Risks and management of dioxin-like compounds in Baltic Sea fish: An integrated assessment

Timo Assmuth and Pauliina Jalonen

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TABLE OF CONTENTS

List of figures

List of tables

23. Tentative semi-quantitative assessment of relative risks from dioxin-like compounds in Baltic Sea fish, other fish and other sources in some Baltic Sea countries and reference regions
24. The influence of some Baltic Sea factors on risks associated with dioxin-like compounds in its fish
25. Comparative assessment of human health impacts of ingredients in fatty Baltic Sea fish, including beneficial effects
26. Distribution of human health risks and benefits from Baltic Sea fish among population segments
27. Characterizations of the treatment of some key uncertainties in relevant authoritative international and national assessments of human health risks from dioxins and in proposals for human tolerable daily intakes
28. Summary evaluation of ecotoxicological risks from dioxin-like compounds in fish-consuming Baltic Sea animals by comparison of body burden or diet levels and effect or no-effect levels for various endpoints and TEqs, estimated from experimental and field studies
29. Policies and instruments relevant to the management of PCDD/Fs and PCBs in Baltic Sea fish, including their sources
30. The status of regulations and actions in the management of PCDD/Fs and PCBs in Baltic Sea countries
31. Implemented and possible new risk management measures for chemicals produced or used in EU that are potential precursors of PXDD/Fs
32. Summary of established and potential reduction measures for dioxin emissions in major source categories, emphasizing industrial and waste management processes
33. Development status of demonstrated dioxin and PCBs treatment technologies applicable to environmental matrices, emphasizing full-scale applications, sediment treatment and destruction (based mainly on OTA 1991, NRC 2001, USEPA 2003a,b, FRTR 2004)
34. Comparative evaluation of main classes of risk management options for dioxin-like compounds in the Baltic Sea, with particular reference to its fish and to technical characteristics of options
35. Estimated social-economic conflict potential due to the EU recommendations for maximum levels of toxic PCDD/Fs in food and feeding-stuff (EC 2002a) for the fisheries industry in Denmark, Finland, Sweden and whole EU, in 1000 t a^{-1} and M€ a^{-1} (from Joas et al. 2001)
36. Summarizing evaluation of two principal alternative approaches to management of risks mainly to human health associated with dioxin-like compounds in Baltic Sea fish, focusing on the intake stage
37. Key policy development and decision making needs, knowledge and opportunities regarding dioxin-like compounds in Baltic fish and environment, emphasizing knowledge use in integrated assessment and evaluation of strategic options.

List of abbreviations and acronyms used repeatedly outside tables, figures and references

Abbreviation	Explanation
a	years
ABS	acrylonitrile butadiene styrene
AC	activated carbon
AC/CC	accession country/candidate country (to EU)
ACOH	acetanilide 4-hydroxylation
ADI	average daily intake
AHH	aryl hydrocarbon (often BaP) hydroxylase
AhR	aryl hydrocarbon receptor
AHRR	aryl hydrocarbon receptor repressor
AMAP	Arctic Monitoring and Assessment Program
AOC	area of concern
ARNT	AhR nuclear translocator
AS	Arkona Sea
AST	aspartate amino transferase
B cell	a lymphocyte derived from bone marrow that provides humoral immunity
BaP	benzo(a)pyrene
BAT	best available technology
BB	Bothnian Bay
BCF	bioconcentration factor
BDD/F	see PBDD/Fs
BDE	bromodiphenyl ether
BDE47	a pentabromodiphenyl ether congener
BDE99	a pentabromodiphenyl ether congener
BEP	best environmental practice
BFR	brominated flame retardant
BMD	benchmark dose
BMD01,10	BMD to 01, 10 % of test organisms
BMF	biomagnification factor
BMI	body mass index
BP	Baltic Proper
BREF	BAT reference
BS	Baltic Sea
BSAF	Biota-sediment accumulation factor
C	carbon
CAD	coronary artery disease
CAFE	Clean Air For Europe program
CALUX-TEq	DR-CALUX-based TEq value
CB	chlorobiphenyl (general)
CCA	Cr/Cu/As salt wood preservative
CD	T cell (cytotoxic leucocyte) marker
CD4+/CD8+	Ratio of T cells that express the CD4 and CD8 antigen, respectively
CDD/F	chlorodibenzo-p-dioxin/ dibenzofuran (gen.)
CE	cholesteryl ester
Cion	European Commission
CERCLA	Comprehensive Environmental Responsibility, Compensation and Liability Act (US)
CFP	Common Fisheries Policy (of EU)
CH_4	methane
CI	confidence interval
Cl	chlorine
CL	confidence limit
CN	chloronaphthalene
CNP	chloronitrofen (a herbicide)
CO	carbon monoxide
CO_2	carbon dioxide
coPBB	coplanar PBB
coPCB	coplanar PCB
CORINAIR	CO-oRdinated INformation on the Environment in the EC – AIR part
CP	chlorophenol (gen.), cf. PCP
Cu	copper
CV	coefficient of variation
CYP	cytochrome P450 phase I enzyme family (e.g. Cyp1a1, Cyp1a2, Cyp1b1)
d	days
1-D, 3-D	one- and three-dimensional
2,4-D	2,4-dichlorophenoxyethanoic acid
DBDE	decabromodiphenyl ether
DCDD	dichlorodibenzo-p-dioxin
DDT (p,p'-DDT)	1,1,1-trichloro-2,2-bis(p-chlorophenyl)ethane
DDE (p,p'-DDE)	1,1-dichloro-2,2-bis(p-chlorophenyl)-ethylene
DFP	2378-CDD/Fs and (dl-)PCBs

DG	Directorate-General (of Cion and EC)
DHA	docosahexaenoic acid (a PUFA)
DK	Denmark
DLC	dioxin-like compound
dlPCB	dioxin-like PCB
DN	Dagens Nyheter
DNA	deoxyribonucleic acid
DOC	dissolved organic carbon
DR-CALUX	H4IIE-luc/DR-Chemical Activated Luciferase eXpression (DLC bioassay)
DRE	dioxin-responsive element
EAF	electric arc furnace
EAP	environmental action program (EU)
EC	European Community
EC50	effective concentration to 50 %
ED	effective dose
ED01,ED10,ED50	effective dose to 1, 10 and 50 %
EEA	European Environmental Agency
EFSA	European Food Safety Agency
EGF	epidermal growth factor
EGFR	epidermal growth factor receptor
EMAS	Eco-Management and Audit Scheme
EMEP	Cooperative Programme Monitoring Eval Long-Range Transp Air Pollut in Europe
EMS	environmental management system
EOCl	Extractable organochlorine
EPA	eicosapentaenoic acid (a PUFA)
EPER	European Pollution Emission Register
ER	estrogen receptor
EROD	7-ethoxyresorufin O-deethylase
EST	Estonia
EU	European Union
EVA	ethylene vinyl acetate (polymer)
FAO	Food and Agricultural Organization (UN)
FI	Finland
fT3	free triiodothyronine
fT4	free thyroxin
GB	Gulf of Bothnia
GC	gas chromatography
GF	Gulf of Finland
GIS	geographical information system
GLBTS	Great Lakes Binational Toxics Strategy
GM	genetically modified
GR	Gulf of Riga
h	hour
HCBz	hexachlorobenzene
HCDF	hexachlorodibenzofuran
HCl	hydrogen chloride
HCH	hexachlorocyclohexane (α-, β- and γ-)
HDL	high-density lipoprotein (cholesterol)
HELCOM	Helsinki Convention (for BS protection)
Hg	mercury
HpCDD	heptachlorodibenzo-p-dioxin
HpCDF	heptachlorodibenzofuran
6-HpCDF	1,2,3,4,6,7,8-heptachlorodibenzofuran
HPLC	high performance liquid chromatography
HPV	high production volume (chemical)
HRGC	high resolution gas chromatography
HRMS	high resolution mass spectrometry
HxCDD	hexachlorodibenzo-p-dioxin
6-HxCDD	2,3,4,6,7,8-hexachlorodibenzo-p-dioxin
HxCDF	hexachlorodibenzofuran
4,6-HxCDF	2,3,4,6,7,8-hexachlorodibenzofuran
6-HxCDF	1,2,3,6,7,8-hexachlorodibenzofuran
IARC	Int Agency for Research on Cancer
IBSFC	Int Baltic Sea Fisheries Commission
ICES	Int Council for the Exploration of the Sea
ICZ	indolo[3,2-b]carbazole
ICZM	integrated coastal zone management
IgA	Immunoglobulin A
IgE	Immunoglobulin E
IFN-gamma	interferon gamma (cytokine mediating resistance to infection/tumorigenesis)
IL	interleukin (inflammatory cytokine)

IMO	International Maritime Organization
IPCS	International Programme for Chemical Safety
IPPC	Integrated Pollution Prevention and Control (e.g. related EU directive)
IQ	intelligence quotient
ISO	International Standardization Organization
I-TEq	International (TCDD) toxicity equivalent
$I\text{-}TEq_{DF}$	I-TEq due to PCDD/Fs
$I\text{-}TEq_{P}$	I-TEq due to PCBs
IUPAC	Int Union for Pure and Applied Chemistry
IVL	Institutet för Vatten- och Luftvårdsforskning
JCP	Joint Comprehensive Program (HELCOM)
JECFA	Joint FAO/WHO Expert Committee on Food Additives
KAT	Kattegat
KemI	Kemikalieinspektionen, Swedish Chemicals Inspectorate
LA	Latvia
LC50	lethal concentration to 50 %
LCAT	lecithin cholesterol acyl transferase
LC	liquid chromatography
LD50	lethal dose to 50 % of test organisms
LT	Lithuania
LOAEL	lowest observed adverse effects level
LOD	limit of detection
LOEL	lowest observed effect level
LOQ	limit of quantitation
LRTAP	Long-Range Transport of Air Pollutants (UNECE Treaty)
LSI	liver somatic index
MACT	maximum available control technology (US)
MCPA	2-methyl-4-chloro-phenoxyacetic acid
MeHg	methyl mercury
$MeSO_2$-CB	methyl sulphone-CB (or –sulfone-CB)
M€	million Euro
mo	month
MROD	methoxyresorufin-O-deethylase
mRNA	messenger-RNA
MSC-E	Meteorological Synthesizing Centre-East
MS	mass spectrometry
MTBE	methyl tertiary butyl ether
n-3, n-6	n-3, n-6 (or ω-3, ω-6) PUFAs
NARAP	North American Regional Action Plan
NEC	no-effect concentration
NF	naphthoflavone
NGO	non-governmental organization
NH_4Cl	ammonium chloride
NIP	national implementation plan (under the Stockholm Convention); nitrofen (herbicide)
NK	natural killer (cell)
NL	The Netherlands
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NOEL	no-observed-effect level
NPDES	National Pollution Discharge Elimination System (US)
NRC	Natl Res Council of the Natl Academies (US)
NS	North Sea
N-TEq	Nordic TEq
0-*o*	non-*ortho* (PCB or PBB)
1-*o*	mono-*ortho* (PCB or PBB)
OCDD	octachlorodibenzo-p-dioxin
OCDF	octachlorodibenzofuran
OECD	Organization of Economic Cooperation and Development
OH	hydroxyl (group/anion)
OH-CB	hydroxylated chlorobiphenyl (chlorobiphenylol)
OR	odds ratio
OSPARCOM	Oslo-Paris Convention (for protection of N Atlantic)
PAH	polyaromatic hydrocarbon
Pb	lead
PBB	polybrominated biphenyl
PBCDD/F	polybromo-chlorodioxin/furan (cf. PCDD/F)
PBDD/F	polybromodibenzo-p-dioxin/dibenzofuran
PBDE	polybromodiphenyl ether
PBDT	polybromodibenzothiophene
PBN	polybromonaphthalene
PBPK	physiologically based pharmacokinetic (model)

PBTA	polybromothianthrene
PCAB	polychloroazobenzene
PCAOB	polychloroazoxybenzene
PCB	polychlorobiphenyl
PCBP	polychlorobiphenylene
PCB-TEq	cf. WHO-TEq_P
PC	polycarbonate
PCCE	poly(cyclohexylene dimethylene cyclohexanedicarboxylate) (polymer)
PCDD	(2,3,7,8-)polychlorodibenzo-p-dioxin
PCDD/F	polychlorodibenzo-p-dioxin/ dibenzofuran
PCDE	polychlorodiethyl ether
PCDF	(2,3,7,8-) polychlorodibenzofuran
PCDT	polychlorodibenzothiophene
PCHB	polychlorohydrazobenzene
PCN	polychloronaphthalene
PCN-TEq	TEq due to PCNs
PCP	polychlorophenol
PCSTB	polychlorostilbene
PCT	polychloroterphenyl
PCTA	polychlorothianthrene
pdf	probability distribution function
PDV	phocine (seal) distemper virus
PE	polyethylene (plastic)
PEC	predicted environmental concentration
4-PeBDF	2,3,4,7,8-pentabromodibenzofuran
PeCDD	pentachlorodibenzo-p-dioxin
PeCDF	pentachlorodibenzofuran
1-PeCDF	1,2,3,7,8-pentachlorodibenzofuran
4-PeCDF	2,3,4,7,8-pentachlorodibenzofuran
PeCP	pentachlorophenol
PHAH	polyhalogenated aromatic hydrocarbon
PL	Poland
PNEC	predicted no-effect concentration
POC	particulate organic carbon
POP	persistent organic pollutant
PP	polypropylene (polymer)
P-PUFA	EPA+DHA
PSU	polyether sulphone (or sulfone) (polymer)
PTS	persistent toxic substance
PUFA	polyunsaturated fatty acid
PVC	polyvinyl chloride
PVC-Cu	polyvinyl chloride coated copper (wire)
PXDD/F	polyhalodibenzo-p-dioxin/ dibenzofuran
QSAR	quantitative structure-activity relationship
QSPR	quantitative structure-property relationship
RA	rheumatoid arthritis
RAPIDS	Regional Air Pollutant Inventory Development Systems (US)
REACH	Registration, Evaluation and Authorization of Chemicals (in EU)
REP	relative (enzymatic) potency
RR	risk ratio
RsD	risk-specific dose
RSD	relative standard deviation
RUS	Russian Federation
SB	Southern Baltic Sea
SCALE	EU environment and health strategy (Science-Children-Awareness-Legal-Evaluation)
SCAN	Scientific Commitee on Animal Nutrition(EU)
SCF	Scientific Committee on Food (EU)
SCOOP	scientific cooperation project (EU)
SD	standard deviation
Se	selenium
SIR	standardized incidence ratio
S-E	South-East
SEM	standard error of measurement
STOA	scientific and technological options analysis
SW	Sweden
S-W	South-West
SYKE	Finnish Environment Institute
2,4,5-T	2,4,5-trichlorophenoxyethanoic acid
TAC	total allowable catch
TBBP-A	tetrabromobisphenol-A
TBDD	tetrabromodibenzo-p-dioxin
TBDF	tetrabromodibenzofuran
TCAB	tetrachloroazobenzene
TCAOB	tetrachloroazoxybenzene
TCB	a PCB mixture brand
TCBP	tetrachlorobiphenylene
TCBT	tetrachlorobenzyltoluene
TCBT87	3,3′,4,4′-Cl4-2-Methyltoluene
TCBT88	3,3′,4,4′-Cl4-5-Methyltoluene

TCDD	(2,3,7,8-)Tetrachlorodibenzo-p-dioxin
TCDF	(2,3,7,8-)Tetrachlorodibenzofuran
TCE	trichloroeth(yl)ene
TCF	total chlorine free (bleaching)
TCHB	tetrachlorohydrazobenzene
TCP	trichlorophenol
2,4,5-TCP	2,4,5-trichlorophenol
2,4,6-TCP	2,4,6-trichlorophenol
TCSTB	tetrachlorostilbene
TCTA	tetrachlorothianthrene
TDI	tolerable daily intake
TeCE	tetrachloroeth(yl)ene
TeCP	tetrachlorophenol
2346-TeCP	2,3,4,6-tetrachlorophenol
TEF	TCDD (toxicity) equivalency factor
TEq	(TCDD) toxic equivalent quantity
TEq_{DF}	toxic equivalent quantity of PCDD/Fs
TEq_{P}	toxic equivalent quantity of PCBs
TMDL	total maximum daily load (to Great Lakes)
TMI	tolerable monthly intake
TOC	total organic carbon
TSH	thyroid stimulating hormone
tT3	total triiodothyronine
tT4	total thyroxin
TTR	tranthyretin
TV	Television
TWI	total weekly intake
UDPGT	uridine diphosphoglucuronyl transferase
UF	uncertainty factor
UK	United Kingdom
UN	United Nations
UNECE	UN Economic Committee for Europe
UNEP	UN Environment Programme
UPOP	Unintentional POP
US	United States
USDA	US Department of Agriculture
USEPA	US Environmental Protection Agency
USFDA	US Food and Drug Administration
UV	ultraviolet (light/radiation)
VCM	vinyl chloride monomer
WHO	World Health Organization (UN)
WHO-TEF	WHO-TCDD toxicity equivalent factor
WHO-TEq	WHO-TCDD toxicity equivalent quantity
$WHO\text{-}TEq_{DF}$	WHO-TEq due to PCDD/Fs
$WHO\text{-}TEq_{P}$	WHO-TEq due to PCBs
ww	wet weight (fresh weight)
Zn	zinc

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by the knowledge and insights in many of these publications, and hope that in combining and evaluating this knowledge and in seeking to make sense of it we have done the authors adequate justice.

The authors are responsible for the information and views presented, and they should not be construed to necessarily represent those of their employer, funding organizations or others.

Preface

In the preface of her book 'fragments of Redemption - Jewish thought and literary theory in Benjamin, Scholem, & Levinas' (Indiana University Press 1991), Susan Handelman perceptively pointed out: "A preface is the first thing a reader of a book encounters, but it is often the last thing the author writes. … The closure of the work is delayed by the author's need to explain a critical question: To whom is this work addressed and for what purpose?"

Regarding the present book, the intended audiences include experts in the disciplines involved, such as toxicology and chemistry, public health and food safety, and fisheries and environmental management. The report may also be of interest, and is hoped to be of use, to other professionals and students, to decision makers in various realms, even to lay persons. Such a readership is foreseen due to the broad and keen interest in dioxins in general and in dioxins in the Baltic Sea and its fish in particular. As elaborated in the book, dioxins have indeed become a 'boundary object' uniting many disciplines and acting as an indicator and a symbol also for other concerns and issues. The potential relevance of the report is consequently not restricted to Baltic Sea or Nordic countries. Baltic Sea dioxins are a case of environmental contamination that has broader links, as other regions and systems affect and are affected by the risks and their management, and indirectly as an example of some universal issues.

As to the purpose, the book purports to synthesize and evaluate knowledge relevant for assessing and managing risks from dioxins and dioxin-like compounds in Baltic fish, including inquiry in what constitutes relevance, and on this basis to analyze scientific and policy aspects of these risks and their management. In lay terms, the intention is mainly to clarify what risks are associated with 'dioxins' in this context and what can be, ought to be, has been and is being done about them and on what grounds.

The extent of the report has been dictated by two main reasons. First, it spans a wide array of topics, from sources, fate and effects of dioxins and, notably, other dioxin-like compounds, to their management and associated policy and communication. Along with risks caused by these compounds in fish, risks of losing health and other benefits from fish are addressed. This broad coverage of problems and solutions is one difference between the report and previous assessments. Second, due to intensive long-time research but also to other activities, there is a wealth of information on these topics, relevant to the present case. An attempt has been made to utilize and discuss this information extensively, albeit in a synthesizing manner.

After the resultant fishing in the sea of knowledge, a picture or rather a series of pictures emerges, although limited, distorted and blurred like a view of a eutrophic sea to subsurface observers. It is evident that further assessment is needed and some opportunities for this have been indicated. Although plenty of information already exists, new basic knowledge is also needed, and can be produced in part by guidance from analyzes of the scientific basis for management and of management itself. Thus, although extensive, this document and the accompanying Annexes that are meant to form a separate web publication should be seen as modest reports of ongoing work at a certain stage of knowledge, and as background and discussion documents.

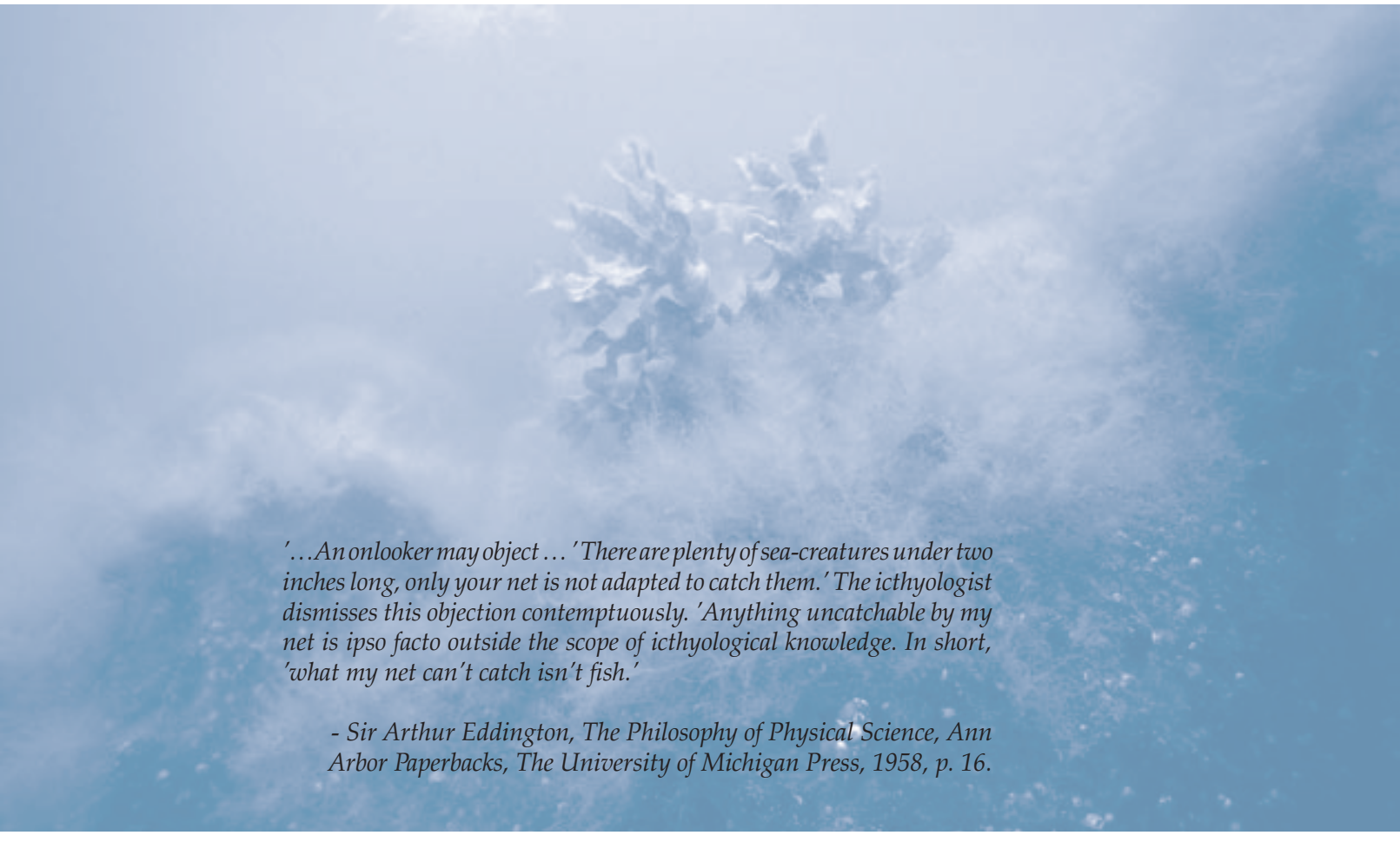
This finally leads to a methodological and philosophical aspect of assessment policy. Rather than providing definite answers, in many occasions the sensible thing has been to identify questions and alternative approaches to their solutions. As part of such a hermeneutic approach to risk and strategy assessment and also as part of continuous communication, a natural focus in the present work has been knowledge and its interactions with policy and decisions in dealing with risks and uncertainties of dioxins in this Baltic Sea context. That is, epistemological issues e.g. in the quality and significance of knowledge and in the modes of inference and argumentation have been stressed throughout. This also involves a self-reflexive approach, including awareness of limitations and deficiencies of the analysts themselves in providing sufficiently broad, detailed and balanced pictures of risks (cf. quote below).

Objective truths also - and in some ways particularly - about risks are elusive, hidden and slippery as many fish. However, open-minded inquiry, interpretation and discourse can help illuminate aspects of risks and guide human activity in dealing with them.

The authors wish that this work will aid in understanding and managing the multi-faceted risks, direct and indirect, associated with dioxin-like compounds in and around the Baltic Sea and its fish.

Helsinki, November 2005

Timo Assmuth and Pauliina Jalonen



'…An onlooker may object … 'There are plenty of sea-creatures under two inches long, only your net is not adapted to catch them.' The icthyologist dismisses this objection contemptuously. 'Anything uncatchable by my net is ipso facto outside the scope of icthyological knowledge. In short, 'what my net can't catch isn't fish.'

- Sir Arthur Eddington, The Philosophy of Physical Science, Ann Arbor Paperbacks, The University of Michigan Press, 1958, p. 16.

Introduction

1.1 General background and historical importance of dioxin problems

'Dioxins' are cyclic halogenated organic chemical compounds most commonly containing two benzene rings united by molecules in a third ring. Dioxins have gained much attention due to their toxicity and long-term health and environmental effects and risks.

Dioxins have been associated in common awareness with contamination cases like the Agent Orange herbicide spreading during the Vietnam war, the Seveso industrial accident in Italy in 1976, and recent food contamination episodes, all of which were widely publicized and caused debates, controversies and conflicts, policy changes, actions and other societal responses. Through such cases dioxins have become a symbol of environmental contamination that is related to more general factors and concerns.

Dioxins and associated risks and impacts have historical precedents and parallels. One of these is methyl mercury (MeHg) that caused alarms in the 1960's due to epidemic human health impairment and also due to ecotoxicity. More importantly in the Baltic Sea, polychlorinated biphenyls (PCBs) and persistent and bioaccumulating chlorinated pesticides have interacted with conceptions of and responses to dioxins. PCBs in particular are ever more integrated with dioxins in assessment and management, as some PCBs elicit dioxin-like toxicity; some of them indeed cause much of this toxicity in the Baltic, as will be shown and discussed in later chapters.

Dioxins are inadvertent by-products of many anthropogenic and even some natural processes. They are formed in most thermal processes involving chlorine e.g. in industry, waste management, energy production and transportation. They are also present as impurities in many chemical products, especially halogenated organic compounds.

'Dioxins' are not clearly defined. Only some of the 2,3,7,8-chlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs) have pronounced toxicity. On the other hand, an increasing amount of substances has been included in dioxin-like compounds, DLCs, encompassing compounds from many other groups than PCDD/Fs. Dioxins in common parlance and also in regulatory contexts include the above PCDD/Fs and increasingly also some dioxin-like PCBs. Dioxins thus cannot be identified by naming a single substance (although often the term has been used in a misleading way in singular) or even a distinct group of substances. This is one important factor that makes it difficult to get a clear picture of the effects, risks and problems they cause.

Dioxins have been studied due especially to accidents in production of chlorinated cyclic chemicals. The first reports of human exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) came after a 2,4,5-trichlorophenol reactor explosion in 1949 in West Virginia, US (see Sweeney and Mocarelli 2000). The first reports of harmful effects of chlorodioxins in the open scientific literature originate around 1960 (Kimmig and Schulz 1957, Jones and Krizek 1962). Thus, they have been known to be hazardous for half a century.

The perceived profile of dioxin risks has extended and evolved since then. The development of chemical analysis methods has allowed detecting lesser amounts and wider arrays of DLCs. Many of them are prone to long-range transport and occur near and far from sources, and accumulate in food chains (e.g., Iwata et al. 1993). TCDD and other DLCs have also been linked to an ever wider spectrum of toxicological effects in more animal species. Some effects occur at low doses. Effects are however not easily attributable to specific causes, and their adversity and significance is often unclear.

Risks from dioxins, DLCs and PCBs have been assessed by many international and national bodies (cf. Annex 11): IARC (1987, 1997), WHO (1991, Van den Berg et al. 1998, Van Leeuwen and Younes 2000), Ahlborg et al. (1989, 1992a,b, 1994), IPCS (1989, 1992, 1994a, 1998, 2001a); SCF (2000, 2001), JECFA (2001); USEPA (1994, 2000a), ATSDR (1998), HCN (1996a), COT (2001), SACN and COT 2004. There is still a lack of consensus on the evidence and the significance of risks of these

compounds. The general problem is to evaluate the cumulative effects from low doses of such mixtures in variable receptor organisms. The occurrence and significance of effects at the level of populations, communities and ecosystems and the relationship with other effects and causative factors are still unclear. In addition to scientific uncertainties, value-laden policy controversies prevail. Assessments depend e.g. on the criteria for data quality and the principles and procedures for interpretation.

In 1999 high levels of dioxin caused by contaminated feed were found in Belgian chicken meat and eggs, leading to product bans throughout Europe (see SCAN 2000). Also other dioxin and PCBs contamination episodes have occurred in food production systems. In 1950's they led to massive death of chickens in US, shown to be related to PCDDs (Higginbotham et al. 1968). The European Community (EC) published an assessment of European background levels of dioxins (Wenborn et al. 1999, cf. SCOOP 2000) stating that despite the reduction in human dietary exposure, some individuals or parts of the community might be "at risk". Together with other food safety scandals in late 1990's (such as the 'mad cow disease') the dioxin issue was thus brought to the European consumer health agenda. The change in the public trust on government authorities contributed to changes in food safety and environmental policies towards more risk adverse and precautionary approaches, prompting dioxin management initiatives especially in the food area.

Increasingly, dioxins have been considered in the context of the broader problem of persistent organic pollutants (POPs) and persistent bioaccumulative toxic substances (PBTs). However, food dioxin problems in particular led to the development of a European strategy for dioxins, furans and polychlorinated biphenyls in food and feeding-stuffs (EC 2001). There are also many other strategies, policies, programs and other initiatives that are relevant for the management of risks from dioxins in the Baltic Sea and its fish, on both EU, global, regional and national level, and in several branches of administration (for more detail, see 7). Dioxins can thus be seen as a "boundary object"[1] (e.g. Star and Griesemer 1989). Such objects bring different communities of practice together to solve a problem of common concern (Wenger 2000). Different communities of practice relevant to dioxins in Baltic Sea, both in research and management, form together a community of interest. However, they have differences in their conceptions of and responses to the dioxin problem and its solutions.

1.2 Particular issues in dealing with dioxins in Baltic Sea fish

1.2.1 Dioxins and dioxin-like compounds in the Baltic Sea

The Baltic Sea is among the marine areas most heavily *loaded* by dioxins and related compounds. It receives dioxins e.g. in precipitation, runoff and sediments, some from contamination decades past (e.g., Verta et al. 2003). Some of the load is degraded or buried in sediments, while a large part cycles in the sea and its food chains including man (cf. 3.3).

Continuous *high levels* of PCDD/Fs are consequently measured in fatty Baltic fish (cf. 3.4.2). Inventories of data on dioxin sources, emissions, levels and exposures in the Baltic Sea region have also been produced, some containing assessment elements (e.g., TWGIM 2004a,b). Other DLCs, such as dlPCBs, have also been found at high levels. The average *human intake* of dioxins from fish and totally has declined, but trends are variable or unclear. Declines of dioxin levels have also been seen in general human populations in Baltic Sea counties. For some population segments such as those consuming much fatty fish the exposures are still high (cf. 3.5.3). The *effects on human health* are still unresolved (cf. 4.2).

High levels of PCDD/Fs and PCBs have also been found in *non-human animals* feeding on Baltic fish, such as seals and fish-eating birds (cf. 3.4). *Adverse conditions* have been found in some of these populations, but are difficult to link conclusively with such exposures (cf. 4.3).

[1] *Similarly, because POPs are a concept simultaneously resident in science and policy worlds, they have functioned as a "boundary object" (Guston et al. 2000) around which international negotiations have coalesced (Selin and Eckley 2003).*

1.2.2 EU strategies and actions on dioxins, furans and PCBs and in related areas

In order to respond to environmental and health risks and subsequent other problems associated with dioxins the Commission of the European Communities (EC) has come up with various strategies, programs, plans, actions and initiatives. In addition to strategies and actions focused specifically on dioxins, others address dioxin-like compounds more generally, especially in the POPs area. For PCBs, measures on waste treatment and disposal have additionally been taken. Some of the strategies have acquired official, authoritative and legally binding status, being directly connected with legal measures (for more detailed discussion of these steering instruments, see Chapter 6).

The authoritative EU instruments specifically addressing PCDD/Fs and PCBs have been laid down in two main parts (cf. 6.3, Annex 13):

- Community strategy on *dioxins, furans and PCBs* (COM(2001)593 final, EC 2001) is a general strategy. Its objectives are to assess the state of the environment with regard to contamination and to reduce human and environmental exposures to these compounds. The strategy consists of two main parts: a) environment; b) feed and food (Van Tongelen 2002, Verstraete 2002). Short- to medium-term and long-term measures have been specified.
- Commission Recommendation (2002)836, EC 2002a) concerning the reduction of the presence of *dioxins, furans and PCBs in feeding-stuffs and foodstuffs* is based on part b) of the general dioxin strategy (cf. above), and addresses specifically the risks from these compounds in human food chains. These recommendations include measures on food and feeding-stuff control, especially through target, limit and action levels of dioxin contents.

As stipulated in the above Community strategy, a report on the progress of the Dioxin Strategy was adopted [COM (2004) 240, EC 2004a]. The activities listed include projects targeted on the new Member States, on integrated environment and health information focussed on the Baltic Sea, on best available techniques, research, limit values for dioxins in feed and food, and on screening methods.

1.2.3 Dioxin risk management by regulation of fish quality and fisheries

The dioxin content in Baltic herring can exceed many-fold the prescribed EU maximum level (e.g., Kiviranta et al. 2003). The inclusion of dlPCBs increases the total level of dioxin toxicity in many cases to more than twice that, and would cause more fish to exceed the levels considered acceptable.

Due to high dioxin levels in Baltic fish, Swedish and Finnish national health and food authorities have given recommendations on fish consumption. As fish is healthy food, an overall reduction in consumption was not recommended, but consumption of different types of fish – from lakes, seas outside the Baltic as well as from the Baltic. It is recommended that young Baltic fish is preferred. The authorities also advise children, young adults and pregnant women to consume only moderate amounts of fatty Baltic fish (e.g., SNFA 2002, 2005, NFAF 2004, cf. Darnerud et al. 2003, 7.4.1).

As the Community dioxin strategy was adopted, Finland and Sweden opposed the EU restrictions on fishing, stressing the socio-economic and cultural importance of fishing and of certain fish products such as herring in national diets, and the health benefits from consumption of fatty sea fish. Sweden and Finland were thus allowed to delay full implementation of EU limits on dioxins in fish. Germany and Denmark accepted the limit values as such; at least in Denmark they were criticized for being too stringent (CPON 13.8.1999). Also in Sweden and Finland the attitudes towards the derogation varied (e.g., DN 28.11.2001). The dioxin problem as such is shared by many countries, but due to the derogations it has a different political status in Finland and Sweden.

On the basis of the *derogations granted* to *Finland and Sweden* in 2002, domestic sale and consumption of dioxin-laden Baltic Sea fish will be allowed until at least 2006. Fish exceeding the EU concentration limit can be sold in Finland and Sweden and outside EU but not to other EU countries. Both countries are required to warn consumers of health risks from dioxins to report on dioxin levels and to abide by rules on dioxin contents in feeding-stuffs (cf. 6.3). The European Commission has proposed that continuous derogations be granted to ensure marketing of Baltic fish on certain terms; this is being discussed at WTO at this writing. The inclusion of four Baltic

Sea bordering countries to EU has increased the amount of countries subject to regulation in connection with EU dioxins. The new member states have been granted varying possibilities for marketing dioxin-contaminated fish.

1.2.4 The need for additional risk assessment and risk management analysis

Previous dioxin assessments have not considered the particular conditions in the Baltic Sea. There has been some assessment of dioxins in the Baltic e.g. within the EMEP (Vulykh and Shatalov 2001) and in the pilot project on dioxins within SCALE (TWGIM 2004a,b), but these have focused on environmental fate and monitoring of human exposures, respectively.

The standard risk assessment methodology for existing substances in the EU (EC 2003a) increasingly considers risks in marine environments, life cycles of substances, and food chain poisoning. However, the procedure is not suited as such to dioxin risk assessment particularly in the case of Baltic fish. The reliance on many simplifying assumptions, limited integration of human and ecological health, and lacking consideration of mixtures, cumulative and internal exposures, higher-level effects, uncertainties and management links are among its restrictions. It is thus appropriately used as additional general guidance in some areas only.

The relevant authoritative assessments collectively cover a rather broad field. However, the focus in most assessments has been on human health (e.g., SCF 2000, 2001, USEPA 2000a). Often dlPCBs have not been treated. With the exception of the analysis of strategies to reduce exposure to dioxins and DLCs in the US food supply by IOM (2003), *none of the assessments has explicitly addressed the links between assessment and management*. Also the analysis of benefits associated with dioxin-laden foodstuffs has been scarce in earlier assessments (SACN and COT 2004, SPCFC 2005).

Risks associated with dioxins in Baltic Sea and its fish thus need to be analyzed more extensively and thoroughly. Instead of routine assessment for single substances, mixtures of DLCs need to be accounted for. A better integration and differentiation of human and ecotoxicological risk assessment is needed. Additional dimensions of risks need to be considered, including qualitative aspects, variations and related uncertainties, and benefits associated with risks.

The Baltic Sea, its natural resources and associated risks are managed also in other connections than dioxins and PCBs or chemicals generally. For instance, marine protection is addressed both under HELCOM and the evolving marine policies of EU (cf. 6.3.4). For fisheries there is an established, partly specialized and advanced management and advisory system under IBSFC (e.g., ICES 2005a,b,c), increasingly coupled with EU Common Fisheries Policy but not with dioxin strategies.

Risks need above all to be assessed more comprehensively along the chain of events from risk causes to management, and putting risks in a broader context.

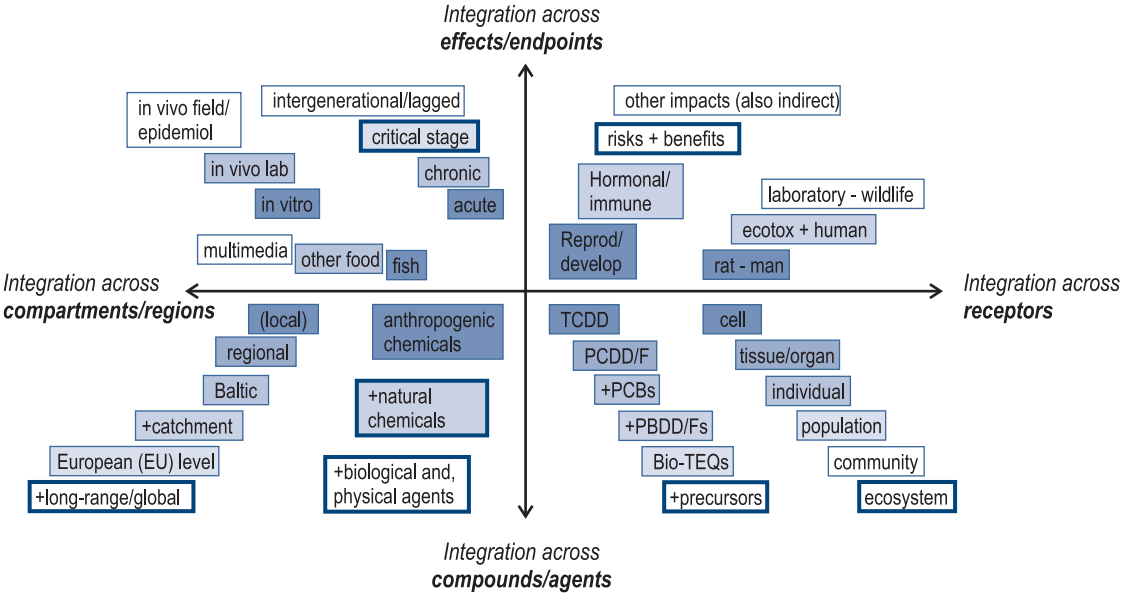


Fig. 1. Levels of integration in assessing risks of dioxins along various dimensions. Those assessment areas routinely included are shown by dark, those more rarely included by light shading. Emerging key areas are shown by bold lining.

This extension of the scope of analysis implies consideration of information in many sectors, disciplines and topic areas (Fig. 1). It also involves questioning of previous assessments along with use of their significant results and insights. The general idea is to proceed beyond them to identify new issues and probe new approaches to risks.

1.2.5 Defining policy-relevant risk analysis questions for dioxin-like compounds in the Baltic

Relevant risk questions are often defined according to the paradigm of risk assessment proceeding from risk identification to management. There are feedbacks, also in the 'structure of inference' (cf. Skolimowski 1966). The order and direction of the process may be altered, reflecting the need to approach risks from several angles. There is thus no one objectively best way. A reverse-mode inference may be useful to define questions and knowledge needs that are relevant to decisions; it may also help resolve interactions between scientific knowledge and policy or decision-making. For the development of quantitative risk management criteria especially for human consumption of fish, a reverse 'upstream' mode of questions and contributing factors may be discerned.

More generally, a set of questions can be formulated to frame the issues in assessing and managing risks from dioxins in the Baltic and its fish. These questions are based on the key entities and factors in risk formation (see next chapters). Any definition of relevant risk questions depends e.g. on the preferences of those asking and on the context of asking and of answering (assessment). The following questions form a logical sequence, but the order is not strict:

- What are the key **DLCs** causing risks and their main sources in the Baltic Sea system
- What are the present and projected **emissions** and immissions of dioxins to the Baltic
- What are the key processes of dioxin **cycling** in the Baltic regarding risks from fish
- What are the **intakes** of dioxins from the Baltic and what doses do they cause
- What **effects** do these dose cause, and what are the most critical effects
- What key **factors** influence the risks and what are the relevant traits of the Baltic Sea
- How do the risks **vary** and develop
- What **benefits** are associated with Baltic fish and what are their relations with risks
- What **policy** principles, goals and initiatives have been invoked
- What contextual and procedural factors affect risk management
- How are dioxins in Baltic fish and associated risks **perceived**
- What is being **presently done** about the risks
- What more can be **done potentially**
- What are conceivable general **strategies** in responding to the risks
- What are the **consequences** of various alternative actions, including socio-economic.

These questions may be summarized in the following general question: *How great and certain are the risks associated with dioxins in Baltic Sea fish, especially for human health but also in a more general perspective, and what are the possible responses to these risks, including alternatives, modifications and complements to present regulatory actions.* Specifically, the question is posed of the basis for allowable fish PCDD/F and dlPCB levels.

1.3 Objectives and methodological approaches of the present work

1.3.1 Goals, objectives and scope

This report has been produced as one deliverable in a project funded by the Nordic Council of Ministers on risk assessment and management strategies of dioxins in Baltic Sea fish. The key general goal of the project is to improve the knowledge base for development and application of relevant strategies. The project thus has the following main operational **goals**:

- to analyze in a many-sided manner the risks to the environment and human health that are associated with dioxins in Baltic Sea fish
- to illuminate the conditions, qualities and impacts of risk management strategies both generally and specifically e.g. in connection with guideline values for dioxin levels in fish.

The **scope** and focus of the project have been defined as follows:

- *Substances:* Focus on toxicologically aggregatable PCDD/Fs and dlPCBs. Directly comparable brominated analogues and other DLCs and stressors have been dealt with to a limited extent or on a more general level for comprehensive risk identification.
- *Environments:* Focus on the Baltic Marine Area. In comparative analysis, some treatment of other seas and fishing areas is included. Inland waters, atmosphere and land areas are included only in a rather generic way
- *Organisms:* Focus on fish and their consumers including man. Other organisms have been treated as appropriate for Baltic Sea fish dioxin assessment and for risk management analysis. In assessing effects, background information is used on other animals as well.
- *Effects:* Focus on AhR aryl hydrocarbon receptor mediated responses. Non-specific disorders have been treated as appropriate. The effects include adverse and beneficial health effects of dioxin-containing fish, and ecological and societal impacts also of management alternatives
- Stages and *parts of risk management:* All stages from formation and release to control and follow-up of dioxins have been included. The focus is on cycling in the sea and subsequent exposures, on risk and impact evaluation and on strategic level aspects of management.

The specific objectives of the analyses reported in the **present publication** are to:

- Provide a comprehensive but focused decision-oriented assessment of risks associated with dioxins in Baltic Sea fish
- Review, summarize and evaluate crucial information, including previous assessments
- Identify methodological and policy issues and approaches to their elucidation.

1.3.2 Approaches and conduct

General

The present assessments are essentially in the form of literature studies and evaluations, primarily based on peer-reviewed scientific information. Multiple schemes were used for appraisal of relevance and quality of information. The work involved document analyses, meta-analyses of existing data and theoretical studies. These were complemented by interviews and observations of views of experts.

Existing frameworks and procedures for risk assessment (cf. 1.4.4) have been utilized as appropriate. They have been modified and extended for better relevance for fish dioxins and for the present task. The analysis has been made predominantly on congener specific data, using WHO-TEq and priority congener approaches for aggregation and focusing of the work.

Previous assessments have been evaluated and utilized. Discussion and appraisal of these documents has been made from a decision and management point of view. Emphasis has been put on philosophical aspects of dioxin assessment and management and on comparative critical analysis of risk and strategy characteristics.

In addition to natural processes and factors, social, technological and other human processes and phenomena have been addressed. Links between these domains have been scrutinized, and the natural scientific analysis is focused on issues seen as crucial for decision-making.

The wide area of the assessment cannot be covered in great detail in the present report. Instead, identification and discussion of key issues in risk evaluation and management is attempted. General issues are also illustrated by examples.

The work was made in 2002-04 with funding from the Nordic Council of Ministers, under the auspices of the Nordic Chemicals Group in collaboration with some other groups and bodies (cf. Acknowledgements).

Data and document evaluation

Scientific literature was complied for evaluation initially by searches in SCIRUS and PubMed databases using several keyword combinations and scoping. They were extended to cover older publications and publications outside the Baltic Sea, depending on their relevance and quality. In some areas, searches were focused on literature published after previous extensive assessments (e.g. USEPA 2000a). Secondary sources were searched among references and more recent publications by authors. The results of these searches were complemented by published sources drawn from earlier projects.

Original scientific sources were emphasized but reviews were also extensively used. Among earlier risk assessments, both official regulatory assessments (cf. 1.1) and reviews in scientific publications were considered. Web sources were additionally searched especially for regulatory, technical, monitoring and risk communication information. The searches especially for data on DLCs in the Baltic were focused on recent, reliable and representative congener-specific analyses published in peer-reviewed papers. Also relevant extended abstracts from Dioxin symposia and aggregating information were utilized. Some unpublished data were obtained from in-house sources mainly.

Expert meetings and networks

Links were established with Dioxin symposia. An ad-hoc meeting on the Baltic Sea was arranged in the Barcelona symposium on 15.8.2002. In Boston on 23.8.2003 a session was arranged in collaboration with other Nordic (food) expert bodies and the EC. Both meeting sessions were reported and utilized in the analysis.

A workshop was held in 12.-13.6.2003 Helsinki on Dioxins in Baltic Sea fish (Annex 13).

