vacuolation and focal necrosis. Thymic atrophy consisting of overall depletion of the lymphoid elements was present in all 50 μg/kg body weight per day rats from which thymus was available and in most animals from the 10 μg/kg body weight per day dose group. No treatment-related alterations were observed in the 1 μg/kg body weight per day group (Hardy et al., 1990), which can be considered as the NOAEL for this study.

The results do not indicate a lower potency of TBDF compared with TBDD, as the studies differed in their experimental design (dosing for 4 weeks, 5 days/week versus 13 weeks, 7 days/week; Wistar versus Sprague-Dawley rats; different animal numbers/dose group).

7.3 Long-term exposure

No long-term exposure studies with PBDDs/PBDFs were available.

7.4 Skin and eye irritation, sensitization, dermal lesions, and acne

A common feature of toxicity of dioxin-like compounds such as PCDDs/PCDFs, PCBs, and PBBs is their hyperkeratotic activity in humans and some animal species (WHO, 1989, 1993, 1994a).

A standard test method for acnegenic activity, the rabbit ear assay first described by Adams et al. (1941), was applied to 2,3,7,8-TeBDD and 2,3,7,8-TeBDF (Pinkerton et al., 1989). Both congeners produced hyperkeratosis at a total dose of 100 µg/rabbit, but not at 10 µg/rabbit (probably repeated application over a 4-week period). The solvent used was not specified. Under the same conditions, combustion residues (soot/char) from a HIPS/DBDE/Sb_2O_3 sample (for the PBDF content, see Table 11) were found to have no acnegenic activity (Pinkerton et al., 1989).

In the case of TCDD, the minimum dose inducing hyperkeratosis after a single administration ranged from 1 µg TCDD/ear to 160 µg TCDD/ear, depending on the vehicle used (Poiger & Schlatter, 1980). During a 4-week test with TCDD, no effect was observed at a total dose of 8 ng/rabbit, and dose-dependent responses were obtained at 0.08–800 µg/rabbit (Schwetz et al., 1973).

7.5 Reproductive and developmental toxicity

7.5.1 Reproductive toxicity

A dose-dependent increase in testes-to-body-weight ratio was seen 28 days after oral administration of single doses of 2,3,7,8-TeBDD (0, 10, 33, 100, or 300 µg/kg body weight; solvent: arachis oil with 5% toluene) to male Wistar rats (n = 5). This effect was seen from 33 µg/kg body weight onwards. Body weight gain was reduced dose-dependently (marginally at 33 µg/kg body weight), but there was no loss of body weight (Ivens et al., 1992).

Decreased spermatogenic activity in the testes and defective or necrotic spermatocytes in the epididymis were found in Wistar rats (n = 10) after daily oral administration of TBDD (in arachis oil) at 3 or 10 µg/kg body weight per day, 7 days/week for 13 weeks. Severe effects were observed at 10 µg/kg body weight per day, and moderate effects at 1 µg/kg body weight per day. The NOEL was 0.1 µg/kg body weight per day (Ivens-Kohl et al., 1989; Ivens et al., 1993).

Adverse effects on the male reproductive system (e.g. reduction in number, size, and organelle content of Leydig cells in adult rat testes) were observed following single intraperitoneal injections of 12.5–50 µg TCDD/kg body weight (Johnson et al., 1994). However, at perinatal exposure, a single oral dose as low as 64 ng TCDD/kg body weight (the lowest maternal dose tested) given to mothers at day 15 of gestation was sufficient to reduce sperm production in the male offspring (Mably et al., 1992). Reduced sperm numbers were also observed in offspring of pregnant rats administered 1 µg TCDD/kg body weight on gestation day 8 or 15 and in offspring of Syrian hamsters dosed with 2 µg/kg body weight on gestation day 11 (Gray et al., 1995).

7.5.2 Developmental toxicity

Several PBDDs/PBDFs were found to be inducers of cleft palate and hydronephrosis in mice. These effects occurred at doses (see Table 47) that produce no or only marginal general maternal toxicity and no fetal mortality (Nagao et al., 1990a,d; Birnbaum et al., 1991).

Table 47. Developmental toxicity of PBDDs/PBDFs

PBDDs/PBDFs (vehicle)	Species (strain) (*n*)[a]	Route (dosing regimen)[b]	Dose (µg/kg body weight)	Effects	Remarks	Reference
2,3,7,8-TeBDD (toluene/DMSO = 1+2, v/v)	mouse (NMRI) (n.sp.)	subcutaneous (single dose on gd 9)	0–90 (5, 10, 30, 50, 90)	**from 5 µg/kg body weight**: F_0: increase in relative liver weight, F_1: cleft palate, ED_{50}: 62 µg/kg body weight (0.123 µmol/kg body weight)	no effect on the number of viable fetuses per litter, fetal weight, or fetal deaths	Nagao et al. (1990a,b)
2,3,7,8-TeBDD (corn oil)	mouse (C57BL/6N) (11–20)	oral (single dose on gd 10)	0–192 (3, 6, 12, 24, 48, 96, 192)	**from 3 µg/kg body weight**: F_0: increase in relative liver weight, F_1: hydronephrosis, ED_{50}: 9 µg/kg body weight (0.018 µmol/kg body weight) **from 48 µg/kg body weight**: F_1: cleft palate, ED_{50}: 65 µg/kg body weight (0.13 µmol/kg body weight)	trend for a decrease in number of live fetuses and an increase in fetal body weight with increasing dose	Birnbaum et al. (1991)

Table 47 (contd).

PBDDs/PBDFs (vehicle)	Species (strain) (*n*)[a]	Route (dosing regimen)[b]	Dose (µg/kg body weight)	Effects	Remarks	Reference
2,3,7,8-TeBDF (corn oil)	mouse (C57BL/6N) (7–22)	oral (single dose on gd 10)	0–4000 (25, 50, 100, 200, 250, 500, 1000, 3000, 4000)	**from 25 µg/kg body weight**: F_0: increase in relative liver weight, F_1: hydronephrosis, ED_{50}: ~12 µg/kg body weight (0.024 µmol/kg body weight) **from 200 µg/kg body weight**: F_1: cleft palate, ED_{50}: 154 µg/kg body weight (0.31 µmol/kg body weight) **from 500 µg/kg body weight**: F_1: increase in fetal mortality and live fetal body weight **4000 µg/kg body weight**: F_0: subcutaneous oedema		Birnbaum et al. (1991)
1,2,3,7,8-PeBDF (corn oil)	mouse (C57BL/6N) (5–11)	oral (single dose on gd 10)	0–4000 (250, 500, 1000, 2000, 3000, 4000)	**from 250 µg/kg body weight**: F_0: increase in relative liver weight **from 500 µg/kg body weight**: F_1: hydronephrosis, ED_{50}: ~340 µg/kg body weight **from 3000[c] µg/kg body weight**: F_1: cleft palate, ED_{50}: 4088 µg/kg body weight	no increase in fetal mortality or fetal body weight	Birnbaum et al. (1991)

Table 47 (contd).

2,3,4,7,8-PeBDF (corn oil)	mouse (C57BL/6N) (9–16)	oral (single dose on gd 10)	0–4000 (25, 50, 100, 200, 400, 800, 1600, 2400, 4000)	**from 25 µg/kg body weight**: F_0: increase in relative liver weight **from 400 µg/kg body weight**: F_1: hydronephrosis, ED_{50}: ~437 µg/kg body weight **from 2400 µg/kg body weight**: F_1: cleft palate, ED_{50}: 3024 µg/kg body weight	no effect on fetal mortality or fetal body weight	Birnbaum et al. (1991)

[a] Number of litters; n.sp. = not specified.
[b] gd = gestation day (first day of gestation designated day zero).
[c] First significant at 3000 µg/kg body weight, statistically ($p < 0.01$) significant at 4000 µg/kg body weight.

Single subcutaneous injections of TBDD or TCDD (5–90 μg/kg body weight) were administered to NMRI mice on day 9 of pregnancy. Maternal and fetal toxicity were assessed on day 18 (Nagao et al., 1990a,d). TBDD and TCDD caused significant dose-related increases in the incidence of cleft palate in the total number of viable fetuses and in the number of litters with cleft palate. On a molar basis, the potency of TBDD for cleft palate induction relative to that of TCDD was found to be 0.6: ED_{50} values were 61.7 μg/kg body weight per day (corresponding to 0.124 μmol/kg body weight per day) for TBDD and 24 μg/kg body weight per day (corresponding to 0.075 μmol/kg body weight per day) for TCDD. Both TBDD and TCDD at the doses given increased maternal liver-to-body-weight ratios. There were no significant effects on maternal weight gain or the number of viable fetuses per litter.

In another study (Birnbaum et al., 1991), the teratogenic effects of TBDD and three PBDFs (TBDF, 1,2,3,7,8-PeBDF, and 2,3,4,7,8-PeBDF) were examined in C57BL/6N mice. Pregnant dams were treated on gestation day 10 with single oral doses of each congener and sacrificed on gestation day 18. Doses ranged from 0 to 192 μg/kg body weight for TBDD and from 0 to 4000 μg/kg body weight for the three dibenzofurans. All compounds produced hydronephrosis at doses below that at which cleft palate occurred (see also Table 47). The LOELs (μg/kg body weight) for hydronephrosis and cleft palate, respectively, were as follows: TBDD: 3 and 48; TBDF: 25 and 200; 1,2,3,7,8-PeBDF: 500 and 3000–4000; and 2,3,4,7,8-PeBDF: 400 and 2400. Embryo/fetal mortality was significantly increased at 500 μg TBDF/kg body weight and higher doses. At 3000 μg TBDF/kg body weight, the few survivors were oedematous. After exposure to TBDD, the number of live fetuses showed a "decreasing trend," which was not significant. Dose-related increases in fetal weights were observed with TBDD and TBDF. However, the increases were significant only for TBDF at doses of 500 μg/kg body weight and higher. Maternal liver weight increased at all dose levels examined for all four compounds. Maternal weight gain was elevated at the highest dose of TBDF (4000 μg/kg body weight), and this increase was due to subcutaneous oedema.

Birnbaum et al. (1991) compared the relative toxicities of the brominated congeners with those of the chlorinated ones. Based on the approximate ED_{50} values, TBDD appeared to be almost half as potent as TCDD in the induction of hydronephrosis in offspring of treated dams (4 µg/kg body weight versus 9 µg/kg body weight; see also Table 47). However, compared on a molar basis, TBDD and TCDD were almost equipotent. A survey of the relative potencies (on a weight basis) of PBDD/PCDD and PBDF/PCDF congeners for the induction of cleft palate is given in Table 48. It can be seen that bromination decreased the activity of TBDD relative to TCDD but increased the potency of TBDF relative to TCDF. The pentaBDFs tested, however, were slightly less potent than the pentaCDFs.

Table 48. Relative potencies of PBDD/PCDD and PBDF/PCDF congeners relative to TCDD for the induction of cleft palate in mice[a]

Congener	Relative potencies for cleft palate[b]	
	H = Br	H = Cl
Dibenzo-*p*-dioxins		
2,3,7,8-TeHDD	0.24	1
Dibenzofurans		
2,3,7,8-TeHDF	0.10	0.05
1,2,3,7,8-PeHDF	0.004	0.03
2,3,4,7,8-PeHDF	0.005	0.09

[a] Adapted from Birnbaum et al. (1991).
[b] Derived from ED_{50} values; on a weight basis.

7.6 Mutagenicity and related end-points

No information was found on the mutagenicity of PBDDs/PBDFs or on related end-points.

Whereas there is limited or conflicting evidence demonstrating the positive mutagenic potential of PCDDs (WHO, 1989), co-mutagenic or co-recombinogenic effects of PCDDs have been demonstrated *in vivo* in the mouse spot test (Fahrig, 1993).

The cell-transforming potential of TBDD has been demonstrated in a host-mediated *in vivo/in vitro* assay with peritoneal murine macrophages (Massa et al., 1990). NMRI mice were intraperitoneally administered 0.39 nmol TBDD (corresponding to 195 ng) or TCDD (corresponding to 125 ng) per mouse. Isolation of resident macrophages 4 days later, cultivation in soft agar for 5–6 days, and evaluation of the clones indicated that the transforming capacity of TBDD was seven times less than that of TCDD (Massa et al., 1991).

7.7 Carcinogenicity

7.7.1 Short-term studies

A permanent cell line was established from peritoneal macrophages of mice treated with TBDD. These cells were tested for their tumorigenicity in athymic nude (nu/nu) mice. Animals given subcutaneous injections of these cells (1×10^6 cells) developed tumours at the injection site 3 weeks later (Massa et al., 1991, 1992a,b).

7.7.2 Long-term studies

PBDDs/PBDFs have not been tested for carcinogenicity in long-term studies.

TCDD was shown to be a multisite carcinogen in both sexes of rats and mice at doses below the maximum tolerated dose (WHO, 1989).

7.8 Other special studies

7.8.1 Immunotoxicity

Thymus atrophy and other signs of immunotoxicity were found to be the main characteristic toxic effects (besides body weight loss or decrease in the rate of weight gain) after exposure to TCDD and were observed in almost all laboratory animals (Vos & Luster, 1989; WHO, 1989). The limited data available for PBDDs/PBDFs from studies with rats, guinea-pigs, and monkeys confirmed the expected immunotoxic potential. Parameters examined were influences on lymphoid tissues,

effects on serum protein levels or other haematological parameters, and alterations of certain lymphocyte subpopulations in peripheral blood. Effects on immunotoxicity after perinatal exposure to PBDDs/PBDFs have not been investigated.

7.8.1.1 *Dibenzo-p-dioxins*

Dose-dependent decreases in thymus weights and atrophy of thymus and other lymphatic tissues were observed in rats after single exposures to 2,3,7,8-TeBDD (Mason et al., 1987a,b; Ivens et al., 1992; see also section 7.1) and to a series of other PBDD or PXDD congeners (Mason et al., 1987a,b) and after subchronic exposure to 2,3,7,8-TeBDD (Ivens-Kohl et al., 1989; Löser & Ivens, 1989; Ivens et al., 1993; see also section 7.2). Effects were found after single doses of 10 µg/kg body weight or higher and after daily doses of 1 µg/kg body weight and higher for 3 months (Ivens-Kohl et al., 1989; Löser & Ivens, 1989; Ivens et al., 1992, 1993). The potency of TBDD for causing thymic atrophy was comparable to that of TCDD (Mason et al., 1987b). 1,3,7,8-TeBDD, 1,2,3,7,8-PeBDD, and 1,2,4,7,8-PeBDD were less active (Mason et al., 1987b). The mixed 2-Br_1-3,7,8-Cl_3DD was as active as TBDD and TCDD; the mixed 2,3-Br_2-7,8-Cl_2DD was the most toxic analogue in this series (Mason et al., 1987a).

Some routine haematological parameters, including serum protein electrophoresis (reduction in number of thrombocytes, prolongation of prothrombin time, slight anisocytosis, poikilocytosis, slight reduction in gamma globulins, etc.), examined in the subchronic (3 months) rat study (doses applied: 0.01–10 µg/kg body weight) might suggest an impact on cellular and, possibly, humoral immunity beginning at daily doses of 1 µg TBDD/kg body weight (Ivens-Kohl et al., 1989).

It has been shown that very low doses of PBDDs and PCDDs affect the immune system of the marmoset monkey (Neubert R. et al., 1990, 1991, 1992, 1993; Neubert, 1991, 1993a). After dibenzo-*p*-dioxin exposure, alterations were seen in lymphocytes from peripheral blood of mature marmosets. The lymphocyte subpopulations showing the most pronounced effects were the "helper-inducer" or "memory" T cells ($CD4^+CDw29^+$ in the marmoset, probably corresponding to the $CD4^+CD45RO^+CDw29^+$ cells in humans) and certain B cell subsets

(e.g. $CD20^+$) (Neubert, 1993a; Neubert et al., 1993). Several weeks after a single subcutaneous dose of 30 ng TBDD/kg body weight (corresponding to 60 pmol TBDD/kg body weight), there was a significant decrease in the percentage and absolute number of the T helper-inducer subpopulation ($CD4^+CDw29^+$) in venous blood of treated marmosets ($n = 3$). This effect on cell-mediated immunity is pronounced in young animals (Vos, 1993). In addition, a significant decrease in the number of B cells ($CD20^+$) was observed (Neubert, 1993a). This is considered to be less dependent on the age of animals (Vos, 1993). Injection of 3 ng TBDD/kg body weight (corresponding to 6 pmol TBDD/kg body weight) did not induce any changes (Neubert, 1993a; Neubert et al., 1993).

With TCDD, a clear dose–response relationship was found at doses of 10 ng/kg body weight and higher, and a questionable response (confined to single individuals) was seen at 3 ng/kg body weight. No effect was detectable with single doses of 1 ng TCDD/kg body weight or lower. It was concluded that, on a molar basis, the potencies of TBDD and TCDD may be similar in this experimental approach (Neubert, 1993a; Neubert et al., 1993).

One mixed halogenated congener, namely TBCDD, was also investigated. No changes in the subpopulations tested were found following a single subcutaneous dose of 3 ng/kg body weight. Treatment with 30 TBCDD/kg body weight decreased the percentage of $CD24^+CDw29^+$ cells and of $CD20^+$ lymphocytes. However, no significant effect was seen in the small number of marmosets used ($n = 3$) on the absolute number of cells per µl of blood. From these data, it appeared that this congener may be less potent than TCDD and TBDD (Neubert, 1993a; Neubert et al., 1993).

7.8.1.2 *Dibenzofurans*

A marked reduction in the size of the thymus, loss of lymphoid cells in the thymic cortex, hypocellularity of bone marrow, lymphoid elements in spleen, and Peyer's patches were seen in guinea-pigs after single oral doses of TBDF (4.7–15.8 µg/kg body weight) that also induced mortality. At a non-lethal dose of 1.6 µg/kg body weight, only mild evidence of thymus lymphoid hypoplasia was noted after the 30-

day observation period (Moore et al., 1979; see also section 7.1). Short-term exposure of Sprague-Dawley rats to daily oral doses of 1–500 µg TBDF/kg body weight (5 days/week for 4 weeks) caused dose-dependent reductions in thymus weights and thymic atrophy at the 10 µg/kg body weight per day and higher doses (Hardy et al., 1990; see also section 7.2).

7.8.2 *Effects on intermediary metabolism: Porphyrin effects*

Hepatic porphyrin accumulation was studied after subchronic dosing (5 days/week for 13 weeks) of female B6C3F$_1$ mice by oral gavage with individual congeners of PBDDs/PCDDs, including 2,3,7,8-TeBDD and 2,3,7,8-TeCDD (Table 49). Dose-dependent increases in total hepatic porphyrins were found for both TBDD and TCDD. The relative porphyrinogenic potencies were determined by the authors using TCDD as a reference (TCDD = 1 and TBDD = 0.4).

Table 49. Total hepatic porphyrin accumulation in female B6C3F$_1$ mice after 13 weeks of exposure to TBDD or TCDD[a]

TBDD		TCDD	
Dose (ng/kg body weight per day)	Hepatic porphyrin accumulation (µg/g)	Dose (ng/kg body weight per day)	Hepatic porphyrin accumulation (µg/g)
0	0.207	0	0.196
30	0.296	0.15	0.204
90	0.346	0.45	0.212
300	0.429	1.5	0.212
900	13.4	4.5	0.222
3000	26.2	15	0.256
		45	0.592
		150	13.2
		450	18.6

[a] Adapted from van Birgelen et al. (1996).

7.8.3 Effects on vitamin A storage

Various brominated or chlorinated aromatic compounds are able to reduce the vitamin A content of the liver (WHO, 1989, 1993, 1994a). A single oral dose of 10 µg TBDD/kg body weight decreased both the concentration and the total amount of vitamin A (retinol) in the liver of adult male Sprague-Dawley rats (n = 5) 4 weeks after the start of the experiment. Reductions in concentration and total amount of vitamin A were 45 and 51%, respectively (Thunberg et al., 1984). A more pronounced effect was elicited by TCDD tested in the same study, the reductions being 88 and 87%, respectively. However, on a molar basis, TCDD was only slightly more potent than TBDD (Thunberg et al., 1984).

7.8.4 Endocrine interactions

Thyroid hormones were affected in rats after single (Ivens et al., 1992; see also section 7.1) and subchronic (Löser & Ivens, 1989; Ivens et al., 1993; see also section 7.2) exposures to 2,3,7,8-TeBDD. Four weeks after single oral doses of 10, 33, or 100 µg TBDD/kg body weight, T_3 was increased and T_4 was reduced dose-dependently in the serum of female and male Wistar rats (n = 5) (Ivens et al., 1992). Wistar rats (10/sex per group) treated with daily doses of 0.01–3.0 µg/kg body weight per day for 3 months had reduced T_4 and increased T_3 levels at 0.1 µg/kg body weight per day and higher (highest dose group of 10 µg/kg body weight per day not examined) (Löser & Ivens, 1989; Ivens et al., 1993; see also section 7.2.1).