A network of dioxin experts from several countries, disciplines and branches of government were set up in connection with the project.

1.3.3 Structure of the report

The report is divided in main blocks on risk assessment and management, flanked by a general introduction and a synthesizing part, and complemented by executive summary, reference and annex sections; annexes will be published only in electronic form. Links between assessment and management have been emphasized and summarizing elements have been included throughout.

The assessment proceeds from risk identification over exposure and effects assessment to risk characterization; a complementary management decision based structure has been applied to some extent. Along with risks, associated impacts and uncertainties are dealt with. The sections on risk and uncertainty identification, effects assessment and risk characterization are relatively detailed. Risk management analysis has been divided in policy contexts, measures and strategies.

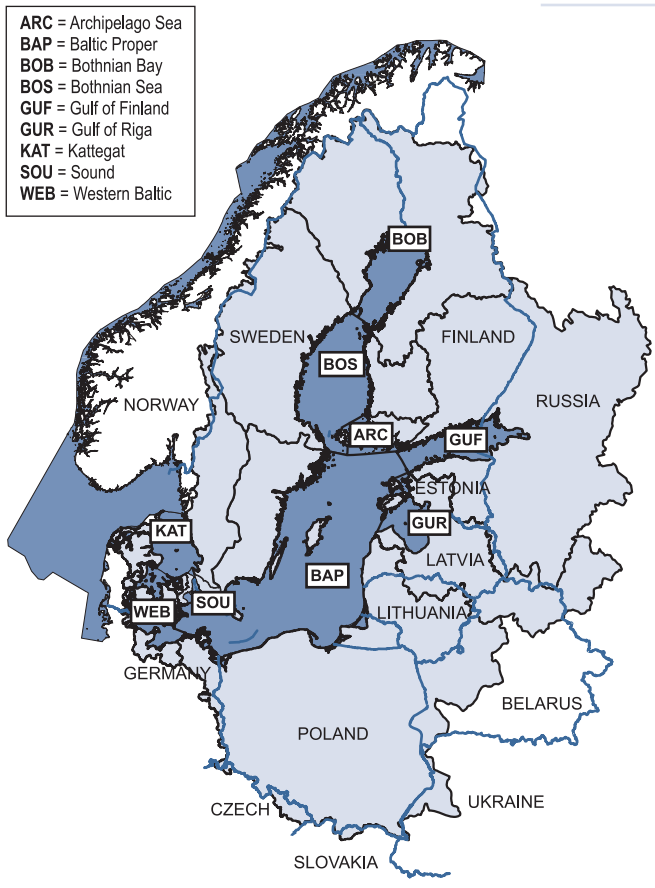


Fig. 2. Baltic Sea and its sub-areas, their catchment areas and national borders from Pekka Kotilainen/SYKE.

1.4 Definitions and related methodological frameworks

1.4.1 Baltic Sea and related other areas

The Baltic Sea (the Baltic in short) is the second largest brackish-water basin after Black Sea. The area of the Baltic Marine Area extends to the Eastern part of the North Sea, including Norwegian, Danish and Swedish sea areas (Fig. 2). It is divided in Gulf of Bothnia (GOB), consisting of Bothnian Bay (BOB) and Bothnian Sea (BOS); Gulf of Finland (GOF); Gulf of Riga (GOR); Baltic Proper (BAP), containing Bornholm and Arkona Basin; West Baltic (WB) or Belt Sea; the Sound; Kattegat (KAT).

The catchment of the Baltic marine area is 4 times the size of the sea itself and includes land in Sweden, Finland, Russian Federation, Poland, Lithuania, Latvia, Estonia, Denmark, Germany, Belarus, Norway, Czech Republic, Slovakia and Ukraine. The four first mentioned comprise 80 % of the catchment, the last four only a very

minor part (Fig. 2). The last five have not ratified the Helsinki Convention. Of these 14 countries, Sweden, Finland, Denmark and Germany were EU member states until 1.5.2004 when also Poland, Latvia, Lithuania, Estonia, Czech Republic and Slovakia joined. The catchment watershed does not present a strict border; the Baltic interacts with a larger area (cf. 2.3.1).

1.4.2 Dioxins and dioxin-like compounds

Dioxins are an inexact name. It is often used of **polychlorinated dibenzo-p-dioxins and dibenzofurans** (PCDD/PCDFs or PCDD/Fs). They are diaromates (but tricyclic), with two benzene rings united by 1 or 2 oxygen bridges (in PCDDs and PCDFs, respectively) and with chlorine substituents. PCDD/Fs comprise 210 different molecules, so-called congeners. The other halogenated dioxins (PXDD/Fs) include thousands of congeners.

Only a fraction of PCDD/Fs is included in the dioxins dealt with in regulatory contexts. They usually mean only those with Cl in lateral 2-, 3-, 7- and 8-positions. They are the most toxic ones as approximated by toxic equivalency factors (TEFs) used to quantify toxicity equivalents (TEqs) relative to 2,3,7,8-TCDD (TCDD) for key vertebrate groups (Table 1). These PCDD/Fs comprise 17 congeners, including TCDD that is best studied. In most cases dioxins occur as mixtures where other congeners are present and more important than TCDD.

Many other chemicals have also been included **in dioxin-like compounds** (DLCs). Some of such DLCs are also commonly called dioxins, and are called so in the present text if confusion does not result. This is most natural in the case of dibenzo-p-dioxins and dibenzofurans with other halogen atoms and other substituents instead of chlorine.

Other DLCs notably include dioxin-like polychlorinated biphenyls (dlPCBs) that have been assigned TEF values (Table 1). Nordic N-TEFs (1988), international I-TEFs (NATO/CCMS 1988a) and WHO-TEFs (1994, 1998) have also been defined, the latter to different animal groups. In the present work, WHO_{98}-TEqs for mammals are used if not otherwise indicated. The WHO-TEqs due to PCBs are termed TEq_P as distinguished from WHO-TEq_{DF} due to PCDD/Fs and from WHO-TEq_{DFP} including both groups (cf. USEPA 2000a). The dlPCBs belong to non-*ortho* (0-*ortho*) and mono-*ortho* (1-*ortho*) congeners. The former are commonly termed coplanar PCBs (coPCBs) due to their structure. The demarcation between dioxin-like and non-dioxin-like PCBs is not clear. The toxicity of DLCs is aggregated by 'Bio-TEqs' based on bioassays.

Also many other substances possess dioxin-like properties and have been suggested to be included in TEF schemes (cf. 2.2.1). While generally halogenated diaromates, DLCs include non-halogenated compounds and other ring structures. Some have been produced industrially, some accidentally. Many natural 'DLCs' in terms of AhR binding and bioactivity have been identified (e.g., Denison et al. 2002, Annex 1), and may play a role for overall dioxin risks. These substances differ in their properties including their ability to activate the AhR and to elicit dioxin-type effects (see e.g. Chen and Bunce 2004).

In summary, not all PCDD/Fs are 'dioxins' in the sense normally used; on the other hand, substances with dioxin-like properties and toxicity are found also in many other classes of compounds. It may thus be illogical and inefficient to limit assessments e.g. only to PCDD/Fs (or to some PCBs); this depends on the context and goals. ***In the present work, dioxins are taken to include 2378-PCDD/Fs and the dlPCBs given TEFs by WHO;*** other DLCs are dealt with cursorily in some connections.

1.4.3 Risks and uncertainties

Risk has been defined in toxicology e.g. as "the probability of an adverse effect in an organism, system or (sub)population caused under specified circumstances by exposure to an agent." (IPCS and OECD 2003, cf. Christensen et al. 2003). Risk is also defined formally as a function of the probability and consequence of adverse event (or process). *Hazard* is partly synonymous with risk but has also other interpretations (cf. risk and hazard assessment). *Impact* is a related term that does not carry strong connotations of probability or potentiality.

Risks have many forms and dimensions. They cannot be captured in a number, or even a probability distribution (pdf) or other quantitative representation (HCN 1996b). The distinction but also the interrelationship of objective and subjective risks is stressed. Risks are further divided in individual and population risks, and in attributable and background risks. Relative risk is used especially in epidemiology, often being rather the odds or risk ratio (Neubert 1997-98). Formal expressions include e.g. R=f(dose, effect) in toxicology, f(p(opportunity loss)) in economics, f(p(load>resistance)) in technology, and f(p(stress>carrying capacity)) in ecology.

Table 1. Nomenclature and current toxic equivalency factors (TEFs) of polychlorinated dibenzo-p-dioxins, dibenzofurans and biphenyls in the WHO TEF schemes (Van den Berg et al. 1998). Note that TEFs are (half) order-of-magnitude estimates of relative potency and depend on value-based judgments and conventions.

Congener name, IUPAC no (for PCBs)	Other abbreviations and trivial names	TEF- WHO_{98} ($TEF\text{-}WHO_{94}$)		
		mammal	bird	fish
2,3,7,8-Polychlorodibenzo-p-dioxins	2378-PCDDs, PCDDs			
2,3,7,8-Tetrachlorodibenzo-p-dioxin	2378-TCDD, 2378-C_4DD (TCDD, 'dioxin')	1	1	1
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	12378-PeCDD, 1-PeCDD (PeCDD, C_5DD)	1 (0.5)	1	1
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	123478-HxCDD, 123478-C_6DD, 4-HxCDD	0.1	0.5	0.1
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	123678-HxCDD, 123678-C_6DD, 6-HxCDD	0.1	0.01	0.1
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	123789-HxCDD, 123789-C_6DD, 9-HxCDD	0.1	0.1	0.1
1,2,3,6,7,8,9-Heptachlorodibenzo-p-dioxin	1236789-HpCDD (HpCDD, C_7DD)	0.01	<	0.001
Octachlorodibenzo-p-dioxin	OCDD, C_8DD	0.0001 (0.001)	-	-
2,3,7,8-Polychlorodibenzofurans	2378-PCDFs, PCDFs			
2,3,7,8-Tetrachlorodibenzofuran	2378-TCDF, 2378-C_4DF (TCDF)	0.1	1	0.05
1,2,3,7,8-Pentachlorodibenzofuran	12378-PeCDF, 12378-C_5DF, 1-PeCDF	0.05	0.1	0.05
2,3,4,7,8-Pentachlorodibenzofuran	23478-PeCDF, 23478-C_5DD, 4-PeCDF	0.5	1	0.5
1,2,3,4,7,8-Hexachlorodibenzofuran	123478-HxCDF, 123478-C_6DF, 4-HxCDF	0.1	0.1	0.1
1,2,3,6,7,8-Hexachlorodibenzofuran	123678-HxCDF, 12368-C_6DF, 6-HxCDF	0.1	0.1	0.1
1,2,3,7,8,9-Hexachlorodibenzofuran	123789-HxCDF, 123789C_6DF, 9-HxCDF	0.1	0.1	0.1
2,3,4,6,7,8-Hexachlorodibenzofuran	234678-HxCDF, 234678-C_6DF, 4,6-HxCDF	0.1	0.1	0.1
1,2,3,4,6,7,8-Heptachlorodibenzofuran	1234678-HpCDF, 1234678-C_7DF, 6-HxCDF	0.01	0.01	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran	1234789-HpCDF, 1234789-C_7DF, 9-HxCDF	0.01	0.01	0.01
Octachlorodibenzofuran	OCDF, C_8DF	0.0001 (0.001)	0.0001	0.0001
Non-*ortho* polychlorobiphenyls	non-ortho PCBs (0-o PCBs)			
3,4,4',5-Tetrachlorobiphenyl, PCB 81	344'5-TeCB, CB 81	0.0001	0.1	0.0005
3,3',4,4'-Tetrachlorobiphenyl, PCB 77	33'44'-TeCB, CB 77	0.0001 (0.0005)	0.05	0.0001
3,3',4,4',5-Pentachlorobiphenyl, PCB 126	33'44'5-PeCB, CB 126	0.1	0.1	0.005
3,3',4,4',5,5'-Hexachlorobiphenyl, PCB 169	33'44'55'-HxCB, CB 169	0.01	0.001	0.0005
Mono-*ortho* PCBs	mono-ortho PCBs (1-o PCBs)			
2,3,3',4,4'-Pentachlorobiphenyl, PCB 105	233'44'-PeCB, CB 105	0.0001	0.0001	-
2,3,4,4',5-Pentachlorobiphenyl, PCB 114	2344'5-PeCB, CB 114	0.0005	0.0001	-
2,3',4,4',5-Pentachlorobiphenyl, PCB 118	23'44'5-PeCB, CB 118	0.0001	0.00001	-
2',3,4,4',5-Pentachlorobiphenyl, PCB 123	2'344'5'-PeCB, CB 123	0.0001	0.00001	-
2,3,3',4,4',5-Hexachlorobiphenyl, PCB 156	233'44'5-HxCB, CB 156	0.0005	0.0001	-
2,3,3',4,4',5'-Hexachlorobiphenyl, PCB 157	233'44'5'-HxCB, CB 157	0.0005	0.0001	-
2,3',4,4',5,5'-Hexachlorobiphenyl, PCB 167	23'44'55'-HxCB, CB 167	0.00001	0.00001	-
2,3,3',4,4',5,5'-Heptachlorobiphenyl PCB 189	233'44'55'-HpCB, CB 189	0.0001	0.00001	-
Di-*ortho* PCBs	di-ortho PCBs (2-o PCBs)			
2,2',3,3',4,4',5-Heptachlorobiphenyl PCB 170	22'33'44'5-HpCB, CB 170	(0.0001)		
2,2',3,4,4',5,5'-Heptachlorobiphenyl PCB 180	22'344'55'-HpCB, CB 180	(0.00001)		

Uncertainty is present in risks, and universally in ontological entities and factual statements. Uncertainty has been defined e.g. as "imperfect knowledge concerning the present or future state of … a system" (IPCS and OECD 2003), but can also pertain to past states. Distinctions can be made between uncertainties that are or are not identified yet, and between risk, uncertainty, indeterminacy and ignorance as classes of imperfect knowledge (Wynne 2002; cf. Hildén 1997b). Also the concept of 'incertitude' has been used (e.g., Harremoës 2003). The following types of uncertainty are often distinguished (e.g., Finkel 1993):

- *Measurement* uncertainty is a commonly recognized type and is reflected by the fact that a risk has not exactly one value, but varies in a range (and probability distribution) around it

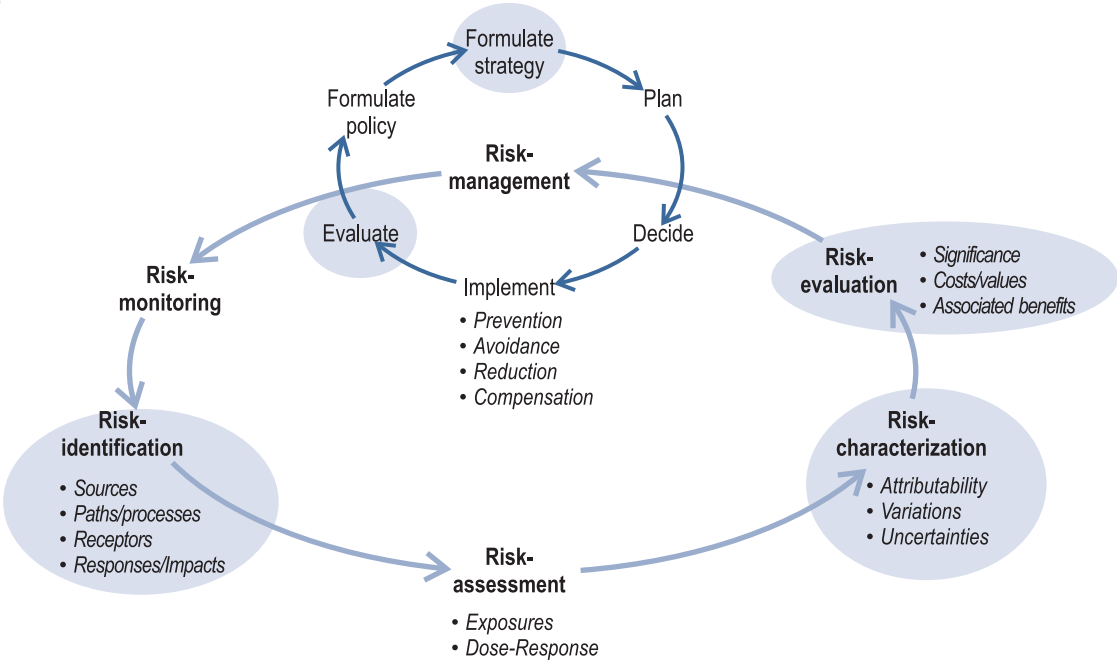


Fig. 3. Schematic presentation of the risk assessment-risk management cycle, indicating key areas and parts of the present analyses (modified and extended from Assmuth and Hildén 2002).

- *Model* uncertainty often has a decisive influence on overall uncertainty but is often underestimated. Model uncertainty may also be of a qualitative nature
- *Decision rule* or epistemological uncertainty; e.g., in framing and reasoning. This is to some part interpretable as model uncertainty and may play a key role in decisions.

1.4.4 Risk assessment and risk management

Risk assessment has acquired interpretations in many areas. According to IPCS and OECD (2003), it is "a process … to calculate or estimate the risk to a … target organism, system or (sub)population … following exposure to a … agent … The … process includes … hazard identification, hazard characterization …, exposure assessment, and risk characterization." Conceptually rather similar definitions and divisions have been used e.g. in EU chemicals control (EC 2003a) and in connection with the precautionary principle. Many of these are based on the environmental risk assessment paradigm by NRC (1983). In the food safety area, work is done on 'Good Risk Assessment Practices' (FAO and WHO 2002b).

Many definitions and approaches are oriented toward toxic chemicals; however, also other kinds of risks are included in assessments. Also in the food safety area, formal risk assessment methods are developing e.g. to address quantitative aspects and uncertainty (Notermans and Mead 1996, Boenke 2001, Dybing et al. 2002).

Risk assessments vary in their contents, information needs and overall methodology. For instance, assessment of sudden risks differs from that of risks from continuing contamination. Assessments can address risks in many dimensions. Many methods can be used, from qualitative to quantitative, deterministic or stochastic. *Strikingly different things are meant by risk assessment* even in toxicology (see e.g. the critique of regulatory risk assessments of dioxins by Neubert 1997-98). Different 'assessment cultures' are thus discernible (see also Horlick-Jones 1998 on 'soft' and 'hard' risk analysis cultures). The harmonization of risk assessment, although needed and possible to a point, is partly an attempt to combat inherently non-harmonizable differences and may conceal crucial distinctions.

Risk analysis can been distinguished as a more general concept. In the terminology of IPCS and OECD (2003), risk assessment is the first part of a risk analysis process that also includes risk management and communication. This is however illogical, as 'analysis' denotes a process of inspection and reflection, not of action. Nevertheless, risk analysis, as established e.g. in many areas of health and safety, does

usually include a decision component that is more extensive and developed than is presently usual within assessment of chemicals.

Integrated risk assessment or integrated assessment has developed in relation with the emphasis on multi-disciplinarity and sector integration. The dimensions and degrees of integration in assessment vary. Integrated risk assessment was defined by USEPA (2002a) as "a process that combines risks from multiple sources, stressors, and routes of exposure for humans, biota and ecological resources in one assessment with a defined point of focus". Also IPCS (2001d) focuses on integrating risks to human and non-human receptors. The concept is broader in the EU environment and health strategy, SCALE (see EC 2004a), including integration of a) information; b) research; c) environmental and health concerns with other policies; d) cycles of pollutants; e) interventions; f) stakeholders.

A related concept is cumulative risk assessment, developed especially by the USEPA (1997b, 2002a) and focusing on mixtures of chemicals, long-term risks and different endpoints, but including other natural aspects of assessment. Still more generally, integrated assessment, as developed especially in connection with global change and precautionary principle, routinely includes socio-economic and management aspects.

The borderline between risk assessment and **risk management** is blurred. They have been separated (e.g., NRC 1983, ECETOC 1993) but it has been increasingly realized that they are interacting and partly fused parts of a complex process with feedbacks between definition of questions, framing of issues, knowledge production and transmittal, and knowledge use in problem-solving. Simultaneously, more nuanced concepts of assessment and management have emerged to avoid their lumping.

Risk management is sometimes used as a narrow term only for technical operations. In the present work, risk management is defined broadly, including non-technical aspects. According to IPCS and OECD (2003) it is a "decision-making process involving … political, social, economic, and technical factors with relevant risk … information … to develop, analyze, and compare … options and to select and implement … regulatory response. ... (it) comprises risk evaluation; emission and exposure control; risk monitoring." (cf. FAO/WHO 1997b, USPCCRARM 1997).

Risk management is commonly taken to include risk prevention, risk avoidance, risk reduction, and risk compensation. In particular, it extends to social and human aspects (HCN 1996b). These overlap with governance that is in some respects more general, encompassing e.g. institutions; on the other hand, it may mostly denote management in the public sector (by governments) and to a lesser extent e.g. in enterprises.

Decisions include policy or management decisions but also decisions in scientific and statistical inference (cf. Dudewicz and Mishra 1988, 629-41) and in other spheres of activity.

Part A: Risk assessment

"So many circumstances of a small and accidental nature are relevant that no broad and simple uniformities are possible"

– Bertrand Russell: Human knowledge: Its scope and limits (1948)

2.1 Conceptualization of risk chains and contexts for decision-relevant risk identification

2.1.1 General considerations

Risks from Baltic Sea fish dioxins are formed in **chains of events** that are divided roughly in sources, transport and receptors; they are also commonly divided in exposures and effects. Additionally, risk management stages may be included. A division between the sea and its surroundings may further be made in framing the system and event chains in risk formation. These in relation to Baltic fish dioxins (Fig. 4) do not constitute a single chain but include interacting elements, co-factors and feedbacks. In this simplified conceptualization, e.g. the levels of management (cf. 6, 8) and the links in the process have not been detailed.

With Baltic Sea fish dioxins, the chain of events more specifically involves 1) formation and releases; 2) transport and fate including accumulation and transformation; 3) effects; and 4) subsequent other impacts also in risk management. The sequence is complicated by the fact that all of these include parts and processes that take place in the sea and outside the sea.

Risk identification according to the present common procedures within environment and health largely revolve around toxic effects. However, risk and hazard identification in a more general sense encompasses other aspects of risks that become crucial also for dioxins especially in a management context. In particular, risks are caused also by management actions e.g. by loss of benefits, although this is often overlooked. Risk identification in this regard is related to the fact that the system entities are linked to a larger context, especially when attempting to make sense of risks in an integrated fashion.

The probabilities of consequences like toxic or beneficial effects take on importance in risk formation and analysis. In the case of dioxins, instead of discrete events a more continuous process takes place. Also the other stages of risk chains involve probabilities and consequences of the respective events and factors. The probability aspect of risks has many levels. In addition to single point estimates of probabilities, their variation and uncertainty need to be considered.

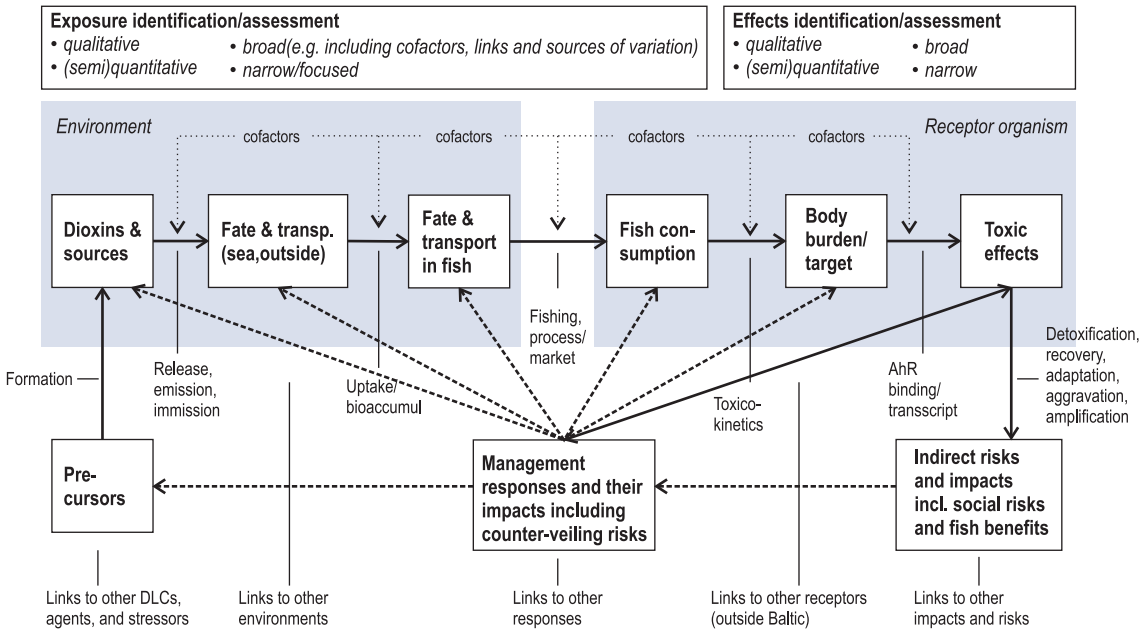


Fig. 4. Conceptualization of event chains and their links in formation of risks associated with dioxin-like compounds in Baltic Sea fish, as a basis for multi-level identification and assessment approaches to these risks.

For instance, carcinogenic risks are commonly expressed as the probability of developing cancer (using e.g. a 10^{-6} lifetime excess risk as a benchmark distinguishing from background risk, cf. Fishbein 1990), but are in reality represented by a range of probabilities, or more fully by a pdf of this single probability. This is approached e.g. by identifying the factors contributing to such variation in risks.

Uncertainties in risks arise in part due to incomplete measurements. However, uncertainty is also caused by imperfect knowledge of the phenomena themselves. For instance, assumptions of the form of dose-response functions may influence dioxin risk estimates more than the spread of dose or response data. Uncertainty in decision rules include technical-level and more fundamental rules, the latter being related e.g. to questions of acceptability of risks and of management goals (see Finkel 1993).

2.1.2 Hazard and risk identification approaches to dioxin-like compounds in the Baltic and its fish

There are many possible approaches to identifying and assessing risks, hazards, impacts and uncertainties associated with dioxins, also with dioxins in Baltic Sea fish. There is no one single universally best approach. Instead, combined and iterative approaches are useful, reflecting the multi-dimensionality of risks, of assessment and of their contexts.

Risk identification is related to the framing of risks and uncertainties and of the questions asked. The balance between breadth and focus varies in the process, especially so that initial broader risk identification is followed by more focused identification and assessment. However, both an 'inbound' focus and a generalizing or comparative 'outbound' focus are needed. Broader assessment is again needed after initial focusing, e.g. in comparative evaluation of results. It may be useful also to start from a focused question (e.g. in risk management) and derive the set of assessment questions from that, in a reverse or 'upstream' assessment approach (cf. 8).

Identification of dioxin risks in connection with Baltic Sea fish may be made within a) compounds or agents; b) matrices or carriers; c) receptors; and d) effects or impacts (Fig. 5). The dimensions of risk that are given particular attention in the present risk identification include:

- What are the dioxin-like compounds considered
- What are the (dietary) intake media
- What are the receptor organisms considered
- What are the effects and consequences considered, and what are their co-factors
- What risk management needs and opportunities exist.

These dimensions span the chain of events from DLCs over receptors to consequences. They are sufficiently separate, although related especially in the case of receptors and effects. The management opportunity dimension may be included in risk characterization or management strategy analysis, and dealt with only to a limited extent in the risk identification stage. The relative speed at which the focusing takes place may vary across the dimensions (Fig. 5, Fig. 3). In accordance with this flexible and multi-dimensional approach to risk identification, no strict formalism is attempted. Instead, the tiers and levels of identification and assessment are presented as an indicative structuring framework only.

Particularly important additional framing questions, included inherently in the above dimensions, include the following:

- How and at what stage the identification of risks is focused on the **Baltic Sea**, i.e. what is the scope in terms of environmental compartments.
- How and at what stage the identification of risks is focused on **risk management**.

The risks associated with dioxins specifically in Baltic fish require assessment approaches that are tailored to elucidate issues in this context, while retaining suitable generality. This means that the particular factors and processes, ecological, social and technological, which are operative in relation to the Baltic Sea and to fish and fishing (not only for human consumption), need to be considered. Therefore, risk and uncertainty assessment approaches e.g. in marine protection, fisheries and food sectors need to be accounted for and applied in dioxin risk assessment and management analysis. The relevant approaches will be described and discussed more specifically in the appropriate sections, in connection with e.g. exposure and effects assessment and management policy and strategy analysis.

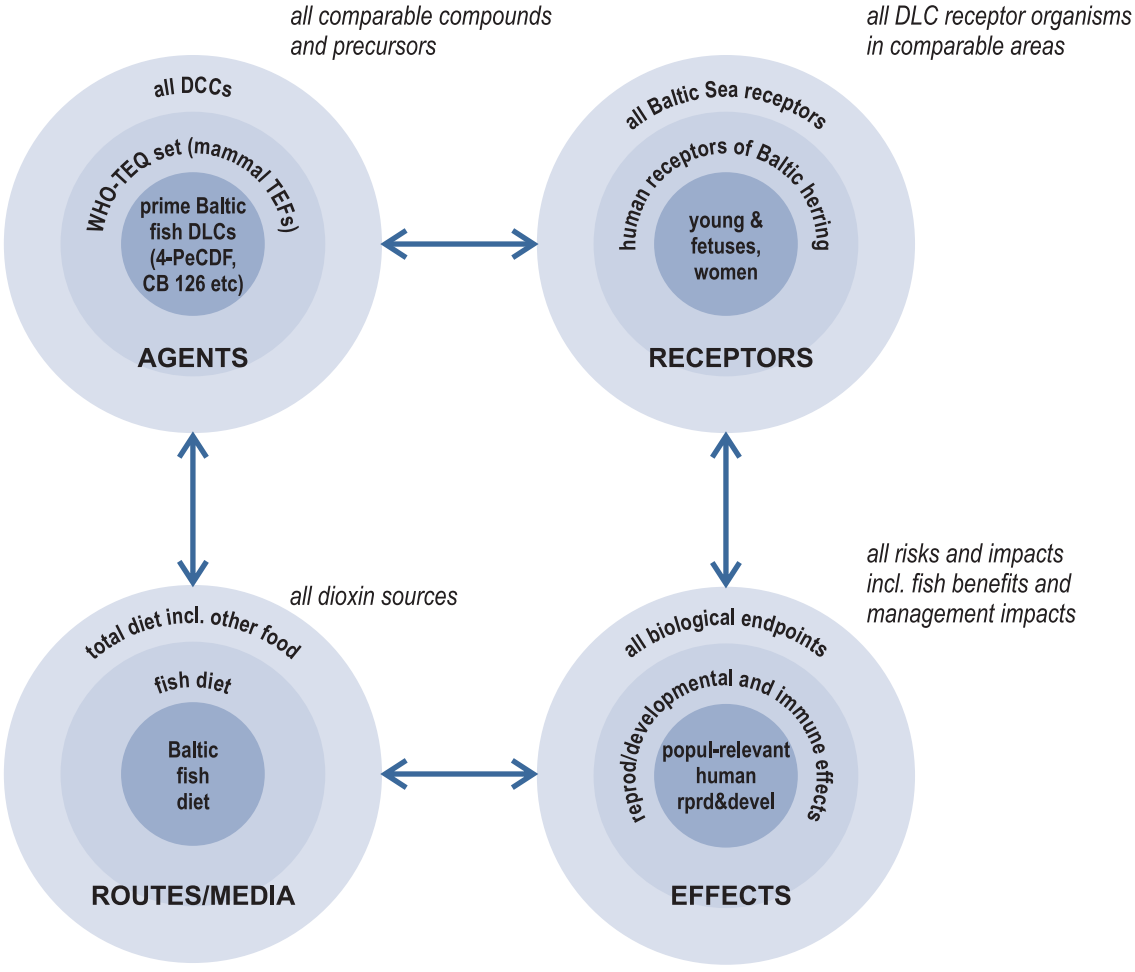


Fig. 5. Focusing assessment within key dimensions of risks associated with dioxin-like compounds in Baltic Sea fish. Note the influences between the dimensions in defining focus.

2.2 Dioxin-like compounds, their precursors and reaction products

2.2.1 Identification of dioxin-like compounds

The criteria for identifying a compound as a DLC vary in time and depend on the context. Dioxins have been most commonly taken to include 2378-PCDD/Fs and certain PCBs, usually three 0-*ortho* and nine 1-*ortho* congeners (cf. Table 1). The criteria internationally agreed (Van den Berg et al. 1998) are structural similarity; binding to AhR; AhR-mediated dioxin-like toxicity; persistence and bioaccumulation. None of these are clear-cut, and such criteria may differ between regulation and research. For comprehensive risk identification it is important initially to include a broad array of potential DLCs, as these may interact with and influence risks of PCDD/Fs. A multitude of other groups of chemicals thus need to be considered, including both man-made chemicals or their reaction products and natural compounds (Table 2, cf. Annex 1).

DLCs may additionally be divided in two main groups on the basis of their chemical structures: a) *Planar* DLCs having a fixed plane, including e.g. PCDD/Fs, 0-*ortho* PCBs and all PCNs, and corresponding other halogenated (brominated and chlorobrominated) compounds; b) *Non-planar* DLCs that allow some rotation of the aryl ring around the plane of the other, including e.g. 1-*ortho* PCBs, PCDEs and corresponding brominated compounds.

An increasing amount of *other compounds* fulfil some criteria for inclusion in DLCs. Some of them are formed naturally (Connor et al. 2004) and have strong dioxin-type activity (Denison and Nagy 2003). Many PAHs, PCNs and PBDEs elicit dioxin-like activity at least *in vitro* (e.g., Chen et al. 2001, Villeneuve et al. 2001, Behnisch et al. 2003).

Bioassays indicating (with variable coverage and specificity) dioxin-like bioactivity, although not specifying the compounds responsible, have exhibited elevated response levels for Baltic Sea samples including fish extracts (e.g., Håkansson et al. 1991).

All in all, *'dioxins' and DLCs encompass many more compounds than the 2,3,7,8-chlorinated PCDD/Fs and dlPCBs subject to most regulatory and monitoring interest.* Thus, not only the PCDD/Fs included in the EU (2001) recommendations or assigned TEFs are addressed if dioxin-type risks are to be comprehensively managed. In general, more meaningful chemically and biologically based definitions of dioxins are needed. This broadening scope also implies a need for compound specificity, and limitations of TEFs and other aggregative concepts for dealing with DLCs. Moreover, in identifying and targeting risks from dioxins also in the Baltic and its fish, it is important to account for the mutual relations of various DLCs as well as for their precursors and reaction products.

2.2.2 Precursors and formation of dioxin-like compounds

Dioxins are formed in various complex and interacting (also overlapping) processes (Table 3). Many are poorly known, e.g. in terms of

Table 2. Basic-level identification of potential dioxin-like compounds with regard to established criteria for assigning toxic equivalency factors (TEFs, van den Berg et al. 1998). The groups included in this WHO TEF scheme are shown in grey and in boldface, as are strong criteria for dioxin likeness and precursor function (cf. Annex 1). Note the many groups of organobromides fulfilling all dioxin criteria, compounds with oxygen (O) bridge substitutes, and polyaromates and naturally produced compounds having some dioxin-like properties. Cf. text.

Substance group	Molecular structure in relation to PCDD/Fs	AhR binding	AhR mediated TCDD toxicity and bioactivity	Persistence and bioaccumulation	Dioxin precursor
PCDD/Fs (2378-chlorinated)	**++**	**+/++**	**+/++ (some weakly)**	**++**	**(++ inherently)**
PCDD/Fs (other congeners)	+	(+) (some)	? (few studies)	+/++	(+) (some)
non- and mono-*ortho* PCBs	**+ (0-o PCBs ++)**	**+/++**	**+/++ (partial)**	**++**	**+**
Di-*ortho* PCBs	(+) (no planarity)	+	(+) (partial)	++	+
PBDD/Fs and PBCDD/Fs	**++ (Br size differs)**	**++**	**++**	**++**	**(++ inherently)**
PFDD/Fs	++	++ (TFDD)	?	(TFDD non-acc)	(++ inherently)
Alkylated PCDD/Fs	+	?	(+, for some)	+	
PCNs (2367-chlorinated)	+ (no O bridges)	+	(+) AhR role unclear	+	+
PCTs (chloroterphenyls)	+	+	?	+	+
TCAB, TCAOB and TCHBs	++ (N/NO bridges)	++	+ (strong AHH induct)	++	
Non- and mono-*ortho* PBBs	**+ (0-o PBBs ++)**	**+/++**	**+/++ (partial)**	**++**	**+**
Dioxin-like di-*ortho* PBBs	(+) (no planarity)	+ ? (PCB analogues)	(+) (partial)	++	+
PCDEs/PBDEs (diphenyl ethers)	+	+ (many congeners)	(+)	+	+
PCDAs/PBDAs, PCDTs/PBDTs	+(+) (low planarity, S)	+	?		
Nitro-PCDDs	+	+	?	?	
PCBPs (chlorobiphenylenes)	+	?	(+) (strong AHH induct)	?	
PAHs	(+) (some)	+ (some congeners)	+	+/+(+)	
Chlorinated PAHs	(+) (some +)	+ (some congeners)	(+) (AHH inducers)	(some unstable)	
TCSTBs (T-chlorostilbenes)	+	+	?	?	
TCBTs (T-chlorobenzyltoluenes)	+	(+) (predicted)	?	?	
HCBz (hexachlorobenzene)		(+)	- (some effects +)	++	+
$MeSO_2$-CBs, OH-CBs (-BBs)	+ (some Cl/Br reloc)	+	+	++ (some)	(+)
Dialkylamino-dinitrobenzenes		+	?	?	
Indoles, e.g. ICZ and derivatives	-	+ (ICZ ++ in vitro)	(+) (AHH, CYP induct)	-	
(Benzo/naphtho)flavones	-	+, - (some NFs, ++)		-	
Tryptanthrins / tryptophan deriv.	-	+		-	
Indigo, indirubin and derivatives	-	+ (synthetic IRs ++)	(+) (AHH, CYP induct)	-	
Halogenated dimethyl bipyrroles	-	(+)		-	
Prostaglandins	-	+ (some)		-	

Explanations of abbreviations (cf. Table 1, Annex 1): AhR=Aryl hydrocarbon receptor; AHH=aryl hydrocarbon hydroxylase; CYP=cytochrome P450 enzyme sub-family; $MeSO_2$-PCBs=methyl sulphone metabolites of PCBs, OH-CBs=hydroxyl metabolites of PCBs, ICZ=indolo[3,2-b]carbazole; NF=naphthoflavone; IR=indirubin.

mechanisms and pathways, contributing substances, determinant factors, reaction rates and speeds, and quantitative yields (cf. Annex 3).

Formation of PCDD/Fs and corresponding dioxins with other halogens is **inadvertent** and often accidental; they are harmful by-products that have been intentionally synthesized by man only for research and analytical purposes. This applies to many other DLCs, too. However, some DLCs including dioxin-like PCBs and PBBs, PCNs and PCTs have been formed also as (originally) desired main ingredients in synthetic chemical products.

PCDD/Fs are formed in chemical reactions from **precursors** such as chlorophenols, chlorophenoxy herbicides, PCBs and chlorobenzenes, e.g. by condensation (McKay 2002). Also chlorobleaching of pulp, water chlorination and chloranil and dyes based on it form PCDD/Fs (e.g., Rappe 1992). PBDD/Fs are formed from corresponding brominated compounds such as PBBs and brominated flame retardants (Annex 3, Sakai et al. 2005).

Importantly, PXDD/Fs are formed in **thermal reactions** that involve halogens and carbon, both in combustion and pyrolysis (Weber and Sakurai 2001) e.g. of wastes and fuels as well as in industrial processes. Precursors here include halogenated aromatic compounds (cf. above), and Cl (or Br) and organic matter (*de novo* formation).

Formation of PCDD/Fs in waste incineration is well known, but incinerators also destroy PXDD/Fs (e.g., Weber et al. 1999). On the other hand, PCDD/Fs may be formed in waste recycling facilities (Tamade et al. 2002). The rate of formation in incineration depends on the conditions and completeness of burning, on catalysts and on alkalinity.

PCDD is easily formed from PeCP (e.g. Addink and Olie 1995). In the Baltic Sea region, a TeCP-TCP-PeCP mixture is a notable PCDF source (Isosaari 2002b, Bergkvist et al. 2005). Burning wood that also contains CCA salts greatly increases TEQ_{DF} yields (Tame et al. 2003b). Other biocides of importance for cumulative emissions include 2,4,5-T. In addition, many other chemical products and processes contain or form PCDD/Fs (cf. Annex 3).

Dioxins are formed in photochemical reactions by dechlorination of OCDD/F, cyclization of o-phenoxyphenols (predioxins), and dimerization of PCPs (Rappe 1992, cf. above). PCDD/Fs are even formed at low rates by enzymatic reactions (Öberg and Rappe 1992, Hoekstra et al. 1999). Nevertheless, by and large PCDD/Fs are products of anthropogenic processes.

PCDD/Fs are thus formed in a) products containing abundant precursors, b) industrial processes; c) intentional energy conversion such as combustion, d) other processes including subsequent reactions of such precursors, e) some natural e.g. enzyme-catalyzed processes.

For purposes of assessment and management, it is instructive to further divide the processes of formation in the following classes (cf. Annex 3); those generally emphasized here are shown in bold:

- **Stationary** and mobile processes
- Point and **diffuse** processes
- Primary and **secondary** processes and sources
- **Ongoing** or terminated processes
- **Production**, use (primary and down-stream) or **waste** management stage processes
- **Open** (ambient) and closed (e.g. industrial) processes
- **Land-based**, sea-based or atmospheric processes
- Processes occurring in **Baltic Sea countries** and elsewhere.

In many technical processes the formation of PCDD/Fs is controlled by the operating conditions, in addition to precursor presence (e.g., Eduljee and Dyke 1996). This is particularly notable in connection with accidental formation and emissions and other such strongly abnormal conditions. On the other hand, formation in non-industrial processes also involves great variation, along with uncertainties regarding the rates and magnitudes of formation. Some PCDD/F fingerprints in sources are well established, such as the dominance of Hx- and HpCDFs in the common Finnish PCP mixture (Assmuth and Vartiainen 1994, Vartiainen et al. 1998) or in certain industrial processes (Tysklind et al. 1989, Rappe 1992).

There is a great variety of pathways in the formation and subsequent reactions of DLCs. This corresponds to the variety of DLCs and their precursors, and is affected by the variability in the environmental, organismal and technological reaction conditions. Some reaction products are known; some are linked through common intermediates. Particular conditions of the Baltic

Table 3. Summary of precursors and sources of PCDD/Fs, dlPCBs and other dioxin-like compounds, with particular reference to the Baltic Sea. Cf. 3.2, Annexes 1 and 3, Rappe (1992), Hagenmaier et al. (1994).

Source	Process and precursor; product or use purpose	General importance as source of DLCs	Baltic Sea relevance	Key homologues or congeners formed
PCP mixtures	2346-TeCP+PeCP+246-TCP fungicide, wood preservative use	wide PCDD/F contamination at sawmills and wood	produced in FI (River Kymijoki); wide past use	C_7DFs, C_7DD (+C_6DFs, C_8DFs)
PeCP	fungicide used as wood, leather and textile preservative	contains and forms PCDD/Fs; emissions along the use chain	FI, SW (see above); still used in EU/BS countries	C_7DDs, C_6DDs, C_8DD, C_7DF, C_8DF
2,4,5-T	phenoxy phenol herbicide (245-TCP derivative) against deciduous trees	contains and forms PCDD/Fs (esp. in thermal reactions)	used in large amounts earlier (also with 2,4,-D)	C_4DD
2,4-D	PCPP herbicide (24-DCP derive.) used esp. against deciduous trees	contains and forms small amounts of PCDD/Fs	some use continues (also RUS)	lower yield of TCDD
2,4-DCAs	2,4-dichloroanilide based herbicides	contain PCABs, PCAOBs	limited use in BS countries	TCAB, TCAOB
hexachlorophene	bactericide	contains and forms PCDD/Fs	past use (e.g. hospitals)	C_4DD
hexachlorobenzene	bactericide and fungicide	contains and forms PCDD/Fs	wide use (esp. in the past)	
Etc chlorobiocides	pesticides (e.g. lindane), herbicides	potential PCDD/F sources	wide past use	var.
PCBs	produced esp. for use in capacitors and transformers	all PCBs form PCDD/Fs; include (and form) dlPCBs	wide past use; stockpiles; still produced in RUS	C_4DF, C_5DFs; CB126, CB118 etc
PCNs	produced as dielectric waxes and for other industrial uses	form PCDD/Fs in thermal reactions; DLCs in themselves	relatively wide past use; stockpiles and diffused	
PBBs	produced for use as flame retardants	form PBDD/Fs (+non-coPBBs) and include coPBBs	wide use (some countries); stockpiles and diffused	var. PBCDD/Fs
Cl and chlorate prod.	mainly chloroalkali process with graphite	forms PCDD/Fs esp. in sludges	SW, FI production	C_4DF, C_5DFs, 6C_6DF
PVC	PVC products, PVC/VCM prod/handling (intermediate)	PCDD/Fs from VCM plant; PCDD/Fs from PVC burning	FI coastal (SW inland) plant; open burning	several incl. 4C_5DF
TCE/TeCE/DCE	solvent production	potential PCDD/F formation		
TBBP-A, DBDE etc	flame retardants, Br-epoxy resins	form PBDD/Fs (also UV light)		
Acetylene	production and pyrolysis	also combustion intermediate	possibly important	probably PCDFs
Inorganic chlorides	e.g. in NaCl	form PCDD/Fs in combustion		
Iron and steel industry	sinters, smelters and furnaces (primary and secondary production)	form PCDD/Fs (especially sintering and scrap smelting)	great BS relevance (many facilities also on coast)	4C_5DF, C_4DF, C_8DD, 4,6C_7DD, CB126
Non-iron metal industry	primary and secondary Al, Cu and Zn/Pb smelters and furnaces		great BS relevance (many facilities also on coast)	4C_5DF; C_4DF from Al smelter
Pulp and paper	chlorobleaching (esp. elemental Cl)	form PCDD/Fs (many stages)	high BS relev. (SW, FI)	C_4DF, C_5DD, 6C_6DD
Water chlorination	perchlorite			
Textile industry	chloroaniline/dioxazine dyes, PeCP	extensive usage	possible BS relevance	C_6DFs, C7DF etc
Chemical/petrochem	precursor processes (aromatic Cl/X)	form PCDD/Fs	refineries, pharmaceuticals	var.
Caprolactam	intermediate in nylon production	high PCDD/F emissions	unlikely to be important	
Waste incineration	municipal, industrial, hazardous, medical, mixed wastes (with Cl/Br)	Transfer, transform and form (de novo) PXDD/Fs, coPXBs	past and present activity in many BS countries	var, e.g. C_4DF/C_5DFs and C_5DD
Waste recycling	e.g. PVC waste oil, solvent, scrap	PCDD/F formation	intense in many BS countries	var.
Landfill disposal	municipal/industrial disposal sites (chemical and biological processes)	Transfer, transform and form PCDD/Fs (e.g. landfill fires)	wide varying activity in BS countries; also landfill fires	var.
Wastewater treatment	sludge treatment	municipal & industrial	potentially relevant	sewerage-dependent
Open burning	uncontrolled e.g. backyard	high potential PCDD/F format.	still common in many	C_4DF, C_5DFs etc
Coal burning	industrial and domestic	PCDD/F format (Cl influence)	BS countries (industrial)	C_4DF, 4C_5DF
Peat burning	industrial (also mixed fuels)	some PCDD/F formation	FI, SW	
Wood burning	industrial, domestic (also mixed fuels)	PCDD/Fs esp. from Cl CCA	many BS countries	C_4DFs-C_6DFs
Straw burning	agriculture	forms PCDD/Fs	some BS countries (DK)	
Gasoline engines	2- or 4-stroke; leaded and esp. unleaded petrol (some with DCE)	some PCDD/Fs; higher from unleaded petrol & slow driving	all BS countries; unleaded esp. in new member states	C_8DD (C_4DF, 4C_5DF)
Diesel engines	vehicles, esp. old trucks and run on soot-reducing Cu doped fuel	PCDD/Fs (esp. slow-pace use)	all BS but small emission	
Accidental fires	esp. incomplete burn. of haloarom.	high potential PXDD/F format.	all BS countries, extensive	var.
Bonfires		form PCDD/Fs	all BS countries	var.
Natural/preindustrial	e.g. peat/wood burning, forest fires	form PCDD/F but at low levels		var.

Sea include the low temperatures which present challenges for prediction of the environmental fate of DLCs (Paasivirta and Sinkkonen 1998, Paasivirta et al. 1999, Chen et al. 2002, 2003) as well as for toxicity assessment. The relative importance of the reactions and pathways is also affected by the particular loads of DLCs.

In Baltic Sea countries, particularly important precursors and formation processes for cumulative emissions include (cf. Table 3, Quass et al. 2004a, Wenborn et al. 1999, Hansen and Hansen 2003, Lassen et al. 2002a-d, 2003, Bergqvist et al. 2005):

- **PCPs**, including PeCP and especially the Finnish mixture used for wood preservation
- **Iron and steel** industry, including sintering plants, steel furnaces and scrap recycling
- **Non-iron** industry: smelters and alloying (Al and especially Cu) and scrap recycling
- **Pulp and paper** industry, especially using bleaching with Cl_2 (Sweden and Finland)
- **Chloroalkali** industry (connected especially with pulp and paper industry)
- **2,4,5-T** (in the main use period 1950-80) and potentially other phenoxyphenyl herbicides
- Uncontrolled (intentional) **burning** and accidental fires
- In some areas, **power generation** and heating (by coal and wood, especially unclean).

2.3 Baltic Sea system compartments, processes and risk factors

2.3.1 System boundaries and interactions of the sea with land areas

The semi-detached Baltic Sea has a permeable **boundary** to the North Sea through the Belt Sea, permitting passage of water and solutes carrying dioxins, and of organisms. The boundary is blurred e.g. as fleets operate both in Baltic and North Sea.

The dioxin **donor area** for the Baltic includes both its watershed and other source areas connected through air. The relative significance of these will depend on the assessment approach (cf. 1.4.4, 2.1). There are important inter-compartmental and sea-catchment interactions. Land-based and estuarine processes thus need to be included at some level in risk assessment for the sea. The dioxins in the catchment area enter the sea especially in rivers (Fig. 6).

The dioxin **efflux** e.g. in fish catches goes mainly to the seaboard countries, including their fish consumers. However, some efflux extends to consumers and areas outside the catchment (see below). The Baltic Sea system is linked to other areas by migrations of animals (see below). These accumulate and transport dioxins both into the Baltic Sea and outside it.

In addition to large islands (Gotland, Öland, Åland, Ösel, Dagö, Bornholm, Rűgen, Sjælland, Fyn, Lolland) there are extensive **archipelagos** (Fig. 6). The Baltic Sea thus extends from pelagic to coastal conditions, and interacts with islands and shallow-water areas. These archipelagos and coasts affect hydrography and ecology as well as fisheries. The consideration of such factors is also dependent on the geographical scale of analysis.

2.3.2 Hydrography and ecology

The Baltic Sea is large (415 000 km^2 for the whole Baltic Marine Area) but relatively shallow; its mean depth is c. 55 m (HELCOM 2002a) and water volume is thus small (20 000 km^3). With a mean water influx of c. 700 $km^3\ a^{-1}$ the water retention time is c. 30 a, or 50 a based on river runoff (Bergström et al. 2001). An inland lake at several stages of its post-glacial development, the Baltic is now semi-enclosed. Water intrusion takes place mainly after North Sea water rise (Stigebrandt 2001). At other times river runoff and outflows dominate over seawater inflow in the water balance. The saltwater influxes have an important influence.

The Baltic Sea is divided in **basins** separated by shallow thresholds (Fig. 6). The current patterns vary due to internal and external and stationary and transient conditions. On the average there is counter-clockwise surface circulation. In addition, deepwater wells up and surface water wells down in a vertical circulation, especially in areas of upwelling and downwelling (Myrberg and Andrejev 2003), transferring also DLCs.

The Baltic is divisible in **depth regimes** by stratification due to temperature and salinity gradients. A surface layer of variable depth is separated by a thermocline. Salinity stratification gives rise to relatively constant haloclines at c.

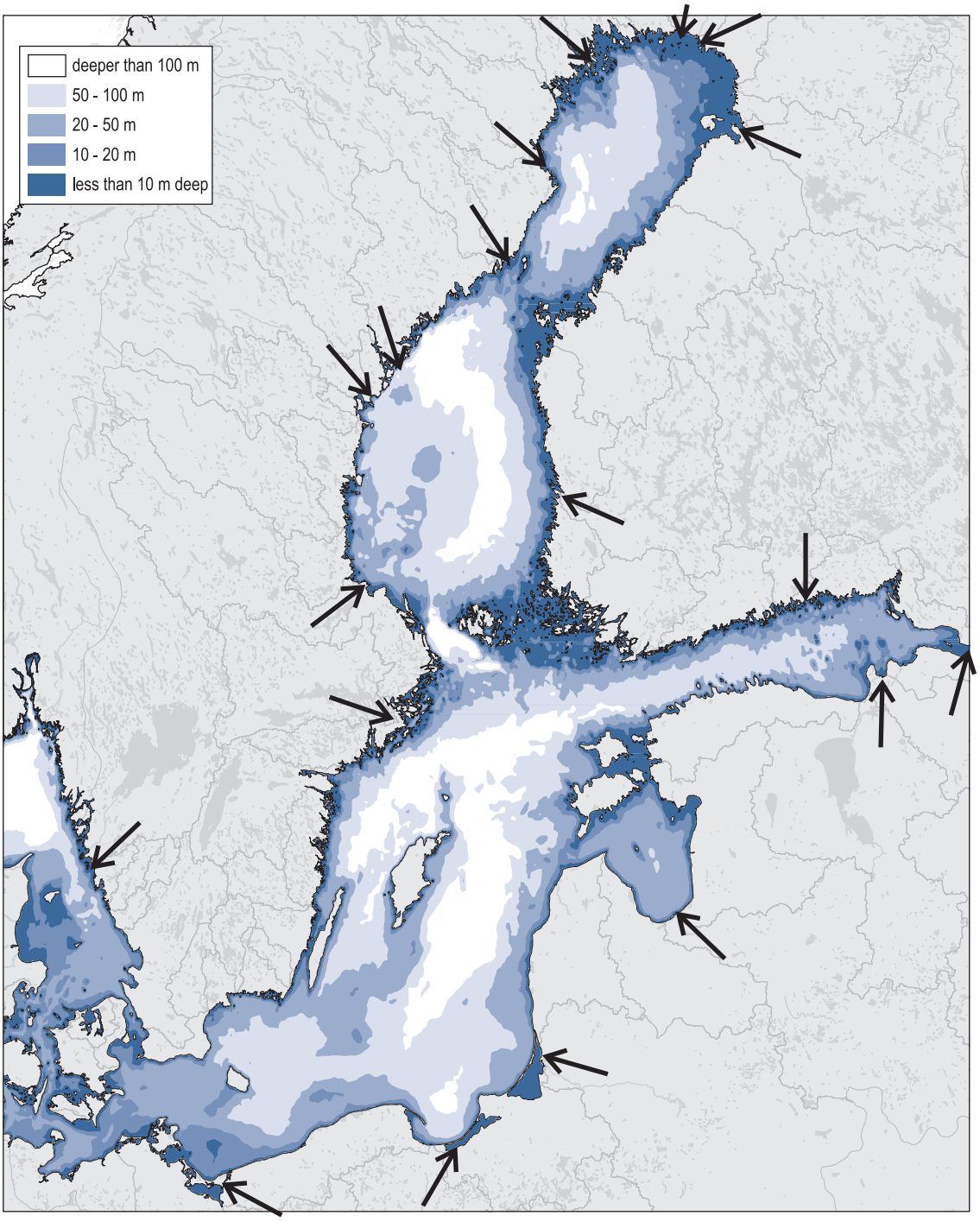


Fig. 6. Bathymetric map of the Baltic Marine Area showing areas within a depth of 10 m (approximate extent of extensive macrophyte beds and dredging), 20 m (average depth of summertime pelagic thermocline and maximum depth of most wave action), 50 m and 100 m (average depths of first and second halocline); lakes in the catchment (no depth grading); boundaries of catchment sub-areas; and major river inlets (arrows). Map source: Data from UNEP Grid Arendal, processed by Samuli Neuvonen/SYKE. Note the geographical distribution of the area below 10 m depth, the river discharge areas, and the continuities and thresholds between deep basins.

50-70 m and 110-150 m. This profoundly affects water quality and currents, and the cycling of DLCs and organisms. These barriers are forced especially by storms, and modified also by bottom topography and currents. In Gulf of Bothnia a comparable stratification does not occur, allowing more efficient mixing and oxygenated bottoms (e.g., Laine 2003). Stratification does not hinder the sedimentation of dioxins in settling particles, but reduces their recycling.

The soft-water surplus renders the Baltic Sea **brackish**. There is a salinity gradient declining from 2 ‰ in the Sounds toward N-E direction to levels approaching sweet-water salinity in Northern Bothnian Bay and Eastern Gulf of Finland. The salinity is low also in coastal

and river inlet areas (HELCOM 2002a). Salinity affects the fate of DLCs directly by physical and chemical mechanisms and indirectly through its influence on the biota that cycles dioxins. Among the main fish species, cod and sprat in particular are dependent on sufficient salinity (e.g., Ojaveer and Lehtonen 2001).

Biological depth regimes are formed by light and bottom type, allowing the development of macrophyte colonies down to 10-15 m. In sediments mixing may extend to >20 cm depth due to bioturbation, waves and currents (Jonsson 2000, cf. Annex 5).

Due to nutrient loading the Baltic Sea has undergone **eutrophication**, increasing e.g. anoxia and TOC accumulation 20-fold over baseline (cf. Jackson et al. 2001), and free nutrient levels several-fold (e.g., Ojaveer and Lehtonen 2001). This impacts DLCs through bioaccumulation in the increased biomass (Gilek et al. 1996, Gunnarsson and Rosenberg 1996), and indirectly e.g. through lowered redox (Bignert et al. 1998) and biomagnification in organically enriched sediments (deBruyn and Gobas 2004). Eutrophication probably contributed to increases also in the biomass and growth of main fish stocks (Ojaveer and Lehtonen 2001); on the other hand, excessive eutrophication lowers them, e.g. through harmful effects of algal growth and anoxia on herring (Aneer 1987, Kääriä et al. 1988).

Organic matter cycles in dissolved (DOC) and particulate (POC) carbon, essentially affecting DLCs. Runoff especially in Northern and Eastern parts is rich on humic substances; most of the organic matter is more easily degraded. Part of the POC with associated DLCs is deposited in sediments and then mineralized, resuspended or buried (Axelman et al. 2001).

The Baltic in general and N-E parts in particular have **low biodiversity**, due to the brackish water, low temperature and young evolutionary age of the sea. The biota is thus subject to considerable stress already due to natural causes. However, productivity also of fish is relatively high due to eutrophication (Ojaveer and Lehtonen 2001). Ecologically, the Baltic is a vulnerable sea due to the particular conditions of salinity, temperature and general hydrochemistry and hydrography.

The species in the various **biotopes** and habitats of the sea, such as pelagic, littoral and benthic zones of the basins, form communities and interact with each other. Despite the low biodiversity and simple trophic structure (see e.g. ICES 2003a,d), the spatial and temporal variation e.g. in life stages and ecology contribute to complexity of the communities.

The biodiversity and species composition change in cycles and fluctuations. In pelagic areas, plankton exhibits seasonal and even diurnal variations. Also the sub-areas differ in terms of ecology due to external and internal factors. Many Baltic Sea species migrate long distances, while others are local and thus vulnerable to and representative of local exposures. Many migrations are related to reproduction and feeding, e.g. for fish (herring, salmon, eel) and birds that include both long-range migrators (e.g. terns, lesser black-backed gull, osprey) and more local migrators. Important influxes of invasive species have occurred, many through humans purposefully (e.g. mink) or inadvertently (especially many invertebrate species transported in bilge water) (Weidema 2000). Also some native species like cormorants have increased explosively.

2.3.3 Fishing, mariculture and other relevant technological processes

Fishing is a key activity for risks of dioxins in Baltic fish. Baltic fishing takes many forms, from recreational and traditional coastal to modern industrial open-sea fishing by large vessels (Fiskeriverket 2001, ICES 2005a,b, Annex 5F). Herring, sprat and cod comprise 95 % of the catch (Table 4). Many stocks, especially cod, are intensively fished. Herring catches have declined to ⅓ of top levels (in late 1970's). Sprat, used largely industrially, has to some degree replaced this loss. Fishing is regulated on the basis of Total Allowable Catches (TAC's) conceded by the International Baltic Sea Fisheries Commission (IBSFC) in negotiations between fishing states, based e.g. on advice from ICES and on EU steering (Annex 11).

An average 20 000 t a^{-1} of mariculture animals were produced in the 1990's in the Baltic Marine Area, mainly Finland and Denmark (HELCOM 2002a). The main species in Finland is rainbow trout, while salmon is cultured in Denmark and blue mussel in Sweden and Denmark. The rainbow trout are mainly fed imported fodder that contains relatively little dioxins e.g. in comparison with herring (Isosaari et al. 2002b) and with the fodder used in Scottish salmon farms (Jacobs 2002a) that have been estimated to

cause considerable dioxin contamination of the fish produce (Hites et al. 2004a, cf.Tuomisto et al. 2004b, and below).

Other present or projected uses and technological risk factors include the following:

- Sediment **dredging** takes place especially for maintenance of harbours and waterways (cf. Annex 5), down to c. 15 m. In comparison with resuspension due to waves (see above), the impacts of dredging on DLCs would seem to be minor. However, dredging is carried out in many contaminated areas; e.g. in the dioxin contaminated river Kymijoki in S-E Finland it has been planned also for remedial purposes (cf. 7.2.3, Annexes 3, 10). Such operations increase the relative significance of dredging for dioxin cycling and subsequent risks and management.
- **Extraction** of marine aggregates: 5 Mm^3 a^{-1} are taken mainly in Denmark (HELCOM 2002a) and affect a minor fraction of DLCs, but may increase
- **Bottom trawling** (see above) impacts the benthic environment by mixing and remobilizing sediments and by disturbing and decimating biota (Floderus and Pihl 1990).
- **Oil and gas production:** Offshore production takes place near Poland's coast and involves emissions that may act as dioxin carriers. Moreover, PAHs are present in mineral oils.
- **Pipelines** and cables: Some cables have been set; notably, a plan has been agreed between Russia and Germany for a pipeline through the Baltic that would affect the sea more extensively
- **Shipping:** Boat traffic causes some DLC emissions, stirs up sediments and disturbs many species, both benthic and surface-dwelling, and thus adds to the effects of dioxins
- **Tourism** and recreation: These impact the ecosystem even if not directly dioxins. They also affect perceptions and requirements concerning the sea, also as to fish and dioxins.

2.3.4 Fluxes of dioxin-like compounds to and from the sea

Runoff in rivers and overland carries solutes and organic and mineral particles to the Baltic. Of particular importance for DLCs is the influx of dissolved (DOC) and particulate (POC) organic carbon. Most DLCs are deposited in near-shore areas, and inter-basin transport is less important than river influx (Miltner and Emeis 2001); uncertainty of carbon fluxes from near-shore areas is however great (Emeis et al. 2002). Some of these influxes are slow due to containment in the catchment, and their significance for risks may change after long lags.

Table 4. Landings of relevant economy fish from the Baltic Sea (ICES subdivisions 22-32) by country in 2004 and 1977 (ICES 2005c). The figures have been rounded to one signifying digit. The two countries with the greatest catches for each species and year have been shown in bold. Note the development of landings.

Country	Landings or mariculture production the Baltic Sea by species, in kt a^{-1}											
	Baltic herring		Sprat		Cod		Salmon[a]		Flounder		Mariculture[f]	
	2004	1977[b]	2004	1977[c]	2004	1977[c]	2004	1977[b]	2004	1977[c]	1998	species cultured
Denmark	<	10	40	7	**20**	**70**	**0.4**	**1**	**3**	2	6	mussels, salmon
Estonia	10	40	30	(10)	1	(6)	<	0.02	<	(0.4)	<	
Finland	10	30	20	7	1	<	0.3	0.1	<	0.2	10	rainbow trout
Germany	4	-	30	20[d]	7	40[d]	0.04	0.08[d]	2	4[d]	<	
Latvia	3	30	50	**(60)**	5	(8)	0.03	0.2	1	(0.7)	<	
Lithuania	2	5	2	(10)	2	(6)	<	0.04	1	(0.2)	<	
Poland	**20**	**60**	**100**	40	**20**	**50**	0.1	<	**8**	4	<	
Sweden	**30**	**50**	**80**	<	3	20	**0.4**	**0.3**	<	0.4	2	salmon, blue mussel
Russia/USSR-Rus	7	30	30	(30)	10	(10)	0.01	0.06	1	(0.3)	<	
Total	90	300	400	200	90[e]	200	1	2	20	10	20	

Explanations: *[a]Includes reared salmon and salmon caught in rivers; [b]For USSR including the Baltic states, 1981 data; [c]For USSR including the Baltic states, estimated based on their shares of the 1992 landings and the total USSR landings in 1977; [d]DDR+GFR; [e]Includes catches unallocated to countries, estimated by ICES; [f]HELCOM 2002a.*

Dioxins enter the Baltic in wet and dry deposition, largely sorbed on particles. The importance and patterns of this load vary according to the DLCs in question, as well as in time, with considerable within-year variation e.g. for PCBs (Agrell et al. 2001). DLCs are also absorbed from vapour phase in air to water.

The PCDD/Fs of most importance have low volatility, but it is higher for lower chlorinated congeners, e.g. for TeCBs. Based on model predictions, average **volatilization** of PCBs from the Baltic has been estimated to be lower than inputs from air (Wania et al. 2001, Axelman et al. 2001).

Industrial and municipal **wastewaters** are discharged directly to the sea (cf. Annex 5). DLCs may enter the Baltic directly also from the following sources:

- Sea-going vessels e.g. in chemicals, waste oils and solid or semisolid wastes
- Offshore activities such as oil and gas production (off the Polish coast)
- Feeding-stuffs to mariculture.

Ocean water from the North Sea enters the Baltic only infrequently in significant quantities and thus carries not a great total influx of dioxins although affected in the deep currents by contaminants in sediments in Kattegat and Skagerrak.

Outflow of water takes place from the Baltic on the surface of the Sounds even when there is influx on the bottom, and removes DLCs, mainly in suspended particles. Dioxins are removed in biota, mainly fish. Most of these are consumed in the catchment and return to the sea in sewage. The main effluxes e.g. in Finland are in sprat to Russian Federation (Abbors 2003). Of other Baltic Sea countries especially Denmark exports fish.

DLCs are removed more permanently through **degradation** (biotic and abiotic) that takes place in all compartments, notably in sediments. The sink due to degradation varies according to the congener, higher chlorinated PCDD/Fs and PCBs being particularly resistant (see below).

Some of the settled dioxins are **buried** to deeper and undisturbed sediment. Especially shallow water sediments are resuspended by waves and currents, some to sediment again. Resuspension rates of around 75 % have been estimated in Baltic basins (Wania et al. 2001).

2.3.5 Processes and properties affecting dioxin cycling and fate in the system

Physico-chemical properties of PXDD/Fs and PXBs vary depending on their structure, especially Cl substitution. Generally, 2,3,7,8-substituted and coplanar congeners and those with more halogens are the most persistent and involatile. Biomagnification may be higher for non-dlPCBs than dlPCBs. PBDD/Fs have higher molecular weights and melting points, lower vapour pressures, water solubilities and Kow values, and weaker C bonds than corresponding PCDD/Fs (IPCS 1998, Birnbaum et al. 2003, Söderström and Marklund 2002).

Biological properties include those in toxico- or pharmacokinetics (cf. 2.5.3); receptor binding activities; and toxicological properties such as potency for the various effects (cf. 4). Within environmental fate and exposure

air↔ ↕	air↔ ↕	air↔ ↕	air↔ ↕	air↔ ↕	air↔ ↕
surface water ↔ plankton planktivor fish (↔) piscivorous fish (↔) seal/piscivor bird ↔	surface water ↔ plankton planktivor fish (↔) piscivor fish (↔) seal/piscivor bird ↔	surface water ↔ plankton planktivor fish (↔) piscivor fish (↔) seal/piscivor bird ↔	surface water plankton planktivorous fish piscivorous fish (↔) seal/piscivor bird ↔	← rivers/lakes anadromous fish piscivor birds	← technosyst/ anthroposphere ↕ humans
↕ bottom water ↔ benthic fish (↔) piscivores (↔)	↕ bottom water benthic fish (↔) piscivores (↔)	↕ bottom water benthic fish (↔) piscivores (↔)	↕ bottom water benthic fish piscivores	↓(↑) soil	↓(↑) soil
↕ surface sediment microb community zoobenthos	↕ surface sediment microb community zoobenthos	↕ surface sediment microb community zoobenthos	↕ surface sediment microb community zoobenthos		
↓(↑) deep sediment	↓(↑) deep sediment	↓(↑) deep sediment	↓(↑) deep sediment	(↕) (low rates) subsoil	(↕) (low rates) subsoil
North Sea	Sounds/S-W Baltic	Baltic Proper	Gulfs (GR, GF, GB)	catchment	long-range area

Fig. 7. Simplified box model of the main compartments involved in the cycling of dioxin-like compounds in the Baltic Sea and adjacent systems. Note that dioxin fluxes take place also by other routes and across boxes.

processes, the inherent biological and physico-chemical properties interact with physiological and ecological properties of the organisms cycling them and exposed to them. In addition to the properties of individual DLCs, their combined properties and interactions both between each other and with other matrices and molecules, including biomolecules, need to be taken into account in risk assessment.

The key environmental **fate processes** involved in the cycling of DLCs (Table 5) include both natural and man-made components. Many processes are inter-connected, and have also indirect consequences for risks. These processes determine the amounts and levels of DLCs in abiotic and biotic compartments of the Baltic, including its fish, and their variations in time and space, and thus the subsequent exposures of human and non-human species.

Bioaccumulation in clupeid (herring and sprat) based food-chains is particularly relevant in the Baltic and the present assessment. The rate of clearance from the organism is a key parameter. Biomagnification varies by compound, animal and conditions, being greater for dlPCB and other PCBs than for PCDD/Fs (van der Oost et al. 2003). On the average DLCs reach equilibria with respect to tissues, and the equilibria change only after a lag. Thus, instead of sudden fluctuations in intakes, long-term cumulative doses are decisive. In some cases rapid clearance takes place, such as through placenta, in eggs or in mother's milk.

Table 5. General characteristics of key processes in the cycling of dioxin-like compounds in the Baltic Sea.

Process type	Specification	Compartments	Influence on DLCs	Determinants (in addition to congener), variations
Physical-chemical				
Volatilization	water-air, solids-air partitioning	land-air, sea-air, sediment-pore gas	removes some from sea; bubble transport in water	- site geometry - meteorology (wind, temperature)
Solubilization	solid-liquid partition	water, other liquids (e.g. body fluids/oils)	generally low for hydrophobic DLCs	- concentration on particles - humus solubilizes; salting-out in estuaries[a]
Photochemical transform/decay	O bond fission; de-Cl, hydroxyl	air, terrestrial surfaces, surface water	-key sink e.g. for PCBs -also toxic products	- radiation intensity, catalysts (e.g. in humic waters)
Other abiotic decay processes	chemical dechlorination	anaerobic sediments	reduces levels, changes congener pattern	- concentration - redox / e shuttles and general microbial conditions[b]
Adsorption/ desorption	on particles / C (soot, organic)	OM/OC (all compartments)	governs particle retention, transport (also sediment)	- compound-specific K_d; sorption sites in matrix - low temperature reduces desorption[c]
Advection/ dispersion	on particles	air, sea, catchment waters	influxes, sea transport, effluxes from the Sounds	currents
Diffusion in liquid phase	for DLCs on colloids/ particles	pore water (+free water/air); body fluids	dispersal; may be notable in sediments	- diffusivity (permeability) of matrix - temperature[d]
Gaseous diffusion	for DLCs in vapor phase	atmosphere to sea and back	net effect is to bind DLCs from air; unknown rates	- temperature
Atmospheric deposition	wet and dry	air-sea, air-land	important influx	particle size and density; meteorological conditions including precipitation
Sedimentation	with particles	sea, lakes/rivers, water to sediment	removes from water phase	particle dimensions and density; OC sedimentation flux (due e.g. to eutrophication)
Resuspension	with particles; also biological	sea, lakes/rivers, sediment to water	remobilizes to water phase	waves/currents, ice action, bioturbation, CH_4 embullition, dredging; bottom quality (depth)
Biological				
Uptake/absorpt (aquat animals)	organisms (passive/active)	-gastro-intest. tract (-dermally, inhalation)	-gut absorption (-fish skin, terrestrial)	-feeding ecology, physiology of organism -medium, conditions (e.g. temperature)
Bioaccumulation	in fat	-all organisms & media -fatty tissues	key carrier to humans and other predators	-compound-specific BAF (cf. K_{ow}) -fraction of OC in sediment, level of exposure
Tissue distribution	in blood, lymph	-all organisms -all compartments	-to adipose, milk, fetus -liver sequestration	-species/strain, age and sex, general condition
Metabolization	-enzymatic (liver and bacterial)	-all organisms (varies) -all compartments	-clearance, mobilizes stores, new DLCs	-species, strain and organ, physiological condition
Excretion/ depuration	-from GIT -metabolic	-all organisms -all compartments	-clears organisms	-also in milk
Biomagnification	-accumulation - depuration	-many organisms -all compartments	concentration in food-chains	fat content and quality, trophic state, feeding habits, general condition; age and sex
Biodegradation	-dehalogenation -O bond break	-bacteria, fungi -animals (cf. above)	removes DLCs but may form toxic metabolites	quantitative mineralization bacteria-dependent; redox (aer or anaer), pH, energy donors

References: *[a]Turner & Rawling 2001; [b]Barkovskii & Adriaens 1996, 1998, Barabas et al. 2004; [c]Johnson & Weber 2001; [d]Paasivirta et al. 1999, Chen et al. 2003.*

2.4 Receptor organisms and risk groups in the Baltic Sea environment

2.4.1 Key organisms in food chains accumulating dioxin-like compounds

A multitude of species is involved with dioxins in the Baltic. Most major taxonomic groups are represented. The key species in ecological and economic terms, their properties and their interactions have been summarized in Fig. 8 and Table 6 (cf. Annex 5 and HELCOM 2002). Fish are of interest due to their use in human consumption and to the biomass of some stocks.

Many animals in all classes are receptors or transmitters of dioxin risks. Invertebrates are mainly transmitters and transformers of dioxins only. Mammals and birds are key vertebrate classes at risk as endotherm animals consuming more food in relative terms than fish. Amphibians are not a key group in the Baltic (cf. Jung and Walker 1997, Gutleb et al. 2000).

Ecological effects of DLCs are directed to several niches in the sea and in the coastal environment where these compounds are transferred in food webs. It has been calculated that benthic organisms and bottom-dwelling fish feeding on them are more contaminated e.g. by PCBs (Campfens and Mackay 1997). However, it has also been found that in endotherm animals, fugacity models have been unsuccessful in predicting the observed higher biomagnification (Norstrom 2002).

Baltic fish species include marine and freshwater species adapted to the brackish water and occupying various habitats from littoral to pelagic and benthic (cf. Annex 5F). There is roughly 10 g of main economy fish per m^2 of sea; most of this biomass is comprised of sprat, industrially fished, and herring. Estimates of their mass in c. 2004 range 1-2 and 0.5-1 Mt, respectively, depending on metric and method. Cod has been decimated especially in the Eastern stock (ICES 2005c). Salmon stocks are not great but valued. They accumulate DLCs especially from sprat (Karlsson et al. 1999). The condition of the stocks varies depending on biological and external factors and fishing pressures (ICES 2005c, Annex 8C). The stocks affect other species and fishing in complex interactions.

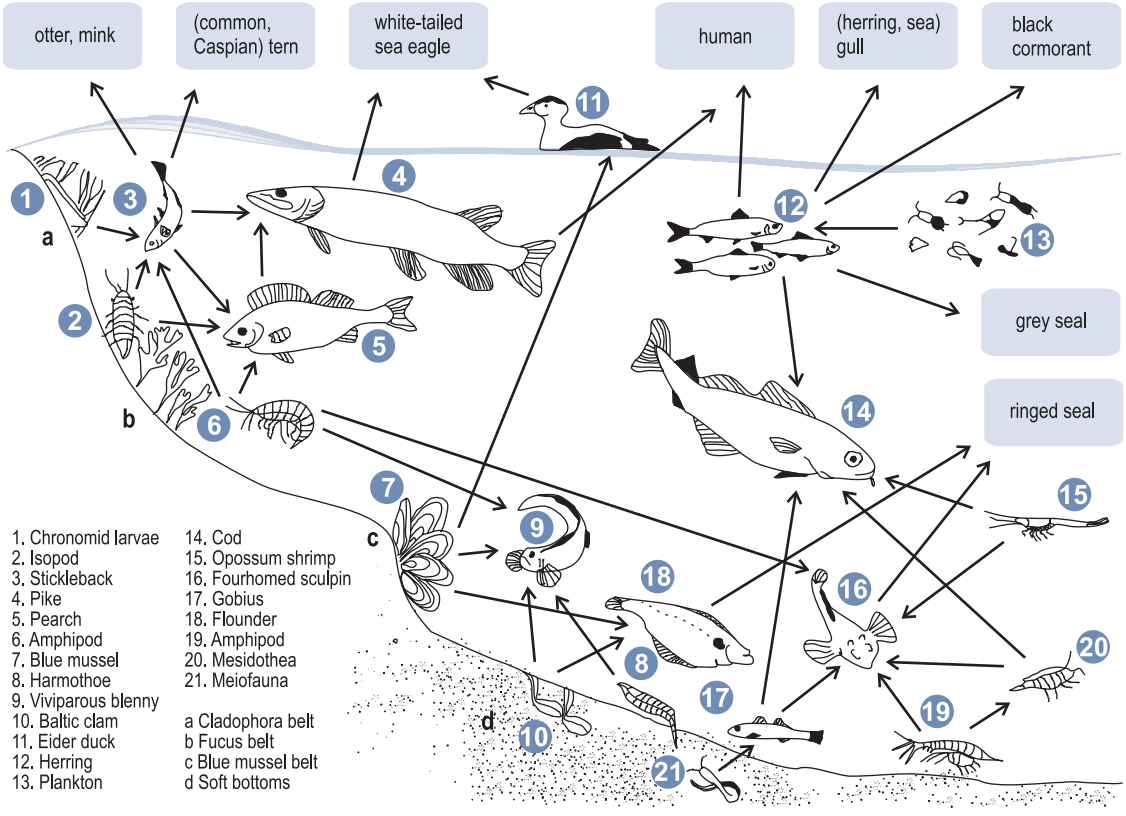


Fig. 8. Mediators and targets of risks from Baltic fish dioxins: Simplified aquatic food web of the key ecological species in the littoral, pelagic and benthic compartments of the Baltic Sea (extended from The Baltic University, www.balticuniv.uu.se/environmentalscience/index.htm).

2.4.2 Particular risk groups

Humans

Some groups of people are particularly vulnerable to dioxins and other DLCs due to high intakes, internal exposures or susceptibilities. These may have genetic, demographic (e.g., gender), life-style such as dietary, or environmental reasons, or, often, many such reasons. Some risk factors are imposed, some voluntary; some biological, some socio-cultural; some are permanent while others are transient.

People may belong to several risk groups, and risk factors may influence each other. This influence may be synergistic and involve positive feedback that leads to non-linearly increasing risks; risk factors may on the other hand have negative feedback and antagonisms. It may be seen that a large array of population segments exist that are at particular risk:

- *Foetuses* exposed *in utero* to dioxins from the mother (or egg) are in many respects in a critical and sensitive stage of development, e.g. neurologically (Brouwer et al. 1995)

Table 6. Key species and groups cycling and exposed to dioxin-like compounds in the Baltic Sea.

Species/group	Distribution	Ecological properties	Role in dioxin cycling and risk formation	Particular sensitivity and risk factors
Bacteria	ubiquitous	chemotrophs, feed on OM	transmitters/-formers of DLCs	degrade DLCs
Cyanobacteria	ubiquitous, also in blooms	autotrophs, N_2-fixing	add to eutrophication, biomass	
Phytoplankton	pelagic, coastal; whole Baltic	primary producers	transmits DLCs to food-chains	cycling depends on species
Copepoda	pelagic, coastal; whole Baltic	food for planktivorous fish	transmits DLCs to food-chains	
Mysis spp. (Crustacaea)	littoral (pelagic-benthic); most of Baltic	key in herring food-chain; vertical migrations	transmit DLCs to (herring) food-chains	food-chain mediated risks
Macoma baltica (bivalve mussel)	benthic (deep open sea); whole Baltic	dominant bivalve in deep benthic macrofauna	transmits DLCs from sediment to fish etc.	food-chain and physically mediated risks
Blue mussel *Mytilus trossulus*	littoral, hard-bottom to c. 20 m deep; BP and SB	important filter-feeder, may dominate littoral benthos	accumulates and transmits DLCs from sediment	efficient turnover of detritus DLCs; harvested (S-W BS)
Saduria entomon isopod	benthic soft-bottom, whole BS	predator on Monoporeia and scavenger; burrowing	high fat content, accumulates and transmits DLCs	food-chain and physically mediated risks
Herring *Clupea harengus*	pelagic except coastal young; whole BS (4 stocks); declined	feeds on zooplankton; some migration (adults)	DLC accumulation in fat especially at higher age	high DLCn levels; important in nutrition and economy
Sprat *Sprattus sprattus*	pelagic clupeid; BP and GF mainly, sustained biomass	feeds in large schools on zooplankton; salmon food	accumulates DLCs in fat at intermediate levels	high dioxin levels; dominant fishery, used mainly for fodder
Stickleback *Gasterosteus a.*	pelagic; BP and GF mainly	feeds on zooplankton; salmon food	may accumulate plenty of DLCs, especially PCBs	important for salmon exposure
Cod *Gadus morhua gallaris*	pelagic in sufficient salinity (esp. S-W BS), 2 main stocks	clupeid predator (prey as roe), spawns at open sea	key pelagic top predator, may control clupeids	liver rich on DLCs; previously important, now unsustainable
Atlantic salmon *Salmo salar*	pelagic except when spawning and as juvenile	pelagic planktivor predator, migrator, river-spawning	magnifies DLCs	high DLCs levels; impaired reprod (maybe unrelated)
Sea (brown) trout *Salmo trutta trutta*	pelagic/littoral except when spawning and as juvenile	predator, short-range migratory, river-spawning	magnifies DLCs	endangered natural rprd; highly sensitive to DLCs
Rainbow trout *Oncorhynchus m.*	coastal; some cultivation areas (fugitive)	farmed, feeds on (mainly) fodder	presently excluded from DLCs cycling and risks in the Baltic	important in food economy, salmon alternative; sensitive
Flounder *Platichthys flesus*	benthic; whole Baltic	feeds on blue mussels and other benthic fauna	fatty; accumulates DLCs	commonly consumed; common lesions in Baltic
Eel *Anguilla anguilla*	coastal benthic; S and Central BS	omnivorous predator; spawn migration to Atlantic	accumulates dlPCBs (esp. CB 126, 118, 156)	seems insensitive to Cyp1a1 induction
Herring gull *Larus argentatus*	coastal, whole Baltic, regional migration	omnivorous predator and scavenger e.g. on herring	accumulates DLCs, especially PCBs	DLC-linked disorders
Lesser black-backed g. *L fuscus*	coastal/surface water; whole Baltic	herring predator, long-range migratory	accumulates DLCs	rprd disorders and mortality (GF) suspected for DLC links
Common tern *Sterna hirundo*	surface water; most of BS	colonial, preys on small fish, long-range migratory	accumulates DLCs, especially PCBs	insensitive to mortal; impaired by DLCs in other regions
Caspian tern *Hydroprogne casp.*	surface water; most of BS (isolated colonies)	colonial, preys on pelagic fish, migratory	accumulates DLCs, especially PCBs; vulnerable colonies	impairment of some stocks in other regions linked with DLC
Black cormorant *Phalacrocorax c.*	surface water and benthic; most of BS	preys on herring & sprat, regional migration, colonial	accumulates DLCs; levels above those in other seas	increased stocks; pollutant-linked disorders (other areas)
Guillemot *Uria aalge*	surface water and benthic, central BS (BP)	colonial, small populations; sprat predator	accumulates DLCs; eggs used in POPs monitoring	mortalities suspected DLC-linked but seem fishing-related
Black guillemot *Cepphus grylle*	benthic, GB	colonial, preys on eelpout	accumulates DLCs	DLC-affected in some other regions but now thriving in BS

Eider *Somateria mollissima*	bottom-feeding diver, whole Baltic	prays mainly on blue mussel, reg/partial migrator	mobilizes DLCs from benthos; females fast when incubating	sensitive to vit A effects; mass mortalities in 1990's
Osprey *Pandion haliaetus*	coasts and inland, surface water feeding; whole Baltic	top predator of fish; migrates to N Africa	accumulates DLCs	
White-tailed sea eagle *Haliaeetus albic.*	coastal, surface water feeding; whole Baltic (scattered populations)	generalist top predator; long-lived, few offspring; reg migrator (young)	accumulates DLCs; levels above those in most other regions	PCB-linked disorders averted, stocks still below normal; may be insensitive
Grey seal *Halichoerus gryp.*	pelagic and coastal; central BS; migrator, seasonal variat	preys on cod, herring etc also in nets; in by-catch	accumulates DLCs, has some metabolic capacity for PCBs	disorders linked to PCBs; recoveries vary, vulnerable
Ringed seal *Phoca hispida*	pelagic and coastal (esp. at breeding); BB (and GF, GR); relatively sedentary	preys on herring, smelt, stickleback, eelpout, crabs; haul out on ice; trapped	accumulates DLCs but has metabolic capacity for some PCBs	disorders linked e.g. to PCBs; var recovery; mass mortalities, other diseases; vulnerable rprd
Harbour seal *Phoca vitulina*	coastal; BP/Kalmarsund and S-W BS (2 populations)	preys on clupeids also in nets; in by-catch	accumulates DLCs but has metabolic capacity	herring-linked immunotox; die-offs; vulnerable rprd
Harbor porpoise *Phocoena phoc.*	coastal in KAT and S-W BS (two distinc populations)	feeds esp. on herring; some caught in by-catch	high accumulation of DLCs; low metabolism for some CBs	endangered; vulnerable rprd and other properties
Otter *Lutra lutra*	terrestrial; mainly inland; whole BS coast sporadically	preys mainly on cyprinid fish and amphibians	accumulates DLCs	sensitive to DLCs; may have suffered; variable recovery
Mink *Mustela vison*	terrestrial/coastal; whole Baltic (fugitive wild stocks)	preys 60 % on fish; terrorizes native species	accumulates DLCs on coast, in farms fed fatty fish	nuisance species fish-feeding; sensitive to DLCs
Human *Homo sapiens*	terrestrial; whole Baltic	ominivore, partly fish-dependant; key impactor and responder	produces/processes DLCs; manages ecosystems e.g. by fisheries	risk groups include fishers, young, reproducing women

- *Breast-fed* children receive a large added dose of DLCs during lactation and are in a sensitive stage. Since breast-feeding is highly beneficial and has been shown to compensate for adverse health effects of DLCs (e.g. Ahlborg et al. 1989), non-breast fed children are however also a risk group in other respects.
- *Juveniles:* Are still in a sensitive stage e.g. as to sexual development, and while not generally receiving high exposures (e.g. in fish dioxins), may have a limited diet (see e.g. Jacobs et al. 2002b) and be susceptible e.g. due to the intensive growth (cf. 5.2.2)
- *Pregnant* women: Are susceptible both as mediators of dioxins to the foetus as well as themselves due to the metabolic changes and stress associated with pregnancy
- Other *women in reproductive age:* Are susceptible both as targets, due e.g. to hormonally conditioned sensitivity, and as mediators of dioxins to their potential future offspring
- *Elderly:* In connection with normal immunosuppression of the elderly, long-term effects of cumulated dioxin exposure on them may become important. The potential importance of this group was also indicated by Neubert (1997-98).
- *Persons with deficiencies*, diseases and other abnormal conditions rendering them vulnerable: Some such conditions are related to fasting that may in itself cause high exposures through dioxins stored in body fat (cf. discussion by Neubert 1997-98)
- *Heavy consumers of Baltic Sea fish* include professional and semi-professional and, increasingly, recreational fishers and their families (Ahlborg et al. 1989, TWGIM 2004a). Consumers of much fatty sea fish have simultaneous benefits; thus, on the contrary, those not using such fish may also be at risk.
- Persons with *other high-dioxin diets:* For cultural or economical reasons, some persons may consume other high-dioxin food items in traditional and modern diets
- *Worker* exposures to DLCs have decreased but some groups continuously receive elevated occupational doses (Ahlborg et al. 1989, Kogevinas 2001)
- *Smokers:* Are at greatly elevated risk due to dioxins and DLCs in smoke and to effects of other components, as shown in both exposure and epidemiological effect studies.
- *Genetically highly susceptible* groups: Persons with anomalous metabolism or sensitivity; some of these factors may be compounded (and hard-distinguished) and lie behind other risk factors mentioned, but may also be separate
- *Other susceptible* groups: These include e.g. combinations of the above, persons exposed to dioxin hotspots and persons

who for some reason such as language are not receptive for information about dietary and other advisories for avoiding exposures.

Other top consumers

It is customary to think anthropocentrically that individual-level variations in risks to non-human animals are not as important as with humans in which also narrowly defined risk groups based on individual and near-individual variation may be relevant. Non-human animals lack occupational exposures, have more limited diet and are unable to switch to other food, and lack cultural adaptability. These animals cannot take dietary advice (de Wit, oral communication 2003); nor do they trade in fish. Many non-human animals are thus vulnerable and difficult to protect, also as their habitat requirements are specific and their habits wild. Their pathological conditions therefore go more easily unnoticed than in humans (Fox 2001).

Non-human animals have partly fundamentally different ecology and considerable physiological differences both with respect to humans and to each other. There is a lack of information on the ecology of many animals also in the Baltic. Therefore, the identification of particular risk groups in non-human animals differs in some respects from those in humans:

- Species with particular ecological or physiological *traits* implying elevated exposures or susceptibility, e.g. in terms of distribution and niches, prey and feeding habits, reproduction, dioxin pharmacokinetics, and sensitivity (cf. Table 6)
- *Embryos and foetuses* are often at particular risk due either to higher exposure (*in utero*, *in ovo* for birds, amphibians and fish) or higher susceptibility due to their sensitive stage of development; this has been studied in detail in experimental animals
- *Neonates and juveniles* are in many respects at a greater risk, either due to higher relative exposures (e.g. being fed certain prey only or reared in a more contaminated environment) or to the generally greater sensitivity in crucial early developmental stages, or to both
- *Reproducing adults* are generally at higher risk due to the common reproductive effects of dioxins, to the reproductive stress and to particular exposures in breeding grounds.
- *Females:* In some species and respects, females are at more risk e.g. because of their hormonal traits (estrous cycles) and vulnerability due to smaller size and reproductive stress and offspring-protecting behaviour; they also transfer dioxins to offspring
- Individuals and groups having particular *deficiencies* and diseases are likely to be hit first or worst by toxic effects. Undernourished animals are typical victims. While partly related to natural variation, this may (if widespread and severe enough) have ecological significance
- *Captive* domestic animals (such as reared salmon, minks, chicken) fed highly and exclusively dioxin-contaminated feedstuffs
- Organisms exposed to local *hot spots:* Especially sessile and territorial species of narrow foraging ranges may be at risk in environments having high dioxin levels and exposures.

2.5 Fish consumption, other intakes and subsequent exposures

2.5.1 Exposure routes

The exposure to PCDD/Fs and PCBs for most **humans** is from diet (cf. 3.5). However, indoor air has been estimated to cause intakes of almost 1 pg TEq kg^{-1} bw d^{-1} (Kohler et al. 2002), i.e. near the lower guideline value. The relative significance of other exposure routes and media may be still greater for some other DLCs and for specific groups such as workers handling dioxin-contaminated materials. The significance of other routes and media may also vary in time and space. For instance, the dietary intakes may be lower for a DLC that is still being produced and has not yet permeated the food production system e.g. in the Baltic Sea.

The routes and patterns of exposure of **non-human animals** are more complex than those of humans. On the other hand, for many animals the selection of pray or food is limited, and Baltic fish may be an exclusive dietary intake route, unlike for humans. The following key points are

to be noted with regard to exposure routes to non-human Baltic Sea animals:

- For many fish, some exposure takes place through *gills.* As DLCs are particle-bound, this is a minor route expect in heavily contaminated environments.
- For bottom-dwelling animals, *dermal* exposure may be a relevant route e.g. for fish with a permeable mucous skin
- Benthic animals and predators on them are exposed to DLCs by *ingestion* of sediment (see e.g. Wenning et al. 2000)
- In *terrestrial* environment, non-human animals are exposed to dioxins from the sea e.g. in fish. These receptors include wild animals and farmed animals given fish-based feeding-stuffs (SCAN 2000). Further diversification of exposure paths is caused by cycling of Baltic Sea DLCs in subsequent stages to other systems and organisms (cf. 2.4).

2.5.2 Consumption of fish and other intakes of dioxin-like compounds

Fish caught from the Baltic is used in many ways by many users (cf. Annex 7A). Uses depend on the species and factors such as size, catch area and time. Industrial uses are distinguished from direct human consumption. Processing to fish meal and oil used in fodder is important. **Cultured fish** and **wild fish** differ; the former are fed fodder containing less contaminated fish.

Baltic Sea fish is used directly or after **processing**. Fish for human consumption is industrially processed to fillets and to other ready-to-eat food. Herring fillets are usually made of large herring and sold with skin. Other types of processed fish include herring and sprat conserves. Smoked herring, salmon, whitefish, flounder and eel are important, treated in cold or hot smoking processes. Previously cod liver was canned but this is restricted due to high contaminant levels (Falandysz et al. 1994d). Traditionally herring is salted, in Sweden also processed through anaerobic digestion to sour herring.