The antiestrogenic potency of a series of PXDDs/PXDFs was examined in cultures of MCF-7 human breast cells (Spink et al., 1994). Two effects, stimulation of the metabolism of 17β-estradiol and inhibition of the estrogen-dependent formation of multicellular foci, were measured as indices of antiestrogenicity. Several tetra- (Br_1Cl_3DDs, Br_2Cl_2DDs) and penta- (Br_1Cl_4DD) halogenated congeners with 2,3,7,8-substitution stimulated estradiol metabolism with a potency similar to that of TCDD. The EC_{50} values of these PXDDs were 0.6–0.8 nmol (TCDD: 0.8 nmol). The focus formation was inhibited by the same congeners, with EC_{50} values ranging from 0.5 to 1.2 nmol (TCDD: 0.3 nmol). Dibenzo-p-dioxins and dibenzofurans

with other substitution patterns were markedly less active (EC_{50} values in nmol for inhibition of focus formation: 17, 8-Br_1-2,3,4-Cl_3DF and 7-Br_1-2,3-Cl_2-DD; 1700, 2,3,7-Cl_3-8-methylDD; >1700, 2,7-Br$_2$DF and several methyl-substituted PCDDs/PCDFs).

7.8.5 *Interaction with drugs and toxicants*

A single oral dose of 2,3,7-TrBDD (25 µg/kg body weight) depressed the plasma disappearance of ouabain and its excretion in bile of male Sprague-Dawley rats 10 days after treatment. In addition, the bile flow was decreased by 2,3,7-TrBDD. TCDD elicited the same effects but more markedly than 2,3,7-TrBDD (Yang et al., 1983).

7.8.6 *Induction of microsomal enzymes*

PBDDs/PBDFs, like TCDD and other environmental contaminants, are potent inducers of certain CYP-dependent enzymes. Most frequently, the activity of the cytochrome CYP1A1 was examined (direct measurements by radioimmunoassay [RIA] or determination of marker enzymes [mostly AHH and EROD]).

The isoenzyme CYP1A2 (determined by RIA) was also found to be inducible. ED_{50} values of 0.8–1 nmol/kg body weight for CYP1A1 induction and 0.2 nmol/kg body weight for CYP1A2 induction in rat liver were estimated after single oral doses of TBDD (Kedderis et al., 1991b, 1992a, 1993). Induction of CYP1A2 enzyme activity has been determined in mouse liver after subchronic dosing by oral gavage with TBDD (Birnbaum & DeVito, 1995) and in liver of marmoset monkeys after single subcutaneous doses of TBDD (Schulz et al., 1996).

Induction of the monooxygenases AHH and/or EROD was observed following single exposure of chicken embryos (Kende & Wade, 1973; Poland & Glover, 1973; Kende et al., 1974; Ramalingam et al., 1986), guinea-pigs (Schmidt & Ivens-Kohl, 1990b), rats (Thunberg et al., 1984; Mason et al., 1987a,b; Nagao et al., 1990b,c; Schmidt & Ivens-Kohl, 1990a,b; Schulz-Schalge et al., 1990, 1991a,b; Kedderis et al., 1991b, 1992a, 1993; Ivens et al., 1992), and marmoset monkeys (Schulz et al., 1993, 1996) and after subchronic dosing by oral gavage in mice (Birnbaum et al., 1993; Birnbaum & DeVito,

1995; van Birgelen et al., 1996) and rats (Ivens et al., 1993), as well as in primary cultures of rat hepatocytes (Blankenburg et al., 1990) and in rat hepatoma H-4-II E cells (Bradlaw & Casterline, 1979; Bradlaw et al., 1980; Denomme et al., 1985, 1986; Mason et al., 1987a; Zacharewski et al., 1988, 1989; Safe et al., 1989a,b). Almost all experiments referred to the tetra- and pentaBDDs/XDDs. Only a few studies (Poland & Glover, 1973; Kende et al., 1974; Denomme et al., 1985) included tri- or diBDDs/XDDs. There was one study (Schulz et al., 1993) on dibenzofurans (pentaBDF) and another (Denomme et al., 1986) on PXDFs.

Several aspects of enzyme induction by PHDDs/PHDFs, with emphasis on the chlorinated congeners, are addressed in Goldstein & Safe (1989) and Neubert (1993b). The general mechanisms of induction of the CYP1A1 enzyme system were discussed by Okey (1990, 1992).

7.8.6.1 *Dibenzo-p-dioxins*

Enzyme induction proceeded dose-dependently at non-toxic concentrations. It was measurable at exposures as low as the pmol range (Schulz-Schalge et al., 1990, 1991b: <100 pmol TBDD/kg body weight in the rat). Induction started soon after exposure and was long-lasting, 1–98 days after a single subcutaneous administration of TBDD, TCDD, and tetraXDDs (Nagao et al., 1990c; Schulz-Schalge et al., 1991a). Maximum EROD induction was seen approximately 7 days after a single subcutaneous exposure of rats to several 2,3,7,8-substituted dibenzo-*p*-dioxins (Nagao et al., 1990c; Schulz-Schalge et al., 1991a). Oral exposure of rats to 1 µg TBDD/kg body weight per day for 91 days also showed maximum induction after 7 days of treatment (Ivens et al., 1993). In contrast, during low-dose oral treatment of mice (30–3000 ng TBDD/kg body weight per day for 13 weeks), a biphasic response was observed, not leading to maximum enzyme induction (Birnbaum et al., 1993).

In a comparison of the decline in EROD activity after single doses of 600 µg TBDD/kg body weight and 300 µg TCDD/kg body weight, the curves were superimposable over a period of 90 days (Nagao et al., 1995/96). When the hepatic EROD activity was related

to the molar concentrations of TBDD and TCDD, the same relationship was also demonstrated. This indicates that both TBDD and TCDD exhibit identical potencies with respect to the induction of EROD, when compared on a molar basis. When compared on a weight basis (as is done for an international toxic equivalent [I-TEQ] approach), TBDD is less potent with respect to this end-point.

Although the liver was the most important organ for enzyme induction, EROD induction by TBDD was also observed in other tissues (lung, skin, kidney) of rats (Ivens et al., 1993) and mice (Birnbaum et al., 1993) and in lung of marmoset monkeys (Schulz et al., 1996).

The induction potency for different PBDD congeners varied over several orders of magnitude. For example, the activities for *in vivo* hepatic AHH induction differed by four orders of magnitude between four brominated congeners, the most active being 2,3,7,8-TeBDD and the least 1,3,7,8-TeBDD (see Tables 50 and 51). Similar structure–activity correlations were seen in other experimental systems (*in vitro* studies, EROD activity, toxic effects) and were discussed in several reviews (Goldstein & Safe, 1989; Safe et al., 1989a,b; Safe, 1990). As with PCDDs (WHO, 1989), the most potent compounds were those having the 2,3,7,8-substitution pattern. In summary, the most effective inducers of *in vitro* and/or *in vivo* AHH or EROD activity were TCDD, TBDD, and TBCDD.

Compared on a molar basis to their chlorinated analogues, the PBDDs and PXDDs have more or less similar potencies (see Tables 50 and 51; Kende & Wade, 1973; Kende et al., 1974; Bradlaw & Casterline, 1979; Thunberg et al., 1984; Denomme et al., 1985; Abraham et al., 1988; Blankenburg et al., 1990; Nagao et al., 1990c; Schulz-Schalge et al., 1990, 1991b). Nevertheless, some differences are apparent between TBDD and TCDD, the congeners most intensively studied. In contrast to TCDD, whose relative induction potency was independent of the tissue examined, TBDD was five times more potent in inducing EROD activity in the liver than in skin and lung following subchronic exposure of mice (Birnbaum et al., 1993).

Table 50. *In vivo* ED_{50} values for AHH induction and toxic effects of several PHDDs in the rat[a,b]

Congeners	*In vivo* ED_{50} (μmol/kg body weight)		
	Inhibition of body weight gain	Thymic atrophy	Hepatic AHH induction
2,3,7,8-TeBDD	0.068	0.034	0.0076
2,3,7,8-TeCDD	0.05	0.09	0.004
2,3-Br_2-7,8-Cl_2-DD	0.012	0.0073	0.000 49
2-Br_1-3,7,8-Cl_3-DD	0.12	0.035	0.0025
1,2,3,7,8-PeBDD	0.87	0.39	0.025
1,2,3,7,8-PeCDD	0.62	0.17	0.031
1,2,4,7,8-PeBDD	12.9	6.17	0.195
1,2,4,7,8-PeCDD	34.0	11.2	2.82
1,3,7,8-TeBDD	252.0	35.5	6.50
1,3,7,8-TeCDD	132.0	100.0	31.2

[a] Adapted from Mason et al. (1987a); Safe et al. (1989b).
[b] Immature male Wistar rats (n = 4), 14 days after single intraperitoneal doses.

Only limited data are available on possible species-dependent variations. The estimated potency for EROD induction of TBDD relative to TCDD ranged from 0.75 to 5.3 in rats (calculated by Birnbaum et al., 1993 from data of Safe, 1990) and from 0.04 to 0.2 in mice (Birnbaum et al., 1993). According to the authors, it is possible that these differences may be indicative of species differences, despite the different experimental designs used in the rat and mice studies.

7.8.6.2 *Dibenzofurans*

2,3,4,7,8-PeBDF was investigated together with its chlorinated counterpart and TCDD in the marmoset monkey. Six days after a single subcutaneous dose of 420 ng 2,3,4,7,8-PeBDF/kg body weight (TCDD: 50–500 ng/kg body weight; 2,3,4,7,8-PeCDF: 240–2400 ng/kg body weight), the caffeine breath test was performed *in vivo*;

Table 51. Comparative survey on CYP1A1 induction (measured as EROD activity *in vitro* or *in vivo* in liver) by 2,3,7,8-substituted TeHDDs

Species (strain)	Details	Parameter	Results[a]			Reference
			TBDD	TCDD	TBCDD	
Rat	*in vitro*: hepatoma H-4-II E cells	EC_{50} (nmol/litre)	0.235	0.080	0.055	Mason et al. (1987a); Safe (1987); Goldstein & Safe (1989); Safe et al. (1989b)
Rat (Wistar)	single intraperitoneal doses	ED_{50} (nmol/kg body weight)	0.355	3	0.347	Mason et al. (1986, 1987a); Safe (1990)
		pED_{50}[b]	9.45	8.16	n.sp.	Mason et al. (1987b)
Rat (Fischer 344)	single oral doses	ED_{50} (nmol/kg body weight)	0.8–1	n.a.	n.a.	Kedderis et al. (1992a)
Rat (Wistar)	single subcutaneous dose of 2 nmol/kg body weight	resorufin (pmol/mg protein per min) after:				Schulz-Schalge et al. (1991)
		- 7 days	6740	5210	4330	
		- 98 days	410	162	105	
Mouse ($B6C3F_1$)	multiple oral doses	relative potency	0.2	1	n.a.	Birnbaum et al. (1993)
Chick embryo	single injections	ED_{50} (pmol/egg)	9.4	11.1	7.35	Ramalingam et al. (1986)

[a] n.a. = not analysed; n.sp. = not specified.
[b] pED_{50} = -log ED_{50}.

1 day later, the animals were sacrificed, and EROD activity was determined in liver microsomes. There was a good correlation between EROD activity in hepatic microsomes and caffeine *N*-demethylation. All three compounds showed inducing capacity, and the ranking order was TCDD > 2,3,4,7,8-PeCDF > 2,3,4,7,8-PeBDF when enzyme activities were compared with the hepatic concentrations (Schulz et al., 1993).

For PXDFs, the AHH and EROD enzyme induction potencies of 8-Br_1-2,3-Cl_2DF and 8-Br_1-2,3,4-Cl_3DF were tested *in vitro* in rat hepatoma H-4-II E cells (Denomme et al., 1986). The EC_{50} values (mol/litre) were similar to those of the fully chlorinated analogues.

7.8.6.3 *Combustion products*

Pyrolysates of several brominated flame retardants, which contained relatively high levels of PBDFs/PBDDs (see also section 3), produced dose-dependent EROD and AHH activity *in vitro* (rat hepatoma H-4-II E cells) (Zacharewski et al., 1988, 1989), as well as in liver microsomes of rats (Zacharewski et al., 1988, 1989; Schmidt & Ivens-Kohl, 1990b) and guinea-pigs (Schmidt & Ivens-Kohl, 1990b) sacrificed 14 days or 4 weeks after single intraperitoneal or oral doses.

7.9 Mechanisms of toxicity — mode of action

The mechanisms underlying the diversity of biochemical and toxic effects of TCDD are being extensively studied, but they are complex and not yet fully understood. Because of similarities in chemical structure and in the pattern of responses, PBDDs/PBDFs are believed to share a common mechanism of action with PCDDs/PCDFs and other related halogenated aromatic hydrocarbons (Poland & Knutson, 1982; Goldstein & Safe, 1989; Mennear & Lee, 1994). Most information is derived from studies with TCDD, the prototypical and best examined compound of this class of chemicals. PHDDs/PHDFs act as multisite toxicants in laboratory animals and elicit species-, sex-, and tissue-dependent responses (Vanden Heuvel & Lucier, 1993). In carcinogenic processes, TCDD and related compounds function as tumour promoters (reviewed in Lucier et al., 1993b; Huff, 1994).

On a cellular basis, TCDD and related compounds cause alterations in cell proliferation and differentiation without a mutational effect on DNA (Goldstein & Safe, 1989; Silbergeld & Gasiewicz, 1989; Nebert et al., 1991; Nebert, 1994). However, there is a possibility that TCDD may influence the DNA-damaging potential of endogenous compounds (Lucier et al., 1993b). Examples of impairment of normal cellular regulatory systems by PCDDs/PCDFs have been compiled (WHO, 1989). Recently, inhibition of gap-junctional intercellular communication by TCDD was reported (De Haan et al., 1993).

At the molecular level, there is growing evidence that most, if not all, biochemical or toxic effects, including carcinogenicity, are mediated through an intracytoplasmic protein, the Ah receptor (Couture et al., 1990; Denison, 1990, 1991; Whitlock, 1990, 1993; Landers & Bunce, 1991; Lucier, 1991; Nebert et al., 1991, 1993; Safe et al., 1991; Hahn & Stegeman, 1992; Poellinger et al., 1992; Andersen et al., 1993; Birnbaum, 1993, 1994; Lucier et al., 1993a,b; Poellinger, 1993; Safe, 1993a; Vanden Heuvel & Lucier, 1993; Whitelaw et al., 1993; Okey et al., 1994; Fernandez-Salguero et al., 1996). However, although the central role of the Ah receptor in mediating TCDD toxicity is generally accepted, some findings may indicate the existence of other mechanisms operating independently of the Ah receptor (Skene et al., 1989; WHO, 1989; Holsapple et al., 1991; Nebert et al., 1993; Okey et al., 1994).

The potential of binding to the cytosolic Ah receptor was confirmed for several mono- through penta-substituted PBDDs and PXDDs (Denomme et al., 1985; Mason et al., 1987a,b; Romkes et al., 1987) and PXDFs (Denomme et al., 1986). Their receptor-binding affinities varied by several orders of magnitude (depending on structure) but were comparable to those of their PCDD analogues (Goldstein & Safe, 1989). As an example, the *in vitro* EC_{50} values for rat hepatic cytosolic receptor-binding potencies of TCDD, TBDD, and TBCDD were found to be 1.00×10^{-9}, 1.50×10^{-9}, and 1.48×10^{-9} mol/litre, respectively (Mason et al., 1987a).

Limited information is available on possible additive, synergistic (overadditive), or antagonistic interactions between different PBDDs/

PBDFs and other xenobiotics. As seen in a rainbow trout early life stage mortality bioassay (see chapter 9), selected PBDD, PBDF, and PBB congeners act additively (Hornung et al., 1996a). All kinds of interactions could be found between PCDDs, PCDFs, and PCBs (summarized by Skene et al., 1989; Safe, 1990; Hornung et al., 1996a).

There have been no studies identified detailing the possible actions of non-2,3,7,8-substituted PBDD/PBDF congeners.

7.10 Experimental data on selected PBDDs/PBDFs and their relevance to the toxicity equivalency factor (TEF) concept

The TEF method is used as a procedure to facilitate the evaluation of complex mixtures of related chemicals. The real concentrations of different congeners in a sample are changed to (by multiplying by a factor called the TEF) and expressed as uniform "toxic" concentrations, as related to a reference substance. A major application of this method is for risk management or regulatory purposes. Advantages and limitations in the use of TEFs were discussed intensively with regard to PCDDs/PCDFs (e.g. Kutz et al., 1990; Ahlborg et al., 1992; Neubert D. et al., 1992; Ahlborg, 1994; DeVito & Birnbaum, 1994; Silbergeld & deFur, 1994). For the 2,3,7,8-substituted derivatives within these compounds, a series of TEF schemes was developed by several agencies in different countries. The schemes differ in the weighting factors for certain congeners. In 1988, an internationally elaborated TEF scheme was published (NATO-CCMS, 1988), presenting the so-called international TEFs (I-TEFs) (see Table 52). It was followed by a modified proposal of Safe (1990, 1993a,b). Additionally, his concept included (besides PCBs, PBBs, and polychlorinated diphenyl ethers [PCDEs]) the PBDDs/PBDFs and PXDDs/PXDFs to which the same factors were attributed as described for the chlorinated analogues. Currently, there are no TEFs for 2,3,7,8-substituted PBDDs/PBDFs that have international agreement.

Table 52. I-TEFs for PCDDs/PCDFs

Congeners	I-TEFs[a]
2,3,7,8-TeCDD	1
2,3,7,8-TeCDF	0.1
1,2,3,7,8-PeCDD	0.5
1,2,3,7,8-PeCDF	0.05
2,3,4,7,8-PeCDF	0.5
1,2,3,4,7,8-HxCDD	0.1
1,2,3,4,7,8-HxCDF	0.1
1,2,3,6,7,8-HxCDD	0.1
1,2,3,6,7,8-HxCDF	0.1
1,2,3,7,8,9-HxCDD	0.1
1,2,3,7,8,9-HxCDF	0.1
2,3,4,6,7,8-HxCDF	0.1
1,2,3,4,6,7,8-HpCDD	0.01
1,2,3,4,6,7,8-HpCDF	0.01
1,2,3,4,7,8,9-HpCDF	0.01
1,2,3,4,6,7,8,9-OcCDD	0.001
1,2,3,4,6,7,8,9-OcCDF	0.001

[a] From NATO/CCMS (1988).

Most of the toxicologically relevant information available refers to the 2,3,7,8-substituted dibenzo-*p*-dioxin analogue pair. Data summarized in Table 53 support the concept of using corresponding TEF values for both analogues. Even if for some end-points slightly lower potencies (on a molar basis, and somewhat more pronounced on a weight basis) were found for 2,3,7,8-TeBDD than for 2,3,7,8-TeCDD, kinetic differences (especially the longer half-life of 2,3,7,8-TeBDD in adipose tissue) may favour the use of identical TEFs for 2,3,7,8-TeBDD and 2,3,7,8-TeCDD (e.g. Neubert, 1993b).

For another analogue pair, namely 2,3,7,8-TeBDF and 2,3,7,8-TeCDF, it was suggested that a higher TEF than that chosen for 2,3,7,8-TeCDF (0.1) be used, possibly around 0.3 on a molar basis or 0.2 on a weight basis (Neubert, 1993b). This proposal was also based on kinetic data — that is, the much longer elimination half-life of 2,3,7,8-TeBDF than of 2,3,7,8-TeCDF observed in rat liver (Neubert, 1993b). Additionally, 2,3,7,8-TeBDF was found to be more potent than 2,3,7,8-TeCDF in producing cleft palate and hydronephrosis in mice (Birnbaum et al., 1991 and references therein). In a fish toxicity

Table 53. A comparative survey of several biological and toxicological parameters for 2,3,7,8-TeBDD and 2,3,7,8-TeCDD (tested in parallel-running experiments)

Parameter	Details	TBDD	TCDD[a]	Reference
Kinetics				
Absorption	rat, oral route	up to 80%	–[b]	Ivens et al. (1992); Diliberto et al. (1993)
Metabolism	rat	about 7%	about 10%	Kedderis et al. (1991b)
Half-life	rat (faeces)	18 days	–[c]	Kedderis et al. (1991a)
	rat (adipose tissue)	58 days	–[d]	Kedderis et al. (1991a)
	human	3–11 years	–[e]	Zober et al. (1992)
Receptor binding (EC_{50})	incubation of cytosolic receptor protein	1.5×10^{-9} mol/litre	1.0×10^{-9} mol/litre	Mason et al. (1987b)
Microsomal enzyme induction				
Binding affinity of CYP1A2	rat liver	9.0 nmol	6.5 nmol	Kedderis et al. (1993)
AHH induction (pED_{50})[f]	rat liver	9.12	8.41	Mason et al. (1987b)
EROD induction (pED_{50})	rat liver	9.45	8.16	Mason et al. (1987b)

Table 53 (contd).