There are great differences in the **consumption** of fish also in Baltic Sea countries (cf. 3.5.1). Professional and recreational fishers and their families are often high consumers. The consumption of Baltic fish is higher in coastal areas due to fish availability and preferences. The average consumption per capita based on the total population differs from the average consumption among fish consumers, as many people never consume fish. In addition to fatty Baltic fish, e.g. imported salmon contributes DLCs (e.g. Lind et al. 2002). Baltic recreational fish is consumed by the fishing population; the amount of recreational fishers is great in both Sweden and Finland, even 25 % of the total population.

Fish is **exported** from Baltic Sea countries (cf. Annex 7). Little data has been found on other than Finnish exports, as Sweden and Denmark also fish in the North Sea. Due to restrictions of marketing contaminated fish, its export may continue to non-EU countries, e.g. Russian Federation.

Non-human animals, wild and farmed, consume Baltic fish, and some depend on it. Some of these animals are not consumed by humans, but others like rainbow trout are (Isosaari et al. 2002b). Fish used as feeding-stuff to edible animals will partly end up in humans (Stark et al. 2002, Easton et al. 2002, Jacobs et al. 2002a,b). Fish and fish oil based fodder has been given especially to pigs and poultry (SCAN 2000). Some Baltic fish has been fed to farmed rainbow trout. However, in Finland during 1990's c. 70 % of herring landings has been given to fur animals (Vartiainen et al. 1997c, Kiviranta et al. 2003). Of these, minks are particularly sensitive to DLCs (see below).

2.5.3 Kinetics of dioxin-like compounds in the body

Not only intakes or ambient environmental or food levels but also many subsequent processes and related risk factors in the exposed organisms govern exposures and thus influence risks. Pharmacokinetics (encompassing kinetics at both toxic and subtoxic dose levels) may be divided in interacting sequences (cf. Annex 7C):

- *Absorption* or uptake (e.g. from gastro-intestinal tract and gills)
- *Distribution* or disposition in the body (transfer between tissues, as well as to offspring)
- *Metabolism,* especially enzymatic conversion in liver to OH and MeHS
- *Excretion* (in both metabolized and non-metabolized form)
- Retention and body burdens
- Resultant internal doses and critical tissue doses.

These processes depend on congeners and biological systems. For instance, some marine mammals have low ability to metabolize some PCBs than other mammals (Norstrom 2002), but some species of seals do metabolize DLCs (cf. 3.4.2, 4.3.2). Moreover, sex and developmental stage as well as individual characteristics play a role for pharmacokinetic behaviour of DLCs. It is also affected by concentration or dose levels of DLCs.

The pharmacokinetics in humans differ in some respects from those in other mammals, due e.g. to different metabolic capacity, tissue distribution and fat contents, overall physiology, and longer (absolute or relative) half-lives for DLCs (cf. 3.5.2). The kinetics in pregnant mothers, foetuses and lactating infants are of particular importance.

The processes and factors operative in dioxin pharmacokinetics and their implications for risk assessment and management have been dealt with in more detail in Section 3 (cf. Annex 7). The pharmacokinetic processes in perinatal exposure are of particular significance, but poorly known especially in humans and young individuals.

2.6 Biological responses to dioxin-like compounds and related stressors

2.6.1 Biochemical and biological basis of dioxin toxicity

DLCs elicit toxic effects primarily through a common pathway, the aryl hydrocarbon receptor, AhR (e.g. Reyes et al. 1992, Okey et al. 1994). As summarized by Birnbaum (1994b), the AhR functions as a transcriptional enhancer, interacting with other regulatory proteins. In fishes more diverse AhRs exist than in mammals (Hansson et al. 2004). AhR mediates altered transcription of a number of genes, including oncogenes and those encoding growth factors, receptors, hormones and metabolizing enzymes. DLCs bind with cytosolic AhR, translocate into the nucleus, dimerize with AhR nuclear translocator (ARNT), a protein ubiquitously expressed in vertebrates and many biological regulatory processes (e.g., Pollenz 2002), and activate genes containing Dioxin Responsive Elements (DRE, cf. Annex 8A). The importance of the corresponding AhR repressor (AHRR) in the AHRR-ARNT complex (cf. Mimura et al. 1999) and of coactivators (Hankinson 2005) has also been realized.

In addition to AhR mediated responses, other action mechanisms of PCDD/Fs have been identified or implied, e.g. for immune effects (Holsapple et al. 1986, Kerkvliet et al. 1990). This is true especially of 1-*ortho* PCBs (e.g., Van den Berg et al. 1998). Also the effects mediated through AhR subsequently include multiple pathways, processes and entities such as Cyp1-type enzymes, other proteins and genes (cf. Annex 8A). The findings that dioxins affect cell proliferation and differentiation, promote tumours and are potent immuno-, developmental, and reproductive toxicants by mechanisms not dependent on cytotoxicity are consistent with the hypothesis that they act by modulating normal cell and tissue growth (Gasiewicz 1997).

Other DLCs can combine dioxin-type activity with other action mechanisms. In particular, PCBs include clearly dioxin-like 0-*ortho* congeners and weakly dioxin-like 1-*ortho* congeners (cf. Table 1) that have also other action mechanisms, as do other PCBs (see e.g. Tan et al. 2004).

2.6.2 Dimensions and continuums of responses

Dioxins are multi-target and multi-dimensional toxicants causing an array of effects in many species, communities and ecosystems. Although many effects have been demonstrated in multiple species, others may be species-specific and have limited human relevance (Table 7). Neubert (1997-98) among others stressed the pronounced inter-species and even inter-strain differences, and detailed some of them in evaluating the scientific basis of risk assessment.

The complexity of effects is accentuated by the heterogeneity of DLCs. Mostly TCDD has been studied but other congeners are present. They generally exhibit qualitatively similar behaviour and effects, but differences exist. The same is true of 0-*ortho* PCBs (Van Leeuwen et al. 2000). They occur together with non-dlPCBs, further complicating the risks. The identification of risks is moreover hampered by polyexposures to still other compounds. Many effects are also not specific and attributable syndromes

but operational definitions of combinations of symptoms (Neubert 1997-98).

Classic and clear toxicological endpoints are linked with preceding and intermediate responses or modulations and effect indicators. Thus, there is a 'vertical' (emergence-related) and a 'horizontal' (system component related) continuum in effects: in the first case from subtle, often biochemical effects to emergent adverse effects, in the other between different systemic

Table 7. Summary of biological effects of dioxin-like compounds based on various sources. Mainly low-dose emergent experimental effects of PCDD/Fs and dlPCBs are included. Species with effects at low doses are underlined. The animals in parentheses denote inconclusively dioxin-attributed effects. Cf. 4.2, 4.3, Annex 8.

Effect type	Endpoint or response	Species mainly studied	Non-TCDD compounds studied			Notes/Baltic Sea relevant factors
			PCDD/F	PCBs	Others	
Reproductive	spermatogenesis	rat, mouse, hamster	x[24]	x[10]	extracts	lowest-dose perinatal effect
	gonad development	rodents, monkeys	C_5DFs[1]	126[1,11]	DLC mix[24]	
	estrous/menstr cycle	rat/rodents (human)		126[11]		BS only tentative
	time to puberty		x[24]	x[24]		
	endometriosis	rodents, monkeys	$4C_5DF$[2]	x[2]	fish (POPs)	unproven linkage in human
	fertility/time-to-pregn	monkey, rodent (man)			p,p'-DDE	not shown in BS
	embryo mortality	mammals, bird, fish		x	OH-CBs	
	sex differentiation	mammals, turtles (man)				(tentative human high dose)
	sex hormones	mammals etc		x		Estr, Test, LH, GnRH, Prog[26]
	sex behavior/parenting	rodent, monkey, bird				
Developmental	genital weight	rat (some), mice		169		seminal, testis/prostate
	tooth development	rat (man), monkey,mink		126[12]		missing, loose, demineral t.
	bone devel/density	rat		126[13]		
	birth weight/size	rodents (human)		x	fish POPs	also litter size (exp animals)
	beak deformations	birds (e.g., terns)		126[14]		
	puberty delay	rat, hamster				cf. sex hormone/rprd effects
	mammary differentiat	rat		126[15]		
	hydronephrosis	rodents (esp. mice)	C_5DFs[3]		PBDFs[25]	lower-dose than cleft palate
	kidney fibrosis	rhesus monkeys				
	cleft palate	mice	C_5DFs[3]		PBDFs[25]	
	cardiac deform & tox.	chicken				not seen in other species
Neurological, behavioral (apart from sex behavior)	motor function, perform	rat (human)		x[16]		sex-related; +hypotonia
	feminized play	monkey (human)				sex-related
	learning	monkey				also non-dlPCBs
	hyperactivity, spontan	rat, mouse		126[17]		also non-dlPCBs
	diet preference behav	rat		126[18]		saccharin use; sex-depend.
Immunological and related	host resistance	rodents, seals (human)	(x)	x		
	immunosuppression	rodents, seals (human)	$4C_5DF$[4]	126 etc[4]		esp. mice
	thymus atrophy	various	x	x	TCAB	hallmark effect
	T and B cells	various		x	TCAOB	subset etc changes
	antibody levels	various				mixed results
Metabolic, non-sexual endocrine and gastrointestinal	thyroid (T4/3/TSH etc)	mammals, birds, fish	$4C_5DF$[5]	x[19]		neurol, reprod effect links
	adrenal etc (HPA axis)	mammals, birds				linked to many other effects
	glucose, lipid metabol	rodents (human)				e.g. cholesterol levels
	vit A homeostasis	rodent, seal, mink, bird		126[20]	fish[23]	related to rprd, immune etc
	vit K homeostasis	rodents				
	wasting/anorexia	all	x	x	TCAB etc	characteristic effect
	porphyria	rodents, birds		x		common effect
	intenstine CCK peptide	rats	$4C_5DF$[6]	126[6]		
	oxidative stress	rodents	$4C_5DF$[7]	126[7]		
	MFO enzyme induction	all	x	x		characteristic effect in some
	thermoregulation	rat	x	x		marker/mediator response
Tumors	var (e.g. liver, lung, lymphoma, soft tissue, thyroid, mammary)	rat/rodents (human)	PeCDD[8], $4C_5DF$[9]	x 126[9]	CB126[9], PBBs, fish	TCDD/DLCs probably weak human carcinogens (e.g. compared to other effects)
Genotoxic/mutagenic	overt genotoxicity	rodents			CB 77, PAH	TCDD not genotoxic
Hepatotoxic	liver lesions	rodents, chick etc			PCNs	
Epidermal	chloracne	rodents, human	x		x	hi-dose (transient) hallmark
	hyperkeratinisation	rabbit, monkey	x		x	hirsutism immune-linked
Cardiovascular	cardiac deformation	chicken				
	cardiovascular pathol	rodents		126[21]	TCAB	
	cardiovasc R factors	rat	x	126[22]		cholesterol, blood p etc
	heart/circulat disease	(human)				

References and explanations *(mainly studies of emergent effects of 4-PeCDF and CB 126 in laboratory animals and mammals and birds feeding on Baltic Sea, cf. 4): [1]Gao & al. 2000; [2]Johnson & al. 1997; [3]Birnbaum & al. 1987a; [4]Harper & al. 1993; [5]Brewster & al. 1988a; [6]Lee & al. 2000; [7]Hassoun & al. 2002; [8]Waern & al. 1991; [9]Walker & al. 2005; [10]Pflieger-Bruss & al. 1999; [11]Muto & al. 2003; [12]Render & al. 2001; [13]Lind & al. 2000b; [14]Hoffman & al. 1998; [15]Muto & al. 2002; [16]Rice 1999; [17]Holene & al. 1998, Eriksson & al. 1998; [18]Amin & al. 2000; [19]van Birgelen & al. 1995b; [20]Håkansson & al. 1994; [21]Jokinen & al. 2003; [22]Lind & al. 2004; [23]Håkansson & al. 1991; [24]Hamm & al. 2003; [25]Birnbaum & al. 1991; [26]E=estrogen, Test=testosterone, LH=luteinizing hormone, GnRH=gonadotropin releasing hormone, Prog=progestrone.*

types of effects (e.g. immune, endocrine, reproductive and developmental, being closely connected). Due to the development of advanced biochemical and other measurement methods, an increasing amount of responses are observed that may not be characterized as adverse.

The identification and assessment of dioxin effects is affected and complicated by socio-psychological reactions to dioxins.

2.6.3 Effect profiles and receptor organisms

In many vertebrates, TCDD and other DLCs including dlPCBs are developmental and reproductive toxicants, immunotoxicants and endocrine modulators, and affect growth and metabolism in many ways (Table 7). The crucial effects include reproductive and neurobehavioral developmental effects in offspring at low maternal exposure. Low-dose immune effects are common in mice. DLC are tumorigenic in many animals and sites, but non-carcinogenic effects are more commonly important.

In humans, unequivocal evidence for causal links between exposure to specific PCDD/Fs and adverse effects has only been produced for chloracne (e.g., USEPA 2000a), at high exposure levels. There is some support for other effects in humans also on lower doses, but much contradicting evidence. The causative role of specific DLCs in all endpoints is complicated by possible polyexposures and confounding factors, and is to a considerably degree dependent on the criteria for data quality and evidence that are adopted, as will be discussed in more detail below and in Annex 8B.

2.7 Compound-specific initial risk identification for in-depth assessment

To identify information gaps an initial comparative assessment of potential DLCs was done (cf. Assmuth 2003) based on ratios between key properties of DLCs and those of TCDD in terms of body burdens and toxicity, and combining the ratios to indices of relative risks. Also other information on the relative importance of the various DLCs in Baltic Sea was accounted for (cf. Annex 1).

Identification of the key congeners was further focused on PCDD/FS and dlPCBs and on those species of top consumers of Baltic fish consumers for which there are body burden data covering both groups of DLCs (Table 8). Through body burdens, the influence of pharmacokinetics on the relative toxicity of the congeners is reduced. However, it can be seen that depending on the TEF system (the receptor animal group considered), great differences are seen in the relative importance of congeners.

Despite the dependence on the TEF system, it can be initially concluded in a generalizing and qualitative manner for the purpose of risk identification that most of the risks to Baltic fish consumers seem attributable to the following congeners within the WHO TEF set:

- *4-PeCDF:* key contributor to total dioxin-like toxicity to most Baltic herring consumers
- *PeCDD:* key contributor to total dioxin-like toxicity and risk especially in humans
- *CB 126:* key contributor to total dioxin-like toxicity in all groups of fish consumers
- *CB 156:* important contributor to total dioxin-like toxicity especially in seals
- *CB 118:* important contributor to total dioxin-like toxicity in seals; a marker PCB.

Literature was searched and evaluated especially regarding these congeners. In some Baltic Sea systems and based on additional factors and prioritization criteria, also other DLCs are important, e.g. TCDD, TCDF, CB 105, CB 77, and CB 169 (Table 8, cf. Annex 1).

Several other substances can also be identified for in-depth risk assessment. These include some other PCBs, some PCNs (the 2,3,6,7-chlorinated congeners CN 48, 54, 66, 67, 70, 73 and 75), some PAHs (cf. 5.2.1) and main metabolites of the above, e.g. OH-PCBs first identified in the field in Baltic seals and guillemot (Jansson et al. 1975). The relative significance of many of these remains difficult to assess due e.g. to lacking basic data on sources, properties, levels and effects.

Table 8. Examples of the relative significance of key congeners of PCDD/Fs and dlPCBs in various species, populations and tissues of humans and non-human animals consuming Baltic Sea fish, based on their contributions to total dioxin-like toxicity as approximated by WHO-TEq$_{DFP}$ for the relevant group of animals. The percentages are rounded to one signifying figure due to order-of magnitude uncertainties in TEFs. The 1st, 2nd and 3rd congener in order of significance are shown in bold italics, bold, and italics, respectively. Cf. Chapters 3 and 5 for more extensive evaluation.

Species, group, region	Average age, a; reproductive status	Tissue sampled	Sampling period	Most important congeners (% of total WHO-TEq$_{DFP}$) based on current TEFs for the relevant fish-consuming animal group					
				PeCDD	4-PeCDF	CB126	CB118	CB 156	Next in rank
Human populations in countries with substantial consumption of Baltic Sea fish									
Fishermen, SW BS coast[a]	51-60	blood plasma	c. 1994	10	***30***	**20-30**	4	*10-20*	TCDD 4-9
Referent males (SW BS coast)[a]	48-57	blood plasma	c. 1994	10	**20**	***30***	5	*20*	TCDD 5
Fishermen, FI coast[b]	58	blood plasma	1997	***20***	**20**	*10*	6	10	6-HxCDD 8
Fishermen, coast+inland, FI[b]	58	blood plasma	1997	***30***	**20**	*10*	3	10	6-HxCDD 10
Women, urban SW near BS[c]	28; primipara	mother's milk	1996-99	9	**10**	***20***	5	*10*	6-HxCDD 4
Women, all regions, SW[d]	29; primipara	mother's milk	1992	10	**20**	***30***	5	*10*	TCDD 8
Women, urban FI[e]	28; primipara	mother's milk	1992-94	*20*	**20**	***20***	5	10	TCDD 7
Baltic Sea living non-human animals									
Ringed seal, well-fed, GOF[f]	8-10 mo	blubber fat	1991-92	10	5	***30***	*10*	**20**	CB105 10[l]
Grey seal, well-fed, GOF[f]	<2 mo	blubber fat	1991-92	3	<	**30**	**30**	***40***	CB105 20[l]
Harbor seal, Kattegat[g]	c. 1	blubber fat	1988	?[g]	?[g]	***30***	*20*	**30**	CB157 10
Harbor porpoise, South BS[h]	?; immature	blubber fat	1985-93	2	2	**20**	***70***	?[h]	CB169 1
White-tailed sea eagle, FI coast[i]	mature	breast muscle	1988-91	4	**10**	***60***	1	3	*CB77 9*
Salmon, BOB, fish TEF based[j]	mature fem	muscle	1988-92	**20**	***70***	*10*	-	-	TCDF 5
Salmon, BOB, bird TEF based[j]	mature fem	muscle	1990-92	1	***10***	*20*	<	?	**CB77 50**
Salmon, mammal TEF based[j]	mature fem	muscle	1990-92	4	20	***70***	*7*	?	TCDF 3
Herring, FI, fish TEF based[k]	small (≤18 cm)	muscle+skin	1993-94	**20**	***60***	2	<	<	*TCDD 5*
Herring, FI, bird TEF based[k]	small (≤18 cm)	muscle+skin	1993-94	*10*	***50***	**10**	<	<	*TCDF 10*
Herring, mammal TEF based[k]	small (≤18 cm)	muscle+skin	1993-94	*10*	***40***	**30**	5	3	TCDF 7

References and notes *(cf. list of abbreviations): [a]Svensson & al. 1995a; [b]Kiviranta & al. 2002a; [c]Glynn & al. 2001; [d]Norén & Meironyté 2000; [e]Kiviranta & al. 1999; [f]Koistinen & al. 1997b; [g]Storr-Hansen & Spliid 1993a, based on PCBs only (WHO-TEqP) but PCDD/Fs may be assumed insignificant; [h]Berggren & al. 1999 data for PCB 156 not reported but may have been absent (cf. Falandysz & al. 1994e); [i]Koistinen & al. 1995a; [j]Vuorinen & al. 1997, based on WHO-TEFs for fish (cf. the fish-TEqs given in publication) [k]Kiviranta & al. 2003; [l]In the earlier data on seals reported by Koistinen & al. 1990, also HxCDFs were important (with dominant TEqs among PCDD/Fs) but few dlPCBs were included in these analyses, and thus total TEqs and the shares of individual congeners can not be estimated.*

2.8 Other risks and impacts of dioxin-like compounds in the Baltic, including indirect risks and benefits from fish

2.8.1 General

Dioxins have psychological effects and may cause mental ailments especially in sensitive persons. Some of these impacts result from fears of contamination or of regulatory and other consequences that are only weakly based on biological responses. Such impacts may be pronounced in heavily contaminated areas and segments of population.

Among social impacts, reactions among stakeholders are caused, some premeditated, some unintended. As information on dioxin risks improves, existing policies and regulations may be developed; they are not fixed facts in the same sense as natural conditions for risks.

Indirect impacts are caused by dioxins in natural, social and combined processes. Arrays of repercussions are caused by dioxin risks and responses to them at all stages, from sources to use and non-use of dioxin-laden fish. Indirect impacts include effects on research and other information generating and processing activities.

Beneficial impacts from dioxin contaminated Baltic fish are accrued to health and other ends, both of humans and other

animals. Losses of benefits associated with fish are examples of counter-veiling risks caused by control of DLCs by reducing fish consumption. Economic losses and risks are caused to fishermen and their families, to the fishing industry and to the economy and society at large. Thus, e.g. the difficulties in devising and implementing compensatory mechanisms belong to counter-veiling social risks. Risks and benefits of dioxins in a food chain are therefore intimately linked and fused.

2.8.2 Health benefits associated with fish contaminated by dioxin-like compounds

The benefits associated with Baltic fish rich in DLCs may be divided in a) benefits to human health from fish consumption; b) benefits to the health of other organisms; c) other benefits to societies. These categories are not exclusive; e.g., health benefits may entail economic benefits, and *vice versa*. However, the last mentioned category is treated separately (2.8.3).

Human health benefits from Baltic fish are of particular importance in integrated and comparative risk assessment and in management. Health benefits from fish in relation to risks have not been explicitly treated in official assessments, with the exception of SPCFC (2005).

Health benefits from fish in general and fatty sea fish in particular include the following (cf. 4.4.2, SPCFC 2005, Annex 8C):

- *Cardiovascular* health: Fish intake has been shown to carry considerable benefits due especially to long-chain n-3 polyunsaturated fatty acids (LC n-3 PUFAs) in fatty sea fish
- *Bone* development: Vitamin D, abundant in fatty fish, prevents bone deficiencies
- *Growth:* Fish consumption enhances foetal growth, postulated to be due to fatty acids
- *Neurological* development: Fatty fish may have a beneficial effect on brain and eye development
- *Immune* system: Fish oils may be protective in inflammatory conditions, e.g. arthritis
- *Metabolism:* Beneficial effects on diabetes are indicated; Se is an important antioxidant
- *Cancer:* Fatty sea fish may have beneficial effects on some cancers.

Many of the health benefits from dioxin-rich fish are associated with fatty fish, others are accrued from fish in general, and may (partly) be obtained also without such fish, e.g. by surrogate formulas. Some of the health benefits from fish presuppose that people deprived of such fish consume more unhealthy food items instead. However, this may not have to be the case (cf. 8, Annex 8C).

2.8.3 Other impacts

Other impacts of dioxins in Baltic Sea fish include the following:

- Fisheries impact the *ecosystem* directly and indirectly, in the stocks fished and through community structure, by-catch, discards and bottom contact. On the other hand, fisheries are instruments in managing (and benefiting from) the system. Fishing restrictions influence the sea, partly conditioned by social factors (e.g. declines in fish consumption).
- Fisheries have *social impacts* through adjacent livelihoods, market chains and social structures. Some of these impacts are felt at the level of national macroeconomies, some are distributed to economically less developed areas e.g. in coastal archipelagos
- Fish and fisheries have *cultural* impacts and importance both in fishing practices themselves and in life-style (also recreation) and culinary habits; they also provide important education of sustainable use of natural resources and natural ways of living (see e.g. IOM 2003).
- Other risks and impacts *in turn influence* risks from dioxins and play a role in their management. For instance, risk management in the fisheries area, typically aiming at optimal and, increasingly, sustainable and integrated (e.g. multi-species) use of the stocks (ICES 2005a,b), crucially influences and modifies what risks are caused by dioxins in the fish.

3.1 Assessment principles and evaluation of the quality of information

3.1.1 General considerations

Exposure processes and their stages

Exposure assessment is the other foot of risk assessment: "Allein die Dosis macht, das ein Ding kein Gift ist." (Paracelsus). The contents vary by case and context. It is common in ecotoxicology to focus assessment to external exposures, even only environmental concentrations (e.g. in standard EU risk assessments), and associated processes of environmental fate. In human health risk assessment also other, subsequent stages of exposure including internal doses have been traditionally specified.

The appropriate contents and conduct of exposure assessment are affected by the characteristics of the substances and of the systems to be assessed, by the state of knowledge on them, and by the purposes of assessment. For DLCs, being accumulative and partly well researched, a detailed, extensive and response-oriented exposure assessment is called for. An attempt is made to balance this detail with a decision-oriented and straightforward approach, to simplify the task as far as possible.

Due to the great variation in pharmaco-kinetic processes (see below) between various DLCs and various species or generally biological systems, body burden is preferable as an exposure metric over daily intake (or, still worse, concentrations in diet or other exposure media). Body burdens also better aggregate cumulative exposures. On the other hand, body burdens may need to be specified e.g. in terms of the levels in critical target organs and tissues.

3.1.2 Data and models

Data relevance and quality

Data relevance and quality are an important part of resolving complexity and variation in risk assessment and risk management. A balance between generalization and specification in data (as in models, see below) is particularly challenging with dioxins, being complex and simultaneously requiring much specific and high-resolution data, and thus easily ending in a 'data-rich, information-poor syndrome'.

The relevance and quality of measurements of DLCs are influenced by many factors and are often poor, due to the inherent properties of these compounds and their low concentrations but also to other reasons such as some sampling difficulties e.g. in some Baltic Sea animals including humans, and costs (which are affected by the first mentioned factors).

Data quality and quality assurance have undergone changes and improvements, e.g. in sampling, bioassays and monitoring. However, improvements in sensitivity and specificity of measurements give an overly optimistic picture. The overall data quality is also often unclear. Rose and Startin (2003) pointed out that details of accuracy and precision are often absent from reports of data on PCDD/Fs and dlPCBs, and that there are a disturbing number of statements in the literature e.g. about differences between foods or locations that probably reflect simply differences in analytical and data assessment methodologies.

In many assessments the relevance and quality of measurements and associated constraints have not been sufficiently taken into account. Data quality considerations guide the selection (and exclusion) of data. On the other hand, by utilizing information on data quality, opportunities can be found for more efficient use and production of data.

Analytical quality depends on a chain of operations including the coverage and quality of sampling and, particularly in the case of dioxins, data analysis, although these are typically subject to less quantitative quality assurance than in laboratory analysis. The total analytical quality and the relative significance of various sources of error can thus be difficult to assess even in studies where laboratory analysis methods are documented. Moreover, a disproportionate attention can be given to the quality of laboratory

analysis in relation to other sources of variation and to other specifications of exposures (see also Grandjean et al. 2004).

Congener-specific measurements of DLCs have been made with a good degree of analytical quality control only since early 1990's, especially at low concentration levels and in difficult matrices, including fatty fish (cf. Annex 2A). Older data may be useful in single cases especially for priority congeners analyzed routinely already earlier (e.g. often TCDD) and in cases where data on aggregated isomers suffice. The relevance and utility of data depends on the use purpose, e.g. considering the overall imprecision of risk estimates.

Other DLCs and still other compounds are often present at orders of magnitude higher levels than PCDD/Fs, and interfere with analysis. They are amenable to simpler analytical methods. However, some groups are still difficult to measure, due in part to co-elution of congeners; this applies e.g. to some PCNs (notably the dioxin-like CN 73 and the non-dioxinlike CN 74), many PCBs (Kiviranta et al. 2002a, Harju et al. 2003), PBDD/Fs (Li et al. 2003) and PCTs (de Boer 2000). Marklund and Söderström (2003) pointed out the possibility that important PCBDD/Fs, formed e.g. in incineration of brominated compounds, escape notice as they are not readily measurable.

It seems the production of data on environmental levels is often seen as an end in itself. Monitoring systems have some tendency to generate more data without sufficient consideration of whether the information is that most crucially needed. Typically the perceptions of and requirements for coverage and precision of measurements are unrealistically high in relation to other uncertainties in management decisions. There is also a common belief that ever more data will provide the essential answers to dioxin risk questions. More generally, a preoccupation with monitoring may sometimes be seen partly as an evasive reaction to difficult risk issues in regulation.

A significant problem is the reporting of TEqs without congener-specific data. The problem is aggravated with dlPCBs, the TEFs for which have undergone particularly great changes. Consequently, TEqs are typically several-fold lower in many Baltic fish consuming animals when calculated using WHO-TEFs instead of earlier TEFs proposed for mammals that included di-*ortho* PCBs and assigned higher relative toxicity to 1-*ortho* PCBs. Regarding such changes it clearly is nonsensical to insist on several decimal places (in TEqs).

Even when congener specific analyses have been made, data on non-2378-chlorinated congeners are usually not given. This usually suffices for toxicological assessment. However, often the non-2378-chlorinated congeners make up the bulk of all PCDD/Fs and may have importance also for the toxic congeners; the information losses in this have been stressed e.g. by Gaus et al. (2002, oral communication 2002).

Considering the great variation in measurements, data on levels and trends have to be evaluated critically especially when pooling data. Comparability may be sufficient for a use purpose within a data set that has been produced using the same methods. However, even in cases of long-term repeated analysis there may have been systematic changes in methods, potentially causing changes in concentration levels that are only apparent. This possibility has been pointed out e.g. in connection with discussions on whether human intake estimates (based on fish levels) reflect real declines in trends (de Wit, oral communication 2003).

The presence and significance of such systematic changes would have to be evaluated on the basis of chemometric data and methods descriptions specifying e.g. LODs and other quality assurance data. It is beyond the scope of the present assessment to access and analyze such primary data (often unpublished). Based on methods studies (see 3.1) and on general factors of perception among data generators, hosts and users, it is the belief of the present authors that the quality and comparability in data are often assumed to be higher than they are in reality or, in any case, than can be ascertained and demonstrated.

In summary, manifold (>100 %) variations and errors in measurements of dioxin levels are encountered especially between laboratories and methods. Analytical quality can be low especially at low concentrations and in difficult matrices and for analytically difficult DLCs. Moreover, such measures of data quality do not yet account for sampling variability.

It is customary to compare data and to draw conclusions at a level of precision that is much higher than that achievable among laboratories, e.g., to the second signifying decimal and distinguishing between levels that comply with or violate limit values only by some 10 %. The appropriate precision and accuracy depend to

a certain extent on the case. A greater precision is thus natural e.g. for estimates of human body weight than of dioxin emissions. Likewise, for resolution of some differences and trends greater precision may be needed and possible (if errors in data to be compared are similar). However, from an integrated risk assessment point of view there exists unnecessary, illusory and even nonsensical precision, e.g. as concentrations have been expressed to the 5th decimal (Anon. 2004, ELICC report) when their toxicity is expressed in orders of magnitude, and when some fate and effect model estimates differ by a factor of 1000.

An important constraint of quantitative assessment of PCDD/Fs and dlPCBs is caused by their low levels in comparison with limits of detection. Hays and Aylward (2003) stressed that a majority of the analyses also in surveys of foods may still result in non-detectable levels, despite increasing sensitivity, influencing estimates of intakes strongly (see Annex 2).

In the present work, the following quality evaluation principles have generally been considered for decisions on inclusion and evaluation of measurement data:

- In chemical analysis, HRGC/HRMS level and good QA/QC, e.g. achieved by participation in inter-calibrations (Annex 11)
- Reported variation metrics and QA/QC procedures (usually peer review publications)
- Quantitative data have been expressed and utilized to the first signifying number only
- Congener-specific data have thus been usually accepted from c. 1990 onwards. Exceptions have been made depending on the case. For instance, reliable quantification of amounts and doses in effects studies has been possible during a much longer time, especially for TCDD; also some earlier analyses of dioxin levels in environmental matrices have been included when the quality has been deemed sufficient for the purpose.

As to the relevance and quality of exposure-related information in terms of Baltic Sea assessment, sometimes the resolution or specification of data is not sufficient to elucidate the particular Baltic Sea aspects. For instance, data on levels in fish especially from Denmark, Germany and Sweden often does not distinguish between Baltic Sea and North Sea stocks. On the other hand, in some cases information from other regions, e.g. on sources of DLCs, may be generalizable to the Baltic, within the constraints of the assessment.

Simulation models and integration of empirical data and theoretical models

Model has been defined as "tool of thinking to formulate hypotheses that have to be verified by observations" (Robert Wetzel). Models in social sciences have a different meaning e.g. as to hypothesis testing, often having more the character of aides for interpretation. Models include not only quantitative mathematical and simulation models but also statistical and conceptual models; the term may also denote a surrogate or indicator system.

Operational and data components of a scientific model can be validated but the theoretical component cannot (Rykiel 1996). It is better to speak of model evaluation against available alternatives, and evaluation of the general assessment process (Upton 1994). However, practice is often characterized by vague, subjective claims that predictions show `acceptable' agreement with data that provide little basis for choosing models (Kirchner et al. 1996). In the present work, models have been evaluated in general terms.

In some cases, fate processes need to be specified. On the other hand, if complexity is regular, it is easier to treat analytically. Some complexity can indeed even be used as an ally in analysis, such as when developing, testing and validating (to the level possible) transport and fate models with the help of congener profiles; such models may also be able to connect sources and immissions (or exposures) and other such entities that are relatively far detached without having to treat the intermediate processes in detail (e.g., Su and Christensen 1997).

The relevance and quality of information and the level of specification and detail in relation to aggregations and generalization depend on the case and purpose of analysis, as with physiological and toxicokinetic models; no standard criteria can be set (see Dybing 2003). In exposure modelling, several areas or stages are distinguished (appropriate models in some of these categories will be described and evaluated in the following).

- Emission and release models e.g. based on emission factors

- Quantitative structure-property relationship (QSPR) models and structure-activity models
- Partitioning models, notable fugacity based
- Environmental fate and transport models
- Biomagnification models, including those based on bioenergetic models
- Intake models
- Pharmacokinetic, e.g. physiologically based models.

Approaches to exposure assessment in the present work can be characterized by the following points or foci:

- Mainly 'top-down' i.e. **body burden** based approach is used
- **Identification** of exposures, contributing factors and associated uncertainties are focused on
- The assessment is largely **comparative**
- The assessment if focused on **human** exposures from fish diet
- For ecotoxicological assessment, mainly the body burden data on key **herring predators** are used
- Some account of **toxicokinetics** is made mainly for humans and well-studied animal models
- **Temporal aspects** of exposures are addressed on a case-by-case basis
- **Variations** in exposures are treated mainly qualitatively and semi-quantitatively (through ranges)
- Attention is given to the resolution and specification of **Baltic Sea** relevant information
- Attention is given to particular **risk groups** and scenarios such as perinatal and heavy fish use
- Specific assessments are presented for **4PeDF and CB 126**; in addition, WHO-TEqs are used
- Environmental fate is addressed without explicit mathematical modelling
- The emphasis is on exposures as **linked with effects and management** processes

3.2 Sources and emissions of dioxin-like compounds to the Baltic Sea

3.2.1 Polychlorodibenzo-p-dioxins and furans

Emission inventories and estimates

The inventories and estimates of emissions of dioxins are severely constrained. There have been many definitions of source categories, both in terms of their boundaries and the level of detail (Annex 3B-C). This reflects different approaches, and comparability may be increased by harmonization or recalculation using different categorization and estimation schemes. However, there are also persistent difficulties due e.g. to the overlap of source categories and to inherent unavailability of some data.

The emission factors used as basis of most calculations are crude, often based on few samples not representative of fluxes or at least not generalizable, and sometimes borrowed. Also the standardized toolkit (UNEP 2005) for POPs emission estimation involves partly very uncertain assumptions, although databased as far as possible. In addition, many data on activity rates and profiles are scarce and unreliable (Wenborn et al. 1999). However, for some processes, increasingly representative and reliable data have been published (Annex 3A).

Some emission estimates are for old installations and poorly comparable. More importance is to be attached to cumulative emissions over time and, as part of them, secondary emissions from earlier fluxes and pools, e.g. in soils, solids and products and wastes (see e.g. Hansen 2000, Hansen and Hansen 2003). The inventory by Bergqvist et al. (2005) for Sweden includes estimates of cumulative emissions (Table 9). However, these do not cover all known important fluxes. The resultant total cumulated loads of PCDD/Fs (of the order of magnitude of 5 kg TEqs based on median values mainly in wastes) may be exceeded also in Sweden by the cumulative emissions of PCDD/Fs in products such as PCP wood preservatives and phenoxyphenol herbicides used, also in Sweden (cf. Annexes 3, 4).

The uncertainties in these inventories were critically evaluated by Quass and Fermann (1997, cf. Quass et al. 2000, 2004a) who preferred

Table 9. Estimates of present and cumulative emissions of total toxic PCDD/Fs (WHO-TEq_{DF}) and ΣPCBs to various environmental compartments or matrices in Sweden from main combined source categories (extended from Bergqvist et al. 2005, cf. Annexes 3, 4). Figures have been rounded to one signifying digit, and main emissions are highlighted. Note the paucity of estimates for many sources and compartments, the importance of waste materials, and the possible overlap of compartments.

Source	Air		Water/sediment		Soil		Waste/landfill		Products	
PCDD/Fs	g TEq a^{-1}	kg TEq cumulat	g TEq a^{-1}	kg TEq cumulative	g TEq a^{-1}	kg TEq cumulative	g TEq a^{-1}	kg TEq cumulative	g TEq a^{-1}	kg TEq cumulative
Chem ind.	<	?	2	**0.3-2**	?	?	0.3	0.7-2	?	c. 10?a
Combust.	**4-70**	1	?	?	?	?	**200-300**	0.2-6	?	?
Forest ind.	1	?	<	0.1-0.8	?	**2-50**	**0.04-60**	**>100 ?b**	<0.9	**>100?b**
Metal ind.	**10-20**	1-4	?	?	?	?	**3-100**	0.1-6	?	?
Other	?	?	0.05-0.6	?	?	?	**0.2-10**	<0.2	?	?
	Air		Water/sediment		Soil		Waste/landfill		Products	
PCBs	g PCB a^{-1}	kg cum	g PCB a^{-1}	kg cumul	g PCB a^{-1}	kg cumul	g PCB a^{-1}	kg cumul	g PCB a^{-1}	kg cumul
Chem ind.	0.001	?	0.3	?	?	?	1	?	-	?
Combust.	**400-1000**	?	?	?	?	?	**4000-10000**	**50-90**	-	?
Forest ind.	?	?	?	?	?	?	?	?	-	?
Metal ind.	?	?	?	?	?	?	?	?	-	?
Other	?	?	9	?	?	?	**6000-20000**	**200-400**	-	?

Explanations: *[a]Including PCDD/F-containing chlorinated chemical products, also imported, such as chlorinated phenoxyphenyl herbicides, mainly 2,4,5-T (c. 3 kg TCDD in Finland, Räisänen & Salkinoja-Salonen 1983); [b]Adopting the estimated cumulative emission from the PCP mixture used for wood preservation (200 kg, cf. Gunnarsson & al. 2005) and the estimated distribution of the corresponding Finnish source of 20 kg I-TEq (Vartiainen & al. 1998).*

emission and activity rate data for various sectors to national reported estimates. Particular uncertainties were continuously identified in emissions from fires; those associated with residential wood and coal burning and medical waste incineration were diminished. Uncertainties were stressed and detailed also by Wenbourn et al. (1999) for emissions to land and water. Bergqvist et al. (2005) evaluated uncertainties and reduced them by additional targeted measurements, but ended up in wide ranges of estimates or inestimable quantities in many cases (Table 9). Yet, misleading certainty and precision prevails; for instance, some official emission estimates give figures with 3-4 signifying numbers, while many estimates for ranges vary by a factor of hundreds (Annex 3, Table 3A1). Even when based on measurements, some emission factor estimates have varied by orders of magnitude, e.g. for uncontrolled waste burning (Gullett et al. 2001).

Great deficiencies and discrepancies are notable in estimates, e.g. those for UNEP by the Baltic Sea countries (Fiedler 1999). Apparent differences and changes in emissions may thus reflect real developments, revised estimation procedures, or both (cf. Annex 3C).

Many inventories cover emissions to air but not to *land and water* (see review by Pacyna et al. 2003). The inventory for EU by Wenborn et al. (1999) produced estimates for emissions to land but much less to water. Also the former were in many cases highly uncertain (e.g. for pesticides and landfills), but seemed to far exceed those to air (cf. Dyke et al. 1997).

The estimated PCDD/F emissions to air from Europe have *decreased* from c. 25 kg I-TEq in 1970 to c. 8 kg I-TEq in 1995 (Breivik et al. 2004). Some estimates have been obtained for dioxin fluxes in river Kymijoki to the Baltic (Verta et al. 2003). Bergqvist et al. (2005) produced databased estimates of Swedish emissions covering all compartments and both present and cumulative emissions for total WHO-TEq_{DF} and ΣPCBs. These estimates illustrate the generally low level of knowledge of emissions (Table 9, Annex 4C).

Most inventories do not address *secondary* sources e.g. from waste disposal, stockpiles or dispersed sources, e.g. in herbicides or textiles (McLachlan et al. 1996). Estimates of some sources have been made in Swedish surveys (de Wit et al. 1990, 1992, de Wit and Strandell 1999, Bergqvist et al. 2005), by Wenborn et al. (1999), and in EU risk reduction strategy for dioxins in PeCP (Anon. 2000a). However, secondary emissions cannot be derived from the fluxes and pools in wastes (Table 9). Generalization from cases to the whole Baltic is also difficult.

Emission data for former *Soviet states* and Poland are scarce and uncertain (Lassen et al. 2002a-d, 2003, Jensen 2003; Kakareka 2002; Holoubek et al. 2003a, Anon. 2004). Geographically specified estimates of PCDD/F emissions have been produced by MSC-E (see Annex 4), but focus on emissions to air. It seems that Poland is presently the major emitter country of dioxins to the Baltic, as Polish emissions readily enter the sea unlike most of those from Russian Federation and Germany. The rough measure of Polish emissions to the sea based on catchment shares was greater than from the other countries combined. However, uncertainties of emissions hamper comparisons between countries. Quass et al. (2004b) provided preliminary data indicating that air emissions from a Polish iron ore sintering plant and also some zinc processing plants were relatively low, due to improvements made. Earlier estimates may thus have been partly exaggerated.

The information on PCDD/F emissions and related uncertainties may be summarized:

- The estimates are variable, contradictory and uncertain, due to both emission factors and activity data; uncertainties (100-fold ranges) need to be pointed out for realism
- Some estimates have greatly exceeded measured emissions; on the other hand, many potentially significant sources have been subject to little estimation and measurement
- Quantitative transport and fate modelling are fundamentally constrained
- The common focus on air gives a faulty picture of emissions
- The inventories usually are limited to PCDD/Fs and WHO-TEq$_{DF}$
- Temporal development of emissions are difficult to elucidate
- Diffuse, secondary and transient emissions become more important in relative terms
- Some major sources can be identified to direct management measures.

3.2.2 Dioxin-like PCBs

Most assessments of PCB emissions do not specify dlPCBs but address total PCBs (e.g., Breivik et al. 2002a, Annex 3C) or the 'indicator' PCBs occurring at highest levels, including usually only CBs 105 and 118. Also the composition of the Aroclor and particularly Clophen mixtures are reported mainly for isomer profiles only, not for individual congeners, and usually giving only concentration ranges, not more exact analytical data (cf. Annex 3C).

PCBs were *produced* mainly in US, several Western European countries and the Soviet Union until 1980's (vf. Annex 3C). Data for imports and exports are available only for OECD countries and for the period 1973-1980. This complicates the assessment of fluxes of PCBs and still more of dlPCBs in Baltic Sea countries. In general, Soviet and some Czech PCBs were used in non-OECD Baltic Sea European countries (cf. Breivik et al. 2002a), and mixtures from OECD producers in Western Baltic Sea countries.

DlPCBs as other PCBs in the environment and also in the Baltic originated largely from the Aroclor, Clophen and similar Eastern European brands used mainly in electric appliances but also for many other purposes. There are data suggesting that primary sources contribute significantly to present emissions, along with those from PCBs accumulated in the environment (Robson and Harrad 2004). In addition, dlPCBs originate from other products including pigments (Litten et al. 2002) as well as combustion sources. The relative importance of such non-Aroclor and non-product sources is growing along with the progress in global and regional phase-out and stockpiles management for PCBs in Aroclors.

DlPCBs are *emitted* from PCB products in use stages, stockpiles or waste management. Closed uses, small capacitor uses, nominally closed uses and open uses are differentiated (Breivik et al. 2002b). The bulk of the PCBs were used in closed systems in large transformators, but considerable emissions have been caused also from the other uses, including small capacitors and open systems, due to higher emissions factors (cf. Annex 3C). Much of the PCBs produced lie in old disposal sites, as wastes of high PCB content are already being diverted to hazardous waste treatment, also in new EU member states and Russian Federation. Subsequent emissions will depend crucially on the disposal technology.

Breivik et al. (2002a) calculated that the global emissions of 22 *main PCBs* (not including dlPCBs) decreased from nearly 400 t $\Sigma PCBs_{22}$ a^{-1} in 1970 to the present level of c. 20 t a^{-1}; these figures include an order-of-magnitude uncertainty (see also Takasuga et al. 2005). Breivik et al. (2002a) specifically gave a cumulative emission figure of 220 t $PCBs_7$ for the Baltic Sea bordering countries, excluding Russian Federation (see also Wodarg

et al. 2004). The Russian emissions to the Baltic (catchment) may roughly equal those from Germany not entering the Baltic; in addition, PCBs enter the sea in long-range transport mainly from Central Europe. It is difficult to estimate how much of the cumulative emissions are retained in wastes and materials and have not yet entered or will never enter the ambient environment.

CB 126 mainly originated in higher chlorinated Aroclors and corresponding Clophen brands (Annex 3C). Of 1-*ortho* PCBs given TEF values, particularly CB 118 was present. The relative importance of the congeners changes: e.g., CB 169 will be increasingly important in food-chains due to its persistence (Jonsson et al. 2003b, cf. 3.3).

Using measurements of congener contents, estimates of ΣPCBs emissions and assumptions of the share of the various PCB brands, rough approximations of emissions of dlPCBs from products can be made (Annex 3C). It can be estimated that of the mid-scenario historical emission of c. 300 t $PCBs_{22}$ from Baltic Sea countries (cf. above), the same fraction 4 % as globally (Breivik et al. 2002b) is comprised of CB 118, i.e. c. 10 t (cf. Table 10). However, the emission may be one order of magnitude higher or lower (Breivik et al. 2002b). These estimates give no indication of the time course of emissions and how much dlPCBs already has entered and might enter the Baltic. The estimates also omit the emissions from more remote areas, but include PCBs that do not affect the sea due to destruction or retention in the catchment. It is assumed that a negligible part of the emissions estimated for Soviet Union and Germany (particularly Western) have entered or will enter the Baltic Sea.

For *non-product sources* scarce and conflicting information is available (cf. Annex 3C). Luthardt et al. (2002) found that dlPCBs contributed c. 16 % to total $WHO\text{-}TEq_{DFP}$ of the air emissions from various thermal sources. It seems possible that a major flux of dioxin toxicity originates from incineration also to the Baltic, not only as PCDD/Fs but also dlPCBs, partly as incineration standard is not high in some countries including former socialist countries.

The information on emissions of dlPCBs can be summarized as follows:

- Cumulative emissions of PCBs seem to have been largest from Sweden, Finland and Denmark; estimates for some areas are however still very crude
- Estimates of air emissions match in some cases rather well air and deposition data for indicator PCBs, but waterborne emissions are very uncertain
- In addition to emissions that have already entered the Baltic, lagged emissions will take place from stockpiles and from the catchment, including sediments
- The contents of dlPCBs in products and in emissions is variable, and emission estimates for key dioxin-like PCBs and $WHO\text{-}TEq_P$ are crude

3.2.3 Other dioxin-like compounds

PBDD/Fs may be present or formed in material containing brominated flame-retardants. Additional processing often increases bromodioxins (IPCS 1998, Annex 1). In PBDEs, PeBDFs have been measured (Hanari et al. 2005), in DBDE e.g. 1-PeBDF and 4-HxBDF, and in bromophenols TBDF. Sometimes no PBDD/Fs were found (e.g., Imai et al. 2003). PBDD/Fs and especially mixed PBCDD/Fs are formed, as are PCDD/Fs, from brominated compounds in thermal processes. These may include burning and pyrolysis in fires, heating in industry, and incineration (Annex 4).

Emissions of PBDD/Fs and PBCDD/Fs in municipal solid waste incinerators has in particular been studied. *De novo* formation takes place on fly ash (e.g., Kawamoto and Ishikawa 2005). PBDD/F and PBCDD/F emissions also originate from traffic and possibly brominated herbicides (Huwe et al. 2003).

As PBDD/F and PBB emissions have been little measured on a congener specific basis (IPCS 1998, Annex 3), emission estimates are difficult to produce. Sakai et al. (2005) tentatively calculated a mass balance for PCDD/Fs at textile factories using HBDE and DBDE, maximally causing a flux of 150 mg PBDD/Fs per batch (mainly to solid wastes). However, few congener-specific or 'PBDD-TEq' based estimates have been produced. Kawamoto and Ishikawa (2005) reported higher TEqs for PBDD/Fs and PCBDD/Fs than PCDD/Fs in boiler ash of a gasification-melting waste treatment plant. The largest yields in thermolytic reactions have been found from PBDEs and bromophenols (IPCS 1998).

Data from Baltic Sea countries are available e.g. on PBDD/F emissions from waste incineration (Söderström and Marklund 2002, Nordsieck and Mücke 2002). Because of the absence of information on PBDD/F fluxes from brominated flame-retardants, a conceivable large source, quantitative assessment of PBDD/F emissions from the Baltic Sea counties is not feasible. A tentative estimate of total PBDD/F emissions from Denmark of 0.01–20 g a^{-1} was quoted by Jensen (2003). The unit has not been given (TEFs have not been officially defined).

For many other DLCs, with the partial exception of PCNs (Lundgren et al. 2002a), the information on emissions and immissions also in the Baltic Sea system is still weaker. For instance, little is known of the emissions of dlPBBs and even dlPAHs and of PCDTs and PCTAs (Sinkkonen 2000) in the Baltic area; relevant information on sources and emissions of other DLCs are summarized in Annex 3.

3.3 Environmental transport and fate of dioxin-like compounds in the Baltic Sea

3.3.1 Fluxes, cycling and transformation

Atmospheric deposition

Measurement-based estimates of atmospheric deposition of *PCDD/Fs* in the Baltic Sea area vary greatly, from <100 g WHO-TEq_{DF} a^{-1} to >2600 g N-TEq a^{-1} for PCDFs (Annex 6). The most relevant and reliable figures (see esp. Vikelsøe et al. 2005) converge at a value of 4 pg WHO-TEq_{DF} m^{-2} d^{-1}, giving a flux of 600 g WHO-TEq_{DF} a^{-1}. The simulation model estimate for 4-PeCDF by Vulykh and Shatalov (2001) is in a similar range. So is the German median deposition at background stations, 3 pg I-TEq m^{-2} d^{-1} (BLAD 2002a). Broman et al. (1991a) and Tysklind et al. (1993) reported that 4-PeCDF makes up c. 20 % of the total TEq in air at rural Swedish coastal sites.

The time trend of PCDD/F deposition has been established from sediment cores; peak immissions occurred in 1960's-1970's (Schramm et al. 1995, Green et al. 2001). Kjeller and Rappe (1995) reconstructed a similar time trend of PCDD/F immissions from Baltic sediments and Jonsson (2000) for PCBs (cf. 3.4.1, Annex 6A). Czub and McLachlan (2004) likewise estimated by a food-chain accumulation model that the levels of the marker PCB 153 in Baltic herring fish peaked in c. 1970.

Estimates of atmospheric deposition of PCDD/Fs may vary by 30 % and those of PCBs by 50 % depending already on the sampling procedure, i.e. rinsing of the collection funnel (Raccanelli et al. 2002). Additional variation and uncertainty in data are caused at other stages and by other factors.

Of *PCBs*, measurements have mainly been made of total and non-dioxin-like PCBs in bulk deposition in the Baltic Sea catchment. Of dlPCBs usually only CB 118 has been sporadically measured (e.g., Agrell et al. 2002, Annex 6A).

Engwall et al. (1999) estimated from unpublished data the background Swedish atmospheric deposition flux of dlPCBs at 0.04 ng WHO-TEq_P m^{-2}, i.e. 2 % of the flux of WHO-TEq_{DF} based on data by Broman et al. (1991a), and amounting to 15 g WHO-TEq_P a^{-1} on Baltic Sea surface. However, the following factors have to be borne in mind in evaluating these data:

- All data are subject to errors, including systematic, from sampling and analysis
- The dlPCB data cited has not been published and thus cannot be properly appraised
- The PCDD/F data came from urban and background areas, that on dlPCBs was rural
- The dlPCB data probably originated from a later period with lower deposition
- The air flux may not reflect the relation between PCDD/F and dlPCBs in all inputs.

Runoff influx

Many runoff fluxes involve long lags, e.g. in lake sediments, before they reach the sea (cf. Suzuki et al. 2000). Dioxin loading on the sea thus depends on the catchment properties and processes, also those indirectly affecting DLCs, like other POPs, especially through organic carbon and hydrological cycles (see e.g. discussion by Breivik and Wania 2002).

Except for river Kymijoki contaminated especially by PCDFs, estimated to cause a flux of c. 40 g WHO-TEq_{DF} a^{-1} (Verta et al. 2003, accepted 2005), and below-detection limit results from River Wisla (Anon. 2004), no data have been found on dioxin runoff to the Baltic.

As a crude long-term estimate of runoff influx based on equilibrium between the catchment and the sea, it may be assumed that c. 50 % of the PCDD/Fs emissions from and deposition on the catchment will enter the sea. The dynamics for these inputs differ from those to the sea directly.

The deposition flux to the Baltic and on its catchment is less than some decades ago, but the flux from the catchment probably is not reduced to similar degree. Thus, the catchment may drain more dioxins than it now receives, postponing the improvement in DLC loading on the sea that ultimately will be caused.

Wastewater discharges

No data have been found on dioxins in wastewater discharges to the Baltic. Bergqvist et al. (2005) reported c. 0.6, 0.1, 0.02 and 0.005 pg WHO-TEq_{DFP} l^{-1} in leachate runoff from a mixed waste landfill, PVC-Cu cable incineration wastes, a PCB ink waste landfill and a HCl factory sludge landfill, respectively, in Swedish Baltic Sea catchment (cf. Annex 6A).

As the purification of dioxins in wastewater and air emissions is intensified, more of the PCDD/Fs formed will be included in solid and semi-solid wastes and directed to sludge application areas, landfills and other disposal sites (including those for ash), and also to recycling. Secondary emissions are then caused to the sea, unless again arrested or destroyed. To an extent there is thus a choice between having dioxins on land or in the sea. Dioxins in sludge are not regulated at EU level; data on these fluxes are scanty and uncertain (Dudzinska and Czerwinski 2003, cf. Eduljee and Gair 1996).

Data on PCDD/Fs in municipal sewage sludge at Swedish purification plants have ranged 1-100 (median c. 20) pg N-TEq g^{-1} dw (Bergqvist et al. 2005, cf. Engwall et al. 1999). The data of Matscheko et al. (2002) on Swedish application areas suggests little loading in sludge, while Vikelsøe (2002) found up to 30 pg TEq_{DF} g^{-1} dw in high-application fields in Denmark. Based on a level of 50 pg WHO-TEq_{DF} g^{-1} dw (Dudzinska and Czerwinski 2003), it was tentatively estimated that 20 g WHO-TEq_{DF} a^{-1} was accumulated in sludge in Poland in 2000, i.e. 1 % of total (airborne) emissions.

As most of the estimated emissions seem to be to air, the total flux in runoff to rivers is likely to exceed that in wastewaters.

Emissions in industrial sewage discharges are highly source-specific. Many are directed to air and enter wastewater treatment only to a minor extent (see Bergqvist et al. 2005). However, the share of the dioxin fluxes in solids and thus later even to runoff may increase (see above).

Seawater concentrations and fluxes

Measurements of *PCDD/Fs* in Baltic surface water have been made (Broman et al. 1992a) but not in the incoming water in SOU. For dlPCBs data mainly on CB 118 only exists in the waters near SOU, and suggest 2- to 5-fold decrease in levels during 1990's; however, seasonal variations preclude definite conclusions of trends (Wodarg et al. 2004). Axelman et al. (2001) reported higher values but not where the samples were taken.

High levels of *dlPCBs* have been reported in biota in Skagerrak and North Sea (de Boer et al. 1993, cf. below) and may reflect high levels in the water phase. The influxes may be assumed to be small in comparison with the efflux due to the positive water balance (Wania et al. 2001). In any case, it seems evident that these budget terms are smaller than those associated with sediments.

Pacyna et al. (2002) estimated by the POPCYCLING-Baltic model that the levels of CB 118 (and CB 153) in the seawater and sediments were over-predicted by a factor of 4 using the high emission scenario, but under-predicted by a factor of 4 using default emissions. This is reasonable accuracy, considering the many sources of uncertainty involved, and that only direct emissions were included. However, subsequent resuspension and bioaccumulation present obstacles to prediction (see below).

Evaporation and vapour absorption

Volatility is low for higher chlorinated PCDD/Fs but may be appreciable for other DLCs. The descriptor of volatility, K_{oa}, also depends on temperature (Pekar et al. 1999, cf. Annex 4). Rates and yields in Baltic Sea conditions are difficult to predict. The model calculations of Bruhn and McLachlan (2002) indicated that it would be impossible to ascertain the volatilization of PCBs from the Baltic with 95 % probability. However, this likelihood depends on the conditions.

Diffusive vapour exchange has been concluded to control the levels of many POPs in the (Laurentian) Great Lakes (Mackay and Bentzen 1997), at times causing net efflux to

atmosphere. No measurements have been made in the Baltic (cf. Wania et al. 1998, 2001, see also Jonsson 2000). Based on theoretical models, vapour absorption was calculated to dominate the influx of PCBs to the Baltic (Axelman et al. 2001); others have estimated it to be smaller (Wania et al. 2001). This influx may be important for low-chlorinated PCBs and some other DLCs (Wania et al. 1998).

Photochemical transformation and decay

Photolysis has been found to be generally lower for *PCDDs* than PCDFs (Atkinson et al. 1991, Chen et al. 2001a,c, cf. Friesen et al. 1996). It depends on light intensity, the presence of other compounds and particles, and temperature.

Kim and O'Keefe (2000) concluded that photolytic dechlorination seems insignificant for PCDD/Fs in aqueous solutions (cf. Isosaari 2004, Annex 4B, 10).

Data on photolysis for *dlPCBs* are difficult to extrapolate to natural conditions. In aqueous suspension, photodegradation was slowest for 0-*ortho* congeners but increased with increasing degree of *ortho* chlorosubstitution (De Felip et al. 1996). The formation of other dlPCBs slows down the decrease in TEq and may even increase it (Miao et al. 1999). PCBs produce OH-derivatives (Tysklind et al. 1993, Rayne et al. 2002).

Sorption and desorption on particles and dissolved substances

DLCs are retained on organic matter and carbon, most of it dissolved (cf. Annex 5). DLCs are mainly settled to sediments if entering the aquatic system in sorbed state (cf. other partitioning processes, Friesen et al. 1995). Dissolved organic matter enhances the solubility of PCDD/Fs (Kim and Lee 2002).

There are indications that the Bothnian Bay is dominated by bacterial production that may play an important role in the accumulation of dioxins and affect their levels up to fish (Broman et al. 1996, Wallberg et al. 1997, 2001). The latter authors pointed out that in present risk assessments such pathways are not considered, likely leading to underestimated contamination at higher trophic levels.

The association of DLCs with organic carbon is indicated by the lower variation in their concentrations in Baltic sediments when normalized to carbon weight (Axelman et al. 2001). The classical paradigm of adsorption where K_{oc} only depends on K_{ow} (e.g., Sabljic 2001) and on TOC or biogenetic carbon has been shown to be inadequate for PCDD/Fs, PCBs and especially PAHs in Baltic sediments (e.g., Bucheli and Gustafsson 2003, cf. Annex 6A). Adsorption is constrained also by soot carbon from combustion, present in small particles that represent 2-20 % of TOC in the Baltic. The omission of soot carbon may lead to estimates of adsorption that are too low by two orders of magnitude for PCDFs and one order of magnitude for PCDDs and dlPCBs (Gustafsson et al. 2003a, Bärring et al. 2002, Annex 4B and 6C). The application of the POPCYCLING-Baltic model (Wania et al. 2000) to 4-PeCDF accounting for soot carbon gave a better match between simulated and measured levels (Mattila et al. in preparation). However, the high recycling rate of sediment PCBs compared to PAHs suggests that the latter are more strongly associated with soot carbon (Axelman et al. 2001).

Sedimentation, sediment mixing and resuspension

Estimates of sedimentation flux of *PCDD/Fs* in the Baltic by similar sediment traps at different locations and depths vary greatly (Broman et al. 1989, 1991b, Näf et al. 1992, Engwall et al. 1997b, cf. Isosaari et al. 2002c, Annex 6C). Such differences are caused by sampling, including its timing (Jonsson 2000), and by assumptions regarding sediment-related processes. While these estimates may account for resuspension from traps, they may miss other remobilization from sediments. As to sedimentation of OC (e.g., Broman et al. 1989), it has been calculated to be much higher than in original estimates in the Baltic (Sandberg 2004, p. 15).

Jonsson (2000) found that sedimentation in largely laminated Baltic Proper sediments was 4-fold greater than in Gulf of Bothnia where bioturbation dominated. Broman et al. (1991b) reported greater sedimentation of I-TEqs in Southern than in other Baltic areas. Emissions from pulp and paper industries have transported PCDD/Fs far into the sea due to the low settling matter in their discharges (Jonsson et al. 1993). On the other hand, it was calculated that most PCDD/Fs from river Kymijoki are retained in the estuary (Verta et al. 2004), and can take decades to reach open sea sediments (cf. Isosaari et al. 2002c). Kjeller and Rappe (1995) estimated a

mean residence time of 2 a for airborne PCDD/Fs in Baltic water before sedimentation, twice that for PCBs calculated by Jonsson (2000).

No direct measurement data on sedimentation of *dlPCBs* in the Baltic have been found. Estimates may be produced from data on indicator PCBs and ΣPCBs (cf. Table 10, Annex 5). Jonsson (2000) calculated a sediment deposition for $\Sigma PCBs_7$ of 15-20 ng km^{-2} a^{-1} (900 kg a^{-1} in the whole Baltic). These estimates involve great uncertainties that may underlie observable differences (e.g., Wania et al. 2001, Strandberg et al. 1998e, cf. 3.3.2). This is a key obstacle in assessment of the cycling of DLCs, as the largest pools and fluxes are associated with sediments. Some of the uncertainties are related to forms and basic assumptions of models. For instance, deBruyn and Gobas (2004) have shown that also the degradation of organic matter in sediment diagenesis may increase remobilization of PCBs manifold especially in shallow systems heavily loaded by organic matter, and may explain discrepancies between observations and predictions of equilibrium models.

For *resuspension* of PCBs rates of around 75 % have been assumed in Baltic basins (Wania et al. 2001), and Axelman et al. (2001) obtained an estimate of >90 %. Heiskanen and Leppänen (1995) estimated that in coastal Gulf of Finland c. 20 % of total C sedimentation was from resuspension; Kankaanpää et al. (1997) gave higher values (cf. Blomqvist and Heiskanen 2001, Annex 5). Jonsson (2000) found ΣPCBs are not very strongly correlated with TOC even in surface sediments.

Physical mixing may be pronounced in shallow estuaries. PCDD/F distribution deep in the sediment, resulting in peaks at depths of >70 cm, can also be due to excessive sedimentation in eutrophic shallow estuaries (Frignani et al. 2001). In offshore areas, mixing depths of generally <20 cm have been reported; in some areas less, especially in laminated sediments (Jonsson 2000).

Bioturbation reaches a depth of c. 10-20 cm on soft sedimentation bottoms (Jonsson et al. 1993, Jonsson et al. 2003b). In other bottoms mixing seems limited, as seen in distinct vertical sediment gradients. Anoxia in deep-water reduces benthic fauna and thus remobilization.

The *biotic remobilization* of PCBs by blue mussel can be considerable (Gilek et al. 1997, Björk 1998, Björk et al. 2000). Engwall et al. (1997b) calculated that blue mussel faeces increased the sedimentation of PCDD/Fs and dlPCBs by 70 %, of Bio-TEqs by 130 %.

It has been concluded that most of the cycling of dioxins is confined to the basins where they enter (Broman et al. 1992a). However, this is dependent on the time scale. In the long term, some PCDD/Fs not degraded will transport to other basins, due to processes ultimately based on entropy laws. Also many organisms cross basin divides affecting the fluxes of, exposures to and risks from dioxins.

Bioaccumulation and biomagnification in the sea

Information on the bioaccumulation of DLCs in the Baltic has been produced in studies of their levels in food-chains (cf. 3.4.2, Annex 6B). The use of equilibrium concentrations in plankton food chain accumulation studies is fundamentally limited for high-chlorinated PCBs (Axelman et al. 1997). Food web and bioenergetics based biomagnification models have been still less validated, and their validation is difficult due also to changes induced in the systems e.g. by eutrophication and fishing (Pacyna et al. 2002). The increase in concentrations per wet weight along the food chain is partly due to greater fat content; if concentrations are compared on fat basis, much less (if at all) biomagnification may be found. There are plenty of data on lipids contents in herring and cod, but little for many other species (3.4, see also Pacyna et al. 2002). Bioaccumulation moreover varies even within one species according to region, age, season etc.

Bioaccumulation in Baltic *herring and sprat* may be summarized as follows (cf. 3.4.2):

- DLCs enter herring in their feed, i.e. mainly macrozooplankton
- DLCs accumulate in the fish mainly in fat and roe (Kiviranta et al. 2003)
- In spawning grounds, DLCs may accumulate in fry and juveniles also from sediments
- There are regional variations e.g. between Northern and Southern parts of the Baltic
- Variations are related to those in fat contents; therefore, some variability is only apparent

DLCs subsequently accumulate in *salmon*, mainly in fatty (anterior) parts. There are also regional variations in dioxin accumulation in salmon. TCDD accumulates in adipose fat and roe (Jones et al. 2001).

Bioaccumulation of PCDD/Fs in multiple species in the Baltic were studied by Rolff et al. (1993) based on a framework for relationships with the trophic level and other explaining variables. PCDD/Fs as a group (in absolute concentrations) did not biomagnify but the TEqs did, due to the congeners (e.g. 4-PeCDF) contributing most to TEqs.

Eutrophication affects the bioaccumulation of DLCs. Olsson et al. (1992b) suggested that reduction in the amount of nutrients will lower the dilution media for DLCs, also of concern for seals. Rappe (1993) similarly hypothesized that the higher PCDD/F levels in herring from Bothnian Bay than Baltic Proper were due to the larger biomass in the Baltic Proper diluting PCDD/Fs. However, eutrophication has other effects on DLC cycling depending e.g. on species composition; it may also impair important fish stocks. Conversely, also other factors than eutrophication influence food chains and bioaccumulation, such as mild winters that have increased herring stocks, and fisheries which may contribute e.g. to the small sizes in some herring stocks (ICES 2005c).

Biomagnification and elimination of PCDD/F in fish consumers has been studied in human subjects mainly in high occupational exposure (cf. 3.5.2, Annex 7). Dietary accumulation factors of c. 10-30 from fish to herring gulls have been reported for TCDD (Van den Berg et al. 1987).

For *4-PeCDF* biomagnification factors of c. 3-4 from plankton to planktivore and from planktivorous to predatory fish such as rainbow trout have been obtained (cf. Annex 4). In the Baltic, biomagnification of TCDD, PeCDD, 4-PeCDF and TEqs has been found to herring and to cod liver (Rolff et al. 1993). In some models of food web biomagnification between estuarine sediments and fish, the overestimation of PeCDF levels has been particularly great (Carrer et al. 2000).

The biomagnification of *dlPCBs* in many aquatic systems is greater than that of PCDD/Fs (e.g., Cook et al. 2003) and of total PCBs (e.g. Lundgren et al. 2004). Bioaccumulation from sediments may be particularly strong for 1-*ortho* PCBs (Naito et al. 2003). Biomagnification of dlPCBs in the Baltic has been observed in some food chains (e.g., Strandberg et al. 1998c,d, Falandysz et al. 2002a,b, see below and Annex 6). The representativeness and specification e.g. in terms of tissues and units of these data present problems.

Strandberg et al. (1998d) found that biomagnification was generally higher in Bothnian Sea than Bothnian Bay in *Mysis*-herring food chains (cf. Lundgren 2003). In Gulf of Bothnia, biota-sediment accumulation factors (BSAF) of c. 2 were found for CB 126 to the amphipod *Monoporeia affinis* (Lundgren et al. 2004). Biomagnification factors (BMFs) were higher for 1-*ortho* than for 0-*ortho* congeners from *Monoporeia* to four-horned sculpins. In the Gulf of Bothnia, BMF was c. 5 for both congener groups from *Monoporeia* to isopod *Saduria entomon* that had highest TEq_P levels. This may be related to its high fat contents in the wild (Sapota 1996, 1997). BMFs from isopod to sculpin were low; biomagnification to cod, a predator on *Saduria*, was not studied. Biomagnification of dlPCBs in some fish-eating birds has been found (cf. 3.4.2, Annex 6).

In Southern Baltic, lipid-based WHO-TEq_P bioaccumulated c. 10-fold to herring, 50-fold to stickleback, and 250-fold to black cormorant (Falandysz et al. 2000b, cf. 3.4.2). Levels of dlPCBs in cormorant muscle and liver were much higher (on lipid basis) than in harbour porpoise blubber and in herring (Falandysz et al. 2002b), suggesting lower metabolization in cormorant. DlPCBs are metabolized in harbour porpoise more easily than many non-dl PCBs (see Karlson et al. 2000).

Pelagic Baltic ciliates feeding on bacteria have been found to play a key role in transferring CB 153 to the food chain (Wallberg et al. 2001). This is important as many assessments assume that DLCs are transferred only from phytoplankton to zooplankton and fish (cf. Broman et al. 1992b), and may be generalizable to dlPCBs of similar chlorination degree.

Of *other DLCs*, bioaccumulation of PCNs has been found in the Baltic (Falandysz et al. 1996a,b, 1997a,b, 1998f, 2000b) generally in the same species and tissues as with dlPCBs. The (tentatively) dioxin-like hexa- and heptachlorinated CN 66/67, CN 69 and CN73 had variable but modest biomagnification in Gulf of Bothnia (Lundgren et al. 2002a). Accumulation is pronounced e.g. in harbour porpoise liver (Ishaq et al. 2000, cf. Annex 6).

Transformation and degradation in the sea

Degradation of specific PCDD/Fs in aquatic environments has been little studied directly. Empirical half-lives of c. 100 d have been obtained for PeCDFs in active anaerobic sediment microcosms (Adriaens et al. 1995, cf. Annex 10). These high rates of dechlorination do not seem applicable to Baltic Sea sediments in most accumulation bottoms (cf. Barabas et al. 2004, Gaus et al. 2002).

Sinkkonen and Paasivirta (2000) cited interpretations by Kjeller and Rappe (1995) that highly chlorinated 2378-PCDD/Fs are essentially non-degradable in Baltic sediments, and published estimates of the half-life of PCDD/Fs, ranging c. 200-700 d in water and 6-10 a in sediment, based e.g. on Mackay et al. (1992). These estimates do not specify congeners, and seem generally highly uncertain. The accumulation of PCDD/Fs e.g. from OCDD (Fu et al. 1999, cf. Fueno et al. 2002) and even from non-2378-chlorinated HpCDDs (Fu et al. 2005) attenuates the reduction of dioxin toxicity.

Some *abiotic* decay of PCDD/F may take place even without light energy, e.g. in sediments. Studies of sediment depth profiles may not detect degradation that occurs due to reductive agents (Adriaens et al. 1996). There is evidence that inorganic compounds including sediment DOC can contribute to degradation of PCDD/Fs (Fu et al. 1999). Such decay processes may not be sufficient to balance even the ongoing influxes of PCDD/Fs to the Baltic or to reduce dioxin toxicity, but may have importance for the longer-term fate of these compounds.

For PCBs, a half-life of 50 a in water and infinite in sediments was used by Wania et al. (2000, 2001) as a default in the POPCYCLING-Baltic model, with no differences between congeners. These are very crude and overly pessimistic estimates. In congener-specific studies in coastal sediments, reductive dechlorination has been found to selectively remove CB 77 and CB 126 with half-lives of c. 8 a (Brown and Wagner 1990, cf. Tiedje et al. 1993/94) which was noted as reduced dioxin toxicity of sediments (Mousa et al. 1998). However, in some of the microbial dechlorination and transformation processes, toxic *ortho* PCBs may be formed.

Weakly dioxin-like PAHs are present in the Baltic (Broman et al. 1990a, 1991b, Blankenship et al. 2000). Many of them are however readily metabolized (Hofelt et al. 2001).

Dioxin cycling and transformation in systems linked with the Baltic Sea

Fluxes of DLCs are directed from Baltic fish to production animals consuming such fish or products based on it. The fluxes may be divided and characterized as follows.

- Some of the flux to the terrestrial system is in unprocessed fish. Feeding Baltic fish to pigs or, still more often, to minks has been practiced e.g. in Finland.
- Much of the industrially processed Baltic fish (especially sprat) has been used as feeding-stuff either as such or more commonly processed to fish meal or oil. Some of these fluxes have been directed to animals in Baltic Sea countries, but also to other EU countries.
- Farmed rainbow trout obtain DLCs in their feed, including both Baltic Sea fish and other ingredients.
- In the catchment, DLCs cycle in solids and products including e.g. PCP-treated wood (Hansen 2000 and Hansen and Hansen 2003). As pointed out e.g. by Duarte-Davidson et al. (1997), the soil (and sediment) burden will take much longer to decline than the direct emissions to air
- PCDD/Fs are transformed and degraded before they enter the sea, and after they are removed from it e.g. in fish. Dehalogenation takes place also in technological systems, depending e.g. on temperature, availability of catalysts and alkalinity of the system (Weber et al. 2002a).

3.3.2 Budgets of PCDD/Fs and PCBs in the Baltic Sea system

PCDD/Fs

From estimates of various budget terms (see above and Annex 6C), tentative and partial budgets may be constructed for Baltic Sea PCDD/Fs and PCBs (Table 10), being order-of-magnitude estimates in many cases. Atmospheric deposition may be estimated with some confidence, but some other influxes, notably in runoff and resuspension from sediments, are difficult to estimate. Numerical uncertainties of budget terms are coupled with the more fundamental difficulty of interpreting terms due to DLC dynamics. The influxes and sinks that may be influenced become important for risk management.

The budgets for specific PCDD/Fs may deviate from those for WHO-TEq$_{DF}$. The relative importance of congeners varies in time (especially as some degrade or transform faster) and in space. This is evident e.g. in Gulf of Finland where most of the loading in runoff is comprised of Hx- to OCDFs, unlike other regions and also unlike the relative fluxes and pools in biota.

On the basis of the budgets it seems that sedimentation of DLCs considerably exceeds atmospheric deposition. This is pronounced for Gulf of Finland PCDD/Fs (Isosaari et al. 2002c). This suggests that other, less known influxes are important, or that the system is in disequilibrium. For PCBs there is evidence of temporal and spatial variations in the ability of sediments to act as sinks (Jonsson 2000). However, it is not known how well the sedimentation estimates reflect actual net sedimentation. Likewise, it is not known how well the flux deposited in surface sediments enters 'final burial' (see above, cf. Axelman et al. 2001 and Wania et al. 2001 for PCBs). Assuming that steady state is approached over extended periods, it would be expected that average net sedimentation would equal influxes (considering other sinks are small). Of other sinks, that due to decay is almost unknown. Part of the decomposition products of PCDD/Fs are DLCs or compounds that may otherwise be hazardous.

The estimates of the total marine pool of DLCs in sediments are highly uncertain, as indicated by those for ΣPCBs (see below). Several calculation approaches are possible and the

Table 10. Summary of estimated current budget terms for toxic PCDD/Fs, dioxin-like PCBs and total PCBs in the Baltic Sea. The values for WHO-TEq$_P$ in parentheses are based on statistical relationships with median values for ΣPCBs (for the other variables the values in parentheses are based on estimates considered of limited reliability). Figures have been rounded to one signifying digit. Note the units. Cf. Annex 6C.

Budget term	WHO-TEq$_{DF}$ (g for pools, g a^{-1} for fluxes)	WHO-TEq$_P$ (g for pools, g a$_{-1}$ fluxes)	ΣPCBs (t for pools, t a^{-1} fluxes)	CB 118 (g for pools, g a^{-1} fluxes)
Influxes				
Atmospheric deposition	400[a], 600[b], 700[c] (>3000)[d]	50[f], 20-40[z]	0.4[f,j] (0.7[t])	70000[o]
Adsorption from air	?	(40)	0.9[u]	
Runoff (river & coast)	40[g]	(10)	0.3[f] (0.2[i]) (0.05[p], >0.007[w])	
Direct discharges	<	(8)	0.2[i]	
Influx from North Sea	<[h]	<[h] (4)	0.1[i]	<[h]
Resuspension	?	(100)	2[i], 3.8[u]	
Outfluxes				
Sedimentation (net)	4000[c,q] (20000)[r]	(200)	1[x], 3[i], 7[k], 4.2[u]	
Volatilization	?	(10)	0.3[i]	
Efflux to North Sea		(6)	0.1[x], 0.2[i]	
Removal in fish	1-3[n]	(1)	0.03[i]	
Degradation (sed, water)	?	?	0 (sed), 0.02 (water)[i]	
Pools				
Mixed surface water	20[c]	(40)	0.6[u], 1[v], 1[1]	10000 (BP)[2]
Bottom water (below thermocline)	<10[n]	(<10)	<0.3[n]	
Marine sediments	>10000[m], (30000)[s]	(1000)	>1[w], 5[u], >20[v], 30[x], 200[l], 250[1]	
Fish biomass	20[n]	(10)	0.3[i], 0.05-1[3]	
Blue mussel biomass	?	(10)	0.3[3]	
Plankton biomass		(30)	0.2-1.5[3]	
Benthic algae biomass	?	(3)	0.7[3]	

References and notes: *[a]Korhonen & al. unpublished, from 13-mo measurements in Utö, Northern Baltic Proper in 2002-2003; [b]Vikelsøe & al. 2005, from 3-a measurements in Bornholm; [c]Broman & al. 1991a, Engwall & al. 1999; [d]Marklund & al. 1991 based on PCDFs; [f]From arithmetic mean of Baltic Sea/coast/river stations by Agrell & al. 2001 and 2002, cf. MSC-E 2005, and from river data and estimates by Axelman & al. 2001; [g]Verta & al. 2003; [h]Influx in seawater assumed negligible; [ii]Model assumptions or simulations by Wania & al. 2001, including diffusive water-sediment exchange, and vapor adsorption in volatilization terms; [k]Strandberg & al. 1998e; [l]Revised from Bergqvist & al. 2005; [m]Isosaari & al. 2002c, for GOF; [n]this study; [o]modeled by Pekar & al. 1999; [p]Based on Falandysz & al. 1998d, 1999b for R. Vistula and the proportion of its catchment to that of the Baltic; [q]Näf & al. 1992; [r]Engwall & al. 1997b; [s]SNV 1987, ref. Paasivirta 1990; [t]Blanz & al. 1999 for Oder only, based on 65-a accumulation period; [u]Axelman & al. 2001; [v]Model estimate by Wania & al. 2001 for top 1 cm of sediment only; [w]Blanz & al. 1999, for Oderhaff, Pomeranian Bight, Arkona Sea and Bornholm basin only; [x]Jonsson 2000; [z]This study, based on Swedish coastal surface sediment data by Bergqvist et al. 2005 and reported range of ΣPCBs deposition in the Baltic Sea area; [1]Schulz-Bull et al. 1995 for the upper 5 cm; [2]Wodarg & al. 2004; [3]Kihlström & Berglund 1978, based on separate data for main fish species, seals, blue mussel, plankton and benthic algae from various sources.*

representativeness of measurement data is not clear. The same applies to pools in the catchment, including those in wastes (see Bergqvist et al. 2005, cf. Table 9, Annexes 4 and 6), in vegetation and in soil. Little data have been found on background soil levels in the catchment and key vegetation types, for estimation of terrestrial pools and long-term fluxes. Considerable amounts of data were reported by Holoubek et al. (2000) and in the ELICC survey (Anon. 2004), but that in BLAD (2002a) seems the only internally coherent extensive dataset to date on the Baltic Sea catchment. The applicability of these data to other Baltic Sea countries is unclear, as e.g. soil and vegetation types differ.

Dioxin-like PCBs

There is little congener-specific information on fluxes of dlPCBs to and from the Baltic (e.g. Axelman et al. 2001, cf. 3.3.1, Annex 6A). Of these PCBs usually only CB 118 has been measured (Table 10). ΣPCBs and indicator PCBs however provide some surrogate basis of assessment.

A linear relationship between ΣPCBs (based on an Aroclor mix) and WHO-TEq_P in Swedish coastal surface sediments was given by Bergqvist et al. (2005, Annex 6); this gives a ΣPCBs/WHO-TEq_P ratio of c. 20000 in the predominant range around medium levels of ΣPCBs. A higher mean sediment ΣPCBs/WHO-TEq_P value of 100000 can be calculated from Sundberg et al. (2005) for a locally PCB-contaminated area that may not be representative of general conditions in the Baltic. It is not clear how well these ratios hold for ΣPCBs/WHO-TEq_P in immissions, considering congener-specific differences in fate. Using a rough estimated ΣPCBs/WHO-TEq_P ratio of 25000, the range of estimates of the atmospheric deposition of ΣPCBs on the Baltic (0.4-1 t a^{-1}, Table 10) translates to 16-40 g WHO-TEq_P a^{-1}. This is around one order of magnitude less than the medium values for atmospheric deposition of PCDD/Fs, which is consistent with the relationship in rural Japan (Ogura et al. 2001). Additional influxes of dlPCBs come especially in runoff and from air gas phase.

Estimates of the total emission may be produced from data on emissions of PCBs and dlPCBs in products (Annex 3), but their contributions to Baltic Sea immissions remain uncertain. However, it seems likely that the total immission of dlPCBs to the Baltic is considerably smaller than that of PCDD/Fs in terms of dioxin toxicity. The relative importance of dlPCBs for dioxin-like toxicity and risks is increased in the later stages in their cycling, especially biomagnification (see below).

Mackenzie et al. (2004) calculated that the standing stock of Baltic cod, sprat and herring was a sink for c. 300 kg ΣPCBs a^{-1} in late 1980s to early 1990s and that the fisheries removed as much as or more than other sinks (i.e., 30 kg ΣPCBs a^{-1}). However, the fluxes to (and from) sediments are probably much larger (cf. PCDD/Fs above). The estimate of Mackenzie et al. (2004) of the relative importance of fisheries in the budget of ΣPCBs is greater than that calculable for PCDD/F removal in catch (ca. 2 g WHO-TEq_{DF} a^{-1}, Table 10, cf. Annex 6C). The pool and removal in fish also seem small in relation to influxes from air (>200 g WHO-TEq_{DF} a^{-1}). Nevertheless, removal in fish does present a means of reducing the load in the sea (cf. 7).

3.4 Environmental levels and trends, and body burdens in Baltic non-human receptors

3.4.1 Abiotic compartments

PCDD/Fs and dlPCBs

Air and water

No congener-specific data allowing resolution of trends or quantification of fluxes have been published for PCDD/Fs in air or atmospheric deposition (cf. 3.3.1). Data exist for some air emissions in Baltic Sea countries. Congener-specific analyses have been made of PCBs in air (Agrell et al. 1999, 2001, 2002, cf. TWGIM 2004a), but not of dlPCBs (except CB 118).

Broman et al. (1991b) reported data on PCDD/Fs in Baltic Sea water (cf. Annex 6A). Seasonal variation may be considerable, e.g. in levels of CB 118 in relation also to plankton dynamics (Bruhn and McLachlan 2002). Thus, single measurements are not very representative.

Sediments

The temporal development of immissions is reflected in unmixed sediment layers, but is constrained by time resolution and by sedimentation processes, e.g. focusing of sediment

to accumulation areas after resuspension and resettling that introduce lags and spatial variations (Broman et al. 1994, Axelman et al. 2001). Resolution is at best several years, as there is some mixing in even layered sediments, and due to dating (Cook et al. 2003). However, the preservation of PCDD/F concentration peaks in sediments indicates that also recent layers may reflect immission trends (cf. Green et al. 2001). Pettersen et al. (1999) showed that also horizontal variation in sediments is great (e.g. 20 % for some PCBs, 2-fold that from chemical analysis).

PCDD/Fs and dlPCBs have been measured in Baltic sediments but not always reported in the scientific literature (cf. Annex 6A). Few sediment profiles have been dated. Differences in sampling, analytical and reporting methods reduce comparability (cf. 3.1). The lack of data on other sediment quality characteristics such as OC moreover hampers interpretation of results.

Kjeller and Rappe (1995) reported data from deep Baltic sediments on *PCDD/Fs* and some dlPCBs (Annex 6A), and Verta et al. (1999a,b, 2003) and Isosaari et al. (2002c) on Gulf of Finland sediments (Fig. 9). Levels of 10–20 pg WHO-TEq_{DF} g^{-1} dw were measured in background areas, while in Eastern Gulf of Finland higher values (up to 400 pg WHO-TEq_{DF} g^{-1} dw) were found particularly near river Kymijoki at sediment depths corresponding to c. 1962-65 (Isosaari et al. 2002c).

Little data have been reported for *dlPCBs* in Baltic sediments, except for CB 118. The levels have ranged from <0.1 (Lundgren et al. 2004) to 2-4 pg WHO-TEq_P g^{-1} dw (e.g., Koistinen et al. 1995c, 1997a; cf. Jensen 2003). The levels of 'total' PCBs have been measured more often (see esp. Jansson 2000, Axelman et al. 2001). In converting data on ΣPCBs to WHO-TEq_P the linear relationship by Bergqvist et al. (2005, see above) can be used. The following results for ΣPCBs may to some extent apply to dlPCBs:

- Increase of $\Sigma PCBs_7$ levels with time has been generally noticeable in dated cores (Jonsson 2000, cf. Nylund et al. 1992, de Wit et al. 1990). Differences in sedimentation conditions may explain this; the levels in laminated sediments have slightly decreased from 1970's to 1990 (Jonsson 2000). There are also indications of slight decreases in $\Sigma PCBs_7$ deposition over time, but temporal and regional variations and factors affecting sedimentation complicate this.
- Slightly higher levels in Southern (ca. 400 ng $\Sigma PCBs_7 g^{-1}$ C) than Northern Baltic (ca. 200 ng $\Sigma PCBs_7$ g^{-1} C) have been reported (Jonsson 2000 and references therein), while in other studies more uniform distribution has been found (Axelman et al. 2001).
- Although PCBs generally attach to OC and are normalized in C, the correlation with TOC is weak especially in deeper sediments (Jonsson 2000).

Local contamination

Dioxin contamination in the Baltic exhibits geographical variation in large and small scales, the latter being largely related to local contamination. Näf et al. (1992) found a gradient in PCDD/F levels in sediment traps around a Swedish coastal town with industrial emissions and Rappe et al. (1989) in sediments near another, but Koistinen et al. (1995c) did not in sediments around a pulp and paper mill in Gulf of Bothnia. Isosaari et al. (2000) calculated the PCDD/F emission from a VCM factory on Finnish coast to be much less than that from river Kymijoki. Elevated levels are also found in other Baltic coastal sediments (Bergqvist et al. 2005, Annex 3).

In river Kymijoki, a stretch of 3 km downstream a PCP-producing factory 100 km from the sea contains c. 4 kg WHO-TEq_{DF} i.e. 30 % of the total 15 kg WHO-TEq_{DF} upstream from Gulf of Finland (Verta et al. unpublished 2005). This total equals the estimated WHO-TEq_{DF} in whole Gulf of Finland (Isosaari et al. 2002c). Elevated dioxin levels are subsequently found also in sediments in the estuary (Isosaari et al. 2002c, Fig. 9). Relatively little 4-PeCDF is discharged, as revealed e.g. by congener patterns and low levels in sediment traps in the estuary (Korhonen et al. 2002).

Data on local contamination by PCBs have been summarized by Bergqvist et al. (2005). In most studies dlPCBs have not been in focus; ΣPCBs, indicator PCBs and some mono-*ortho* PCBs (mainly CB 118) have been reported. A notable source of PCBs including CB 126, CB 169 and CB 77 has been river Emån (Asplund et al. 1990); most of them have been intercepted in lake sediments. Contamination by PCBs has been reported also in pulp and paper mill waste deposits and sediments in Örserumsviken on Swedish Bothnian Sea coast (Sundberg et al. 2005), but other compounds had a much greater contribution to toxicity as indicated by biassays.

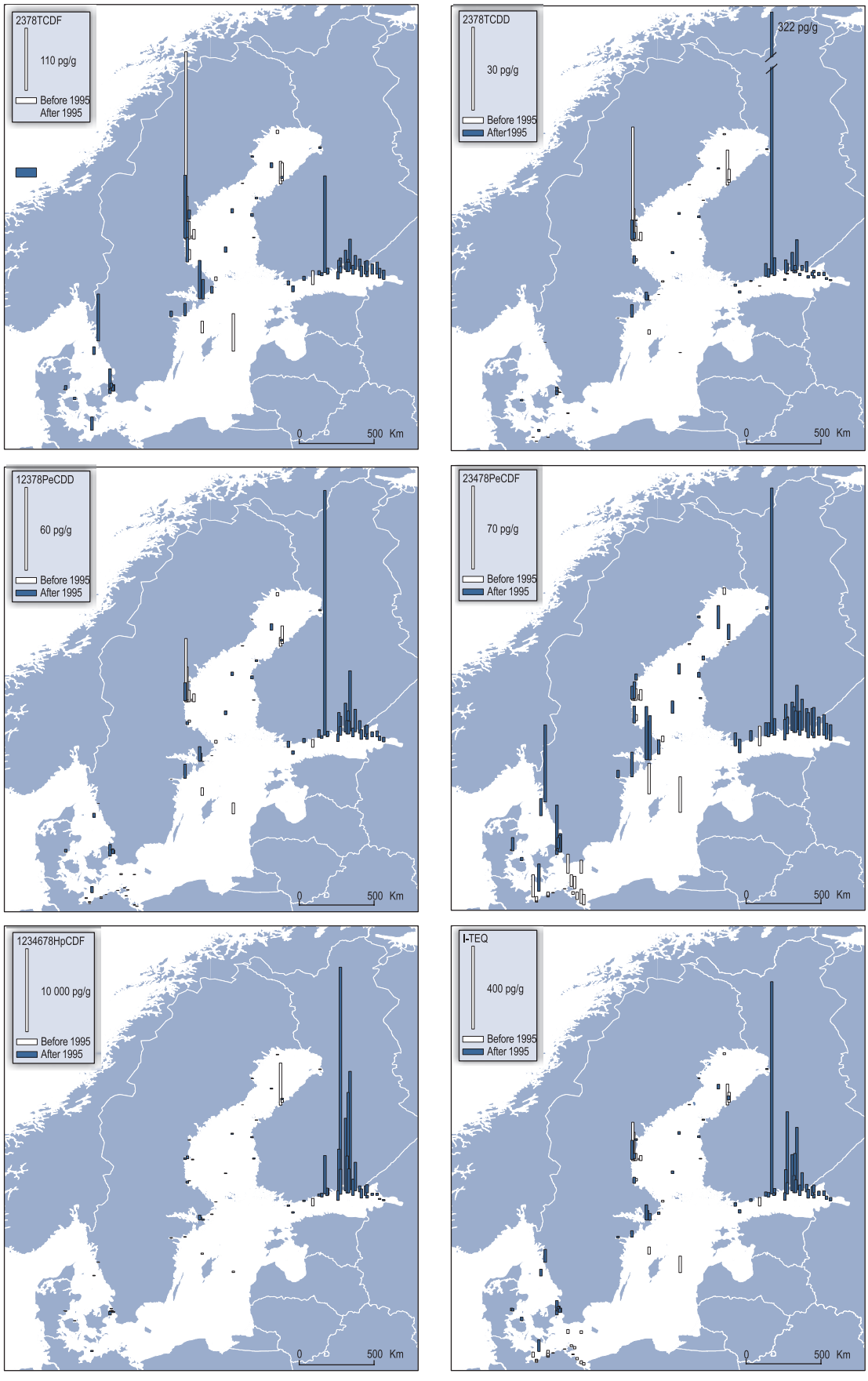


Fig. 9. Regional distribution of some key PCDD/F congeners and total PCDD/F toxicity (I-TEq) in surface layers of Baltic Sea sediments. From Verta et al., accepted, based on available data from various sources. Recent and previous data (in pg I-TEq g^{-1} dw) have been indicated. Note the high levels in some coastal sediment areas of limited extent as compared with deep-sea sediments.

Other dioxin-like compounds

Data on other DLCs in the Baltic has been summarized in Annex 6A. Mainly PCNs have been studied. Elevated levels of 4-PeBDF have been preliminarily reported in Swedish lake sediments reflecting long-range transport (Hagberg et al. 2005).

3.4.2 Biota

PCDD/Fs and dlPCBs

Herring

Many limitations and inconsistencies in sampling, analysis and reporting constraint assessment of PCDD/Fs and dlPCBs in Baltic herring, as illustrated by the Finnish data (Hallikainen et al. 2004) and largely applicable also to Swedish (e.g. SNFA 2004, 2005), German (e.g., Karl et al. 2002) and other data (e.g., Shelepchikov et al. 2005):

- Measurements from *Gulf of Bothnia* in particular have been made only during the last few years
- *Spring and fall* herring, displaying some differences, have been reported mainly starting in 2002
- *Age class* specification in earlier data has usually not been made
- *DlPCBs* were included in herring dioxin analysis only in late 1990's
- In some reports *only TEqs* have been given instead of the congener specific data
- Sometimes only *wet or lipid weight* based data have been reported.

Summary data on WHO-TEq_{DF} in Baltic herring have been reported (Tuomisto et al. 2004a, cf. de Wit et al. 1992, Annex 6B). No trends could be ascertained in these data. This contrasts with the decreased trends in the more long-term data on ΣPCBs (Bignert et al. 1998, Strandberg et al. 1998d) and also with the declining PCDD/F-TEqs in guillemot eggs (see below). The spatial, physiological and age class related variation of herring may partly explain these differences and also make comparisons of measurements difficult. Comparable data including dlPCBs and resultant total dioxin-like toxicity are very scarce (see e.g. Kiviranta et al. 2003).

Although the age (and fat content) of herring has not always been reported, it seems that the levels of ΣPCBs in herring peaked in 1969-1970 (and 1979), having been c. 4-fold lower in mid-1960's and 10-fold lower from mid-1980's (Strandberg et al. 1998b,d, cf. Annex 6B). It may be assumed that the development of dlPCB levels in herring was roughly similar. These diminishing trends were noted already by Paasivirta and Linko (1980).