EROD induction (molar basis)	rat liver - after 7 days - after 98 days	6740 pmol resorufin/mg protein per min 410 pmol resorufin/mg protein per min	5210 pmol resorufin/mg protein per min 162 pmol resorufin/mg protein per min	Schulz-Schalge et al. (1991a,b)[g]
EROD induction (relative potency, molar basis)	rat liver	identical dose–effect and enzyme concentration curves		Nagao et al. (1995/96)
EROD induction (ED_{50})	chick embryo liver	9.4 pmol/egg	11.1 pmol/egg	Ramalingam et al. (1986)
EROD induction (relative potency, molar basis)	mouse liver (subchronic exposure)	0.2	1	Birnbaum et al. (1993); Birnbaum & DeVito (1995)
ACOH[h] induction (relative potency, molar basis)	mouse liver (subchronic exposure)	0.2	1	Birnbaum & DeVito (1995)
EROD induction (relative potency, molar basis)	mouse lung (subchronic exposure)	0.1	1	Birnbaum & DeVito (1995)
EROD induction (relative potency, molar basis)	mouse skin (subchronic exposure)	0.04	1	Birnbaum et al. (1993); Birnbaum & DeVito (1995)
EROD induction (relative potency)	mouse liver (subchronic exposure)	0.31	1	Van Birgelen et al. (1996)
ACOH induction (relative potency)	mouse liver (subchronic exposure)	0.11	1	Van Birgelen et al. (1996)

Table 53 (contd).

Parameter	Details	TBDD	TCDD[a]	Reference
Hepatic porphyrin accumulation (relative potency)	mouse liver (subchronic exposure)	0.4	1	Van Birgelen et al. (1996)
Body weight loss (pED_{50})	rat	7.17	7.28	Mason et al. (1987b)
Immunotoxicity				
Thymic atrophy (pED_{50})	rat	7.47	7.03	Mason et al. (1987b)
Changes in certain lymphocyte subpopulations (NOEL)	marmoset monkey	3 ng/kg body weight (6 pmol/kg)	1 ng/kg body weight (3 pmol/kg)	Neubert (1993a)
Developmental toxicity				
Cleft palate (ED_{50})	mouse (NMRI)	62 ± 5 µg/kg body weight (0.123 µmol/kg body weight)	24 ± 1 µg/kg body weight (0.075 µmol/kg body weight)	Nagao et al. (1990a)
	mouse (C57BL/6N)	65 µg/kg body weight	(15 µg/kg body weight)[j]	Birnbaum et al. (1991)
Hydronephrosis (ED_{50})	mouse (C57BL/6N)	9 µg/kg body weight	4 µg/kg body weight (estimated)	Birnbaum et al. (1991)

Table 53 (contd).

Tumorigenicity (short-term test: relative cell-transforming potential, molar basis)	*in vivo*/*in vitro* assay with murine macrophages	0.14	1	Massa et al. (1991)
Mortality (LD_{50}, relative potency, molar basis)	Rainbow trout (early life stage)	1.14–2.54	1	Hornung et al. (1996b)[j]

[a] – = not performed.
[b] Comparable data from other TCDD studies (WHO, 1989); 88% measured by Diliberto et al. (1996).
[c] Comparable data from other TCDD studies (references in Kedderis et al., 1991a; Weber et al., 1993).
[d] Less than values from other TCDD studies (17–25 days; references in Kedderis et al., 1991a).
[e] Comparable data from other TCDD studies (5–11 years, references cited in chapter 8).
[f] $pED_{50} = -\log ED_{50}$ (molar basis).
[g] Data in agreement with results of Abraham et al. (1988) and Nagao et al. (1990b).
[h] ACOH = acetanilide-4-hydroxylase.
[i] Measured by the same laboratory (Birnbaum et al., 1989).
[j] See chapter 9 for information about the study.

assay (see chapter 9), 2,3,7,8-TeBDF was ninefold (molar basis) more potent than 2,3,7,8-TeCDF (Hornung et al., 1996b).

Considering the few data available on the higher halogenated congeners and the wide range of potencies leading to the established TEFs, the (preliminary) use of the same TEF values for the other PBDD/PBDF or PXDD/PXDF congeners as described for the chlorinated analogues appears to be justified.

8. EFFECTS ON HUMANS

There is little information available on the effects of PBDDs/PBDFs on human health. The main human health features discussed in connection with PCDDs/PCDFs are immunotoxicity (Lorenzen & Okey, 1991; Vos, 1993), developmental toxicity (Sweeney, 1994), neurotoxicity (Peper et al., 1993), carcinogenicity (Kogevinas et al., 1993; Bertazzi & di Domenico, 1994; Hardell et al., 1994), and skin, liver, and gastrointestinal toxicity (Mennear & Lee, 1994). Recently, a brief critical review of the short- and long-term non-cancer health effects of PCDDs/PCDFs was given by Sweeney et al. (1993). The relevance of biochemical effects of PCDDs/PCDFs has also been addressed (Lucier, 1991).

8.1 General population exposure

There are no data for the general population on exposure to or effects of PBDDs/PBDFs.

8.2 Occupational/accidental exposure

Two cases of acute health problems due to 2,3,7,8-TeBDD/TeCDD exposure have been described (see also section 5.3.2). The first case refers to an American chemist who developed serious illness, including chloracne, headaches, and back and leg pain, after synthesizing TBDD/TCDD (Schecter, 1992). The second case was that of a Japanese student who suffered from very severe acne-like eruptions on his cheeks and chin following synthesis of PCDFs and PBDFs (Asahi & Urabe, 1987).

In the course of a morbidity study, male personnel of a chemical plant with documented exposure to PBDDs/PBDFs (see section 5.3) originating from the use of brominated flame retardants (OBDE and DBDE) were subjected to a general health examination, including special immunological tests (Zober et al., 1992). The measurements of the exposed (n = 21; exposed for up to 13 years) and the control (n = 42; employees of a similar resin production plant but with no use of PBDEs within the plant) groups included cellular and humoral parameters (distribution of lymphocyte subsets, concentrations of

immunoglobulins, immune complexes, complements C3 and C4, and antinuclear antibodies). Functional abnormalities of the immune system were not investigated. Only one person with the highest TBDD/TBDF burden (478/112 pg/g blood lipid) showed some changes (high concentrations of complement C4; low total lymphocyte, T cell, T helper cell, and natural killer cell counts) but did not have clinical symptoms attributable to an immunodeficiency disease. The health problems of this 54-year-old worker (hypertension, low back pain, hyperuricaemia, and signs of an old inactive tuberculosis) were thought to be not dioxin-dependent. No notable differences from the control group were found with the other participants with lower blood lipid levels (<208 pg TBDD/g blood lipid; <58 pg TBDF/g blood lipid). Evaluating the group results as a whole, complement C4 concentrations increased significantly with increasing concentrations of both TBDD and TBDF. Marginal associations independent of the person with the highest exposure were seen with C3 concentrations (increased with TBDF concentrations). However, retesting of the one person at a later date showed that his C4 levels had dropped. Altogether, the effects observed were not considered to be indicative of an impact of PBDDs/PBDFs on the immune system.

A shortcoming of this study was a lack of measurements of PBDD/PBDF concentrations (and concentrations of related compounds, such as PCDDs/PCDFs or PCBs) in the control group. In addition, the internal exposure was related to only two congeners, namely TBDD and TBDF, although a lot of other homologues were identified at the corresponding workplace (see section 5.3.1). This was partly due to analytical limitations (lack of reference substances) and partly due to the high weighting factor of TBDD in toxic equivalent (TEQ) calculations.

Additional clinical laboratory tests (liver function tests, lipid and glucose measures, thyroid function parameters, haematological and coagulation indicators) were performed within the same study cohort. They did not reveal any remarkable differences between the exposed (n = 38–42) and the control (n = 40–42) groups (Ott & Zober, 1996).

Although there are indications for excesses of cancer mortality seen in workers exposed to TCDD (e.g. Zober et al., 1990; Fingerhut et al., 1991; Manz et al., 1991; Becher et al., 1996), there are no reports of cancer mortality caused by PBDDs/PBDFs.

8.3 Subpopulations at special risk

No data are available with regard to the effects of PBDDs/PBDFs on high-risk subpopulations.

As discussed for PCDDs/PCDFs (Helge, 1993), fetuses, newborn babies, and children are at a higher risk of exposure for several reasons (placental and milk transfer, contact with soil and dust, immunological and physiological immaturity, lack of fat depots, etc.), and developing systems, based on experimental evidence, may be more sensitive to the adverse effects of halogenated aromatic compounds than developed systems of adults.

9. EFFECTS ON OTHER ORGANISMS IN THE LABORATORY AND FIELD

There are limited reports on the effects of PBDDs/PBDFs on microorganisms, plants or invertebrate species. Regarding vertebrates, data from a fish early life stage mortality bioassay are available (Hornung et al., 1996a,b).

Newly fertilized rainbow trout eggs were injected with a series of PBDDs, PXDDs, and PBDFs (see Table 54). All congeners tested but 2,7-DiBDF caused mortality in the rainbow trout sac fry by a blue sac syndrome. The signs of toxicity were identical to those produced by TCDD and included yolk sac oedema, pericardial oedema, multifocal haemorrhages, reduced growth, and craniofacial malformations. 2,3,7,8-TeBDD showed the highest potency among the brominated congeners and was also more potent than 2,3,7,8-TeCDD. Similarly, 2,3,7,8-TeBDF was more potent than 2,3,7,8-TeCDD, whereas the other PBDDs/PBDFs were equipotent (or less potent) than identically substituted PCDDs/PCDFs (Hornung et al., 1996b).

The interactions between pairs of PBDD and PBDF congeners were also studied using the rainbow trout sac fry (early life stage) mortality bioassay (Hornung et al., 1996a). The tested pairs were as follows: 2,3,7,8-TeBDD and 1,2,3,7,8-PeBDF; 2,3,7,8-TeBDD and 1,2,3,7,8-PeBDD; and 1,2,3,7,8-PeBDF and 1,2,3,7,8-PeBDD. As with the individual PBDD and PBDF congeners, their mixtures also produced TCDD-like toxicity and mortality in the rainbow trout sac fry. The rank order for LD_{50} in the individual congeners tested, from lowest to highest, was as follows: 2,3,7,8-TeBDD < 1,2,3,7,8-PeBDD < 1,2,3,7,8-PeBDF. The interactions between each of the tested pairs were additive in causing sac fry mortality.

Laboratory studies with the related PCDDs/PCDFs showed adverse effects on fish and avian species, especially on the early life stages (Cook et al., 1991; Walker & Peterson, 1991, 1994; Peterson, 1993; Peterson et al., 1993). Correlations between environmental exposure to polyhalogenated aromatics (PCDDs/PCDFs, PCBs) and the decline of some populations of marine and freshwater mammals, fish, and fish-eating birds or symptoms such as reproductive dysfunc-

Table 54. Early life stage mortality in rainbow trout (*Oncorhynchus mykiss*) caused by PBDDs, PXDDs, and PBDFs[a]

Congener	Blue sac syndrome[b]	LD_{50}[c] (ng/g egg)	Rainbow trout strain
Dibenzo-*p*-dioxins			
2,3,7-TrBDD	+	18.9	Erwin
	+	15.6	McConoughy
2,3,7,8-TeBDD[d]	+	0.222	Eagle Lake
	+	0.264	Eagle Lake
	+	0.158	Erwin
	+	0.122	Arlee
1,3,7,8-TeBDD	+	29	Erwin
1,2,3,7,8-PeBDD	+	4.16	Eagle Lake
	+	4.92	Erwin
1,2,3,4,7,8-HxBDD	+	63.7	Arlee
2,8-Cl_2-3,7-Br_2-DD	+	0.448	Erwin
2,3,7-Cl_3-8-Br_1-DD	+	0.410	Erwin
Dibenzofurans			
2,7-DiBDF	-	>597[e]	Erwin
2,3,7,8-TeBDF	+	1.5	Erwin
2,3,4,7,8-PeBDF	+	6.19	Erwin
1,2,3,7,8-PeBDF	+	9.56	Erwin
1,2,3,4,7,8-HxBDF	+	247	Erwin

[a] Modified from Hornung et al. (1996b).

[b] TCDD-like toxicity grossly identical to blue sac syndrome was characterized by sac fry mortality that was preceded by yolk sac oedema, pericardial oedema, multifocal haemorrhages, growth retardation, and craniofacial malformations.

[c] Based on cumulative hatching and sac fry mortality (for fiducial limits, see original); eggs (n = 30 per dose) injected with seven graded doses of congener incorporated into phosphatidylcholine liposomes.

[d] For comparison: LD_{50} values for 2,3,7,8-TeCDD: 0.171 ng/g egg (Shasta strain); 0.374 ng/g egg (Arlee strain) (Walker & Peterson, 1991; Zabel et al., 1995).

[e] No signs of toxicity occurred at the highest egg dose tested.

tion or other lesions were investigated in a number of field studies (Gilbertson, 1989; Rappe, 1993).

Recent results of the research on the effects of dibenzo-*p*-dioxins and dioxin-like compounds on wildlife support the conclusion that there are real-world adverse effects on wildlife caused by these compounds. Although PCDD/PCDF have contributed to these effects, in most locations the major effects are due to the dioxin-like non- and mono-ortho-substituted PCB congeners. Effects have been reported in mammals and birds as well as in fish. While the current effects are subtle, there is no assimilative capacity for TEQs in the global environment. Trends in TEQs in industrialized regions are continuing to decline. Concentrations of TEQs in remote areas, such as the Arctic and open ocean, may not yet have reached their maximum. The contribution of PBDDs/PBDFs to these effects is unknown (Giesy et al., 1994; Fiedler & Van den Berg, 1996).

10. EVALUATION OF HUMAN HEALTH RISKS AND EFFECTS ON THE ENVIRONMENT

There is much less information on PBDDs/PBDFs than on their chlorinated analogues. The analytical methods for separating and identifying the individual brominated congeners are less advanced than those for their chlorinated analogues, and only a few standards are available. Present analytical methods are able to quantify total brominated homologue groups and are able to detect but not quantify the mixed brominated/chlorinated congeners.

10.1 Hazard evaluation

Data on bioconcentration and biomagnification are lacking. The physicochemical properties of PBDDs/PBDFs and extrapolations from PCDDs/PCDFs suppose that PBDDs/PBDFs are enriched in carbon- and fat-rich environmental compartments in a manner similar to PCDDs/PCDFs. Atmospheric transport occurs in the vapour and particulate phases, the ratio depending on molecular weight.

Photochemical degradation of PBDDs/PBDFs and PXDDs/PXDFs adsorbed on surfaces was much slower than that in organic solvents. Under environmental conditions, debromination was found to be a slow process (several months in soil).

From the available data, it is apparent that PBDDs/PBDFs have a toxic potential similar to that of PCDDs/PCDFs.

Generally, the studies performed with 2,3,7,8-substituted PBDDs/PBDFs showed typical TCDD-like effects in experimental animals and a persistence in animal and possibly human tissues apparently similar to that seen with PCDDs/PCDFs. For some end-points, quantitative differences were found. For example, under certain experimental conditions, 2,3,7,8-TeBDF was more potent than 2,3,7,8-TeCDF in inducing cleft palate and hydronephrosis in mice following prenatal exposure. 2,3,7,8-TeBDD was somewhat less potent than its chlorinated counterpart with regard to short-term toxicity, cell-transforming capacity, antiestrogenic activity, and decreasing vitamin A storage. Both brominated congeners had longer elimination half-

lives in rat adipose tissues than the chlorinated ones. More or less comparable activities of these and some other PBDDs/PBDFs with their chlorinated analogues were seen for receptor binding, enzyme induction, and immunotoxic effects.

The few data available on trisubstituted PBDDs suggest a higher metabolic rate and lower short-term toxicity compared with the 2,3,7,8-substituted congeners. Limited data are available for other non-2,3,7,8-substituted PBDD/PBDF congeners. Studies with 1,2,7,8-TeBDF demonstrated higher rates of metabolic elimination in rats compared with 2,3,7,8-TeCDF or 2,3,7,8-TeBDD.

Long-term toxicity studies as well as perinatal exposure and multigeneration studies are also lacking for the 2,3,7,8-substituted PBDDs/PBDFs. Nevertheless, the possibility of adverse effects — for example, carcinogenicity, reproductive/developmental toxicity, and neurotoxicity — cannot be excluded, based on the similarity in results obtained in short-term studies with PBDDs/PBDFs compared with PCDDs/PCDFs. However, in a comparison of relative potencies, 2,3,7,8-TeCDD was approximately two times more potent than 2,3,7,8-TeBDD in inducing the accumulation of total hepatic porphyrin in mice following subchronic (13-week) exposure.

Congeners with a high degree of bromination (>penta) were usually not included in toxicity studies (owing to a lack of pure congeners), although they were detectable in workplace air and other samples.

Exposure to single congeners of PBDDs/PBDFs and PXDDs/PXDFs has resulted in TCDD-like toxic effects and mortality in the rainbow trout sac fry early life stage mortality bioassay. Pairs of PBDDs/PBDFs have been shown to have additive effects on the mortality of rainbow trout fry. In these bioassay studies, single exposures to 2,3,7,8-TeBDD and 2,3,7,8-TeBDF were more potent than single exposures to the chlorinated analogues.

There are no pertinent data on microbial degradation. Photochemical debromination is thought to be the major transformation process for PBDDs/PBDFs and PXDDs/PXDFs. As was shown in

laboratory experiments, PBDDs/PBDFs and PXDDs/PXDFs were degraded in organic solvents after irradiation with sunlight or UV light. The easier loss of bromine than of chlorine resulted in the formation of the PCDDs/PCDFs when PXDDs/PXDFs were exposed to light.

10.2 Exposure evaluation

As PBDDs/PBDFs are not known to occur naturally, their presence is indicative of thermal or photolytic degradation or thermal transformation of brominated chemicals, many of which are used as flame retardants. The occurrence of such chemicals in consumer products (e.g. electrical appliances, textiles, petrol) may, in some cases, result in a risk for the population.

The database on the occurrence of PBDDs/PBDFs in ambient air and dust is too small for detailed comparisons with PCDDs/PCDFs and estimations of their impact on human health. Only one 2,3,7,8-substituted PBDF congener was detected in ambient air. Automobile exhaust was found to be a diffuse source of PBDDs/PBDFs and PXDDs/PXDFs (along with PCDDs/PCDFs), if halogenated scavengers are used in the petrol.

Because typical municipal waste generally contains much more chlorine than bromine, the formation of mixed and completely chlorinated compounds from brominated precursors is possible and was confirmed experimentally. Modern waste incinerators are capable of realizing very low emissions of PCDDs/PCDFs; however, this modern technology is not applied in all countries.

Emissions (and effluents) of PBDDs/PBDFs into the environment from plants (processing organic chemicals, recycling plastics and metals), waste incinerators, etc. are hardly documented, but may vary depending on the industrial hygienic standards (including waste disposal practices). A limit value of 0.1 ng TEQ/m^3 for PCDDs/PCDFs in exhaust gases of municipal waste incinerators was established by some countries.

A critical source for release of PBDDs/PBDFs and PXDDs/PXDFs into the environment is accidental fires of materials containing brominated compounds. Concentrations measured depend on the specific fire situation and can be very high. The highest concentrations are found in the solid residues.

PBDDs/PBDFs have been found in indoor air (maximum PBDF sum concentration, up to hepta: 1.27 pg/m^3) and dust samples of rooms equipped with electronic appliances and in house dust. Whereas 2,3,7,8-substituted congeners were not detected in the air samples, they were detectable in dust samples (tetra- to pentaBDFs: up to 0.07 μg/kg). Dust collected in computer rooms contained comparable amounts of PBDDs/PBDFs and PCDDs/PCDFs (e.g. maximum sum concentrations: each about 5 μg/kg).

Potential exposure pathways for the general population, clean-up personnel, and fire personnel may originate from accidental fires where bromine-containing plastics are involved. Gas, smoke, and residue samples as well as firemen's trousers were found to contain PBDDs/PBDFs (mainly PBDFs), including a portion of up to 20% 2,3,7,8-substituted congeners, and PXDDs/PXDFs. Generally, the analyses of fire incidents showed a wide variation of contamination in the μg/kg (residue), ng/m^2 (smoke condensate), or ng/m^3 (gas) range, with very high peak concentrations possible. In several residue samples, the sum concentrations of seven 2,3,7,8-substituted congeners (2,3,7,8-TeBDF, 2,3,4,7,8-PeBDF, and five tetra- to hexaBDDs) were greater than 5 μg/kg. In one television fire incident, the sum concentrations of these congeners in residues amounted to 1100 μg/kg. Additionally, this sample contained high concentrations of 1,2,3,7,8-PeBDF (1860 μg/kg) and of 1,2,3,4,7,8-HxBDF (1900 μg/kg). PBDF concentrations measured at experimental fires generally were higher than those measured at real fires and may be considered as worst-case examples. The area contaminant concentrations of PBDFs resulting from experimental or real television fires were in a similar range as found for PCDDs/PCDFs after fires involving their precursors. Fires involving televisions or computers may produce higher PBDD/PBDF than PCDD/PCDF concentrations — for example, maximum sum concentrations of 5600 μg/kg (13 μg

TEQ[a]/kg) versus 320 μg/kg (2.3 μg I-TEQ/kg) in soot samples after a fire in a computer room.

Potential issues of concern are fires and suspected leaching processes at waste disposal sites (additional presence of organic solvents and other contaminants).

One study reported on the analysis of PBDDs/PBDFs and PXDDs/PXDFs in a sample of muscle from a salmon from the Baltic Sea and a pooled sample of human milk from Sweden. The concentrations were found to be below the detection limit (0.3 ng 2,3,7,8-TeBDF/kg and 0.4 ng 2,3,7,8-TeBDD/kg), suggesting a very low exposure to PBDDs/PBDFs and PXDDs/PXDFs. On the other hand, a series of 2,3,7,8-substituted PCDDs/PCDFs were detected in these samples.

Workplaces identified as involving a risk of exposure to PBDDs/PBDFs (and PXDDs/PXDFs) include mainly those in the plastic and recycling industry using brominated flame retardants or products containing them and those of firemen and clean-up personnel associated with fires.

Recent monitoring data from three plastic plants showed concentrations between 260 and >10 000 pg/m^3 (sum of mono- to hexaBDDs/BDFs). Concentrations of eight 2,3,7,8-substituted congeners (three tetra- to pentaBDFs and five tetra- to hexaBDDs) ranged from 0.11 to 18 pg TEQ[b]/m^3 at permanently operated workplaces. At periodically operated workplaces (maximal stay: 1 h/day), a maximum concentration of the eight congeners of 954 pg TEQ[b]/m^3 has been measured. There are no occupational threshold limits for PBDDs/PBDFs.

PBDD/PBDF concentrations currently measured in plastics, recycled products, electronic scrap, and other waste samples were

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[a] Calculated by Schacht et al. (1995) using PCDD/PCDF I-TEFs for the brominated congeners.

[b] Calculated by Kieper (1996) using PCDD/PCDF I-TEFs for the brominated congeners.

considerable. In view of the growing worldwide production and use of brominated flame retardants (estimated worldwide demand in 1992: 150 000 tonnes; OECD, 1994) as additives to a series of polymers, it can be assumed that the amount of bromine-containing waste will be increasing in the future. In particular, electronic scrap from casings and printed circuit boards of computers, etc., flame-retarded with brominated compounds will reach the waste streams and then be a potentially major source of PBDDs/PBDFs (and PXDDs/PXDFs and PCDDs/PCDFs).

A potential hazard can arise in chemical research laboratories performing special syntheses. Distillation residues, other wastes, and equipment were found to be contaminated by PBDDs/PBDFs and/or PXDDs/PXDFs.

Few human monitoring data are available. One study published blood monitoring data of personnel from an industry using PBDEs. The analyses showed (1) uptake of PBDD/PBDF congeners, (2) the presence of 2,3,7,8-substituted congeners, which were not or hardly detectable in the corresponding workplace air samples, and (3) estimated half-lives typical for dioxin-like compounds.

The main route of human exposure to PCDDs/PCDFs for the general population is via food intake (more than 95%). Whereas the presence of PCDDs/PCDFs was confirmed in most foodstuffs, PBDDs/PBDFs (≥tetra) could only be detected but not quantified in shellfish, fish, and cow's milk. 2,3,7,8-Substituted congeners could not be identified.

For the general environment, it was found that the concentrations of PBDDs/PBDFs are much lower than those of the chlorinated analogues (tetra- through octahalogenated), the homologues with four or more bromine atoms being hardly detected in environmental samples. At present, there are only few data, but these data do not indicate an accumulation of PBDDs/PBDFs or PXDDs/PXDFs along the terrestrial or the aquatic food-chain. However, the limited data set does not allow for environmental trend analysis.

Currently, most PBDDs/PBDFs and their precursors are bound into products and have not yet reached waste streams and the environment.

10.3 Risk evaluation

PBDDs/PBDFs have been detected in air, dust, soil, sediment, sewage sludge, grass, and fish, but not in the general human population. This limited occurrence is in contrast to the ubiquitous PCDDs/PCDFs. Generally, concentrations measured were low.

The major risk group for exposure to PBDDs/PBDFs has apparently been workers involved in the production and application of brominated flame retardants (e.g. extruder personnel). In these persons, clearly increased body burdens of 2,3,7,8-TeBDD and 2,3,7,8-TeBDF have been measured. Another possible source of exposure may be automobile exhaust, as a result of incomplete combustion of bromine-containing materials. No data on increased body burdens have been reported (and apparently body burdens have not been measured). For the general population, there seems to be a very low risk of exposure, compared with the risk of exposure to PCDDs/PCDFs. In the few samples measured (breast milk), no PBDDs/PBDFs were detected, whereas at least 100 times higher PCDD/PCDF levels were present.

Within the group of workers with clearly increased PBDD/PBDF body burdens, no clinical adverse health effects were reported, and only a few data for laboratory values of the volunteers were outside the normal reference range. Although no exact data on elimination half-lives have been reported for humans, the data available indicated a considerable persistence of the 2,3,7,8-TeBDD/TeBDF congeners within the human organism.

From all the information currently available, it can be concluded that the potential of the PBDDs/PBDFs for biological (e.g. enzyme induction) and toxic actions is very similar to that of the PCDDs/PCDFs.

It is difficult to compare the potency of PBDDs/PBDFs with that of their chlorinated analogues, as the database for the individual PBDDs/PBDFs is very small, with respect to both data from animal studies and observations in humans. With a few exceptions in most of the systems tested, the few evaluated PBDD/PBDF congeners were less potent than the corresponding chlorinated congeners. Kinetically, however, some of the brominated substances exhibited a higher persistence within the mammalian organism compared with the corresponding chlorinated substances. Therefore, as judged from data on 2,3,7,8-TeBDD, a TEF of 1.0 (equal to that of TCDD) would represent a conservative approach. For 2,3,7,8-TeBDF, a somewhat higher TEF may be suggested, as the substance has been shown in rodents to have a clearly longer elimination half-life compared with 2,3,7,8-TeCDF; a TEF of 0.2–0.3 may be justified.

The current limited experimental database does not allow a complete hazard assessment and the recommendation of a safe level of exposure to PBDDs/PBDFs for the general population.

However, if a comparison of the health impact of PBDDs/PBDFs with their chlorinated analogues is needed, the data published on 2,3,7,8-TeBDD may be taken as an example. For 2,3,7,8-TeBDD, a NOAEL of 10 ng/kg body weight per day may be established in a 13-week study in rats. This value compares with the NOAEL of 10 ng/kg body weight per day for 2,3,7,8-TeCDD, as derived from the 13-week study in rats.

The very limited data set available for concentrations of PBBDs/PBDFs in environmental compartments makes it impossible to conduct a proper risk evaluation for the environment. However, the few data on levels of these substances indicate that they are much lower than levels of their chlorinated counterparts. The assumption that both brominated and chlorinated dibenzo-*p*-dioxins and dibenzofurans act through a common mechanism and that their potency is not greatly dependent on the nature of the halogen atom (chlorine or bromine) will lead to the conclusion that the PBDDs/PBDFs will contribute marginally to the total "dioxin" effect.

11. CONCLUSIONS AND RECOMMENDATIONS FOR PROTECTION OF HUMAN HEALTH AND THE ENVIRONMENT

11.1 Conclusions

PBDDs/PBDFs are contaminants that are more or less similar to PCDDs/PCDFs in their persistence and toxicity. Therefore, humans and the environment should be protected from them. Exposure of the general population to PCDDs/PCDFs appears to be greater than exposure to PBDDs/PBDFs. Limited biomonitoring information indicates very low residues, compared with PCDDs/PCDFs. Brominated flame retardants and their precursors appear to be a main source of PBDDs/PBDFs.

A limited experimental database exists for PBDDs/PBDFs and would therefore exclude an attempt at a complete hazard identification. Current information does not allow a quantitative risk assessment, although toxicological similarities appear to exist between certain PBDD/PBDF congeners and their corresponding chlorinated homologues. On an interim basis, it is suggested that current I-TEFs for the 17 2,3,7,8-substituted PCDD/PCDF congeners be applied to the comparable brominated and mixed halogenated congeners.

11.2 Recommendations

Owing to the accumulating and toxic potential of some PBDDs/PBDFs, every effort should be made to prevent exposure of humans to, and pollution of the environment by, these compounds.

Brominated flame retardants should not be used where suitable replacements are available, and future efforts should encourage the development of further substitutes.

Appropriate precautions, including monitoring, should be taken both to protect workers from exposure to PBDDs/PBDFs and to prevent their release into the environment in emissions and effluents.

Disposal of industrial wastes, fire residues, and consumer products containing brominated compounds should be controlled to minimize environmental contamination by PBDDs/PBDFs and their precursors. All products flame-retarded with bromine compounds should be labelled and disposed of only in properly constituted waste incinerators working at consistent operating conditions, to avoid the release of PBDDs/PBDFs.

The use of leaded petrol, which necessitates the use of halogenated scavengers, should be avoided.

Selected PBDD/PBDF congeners (2,3,7,8-TeBDD/TeBDF) should be included in ongoing dioxin monitoring programmes to enhance the existing database.

12. FURTHER RESEARCH

Analytical methods, including screening techniques, should be improved. Interlaboratory comparisons should be undertaken to validate methodologies.

As the experimental database is limited, comparative toxicological and ecotoxicological studies with selected PBDD/PBDF congeners should be performed with respect to both identifying appropriate adverse- and no-adverse-effect levels and improving the interim TEF recommendation.

13. PREVIOUS EVALUATIONS BY INTERNATIONAL BODIES

There have been no previous evaluations by international bodies.

(A toxicological evaluation of PBDDs/PBDFs was prepared by the German Federal Health Office [Appel, 1991, 1993].)

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APPENDICES

APPENDIX I: Directives/Test Rules Concerning PBDDs/PBDFs

Table 2 (in chapter 2) gives the PBDD/PBDF congeners substituted with bromine in the 2,3,7,8-positions. As these are the most toxic congeners, in some investigations only these congeners are determined. The German Dioxin Directive (1994) has established temporary and permanent limitations on the concentrations of certain 2,3,7,8-substituted PBDDs/PBDFs in products brought to the German marketplace (see also Anon, 1996). For the first 5 years (i.e. until 15 July 1999), the sum concentration of the four PBDD/PBDF congeners listed in category A must be less than 10 ppb, and the sum concentration of the eight congeners in categories A and B must be less than 60 ppb. After this time, the limits become 1 ppb and 5 ppb, respectively. Restrictions on PCDDs/PCDFs are already established.

Category A	*Category B*
2,3,7,8-TeBDD	1,2,3,7,8-PeBDF
2,3,7,8-TeBDF	1,2,3,4,7,8-HxBDD
1,2,3,7,8-PeBDD	1,2,3,6,7,8-HxBDD
2,3,4,7,8-PeBDF	1,2,3,7,8,9-HxBDD

In 1987, the US EPA issued a Test Rule requiring manufacturers and importers of certain halogen-containing chemicals to analyse their products for 2,3,7,8-substituted PHDDs/PHDFs. Specific limits of quantitation and analytical requirements concerning recoveries and precision were included in the Test Rule. Among the chemicals tested were decabromodiphenyloxide and TBBPA.

Analyte	*Limit of quantitation (ppb)*
2,3,7,8-TeBDD	0.1
1,2,3,7,8-PeBDD	0.5
1,2,3,4,7,8-HxBDD	2.5
1,2,3,6,7,8-HxBDD	2.5
1,2,3,7,8-HxBDD	2.5
1,2,3,4,6,7,8-HpBDD	100
2,3,7,8-TeBDF	1
1,2,3,7,8-PeBDF	5

2,3,4,7,8-PeBDF	5
1,2,3,4,7,8-HxBDF	25
2,3,4,6,7,8-HxBDF	25
1,2,3,7,8,9-HxBDF	25
1,2,3,4,6,7,8-HpBDF	1000
1,2,3,4,7,8,9-HpBDF	1000

APPENDIX II: Apparatuses and Terminology Used in Thermolysis Experiments

Apparatuses used for thermolysis experiments are given below:

DIN oven: According to German standard DIN 53436. Open horizontal quartz tube (1 m long, 4 cm in diameter) with a gas flow in one direction. The sample (mg to a few g) is placed on a quartz plate within the tube, and a ring oven moves outside along the tube (1 cm/min). The design of the DIN apparatus simulates real fire situations ranging from smouldering to open flame.

BIS oven: Same as DIN apparatus but stationary oven.

VCI oven: Vertical combustion apparatus with two heating zones. The oven is heated to a defined temperature, and then the sample (approximately 50 mg) is dropped via a valve into the combustion zone. During the experiment, an air flow is maintained. Residence times range from a few seconds up to 10 min. Owing to the small sample size, the limit of quantification is relatively high. The VCI apparatus simulates an instant fire.

In general, the combustion gases are adsorbed on XAD resin, and the solid residues can be analysed for PBDDs/PBDFs as well. Typical temperature ranges for all experiments are from 300 to 800 °C.

Quartz tube: The sample is sealed into a quartz tube and then heated to a defined temperature.

Terms referring to thermal treatment are used as follows:

Thermolysis: General term for treating materials, compounds, etc. at elevated temperatures. No specification is given for oxygen content, technology, etc.

Combustion: General term for thermal treatment, more specifically in the presence of oxygen or air.

Incineration: Refers to large-scale plants as used, for example, in the incineration of municipal solid waste, hazardous waste, sewage sludge, clinical waste, etc.

Pyrolysis: Thermal treatment under exclusion of oxygen/air.

RÉSUMÉ

1. Identité, propriétés physiques et chimiques et méthodes d'analyse

Les polybromodibenzo-*p*-dioxines (PBDD) et les polybromodibenzofuranes (PBDF) sont des composés aromatiques de structure quasiment plane. Il peut y avoir théoriquement 75 PBDD et 135 PBDF. En outre, un grand nombre de dérivés halogénés mixtes — 1550 bromo/chloro dibenzo-*p*-dioxines (PXDD) et 3050 bromo/chloro dibenzofuranes (PXDF) — sont également envisageables sur le plan théorique. En raison de la complexité des méthodes d'analyse et de la rareté des substances de référence utilisables à des fins analytiques, il n'est possible de rechercher et de doser qu'un petit nombre de ces composés. Les dérivés les plus toxiques sont ceux qui sont substitués en position 2, 3, 7 et 8. Il existe ainsi 7 PBDD et 10 PBDF substitués en 2, 3, 7 et 8 et on peut également envisager 337 PXDD et 647 PXDF substitués aux mêmes positions.

Les PBDD et les PBDF ont une masse moléculaire plus élevée que celle de leurs homologues chlorés. Leur point de fusion est élevé et leur tension de vapeur est faible, tout comme leur solubilité dans l'eau. Ils sont généralement solubles dans les graisses, les huiles et les solvants organiques. On ne possède que très peu de données expérimentales sur les propriétés physiques et chimiques de ces composés.

Les PBDD et les PBDF sont photolysés plus rapidement que les dibenzo-*para*-dioxines polychlorées (PCDD) et les dibenzofurannes polychlorés (PCDF). Ils présentent une bonne stabilité à la chaleur. Les températures de formation ou de destruction des PBDD/PBDF dépendent d'un certain nombre de facteurs, entre autres, la présence ou l'absence d'oxygène, de polymères ou d'additifs retardateurs de flamme comme le trioxyde d'antimoine (Sb_2O_3).

En présence d'un excès de chlore celui-ci se substitue au brome pour donner des PXDD/PXDF.

En raison du caractère toxique de ces composés et de leur sensibilité à la photolyse, des précautions sont en prendre lors de l'échantillonnage et de l'analyse. Il faut en particulier faire appel à des

méthodes extrêmement sensibles, sélectives et spécifiques (comme la chromatographie en phase gazeuse couplée à la spectrométrie de masse) en raison du nombre très élevé de dérivés. Les méthodes d'échantillonnage sont identiques pour toutes les dibenzo-*p*-dioxines et tous les dibenzofuranes polyhalogénés (PHDD et PHDF), toutefois la séparation et le dosage des PBDD et des PBDF (et des PXDD et des PXDF) sont un peu différents de ceux de leurs homologues chlorés. Les PBDD/PBDF ont une masse moléculaire plus élevée et un temps de rétention chromatographique plus long que leurs homologues chlorés. En outre, la configuration des motifs isotopiques observés en spectrométrie de masse et les interférences sont également différents. Le très petit nombre de substances de référence limite considérablement les possibilités d'identification des dérivés polybromés. Pour la même raison, la recherche et le dosage des dérivés halogénés mixtes sont quasiment impossibles.

2. Formation et sources d'exposition humaine et environnementale

On ne connaît pas de PBDD/PBDF d'origine naturelle. On ne les produit pas non plus délibérément (sauf à des fins scientifiques): ce sont en général les sous-produits involontaires de divers processus. Ils peuvent se former au cours de réactions chimiques, photochimiques ou thermiques à partir d'un certain nombre de précurseurs ou être synthétisés *de novo*.

Ils sont présents à l'état d'impuretés dans divers dérivés organiques bromés, comme les bromophénols et en particulier, dans les retardateurs de flamme, comme les polybromodiphényléthers (PBDE), le décabromodiphényle (décaBB ou DBB), le 1,2-bis(tribromophénoxy)éthane, le tétrabromobisphénol A (TBBPA) etc. On en a mis en évidence dans des résidus de distillation de divers bromophénols et bromoanilines ainsi que dans des déchets de laboratoires de chimie.

On a décelé la présence de PBDF et, en moins grande quantité, de PBDD, dans les produits de photodécomposition de composés organiques bromés tels que les PBDE et les bromophénols.

Les essais de thermolyse effectués en laboratoire ont mis en évidence la formation de PBDD/PBDF à partir de bromophénols, de PBDE, de polybromobiphényles (PBB) et autres dérivés bromés utilisés comme retardateurs de flamme (à l'état pur ou dans une matrice polymère). Les rendements obtenus variaient dans de larges proportions, allant de zéro à des valeurs maximales de l'ordre du g/kg (dans le cas des PBDE). En général, il y avait beaucoup plus de PBDF que de PBDD. On a constaté que pour une série de retardateurs de flamme, la température optimale de formation des PBDF se situait entre 600 et 900 °C. La présence de polymères ou de synergisants (par ex. Sb_2O_3) a eu pour effet de diminuer la température de optimale de formation, la ramenant aux alentours de 400 °C. Outre la température et la présence d'une matrice polymère ou de synergisants, d'autres facteurs tels que les métaux ou oxydes métalliques, l'eau, l'oxygène et le système de combustion utilisé ont également eu une influence sur la nature et la proportion des PBDD ou PBDF obtenus. Dans le cas de mélanges ternaires constitués d'un PBDE, d'une matrice polymère et de Sb_2O_3, ce sont des tétrabromodibenzofuranes (tétraBDF ou TeBDF) que l'on a surtout obtenus. On a trouvé des 2,3,7,8-PBDD/PBDF (tétra et hepta) à diverses concentrations; par exemple du 2,3,7,8-TeBDF à des concentrations allant jusqu'à 2000 mg/kg dans les produits de pyrolyse de polymères contenant de l'octabromodiphényléther (octaBDE ou OBDE).