Specific data on dlPCBs in herring of known properties have been published since (e.g., Kiviranta et al. 2003). It seems that the levels in herring of comparable age and from the same area have fluctuated with no clear trend. As with PCDD/Fs, this contrasts with the continuous declining trend in levels of ΣPCBs in representative populations and tissues in some consumers of Baltic herring, including guillemots (see below and Olsson et al. 2005) and humans (cf. 3.5.3).

Also recent analyses (Karl et al. 2002, Bjerselius et al. 2003, SNFA 2004, 2005, Hallikainen et al. 2004) have limited comparability due to variable or unreported sample sizes, locations and times, age, gender and length, and coverage of dlPCBs. Kiviranta et al. (2002b, 2003) could not discern a trend from early 1990's to a decade later in dlPCBs and TEq_P which suggests that declines have levelled off; it was pointed out that changes in analytical methods may have caused decreases that are only apparent. Karl and Ruoff (2004) reported a declining trend from 1996 to 2003 in German catches of spring herring, but likewise noted that this may be due to artifacts like variations in size and fat content. In addition, some of the apparent changes in TEq_P may be due to different TEFs.

Some regularities in variations of TEq levels in Baltic herring can nevertheless be noted:

- PCDD/F levels correlate positively with *age*, relatively independently of fat content (see also Roots et al. 2003). The length corresponding to 4 ppb ww WHO-TEqDF is c. 17 cm, usually of 3-year old fish. The correlation is less linear and strong in some other data, which limits the utility of age as a proxy for TEqs. If the limit for WHO-TEq were retained, even smaller fish would exceed it, and for these the age correlation is low.
- *Regionally* levels generally increase in herring of similar age toward East (Karl et al. 2002, Karl and Ruoff 2004) and North. However, WHO-TEq_{DF} correlates more with fat contents than area (Vuorinen et al. 2004), and regional variation is in general different

when expressed on a more comparable lipid and age normalized basis (cf. 5.2.4, Bignert et al. 2005)
- Dioxin levels seem to differ *seasonally* between fall and spring catches especially in the Archipelago Sea (Hallikainen et al. 2004, cf. data for PCBs by Asplund et al. 1990).
- Dioxin contents differ in different *parts* of fish; a large part of the dioxins is contained in the fat in skin (see also Aune et al. 2003, and for smoked herring Hallikainen et al. 2004).
- Bjerselius et al. (2003) reported a correlation between the *gender* of herring and WHO-TEq_{DF} (fw), females having on the average 10 % higher levels at equal age
- *DlPCBs* have contributed c. 30-50 % to total WHO-TEqs in herring, less in relative terms than in some other species (Bjerselius et al. 2002b, Hallikainen et al. 2004). Asplund et al. (1990) reported that the share of PCBs of total dioxin toxicity in herring was higher (cf. Annex 6B); this may suggest that the relative importance of PCBs has diminished. The share of CB 126 has been on the average 60-70 % of WHO-TEq_P (Ankarberg et al. 2004, Kiviranta et al. 2003).
- Some data on *ΣPCBs* (e.g. Vuorinen et al. 1998b) indicate a decline during 1980's in herring muscle PCB levels in the Gulf of Bothnia to c. 20 % of the level in 1979-80.

Summarizing, high levels of up to 30 pg WHO-TEq_{DFP} g^{-1} ww have been still reported in Baltic herring, especially old fish. Most of the dioxin toxicity has been due to PCDD/Fs, but in some cases PCBs have contributed over 50 %. The PCDD/F and dlPCB contents are correlated with fat and age.

Salmon

Data on the levels of PCDD/Fs and dlPCBs in Baltic salmon are available mainly in TEqs, not on congener basis, from SNFA (2004, 2005, cf. Bjerselius et al. 2003) and Hallikainen et al. (2004). DlPCBs have been reported for only part of the Swedish data and with imprecise catch times. No trends can be established for PCDD/Fs or dlPCBs in salmon (cf. Larsson et al. 1996).

The salmon data can be summarized as follows (cf. Annex 6B):
- In Gulf of Bothnia, 8-15 pg WHO-TEq_{DFP} g^{-1} ww was found in Swedish (2001) and 2-3 fold more in Finnish (2002) data; on lipid basis the relationship is reversed. In Gulf of Finland, 20–25 pg WHO-TEq_{DFP} g^{-1} ww were measured, in Southern Baltic Proper, c. 10 pg WHO-TEq_{DFP} g^{-1} ww
- Most dioxins are located in anterior parts
- No clear correlation with age or gender seems to exist
- In Swedish data on 1-3 year old salmon, PCBs have contributed c. half of the total WHO-TEqs (Ankarberg et al. 2004)
- Higher DLC levels in reared than natural salmon are found (Lundstedt-Enkel et al. 2002).

All in all, no clear and consistent trends are distinguishable in time or space in salmon dioxin and PCBs contents that reach high levels, ranging from c. 5 to 30 pg WHO-TEq_{DFP} g^{-1} ww.

Other fish

Increasing measurements have been made of PCDD/Fs and dlPCBs in fish species in the Baltic (cf. Annex 6B). Data from Russian Federation, Poland and the Baltic States are also increasing. However, generally the data are infrequent and their representativeness uncertain. In addition to surveys of food fish, measurements have been made in ecotoxicological studies (Falandysz et al. 2002a,b, Lundgren et al. 2003a) and in investigations of local contamination (e.g. Verta et al. 1999a,b, Isosaari et al. 2000, Korhonen et al. 2001). ΣPCBs have also been reported in many species of fish.

The data generally do not allow trend analysis, but illustrate the overall levels and variations. However, the long-term data on PCDD/Fs and dlPCBs in cod liver oil from Southern Baltic (Kannan et al. 1992 and esp. Falandysz et al. 1994d) allows some resolution of a declining trend in total dioxin toxicity. Earlier trend data on ΣPCBs in pike (Moilanen et al. 1982) have also demonstrated the decline in lipid-based concentrations from mid-70's to early 80's (cf. Jensen et al. 1977a). It may be assumed that the trends in ΣPCBs reflect, with some uncertainty, those of dlPCBs.

In sprat, levels comparable to or even higher that those in herring are found, depending e.g. on age, with increasing levels in old fish (Vuorinen et al. 2002, Hallikainen et al. 2004).

In many freshwater species, the levels in the Baltic are 2-3-fold higher than those in inland waters, except in vendace (Vuorinen et al. 2002, 2005,

Hallikainen et al. 2004). Lean fish such as perch has low WHO-TEq_{DF} levels even near industrial sources (Olsson et al., unpublished 2005), while many fatty fish have had levels comparable with herring and even salmon, those in smelt exceeding these several-fold (Koistinen et al. 2002). In eel, dlPCBs have contributed almost all of the 3-8 pg WHO-TEq_{DFP} g^{-1} ww (Bjerselius et al. 2002b, cf. Atuma et al. 1996a). Levels depend also on exposure patterns and metabolism (Lundstedt-Enkel et al. 2002).

In summary, it can be noted that

- Levels of PCDD/Fs and dlPCBs approaching or exceeding the EC (2002a) action limits have been measured in several fish species, especially if dlPCBs are included; such species include sea trout, eel, sprat, flounder, whitefish, bream, smelt, eelpout and three-spined stickleback and lamprey.
- Limited information is available on geographical, time and tissue distributions in many species; in general, dioxins are accumulated in fatty tissues, including liver and roe.

Seals and other marine mammals

The presence and distributions of DLCs and PCBs in the Baltic have been most commonly studied in seals, since 1960's as to ΣPCBs (Jensen et al. 1969). These studies have variable representativeness and specificity as to age class, sex and condition of the seal specimens, and variable coverage and specificity as to compounds analyzed (Olsson et al. 1992a). Many data are for juvenile animals that have been considered both more representative (e.g., Bignert et al. 1989) and in some respects less representative (Nyman et al. 2003) than adults in contamination description.

In Baltic **grey seals**, PCDD/Fs were measured e.g. by Bignert et al. (1989), Bergek et al. (1992) and Olsson et al. (1994). The levels in seals suffering from uterine occlusions were not higher than in other seals, unlike with PCBs (cf. 4.3.2). Olsson et al. (1994) did not find CB 118 even in highly contaminated females. As levels of dlPCBs were not reported, these data are of limited use in assessment of dioxin-like toxicity, but serve some comparative purposes (cf. 5.4, Annex 6B). Koistinen et al. (1997a) who analyzed also dlPCBs obtained WHO-TEq_{DFP} levels of 400-800 pg g^{-1} lw in the blubber of starved and c. 200 pg g^{-1} lw in normal grey seals, almost totally contributed by PCBs 118, 156 and 126. Nyman et al. (2003) found Bio-TEqs (by the CALUX assay) of 100 pg g^{-1} lw in plasma of Baltic grey seals.

In **ringed seals**, the levels of PCDD/Fs reported by Bignert et al. (1989), Bergek et al. (1992) and Olsson et al. (1994) were 3- to 10-fold higher than in grey seals. Koistinen et al. (1997a) who analyzed also 0-*ortho* and 1-*ortho* PCBs obtained levels of 300-400 pg WHO-TEq_{DFP} g^{-1} lw in blubber of normal and 500 pg WHO-TEq_{DFP} g^{-1} lw in starved animals; PCBs contributed 70-90 % of total dioxin toxicity. Accounting for dlPCBs, the TEqs were not clearly higher than in grey seals in this material. Nyman et al. (2003) reported levels of c. 100-200 pg Bio-TEqs g^{-1} lw in liver and 200 (40-400) pg Bio-TEqs g^{-1} lw in plasma of Baltic ringed seals collected from ice in Bothnian Bay in 1996-98; the Bio-TEqs were below instrumental TEqs based on analyses of PCBs in liver.

In juvenile Baltic **harbour seals** collected in 1983-87, Bignert et al. (1989) measured much lower PCDD/F levels (c. 20 pg WHO-TEq_{DF} g^{-1} blubber lw) than in ringed seals; PCBs were however not analyzed. Storr-Hansen and Spliid (1993a) showed their importance in harbour seals found in Kattegat. Lipid contents of the blubber samples were not given; lipid based levels may have been nearly twice as high in starved animals (cf. Bignert et al. 1989). The average WHO-TEq_{P} levels were c. 100 pg g^{-1} ww, 5-fold more than the WHO-TEq_{DF} (lipid based) in the specimens from these waters studied by Bignert et al. (1989), but 5-fold less than the reported TEQs based on TEFs proposed by Safe (1990). The dioxin-like toxicity was contributed mainly by CB 126 and CB 156, according to WHO TEFs for mammals.

In an experimental study with juvenile harbour seals fed Baltic herring, levels of 200-300 pg TEQ_{DFP} g^{-1} blubber were reported (Ross et al. 1995, 1996c) based on TEFs suggested for seals by Ahlborg et al. (1994). In the Baltic herring feed, the level was c. 400 pg TEQ_{DFP} g^{-1} muscle. Again, these values may be several-fold greater than TEqs based on WHO TEFs.

Total PCBs were measured in Baltic seals since late 1960's (Jensen et al. 1969) using variable methods and standards. Olsson et al. (1974) provided data indicating that the levels were 20-fold higher in 1970 than in 1950-55. Lipid-based bioaccumulation factors from herring were c. 8. Consistent age dependency or regional trends were not found. Nyman et al. (2002) noted that the levels in adult ringed seals had decreased from the c. 200 ppm ΣPCBs lw found by Blomqvist et al. (1992) to a tenth. The development also

Table 11. Summary data on levels of key PCDD/Fs and dlPCB, total dioxin-like toxicity as WHO-TEqs (for mammals and birds) and ΣPCBs in Baltic seals. Arithmetic means and ranges in pg g^{-1} lw (for CB 118, CB 156 and ΣPCBs in µg g^{-1} lw). The values have been rounded up to one signifying digit. Cf. text. For more extensive compilation of data on body burdens in Baltic wildlife including birds, see Annex 6B.

Species, age, sex, cond.	Tissue	Area	Period	WHO-TEq_{DF}	WHO-TEq_{P}	Bio-TEq	4-C_5DF	CB 126	CB 118	CB156	ΣPCBs
Harbor seal	blubber	BAP	1983-87	20[1]			10[1]				
Harbor seal, juv healthy	blubber	KAT	<1988	≥10[5]			5[5]				20[2]
Harbor seal, juv healthy	blubber	S BP	<1988	≥20[5]			9[5]				40[2]
Harbor seal, juv epizootic	blubber	S BS	<1988	≥20[5]			10[5]				30-40[2]
Harbor seal, M, 1 a	blubber	KAT	1988		c. 100 ww[4]			400 ww[4]	0.2 ww[4]	0.06 ww[4]	10 ww[4]
Harbor seal, F, 1 a	blubber	KAT	1988		c. 100 ww[4]			300 ww[4]	0.2 ww[4]	0.09 ww[4]	10 ww[4]
Gray seal, juvenile	blubber	BAP/	1983-7	20-40[1]			10-20[1]				
Gray seal, juvenile	blubber	GUF/	1981,-87	<[6]			<9[6]	600-800[6]			
Gray seal, mature F	blubber	ARC	1981,-87	500-1500[6]			70-600[6]	2000-3000[6]			
Gray seal, juvenile	blubber	GUF	1991-92	20[7]	150[7]		<[7]	600[7]	0.6?[7]	0.2[7]	10[7]
Gray seal, juv healthy	blubber	S BP	<1988	≥30[5]			20[5]				80[2]
Gray seal, mature M	blubber	BP?	1979-90	≥20[5]			9[5]				1002
Gray seal mat. F, healthy	blubber	BP?	1979-90	≥20[5]			c. 10[5]				200-400[2]
Gray seal, mat. F, occl.	blubber	BP?	1979-90	≥20[5]			6[5]				500-900[2]
Gray seal, F, starv. occl.	blubber	BP?	1979-90	≥30[5]			8[5]				2500[2]
Gray seal, mature M	blood	GUB	1996-98			c. 100[3]					40[3]
Gray seal, mature F	blood	GUB	1996-98			c. 100[3]					50[3]
Ringed seal, juv healthy	blubber	GUB?	1979-90	≥70[5]			40[5]				20[2]
Ringed seal, mature M	blubber	GUB?	1979-90	≥200[5]			80[5]				100-300[2]
Ringed seal, juvenile	blubber	GUF/	1981,-87				<-30[6]	1000-2000[6]			
Ringed seal, mature F	blubber	ARC	1981,-87	200-800[6]			30-100[6]				
Ringed seal, juvenile	blubber	BAP/	1983-87	90-400[1]			50-100[1]				
Ringed seal, juvenile	blubber	GUF	1991-92	70[7]	150[7]		40[7]	1000[7]	0.4[7]	0.1[7]	10[7]
Ringed seal, mature M	blood	GUB	1996-98			200[3]					80[3]
Ringed seal, mature F	blood	GUB	1996-98			200[3]					40[3]
Ringed seal F, occluded	blood	GUB	1996-98			200[3]					60[3]

References and explanations: *[1]Bignert & al. 1989; [2]Blomkvist & al. 1992, quantitated as Aroclor 1254; [3]Nyman & al. 2002, 2003, ΣPCBs quantitated as Clophen A50, Bio-TEqs reported for both sexes together; [4]Storr-Hansen & al. 1993a, on seals found dead during epizootic outbreak, ΣPCBs quantitated by HRGC-ECD; [5]Bergek & al. 1992; [6]Koistinen & al. 1990; [7]Koistinen & al. 1997a; grey and ringed seals <2 mo and 1-3 a of age, respectively, ΣPCBs quantitated by HRGC-MS.*

depends on sex, condition and lipid contents (e.g., Koistinen et a. 1997b). Ross et al. (1996c) found levels in captive juvenile harbour seals 2-fold those reported by Blomqvist et al. (1992), which may be due e.g. to prolonged accumulation in the wild. Notwithstanding this variability, Roos et al. (1998) estimated that average declines from c. 1970 in the levels of ΣPCBs in blubber of juvenile Baltic grey seals were less than in fish (c. 4 and 10 % yearly, respectively), and were still less in pups (2 % per year).

As noted by de Wit et al. (1992), the N-TEq levels reported by Bignert et al. (1989) for Baltic seal species were similar on lipid basis to those in herring, indicating no biomagnification of toxic PCDD/Fs to seals.

Several OH-CB and $MeSO_2$-CB *metabolites* of PCBs have been identified (Haraguchi et al. 1992, Bergman et al. 1994a, Janák et al. 1998, see also Olsson et al. 1994) in Baltic seals that have variable ability to metabolize PCBs (cf. Annex 7C). The parent compound of most of the $MeSO_2$-CBs was the non-dioxinlike CB 101, and most products had di-*ortho* structure. However, also dlPCBs, present at lower levels, may be metabolized to $MeSO_2$ and OH derivatives.

In **harbour porpoises** in Baltic and nearby sea areas (e.g. KAT), blubber levels of PCDD/Fs and dlPCBs were reported by Berggren et al. (1999). CB 118 contributed up to 80 % to total WHO-TEq_{DFP} levels of c. 100 pg g^{-1} lw in both mature and immature porpoises. In immature porpoises, Falandysz et al. (1994e) likewise found CB 118 and CB 126 dominated the WHO-TEq_{P} levels of 100-300 pg g^{-1} lw blubber and 200 pg g^{-1} lw liver, while PCDD/Fs were insignificant. Falandysz et al. (2002b) reported an average 50 pg WHO-TEq g^{-1} lw in blubber of porpoises collected in 1991-93, based only on 1-*ortho* PCBs (cf. 5.4.3 and Annex 6B).

Summarizing, DLCs and particularly dlPCBs have been found in elevated concentrations in all marine mammal species in the Baltic. Typically, values around 100 pg WHO-TEq_{DFP} g^{-1} lw have been measured. The levels and congener profiles in these species have varied considerably depending on properties like age. In addition to such real variation, the estimated TEq_{DFP} have exhibited great (sometimes 5-fold) apparent differences depending on the TEFs assigned to 1-*ortho* PCBs.

Fish-eating birds

Guillemots near Gotland, in a small local population of low genetic variation, have been monitored a long time for persistent pollutants and population dynamics. Levels of PCDD/Fs in guillemot eggs declined from early 1970's to c. 1/3 of initial levels, but slowed down already in mid-1980's and may have stabilized (Bignert et al. 1998, Olsson et al. 2005, see also de Wit et al. 1992). The declines already from c. 1970 suggest PCDD/F levels declined also in fish that have not been monitored for as long by comparable methods (cf. Annex 6B).

The decline of ΣPCBs has been still more pronounced, from c. 300 to 50 μg g^{-1} lw from early 1970's to 1990's, but slower than that of DDT metabolites, presumably due to continued sources of PCBs (Bignert et al. 1998). Odsjö et al. (1997) noted that the decline of PCDD/F levels started earlier than that of PCBs for which it happened from late 1970's onwards.

In Gulf of Bothnia **black guillemots** dlPCBs have been found to contribute most to the total dioxin toxicity of c. 5 ng TEq g^{-1} lw (Koistinen et al. 1995a, based on TEFs proposed by Safe).

Broman et al. (1992b) and Rolff et al. (1993) reported levels of c. 20 pg TEq g^{-1} dw in **eiders** (common eider ducks) from the Baltic Sea. The former authors found that out of the total PCDD/Fs consumed, only 10% were retained in the body, while of the 2378-substituted isomers 60 % were retained. No data on dlPCBs or other PCBs have been found.

Black cormorants from by-catch in SB had PCB-TEq levels of c. 7 pg g^{-1} ww and c. 3000 pg g^{-1} lw in both breast muscle and liver, based on TEFs for birds, and mainly contributed by CB 126 and CB 118 (Falandysz et al. 2002b).

Levels of DLCs in **herring gulls** have not been reported from the Baltic (cf. studies in the Great Lakes, Annexes 6B, 8D). The levels of ΣPCBs in herring gulls in S-W Finnish archipelago in 1970's were 10-fold higher than in Arctic terns; it was considered plausible that their body burdens reflected exposures in Southern Baltic wintering grounds (Lemmetyinen et al. 1982).

In chicks of **lesser black-backed gulls**, the levels of WHO-TEq$_P$ were around 20 ng g^{-1} lw in diseased Gulf of Finland and Bothnian Bay populations, rather similar as in the healthy Gulf of Finland population (Hario et al. 2004, Annex 8D). Based on the TEFs by Safe (1990) the levels were higher in the diseased Gulf of Finland site than in the others, but were expressed on wet weight basis and thus affected by the variable lipid content.

In **little terns** in Southern Baltic coast, levels of dlPCBs were found to have decreased from 1970's-1980's (Thyen et al. 2000, cf. Annex 6B).

Organochlorides including ΣPCBs have been measured in **Arctic terns** in S-W Finnish archipelago (Lemmetyinen and Rantamäki 1980, Lemmetyinen et al. 1982). The ΣPCB levels in livers of males were 2-fold higher than in females and chicks, but much below those in herring gulls. Notably, the levels were similar in Arctic terns nestling in Lapland and thus exposed to Baltic fish to a much smaller extent (cf. 4, 5 and Annex 8D).

Birds of prey

White-tailed sea eagle is the bird species most commonly analyzed for PCD/Fs, PCBs and DLCs in the Baltic Sea region. High levels have been reported mainly in eggs and in muscle of dead eagles (Tarhanen et al. 1989, de Wit et al. 1992, Falandysz et al. 1994a, Koistinen et al. 1995a, 1997b, cf. Annex 6B). The levels have depended in part on analytical methods and on whether and what PCBs have been included. Tarhanen et al. (1989) only analyzed 0-*ortho* PCBs in addition to PCDD/Fs, and de Wit et al. (1992) only PCDD/Fs, finding 10- to 30-fold biomagnification of lipid-based N-TEqs from pike to eagle eggs, comparable with that from herring to guillemot eggs. Falandysz et al. (1994a) reported levels in wet weight but not lipid contents. Moreover, lipid-normalized levels of PCBs reported e.g. by Koistinen et al. (1995a) and Koistinen et al. (1997b) differ by orders of magnitude (see Annex 6B). Thus, comparisons and evaluations of the data are fraught with difficulties (cf. 5.4).

No time trends could be seen in levels of WHO-Teq$_{DF}$ or ΣPCBs in East German coastal white-tailed eagles (Kannan et al. 2002a). Tarhanen et al. (1989) and de Wit et al. (1992) found that the levels of the analyzed DLCs were 5- to 20-fold higher (depending on unit) in the Baltic than in Lapland, presumably unexposed to Baltic fish. Koistinen et al. (1995a) found that CB 156 contributed most to the total dioxin-like toxicity of PCDD/Fs and PCBs in this species.

Olsson et al. (2000b) analyzed CB 118 and some methylated PCBs (mainly 4-MeO-CB 187) in the blood of white-tailed sea eagle nestlings. The contribution of dlPCBs has been highest to

the total TEqs, reaching 100 ng g^{-1} lw in eggs and muscle (Paasivirta 1990).

In **ospreys**, PCDD/Fs in breast muscle have been reported in the Baltic Sea area only from inland sites or pooled specimens from Baltic and inland areas (Annex 6B).

In summary, high levels of PCDD/Fs and dlPCBs have been measured in Baltic Sea living wildlife. The levels have however varied greatly by species, age, tissue and condition, and generalizations and comparisons are thus difficult to make. Also the relative importance of the various congeners and groups has varied; generally dlPCBs have been more important than PCDD/Fs. It seems on the basis of some long-term data that the levels have decreased several-fold.

Other dioxin-like compounds

Bioaccumulation in Baltic biota is expected of many other DLCs. However, data are scarce, and predictions are constrained by the paucity of information on fate properties and processes.

PCNs accumulate in Baltic food-chains (Falandysz et al. 1996a,b, 1997a,b, Lundgren et al. 2003a, 2004, Lundgren 2003, cf. Annex 6B). White-tailed sea eagles on the Baltic coast had up to 20-fold higher levels of CN 66/67 in liver than inland eagles had (Falandysz et al. 1996a). Levels in black cormorant (Falandysz et al. 1997a) were comparable to those in sea eagles.

Wiberg et al. (1992) found no *PBDD/Fs* in Swedish salmon, osprey or mother's milk at a LOD of 20 ppt lw. Also Haglund et al. (2005) a lower LODs of 0.1-2 pt lw could not detect key 2,3,8,7-PBDD/Fs in Baltic herring and other fish. Malmvärn et al. (2005) tentatively identified two TBDDs at a level of 2 ppt lw in *Mytilus edulis*. The decline of human serum levels of some PBDD/Fs in Japan (Choi et al. 2003) supports the notion that PBDD/Fs do not bioaccumulate as strongly as PCDD/Fs.

3.5 Human exposures to dioxin-like compounds in Baltic Sea fish and other sources

3.5.1 Consumption of fish and fish products and intakes of dioxin-like compounds

Fish consumption estimates

Various data, estimates and proxies of human consumption of fish and fish products are available for Baltic Sea countries (Annex 7A). These data include landings, amounts available for consumption on the market, and actually consumed amounts (Fig. 10).

The landings differ greatly from the amounts available on the market for human consumption, as many fish are used industrially for other purposes. The total landings of fish in Sweden have remained at c. 30 kg a^{-1} per capita since 1980, and since 1970 in Finland (20 kg a^{-1} per capita in Denmark), i.e. c. 80 and 50 g d^{-1}, respectively (NMR 2001). The amounts consumed as fish in human diet have been 3-4 fold lower, c. 20 g d^{-1} per capita. The share consumed as fish varies by species, being low for sprat and high for salmon. The share of Baltic fish consumption is difficult to estimate for Sweden and Denmark as fish are also obtained from the Atlantic; however, it can be approximated from the data on catches and landings from the Baltic Sea for some important species such as herring (cf. Annex 7A).

The data from dietary surveys are based on food basket, duplicate portion, dietary recall and other studies that also have methodological problems (IPCS 2000, cf. Annex 7A). As noted e.g. by Lind et al. (2002), consumption by non-respondents in survey studies is poorly known. Estimates of consumed amounts depend on both consumption frequency and portion size (c. 100 g for adults and most Baltic fish dishes but varying individually, cf. SPCFC 2005). For instance, the higher estimates obtained by Svensson et al. (1995a) for consumption of (fatty) fish by their referents than in the Riksmaten 1997-98 survey (Becker and Pearson 1999) may be due largely to the high average age of the referents.

Consumption patterns vary between different fish species and products, regions, and age and other groups (e.g. gender, profession) of the consumers. The distribution of fish consumption

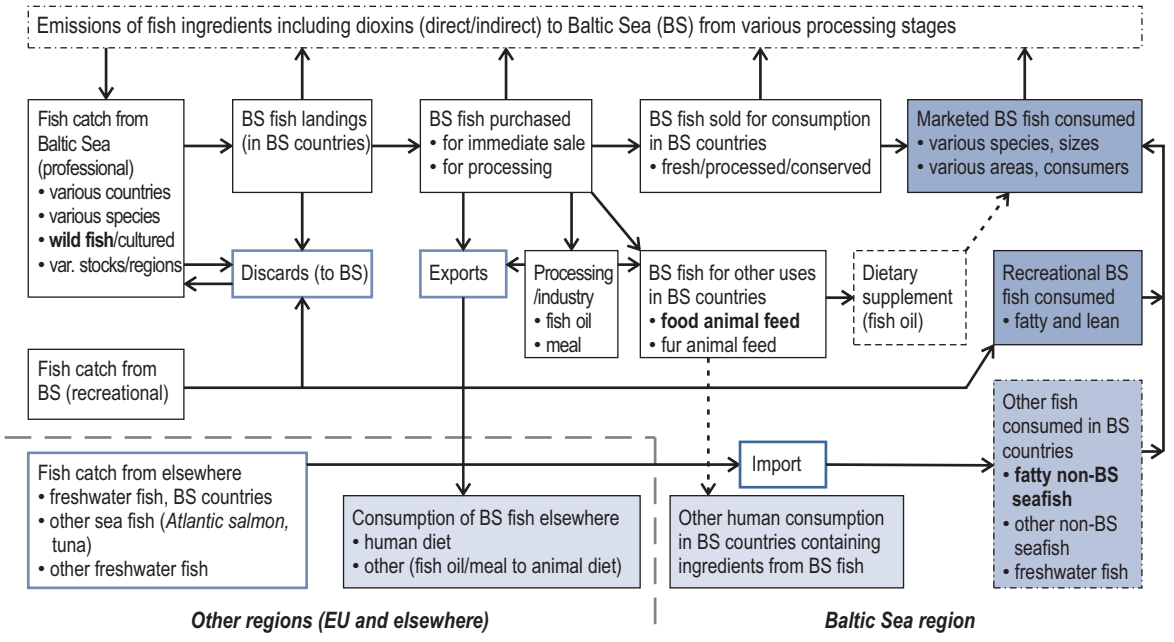


Fig. 10. Simplified flow diagram of the production, distribution and consumption of fish from the Baltic and other regions, emphasizing systems and flows relevant for human exposures to dioxin-like compounds. Important and sensitive entities, items processes have been indicated by shading and boldface. Note the (permeable) regional system boundaries. Cf. text.

is skewed, as many people eat no fish, most people eat moderately and only some people large amounts (Lind et al. 2002). The amounts consumed per capita in the total population thus differ from those in the population consuming. The consumption of fatty Baltic fish depends especially among women on age, young women consuming much less than old women. Coastal fishers in Finland chose herring (usually >3 a old) as their most favoured species twice as often than inland fishermen did (Kiviranta et al. 2002); this tendency is notable among other consumers (Männistö et al. 2003).

A key deficiency is that *children* have seldom been studied. The data by Adolf et al. (1994) were for whole Germany; thus, the Danish survey of 1995 (Fødevarestyrelsen 1997, cf. Annex 7A) seems the only one giving fish (and fatty fish) consumption for the youngest age classes. According to some appraisals the consumption of herring by children and adolescents is low in comparison with the general population (Bjerselius and Darnerud, oral communication 2002-2003).

There are indications that fish consumption has decreased, e.g. in Denmark from 1995 to 2001 among both women and men of various ages, but without simultaneous increase in the consumption of meat that has on the contrary decreased even more (Fagt et al. 2002).

Intakes of PCDD/Fs and dioxin-like PCBs

Total intakes

Dietary intakes represent by far most of the dioxin intake for the majority of people also in Baltic Sea countries. Comparative data on dietary intakes of PCDD/Fs have been provided for some Baltic Sea countries in the SCOOP study (2000) and by SPCFC (2005, cf. Annex 7B). More detailed data have been provided by Lind et al. (2002) for the general Swedish adult population. In addition, food survey and market basked studies have been made (Kiviranta et al. 2001, 2004). Alternatively, body burdens can be measured in subjects with known fish consumption (cf. 3.5.3).

Regarding these estimates the following points need to be mentioned:

- **Consumption** is affected by uncertainty e.g. from short survey durations (SPCFC 2005) in relation to temporal variations, and variations in consumptions between age groups and other groups
- **Contents** of DLCs have been often based on substituting limit of detection (LOD) for non-detects (e.g., SCOOP 2000, cf. Lind et al. 2002). As shown by Kiviranta et al. (2004), treatment of non-detects significantly influences intake estimates. Also other uncertainties are included, e.g. due to pooled samples and extrapolation (Lind et al. 2002).

It seems evident that the total intakes of WHO-TEq$_{DF}$ have decreased in the Baltic Sea countries for which there are extensive data. The development of dlPCB intakes is more difficult to estimate. Kiviranta et al. (2001) analyzed PCDD/Fs and PCBs in common Finnish diet ingredients and food consumption data from a dietary recall study, calculating a total intake of c. 50 pg WHO-TEq$_{DF}$ d^{-1}, c. half of the earlier estimate in 1992. The calculated dlPCB intake was also c. 50 pg WHO-TEq$_{P}$ d^{-1}, resulting in a total intake of 100 pg WHO-TEq$_{DFP}$ d^{-1} (c. 1 pg TEq kg^{-1} bw d^{-1}), i.e. within the TDI range proposed by the WHO. Additional studies are needed e.g. among population sub-groups.

Intakes from Baltic Sea fish and other fish

Intakes of DLCs from Baltic fish have been assessed by combining fish consumption and concentration estimates. The general uncertainties regarding consumption and contents (cf. above) are compounded in the case of fish by several factors:

- Of Baltic Sea countries, SPCFC (2005) used only Swedish data but gave no source; a distinction was made between salmon and herring from the Baltic and other areas
- There is information suggesting that the reported consumption of Baltic salmon has been exaggerated by respondents in dietary surveys, up to 5-fold (Lind et al. 2002, Vaz 1995)
- Little information exists on the consumption of some fish products that may contribute significantly to total intakes, such as eel for which Lind et al. (2002) used older data
- Even for herring, intake estimates have been based on limited geographical coverage of stocks (Lind et al. 2002) and of different age classes of herring (cf. TWGIM 2004a)
- There is little information on the consumption of fish caught off-market, such as recreational and semi-professionally caught fish (Lind et al. 2002)
- Registration of non-respondents as non-consumers may have caused underestimation of consumption (Lind et al. 2002).

The uncertainties of the consumption of fish may exceed those regarding fish dioxin contents. The distribution in consumption of the key fish species, stocks and tissues between populations in particular needs to be clarified for more realistic assessment of intakes, including those by high-risk groups (cf. Lind et al. 2002, below).

Prior to ingestion, levels and amounts of DLCs may change in food *processing*. Karl et al. (2002) did not find great differences in dioxin levels between fresh fish and corresponding fish products, but others have reported such changes (cf. Annex 10). Alterations in intake may mainly depend on what part of the fish is consumed, e.g. the parenchymal skin fat of herring.

In addition to DLCs in fresh fish, DLCs from the Baltic are consumed by humans in:

- **Farmed fish:** Data on dioxins were published by Isosaari et al. (2002b, 2003). Intakes of such fish are poorly known (cf. SPCFC 2005). Levels of DLCs are lower than in wild salmonids but rainbow trout carries total amounts
- **Fish oil** based dietary supplements: Some of these are based on fish from other areas than the Baltic. The contribution of Baltic fish is difficult to quantify.
- Other industrially processed directly fish-based products
- Food production animals that have been fed fish, fish meal or fish oil from the Baltic, e.g. pigs and poultry (cf. SCAN 2000).

In **Sweden**, detailed estimates of PCDD/F and dlPCB intakes by the general adult population have been produced from the Riksmaten 1997-98 data and from measurements of levels in various categories of fish (Table 12, cf. Lind et al. 2002, Annex 7B). Intakes by high consumers of fatty Baltic fish have also been estimated in studies in particular populations, mainly East coast fishermen (Svensson et al. 1995a, Asplund et al. 1994, Hagmar et al. 2004a,b) and their wives (Rylander et al. 1995), and among referents.

The average share of fish of total intakes of WHO-Teq$_{DFP}$ by the general population, c. 55 %, was below that in Finland (cf. Kiviranta et al. 2004, below). Fatty Baltic fish was the single food category that contributed most to intakes of WHO-TEqs. Estimates depend on the frequency, amount, species and quality of fish consumed that in turn are affected e.g. by dietary advice and vary among population groups. The advised frequencies of one fish meal a month only for consumption of fatty Baltic fish by women in child-bearing age is calculated to correspond to 4 g d^{-1} (Darnerud et al. 2003).

Table 12. Estimated average contributions of different categories of Baltic fish and other foods to present intakes of dioxin-like toxicity by some populations in Baltic Sea countries (mainly based on SCOOP 2002, Lind et al. 2002, Kiviranta et al. 2004 and SPCFC 2005). Cf. text, Table 12 and Annexes 7A, 7C.

Population, fish consumption, age, region	Average intake (95 % percentile, or mean +2SD), pg WHO-TEq_{DFP} d^{-1}, from various categories of dietary items							
	BS herring	BS salmon	Other wild BS fish	BS rainbow trout	All fish and shellfish	Fish fodder based foods	Other foods	Total diet
Adults, general, FI	48			14	81	?	19	100
Women, general, SW[2]	30 (130)				52 (150)	?	37 (50)	89 (200)
Women, general, 17-30 a[3]	11				28	?		72
Men, general, SW[2]	32 (130)				57 (200)	?	53 (50)	110 (240)
Men, general, 17-30 a[3]	20				38	?		110
Men, high consumer, SW[4]	100				170			
Men, high consumer, SW[5]	30 (100)	17 (60)						
Women, >consumer, SW[6]								
Adult, high consumer, FI[7]								
Children <10 a, general[8]	?	?	?		3	?		
Children, > consumer	?	?	?			?		

References and explanations: [1]From Kiviranta & al. 2001; in parentheses, estimates for other categories based in part on consumption data by FGFRI (2003); [2]Lind & al. 2002; [3]Lind & al. 2002, average of means for age classes 17-20 and 20-30 a; [4]Asplund & al. 1994, for consumption of all fatty fish by fishermen or fish industry workers in SE Sweden; [5]Svensson & al. (1995a), for East coast fishermen, cf. Hagmar & al. 2004b and 95 % percentiles reported by Lind et al. 2002; [6]Rylander & al. 1995; [7]Based on information on consumption by coastal fishermen by Kiviranta et al. 2002 and on mean intake + 2SD in data on both genders of the general population by Kiviranta & al. 2004; [8]Rough estimate based on fatty fish consumption data for Danish children by Fødevarestyrelsen 1997, cf. Annex 7A, and average levels and intakes of WHO-TEQs in Swedish and Finnish fish food baskets.

The contribution of dlPCBs to total WHO-Teq_{DFP} in intakes from fish was at c. 50 % comparable to that in Finland (cf. Kiviranta et al. 2004). DlPCBs had a higher share of total WHO-TEq_{DFP} in fish than in other dietary items (Lind et al. 2002).

In **Finland**, Kiviranta et al. (2004) estimated based on a food basket study and the Findiet survey (cf. Annex 7C) that of the total dietary intake by adults of c. 60 pg TEq-WHO_{DF} d^{-1}, c. 90 % came from fish; for dlPCBs, the share of fish was c. 70 % of the total intake of c. 60 pg TEq-WHO_{p} d^{-1}. Thus, c. 80 % of dioxin toxicity in human dietary intakes (and almost as great a share of all intakes) originated from fish. Kiviranta et al. (2001) specifically estimated on the basis of dietary recall studies that herring contributed most, c. 50 pg WHO-TEq_{DFP} d^{-1} (70 % of this from 4-PeCDF), rainbow trout 10 (70 % from CB 126) and other fish 20 pg WHO-TEq_{DFP} d^{-1} to the total intake by adults (Table 12). The shares of the fish categories of total intake vary e.g. between regions and age classes.

In **Germany**, data are available for PCDD/Fs on total dietary intakes and on different food items (for 1998) e.g. as reported by SCOOP (2000, cf. Annexes 7B, 7C). These data are for populations not consuming great quantities of fish in general, and the share of fish of total PCDD/F intakes has been c. 10 %. The contribution of Baltic fish to total intakes is still smaller except for some coastal populations.

In **Denmark**, the contribution of fish to dioxin intakes was not reported in SCOOP (2000). The share of Baltic fish is difficult to specify as much of the fish on the consumer market comes from North Sea. It has been estimated that dlPCBs contribute more than PCDD/Fs to WHO-TEq_{DFP} levels in the Danish diet (Fødevaredirektoratet 1999).

In **Poland**, some data are available on PCDD/F levels in food items including Baltic Sea fish and fish oil (Falandysz et al. 1994d, 2002a), but intake estimates have not been found. Such figures depend heavily on the amounts consumed and other assumptions.

Intakes in Baltic fish based products

Human intakes of PCDD/Fs and dlPCBs in animal foods produced with feeding-stuffs based on Baltic Sea fish are potentially important for the overall risks from this fish. Greatest concern has been associated with fish meal and fish oil (SCAN 2000, SPCFC 2005). There is a lack of information on the use of fish-based feeding-stuffs specifying Baltic Sea fish (sprat and herring) in the production of food animals. Also the specification of congeners in addition to TEqs is important (Verstraete 2002). It is known

that during the 1990's most of Finnish herring landings were fed to fur animals.

The flux of PCDD/Fs in Finnish rainbow trout fed Baltic herring and sprat has been subject to some studies (Vartiainen and Hallikainen 1995, Isosaari et al. 2002b); the levels of PCDD/Fs and PCBs and resultant human intakes in rainbow trout have not been very high.

The influence of fish-based feeding-stuffs on human dietary intakes in EU has been assessed at an aggregating level (SCAN 2000). It may be expected that the increased control of feeding-stuffs have reduced such uses of Baltic fish. It is on the other hand known that much of the sprat catch e.g. of Finland has in recent years been sold to Russia for processing to human food (Abbors 2003). It is conceivable that some use of Baltic fish also in fodder for human food animals takes place in Baltic Sea countries and elsewhere based on exports.

Intakes of other dioxin-like compounds in fish

The intakes of *PBDD/Fs* and PBCDD/Fs from Baltic Sea fish cannot be estimated due to the lack of data and evaluated models (cf. Annex 4). However, no 2,3,7,8-brominated congeners have been reported in Baltic fish.

The share of *PCNs* of total I-TEq (in intakes) has been deemed small, around 2-5 %, but this may depend on area, species, etc.

3.5.2 Absorption, distribution, metabolism and excretion

The phamacokinetics of DLCs include toxicokinetics and kinetics at non-toxic doses (Neubert 1997/98), and operate at many levels from cellular to body (Safe 1989b, Hu and Bunce 1999, USEPA 2000a). They play a key role for critical doses, and thus for effects and risks (Van den Berg et al. 1994). Risk assessment based only on intakes may therefore be severely deficient (van Birgelen et al. 1995a, DeVito et al. 1997).

Pharmacokinetics of DLCs in the body involves various linked processes and compartments, united by blood. There are dynamic balances between these compartments and between intakes and outputs (Fig. 11).

Pharmacokinetics causes differences between congeners in apparent toxicity (DeVito et al. 1995). Petroff et al. (2001) even maintained that the most critical difference between rats and humans regarding risks from dioxins is their kinetics. Emond et al. (2003a, 2004) estimated by a modification of the Wang et al. (1997) model for some DLCs elimination rates 20-fold those of TCDD in rats while the relative potency factors can vary over 2 orders of magnitude.

Generally, the kinetics of DLCs *varies* between species, developmental stages, gender, tissues (aggregated in whole-body elimination)

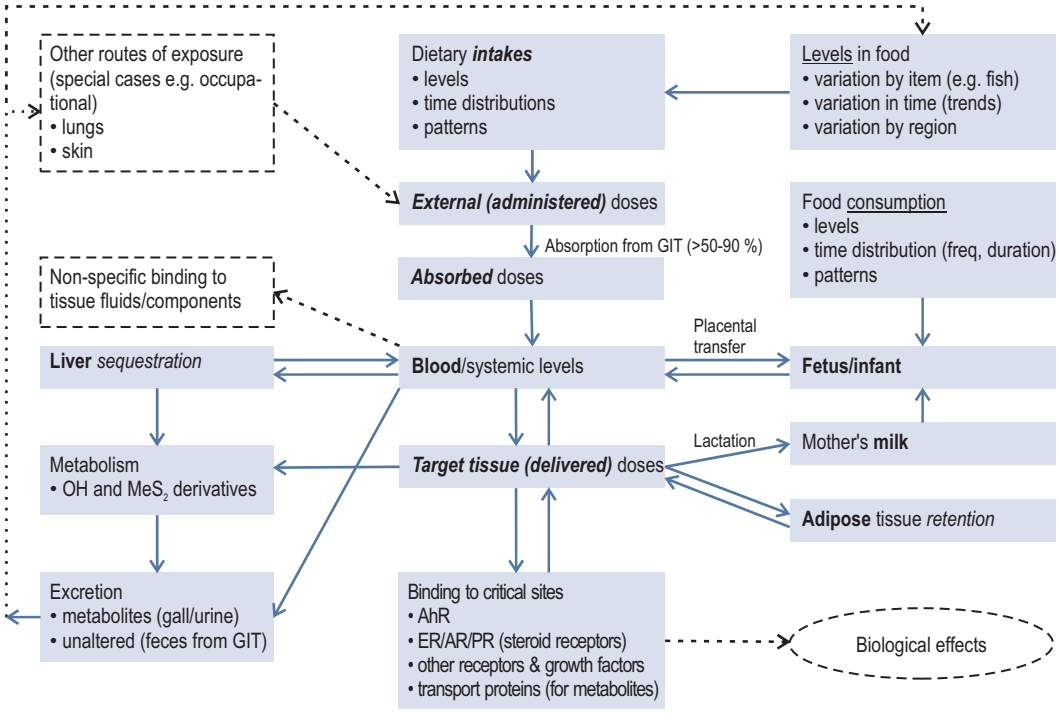


Fig. 11. The contents of integrated human exposure assessment applied to dioxin-like compounds (modified from EUSSC 2000). Key ***dose concepts***, **compartments**, *processes* and some variations have been indicated.

and nutritional and general conditions (Van der Molen et al. 2000, Geyer et al 2002, cf. Annex 7E). Kinetics also depends on congeners and dosing. Differences are associated e.g. with liver retention (van der Plas et al. 1998).

Most human data have been obtained in studies of occupational and accidental (poly)exposures (e.g., Flesch-Janys et al. 1996). Pharmacokinetics in wildlife will be additionally discussed in connection with ecotoxicological effects assessment (cf. 4.3.2).

There is less information on the pharmacokinetics of *other DLCs* than TCDD, including PCBs, and on mixtures of DLCs especially in chronic exposures (Abraham et al. 1989, Krowke et al. 1990, Darnerud et al. 1996b, DeVito et al. 1998, Kodavanti et al. 1998, van der Plas et al. 1998, Chen et al. 2001, Korner et al. 2002). The total effect of congener-specific kinetics is thus difficult to elucidate. Exposure to mixtures can alter the kinetics of individual compounds (Van den Berg et al. 1994). Observed over- or underestimation of effects from TEq doses may be due also to kinetic interactions.

Pharmacokinetics of DLCs *in utero* and maternal-foetal disposition has been studied mainly in rats (Hurst et al. 2000b, Emond et al. 2004). The data of Hurst et al. in Long-Evans rats are crucial for quantitative risk assessment, as they studied disposition from dams to foetuses in dosing schemes that corresponded with the lowest-dose effect studies (cf. 4.2.3).

In *humans*, data have been provided on foetuses and (more commonly) infants e.g. by Abraham et al. (1996), Patandin et al. (1999), Heudorf et al. (2002), and Guvenius et al. (2003) (cf. Annex 7E). Based on such studies, quantitative models of dioxin pharmacokinetics have been developed (Kreuzer et al. 1997, Van der Molen et al. 2000, Lorber and Phillips 2002, see also Alcock et al. 2000 and LaKind et al. 2000).

In *wildlife* species of Baltic Sea relevance, pharmacokinetics has been mainly studied in salmonid fish (e.g., Kleeman et al. 1986a, Fitzimmons et al. 2001, Jones et al. 2001). There are experimental data also for minks (e.g., Heaton et al. 1995, Hochstein et al. 2001) and seals (e.g., Sormo et al. 2003, cf. Hickie et al 1999, 4.3.2, Annex 7C). Metcalfe and Metcalfe (1997) found that although the metabolic clearance of CB 126 was higher than that of some other PCB congeners in a fish-herring gull food chain, the levels in gulls dominated the total TEqs. The validated model by Drouillard et al. (2001) for biomagnification and elimination of mono-*ortho* PCBs in kestrels accounts for body fat store changes e.g. during migration or reproductive output, in addition to congener structure. Of CB 118, 70 % was cleared in 1 a. Drouillard et al. (2003) also developed and validated a bioenergetic model for PCB kinetics in herring gull embyos that was sensible to the mass of yolk lipids in addition to the mass of PCBs deposited in egg; predicted PCB levels were highest during pipping or soon after. Generalization of these models to other bird species is uncertain, but they indicate that tissue- and age-specific kinetics may need to be accounted for.