Dans l'industrie des matières plastiques, des températures élevées (150–300 °C) peuvent être atteintes au cours de divers processus. L'étude des vapeurs qui s'échappent des presses à injecter, extrudeuses etc. lors du thermoformage de plastiques tels que les résines ABS (acrylonitrile-butadiène-styrène) ou PBT (téréphtalate de polybutylène) contenant divers retardateurs de flamme bromés, montre que des PBDD/PBDF peuvent se former à ces températures. Ce sont l'OBDE et le décabromodiphényléther (décaBDE ou DBDE) qui donnent naissance aux plus grandes quantités de PBDD/PBDF, les PBDF étant les plus abondants. La quantité de TBBPA ou de TBPI (bis-tétrabromophtalimide-éthylène) sont beaucoup plus faibles (de plusieurs ordres de grandeur). Lors du thermoformage de résines ABS contenant du bromostyrène ou du 1,2-bis(tribromophénoxy)éthane comme retardateurs de flamme, on n'a pas décelé de PBDD/PBDF. En ce qui concerne les autres homologues substitués en 2,3,7,8, soit on ne les a pas recherchés (en présence de DBDE), soit on en a trouvé des

traces (en présence d'OBDE), soit on n'est pas parvenu à les mettre en évidence (en présence de TBBPA et de TBPI).

On a analysé divers plastiques à différents stades de leur mise en forme, à la recherche de PBDD/PBDF. Il s'agissait soit de poudres à mouler sous forme de granulés soit de pièces moulées dont on connaissait les additifs retardateurs de flamme ainsi que de divers objets de l'électroménager gris ou brun (téléviseurs, imprimantes, ordinateurs) dont on ignorait quels additifs ils pouvaient contenir. C'est dans les produits contenant des PBDE comme retardateurs de flamme que l'on a trouvé la plus grande quantité de PBDD/PBDF. Ces teneurs étaient de l'ordre de plusieurs milliers de µg/kg, c'est à dire qu'elles dépassaient de plusieurs ordres de grandeur la concentration observée dans les autres systèmes polymère/retardateur de flamme. Les quantités formées dépendaient de la température et de la durée des divers processus: adjonction de l'additif, extrusion ou moulage par injection. Là encore et à quelques exceptions près, ce sont les PBDF qui prédominaient par rapport aux PBDD, les dérivés les plus substitués (plus de quatre bromes) étant présents en abondance. Ce sont les pentabromodibenzofuranes (pentaDBF ou PeDBF) et les hexabromodibenzofuranes (hexaBDF ou HxBDF) dont les concentrations étaient les plus élevées. Dans des gainages de plastique, la concentration de ces derniers atteignait 3000 µg/kg. Les supports de circuits imprimés contenaient des tétra- et pentaBDF aux concentrations maximales respectives de 1300 et 1400 µg/kg. La concentration totale en PBDF (mono à hexa) se situait dans les limites de 3,6 à 3430 µg/kg. Les PBDF substitués en 2,3,7,8 n'ont pas été dosés ou bien n'étaient pas décelables ou encore étaient présents à des concentrations trop faibles. La concentration maximale des PBDF substitués en 2,3,7,8 (tétra à hexa) dans des gainages et des supports de circuits imprimés allait de 11 µg/kg (tétra) à 203 µg/kg (hexa).

Les mesures effectuées pour déterminer si des PBDF sont libérés par les téléviseurs et autres appareils de ce genre pendant leur fonctionnement ont montré que la concentration de ces composés dans l'air allait de zéro (non décelable) à 1800 pg de PBDF totaux (tétra à hexa) par appareil.

La combustion de produits contenant des composés bromés provoque un dégagement de PBDD/PBDF. Lors d'essais au cours

desquels on avait reproduit les conditions d'un véritable incendie, on a constaté que des appareils électriques tels que téléviseurs, imprimantes, terminaux d'ordinateurs et leurs gainages ou boîtiers laissaient des résidus contenant de fortes concentrations de PBDF (mono à hexa), atteignant plusieurs milliers de mg/kg. Ces concentrations étaient également élevées dans les fumées (jusqu'à 1700 $\mu g/m^3$) et leurs condensats (plusieurs centaines de $\mu g/m^2$). La concentration des PBDD était égale à 3% de celle des PBDD/PBDF. Celle de l'isomère substitués en 2,3,7,8 n'atteignait pas 3% de la concentration totale des PBDD/PBDF. Les penta- et hexaPBDF substitués en 2,3,7,8 ont fourni entre 1 et 16% du total correspondant. Lors d'essais d'incendie de véhicules, on a trouvé dans les résidus, des concentrations en PBDF (mono à octa) pouvant aller jusqu'à 4,3 µg/kg.

Au cours d'incendies réels dans des résidences privées (téléviseur étant en cause), dans des immeubles de bureaux (ordinateur en cause) ou d'autres bâtiments, on a trouvé des concentrations généralement inférieures à celles que l'on avaient obtenues expérimentalement comme indiqué ci-dessus, la composition étant toutefois qualitativement similaire. On a mis en évidence des PBDF dans la presque totalité des échantillons; par contre, on n'a pas toujours trouvé des PBDD. Lorsqu'elles étaient présentes, c'était à faible concentration. La concentration des PBDF dans les résidus de combustion était généralement de l'ordre de plusieurs µg/kg (faible à élevée) mais on a observé des valeurs maximales (somme des dérivés mono à hexa) pouvant atteindre 107 mg/kg. A proximité immédiate du lieu des incendies, on a décelé la présence d'une contamination par des PBDF (mono à hexa) à des concentrations allant la plupart du temps de 0,1 à 13 $\mu g/m^2$. En outre, on a pu déceler la présence de PXDD/PXDF à des concentrations significatives. La proportion des PBDD/PBDF substitués en 2,3,7,8 était relativement faible dans la majeure partie des échantillons étudiés. Par exemple, lors d'incendies impliquant des téléviseurs, les proportions maximales étaient respectivement égales à 3, 10 et 18% du total des tétra-, penta- et hexaBDF. Des prélèvements de suie effectués après l'incendie d'une salle d'ordinateurs contenaient des tétra- et des pentabromodibenzo-*p*-dioxines (tétra/pentaBDD ou TeBDD/PeBDD) ainsi que des tétra- et pentaBDF à diverses concentrations, la plus élevée (48 µg/kg) étant celle du 2,3,7,8-TeBDF (TBDF).

On a mis en évidence des PXDD dans les cendres d'une chaudière à bois. On n'a toutefois pas précisé de quelle sorte de bois il s'agissait (traité ou non traité). On ne dispose d'aucune donnée sur l'incinération d'autres combustibles comme le charbon, la tourbe ou le mazout.

On a signalé la présence de PBDD/PBDF ou de PXDD/PXDF dans les cendres volantes et les gaz émis par les incinérateurs municipaux ou hospitaliers ou encore ceux que l'on utilise pour détruire les déchets dangereux. La plupart de ces composés prennent probablement naissance dans l'incinérateur lui-même, soit à partir de précurseurs qui réagissent aux températures élevées engendrées par les flammes, soit par synthèse *de novo* à basse température dans la zone de post-combustion de l'appareil. On explique la formation des PXDD/PXDF par un échange important entre atomes de brome et de chlore (échange avec des composés chlorés présents dans les déchets), comme on peut l'observer expérimentalement dans un certain nombre de cas. Les concentrations de PBDD/PBDF et PXDD/PXDF mesurées dans les cendres volantes des incinérateurs sont de l'ordre du ng/kg ou du µg/kg. Dans la plupart des cas, on constate que la concentration des dibenzo-*p*-dioxines dépasse celle des dibenzofuranes et que les PXDD/PXDF sont plus abondants que les PBDD/PBDF. Parmi les homologues substitués en 2,3,7,8, on a trouvé une dibenzo-*p*-dioxine tétrahalogénée mixte (tétraXDD ou TeXDD) (Br_2Cl_2DD).

L'analyse d'échantillons de déchets provenant de décharges a révélé la présence de PBDD/PBDF et de PXDD/PXDF à des concentrations allant de plusieurs centaines à plusieurs milliers de ng/kg de poids à sec. La concentration des dibenzo-*p*-dioxines (jusqu'à 580 ng/kg) était inférieure à celle des dibenzofuranes (jusqu'à 4230 ng/kg). En ce qui concerne les homologues, ce sont les dérivés les moins halogénés (mono à téta) qui prédominaient. Dans les déchets de laboratoires de chimie, on a relevé la présence de PBDD/PBDF dont la concentration maximale atteignait 15 500 ng/kg (dans le cas des hexaBDF).

On a décelé la présence de PBDD/PBDF dans des plastiques (avec ou sans métaux) à différents stades de leur recyclage. Les échantillons provenaient pour la plupart de matériel de bureau, de supports de circuits imprimés et autres types de matériel électronique

au rebut. Dans certains cas, la concentration totale des huits homologues retenus (substitués en 2,3,7,8) atteignait 65 µg/kg. On a constaté que la récupération des métaux pouvait aussi être une source de PBDD ou de PXDD/PXDF. Des PBDD/PBDF ont été mis en évidence dans industries textiles faisant usage de derivés bromés comme retardateurs de flamme. On a également décelé des PBDF dans des gaz d'échappement, dans des textiles avant et après traitement ainsi que dans des dépôts de cheminée.

Des PBDD/PBDF et des PXDD/PXDF (ainsi que des PCDD/PCDF) ont été décelés dans les gaz d'échappement de moteurs utilisant de l'essence au plomb, dans ceux de moteurs utilisant de l'essence sans plomb avec ou sans catalyseur, ainsi que dans ceux de moteurs diesel. Etant donné l'utilisation d'agents de balayage bromés ou chlorés (dibromo- et dichloroéthane) comme additifs à l'essence au plomb, c'est dans celle-ci que l'on trouve la plus forte concentration de PHDD/PHDF (plusieurs milliers de ng/m^3). L'essence sans plomb donne lieu à des émissions beaucoup moins importantes de PHDD/PHDF (inférieures d'environ deux ordres de grandeur). L'épuration catalytique des gaz permet de réduire encore ces valeurs. La concentration est un peu plus élevée pour les moteurs diesel que pour les moteurs à explosion (moteurs à allumage commandé) fonctionnant à l'essence sans plomb. On a constaté que les PBDD/PBDF étaient plus abondants que les PXDD/PXDF et que les PCDD/PCDF dans les gaz résultant de la combustion d'essence au plomb. En général, la concentration des dibenzofuranes était supérieure à celle des dibenzo-*p*-dioxines, avec prédominance des homologues les moins substitués (mono à tri). On a observé une composition analogue dans les résidus adhérant à la paroi des pots d'échappement.

3. Transport, distribution et transformation dans l'environnement

On ne possède guère de données sur le transport et la distribution des PBDD/PBDF dans l'environnement. En général, leurs propriétés physicochimiques suggèrent certaines similitudes avec les PCDD/PCDF. On peut donc s'attendre, en cas de libération dans l'environnement, à ce qu'ils se répartissent de préférence dans les compartiments riches en carbone ou en corps gras, comme c'est le cas des PCDD/PCDF.

Le transport des PBDD/PBDF aéroportés s'effectue soit sous la forme de gouttelettes soit en phase gazeuse, le coefficient de partage étant fonction du degré de bromation.

On ne dispose d'aucune donnée expérimentale sur la migration des PBDD/PBDF dans l'eau ou le sol. Dans le cas des PBDF (tri à penta) on a observé une adsorption aux sédiments. Du fait que les PBDD/PBDF sont peu solubles dans l'eau, le lessivage à partir du sol devrait être limité tout en étant susceptible de s'accroître en présence de solvants organiques ou d'acides humiques.

Il n'existe pas non plus de données expérimentales sur les processus de transport et de distribution des PBDD/PBDF entre les divers compartiments du milieu et les biotes ou encore à l'intérieur de ces biotes. En s'appuyant sur la valeur élevée du coefficient de partage octanol/eau calculé pour un certain nombre de PCDD/PCDF, PBDD/PBDF et PXDD/PXDF, on peut s'attendre à ce que la biodisponibilité des PCDD/PCDF soit également élevée.

On a étudié la photolyse des PBDD/PBDF et celle des PXDD/PXDF, soit au laboratoire dans des solvants organiques ou sur des surfaces de quartz, soit à l'extérieur sur le sol et sur des particules de suie ou de poussière. C'est dans ces dernières conditions, plus représentatives de la réalité environnementale, que l'on a observé les réactions les plus lentes. La débromation réductrice s'est révélée être l'une des principales voies de décomposition photochimique. La vitesse de décomposition dépend du nombre et de la position des atomes de brome. En général, les dérivés les plus substitués ou substitués sur les chaînes latérales sont ceux dont la demi-vie est la plus courte. Le calcul de la demi-vie donne des valeurs qui sont de l'ordre de quelques minutes (rayonnement solaire direct ou UV et ampoules de quartz), de quelques heures (pellicules solides, particules de suie ou poussières et rayonnement solaire) et enfin de quelques centaines ou milliers d'heures (échantillons de sol et rayonnement solaire). Par exemple, la valeur calculée de la demi-vie du 2,3,7,8-TeBDD (TBDD) est de 0,8 minutes en présence de lumière solaire (en solution dans un solvant organique) ou de 32 h (dispersé sous la forme de pellicules solides). On estime que la demi-vie des différents isomères du tétraBDD est de 3 à 6 mois à la surface du sol. Si on les compare aux PCDD/PCDF, on constate que les homologues bromés

sont photochimiquement moins stables. Dans le cas des PXDD/PXDF, ce sont les atomes de brome qui s'éliminent préférentiellement lors de la photolyse pour donner naissance à des PCDD/PCDF dont la demi-vie photolytique est plus longue. Cette transformation des PXDD/PXDF en PCDD/PCDF se produit également pendant l'incinération.

Les PBDD/PBDF semblent être peu biodégradables.

La présence de PBDD/PBDF chez l'Homme et les animaux, comme le révèlent un certain nombre d'études, est l'indication de leur capacité de bioaccumulation. Lors d'études d'alimentation de type subchronique, on a constaté que la 2,3,7,8-TeBDD s'accumulait dans l'organisme du rat. On ne connaît pas la valeur des facteurs de bioaccumulation, de bioconcentration ou de bioamplification des PBDD/PBDF et des PXDD/PXDF.

4. Concentrations dans l'environnement et exposition humaine

Jusqu'ici, on ne s'est que rarement préoccupé d'inclure les PCDD/PCDF et PBDD/PBDF dans les programmes de surveillance de l'environnement. Les quelques études dont on dispose indiquent qu'ils n'y sont pas très souvent présents.

Dans l'air ambiant, on trouve plus fréquemment des PBDF que des PBDD. Seules des PBDD peu bromées (mono à tétra) ont été décelées à des concentrations allant de non décelable à environ 0,85 pg/m^3; il s'agissait en l'occurrence de monobromodibenzo-*p*-dioxines (monoBDD ou MoBDD) dans l'air d'un tunnel routier et d'un garage souterrain. Dans le cas des PBDF, on a trouvé des homologues mono- à hexabromés à des concentrations allant de non décelable à 74 pg/m^3. Par exemple, la concentration moyenne des PBDD/PBDF totaux (tri à hexa) mesurée en Allemagne dans un tunnel routier, en centre ville et dans une banlieue était respectivement égale à 23 pg/m^3, 2 pg/m^3 et 0,59 pg/m^3; on n'a pas constaté la présence de 2,3,7,8-TeBDD et la concentration maximale des 2,3,7,8-TeBDF et des 1,2,3,7,8-PeBDF était respectivement égale à 0,28 et 0,08 pg/m^3. On a mis en évidence des PXDF dans des échantillons d'air de rues à grande circulation, à des concentrations allant jusqu'à 41 pg/m^3 (Cl_1Br_1DF). Dans des échantillons de poussières extérieures (provenant pour la plupart

d'autoroutes), on a également constaté la prédominance de PBDF/PXDF (concentrations maximales de plusieurs ng/kg) par rapport aux PBDD/PBDF (concentrations maximales jusqu'à quelques centaines de ng/kg).

Dans des échantillons d'air prélevés dans des pièces équipées d'un certain nombres d'appareils électroniques (téléviseurs ou moniteurs d'ordinateurs), on a constaté la présence de PBDF (tétra à hepta) à une concentration totale allant de 0,23 à 1,27 pg/m^3. On n'a pas décelé de PBDD. Dans des échantillons de poussière provenant de salles d'ordinateurs, on a trouvé des PBDF à la concentration totale de 2,4 à 5,5 µg/kg. Contrairement à ce que l'on a observé dans l'air, il y avait prédominance des hexaBDF et des heptaBDF (HpBDF) (heptabromodibenzofuranes). C'est seulement dans les échantillons de poussière que l'on a relevé la présence de faibles concentrations de tétraBDD (jusqu'à 1 µg/kg), de tétraBDF substitués en 2,3,7,8 et de pentaBDF (jusqu'à 0,07 µg/kg). Dans un échantillon de poussière ménagère, la concentration des PBDF était plus basse d'un facteur 10. La concentration totale des PBDD/PBDF était égale à celle des PCDD/PCDF dans la poussière de salles d'ordinateurs mais elle était plus faible dans la poussière ménagère. La poussière d'un garage souterrain contenait moins de PBDF (mono et di) et de PXDF (di à tétra) faiblement bromés, avec une concentration maximale de 4,3 µg/kg pour les dibenzofuranes dihalogénés mixtes (DiXDF).

On ne dispose d'aucune donnée sur la concentration des PBDD/PBDF dans l'eau.

Dans des sédiments fluviaux et marins prélevés au voisinage d'un site industriel, on a trouvé des tétraBDD (jusqu'à 0,006 µg/kg de poids sec) ainsi que des PBDF (tétra à hexa) (concentration totale allant jusqu'à 0,37 µg/kg de poids sec). Dans les sédiments provenant d'un réseau de drainage routier, on a trouvé des PBDF (concentration totale des mono à tri: 2,5 µg/kg; concentration totale des tétra à hepta: 0,3 µg/kg) et des PXDF (concentration totale des di et des tri: 1,85 µg/kg), mais pas de PBDD.

De même, des échantillons de sol prélevés à proximité d'une autoroute contenaient des monobromodibenzofuranes (monoBDF ou MoBDF) et des dibromodibenzofuranes (DiBDF) (total: 1,3 µg/kg),

des téta- et des pentaBDF (total: 0,02 µg/kg) et des PXDF (total: 1 µg/kg), mais pas de PBDD. Des échantillons de sol prélevés sur un site d'incinération à proximité d'une usine de récupération de métaux contenaient des PBDF à une concentration totale allant jusqu'à 100 µg/kg, mais pas de PBDD non plus. Dans une série d'échantillons de boues d'égout provenant de stations d'épuration municipales, on a trouvé une teneur totale en PBDF allant de non décelable à 3 µg/kg. Dans un cas, on a décelé des traces de tétraBDD et de 2,3,7,8-TeBDF. Un échantillon de compost s'est révélé à peu près exempt de PBDD/PBDF (tétraBDF <0,003 µg/kg).

On ne possède aucune donnée quantitative sur la teneur des denrées alimentaires en PBDD/PBDF.

Dans de l'herbe et des aiguilles de pin prélevées à proximité d'une autoroute, on a décelé la présence de PBDF/PXDF faiblement halogénés (mono à téta) et de traces de PBDD/PXDD (mono à tri).

On n'a pas trouvé de traces de PBDD/PBDF dans les rares échantillons biologiques de faune et de flore sauvages dont on disposait.

Dans du lait de vache provenant de fermes situées à proximité d'une installation municipale d'incinération, on pense avoir trouvé des dérivés halogénés qui seraient des tribromodibenzofuranes (triBDF ou TrBDF), un tétraBDF et un pentaBDF (sans substitution en 2,3,7,8).

Dans les rares échantillons de tissus adipeux et de lait humain provenant de la population générale qui ont été analysés à la recherche de PBDD/PBDF, on n'a pas trouvé trace de ces produits.

Une contamination par des PBDD/PBDF est possible sur divers lieux de travail où l'on produit, transforme, utilise ou évacue certains retardateurs de flamme ou produits qui en contiennent, notamment quand sont mis en oeuvre des processus nécessitant des températures élevées. Le degré d'exposition des travailleurs dépend non seulement des composés en cause mais encore de la qualité de l'air et de la ventilation. On ne possède que peu de données résultant de la surveillance de divers lieux de travail tels qu'usines de production ou de transformation de matières plastiques, bureaux ou studios comportant un

grand nombre d'appareils électriques en fonctionnement continu ou unités de recyclage (notamment installations de récupération du cuivre). En général, on constate que les PBDF sont plus abondants que les PBDD et que leur concentration est maximale dans les ateliers où sont produits des polymères contenant des DBDE. On a pu mettre en évidence des PBDF/PBDD substitués en 2,3,7,8 dans de nombreux échantillons. Une contamination par ces composés a également été constatée dans un laboratoire de chimie, dans la partie d'une paillasse située au-dessous de la hotte. On manque de données de contrôle relatives aux installations d'incinération des déchets.

5. Cinétique et métabolisme

La plupart des études concernent la 2,3,7,8-TeBDD et, dans une moindre mesure, le 1,2,3,7,8-TeBDF. La demi-vie a également été calculée pour un certain nombre d'homologues.

Après administration à des rats par voie buccale, intratrachéenne ou par application cutanée, on a constaté que la 2,3,7,8-TeBDD était résorbé dans une proportion qui dépendait de la voie d'administration et de la dose. Par exemple, une dose unique de 1 nmol de 2,3,7,8-TeBDD par kg de poids corporel a été résorbée à hauteur de 80% (voie buccale ou intratrachéenne) ou de 12% (voie percutanée). L'absorption percutanée de 1 nmol de 1,2,7,8-TeBDF par kg de poids corporel a été d'environ 29%. *Per os,* la TeBDD est résorbée dans une proportion comparable à celle de la 2,3,7,8-tétrachlorodibenzo-*p*-dioxine (2,3,7,8-TeCDD ou TCDD). Par contre, l'absorption percutanée de ce même composé (2,3,7,8-TeBDD) a été à peu près égale aux deux tiers de celle d'une dose équimolaire de 2,3,7,8-TeCDD.

Lorsqu'on les administre à des rats par n'importe quelle voie, la 2,3,7,8-TeBDD ou le 1,2,7,8-TeBDF se répartissent dans tout l'organisme, s'accumulant de préférence dans le foie et les tissus adipeux, puis, dans l'ordre, dans la peau et les muscles. Par exemple, 3 jours après administration de doses uniques de 2,3,7,8-TeBDD (1 nmol par kg de poids corporel) on observait, dans ces tissus, une répartition dans les proportions respectives de 20%, 20%, 11% et 4%, alors que le thymus et les surrénales n'en contenaient respectivement que 0,03% et 0,4%. Chez le rat, la répartition de la 2,3,7,8-TeBDD entre le foie et les tissus adipeux dépendait de la dose, du mode

d'exposition, et du temps écoulé depuis l'administration. Le rapport concentration dans le foie/concentration dans les tissus adipeux mesuré dans différentes conditions allait de 0,2 à 6,5 (doses uniques de 2,3,7,8-TeBDD administrées à des rats). On ne possède aucune donnée expérimentale concernant la transmission des PBDD/PBDF à la progéniture.

On a décelé la présence de métabolites de tétraBDD/BDF dans la bile et les matières fécales de rats. Ces métabolites se forment principalement par hydroxylation du noyau aromatique et débromation hydrolytique. Le taux de métabolisation (déterminé indirectement par le taux d'excrétion biliaire) était différent selon qu'il s'agissait de 2,3,7,8-TeBDD (environ 7%) ou de 1,2,7,8-TeBDF (environ 50%). Trois jours après l'administration par voie intraveineuse d'une dose de 2,3,7,8-TeBDD égale à 1 nmol par kg de poids corporel, 14% de la dose initiale étaient retrouvés sous forme de métabolites dans les matières fécales des rats.

L'élimination et l'excrétion de la 2,3,7,8-TeBDD a été étudiée chez le rat en utilisant différentes voies d'administration: voies buccale, intraveineuse, intratrachéenne et percutanée. Dans toutes les études, la principale voie d'élimination a été la voie fécale. La radioactivité éliminée allait de 2% (voie percutanée) à 42% (voie buccale) de la dose initiale, c'est-à-dire 1 nmol de (^{3}H)2,3,7,8-TeBDD par kg de poids corporel dans les échantillons de matières fécales, et de 0,2 à 1% dans les échantillons d'urine. De même, l'étude du 1,2,7,8-TeBDF sur des rats a également montré que l'excrétion se fait essentiellement par la voie fécale, la dose initiale administrée par voie intraveineuse, buccale ou percutanée n'étant excrétée qu'à hauteur de 2 à 3% dans les urines. Au cours des premiers jours qui ont suivi l'administration par voie buccale, les composés ont été principalement éliminés tels quels dans les matières fécales ainsi que dans la bile. La fraction de la dose initiale de 2,3,7,8-TeBDD retrouvée dans les matières fécales des rats après administration de 1 nmol de ce composé par kg de poids corporel était respectivement égale à 53% (voie buccale), 43% (voie intratrachéale) et 10–20% (voie intraveineuse). Quelques jours après l'administration de 2,3,7,8-TeBDD par voie buccale (1 nmol/kg de poids corporel), environ 20% du composé initial ont été éliminés sous forme inchangée.

Pour un certain nombre de PBDD/PBDF, on possède des données sur la rétention et la vitesse d'élimination. En particulier, on sait que chez le rat, la charge relative de l'organisme en 2,3,7,8-TeBDD (et homologues) dépend de la voie d'administration et de la dose administrée, traduisant ainsi la variation du degré de résorption. On a calculé la demi-vie d'un certain nombre de PBDD/PXDD et PBDF présents dans divers tissus et dans les matières fécales de rats. Elle s'échelonne entre 1 jour (élimination du 1,2,7,8-TeBDF présent d l'organisme) et 99 jours (élimination du 2,3,4,7,8-PeBDF présent dans le foie). Le calcul de la demi-vie de la 2,3,7,8-TeBDD présente dans le foie, les matières fécales et les tissus adipeux donne les valeurs respectives de 17, 18 et 58 jours, valeurs qui sont du même ordre que celles de la 2,3,7,8-TeCDD dans le cas du foie et des matières fécales mais plus de deux fois plus élevés que dans le cas des tissus adipeux. Malgré des différences de rétention au cours des premiers jours, le 2,3,7,8-TeBDF et le 2,3,7,8-tétrachlorodibenzofurane (2,3,7,8-TeCDF ou TCDF) ont une demi-vie comparable au niveau du foie.

Comme dans le cas des PCDD/PCDF, le calcul donne une demi-vie beaucoup plus longue chez l'Homme que chez l'animal. Les estimations sont les suivantes: 3 à 11 ans (moyenne 5,9 ans) pour la 2,3,7,8-TeBDD et 1 à 2 ans (moyenne 1,5 ans) pour le 2,3,7,8-TeBDF. On a également pu se rendre compte de la persistance de ces composés dans le cas d'un chimiste qui avait préparé de la 2,3,7,8-TeBDD et de la 2,3,7,8-TeCDD en 1956. Trente-cinq ans après, son sang contenait encore une quantité importante de 2,3,7,8-TeBDD.

6. Effets sur les mammifères de laboratoire et les systèmes d'épreuve *in vitro*

La plupart des études ont porté sur la toxicité de la 2,3,7,8-TeBDD, mais on dispose tout de même de quelques données sur les autres PBDD/PBDF et PXDD/PXDF.

La 2,3,7,8-TeBDD a des effets analogues à ceux de la 2,3,7,8-TeCDD, notamment un syndrome de dépérissement, une atrophie du thymus et une action toxique sur le foie. On a observé au niveau du foie des lésions caractéristiques d'une péliose hépatique, lésions qui n'ont pas été observées chez le rat après exposition à la 2,3,7,8-TeCDD. De par leur nature (létalité, histopathologie, poids du foie et

du thymus), les lésions ou effets toxiques observés chez le cobaye et le rat après une brève exposition au 2,3,7,8-TeBDF étaient analogues à ceux observés après exposition au 2,3,7,8-TeCDF.

La 2,3,7,8-TeBDD agit sur le système endocrinien. Chez le rat, on a observé une modification des hormones thyroïdiennes présentes dans la circulation ainsi qu'une diminution de la spermatogénèse.

La DL_{50} par voie orale (période d'observation de 28 jours) de la 2,3,7,8-TeBDD pour le rat Wistar est d'environ 100 µg/kg de poids corporel dans le cas des femelles et d'environ 300 µg/kg p.c. dans le cas des mâles. Celle de la 2,3,7,8-TeCDD tirée d'autres études varie de 22 à >3000 µg/kg p.c. Des doses équimolaires de 2,3,7,8-TeBDF et de 2,3,7,8-TeCDF ont entraîné une mortalité comparable chez des cobayes. Par exemple, on a observé une mortalité de 100% après administration de 2,3,7,8-TeBDF (0,03 µmol/kg p.c., 15,8 µg/kg p.c.) et de 2,3,7,8-TeCDF (0,03 µmol/kg p.c., 10 µg/kg p.c.). On a noté la présence de lésions prépéliotiques et une modification des hormones thyroïdiennes chez des rats qui avaient reçu une dose unique de 100 µg de 2,3,7,8-TeBDD par kg de poids corporel.

Chez des rats Wistar ayant reçu pendant 13 semaines de la 2,3,7,8-TeBDD par voie orale, on a relevé des signes de réduction de la spermatogénèse, la présence de spermatocytes anormaux ou nécrosés, les signes d'une péliose hépatique grave ainsi qu'une modification des hormones thyroïdiennes circulantes et du poids des organes. La dose sans effet nocif observable (NOAEL) a été trouvée égale à 0,01 µg/kg p.c. par jour.

Du 2,3,7,8-TeBDF administré par voie orale à des rats Sprague-Dawley pendant 4 semaines a provoqué un retard de croissance lié à la dose et des anomalies histopathologiques au niveau du foie et du thymus. La NOAEL a été estimée 1 µg/kg p.c. par jour.

Chez des souris qui avaient reçu des 2,3,7,8-PBDD/PBDF administrés par voie orale ou sous-cutanée on a noté, pour certains de ces produits, des effets délétères sur le développement à des doses non toxiques pour les mères et non létales pour les foetus. La dose la plus faible (en µg/kg p.c.) produisant un effet observable (LOEL) – à savoir une hydronéphrose et une fente palatine – après administration d'une

dose unique par voie orale à des souris gravides a été trouvée respectivement égale à: 3 et 48 pour la 2,3,7,8-TeBDD, à 25 et 200 pour le 2,3,7,8-TeBDF, à 400 et 2400 pour le 2,3,4,7,8-PeBDF et à 500 et 3000–4000 pour le 1,2,3,7,8-PeBDF. On a constaté que la 2,3,7,8-TeBDD et la 2,3,7,8-TeCDD avaient pratiquement la même aptitude à induire une hydronéphrose lorsqu'on utilisait la mole comme unité; par contre, sur une base pondérale, les isomères bromés se révélaient moins aptes que les isomères chlorés à produire une hydronéphrose ou une fente palatine. Le 2,3,7,8-TeBDF, en revanche, était plus actif que le 2,3,7,8-TeCDF.

On n'a trouvé aucune donnée sur la mutagénicité des PBDD/PBDF ni sur des points d'aboutissement toxicologiques en rapport avec des propriétés mutagènes.

On ne dispose d'aucune étude sur la toxicité ou la cancérogénicité à long terme des PBDD/PBDF. Une épreuve de transformation cellulaire sur macrophages péritonéaux murins a donné un résultat positif avec la 2,3,7,8-TeBDD. Cependant, l'activité transformante de la 2,3,7,8-TeBDD était sept fois moins forte que celle de la 2,3,7,8-TeCDD. En injectant les cellules ainsi obtenues à des souris *nude* par voie sous-cutanée, on a observé l'apparition ultérieure de tumeurs.

Après injection d'une série de PBDD et de PXDD (tétra et penta) par voie intrapéritonéale à des rats Wistar mâles immatures, on a constaté une perte de poids au bout de 14 jours. En se basant sur la valeur de la DE_{50} (exprimée en moles), on a constaté que les composés les plus toxiques étaient la 2,3,7,8-TeBDD, la 2-Br_1-3,7,8-Cl_3-DD et la 2,3-Br_2-7,8-Cl_2-DD (TBCDD), qui ne sont substituées que sur les quatre positions latérales. Pour l'activité relative des autres PBDD étudiées, on a trouvé l'ordre suivant: 2,3,7,8- > 1,2,3,7,8- > 1,2,4,7,8- > 1,3,7,8-DD. Selon d'autres études, il n'y aurait que peu de différence entre la 2,3,7,8-TeCDD et la 2,3,7,8-TeBDD en ce qui concerne la valeur de la DE_{50} (exprimée en moles) pour la perte de poids, l'atrophie du thymus et l'induction des enzymes hépatiques.

Une atrophie du thymus et d'autres signes d'immunotoxicité (par ex. dans les paramètres hématologiques et aussi des modifications dans certaines sous-populations de lymphocytes) ont été observés chez le rat après exposition à plusieurs PBDD/PXDD et au 2,3,7,8-TeBDF

ainsi que chez le singe marmouset (*Callithrix jacchus*) après exposition à la 2,3,7,8-TeBDD et à la TBCDD. On en a conclu qu'exprimée en moles, l'activité de la 2,3,7,8-TeBDD était comparable à celle de la 2,3,7,8-TeCDD chez le rat et le singe. Par exemple, on a observé un effet sensible sur certaines sous-populations de lymphocytes simiens après injection sous-cutanée d'une dose unique de 30 ng de 2,3,7,8-TeBDD par kg de poids corporel, le même effet étant obtenu avec une dose de 10 ng de 2,3,7,8-TeCDD par kg p.c. On n'a pas étudié les effets immunotoxiques d'une exposition périnatale aux PBDD/PBDF.

Après avoir administré de manière subchronique de la 2,3,7,8-TeBDD ou de la 2,3,7,8-TeCDD par gavage à des souris, on a constaté un accroissement des porphyrines hépatiques totales qui dépendait de la dose.

Après administration à des rats d'une dose unique de 2,3,7,8-TeBDD et de 2,3,7,8-TeCDD, on a observé une réduction de la concentration et de la quantité totale de vitamine A dans le foie, la 2,3,7,8-TeBDD ayant une activité (exprimée en moles) un peu inférieure à celle de la 2,3,7,8-TeCDD.

Lors d'une épreuve sur oreille de lapin effectuée avec de la 2,3,7,8-TeBDD et du 2,3,7,8-TeBDF, on a observé une hyperkératose à la dose de 100 µg/animal mais pas à la dose de 10 µg/animal. Dans le cas de la 2,3,7,8-TeCDD, la dose sans effet observable (NOEL) était de 0,01 µg/animal.

On a constaté que plusieurs homologues tétra- (Br_1Cl_3DD, Br_2Cl_2DD) et penta- (Br_1Cl_4DD) halogénés substitués en 2,3,7,8 avaient une activité antiestrogénique analogue à celle de la 2,3,7,8-TeCDD, comme l'a montré l'observation de cultures de cellules humaines de cancer du sein.

Chez le rat, la 2,3,7-tribromodibenzo-*p*-dioxine (2,3,7-triBDD/TrBDD) a réduit l'élimination de l'ouabaïne du plasma et son excrétion par la voie biliaire; elle a également agi comme anticholagogue, mais dans une moindre proportion que la 2,3,7,8-TeCDD.

Les PBDD/PBDF et les PXDD/PXDF sont de puissants inducteurs de certaines enzymes microsomiennes dépendant du cytochrome P-450 (CYP). On a obtenu une DE_{50} de 0,8–1 nmol/kg p.c. pour l'induction de la CYP1A1 et d'environ 0,2 nmol/kg p.c. pour l'induction de la CYP1A2 dans le foie de rat après administration d'une dose unique de 2,3,7,8-TeBDD par voie orale. L'induction de la CYP1A1 (arylhydrocarbure-hydroxylase [AHH] ou de l'ethoxyrésorufine-*O*-déséthylase [EROD]) a été observée chez diverses espèces et un certain nombre de tissus *in vivo* ainsi que dans des cultures de cellules de rat *in vitro*. Divers homologues se sont révélés actifs à cet égard, de même que les produits de pyrolyse de certains retardateurs de flamme. En général, l'induction des enzymes se produisait à des concentration non toxiques, elle dépendait de la dose, commençait peu après l'exposition et était de longue durée. Elle était mesurable à des concentrations de l'ordre de la picomole. L'activité inductrice différait de plusieurs ordres de grandeur d'un homologue à l'autre, en fonction de la structure chimique. Les inducteurs les plus actifs étaient la TCDD, la TBDD et la TBCDD. Comparées à leurs homologues chlorés, les PBDD et les PXDD avaient une activité (exprimée en moles) à peu près équivalente. Contrairement à la TCDD, dont l'activité inductrice relative s'est montrée indépendante du type de tissu examiné, la TBDD a fait preuve d'en activité inductrice de l'EROD cinq fois plus élevée dans le foie que dans le tissu cutané ou pulmonaire, lors d'études comportant l'exposition de souris dans des conditions de subchronicité. Chez le singe marmouset, l'activité inductrice de l'EROD s'établit comme suit: TCDD > 2,3,4,7,8-pentachlorodibenzofurane (2,3,4,7,8-pentaCDF/PeCDF) > 2,3,4,7,8-PeBDF, l'activité des enzymes étant comparée à la concentration dans le foie. Des épreuves *in vitro* sur cultures de cellules de rat ont donné des valeurs analogues pour la CE_{50} des PXDF et des PCDF correspondants, l'effet examiné étant l'induction de l'AHH et de l'EROD.

On pense que les PBDD/PBDF ont le même mode d'action que les PCDD/PCDF et les d'autres hydrocarbures aromatiques halogénés de type voisin. La fixation sur le récepteur cytosolique aux hydrocarbures aromatiques, qui joue un rôle central dans la toxicité des composés du type 2,3,7,8-TeCDD, a été confirmée pour plusieurs PBDD et PXDD/PXDF. Leur affinité pour ce récepteur variait de

plusieurs ordres de grandeur, mais elle était comparable à celle de leurs homologues chlorés.

7. Effets sur l'Homme

On ne dispose d'aucune donnée sur l'exposition humaine aux PBDD/PBDF ni au sujet de leurs effets sur la santé de la population dans son ensemble.

Deux cas d'effets aigus dus à une exposition à la 2,3,7,8-TeBDD/TeCDD ont été rapportés, avec différents symptômes, dont une chloracné.

Lors d'une autre étude, des employés de sexe masculin travaillant dans une usine chimique et effectivement exposés à des PBDD/PBDF provenant de l'utilisation de retardateurs de flamme (OBDE et DBDE), ont été soumis à des épreuves immunologiques et autres examens de laboratoire. Malgré la présence de légères modifications dans les paramètres immunologiques, leur état général ne laissait en aucun cas penser que la teneur de leur organisme en 2,3,7,8-TeBDD/TeBDF ait eu des effets sur leur système immunitaire.

On ne connaît aucun cas de cancer mortel qui serait dû aux PBDD/PBDF.

8. Effets sur les autres êtres vivants au laboratoire et dans leur milieu naturel

On ne possède que des données limitées sur les effets que les PBDD/PBDF peuvent avoir sur les microorganismes, les plantes, les invertébrés et les vertébrés.

On a soumis une série de PBDD/PBDF et leurs homologues chlorés à une épreuve biologique sur alevins de truite arc-en-ciel (*Onchorhyncus mykiss*) qui a mis en évidence l'activité de ces composés. L'épreuve a montré, entre autres, que l'activité des PBDD/PBDF diminue à mesure qu'augmente de degré de substitution par le brome. La 2,3,7,8-TeBDD et le 2,3,7,8-TeBDF sont plus actifs que leurs homologues chlorés respectifs.

RESUMEN

1. Identidad, propiedades físicas y químicas y métodos analíticos

Las dibenzo-*p*-dioxinas polibromadas (PBDD) y los dibenzofuranos polibromados (PBDF) son compuestos aromáticos tricíclicos casi planares. En teoría existen 75 PBDD y 135 PBDF. Además es en teoría posible la existencia de un amplio número de congéneres halogenados mixtos: 1550 dibenzo-*p*-dioxinas bromadas/cloradas (PXDD) y 3050 dibenzofuranos bromados/clorados (PXDF). Dada la complejidad de los procedimientos analíticos y la escasez de normas analíticas de referencia, sólo se ha podido identificar y determinar un pequeño número de estos productos. Los congéneres más tóxicos son los sustituidos en las posiciones 2,3,7 y 8. Existen 7 PBDD sustituidos en las posiciones 2,3,7 y 8 y 10 PBDF sustituidos en las posiciones 2,3,7 y 8, así como 337 posibles PXDD sustituidos en las posiciones 2,3,7 y 8, y 647 posibles PXDF sustituidos en las posiciones 2,3,7 y 8.

Los PBDD y PBDF tienen mayores pesos moleculares que sus análogos clorados, altos puntos de fusión, bajas presiones de vapor y bajas solubilidades en agua. En general son solubles en grasas, aceites y disolventes orgánicos. Existen escasos datos experimentales sobre las propiedades físicas y químicas de los PBDD y PBDF.