Little direct information is available on *absorption* of DLCs in humans. Absorption of TCDD from diet seems to be comparable to that in rodents, being >80 %, but absorption is lower for higher chlorinated PCDD/Fs (van den Berg et al. 1994).

DLCs are *disposed* between tissues mainly according to their fat contents (see esp. Van den Berg et al. 1994, USEPA 2000a). Liver plays a key role in disposition and transformation as the enzymes responsible for detoxification largely reside there. Blood provides the link between tissues. Significant transfer results in many mammals also through placenta and especially milk (see below). In addition, DLCs are distributed to genitals, lymph tissues, some endocrine glands including thymus, and skin epithelium, all of which are important sites of dioxin action.

PCDD/Fs and related compounds are *excreted* mainly in faeces (e.g., Schrey et al. 1998). Moser and MacLachlan (2002) showed that the net uptake from a fish meal may be compensated by excretion during the rest of the week. However, the considerable individual and temporal variability in net excretion constraints uptake models for the digestive tract.

Perinatal exposure is of particular interest (Kreuzer et al. 1997). Infants receive a proportionally great dietary dose especially in milk. On the other hand, as pointed out by Clench-Aas et al. (1992), this load represents 5 % of the total dose through a 70-year lifetime, and the concentration in fatty tissue of an infant does not increase because the percentage of this tissue increases rapidly during the first year of life. TCDD half-life is short in infants but increases to c. 10 a between the age of 40 and 60 a; that of some other congeners such as 4-PeCDF has been estimated to be still longer.

Some of the relevant findings regarding *young* stages may be summarized:

- Infants absorb >95 % of most PCDD/Fs including 4-PeCDF and PCBs from breast milk (Dahl et al. 1995).
- PCDD/Fs and dlPCBs are transferred over the placenta to foetuses *in utero*. 4-PeCDF and PeCDD may not be as readily transferred to foetuses as TCDD, while the relative risk of HxCDDs to the offspring may be increased (Van den Berg et al. 1987). However, the contribution of breast-feeding to total body burden is in most cases smaller than that of gestational exposure
- Children receive per unit body weight a high DLC load that is diluted by growth but persists at noticeably elevated levels even to adulthood
- Brown adipose tissue in infants is important for DLC kinetics (e.g. Rozman et al. 1987)
- An age-dependent half-life model for infants and young children may predict lower body burdens than by assuming a constant half-life (Paustenbach 2004, see also Kerger et al. 2004).

Pharmacokinetic *models* involve uncertainties (Maruyama et al. 2003) but have been advanced. Carrier et al. (1995a) noted that the model of Leung et al. (1990) to describe TCDD disposition in rodents operates on parameters not available in humans or for other congeners. Carrier et al. (1995a,b) developed robust models based on readily available parameters and accounting for non-linear dependency on dose, testing, them in acute and chronic exposures at various doses for multiple species, also humans. Additive behaviour of PCDD/Fs in low doses was assumed (cf. Aylward et al. 2005). Carrier et al. (1995a,b) specifically accounted for the fact that body burdens will be higher in lean than obese persons, and concluded that caution is needed in deriving clearance rates solely from tissue lipid data. It was also shown that as body burdens decrease, the adipose half-lives increase, e.g. for 4-PeCDF in humans from 1 a at higher doses to >30 a at background doses (cf. Brewster and Birnbaum 1987, 1988, Brewster et al. 1988a, Annex 7E).

DLCs in the body, although reacting slowly to altered intake, are still in a *dynamic* state. Growth causes some dilution and redistribution. Marked changes in body levels of DLCs are seen in mammals during pregnancy as mothers release much of their body burden to the child through placenta and in milk (Krowke et al. 1990). Chen et al. (2001) specifically reported dose-dependent transfer from rat dams to pups of a DLC mixture approximating that in food, concluding that extrapolation from high to low doses may under-predict the exposure of offspring.

Little is known of kinetics of *4-PeCDF* in humans. In rodents and monkeys, 55 and 30 %, respectively, has been retained in liver (Van den Berg et al. 1994). Brewster and Birnbaum (1987) provided data indicating that the 4-substituted Cl in 4-PeCDF hinders its metabolization, resulting in high accumulation. Pluess et al. (1987) showed that methyl ether metabolites were formed in the rat. Chen et al. (2001) found that the behaviour of also 4-PeCDF was dose dependent and that extrapolation of high-dose data should be made with caution (cf. 5.3, 5.5.1).

For *PeCDD*, the scaling from rodents to humans is uncertain; proposed relationships (Geyer et al. 2002) result in half-lives from 5 to 13 a. Armitage and Cousins (2005) estimated half-life of c. 9 a using a simple bioaccumulation model that explains the increase in half-lives with increasing total body fat content and the change in elimination kinetics with age.

The pharmacokinetics of *dlPCBs* in some respects differ from those of PCDD/Fs, e.g. through metabolites (cf. Annex 7E). In general, the kinetics of dlPCBs is related to the degree of chlorination and dioxin-like activity. CB 126 and CB 169 thus accumulate to greater extent than CB 77 (Feeley and Jordan 1998). Also higher chlorinated congeners like CB 169 are readily transferred to offspring in milk (Chen et al. 2001). DeVito et al. (1998) reported greater hepatic sequestration in mice for CB 126 and 4-PeCDF than for TCDD. Öberg et al. (2002b) on the other hand found retention in rat liver was higher for PCDD/Fs than PCBs; thus the contribution of PCDD/Fs to total TEqs in liver was higher than in the fish oil diet. The modelling results of Maruyama et al. (2004) indicated that the transfer of CB 126 (and CB 169) to infant blood was over-predicted in comparison with the fit for PCDD/Fs, e.g. 4-PeCDF. Data on mono-*ortho* PCBs are scarce. Of Aroclor 1254 congeners, CB 105, 118 and 156 seem to predominate in rat brains, on concentration basis (Kodavanti et al. 1998).

The kinetics of *PBDD/Fs* is poorly known. There is evidence that PBDD/Fs are more vulnerable to metabolic attack (Birnbaum et al. 2003), but have comparable persistence in some tissues (Zober et al. 1992, Nagao et al. 1995).

3.5.3 Body burdens and contributions from fish

PCDD/Fs and dlPCBs

General

Congener-specific data on human body burdens of PCDD/Fs and dlPCBs in Baltic Sea countries, mainly in blood fractions and mother's milk, have been reported in the general population and in some specific groups including high consumers of Baltic fish already during several decades (cf. the compilation by Jensen 1987). There are practically no data on children.

The congeners analyzed vary; many studies and TEqs reported have not included dlPCBs. The contribution of fish intake can be deduced only indirectly also in the high consumers as they have been exposed to other sources. The time course of intakes and body burdens is also poorly known; there have however been some prospective studies of cohorts (Hagmar et al. 2004b).

Mother's milk

Milk represents an important pool of DLCs in humans as in other mammals, since much of the body burden in mothers is excreted in milk (cf. 3.5.2). Mother's milk makes up most of the load of DLCs on suckling infants and has been estimated to have some influence on body burdens until adulthood (Ayotte et al. 1996, Lorber and Phillips 2002). The latter authors modelled a half-life in infants much lower than in adults, but noted that its health consequences are unclear.

Levels of PCDD/Fs, dlPCBs and related compounds in human milk have been monitored also in Baltic Sea countries (Norén and Lundén 1992, Becher et al. 1995, Atuma et al. 1998a, Kiviranta et al. 1999, Glynn et al. 2001, 2003, Guvenius et al. 2003, cf. Annex 7D). The data produced allow some resolution of trends (Norén and Meironyté 2000). For Baltic States and Poland, scarce data are available (Becher et al. 1995).

In general, levels in human milk in Finland and Sweden are lower than e.g. Germany and the Netherlands (Atuma et al. 1998a), the WHO-TEq_{DF} level in Finland being c. half of that in the latter (Malisch and van Leeuwen 2003). Six-fold variations between maxima and minima have been reported (Glynn et al. 2003). Regional differences are discernible (cf. Annex 7D). Some metabolites of dioxin-like mono-*ortho* PCBs have been found (Guvenius et al. 2003).

WHO-TEq levels in mother's milk have decreased in all the above countries (Kiviranta et al. 1999, BLAD 2002a), e.g. in Sweden by c. 15 % yearly (Norén and Meironyté 2000, cf. Glynn et al. 2003). The decline in WHO-TEq_{DF} among primiparous women in Uppsala country during 1996-2004 was lower but still 6-8 % yearly (Lignell et al. 2005). The declines are not uniform, varying e.g. by congener and being slower e.g. for CB 118 (Norén and Lundén 1991, Becher et al. 1995, Norén and Meironyté 2000).

No studies seem to have specified the relationships between human consumption of fatty fish and levels of DLCs in mother's milk. Vartiainen et al. (1997b) suggested that some of the differences found between levels in Southern and Central Finland were due to intake of Baltic Sea fish.

Blood levels and fish consumption

Svensson et al. (1991) found that plasma levels of several PCDD/Fs were significantly higher in Swedish men with a high intake of Baltic fish than in those who consumed less, concluding that fatty Baltic fish is an important source of PCDD/Fs in persons who eat fish regularly. It was found that fishermen on the East coast had higher levels of PCBs and PCDD/Fs than West coast and other referents (Svensson et al. 1995a). Asplund et al. (1994) likewise reported statistically significant correlations between blood levels of PCBs (including CB 77, 126 and 169) and intake of fatty fish by men from S-E Sweden (cf. Glynn et al. 2000a, Annex 7D). Elevated levels of CB 126 were obtained by intakes of >400 g fat fish a week. Thus, fish from the Baltic was concluded to be a major source of these compounds in Swedes. The contribution of dlPCBs to total N-TEqs in Sweden has been high. Grimvall et al. (1997) found that CB 126 and CB 156 both contributed >40 % of the total of c. 30 ng WHO-TEq_{P} g^{-1} plasma lw among wives to fishermen living on Swedish East coast.

There is considerable variability in the relationship between body burdens of DLCs and fish intake. Sjödin et al. (2001) found that plasma levels of CB 118 and CB 156 were almost the same in Swedish men reportedly consuming high amounts of fatty fish from the Baltic as in those consuming only moderate amounts; the levels of the key metabolite 4-OH-CB 107 were even lower in 'high-consumers' than in 'moderate-

consumers'. The situation was different among Latvian men: the levels of CB 118 and CB 156 in high consumers were 4- to 5-fold higher than in moderate consumers. This may be explained by differences in the study groups, e.g. timing of intakes and age distributions (the Latvians being on the average 6 a older), as well as by different intakes from other sources (including other than fatty Baltic fish). Such findings emphasize the difficulty of drawing firm conclusions about the quantitative importance of Baltic fish consumption for body burdens and risks on the basis of approximate and non-specific information on exposures, and uncertain representativeness and comparability of study populations.

The complex relationship between DLCs and Baltic fish consumption was shown by Hagmar et al. (1998) who reported that among 30 delivering mothers from the SW Finnish archipelago, stated fish intake was uncorrelated with blood concentrations of ΣPCBs and only weakly correlated with indicator CB 153. A reasonable explanation was the previous rapid decline of PCBs in Baltic fish, resulting in less impact of fish intake on the body burdens of PCB in relatively young women. The body burdens of dlPCBs and PCDD/Fs in younger populations consuming fatty Baltic fish may reflect diminished levels in the fish, while those in elderly persons are more strongly influenced by previous higher exposures, as shown also by Grimvall et al. (1997) for dlPCBs in the plasma of different age groups among women consuming Baltic fish.

Hagmar et al. (2004b) found in follow-up studies of 39 Swedish men with variable intake of fatty Baltic fish that serum levels of WHO-TEq_{DF} had on the whole remained unaltered from 1987 to 2002. There were some differences in trends according to fish intake: the mean and median levels in moderate consumers had decreased by c. 20 %, but in high consumers the mean levels had increased. Notably, the levels of CB 126 had markedly decreased in all groups, to <20 % in moderate consumers (and non-consumers) and also in high consumers to c. 30 % of the level in 1987. Although WHO-TEq_{P} levels were not reported in 1987, it seems likely that they had followed a similar trend, as CB 126 contributes much to total dioxin-like toxicity and correlates with other dlPCBs. Some decrease from 1991 to 2001 was also seen in the marker CB 153. These findings indicate that dlPCBs may be less persistent than PCDD/Fs in humans, and that their relative contribution to total dioxin risks, being presently c. 50 % in many populations, may decrease.

Kiviranta et al. (2002a) measured plasma levels of PCDD/Fs and PCBs in fishermen from the Finnish coastal and inland areas. The levels correlated with the frequency of fish meals and consumption of Baltic fatty fish; also age was a significant predictor. The body burden reached a median of c. 200 pg WHO-TEq_{DFP} g^{-1} fat for Baltic Sea fishermen, and a maximum of c. 400 pg g^{-1}, i.e. levels found in Seveso. Adding PCBs, the median body burden increased to 300 (max. 900) pg WHO-TEq_{DFP} g^{-1} fat. PCBs thus contributed

Table 13. Summary of representative data on circulating blood levels of dioxin-like compounds and toxicity in human males in relation to consumption of Baltic Sea fish. Cf. Annex 7D.

Country/ region	Age (a) (x/ median, range or ±SD)	Exposure to BS fish	Sampling period	Concentrations of key congeners and toxicity equivalents (pg g^{-1} lw); the former have been been rounded to two, the latter to one signifying digits (nr = not reported)					
				PeCDD	4-PeCDF	CB 126	CB 156	WHO-TEq_{DF}	WHO-TEq_{DFP}
SW/BOS	M, 55±11[1]	>	c. 1994	35	200	600	76	200	300
SW/BP	M, 51±12[1]	>		20	110	560	68	90	200
SW/S BS	M, 60±16[1]	>		33	160	1100	170	200	300
SW/West	M, 53±14[1]	< (> to fish)		11	47	370	59	50	100
SW	M, 48-7±9-16[1]	-		12	46	330	44	50	100
SW/East	M, 43 (30-53)[2]	>	c. 1993	nr	nr	790	90	80	200
SW/East	M, 42 (25-56)[2]	- (no fish)		nr	nr	220	40	20	70
SW/East	M, 55 (34-62)[3]	high	2002	nr	nr	270	nr	70	c. 50
SW/East	M, 51 (41-64)[3]	moderate	2002	nr	nr	70	nr	30	c. 12
SW/East	M, 53 (34-64)[3]	-	2002	nr	nr	40	nr	20	c. 9
FI	M, 59 (27-76)[4]	>[c]	1997	79	130	400	71	220	340
FI	M, 57 (42-75)[4]	<[c]		43	76	200	43	140	200

[1]Svensson & al. 1995a; [2]Asplund & al. 1994, WHO-TEqs calculated assuming similarity with reported N-TEqs and I-TEqs; [3]Hagmar & al. 2004b, follow -up of the cohort studied by Asplund et al. 1994; [4]Kiviranta & al. 2002a; means of the values for 'exposed' and 'coastal' and of 'unexposed' and 'inland' fishermen, repectively.

almost half the total body burden in terms of dioxin toxicity. Individual variation in PCDD/F (but not PCB) congener patterns could be associated with those in the fish species reported to have been consumed. Linear regression models for ln WHO-TEq_{DF}, ln WHO-TEq_{P} and ln WHO-TEq_{DFP} explained 50-60 % of the variability. Age was the only significant predictor of ln WHO-TEq_{DF} whereas age, amount of fish eaten, and place of residence were predictors of the other TEq measures. These data are augmented by a comparison with levels in the general population (Tuomisto et al. 2004a).

Becher et al. (1995) found a mean WHO-TEq_{DFP} of c. 50 pg g^{-1} ww in Lithuania. PCBs contributed 2-3-fold more than PCDD/Fs. WHO-TEq_{DF} levels had decreased by c. 40 % since 1985-86, those of ΣPCBs had remained unchanged or only slightly decreased, deviating from in Swedish high consumers of fatty Baltic fish (Hagmar et al. 2004b, cf. above).

Glynn et al. (2005) reported preliminary data on marker PCBs in pregnant women in relation to intake of fatty Baltic fish. A slightly but significantly higher serum CB 118 level was found in the women reporting the highest intake. There was no clear gradient in serum levels with intake, the CB 118 (and CB153) levels being lowest among moderate consumers. The levels of CB 156 were still elevated in mothers who had breast-fed during the longest time in their infancy, indicating that exposure to such PCBs can be noticed over two decades later.

Adipose tissue

Data on PCDD/Fs and dlPCBs in human adipose fat was reported by Kiviranta et al. (2005) based on samples taken 1997-99 from 420 individuals with a mean age of 44 a from Southern Finland. The median level in coastal dwellers, c. 50 pg WHO-TEq_{DFP} g^{-1} lw (2/3 from PCDD/Fs), was >20 % higher than that inland. This was concluded to be due to fish intake, on the basis of variations in total TEq levels and congener patterns. In regression models, age, duration of lactation by women, consumption of Baltic herring but also of farmed trout or herring and living area with respect to the coast were highly significant predictors of WHO-TEqDFP; some of these explaining variables may be mutually correlated. Assuming that maximum levels would be reached by 40 a at constant exposure, it was concluded from the observed age dependency (the absence of a plateau in the age-TEq relationship) that the levels of in Finns have decreased during the last 30 a or so, as found in also other studies.

Wallin et al. (2003) studied the development of serum levels of CB 153 (shown to be a marker of all PCBs, Glynn et al. 2000a) from 1991 to 2001 in 39 men. Although there was an average decrease of 30-40 % during this period in CB 153 levels, it was not related to fish consumption or age, but instead to body mass index, indicating slow body removal. There was also considerable inter-individual variation in the temporal development of serum levels. CB 156 contributed most to WHO-TEqs among the 1-*ortho* PCBs in both adipose and liver in a Swedish population (Guvenius et al. 2002); these authors also reported several PCB metabolites in these tissues.

The reported record blood lipid levels of c. 100 ppb TCDD (AP 15.12.2004) in the Ukrainian president Yushchenko poisoned in 2005 corresponds to a dose of c. 2 mg TCDD, roughly equal to the WHO-TEQ_{DFP} content in c. 500 t of large Baltic herring. Assuming no elimination, this could be obtained from a daily consumption of 3 kg of such herring during 50 a; taking into account elimination, the average daily consumption resulting in such levels would be several-fold higher.

Other DLCs

Some studies have been made of PCB metabolites in humans, e.g. 4-OH-CB 107 produced from CB 105 and CB 118 (Sjödin et al. 1998, Hovander et al. 2002, Guvenius et al. 2002). Metabolites of primarily non-ortho PCBs including CB 126 have not yet been reported in humans in the Baltic Sea countries.

For *PBDD/Fs* (and PBBs) there are almost no human data from Baltic Sea countries. Kotz et al. (2005) reported that the average 'PBDD/F-TEq' in human milk also from Finland was within 0.3-1 pg g^{-1} lw, c. 10 % of the corresponding WHO-TEq_{DF} (and only 5 % of WHO-TEq_{DFP} including dlPCBs). Choi et al. (2003a) reported that the median levels of the sum of TBDD, TBDF and 4-PeBDF in Japanese adipose tissues in 1970 and 2000 were 5 and 3 pg/g lw, respectively; they had thus not increased with time, although the levels of PBDEs in the same tissues had increased strongly. These results suggest that PBDD/Fs may not necessarily become a great human health problem despite increased exposure to PBDEs.

4.1 Assessment principles and evaluation of the quality and relevance of information

4.1.1 General considerations

Elucidation of effects is the crux of risk assessment also for DLCs. Without detailed information on effects risks cannot be adequately identified, estimated and characterized, and their certainty and significance evaluated. Both the biological forms and basis of effects and the rationales used in interpreting them need scrutiny. Assessment of effects is crucial also in evaluating management options. In particular, the information and inference about effects in authoritative risk assessments underlying recent regulatory risk management strategies need to be appraised, taking into account additional information and judgments and modifying the assessments with a view of the Baltic Sea. While an extensive evaluation of the evidence for biological effects of DLCs is beyond the scope of this work, a scrutiny of much of it is essential for the present assessment.

Effect mechanisms play an important role in risk assessment. Mechanistic information may e.g. focus assessment, and clarify variability in risks. Although TCDD is among the most studied chemicals, many aspects of the effects of TCDD and other DLCs are still not well known. New mechanisms, pathways and factors are being discovered, and may alter risk assessment and provide new insights in risks. However, detailed mechanistic information is not required for all assessment purposes. There may be a preference for identifying and accounting for mechanisms that increase risks over those (e.g., detoxification, adaptation, repair) that are conducive to decreasing risks. Thus, the mechanisms and factors of significance, risk-increasing of risk-decreasing, need to be sorted out from less significant ones, and the level of treatment of mechanisms needs to be adjusted.

Usually it is assumed that the effects of DLCs are mediated and thus defined by the **AhR**. This has been questioned based on empirical evidence (e.g., Bol et al. 1989), and less clearly AhR dependent mechanisms have been implicated (cf. Annex 8A). Among PCBs, especially 1-*ortho* congeners have partly different mechanisms from PCDD/Fs (Kodavanti et al. 1995). Part of this complexity is related to the variability in AhR and related genes (Pohjanvirta et al. 1999). As discussed by Puga et al. (2000, 2002), the activated AhR acts as a sensor and cell cycle checkpoint that commits cells exposed to adverse environmental stimuli to arrest before the onset of DNA replication. Hanlon et al. (2005) concluded that TCDD can affect gene expression through multiple mechanisms including AhR complex formation with retinoblastoma protein, stimulation of growth factors, effects on kinases and interference with AhR-nonrelated nuclear receptors.

Ever more subtle effects and action mechanisms of DLCs at **biochemical level** are being identified, also in human target tissues *in vivo* at ambient exposure levels (e.g., Grassman et al. 2002). The variation in effects is also increasingly characterized, e.g. in connection with the genetic basis of susceptibility (e.g., Puga et al. 2000). Explanations and predictions of effects thus become more accurate. However, there are continuously great uncertainties due to the multi-dimensional variation in such responses. These complicate the assessment of what, if any, the overt toxic or other more clearly adverse effects are in a particular case. Thus, the actual health significance, to humans and other animals, of many of the experimentally observed biochemical effects and mechanisms is unclear.

Dose-response relationships and functions are also in many cases still unclear, although information on them is crucial for quantitative risk assessment. For instance, Van Leeuwen et al. (2000) showed that among 45 rodent studies of non-cancer effects, 21 exhibited linear, the rest non-linear dose-response functions; biochemical responses were mainly linear, most of the clearly adverse endpoints non-linear. For rodent cancer studies, ED01 of 10-700 pg g^{-1} bw was estimated. However, much greater differences in effective doses have been obtained in connection with cancer effects (see below). According to Neubert

et al. (1992a), not a single published study existed in which the dose-response curves for a clearly toxic effect of even a pair of PCDD/F congeners had been analyzed where *sufficient* (emphasis added) points were provided to establish the shape of the curve.

Effects of dioxins particularly in a risk management context include **non-toxicological and other non-biological** effects. Psychological, social, economic, technological or research-related effects arise to a considerable degree from toxicological effects. However, some impacts may be significantly caused and conditioned by other factors, such as perceptions, including fears and wishes, and by policies (which are essentially reflections of perceptions of policy-makers and based not only on scientific information).

Information on effects comes from empirical studies, including *in vivo* experiments and field studies, *in vitro* studies using cell and tissue cultures (or xenografts as a semi- *in vivo* approach) and from theoretical and modelling studies (cf. 4.1.4). Human data are stressed here, as most information from controlled studies of dioxin effects comes from animals, and also many risk assessments have relied on such information, even exclusively (e.g., SCF 2001, cf. USEPA 2000a). Usually insights are obtained by both data from existing sources or original studies and by models. The experimental information on many effects has been obtained after single high doses and not chronic low exposure. On the other hand, much of the information is based on lifetime or cumulated exposures, not on critical development stages (Brouwer et al. 1995). Both factors imply limitations to risk assessment.

Effects assessment overlaps with some areas of **exposure assessment**. The analysis of doses is partly included therein, but is also an integral part of effects assessment particularly in the case of effective tissue doses that result from pharmacokinetics, and still more in the case of dose-response functions. Also here a continuum exists, in accordance with the exposure-dose-response paradigm used traditionally mainly in human toxicological assessment.

4.1.2 Attributability of risks to causes and specification of the role of dioxin-like compounds in Baltic fish

A key constraint in assessing the risks from dioxins in a setting such as Baltic fish is the **low resolution of causality** and of attributable effects in human studies. Thus, although studies in human populations are the ultimate proof of dioxin risks to humans, they are inherently limited in their ability to detect such effects. One reason for this is the ecological complexity of humans e.g. in terms of food choice, even beyond the apparent complexity that is caused by our greater familiarity with human conditions than with those of other animals.

Several systems have evolved for resolution of causation and for related inference, especially in human epidemiology. The most commonly applied systems and criteria include those by Hill (1965) and by Susser (1977 reprinted 1995, 1986, 1991, cf. Table 14). The latter have been adopted for ecoepidemiological risk assessment (Fox 1991, McNabb and Fox 2003, cf. Suter 1993 and 4.3). Also other principles and procedures for epidemiological causal inference have been developed, both nationally (e.g. USEPA 2000a) and internationally (IARC, WHO) and with more or less explicit formalism, although mainly qualitative and variable as shown by Weed (2002).

Susser (1991) emphasized the distinction between essential properties of a cause and the criteria useful for deciding on the presence of these properties in a given case. For a pragmatic epidemiology in which all determinants serve as causes, their essential properties were held to be a) association, b) time order, and c) direction, in an ascending hierarchy. Criteria for association are probabilistic and can be enhanced by strength and consistency. Given association, criteria for time order of the relevant variables follow from access to observation, which is dependent on design. Given association and time order, causal direction calls on an array of criteria: consistency and survivability, strength, specificity in cause and in effect, predictive performance, and coherence in all its forms.

Information quality oriented criteria make an important complementary approach. Of these, an important reference is the guidance of USPSTF (1996), classifying I: Evidence from ≥1 properly designed, randomized controlled trial; II-1: well-designed controlled trials without randomization (neither of these applicable to DLCs); II-2: well-designed cohort or case-control studies, preferably from ≥1 centre or group (seldom applicable to DLCs); II-3: multiple time series with or without intervention, or dramatic results in uncontrolled experiments; III: opinions of respected authorities, based

on clinical experience, descriptive studies, or reports of expert committees. Such criteria can be seen as general examples of a study quality emphasis. This can be fused with other sets of criteria, such as those by IARC: study designs; quality of studies; other studies (e.g., non-human animal); plausible inferences about mechanism of action; and other causal criteria (strong association, replication, consistency). The latter include some of the substantive criteria by Hill and Susser, within a methodological framework. These evaluation schemes are important also in the context of intervention evaluation (cf. Rychetnik and Frommer 2000, Rychetnik et al. 2002), of which e.g. the levels and transferability of evidence are relevant in the present context.

A key difficulty in characterizing risks from dioxins and in attributing them to Baltic fish consumption is that **specification of exposures** has been poor in studies among populations consuming such fish. This difficulty is common also in other comparable populations such as Great Lakes fish consumers (see e.g. Anderson et al. 1998 and especially Schell et al. 2001). This applies to both specification of the compounds analyzed and to exposure measures, ranging from proxies in the form of fish consumption frequency and quantity to more specific and informative measures of the actual body burden in representative tissues (cf. 3.4.2, 3.5.3, 4.3.2).

Poor specification of exposures is compounded by poor **specification of effects**. As pointed out e.g. by Neubert (1997-98), there is no clear-cut syndrome or even a set of symptoms characteristic of dioxin toxicity, but instead a suite of non-specific, often broad and usually highly variable adverse or anomalous conditions. This makes it difficult to draw conclusions regarding effects and risks.

Even with improved specification of exposures, a key difficulty is caused by the fact that dioxins and DLCs constitute a complex **array of exposures**. Many of them are correlated, including dioxin-like and non-dioxin-like substances. Distinguishing between their effects will thus require, among other things, that large numbers of subjects be studied (Longnecker et al. 2000) and that exposures are explicitly measured. Effects of other non-TCDD DLCs are extrapolated by TEFs despite their limitations (cf. 5, Annex 8B).

Dioxin effects are **multifactorial**, being influenced by an interacting host of other factors including biological and physical challenges and also balancing mechanisms, internal and external. The difficulty caused by multifactoriality is related to the risk of 'barking at the wrong tree' that is relevant for management in many cases. Also non-specificity of effects is important; it may in comparison be illustrated by whether one should bark at all (or to any direction at all trees).

The lacking specification of exposures and effects jointly produce a network of causal relationships that is complex, overlapping and entangled, conceptually and physically. As pointed out by Van Leeuwen et al. (2000), for the 1-*ortho* PCBs certain endpoints like cancer, thyroid alterations and neurotoxicity could arise by both AhR mediated and other mechanisms, the latter potentially disqualifying these effects and cause-effect relationships as 'dioxin effects', although at the same time potentially adding to the aggregated toxic effects and risks of also DLCs.

Table 14. Criteria for the weight of evidence on cause-effect relationships in epidemiological studies, modified from the criteria by Hill (1965) and Susser (1986, 1991). Cf. the criteria for dioxins listed by Neubert (1997-98) and USEPA (2000a).

Criteria	Specifications and examples	Evaluations particularly for dioxins
Strength of association	OR or RR estimates and their CI's[a]	for most endpoints low OR's, large CI's
Consistency of association	negative or positive (or inconclusive) upon replication	for most endpoints variable
Specificity of effect	misclassification, biological confounders	poor (no clear syndrome)
Specificity of exposure[b]	exposure proxies, total CBs, non-specified PCDD/Fs	usually very poor (mixtures, proxies)
Temporal sequence/time order	exposure prior to effect or diagnosis	ffor chronic exposures/effects often unclear
Biological gradient	dose-response relationship, geogr gradient	often unclear dose-response
Rationale of effect[c]	biological plausibility and mechanistic considerations	considerable support for effect hypothesis
Coherence	no conflict with current knowledge	governs the overall plausibility
Experimental evidence	manipulations, interventions and controls	'gold standard'
Analogous evidence	commensurate other agents	commensurability and generalization limits

[a]OR=odds ratio, RR=relative risk, CI=confidence interval (at a specified confidence level/probability); [b]Not included in the original Hill criteria, may be seen as a subclass of general specificity; [c]Related to more general information-oriented criteria (see text).

The published studies of adverse effects from dioxins in Baltic fish have limited **consistency**. This applies to studies of dioxin effects in general, especially in field populations. Therefore, negative or positive findings may be spurious. Confirmation of results from toxicological effects of DLCs has in general not yet proceeded very far.

For evaluation of the coherence and **biological plausibility** of evidence, the action mechanisms are crucial, especially in the low-dose region. An issue here is the linearity of the dose-response function. Kohn et al. (1994) showed that inclusion of AhR induction by TCDD produced a linear low-dose response of Cyp1a1. This arose from the net effect of sub-linear response of Cyp1a1 mRNA to the concentration of the Ah-TCDD complex and supra-linear response of the protein concentration to the mRNA level, illustrating the importance of biological realism in dose-response modelling. However, other dose-response forms are possible, for some endpoints and systems, and may deviate from linear control by the AhR and the genes (such as Cyp1 enzymes) it affects. **Hormesis** is particularly challenging, also for perception of risks and for policy (Slovic 1998). The assumption of higher risk from a higher dose is implicit in most toxicological reasoning and decision-making about toxicological risks; it is implicit also in the thinking and choices that influence the generation of basic information on both exposures and effects. However, it has been demonstrated by Calabrese and Baldwin (1998, 2003) and Calabrese et al. (1999) that hormesis is a common phenomenon for many substances and effects, including those of TCDD and PCBs, and may be even more common than the default assumptions of a threshold (or linear) dose-response. As pointed out by Rodricks (2003), this does however not deny the existence of a threshold at higher doses.

Hormesis can be seen as one of a variety of factors causing **differential susceptibility** between individuals (Calabrese and Baldwin 2002). The stance adopted to such effects can not be decided on the basis of the prevailing effects and risk assessment paradigm requiring monotonously growing dose-response relationships as an important part of the overall evidence (Susser 1986); it may not be definable on any 'scientific' grounds to the extent as with other effects, and instead hinges largely on the overall conservativeness and precaution towards risks. Mundt and May (2001, ref. by Rodricks 2003) have noted that although hormesis-like dose-response functions have been observed in epidemiology, it is generally not possible to discern whether they reflect true hormesis, but if hormesis is ignored in analyzing such data, relative risks above the hormetic region are systematically overestimated.

Establishing causality is particularly difficult in the area of **nutrition**, due e.g. to the exposure to complex mixtures and to the presence of many often weak dietary factors that excludes the application of classic criteria for causality in many cases (Byers 1999, Potischman and Weed 1999). Causation is still more difficult to resolve with environmental contamination, being present as complex mixtures at variable and usually low doses.

4.1.3 Weight of evidence and data quality

The question of the necessary and attainable weight of evidence for effects is crucial especially in causal inference about the roles of dioxins in such effects. Neubert (1994b) pointed out that the quality of the toxicological data also on DLCs is far below today's standards required for preclinical and clinical data on medicinal substances. He stressed that the quality and predictive power of the data on possible effects of TCDD and other DLCs in humans vary widely, from adequate to unacceptable, but this is often ignorantly or perhaps even deliberately disregarded, and such divergent data are frequently given the same weight in risk assessments. COT (2001) considered that most of the regulatory toxicology studies were conducted >20 a ago and are adequate for determining NOAELs, while recent studies were conducted on non-standard protocols and often also failed to identify NOAELs. A question arises whether for some reasons (e.g. precaution) different quality standards may be applied to the evidence used in assessing risks from dioxins and other compounds.

The relevance of evidence generally increases with the level of biological organization. The limitations *in vitro* methods are particularly important. For some responses and systems, *in vitro* and *in vivo* responses are rather similar e.g. in terms of the relative potency of congeners (e.g., Richter et al. 1997). However, great discrepancies have also been found. For example, Davis and Safe (1991) demonstrated up to 15000-fold differences in the *in vivo* immunotoxicity of a series of

Increasing realism ----------->
Increasing data cost ----------->

Realism increase	Exposure/dose measure	Response measure						
		SAR prediction	Biochem indication	Lab effect, single high-D	Lab, single low-dose	Lab, mix low-dose	Field, nonspecific	Field, actual popul effect
↓	Precursor act.	prescreen						
↓	Emission proxy							
↓	Env level/flux		< ---------- many ecotox tests ---------- >					
↓	Bioavail fraction		< ---------- some ecotox tests ---------- >					
↓	Intake		< ---------- many lab animal tests ---------- >					
↓	Absorbed dose		< ---------- some lab animal tests ---------- >				field study	
	Delivered dose		< ------ lab tests, good exposure/toxicokinetic data ------- >					epidem study, good expo data

Fig. 12. Levels of realism in empirical measures and theoretical models of dioxin risks.

PCDD/Fs but in a mouse *in vitro* model their immunosuppressive potencies were comparable. Also sample pre-treatment may much influence the outcomes of *in vitro* bioassays, such as of AhR binding (e.g., Hu et al. 1995). The weight of evidence of effects increases from *in silico* model predictions to *in vitro* information, and to *in vivo* laboratory, semi-field (captive) and field information (Fig. 12).

A key issue is the **relevance of responses**. The adversity and overall significance, in terms of health outcomes (human and ecological), of many responses observed after exposure to DLCs are unclear. In particular, biochemical responses are often indirect, early, variable markers of effects that may or may not be related and translated to classical toxicological endpoints or other emergent adverse effects of clear physiological, clinical or ecological importance. An ever-greater array of (ever more subtle) responses is measurable. Kim et al. (2002) stated that although the biochemical responses described such as enzyme induction have not been directly linked with adverse health effects, the "body of scientific evidence" relating TCDD-induced effects to events believed to be involved in both cancer and non-cancer processes "warrant concern" for any Margin Of Exposure <10. However, the strength of the evidence on specific responses and their toxicological implications varies, and so will the warranted relative concern for the corresponding MOE. The selection of a benchmark level in the dose-response curve and in the measured population variation (e.g., ED01 or ED10) as well as in the variation in the real population (i.e., the level of protection allowed to deviating high-risk groups) is a matter of judgment (cf. 5.5.1). Moreover, the body of evidence for non-TCDD compounds, including those thought to be of greatest toxicological significance in Baltic fish, is much weaker.

It is commonly held that **epidemiological** studies are the most heavily weighing evidence against which other results must be gauged. This presents an inherent complication as many toxic effects are difficult and some are impossible to establish in epidemiological studies. Specifically, most human studies of effects of DLCs have been small case-control or prospective cohort studies, the power of which is small in detecting effects, while no randomized controlled trials have been made due to the nature of the problem. Thus, the strength of the evidence is considerably less than that commonly required e.g. in studies of beneficial health effects from dietary or therapeutic interventions (cf. 4.4.2).

There is some gap between risk assessments based on experimental animal studies and epidemiological studies. Risks in free-living animals can differ from those in experimental animals. An underestimation of risk based on experimental studies may be common due e.g. to the shorter duration of exposures (Fox and Grasman 1999). Regulatory assessments tend to favour the selection of pivotal studies from animal experiments (cf. 5.5, Annex 8B). A contributing reason is the availability of more detailed exposure information and the possibility to control confounders.

4.1.4 Approaches to effects assessment in the present work

The assessment in this connection is mainly structured according to the various kinds of effects. In accordance with USEPA (2000a), effects on adults and young are in some cases

specified. The various DLCs are treated within this structure, with some differentiation between PCDD/Fs and other DLCs. In later parts of the assessment, complementary divisions and structures are applied.

Emphasis is given to effects studies where

- Effects in **human populations** have been investigated
- Exposures have been at a **low or medium** dose level (background or near background)
- Exposures have occurred to a great extent through **fish**, especially Baltic Sea fish species
- Exposures have been **specified** as to congeners (esp. 4-PeCDF and CB 126) and tissue burdens
- Effects on particular **risk groups** such as early developmental stages are addressed
- **Emergent** (i.e. above biochemical level), severe and non-transient effects have been studied
- **Effect mechanism** considerations have been included
- **Variations and uncertainties** of effects have been explicitly addressed
- Study size and **design** allow resolution of effects
- **Conflicting** but well-founded results are given
- Information on confounders and confidence intervals or other **statistical data** are given
- Risk assessment and management **implications** and links have been indicated.

The USEPA (2000a) reassessment has been largely built upon. It has been augmented by evaluations of later studies and reviews and by additional consideration of inference in establishing effects and attributing them to DLCs. Human studies have been emphasized due to their weight and relevance in the present case. In experimental animal studies emphasis is put on pivotal studies in terms of dose level and effect, for risk characterization. *In vitro* studies are presented when pertinent to a comparative risk assessment focused on human effects.

The assessment and discussion is kept at a rather general level, and illustrating general points by examples. No comprehensive evaluation of effects data is attempted. However, it is necessary to discuss the evidence for effects of DLCs rather extensively and in detail also to discern the relevance and implications of the information specifically on Baltic Sea. Implications and issues in resolving and responding to the quality, significance and limits of knowledge are focused on. Some of the questions related to causal inference will be dealt with in risk characterization (cf. 5).

4.2 Effect types and levels, emphasizing human populations and fish consumption

4.2.1 General considerations

The effects of DLCs include **many kinds** of toxicological responses and endpoints. Their profile is influenced by the compound, the biological system in question and the overall conditions. Partly because the biological basis of the effects is largely shared at the level of receptor-mediated mechanisms, there are close links between the various effects. This is due also to the interconnectedness of biological systems, including regulation mechanisms. Therefore, effects cannot be unambiguously delineated and separated.

Despite such links and overlaps between effects, the following assessment is organized partly according to endpoints. Within this structure, receptors (species and also genders) are in some cases specified. The section dealing with free-living populations is organized by receptor organisms. The latter structure is due to the need to assess the biological and practical significance of effects.

Dioxin effects are multi-dimensional, **complex and partly ambiguous**. They are not clearly definable, single, constant and specific outcomes. Instead, they are in many cases ambiguous syndromes that may be interconnected. They may also be intermediate conditions or modulations of physiology or of biochemical level functions, the significance and adversity of which are not clear.

Effects may appropriately be defined as a **continuum** from AhR binding and other such initial processes, over subsequent modulation of biochemical functions, to emergent individual level and further population and community level effects, in accordance e.g. with USEPA

(2000a). However, as pointed out e.g. by SAB (1995), this continuum is not uniform and does not imply that more emergent and severe effects necessarily follow after initial subtle alterations. Tonn et al. (1996) stressed that impaired immune responsiveness need not have pathological consequences; this is still more true of markers of such responses or alterations. Some biochemical effects such as those on thyroid and retinoid homeostasis are however crucial for many overt biological effects (Simms and Ross 2000).

The effects profile and still more so the relative magnitude of the various effects depend on the **biological system**, involving great variations between taxa, individuals and tissues (e.g., McConnell 1989). Notably, some species and strains are much more sensitive than other species and strains to some effects of dioxins, but may be much less sensitive to other effects. Such dependence on the biological system causes great complexity in the effects of dioxins.

The variation in effects due to the biological system is compounded by variability in the potency and toxicological profile of the various **DLCs**. For instance, fish tend to be less sensitive than mammals to PCBs and particularly 1-*ortho* PCBs, due possibly to different AhR isoforms (e.g. Abnet et al. 1999a, cf. 5.2). Of dlPCBs, toxicological information is available mainly on CB 77 and CB 126, less on CB 169, CB 118 and especially the key 1-*ortho* congener e.g. in humans, CB 156 (as shown e.g. by the 523, 416, 163, 150 and 78 PubMed citations, respectively, using various congener names).

The effects of PBDD/Fs are particularly poorly known (Birnbaum et al. 1991, 2003, Zober et al. 1992). Their potency seems comparable with that of the corresponding PCDD/Fs, and higher to mammals than fish (Mason et al. 1987, Olsman et al. 2005). Weber and Greim (1997) considered that due to the overall similarity in action of chlorinated and brominated DD/Fs, assessments should be based on molar body burdens not discriminating the halogen.

Effect assessment depends on the **mechanisms and modes of action** of DLCs for the effect (and biological system) in question. A key basis of the effects of DLCs is their binding to AhR and subsequent effects in the genes and biochemical entities and processes influenced by AhR, including ARNT and AHRR and mixed-function oxidizing Cyp1 enzymes (Goldstein and Safe 1989, cf. 2.6.1). In addition and subsequent to binding to AhR, the following interrelated effect mechanisms may be noted (cf. Annex 8A):

- DLCs interfere with cellular signalling in regulation of biological processes, including growth and proliferation of cells. **Growth factors** such as epidermal growth factor are key mediators of many dioxin effects. **Cytokines** play an important role as well.
- **Hormonal regulation** is involved in effects of DLCs. In particular, they bind to estrogen receptor (ER). Also other sex hormone related mechanisms such as androgens and mPR (membrane-bound progesterone receptor) may be involved (see Annex 8A). DLCs also influence thyroid functions, in part in association with sex hormone mechanisms.
- Some **other effects** and effect mechanisms may also be important, e.g. direct effects on neural tissue and blood.

The specific implications of the effect mechanisms, pathways and modes of action will depend on how information on them is used in risk assessment. This is reflected in the evaluation of the weight of evidence in causal inference (cf. 4.1.2, 4.1.3). In principle, effect mechanisms may critically influence the **dose-response functions** and thus risk levels. For chemicals that bind reversibly to receptors, some of the assumptions underlying current methods of risk assessment, such as monotonic dose-response curves and existence of a threshold dose, are invalid. The former assumption has been invalidated e.g. with the evidence for hormetic effects of TCDD on immune system (see below). Richter and Vom Saal (2003) also pointed out the implications of the fact that low levels of TCDD can interfere with the stimulatory effect of oestrogen.

Biochemical effects in experimental animals have been generally reported at body burden levels of 3-10 pg TCDD g^{-1} bw (Ma et al. 1992). Specifically, effects on Cyp1a1, Cyp1a2 and Cyp1b1 mRNA and protein are found at 2-15 ppt TCDD in rat liver and 25 ppt TCDD in mouse liver (Gastel 2001), i.e. up to 2 orders of magnitude before the onset of pathology. Based on the preliminary report of correlations by Grassman et al. (2002), Cyp1a1 and Cyp1a2 induction seems to become elevated in human liver at median levels of 90 ppt (lw) and mean levels of 120 ppt (lw) WHO-TEq (including only 0-*ortho* PCBs), presumed to represent typical background exposures. A key question is what, if any, are

the implications of such subtle biochemical level responses to TCDD especially for humans and other free-living animals exposed to mixtures of DLCs. In addition, a difficulty of interpreting such biomarkers especially in field situations is their non-specificity; DLCs are not the only substances and factors influencing Cyp1a levels (e.g. Sleiderink et al. 1995).

The **overt effects** of PCDD/Fs and DLCs observed at lowest doses in experimental animals are disorders of reproductive, neurological and overall development (Brouwer et al. 1995, USEPA 2000a, Birnbaum and Tuomisto 2000). Effects on the immune system are also caused by very low TCDD doses. DLCs modulate multiple hormonal systems (Feeley 1995, Birnbaum 1995a,b, Brouwer et al. 1998a, 1999). Effects on the thyroid may be involved in developmental toxicity (Brouwer et al. 1995, MacLusky et al. 1998). Carcinogenicity of TCDD is pronounced in some non-human animals. Also hepatic and metabolic effects are caused at relatively low doses in many species.

Overt adverse effects occur at generally one order of magnitude higher **body burdens** than biochemical responses, i.e. at 30-70 pg g^{-1} bw in most vertebrates (DeVito et al. 1995, Gray et al. 1998, Van Leeuwen et al. 2000, USEPA 2000a, SCF 2001, Annex 8B). However, neurological effects in rodents at still lower body burdens or doses have been reported (see below). Most of the low-dose developmental effects have been seen in offspring after perinatal exposure. Assuming with Van Leeuwen et al. (2000) simple first-order kinetics of elimination, steady-state conditions, applicability of the TEF concept to pharmacokinetics, and equivalent body burden-effect relationships across species, all of which assumptions may be questioned, the above body burdens are calculated to correspond to a range of human daily intakes of 10-40 pg kg^{-1} bw d^{-1}. This range of effect levels was the basis of the tolerable daily intake (TDI) values proposed by WHO (1998) after application of a compound safety factor of 10 (cf. 5.5.1). The WHO consultation also noted that subtle effects might be occurring in the general human population at current background levels of DLCs, even if not explicitly addressing and including dlPCBs. Some estimates of effect levels, especially in the low end of sensitivity distributions, have been below the most commonly calculated levels given above (cf. Annex 8B).

The evidence from **epidemiological studies** for adverse effects attributable to DLCs in humans is still inconclusive for most effects (e.g., Van Leeuwen et al. 2000, USEPA 2000a). This evaluation essentially depends on the criteria used for evaluating the quality and strength of such studies and in proving causality (cf. 4.1.2, 4.1.3 and see below).

Biomarkers of dioxin effects include a variety of biochemical measures with varying sensitivity, specificity and function and thus variable information content. Induction of detoxifying enzymes, especially of the Cyp1a subfamilies in liver and the related markers such as EROD, MROD and AHH, has been shown in many systems as a result of or in association with exposure to DLCs, both experimentally (e.g., Staskal et al. 2005) and in the field (cf. Annex 8A). However, the specificity and applicability of these enzymes as biomarkers of DLC effects is questionable e.g. in some species of birds (Kennedy et al. 2003a).