La fotolisis se produce con más rapidez en el caso de los PBDD y PBDF que en las dibenzo-*p*-dioxinas policloradas (PCDD) y los dibenzofuranos policlorados (PCDF). Los PBDD y PBDF son termoestables. Las temperaturas de formación y destrucción de los PBDD y PBDF dependen de varias condiciones, que incluyen la presencia o ausencia de oxígeno, polímeros y aditivos pirorretardantes, como el trióxido de antimonio (Sb_2O_3).

En presencia de cloro en exceso, el bromo es sustituido por cloro para dar PXDD y PXDF.

Teniendo en cuenta el carácter tóxico de estos productos y sus propiedades fotolíticas debe actuarse con cuidado en el curso del muestreo y el análisis. Dado el alto número de congéneres de los PBDD y PBDF se necesitan métodos de análisis muy sensibles,

selectivos y específicos (cromatografía de gases o espectrometría de masas). Los procedimientos de muestreo son idénticos en el caso de todas las dibenzo-*p*-dioxinas polihalogenadas (PHDD) y los dibenzofuranos polihalogenados (PHDF), pero la separación y determinación de las PBDD y los PBDF (y las PXDD y los PXDF) difiere ligeramente de las correspondientes a sus análogos clorados. Los PBDD y PBDF tienen pesos moleculares más altos y mayores periodos de retención en la cromatografía de gases que los análogos clorados, así como distintos tipos de agrupación isotópica en espectrometría de masas y de compuestos de interferencia. La identificación exacta de determinados congéneres bromados es muy limitada debido al escaso número de patrones de referencia actualmente disponibles. Por el mismo motivo, la determinación de los congéneres halogenados mixtos es casi imposible.

2. Formación y fuentes de exposición humana y ambiental

No se conoce la presencia natural de PBDD y PBDF. No se producen de modo intencional (excepto para fines científicos), pero aparecen como productos indeseados en distintos procesos. Pueden formarse por reacciones químicas, fotoquímicas o térmicas a partir de precursores y en la llamada síntesis *de novo*.

Se han hallado PBDD y PBDF como contaminantes en productos químicos orgánicos bromados (por ej., bromofenoles) y, en particular, en pirorretardantes, como los éteres difenílicos polibromados (PBDE), el decabromobifenilo (decaBB o DBB), el 1,2-bis(tribromofenoxi)-etano, el tetrabromobisfenol A (TBBPA) y otros. Se han hallado en residuos de destilación de algunos bromofenoles y bromoanilinas y en desechos de laboratorios químicos.

Los PBDF y, en menor grado, las PBDD se han hallado como productos de degradación fotoquímica de sustancias químicas orgánicas bromadas, como los PBDE y los bromofenoles.

En experimentos de termolisis en laboratorio se ha observado la formación de PBDD y PBDF a partir de bromofenoles, PBDE, bifenilos polibromados (PBB) y otros pirorretardantes bromados (puros o en matriz de polímero). Se observó una amplia gama de

rendimientos, desde 0 hasta los valores máximos (alcanzados a partir de los PBDE) en la gama de g/kg. Por lo general, los PBDF son mucho más abundantes que las PBDD. La temperatura óptima de formación de PBDF en una serie de pirorretardantes puros se situó en la gama de 600–900 °C. La presencia de polímeros o productos sinérgicos (por ej., Sb_2O_3) produjo la disminución de la temperatura óptima de formación (hasta 40 °C). Además de la temperatura y la presencia de productos sinérgicos o de una matriz de polímero, varios otros factores, tales como la presencia de metales, óxidos metálicos, agua y oxígeno, y el tipo de aparato de combustión utilizado, influyeron en el rendimiento y el tipo de PBDD y PBDF obtenidos. En las mezclas ternarias de PBDE, matriz de polímero y Sb_2O_3, los tetrabromodibenzofuranos (tetraBDF o TeBDF) fueron con frecuencia el grupo homólogo más abundante. Se hallaron en concentraciones variables PBDD y PBDF sustituidos en las posiciones 2,3,7 y 8 (tetra a hepta); por ejemplo, se halló el 2,3,7,8-TeBDF en concentraciones de hasta 2000 mg/kg en pirolizados de polímeros que contenían éter de octabromodifenilo (octaBDE u OBDE).

En la fabricación de plásticos se producen altas temperaturas (150–300 °C) en varios procesos. Los estudios de los vapores de escape de máquinas de tratamiento de polímeros, como el acrilonitrilo-butadieno-estireno (ABS) y el tereftalato de polibutileno (PBT), que contenían distintos tipos de pirorretardantes bromados, mostraron que pueden formarse PBDD y PBDF (di a octa) a esas temperaturas. El OBDE y el éter de decabromodifenilo (decaBDE o DBDE) produjeron las mayores cantidades de PBDD y PBDF, consistiendo la porción principal en PBDF. Las concentraciones observadas en el caso de TBBPA o de etileno de bis-tetrabromo-ftalimida (TBPI) eran inferiores en varios órdenes de magnitud. No se hallaron PBDD ni PBDF en el curso del tratamiento de ABS pirorretardado por medio de estireno bromado o 1,2-bis(tribromofenoxi)etano. Los congéneres sustituidos en las posiciones 2,3,7 y 8 no se determinaron (fabricación de DBDE), se hallaron sólo en concentraciones infinitesimales (fabricación de OBDE) o no se detectaron (fabricación de TBBPA y TBPI).

Se analizó la presencia de PBDD y PBDF en varios materiales plásticos en distintas etapas de fabricación. Comprendieron resinas (granuladas) y partes moldeadas, cuyos aditivos pirorretardantes eran

conocidos, así como muestras de dispositivos eléctricos comerciales (televisores, impresoras, ordenadores), cuyos aditivos pirorretardantes eran desconocidos. Se hallaron las mayores concentraciones de PBDD y PBDF en los materiales pirorretardados con PBDE, en la gama de varios miles de µg/kg, excediendo así en varios órdenes de magnitud a las concentraciones de otros pirorretardantes/sistemas de polímero. Los factores que influyen en la cuantía de la formación son la temperatura y la duración de procesos tales como el mezclado, la extrusión y el moldeo. También en este caso dominan los PBDF, con algunas excepciones, sobre las PBDD, prevaleciendo los derivados muy bromados (>tetra). Las máximas concentraciones se observaron en el caso de los pentabromodibenzofuranos (pentaBDF o PeBDF) y los hexabromodibenzofuranos (hexaBDF o HxBDF). Los últimos alcanzaron concentraciones tan altas como 3000 µg/kg en piezas de revestimiento. Los tableros de circuitos impresos contenían tetra- y pentaBDF en concentraciones máximas de 1300 y 1400 µg/kg, respectivamente. Las concentraciones totales de PBDF (mono a hexa) se hallaban en la gama de 3,6–3430 µg/kg. Los PBDD y PBDF sustituidos en las posiciones 2,3,7 y 8 no se determinaron, eran indetectables o se hallaban en concentraciones relativamente bajas. Las concentraciones máximas de PBDF sustituidos en las posiciones 2,3,7 y 8 (tetra a hexa) en revestimientos o tableros de circuitos impresos eran de 11 µg/kg (tetra) a 203 µg/kg (hexa).

Los experimentos destinados a determinar si se liberaban PBDF de los televisores o aparatos análogos durante el uso mostraron la presencia de concentraciones en el aire que iban de los niveles indetectados a 1800 pg de PBDF totales (tetra a hexa) por aparato.

La combustión de productos que contenían compuestos bromados produjo la emisión de PBDD y PBDF. Las pruebas experimentales de incendio que simulaban condiciones reales utilizando aparatos eléctricos tales como televisores, impresoras, terminales de ordenador, y sus receptáculos, permitieron hallar altas concentraciones de PBDF (mono a hexa) en los residuos de la combustión (miles de µg/kg), en el condensado del humo (centenares de $\mu g/m^2$) y en el humo (hasta 1700 $\mu g/m^3$). Las concentraciones de PBDD fueron del 3% aproximadamente de los niveles detectados de PBDD y PBDF. El isómero 2,3,6,8-sustituido se hallaba sobre todo por debajo del 3% del total de tetraBDF. Los penta- y hexaBDF 2,3,7,8-sustituidos dieron del 1 al

16% de los totales correspondientes. La combustión de vehículos de prueba produjo concentraciones de PBDF (mono a octa) de hasta 4,3 µg/kg en los residuos del incendio.

En el curso de incendios reales en residencias privadas (con inclusión de televisores), oficinas (con inclusión de ordenadores) y otros edificios, las concentraciones medidas se hallaban en la mayoría de los casos por debajo de los valores observados en los modelos experimentales antes descritos, pero la composición cualitativa de las muestras era análoga. Se hallaron PBDF en casi todas las muestras, pero no siempre se detectaron PBDD; si se encontraban, sus concentraciones eran bajas. Las concentraciones de PBDF en los residuos de la combustión se hallaban principalmente en la gama de µg/kg (bajas a altas), pero también se observaron valores máximos (suma de mono a hexa) de hasta 107 mg/kg. Las concentraciones contaminantes de PBDF (mono a hexa) en las cercanías del lugar del incendio variaban entre 0,1 y 13 µg/m^2 en la mayoría de los casos. Pudieron detectarse además concentraciones significativas de PXDD y PXDF. La proporción de PBDD y PBDF sustituidos en las posiciones 2,3,7 y 8 era relativamente baja en la mayoría de las muestras examinadas. Por ejemplo, se registraron proporciones máximas del 3, el 10 o el 18% de los totales correspondientes de tetra-, penta- o hexaBDF, respectivamente, en los incendios que comprendieron televisores. Las muestras de cenizas recogidas después de un incendio en una sala de ordenadores contenían tetra- y pentabromodibenzo-*p*-dioxinas sustituidas en las posiciones 2,3,7 y 8 (tetra/pentaBDD o TeBDD/PeBDD) y tetra- y pentaBDF, con una concentración máxima de 48 µg/kg en el caso del 2,3,7,8-TeBDF (TBDF).

Se detectaron PXDD en la ceniza de una caldera de combustión de madera. Sin embargo, no se especificó el tipo de madera (tratada o sin tratar). No se dispuso de datos sobre la incineración de otros combustibles, como carbón, turba o fueloil.

Se señaló la presencia de PBDD/PBDF y/o PXDD/PXDF en cenizas volantes y en los gases de combustión de incineradores municipales, de hospital o de desechos peligrosos. La mayor parte de esos productos estaban formados probablemente en el propio incinerador, a partir de precursores a altas temperaturas en la llama o por síntesis *de novo* a temperaturas bajas en la zona poscombustión del

incinerador. La formación de PXDD y PXDF se explica por las amplias reacciones de intercambio bromo-cloro (con donantes de cloro en los desechos) observadas en varias condiciones de prueba. Las cantidades de PBDD/PBDF y PXDD/PXDF medidas en las cenizas volantes de los incineradores se hallaban comprendidas en la gama de ng/kg a µg/kg. En la mayoría de los casos, las concentraciones de dibenzo-*p*-dioxinas excedían a las de los dibenzofuranos, siendo los PXDD/PXDF más abundantes que los PBDD/PBDF. Entre los congéneres 2,3,7,8-sustituidos se halló una dibenzo-*p*-dioxina tetrahalogenada mixta (tetraXDD o TeXDD) (Br_2Cl_2DD).

Los análisis de muestras de desechos procedentes de varios vertederos mostraron la presencia de PBDD/PBDF y PXDD/PXDF en concentraciones de varios centenares a varios miles de ng/kg de peso en seco. La concentración de las dibenzo-*p*-dioxinas (hasta 580 ng/kg) era inferior a la de los dibenzofuranos (hasta 4230 ng/kg). Por lo general, la gama de homólogos estaba dominada por los derivados menos halogenados (mono a tetra). Los desechos de laboratorios químicos contenían PBDD y PBDF con una concentración máxima de 15 500 ng/kg en el caso de los hexaBDF.

Se hallaron PBDD y PBDF en materiales plásticos (con o sin metales) en varias etapas de reciclado. Las muestras procedían principalmente de maquinaria de oficina, tableros de circuitos impresos y otras chatarras electrónicas. En algunos casos, la suma de las concentraciones de 8 congéneres PBDD/PBDF seleccionados con sustituciones en las posiciones 2,3,7 y 8 llegaba a 65 µg/kg. También se observó que la recuperación de metales era una fuente de PBDD y/o PXDD/PXDF. Igualmente se detectaron PBDD y PBDF en la industria textil, en la que se utilizaban pirorretardantes bromados. Se hallaron PBDF en el aire de salida de industrias textiles, antes y después del procesado, y en sedimentos de chimenea.

Se han detectado PBDD/PBDF y PXDD/PXDF (junto con PCDD/PCDF) en los gases de escape de motores que utilizan gasolina plomada, en los gases de escape de motores que usan gasolina sin plomo con o sin convertidores catalíticos y en los gases de escape de motores diesel. Teniendo en cuenta que la gasolina plomada contiene productos de limpieza bromados y clorados (dibromoetano y dicloroetano), las mayores concentraciones de PHDD y PHDF (varios miles de

ng/m^3) se encuentran en este tipo de gasolina. La gasolina sin plomo produce emisiones muy inferiores de PHDD y PHDF (aproximadamente inferiores en dos órdenes de magnitud). Tras la limpieza catalítica de los gases se observa una nueva reducción. Los valores correspondientes a los motores diesel eran ligeramente superiores a los hallados en los motores Otto (motores de encendido por chispa) que funcionan con gasolina sin plomo. En los gases de escape procedentes de la combustión de gasolina plomada, los PBDD/PBDF eran más abundantes que los PXDD/PXDF y PCDD/PCDF. En general, las concentraciones de los dibenzofuranos excedían a las de las dibenzo-*p*-dioxinas, con un predominio de los homólogos de baja sustitución (mono a tri). Se han hallado distribuciones análogas en los residuos adheridos a los silenciadores de escape.

3. Transporte, distribución y transformación en el medio ambiente

Se dispone de datos muy escasos sobre el transporte y la distribución en el medio ambiente de los PBDD y PBDF. Por lo general, sus propiedades fisicoquímicas permiten pensar en analogías con los PCDD y PCDF. Por consiguiente, si pasan al medio ambiente, pueden estar de preferencia distribuidos en compartimentos ricos en carbono y grasas, como sucede con los PCDD y PCDF.

El transporte por el aire de PBDD y PBDF se realiza en forma de partículas y en fase de vapor, dependiendo la relación de partición del grado de bromación.

No se dispone de datos experimentales sobre el movimiento de los PBDD y PBDF en el agua o el suelo. En el caso de los PBDF (tri a penta) se ha señalado la adsorción al sedimento. Debido a la baja hidrosolubilidad de los PBDD y PBDF, la filtración por el suelo puede estar limitada, pero aumentar en presencia de disolventes orgánicos o ácidos húmicos.

No se dispone de datos experimentales sobre los procesos de transporte y distribución de los PBDD y PBDF entre el medio ambiente y los biota o dentro del los biota. Basándose en la existencia de análogos coeficientes elevados de partición octanol/agua, calculados

para determinados PCDD/PCDF, PBDD/PBDF y PXDD/PXDF, se supone una biodisponibilidad comparable a la de los PCDD y PCDF.

Se estudió la fotolisis de los PBDD/PBDF y PXDD/PXDF en disolventes orgánicos y sobre superficies de cuarzo en el laboratorio, así como en el suelo y en partículas de hollín (y polvo) al aire libre. Se observaron las reacciones fotolíticas más lentas en estas últimas condiciones, más pertinentes respecto al medio ambiente. Se observó que la desbromación reductora era la principal vía metabólica. La tasa de descomposición de los distintos congéneres depende de su tipo de sustitución del bromo. Por lo general, los congéneres muy bromados y los que poseen bromo en posiciones laterales tienen semividas más breves. Las semividas calculadas eran del orden de minutos (empleo de luz solar directa o luz ultravioleta [UV] y de viales de cuarzo), horas (empleo de láminas sólidas o de partículas de hollín o polvo y luz solar) o de centenares a miles de horas (empleo del suelo y luz solar). Por ejemplo, las semividas inducidas por la luz solar estimadas para la 2,3,7,8-TeBDD (TBDD) eran de 0,8 min (en solución orgánica) o de 32 horas (en dispersión como láminas sólidas). Se calculó una semivida de 3–6 meses para los isómeros tetraBDD en el suelo superficial. En comparación con los PCDD y PCDF, los correspondientes compuestos bromados presentaban menos estabilidad fotoquímica. Los PXDD y PXDF pierden de preferencia sus átomos de bromo durante la fotolisis, siendo transformados en PCDD y PCDF, que tienen semividas fotolíticas más largas. Esa transformación de PXDD/PXDF en PCDD/PCDF se produce también durante los procesos de incineración.

Los PBDD y PBDF parecen ser escasamente degradables por la acción de los microorganismos.

Como se ha observado en algunos estudios, la presencia de PBDD y PBDF en animales y seres humanos indica su potencial de acumulación. La 2,3,7,8-TeBDD se acumula en ratas durante la administración subcrónica. No se dispone de los factores de bioacumulación, bioconcentración o bioamplificación de los PBDD/PBDF o PXDD/PXDF.

4. Niveles ambientales y exposición humana

Hasta la fecha, en contraste con los PCDD y PCDF, los PBDD y PBDF no se han incluido con frecuencia en programas de vigilancia. Los pocos estudios realizados muestran una aparición limitada.

En el aire ambiental, los PBDF se encuentran con más frecuencia que las PBDD. Sólo se han detectado PBDD bromados inferiores (mono a tetra) en concentraciones que iban de las indetectadas a las de 0,85 pg/m^3 aproximadamente para las monobromodibenzo-*p*-dioxinas (monoBDD o MoBDD) en un túnel de carretera y en un garaje subterráneo. Entre los PBDF se hallaron homólogos mono a hexabromados, en concentraciones que iban del nivel indetectado a 74 pg/m^3. Por ejemplo, en Alemania se midieron las concentraciones (valores medios) de los PBDD y PBDF totales (tri a hexa) en un túnel de carretera, en el centro de una ciudad y en una zona suburbana, obteniendo valores de 23 pg/m^3, 2 pg/m^3 y 0,59 pg/m^3, respectivamente; no se detectó la 2,3,7,8-TeBDD y las concentraciones máximas de 2,3,7,8-TeBDF y 1,2,3,7,8-PeBDF fueron de 0,28 pg/m^3 y 0,08 pg/m^3, respectivamente. Se hallaron PXDF en muestras de aire en zonas de tráfico en concentraciones de hasta 41 pg/m^3 (Cl_1Br_1DF). En las muestras de polvo tomadas al aire libre (principalmente en carreteras) se observó también un predominio de PBDF y PXDF (valores máximos de varios miles de ng/kg) respecto a las PBDD y PXDD (valores máximos de hasta unos centenares de ng/kg).

Las muestras de aire tomado de locales equipados con distintos dispositivos electrónicos en funcionamiento (televisores o monitores de ordenador) mostraron la presencia de PBDF (tetra a hepta) en concentraciones totales que iban de 0,23 a 1,27 pg/m^3. No se detectaron PBDD. Las muestras de polvo recogidas en un local de ordenadores dieron concentraciones totales de PBDF de 2,4–5,5 µg/kg de polvo. En contraste con el aire, la distribución homóloga en el polvo está dominada por los hexaBDF y los heptabromodibenzofuranos (heptaBDF o HpBDF). Sólo en las muestras de polvo se hallaron concentraciones bajas de tetraBDD (hasta 1 µg/kg) y de tetra y pentaBDF sustituidos en las posiciones 2,3,7 y 8 (hasta 0,07 µg/kg) detectables. Las concentraciones de PBDF en una muestra de polvo doméstico eran inferiores en un factor de 10. La concentración sumada de PBDD y PBDF fue igual a la de PCDD y PCDF en el polvo tomado

de locales de ordenadores, pero inferior a la de PCDD y PCDF en el polvo doméstico. El polvo tomado en un garaje subterráneo contenía PBDF (mono y di) y PXDF (di a tetra) halogenados inferiores, con una concentración máxima de 4,3 µg/kg en el caso de los dibenzofuranos dihalogenados mixtos (DiXDF).

No se dispone de datos sobre las concentraciones de PBDD y PBDF en las muestras de agua.

En las muestras de sedimentos de río y mar tomados en una zona industrializada se detectaron tetraBDD (hasta 0,006 µg/kg de peso en seco) y tetra a hexaBDF (en conjunto hasta 0,37 µg/kg de peso en seco). El sedimento procedente de un drenaje de carretera contenía PBDF (suma de mono a tri: 2,5 µg/kg; suma de tetra a hepta: 0,3 µg/kg) y PXDF (suma de di y tri: 1,85 µg/kg), pero no PBDD.

Asimismo, las muestras de suelo tomadas cerca de una carretera contenían monobromodibenzofuranos (monoBDF o MoBDF) y dibromodibenzofuranos (DiBDF) (suma: 1,3 µg/kg) tetra y pentaBDF (suma: 0,02 µg/kg) y PXDF (suma: 1 µg/kg), pero no PBDD. Las muestras de suelo tomadas de un terreno de incineración y cerca de una fábrica de recuperación de metales dieron concentraciones totales de PBDF de hasta 100 µg/kg, pero sin detectar PBDD. En una serie de muestras de fango de alcantarillado procedentes de plantas municipales de tratamiento de aguas residuales se hallaron concentraciones totales de PBDF comprendidas entre niveles indetectados y 3 µg/kg. En un caso se hallaron valores infinitesimales de tetraBDD y 2,3,7,8-TeBDF. Una muestra de abono biológico estaba casi exenta de PBDD y PBDF (tetraBDF: <0,003 µg/kg).

No se dispone de datos cuantitativos sobre las concentraciones de PBDD y PBDF en los alimentos.

En muestras de hierba y de agujas de pino recogidas cerca de carreteras se encontraron PBDF y PXDF halogenados inferiores (mono a tetra) y valores infinitesimales de PBDD y PXDD (mono a tri).

No se han encontrado PBDD ni PBDF en las escasas muestras de animales o plantas silvestres analizados.

En la leche de vaca recogida en granjas lecheras cerca de una instalación incineradora de desechos municipales se identificaron de modo provisional tribromodibenzofuranos (triBDF o TrBDF), un tetraBDF y un pentaBDF (no tenían el tipo de sustitución en las posiciones 2,3,7 y 8).

No se han detectado PBDD ni PBDF en las escasas muestras analizadas de tejidos adiposos humanos o de muestras de leche procedentes de la población general.

Es posible la contaminación por PBDD y PBDF en distintos lugares de trabajo en donde se procede a producir, elaborar, utilizar o eliminar ciertos pirorretardantes o sus productos, en particular si se emplean altas temperaturas. La magnitud de la exposición del trabajador depende no sólo de los productos utilizados sino también de la calidad del aire y de las condiciones de ventilación. Se dispone de escasos datos de vigilancia del lugar de trabajo procedentes de instalaciones de producción o elaboración de plásticos, de oficinas o de estudios con un alto número de dispositivos eléctricos en funcionamiento continuo y de instalaciones de reciclado (incluidas plantas de reciclado de cobre). Por lo general, los PBDF eran más abundantes que las PBDD y las concentraciones en el aire de PBDF eran superiores en los lugares de producción de polímeros que contenían DBDE. En numerosas muestras se detectaron PBDF y PBDD con sustituciones en las posiciones 2,3,7 y 8. También se halló contaminación por PBDD y PBDF en la zona de trabajo comprendida debajo de la chimenea de humos de un laboratorio químico. Se carece de datos de vigilancia procedentes de instalaciones de incineración de desechos.

5. Cinética y metabolismo

La mayor parte de los estudios se refieren a la 2,3,7,8-TeBDD y, en menor cuantía, al 1,2,7,8-TeBDF. Los cálculos de la semivida han comprendido algunos congéneres adicionales.

La 2,3,7,8-TeBDD se absorbió en ratas después de la administración oral, intratraqueal y cutánea, variando el porcentaje de absorción conforme a la vía y la dosis. Las dosis únicas de 1 nmol de 2,3,7,8-TeBDD/kg de peso corporal condujeron a la absorción del

80% (vías oral e intratraqueal) o el 12% (vía cutánea) de la dosis administrada. La absorción cutánea de 1 nmol de 1,2,7,8-TeBDF/kg de peso corporal fue del 29% aproximadamente. La absorción oral de 2,3,7,8-TeBDD pareció ser comparable a la de la 2,3,7,8-tetraclorodibenzo-*p*-dioxina (2,3,7,8-TeCDD o TCDD). Sin embargo, la absorción cutánea de 2,3,7,8-TeBDD fue la tercera parte aproximadamente de la dosis equimolar de 2,3,7,8-TeCDD.

La 2,3,7,8-TeBDD o el 1,2,7,8-TeBDF administrados a ratas, por cualquier vía, se distribuyeron por todo el organismo, hallándose los principales depósitos en los tejidos hepático y adiposo, seguidos de la piel y el tejido muscular. Por ejemplo, 3 días después de la administración de dosis orales únicas de 2,3,7,8-TeBDD (1 nmol/kg de peso corporal), las porciones halladas en esos tejidos eran del 20%, el 20%, el 11% y el 4% respectivamente, mientras que el timo y las glándulas suprarrenales contenían el 0,03% y el 0,4%, respectivamente, de la dosis administrada. La partición de la 2,3,7,8-TeBDD entre el hígado y el tejido adiposo de ratas estaba influida por la dosis, la vía de exposición y el tiempo transcurrido después de la administración. Las relaciones entre las concentraciones del hígado y el tejido adiposo medidas en distintas condiciones variaban entre 0,2 y 6,5 (gama para dosis únicas de 2,3,7,8-TeBDD en ratas). No se dispuso de datos experimentales sobre la transferencia de PBDD y PBDF a las crías.

Se hallaron metabolitos de tetraBDD/BDF en la bilis y las heces de ratas. Se formaron principalmente por hidroxilación aromática y debromación hidrolítica. La tasa de metabolismo (determinada indirectamente como tasa de excreción biliar) difería entre la 2,3,7,8-TeBDD (el 7% aproximadamente) y el 1,2,7,8-TeBDF (el 50% aproximadamente). Tres días después de la administración intravenosa de una dosis de 2,3,7,8-TeBDD (1 nmol/kg de peso corporal), el 14% de la dosis administrada se halló en forma de metabolitos en las heces de ratas.

Se estudiaron en ratas la eliminación y excreción de la 2,3,7,8-TeBDD utilizando las vías de administración oral, intravenosa, intratraqueal y cutánea. En todos los estudios, la principal vía de eliminación fue las heces, variando la radiactividad eliminada entre el 2% (vía cutánea) y el 42% (vía oral) de la dosis administrada (1 nmol de [^{3}H]2,3,7,8-TeBDD/kg de peso corporal) en muestras de heces, y

entre el 0,2 y el 1% en muestras de orina. Asimismo, en estudios del 1,2,7,8-TeBDF en ratas, la excreción se produjo principalmente por las heces y sólo se eliminó por la orina el 2–3% de las dosis intravenosa, oral o cutánea. En los primeros días que siguieron a la administración de las dosis orales, el material no absorbido y la excreción biliar parecieron ser las principales fuentes de sustancia eliminada por las heces. Las porciones de 2,3,7,8-TeBDD original hallado en heces de ratas después de la administración de 1 nmol de 2,3,7,8-TeBDD/kg de peso corporal fueron del 53% (vía oral), el 43% (vía intratraqueal) y el 10–20% (vía intravenosa). Pocos días después de la administración oral de 2,3,7,8-TeBDD (1 nmol/kg de peso corporal), el 20% aproximadamente de la dosis administrada se eliminó como sustancia original.

Se dispone de datos sobre la retención y el ciclo biológico en el caso de algunos PBDD y PBDF. En las ratas, la carga corporal relativa de 2,3,7,8-TeBDD (y otros congéneres) depende de la vía y de la dosis administrada, mostrando diferencias en la absorción. Se calcularon las semividas de varios PBDD/PXDD y PBDF en distintos tejidos y en heces de ratas. Variaron entre un día (1,2,7,8-TeBDF en el organismo en conjunto) y 99 días (2,3,4,7,8-PeBDF en el hígado). Las semividas calculadas de 17,18 y 58 días para la 2,3,7,8-TeBDD en el hígado, las heces y el tejido adiposo, respectivamente, fueron análogas a las señaladas para la 2,3,7,8-TeCDD en el hígado y las heces, pero superiores (en un factor de >2) a las registradas para la 2,3,7,8-TeCDD en el tejido adiposo. Pese a las diferencias en la retención inicial, las semividas de la 2,3,7,8-TeBDF y el 2,3,7,8-tetraclorodibenzofurano (2,3,7,8-TeCDF o TCDF) en el hígado fueron comparables.

En lo que respecta a los PCDD y PCDF, las semividas calculadas en personas son mucho más largas que las correspondientes a ratas. Se dispone de estimaciones de 3–11 años (promedio: 5,9 años) para la 2,3,7,8-TeBDD y de 1–2 años (promedio: 1,5 años) para el 2,3,7,8-TeBDF. También se observó la persistencia de esas sustancias en el caso de un químico que sintetizó 2,3,7,8-TeBDD y 2,3,7,8-TeCDD en 1956. A los 35 años de la exposición se hallaron en su sangre concentraciones muy elevadas de 2,3,7,8-TeBDD.

6. Efectos en mamíferos de laboratorio y en sistemas de pruebas *in vitro*

La mayor parte de los estudios se refieren a la toxicidad de la 2,3,7,8-TeBDD, pero también se dispone de alguna información sobre otros PBDD/PBDF y PXDD/PXDF.

La 2,3,7,8-TeBDD produjo efectos típicos análogos a los de la 2,3,7,8-TeCDD, incluidos el síndrome de consunción, la atrofia tímica y la toxicidad hepática. Se observaron además lesiones hepáticas descritas como púrpura hepática, que no se habían registrado después de la exposición de ratas a la 2,3,7,8-TeCDD. El tipo de lesiones (mortalidad, histopatología, pesos del hígado y el timo) hallado en cobayos después de una sola exposición y en ratas después de la exposición a corto plazo al 2,3,7,8-TeBDF fue análogo al observado en el caso del 2,3,7,8-TeCDF.

La 2,3,7,8-TeBDD mantiene una interacción con el sistema endocrino. En ratas se han observado alteraciones relacionadas con la dosis en las hormonas tiroideas circulantes y alteración de la actividad espermatogénica.

La DL_{50} oral (periodo de observación de 28 días) de la 2,3,7,8-TeBDD en ratas Wistar fue de 100 µg/kg de peso corporal aproximadamente en las hembras y de 300 µg/kg de peso corporal aproximadamente en los machos. Los valores de la DL_{50} oral para la 2,3,7,8-TeCDD obtenidos en otros estudios variaron entre 22 y >3000 µg/kg de peso corporal. Las dosis equimolares de 2,3,7,8-TeBDF y de 2,3,7,8-TeCDF dieron tasas de mortalidad comparables en cobayos. Por ejemplo, se observó una mortalidad del 100% después de la administración de 2,3,7,8-TeBDF (0,03 µmol/kg de peso corporal, 15,8 µg/kg de peso corporal) y de 2,3,7,8-TeCDF (0,03 µmol/kg de peso corporal, 10 µg/kg de peso corporal). Se observaron en ratas lesiones prepurpúreas y modificaciones de las hormonas tiroideas después de la administración de una sola dosis de 100 µg/kg de 2,3,7,8-TeBDD/kg de peso corporal.

En ratas Wistar a las que se administró 2,3,7,8-TeBDD por vía oral durante 13 semanas se observaron signos de disminución de la espermatogénesis, presencia de espermatocitos defectuosos y

necróticos, signos de púrpura hepática grave y modificaciones de las hormonas tiroideas circulantes y de los pesos de los órganos. El nivel de efectos adversos no observados fue de 0,01 μg/kg de peso corporal por día.

La administración oral de 2,3,7,8-TeBDF a ratas Sprague-Dawley durante 4 semanas provocó retraso del crecimiento dependiente de la dosis y lesiones histopatológicas en el hígado y el timo. El nivel de efecto adverso no observado fue de 1 μg/kg de peso corporal por día.

En ratones se observó la aparición de toxicidad en el desarrollo en el caso de algunos PBDD y PBDF sustituidos en las posiciones 2,3,7 y 8 administrando dosis subcutáneas y orales que no provocaron toxicidad materna ni mortalidad fetal. Los niveles de efectos mínimos observados (en μg/kg de peso corporal) para la hidronefrosis y el paladar hendido, después de una sola dosis oral, en ratonas gestantes fueron, respectivamente, los siguientes: 3 y 48 para la 2,3,7,8-TeBDD, 25 y 200 para el 2,3,7,8-TeBDF, 400 y 2400 para el 2,3,4,7,8-PeBDF, y 500 y 3000–4000 para el 1,2,3,7,8-PeBDF. En comparación con la base molar, la 2,3,7,8-TeBDD y la 2,3,7,8-TeCDD presentaron casi la misma actividad en la inducción de la hidronefrosis. Al efectuar la comparación con el peso, los isómeros bromados fueron en general menos potentes que los clorados en la inducción de la hidronefrosis y el paladar hendido. Sin embargo, el 2,3,7,8-TeBDF fue más activo que el 2,3,7,8-TeCDF.

No se halló información sobre la mutagenicidad de los PBDD y PBDF o puntos finales conexos.

No se dispuso de estudios sobre la toxicidad y la carcinogenicidad a largo plazo con PBDD y PBDF. La 2,3,7,8-TeBDD resultó positiva en una prueba de transformación celular utilizando macrófagos peritoneales murinos. Sin embargo, la actividad transformadora de la 2,3,7,8-TeBDD fue siete veces menor que la de la 2,3,7,8-TeCDD. Más tarde aparecieron tumores en ratones lampiños tras la inyección subcutánea de las estirpes celulares establecidas resultantes.

La administración intraperitoneal de una serie de varias PBDD y PXDD (tetra y penta) a ratas Wistar inmaduras de sexo masculino produjo pérdidas de peso 14 días después de la inyección. Basándose

en los valores de DE_{50} molar, las sustancias más tóxicas ensayadas fueron las 2,3,7,8-TeBDD, 2-Br_1-3,7,8-Cl_3-DD y 2,3-Br_2-7,8-Cl_2-DD (TBCDD), con sustituciones sólo en las cuatro posiciones laterales. Las actividades relativas de las demás PBDD examinadas siguieron el siguiente orden: 2,3,7,8- > 1,2,3,7,8- >1,2,4,7,8- > 1,3,7,8-DD. En otros experimentos sólo se observaron ligeras diferencias en los valores de la DE_{50} (sobre una base molar) para la pérdida de peso total, la atrofia tímica y la inducción de las enzimas hepáticas entre la 2,3,7,8-TeCDD y la 2,3,7,8-TeBDD.

Se observaron atrofia tímica y otros signos de immunotoxicidad (por ej., parámetros hematológicos y alteraciones de ciertas subpoblaciones de linfocitos) con la administración de varias PBDD/PXDD y de 2,3,7,8-TeBDF en la rata y con las 2,3,7,8-TeBDD y TBCDD en el mono titi (*Callithrix jacchus*). Se llegó a la conclusión de que, sobre una base molar, la actividad de la 2,3,7,8-TeBDD es comparable a la de la 2,3,7,8-TeCDD en ratas y monos. Por ejemplo, se observó un efecto notable en cierta subpoblaciones de linfocitos en monos después de una sola dosis subcutánea de 30 ng de 2,3,7,8-TeBDD/kg de peso corporal en relación con 10 ng de 2,3,7,8-TeCDD/kg de peso corporal. No se han investigado los efectos sobre la immunotoxicidad después de la exposición perinatal a los PBDD y PBDF.

Tras la administración subcrónica de 2,3,7,8-TeBDD o 2,3,7,8-TeCDD por cebado oral de ratones se produjo un aumento dependiente de la dosis en las profirinas hepáticas totales.

Dosis orales únicas de 2,3,7,8-TeBDD y 2,3,7,8-TeCDD produjeron reducciones en la concentración y la cantidad total de vitamina A en el hígado de ratas, siendo la 2,3,7,8-TeBDD ligeramente menos potente que la 2,3,7,8-TeCDD (sobre una base molar).

La 2,3,7,8-TeBDD y el 2,3,7,8-TeBDF produjeron hiperqueratosis en la oreja del conejo en una dosis de 100 µg/conejo, pero no con 10 µg/conejo. El nivel de efecto no observado para la 2,3,7,8-TeCDD fue de 0,01 µg/conejo.

Se observó que varios congéneres halogenados tetra (Br_1Cl_3DD, Br_2Cl_2DD) y penta (Br_1Cl_4DD) con sustitución en las posiciones 2,3,7 y 8 presentaban una actividad antiestrogénica análoga a la de la de

2,3,7,8-TeCDD, examinada en cultivos de células de cáncer mamario humano.

En ratas, la 2,3,7-tribromodibenzo-*p*-dioxina (2,3,7-triBDD/TrBDD) reducía la desaparición de la uabaína del plasma, su eliminación por la bilis y el flujo biliar en una amplitud ligeramente inferior a la observada con la 2,3,7,8-TeCDD.

Los PBDD/PBDF y PXDD/PXDF son potentes inductores de ciertas enzimas microsómicas dependientes del citocromo P-450. Se calcularon valores de DE_{50} de 0,8–1 nmol/kg de peso corporal para la inducción del citocromo P-1A1 y de 0,2 nmol/kg de peso corporal aproximadamente para la inducción del citocromo P-1A2 en el hígado de rata tras la administración oral de dosis únicas de 2,3,7,8-TeBDD. Se observó la inducción del citocromo P-1A1 (inducción de la hidroxilasa de arilhidrocarbono y/o la etoxirresorrufina-*O*-desetilasa) en distintas especies y tejidos *in vivo* y en cultivo celular de rata *in vitro*. Se observó que distintos congéneres eran activos, así como los pirolizados de ciertos pirorretardantes. Por lo general, la inducción enzimática dependía de la dosis en concentraciones no tóxicas, comenzaba después de la exposición y era duradera. Resultó mensurable en exposiciones tan bajas como las situadas en la gama de pmol. La actividad inductora varió en varios órdenes de magnitud para distintos congéneres, en función de su estructura química. Los inductores más potentes fueron las TCDD, TBDD y TBCDD. En comparación (sobre una base molar) con sus análogos clorados, las PBDD y PXDD tenían más o menos igual actividad. En contraste con la TCDD, cuya actividad inductora relativa era independiente del tejido examinado, la TBDD era cinco veces más activa en la inducción de la etoxirresorrufina-*O*-desetilasa en el hígado que en la piel y el pulmón después de la exposición subcrónica de ratones. La clasificación de la inducción de la actividad de la etoxirresorrufina-*O*-desetilasa en monos titís fue de TCDD > 2,3,4,7,8-pentaclorodibenzofurano > 2,3,4,7,8-pentaCDF/PeCDF > 2,3,4,7,8-PeBDF cuando se compararon las actividades enzimáticas con las concentraciones hepáticas. En las pruebas *in vitro* con cultivos de células de rata se obtuvieron valores de la CE_{50} molar análogos para las actividades de inducción de la hidroxilasa del arilhidrocarbono y de la etoxirresorrufina-*O*-desetilasa entre los PXDF y PCDF correspondientes.

Se estima que los PBDD y PBDF comparten un mecanismo común de acción con los PCDD y PCDF y otros hidrocarburos aromáticos halogenados. Se confirmó el enlace con el receptor de hidrocarburos aromáticos citosólico, que desempeña una función central en la mediación de la toxicidad afín a la de la 2,3,7,8-TeCDD, en el caso de varios PBDD y PXDD/PXDF. Sus afinidades de enlace con los receptores variaron en varios órdenes de magnitud, pero fueron comparables a las de sus análogos clorados.

7. Efectos en el ser humano

No se dispone de datos sobre la exposición de seres humanos a los PBDD y PBDF o sobre sus efectos en la salud de la población general.

Se han registrado dos casos de problemas de salud agudos debidos a la exposición a 2,3,7,8-TeBDD/TeCDD, con síntomas que comprendían el cloroacné.

En otro estudio, el personal masculino de una fábrica de productos químicos con exposición documentada a los PBDD y PBDF procedentes del uso de pirorretardantes bromados (OBDE y DBDE) fue sometido a pruebas de laboratorio inmunológicas y clínicas adicionales. Aunque se observaron indicios de modificaciones menores de los parámetros inmunológicos, la evaluación global de su estado de salud no mostró un efecto de la carga corporal de 2,3,7,8-TeBDD/ TeBDF sobre el sistema inmunitario.

No existen informes sobre la mortalidad cancerosa producida por los PBDD y PBDF.

8. Efectos en otros organismos en el laboratorio y en el medio ambiente

Sólo se dispone de información limitada sobre los efectos de los PBDD y PBDF en microorganismos, plantas, invertebrados o especies silvestres vertebradas.

En una biovaloración de la mortalidad precoz de pececillos de trucha irisada (*Oncorhynchus mykiss*), se ensayó una serie de

congéneres de PBDD y PBDF, que resultaron activos. Esta biovaloración demostró también que tanto las PBDD como los PBDF tienen menor actividad al aumentar la sustitución por bromo. Tanto la 2,3,7,8-TeBDD como el 2,3,7,8-TeBDF eran más activos que sus análogos clorados.

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