4.2.2 Developmental effects

General

Developmental effects, including effects on the developing nervous, immune, reproductive and related hormonal systems and functions (IPCS 2001c, p. 81-2), are commonly considered to be the effects of dioxins that occur at lowest doses (Peterson et al. 1993, Birnbaum 1995a,b, Birnbaum and Tuomisto 2000, USEPA 2000a, SCF 2001). These effects and dose-responses in rodents are particularly well characterized, and have served as the reference point in assessments of human health risks of dioxins e.g. in the EU (SCF 2001).

All main **types** of developmental manifestations (reduced viability, structural alterations, growth retardation, functional alterations) have resulted from dioxin exposure, but the effects at lowest doses mainly include prenatal mortality and functional alterations in learning and sexual behaviour (e.g. Bowman et al. 1989a), and changes in reproductive development in rats (Peterson et al. 1993). Increasing evidence has been published on abnormal development of also structures such as teeth and bones in mammals (see below).

Developmental effects are particularly significant and difficult to elucidate as they may arise from exposures during short and critical early **periods** in development, especially in the initial stages of foetus and during infancy, as shown for rats e.g. by Ohsako et al. (2002). Effects may thus arise in offspring from parental

exposure (cf. Hardell et al. 2004, below). Such effects are not readily tractable in an explicit and quantitative manner within a regular risk assessment paradigm.

USEPA (2000a) concluded that no epidemiological evidence makes a *direct* association between TCDD or TCDD-related agents and effects on human development, but that the accumulated evidence is *suggestive* of such an effect (emphasis added). However, Birnbaum (1995a,b) pointed out that such adverse effects have been detected in highly exposed populations (e.g., Lucier et al. 1990), that it may not be sufficient to look only at overt alterations and disorders, that some studies suggest sub-clinical effects may be present even at or near background exposure at least in part of the population, and that subtle effects may not be readily revealed under normal conditions but only under stress. In humans consuming Baltic Sea fish, some suggestive but no convincing evidence has been published of associations between pronounced developmental disorders and exposure to DLCs, as explained below.

Reproductive development

Effects on the development of **male** reproductive system have been observed in rats at remarkably low doses (Mably et al. 1992a-c), then also in other strains and species (Bjerke et al. 1994, Gray et al. 1997a, Ohsako et al. 2001, 2002, cf. USEPA 2000a, Annex 8B). These effects include delayed onset of puberty, reduction in testis and sex accessory gland weight and penis size, and effects on sperm parameters and sexual behaviour. In hamsters also sperm granulomas were found (Gray and Ostby 1995). Preliminary evidence for testes oedema after comparable exposure of monkey to TCDD has also been published (Sumida et al. 2005).

The lowest effect levels in rats were 50-60 pg TCDD g^{-1} bw single dose, corresponding to a maternal body burden of 0.3-0.4 ng TCDD g^{-1} bw. The body burdens in the offspring in these studies varied from c. 5 pg TCDD g^{-1} ww to >200 pg TCDD g^{-1} ww (Gray et al. 1997a,b, Maruyama et al. 2004). Based on the more relevant continuous dosing study, Faqi et al. (1998) concluded that the LOAEL and NOAEL can be much lower than the estimated daily dose of 0.8 pg g^{-1} bw d^{-1} in the rat, and below the chronic LOAEL and NOAEL estimates of Murray et al. (1979) for reproductive toxicity. However, most of these studies only partially resemble human exposures (e.g., Brouwer et al. 1995). Faqi et al. (1998) also noted that effects on sperm parameters, among the most sensitive endpoints in rats after perinatal exposure to TCDD, remain to be ascertained in humans. The same goes to the similar effects of dlPCBs, especially CB 118, in rats (Pflieger-Bruss et al. 1995). Regarding interspecies extrapolation, it is notable that indications for reduced area of seminiferous tubules have been obtained at a dose of 30 pg TCDD g^{-1} bw in monkeys (Sumida et al. 2005).

In **humans**, male developmental reproductive effects clearly associated with exposure to DLCs have not been reported (but see below other reproductive effects). The finding of Hardell et al. (2004) of increased risk for testicular cancer after exposure of case mothers (not cases) to PCBs, including CB 156, and especially PCB-TEqs, is tentative and requires confirmation, although being supported e.g. by time course and mechanistic considerations and by experimental data.

The most sensitive dose-dependent effects on reproductive development in **female** animals (rats) were morphological alterations in vagina at 200 pg TCDD g^{-1} bw, i.e. c. 4-fold higher than for males (cf. Gray et al. 1997b, Annex 8B). This LOAEL, corresponding with maternal body burdens of c. 90 pg g^{-1} bw and calculated human daily intake of 40 pg kg^{-1} bw d^{-1}, was used in deriving a human TDI in Japan (Sumida et al. 2005). Gray et al. (1997b) found accelerated eye opening already at lower doses (Table 16). In addition, e.g. Muto et al. (2002) and Vorderstrasse et al. (2004) reported effects of CB 126 in rats and TCDD in mice, respectively, on mammary differentiation after perinatal exposure to low doses.

In **humans**, effects of DLCs on female sexual development have been studied in connection with the onset of puberty. Den Hond et al. (2002) found in Dutch girls that a doubling of the serum Bio-Teq level doubled the odds of not having reached the adult stage of breast development. These results however require confirmation (see below reproductive effects).

Bone and tooth development

Abnormal development of **teeth** after perinatal exposure to TCDD has been found in sensitive rats at low doses of 30 pg TCDD g^{-1} bw (Kattainen et al. 2001, cf. Annex 8B). Alaluusua et al. (1993)

hypothesized that such outcomes may be linked to effects on vitamin A (cf. below). In rhesus monkeys, the doses with such effects have been 10-fold higher (Yasuda et al. 2005).

In **humans**, associations have been reported between tooth mineralization defects and exposure in mother's milk to PCDD/Fs (Alaluusua et al. 1996a,b), to dlPCBs and to WHO-TEq_{DFP} (Alaluusua et al. 1999, 2002). Increased prevalence of defects was seen at c. 50-100 pg TEq g^{-1} milk fat in the general Finnish population, in which prolonged nursing has been particularly common. Alaluusua et al. (1996a) assumed that also the high prevalence of tooth mineralization defects in Swedish children born in early 1970's might in part have been caused by the top exposures at that time. This evidence was evaluated critically by USEPA (2000a). Additional evidence however was published on tooth development effects in Seveso where also hypodontia at high doses was observed (Alaluusua et al. 2004). Studies of such developmental effects among consumers of Baltic fish have not been reported.

Effects of TCDD and DLCs including CB 126 on the development of **bone** properties have been studied in rodents (Lind et al. 2000a,b, Jämsä et al. 2001, Miettinen et al. 2003, Stern et al. 2003), also *in vitro* (Natunen et al. 2005). A preliminary study in consumers of Baltic fish (Hagmar et al. 2004a) found no association between such properties and exposure to CB 153 or p,p'-DDE for which Glynn et al. (2000b) reported a weak association (cf. Stern et al. 2004). However, Rylander et al. (2003) observed elevated incidence of osteoporotic fractures in Swedish East coast fishermen's wives, giving indirect support for a role of some Baltic fish contaminants in bone development (cf. 4.3.2). Miettinen et al. (2005) found that effects on rat bone density were mainly reversible. It should be noted that vitamin D, abundant in fatty sea fish, has beneficial effects on bone development (cf. 4.4.2).

Growth

One of the conspicuous effects of TCDD and DLCs in experimental animals is retardation of body weight gain and growth, in an anorectic wasting syndrome. Growth retardation is also often seen in a reduced size of offspring. In humans, low birth weight has been reported in populations with high intake of fish but with inconsistent results (see review by Kimbrough and Krouskas 2001, cf. Annex 8B; see also Karmaus and Zhu 2004, Fristad et al. 2004).

Associations between exposure to PCBs and related compounds and reduced **birth weight** have been reported in several studies among Swedish East coast fishers and their families who had consumed large amounts of fatty Baltic fish (Hardell et al. in Brouwer et al. 1995, Rylander et al. 1995, 1996, 1998). This evidence was considered inconclusive by clinical standards, due e.g. to the lack of consideration of confounders and exposure specification, by Kimbrough and Krouskas (2001); these authors did not account for the report of low birth weight among the offspring of East coast fishermen's sisters by Rylander et al. (2000). Also the comparison of groups stratified by maternal plasma CB 153 levels (Rylander et al. 1998) supports an association. Rylander et al. (1996) pointed out that the period of maternal exposure seems to play a role. However, the results of Rylander et al. (2005) give no clear evidence of long-term (up to 7 a of age) growth retardation. Collectively, these studies support but do not prove a slight association between low birth weight and maternal exposure to contaminants in fatty fish, also PCBs.

Patandin et al. (1998) reported a statistically significant correlation between lower birth weight and *in utero* (but not lactational) exposure to PCBs and dioxins in Dutch children representing the general population, partly consistent with results from studies of laboratory animals and human high accidental exposures. However, decreased birth weight was correlated with ΣPCBs, not TEq_{DFP}. The predicted decrease in birth weight at a level of ΣPCBs of 0.5 ng g^{-1} ww in cord blood that corresponds to 2-fold that in Swedish neonates (Guvenius et al. 2003), as would be expected in children to moderate consumers of Baltic fish (Asplund et al. 1994), was c. 400 g for non-smoking and non-drinking mothers, and 700 g for smoking and drinking mothers. These can be considered high estimates of effect, with a view of the results on heavy consumers of Baltic fish (see above) and on neonates to Great Lakes fish consumers suggesting more moderate effects at higher plasma levels of ΣPCBs (Karmaus and Zhu 2004) or even opposite (weight-increasing) effects (Dar et al. 1992). There are also indications of favourable effects of fish diets on birth weight (Lucas et al. 2004) and head circumference (Helland et al. 2003, cf. 4.4.2).

The specific effect of DLCs on birth weight remains to be established. Also p,p'-DDE and

other contaminants in fish may be implicated (Weisskopf et al. 2005, cf. Karmaus and Zhu 2004). The slight association between decreased birth weight and TEq doses received by Finnish breast-fed boys vanished when analysing only the more comparable primipara infants (Vartiainen et al. 1998) that are also more heavily exposed (cf. 3.5.3). This group may on the other hand not reveal effects from *in utero* exposure (e.g., Patandin et al. 1998). However, the similar lack of association among Seveso neonates (Eskenazi et al. 2003) also suggests that DLCs have only marginal effects on birth weight.

Neurological development

Developmental effects of dioxins after perinatal exposure include the **interrelated aspects** of brain and neurological development, thyroid metabolism and immune development (e.g., Boersma and Lanting 2000, ten Tusscher and Koppe 2004). According to Kakeyama and Tohyama (2003) the evidence in general suggests that dioxins exhibit endocrine-disrupting action on the gonadal and thyroid axes, and 'neural-disrupting action' on transmission and neural network formation.

Such effects have been found in humans after accidental **high exposure** to PCDFs and dlPCBs (Rogan et al. 1988, Chen and Hsu 1994, cf. Aoki 2001). Associations between human neurological development and exposure to DLCs or related compounds in seafood have also been published from the Great Lakes (Jacobson et al. 1985, 1990a,b, 1992, 1999, Jacobson and Jacobson 1996, 1997, Annex 8B) and the Arctic (Grandjean et al. 2001). The Great Lakes evidence has been criticized on many grounds. Usually other contaminants have been involved, and often only ΣPCBs have been measured.

Some associations between suboptimal neurological development and *in utero* or lactational exposure to DLCs (TEqs) have been found also at exposure levels near those in the **general population** (Huisman et al. 1995a, Koopman-Esseboom et al. 1996, cf. Brouwer et al. 1995 and Annex 8B). Koppe (1995) estimated that resultant neurological alterations might have affected 10 % of Dutch children. By contrast, Winneke et al. (in Brouwer et al. 1995) concluded that the (high) Dutch background perinatal exposure to PCBs or PCDD/Fs in infants is not related to a *serious* delay in neurodevelopment (emphasis added). Huisman et al. (1995a) stressed that severe deficiencies were absent and it is premature to advice against breast-feeding, considering its many benefits. Koopman-Esseboom et al. (1997) concluded that 'overt' neurological abnormalities are not caused by direct effects of PCDD/Fs or PCBs or by altered thyroid levels induced by them, but perinatal exposure could still have an impact on neurological development that may result in learning and behavioural problems later. Ilsen et al. (1996) on the other hand found signs of enhanced neuromotor maturation, unlike hypotonia, in 2.5-year old children after perinatal exposure to background TEq levels but no beneficial effects of breast-feeding; they noted that similar effects have been linked with p,p'-DDE. Also Brouwer et al. (1999) concluded that overall the data suggest subtle neurodevelopmental delays in human infants exposed *in utero* to background levels of PCBs (dioxin-like or other). These studies and assessments illustrate the variable findings and equivocal interpretations of associations between low exposures to DLCs (or other contaminants and factors) and neurological development.

Brouwer et al. (1995) summarized the evidence showing that the lowest effect levels for developmental neurobehavioral endpoints based on I-TEq body burdens in animals are within background human body burdens, and subtle adverse effects on neurobehavioral development have also been observed in children exposed to background levels. Lim et al. (2004) likewise concluded that these effects were limiting the margin of exposure for humans (infants). The evidence for neurological and cognitive effects in humans has however been questioned based e.g. on normal variation, confounders, lacking consistence between studies, poor exposure measures and other methodological problems (Middaugh and Egeland 1997, Kimbrough and Krouskas 2001, 2003, Kimbrough et al. 2001, Schell et al. 2001; cf. USEPA 2000a). There is evidence that lowered IQ and other neurological and cognitive deficits in offspring to heavy consumers of contaminated fish may be linked with non-dlPCBs and MeHg (Stewart et al. 2003, ten Tusscher and Koppe 2004, cf. 5.4.2 and Annex 8B). In some studies of neurobehavioral development in Dutch child cohorts, exposure to specifically DLCs was not measured (Vreugdenhil et al. 2002a,b, 2004).

Among consumers of fatty Baltic fish, Rylander and Hagmar (2000) found no effects on the psychometric development of boys exposed perinatally to DLCs and other contaminants

through maternal consumption of such fish (cf. the evidence for thyroid effects, below).

In **experimental animals**, effects on neurological development have been observed at remarkably low TCDD doses in rats and rhesus monkeys (Table 16, cf. Amin et al. 2000, Schantz and Bowman 1989, Schantz et al. 1989, 1992). In monkeys, object learning was impaired in offspring after maternal dietary exposure to 0.15 pg g^{-1} bw d^{-1}, corresponding to a maternal body burden of 20-30 pg g^{-1} bw (cf. Brouwer et al. 1995), and to a human daily intake of c. 15 pg kg^{-1} bw d^{-1}. The monkeys were exposed by continuous low doses and are closely related to humans. A maternal dose level of 20 pg g^{-1} altered neuromotor function in rats; mean ED10 estimates were still lower (Hojo et al. 2002). In mice, Kuchiiwa et al. (2002) found TCDD decreases serotonin-immunoreactive neurons of male offspring at a maternal dose of 5 pg g^{-1} bw, suggesting such perinatal exposure may act as a neuroteratogen.

There is also a body of evidence for neurological effects of **PCBs**, although dlPCBs have been analyzed more seldom (see Rogan and Gladen 1992, Tilson and Kodavanti 1997, Rice 1999, Rice and Hayward 1999, Boersma and Lanting 2000). The focus has been on neurodevelopment and related thyroid and immune effects (see below and Annex 8B). Associations between body burdens of PCBs and memory and learning have been reported in studies among cohorts exposed to Great Lakes fish, both children (Jacobson et al. 1985, 1992, cf. above) and adults (Schantz et al. 2001). Many PCBs and PCB mixtures including non-dioxinlike congeners are neurotoxicants *in vivo* (e.g., Rice 1999), also by mechanisms that are not mediated by AhR. Their neurotoxicity thus masks and blurs that of DLCs.

Other teratogenic and developmental effects

Among structural disorders in development, one of the most sensitive and pathognomonic effects of PCDD/Fs and PBBD/Fs and other DLCs in experimental studies is **cleft palate** and **hydronephrosis** in mice (Birnbaum et al. 1987a,b, 1991, cf. Annex 8B). In humans, Barrow et al. (2002) found no relationship with cleft palate and a marker of the ARNT2 gene related to effects of DLCs.

A preliminary report has been published of a high prevalence of grave congenital malformations in offspring to Vietnamese mothers exposed to TCDD in Agent Orange (Le and Johansson 2001).

Also other renal developmental effects have been reported, e.g. interstitial and peripelvic fibrosis in rhesus monkey offspring after a maternal dose of 300 pg g^{-1} bw (Fukusato et al. 2005).

Effects of DLCs on **skin** development are notable. Chloracne has been often considered the only effect linked conclusively and consistently with exposure to dioxins, at high doses (e.g., Neubert 1997-98, USEPA 2000a, Greene et al. 2003), as highlighted by the poisoning of the Ukranian president in 2005; chloracne has been used as a non-specific marker of exposure to TCDD (e.g., Moshammer and Neuberger 2000). Also other skin effects of TCDD, typically involving hyperkeratinisation, are common in many animals. They have a mechanistic basis in the effects on epidermal growth factors (Berkers et al. 1995, cf. 8A).

In summary, developmental effects of dioxins in non-human animals are found at low doses (ca. 30 pg TCDD g^{-1} maternal bw) in the reproductive system of offspring perinatally exposed. There is also evidence for effects on rat locomotor development after still lower perinatal exposure, depending e.g. on the dose-response modelling and metrics adopted. Biochemical responses, some of which have unclear physiological significance but may indicate emergent adverse developmental effects, are found at even lower internal doses, being within a factor of ten of current background levels in the human populations. In humans mainly effects on tooth and neurological development have been repeatedly found at low exposures to PCDD/Fs, but especially the latter effects have been associated with confounding factors. The associations between PCBs and developmental disorders, including birth weight and neuro-behavioural and immune development, have generally not been confirmed by congener specification. Some indications of abnormal development in human and non-human populations exposed to DLCs through consumption of contaminated fish also in the Baltic have been obtained, but even in these studies the exposure to a mixture of compounds, in addition to other confounders and study limitations, complicates the inference and precludes firm conclusions.

4.2.3 Reproductive effects

Many reproductive effects have been found or suspected to be associated with dioxins mainly in non-human animals. Such effects include reduced fertility and reproductive cycles, sperm parameters, lower hatch rates, altered sex determination and differentiation, and changes in sex hormone homeostasis. Some reproductive effects are caused by low doses of PCDD/Fs or DLCs. Many of these effects have only been ascertained in non-human animals, and for some species and effects even this evidence is inconclusive or has unclear significance. Some effects seen in epidemiological studies on the other hand are related to PCBs, not necessarily dioxin-like. Reproductive effects per definition have great importance both for human health and ecologically. They have thus also symbolic and societal in addition to biological significance.

Reproductive effects of dioxins are in many cases **related to developmental** effects to the degree that they cannot be separated (cf. 4.2.2). This holds e.g. for some effects mediated by sex hormones that are also involved in (early) development. In the following, such effects are dealt with that are directly related to reproductive organs and functions.

In general, no reproductive effects have been regarded as causally linked with dioxins or DLCs in **humans** (e.g., USEPA 2000a, Sharara et al. 1998, Birnbaum and Tuomisto 2000; see also Annex 8B). There is support for such hypotheses mainly from associations between high exposures of young males to TCDD (and related compounds) and sex ratio of offspring (Mocarelli et al. 2000, Ryan et al. 2002, Moshammer and Neuberger 2002, see also Warmerdam and Greene 2002, Tiido et al. 2005). Some of the inconsistencies and gender asymmetries in such reproductive effects may be explained by ovopathy (Jongbloet et al. 2002).

In populations consuming contaminated fish from the Baltic (fishermen and their families on Swedish East coast), inconsistent results of reproductive effects and associations with DLCs have been obtained, due in part to poor specification of exposures and to study limitations (Hagmar et al. in Brouwer et al. 1995, Axmon et al. 2000a,b, 2001, 2002, 2004a,b). This is the case also with studies of such effects from fish consumption in the Great Lakes (e.g., Mendola et al. 1997, Buck et al. 1999, 2000, Courval et al. 1999, McGuinness et al. 2001, Annex 8B). The effects studied include altered menstrual cycles, time to pregnancy, fertility, spontaneous abortion, sex ratio shifts and, most recently, sperm quality.

Rignell-Hydbom et al. (2003, 2004) presented data suggesting an association (formally not significant) between the level of CB 153 and sperm motility among Swedish East coast fishermen as compared to West coast fishermen. Rignell-Hydbom et al. (2005a) also found that exposure to PCBs may have a slight negative effect on sperm chromatin structure in this cohort (cf. Rignell-Hydbom et al. 2005b). Richthoff et al. (2003) likewise found a weak but significant association of plasma CB 153 with sperm motility and biologically active free testosterone in a general Swedish population. None of these studies or that by Dallinga et al. (2002) on PCB metabolites and sperm quality proves the involvement of DLCs. However, the putative effect on sex ratio shift found by Hagmar et al. in Brouwer et al. (1995) is supported by data on increased X chromosome proportion in the sperm of the Swedish East coast fishermen in correlation with PCBs and p,p'-DDE (Tiido et al. 2005), and is linked to DLCs by the epidemiological evidence for TCDD effects on sex ratio (see above and Annex 8B).

Reproductive effects of TCDD and related compounds in **male experimental animals** include decreased spermatogenesis, decreased testis weight and abnormal morphology, and reduced fertility (see above). Suppression of spermatogenesis is not as sensitive in post-weanling animals as in the developing reproductive system, doses of 1 pg g^{-1} d^{-1} being required (USEPA 2000a). Fujita et al. (2002) and Kuroda et al. (2005) reported data indicating that dioxin sensitive genes are involved in sperm and penis development and may be targets of reproductive toxicity of dioxins in humans.

The primary effects of TCDD on **female** reproduction in experimental animals include decreased fertility, inability to maintain pregnancy, and decreased litter size. TCDD also affects ovulation (Petroff et al. 2001). It has been stated (USEPA 2000a) that too exclusive focus has been put on male reproductive effects. In humans the evidence is equivocal. Such effects are not found even after high TCDD exposure (Eskenazi et al. 2002a, 2003, Warner et al. 2004, Warner and Eskenazi 2005). Endometriosis, a common gynaecological condition associated with infertility, has been observed especially in monkeys exposed to DLCs (Rier et al. 1993, 2001a,b, cf. Johnson et al. 1997), also together

with immune alterations (Rier et al. 2001b) and at body burdens potentially relevant for humans (Rier et al. 2001a, Rier 2002, Rier and Foster 2003). Such effects in humans have been considered unlikely based on the primate data (Guo 2004).

In humans, associations between exposure to DLCs and endometriosis were reported for the first time by Heilier et al. (2005) who distinguished between peritoneal and deep endometriotic growth (adenolytic nodules) and obtained OR of 3.3 (95% CI 1.4-7.6) after adjusting for various confounders. The authors evaluated that the lack of association in other studies (Pauwels et al. 2001, Eskenazi et al. 2002b, Tsukino et al. 2005, cf. Annex 8B) may have been due e.g. to the inclusion of deep endometriotic cases in controls.

As to **mechanisms**, most reproductive effects of DLCs seem mediated by the AhR and subsequent interference in transcription. Low-dose reproductive effects of DLCs in rodents involve the hypothalamus-pituitary-thyroid axis but also direct effects on gonads (Bookstaff et al. 1990a,b, Li et al. 1997, Wolf et al. 1999, Kuriyama et al. 2003). It is well documented that DLCs are weakly antiestrogenic (see e.g. Umbreit and Gallo 1988, Safe and Krishnan 1995). They exert estrogenic effects through ER signalling modulated by the AhR (Ohtake et al. 2003, Selmin et al. 2005, cf. Pliskova et al. 2004). Some tissues in turn exhibit a potentiation of TCDD effects by estrogens (see Petroff et al. 2001). Binding on progesterone receptor (PR) is also involved in female reproductive effects (cf. Annexes 8A, 8B). Androgen receptor (AR) and hormones play a role in male reproductive effects. These are associated with and noticeable e.g. in the prostate.

In summary, reproductive effects of various kinds, including hormonal, physiological and behavioural effects in both sexes and in multiple species, have been among the endpoints found at low levels of exposure to DLCs, especially in rats as a result to perinatal TCDD exposure. The safety margin for comparable human effects is small, if equating human body burdens and effective doses in rats. The studies of reproductive health among persons who have consumed high amounts of fatty Baltic Sea fish or other fish contaminated with dioxins and PCBs have produced mainly inconclusive results e.g. regarding effects on fertility. There are indications of sex ratio shifts (feminization) among offspring to heavy consumers of fatty Baltic fish, supported by associations between similar sex ratio shifts and high- to intermediate-exposure to TCDD and also by some mechanistic information. Such effects, if caused by DLCs in Baltic fish, have affected the next generation after the past peak exposure. The evidence is growing also for an association between sperm quality and exposure to contaminants in Swedish fishermen who consumed much fatty Baltic fish, but this does not necessarily implicate DLCs as a cause, and the clinical significance is not clear.

4.2.4 Immune effects

Suppression of immune responses and other alterations in immune system components and functions are among the effects of dioxins that are caused at low doses in experimental animals, as summarized e.g. by Neubert et al. (1994b), Vos et al. (1997-98),and Kerkvliet (2002); see also Birnbaum and Tuomisto (2000). These studies have triggered much controversy; part of it may be due to the fact that the most appropriate approaches have not been applied (Holsapple et al. 1991).

DLCs affect both humoral and cellular immune functions and cause changes in both innate and acquired immunity (Vos and Luster 1989, Holsapple et al. 1991). Effects of TCDD on hemopoiesis have been much studied after reports by Luster et al. (1980) of suppressive changes in bone marrow and immunological parameters of mice, shown later to be mediated by the AhR (e.g., Masten and Shiverick 1996). In addition to T helper cells, attention has been given to effects on B cells (e.g., Nikolaidis et al. 1988a,b). The immune system is intricately linked with hormonal and other regulatory systems and processes, and also with some cancers. Many immune effects are most conspicuous in young animals.

In **humans**, variable and unclear results of immune effects from exposure to DLCs have been obtained (Annex 8B). The slight effects after occupational exposure studied by Neubert et al. (2000) vanished when confounders were included in the analysis. These authors noted that this might indicate a smaller susceptibility of humans than other animals, but underlined

the difficulty of comparison. The decrease of plasma IgG with increasing TCDD level in Seveso was significant also after controlling for key confounders (Baccarelli et al. 2002, Mocarelli et al. 1986). The latter authors stressed that the clinical significance is uncertain despite the evidence for the protective capacity of IgG against microbes, as the IgG levels observed were far above those in patients with antibody immunodeficiency disorders. It was speculated that the finding might reflect a broader alteration of the immune system, as indicated by other studies also in non-human species.

Immune disorders from **high exposures** to DLCs have been reported in offspring of the Ye-Cheng and Yusho populations accidentally exposed to rice oil contaminated by PCBs, including dlPCBs, and PCDFs (Lu and Wu 1985, Annex 8B). Chao et al. (1997) observed significantly elevated prevalence of otitis media at a body burden of 30 pg TEq g^{-1} serum, corresponding with the LOAEL for monkey immunosuppression reported by Neubert et al. (1992b). Yu et al. (1998) found elevated prevalence of influenza and Chang et al. (1982) bronchitis in these subjects. Effects on immune markers were also seen in Koreans exposed to Agent Orange in Vietnam (Kim et al. 2003). Kimbrough and Krouskas (2001) however pointed out that in the Yu-Cheng cohort e.g. the effects of breast-feeding (also beneficial) on developing immunocompetence were not taken into account.

Some results suggesting links between DLCs and immune anomalies (levels of B or T cells) have been obtained in consumers of Baltic fish (Svensson et al. 1994, Hagmar et al. 1995). The former authors found significant negative associations between natural killer (NK) cells and plasma CB 118, CB 126 and p,p'-DDE levels (but not with PCDD/Fs). The latter concluded that the biological relevance of the observed associations with T cell indices may be minor and the causes could not be pinpointed, as also Me-Hg, Se and n-3 PUFAs play a role for immunological conditions. The strongest association was seen for consumption of pike, not a fatty fish species, suggesting other factors than PCDD/Fs or PCBs.

In the Arctic, associations between the frequency of infections and exposure to PCBs and other organochlorides in seafood have been reported (Dewailly et al. 2000; Heilmann 2003, Dallaire et al. 2004). In these studies dlPCBs were not specifically investigated. Reservations as to normal variation, confounders and co-factors, and significance of the findings constraint conclusions from such studies (Kimbrough and Krouskas 2001, see also Neubert et al. 1994b, Neubert 1997-98, Baccarelli et al. 2002).

Also at **near-background exposure**, indications have been obtained of associations between DLCs and immune functions or markers (Weisglas-Kuperus et al. 2000, Van den Heuvel et al. 2002, cf. Annex 8B). ten Tusscher et al. (2003) reported that the immune alterations they observed in the perinatally exposed Dutch children cohort persisted at an age of 8 a. These findings are consistent with many effects found in experimental animals. However, their significance and causes are not clear.

Among **non-human animals**, low-dose immune effects of dioxins have been found especially in sensitive mice (e.g., Clark et al. 1983, Harper et al. 1993, Burleson et al. 1996, Luebke et al. 2002, Vorderstrasse et al. 2003, cf. Annex 8B), but not always reproducibly (Nohara et al. 2002). Immune effects of TCDD have also been reported in rats (e.g., Neubert et al. 1993b, Gehrs and Smialowicz 1999) and marmoset monkeys (Neubert et al. 1992b, 1993a,b), and of PCBs in rhesus monkeys (Tryphonas et al. 1991a).

As reviewed by Vos and Van Loveren (1998), TCDD has been shown to suppress resistance to infectious diseases (cf. Holsapple et al. 1991 and Annex 8B). The relationships between immune responses and host resistence to infectious diseases are however not straightforward and consistent. House et al. (1990) and Burleson et al. (1996) found reduced resistance to mortality from influenza in B6C3F1 mice exposed to only 10 pg TCDD g^{-1} bw. On the basis of their deviating results, Nohara et al. (2002) stressed the difficulty of establishing a LOAEL for effects of TCDD on resistance to influenza.

Extrapolation of such findings to humans and other animals is particularly difficult due to the differences and complexities of immune systems and responses (see e.g. Neubert et al. 1994b, Burleson et al. 1996). There are indications of comparable sensitivity of humans and experimental animals to immune effects of dioxins, especially regarding thymic effects (de Heer et al. 1995, cf. Vos et al. 1997-1998). Vos and Van Loveren (1998) concluded still more confidently that PCBs and related compounds cause immune alterations in humans, and these correlate with the findings in experimental animals. However, the relative susceptibility of different species to different immune responses

and markers remains non-quantifiable, due both to lacking exposure data and to the complexity and variability of effects.

Immune effects pose particular challenges to the **TEF concept** (Neubert 1992a, 1994b, Vos et al. 1997-1998, Levin et al. 2005b). The immunoresponse potency of 4,6-HpCDF in mice can be only one order of magnitude less than that of TCDD (Dickerson et al. 1990). Mayura et al. (1993) found that the immunotoxicity of dlPCBs, especially CB 169 and CB 126, was much closer (even equal) to that of TCDD than with teratogenic effects in male mice. Levin et al. (2005b) reported that PCBs modulated marine mammal phagocytosis *in vitro* in a fashion incompatible with TEFs and differently from mouse cells. Such data suggest that immune effects may not be correctly assessed by standard TEFs, and this may be so also in the Baltic. Van den Berg et al. (1998) noted that the possible greater TEF of 4-PeCDF due to immune effects (Harper et al. 1995a) is particularly important. Burleson et al. (1996) made the point that decreases in several immunological functions, not statistically significant alone, may together result in significant immunosuppression; this may be countered by that also attenuating factors acting in concert can be involved.

The **mechanisms, measures and markers** of immune systems and functions and of DLC effects on them are yet poorly defined (see Annexes 8A, 8B). Several hypotheses have for instance been presented to explain the mechanisms in the reduction of the number of lymphocytes in peripheral blood seen e.g. in monkeys (Riecke et al. 2003). Yao et al. (1995) found that TCDD can activate expression of human HIV-1 gene expression *in vitro* by various mechanisms, stressing the importance such compounds may have in the progression of AIDS, and provided data suggesting two different signal transduction pathways.

Most of the immune effects of DLCs seem mediated by the AhR (e.g., Harper et al. 1993). However, e.g. Nagai et al. (2005) have stressed the pronounced variation in immune effects, and concluded that the presence of many factors modulating AhR function in cell type specific manner is implied. Importantly, Kerkvliet et al. (1990) obtained data suggesting the involvement of an AhR-independent component of immunosuppression by TCDD in mice. Kerkvliet and Brauner (1990) concluded that the lymphocyte subset markers commonly used may not be appropriate for AhR mediated immune effects and that the absence of subset changes does not preclude functional immunosuppression. Kerkvliet (2002) proposed a new paradigm for the mechanism of immunotoxic action of TCDD, moving from a focus on the suppression of immune functions to focusing on the inappropriate activation of cells, leading to anergy or cell death, and the consequent premature termination of the immune response.

Especially with immune effects of TCDD, hormesis is seen in dose-response relationships (Neubert et al. 1992a, Fan et al. 1996). McGrath et al. (1995) stressed the potential importance of other non-linear dose-response forms, such as inertial responses in the low-dose range. The implications of such responses for human effects are unclear but potentially great, although complex and not only increasing risks (cf. 5.2.1, 5.4).

In summary, a growing body of information exists on alterations and also impairments of human immune functions in association with exposure to DLCs even at levels near the background in the general population. This information is complemented by findings of some similar and some dissimilar responses in non-human animals. Some evidence is also available for associations between immune disorders or anomalies in mammals and consumption of fish contaminated by DLCs, including Baltic Sea fish. Much of the above information is as yet uncertain and difficult to interpret. Due to the non-specificity and multiplicity of exposures, the lack of information on natural variability in immune systems and markers, the multifactoriality of these alterations, the poorly known dose-response relationships and the lacking understanding of the impacts (if any) at population level, assessment of resultant human health risks is severely limited. Nevertheless, it seems prudent not to dismiss the information on immune responses from regulatory risk assessment to the present extent, but to take it into account as an additional factor warranting safety and additional research.

4.2.5 Thyroid effects

TCDD and other DLCs have been shown to modulate hormonal systems in many animals. These effects involve both sex hormones and other such as thyroid hormones; it should be observed that these hormonal regulation systems also have interactions. MacLusky et al. (1998) stressed that interactions between different hormone-sensitive systems may contribute to the broad spectrum of responses observed after perinatal exposure, pointing out that interactions between the effects of sex steroids, corticosteroids, and thyroid hormone are known to influence the development of the central nervous system (see above).

Effects on the thyroid gland and thyroid functions such as homeostasis of thyroid hormones and thyroid hyperplasia have been caused by DLCs in **experimental animals** (e.g., Nishimura et al. 2002, 2003, see also Brouwer et al. 1995), and have been found to be AhR mediated (Nishmura et al. 2005a). There is evidence for associations of thyroid effects and exposure to DLCs also in wildlife (see 4.4). Brouwer et al. (1998a) concluded that DLCs are able to disrupt the thyroid system at a multitude of interaction sites, which may have a profound impact on normal brain development in animals including human infants. The evidence in humans is considered inconclusive, observed thyroid effects often being unclear, slight or transient.

Thyroid effects are **connected** with several other regulatory functions, such as growth, metabolism, development, notably neurological, and immune functions (Brouwer et al. 1998b). They are also associated with reproduction and vitamin A states (e.g., Brouwer et al. 1989, Rolland 2000). Hypothyroidism induced by TCDD can be viewed as a protective response to reduce the insult by TCDD, as shown in rats by Pazdernik and Rozman (1985).

In **humans**, usually only slight alterations in thyroid function have been found to be associated with exposures to dioxins, and then usually high exposures except for some studies in breast-fed infants (see below). Pavuk et al. (2003) evaluated published human studies based e.g. on the review by Karmaus (2001). Of the 19 studies that examined thyroid effects of polyhalogenated aromatic compounds, six indicated a significant increase in TSH, five a decrease of T4 or T3, four an increase of T4 or T3, three a higher production of thyroid antibodies, two an increased thyroid volume and only one a higher prevalence of thyroid disease; four found no significant change in thyroid hormone levels. Thus, the collective evidence may be regarded as inconclusive (see also Kogevinas 2001).

Some associations between thyroid states with exposure to dioxins and DLCs have been found also at **low levels** of perinatal exposure (Pluim et al. 1992, 1993, Koopman-Esseboom et al. 1994, Nagayama et al. 1998b, 2005a, cf. Annex 8B). The causes, prevalence, persistence and significance of such effects are unclear. Kimbrough and Krouskas (2001) questioned the strength of the evidence, stressing the lacking account of natural variation and confounders; exposure specification is also a problem. Some effects have been transient and subtle (Ilsen et al. 1996, cf. ten Tusscher and Koppe 2004). As discussed by the latter authors, even transient effects may however be related to disturbances in sex hormone and neurological development.

Among fishermen with high consumption of Baltic fish, Hagmar et al. (2001a) reported that contaminant levels in plasma were not associated with thyroid (or other studied) hormone levels. In contrast, Hagmar et al. (2001b) found among fishermen's wives that plasma CB 153 level was negatively correlated with TT3 concentrations ($p<0.001$) also after age adjustment, giving some support for the notion that dietary exposure to fish contaminants might weakly affect peripheral thyroid hormone concentrations in adult women. In neither study were exposures to dlPCBs, PCDD/Fs or other DLCs specifically measured, and the results are also therefore only suggestive.

The effects of DLCs, mainly TCDD, on thyroid and on thyroid hormone (T3, T4, TSH) states have been studied in many **non-human** species, including monkeys (Brewster et al. 1988a, Van den Berg et al. 1988) and particularly rats (e.g., Ness et al. 1993, Van Birgelen et al. 1995b, Nishimura et al. 2002, 2005b). Results of the magnitude and direction of change in hormone levels have varied. Brouwer et al. (1998b) summarized the evidence as suggesting that pure or mixed organohalogens interfere directly with the thyroid, with thyroid hormone metabolizing enzymes, and with the plasma transport system of thyroid hormones in experimental animals and their offspring. Rolland (2000) concluded that some studies have found adverse health effects in wildlife associated with exposure

to polyhalogenated aromatic hydrocarbons, especially DLCs, and altered thyroid and retinoid status, but a direct causal relationship between these effects and thyroid and retinoid changes has not been demonstrated. Some results suggest however that human and rodent effects particularly of hydroxylated DLCs may differ profoundly (Lans et al. 1994).

Nishimura et al. (2002) found that a single dose of 2 ng TCDD g^{-1} bw in female Sprague-Dawley rats decreased serum TT4 and FT4 levels, while 1 ng g^{-1} bw induced the UGT1 gene that catalyzes the formation of T4-glucuronides. In pregnant Holzman rats, 0.8 ng g^{-1} bw caused hyperplasia of the thyroid of offspring in the study of Nishmura et al. (2003) who also showed that lactational rather than gestational exposure was decisive. No data on body burdens was reported, but it was evaluated that the exposures may have relevance for human populations, despite the evidence for greater susceptibility of rats to thyroid effects.

It is of particular interest that early thyroid function affects **neurological development**. Many studies have postulated anomalous neurological development to be linked with DLCs, PCBs (including non-dioxinlike) and other contaminants (see above). Effects of altered thyroid functions on neurological development have been studied also in association with exposure to contaminants in fish (e.g., Hagmar et al. 2001b), but evidence for clear links with DLCs is inconclusive and their relevance unclear (cf. Annex 8B). Often exposures to total PCBs and their approximate indicators such as CB 153 have been measured instead of specific dlPCBs, PCDD/Fs or other DLCs.

PCB metabolites compound the effects of PCDD/Fs and dlPCBs, and complicate their elucidation. It is unclear what the relative role of dlPCB metabolites is in comparison with those of other PCBs. Based on human *in vitro* and rodent *in vivo* studies, OH-CBs formed from both dlPCBs and other PCBs accumulate in maternal and foetal body compartments and may interfere with thyroid functions (Lans et al. 1993, 1994, Meerts et al. 2002). Maternal exposure during pregnancy results in foetal transfer of hydroxylated metabolites that compete with T4 for plasma transthyretin (TTR) binding sites (Darnerud et al. 1986, Purkey et al. 2004). However, in humans TTR plays a minor role in comparison to thyroid binding globulin (Kimbrough and Krouskas 2001).

All in all, the evidence for effects of DLCs on thyroid hormone states, while reported often also in humans, sometimes at low doses, and having partly well-characterized mechanistic basis and potentially great significance e.g. for neurological development, is variable and difficult to interpret. The extent, causes and implications of such effects among consumers of Baltic fish, especially females and their offspring, also require additional study before firm conclusions can be made.

4.2.6 Carcinogenic effects

Especially since the reports of Van Miller et al. (1977) and Kociba et al. (1978) that TCDD causes malignant tumours in rat liver at low doses, attention has been directed to carcinogenicity of dioxins. Tumours are caused by TCDD and other DLCs in multiple animals and sites, often at doses of c. 1 pg TCDD g^{-1} bw d^{-1}. This is roughly as low as for other effects including biochemical responses (e.g., van Birgelen et al. 1995a). However, it has been regarded that the carcinogenic risk of TCDD in humans is below that for other effects (e.g., SCF 2001, SACN and COT 2004). The extrapolation to humans is questionable and together with tumour type and dose-response function specifications presents a key obstacle in quantitative cancer risk assessment.

Qualitative **classifications** of dioxins as to carcinogenicity have been produced by IARC (McGregor et al. 1998), evaluating TCDD as carcinogenic to humans (group 1) based on limited evidence on workers heavily exposed in accidents, and sufficient evidence in experimental animals (cf. Kogevinas et al. 1997, Annex 8B). The evaluation also considered that TCDD is a multi-site carcinogen in experimental animals, shown to act through a mechanism involving the AhR in humans as in experimental animals, and that tissue levels of TCDD are similar in human populations in which an increased overall cancer risk was observed and in exposed rats that developed tumours. Notably with regard to Baltic fish, other PCDD/Fs were considered non-classifiable as to human carcinogenicity. A re-evaluation (Steenland et al. 2004) reinforced this, pointing out the supportive specific human data for several tumours in high-exposure cohorts (e.g., Warner et al. 2002, Pesatori et al. 2003, cf. Crump et al. 2003), for melanoma also in the Ranch Hand cohort that had a mean blood

level of 50 ppt TCDD lw in late 1980's (Akhtar et al. 2004) as compared to over 200 ppt TCDD in occupationally and accidentally exposed key cohorts (and to over 200 ppt TEq in top Baltic fish consumers). Akhtar et al. (2004) and Pavuk et al. (2005) also found increased incidence of prostate cancer in this cohort.

Even though TCDD and possibly (even probably) other DLCs seem carcinogenic also to humans, the **potency** depends e.g. on type of tumour and has varied between studies. Fingerhut et al. (1991) did not confirm the high risk ratios (RR) reported in many previous studies in US workers. Typically RR or odds ratio (OR) values of <2 with confidence intervals starting around 1 have been obtained in epidemiological studies. Steenland et al. (2004) evaluated that the inclusion of lower exposures in these cohorts reduces the uncertainties involved in extrapolation from high doses, that the cohort-internal comparisons are unlikely to be affected by confounders, and that positive dose-response relationships support causality. On the two first mentioned points their judgment however seems overly confident. Yamaguchi (1999) showed that there is an appreciable chance that such elevations in cancer risk as found for TCDD are caused also by random factors, concluding that estimation of the absolute cancer risk in human populations should be made with caution. If applying strict criteria on the quality of studies (e.g. nested case-control designs, specification of individual body burdens), many of the claims for an association of cancer with PCDD/Fs and PCBs are considerably weakened. Also the direction of association between exposure to TCDD and some types of tumours is inconclusive, e.g. as regards breast cancer (see Warner et al. 2002 and below).

The **mechanisms** of dioxin tumorigenesis have important bearing on quantitative cancer risk assessment through the selection of dose-response models. It has generally been agreed that TCDD and other dioxins are cancer promoters, but not initiators (e.g., McGregor et al. 1998). This has been based also on their apparent lack of genotoxicity. Thus, the assumption of a no-threshold dose-response function and use of the common default linearized multistage model have often been deemed inappropriate. The more precautionary criteria used by USEPA (2000a) led to estimates of the human cancer risk that differ greatly from those of other agencies (Hays and Aylward 2003). Information on the mechanisms of the tumorigenicity of DLCs and its links with the AhR complex, cytokines and growth factors and overall responses to cellular stress (Matsumura 2003) may clarify risks, and molecular epidemiology may reveal factors of susceptibility to cancer in humans (Landi et al. 2005). The effects of DLCs e.g. on gap junctional intercellular communication seem related to tumour promotion (Warngard et al. 1996).

Another fundamental issue in connection with mechanisms of tumorigenesis is the applicability of results of **animal models** to humans. This affects already the definitions of tumours to be considered, and interspecies dose-response generalizations. The fact that TCDD is a carcinogen in many species and the evidence for phylogenetically conserved sites and mechanisms of action has strengthened the basis of extrapolating between species (McGregor et al. 1998, cf. above). This however does not resolve whether and what interspecies extrapolation more precisely is appropriate (cf. 5.2.1).

Uncertainties in quantitative cancer risk assessment based on animal studies can be illustrated by the re-evaluation of the histopathological findings of Kociba et al. (1978) by a panel using revised NTP criteria and finding 2/3 fewer tumours (Goodman and Sauer 1992). As shown by the latter authors, this changed potency estimates by an order of magnitude, as the NOAEL for hepatocellular carcinomas was 10 and not 1 pg g^{-1} d^{-1}. Further, the aggregation of tumour forms had a major impact; estimates of risk-specific dose (RsD) at 10^{-6} risk level based on the original histopathology criteria were 10 pg g^{-1} d^{-1} combining carcinomas and hyperplastic nodules, 150 pg g^{-1} d^{-1} when only carcinomas were considered. On the revised criteria, RsD was 80 pg g^{-1} d^{-1} when adenomas and carcinomas were combined and 25 000 pg g^{-1} d^{-1} considering only carcinomas. As the Moolgavkar-Venzon-Knudson two-stage model is intended to predict malignant tumours only, the appropriate human RsD was estimated at 25 000 pg g^{-1} d^{-1}. As the model does not account for e.g. pharmacokinetics, the RsD was judged to be higher still (Paustenbach et al. 1991). The plausible ranges for RsD based on original and re-evaluation data were 70-3000 and 100-50000 pg g^{-1} d^{-1}, respectively. Even the lowest plausible RsD was thus 10-fold greater than the USEPA RsD of 6.4 pg g^{-1} d^{-1} based on the linearized multistage model, and the RsD could well be more than 1000-fold greater.

Dose conversion from rats to humans based on skin surface area was also critically examined by Keenan et al. (1991). Using body weight, a more appropriate basis, but lumping tumours and settling for the linearized multistage model, the re-evaluated histopathology data produced carcinogenic potency estimates at least 16 times less than originally estimated.

Evaluations of the carcinogenic potency of TCDD (and, by extension, TEq) have additionally been based on meta-analyses of **occupational** cohorts. Crump et al. (2003) found a statistically significant ($p=0.02$) trend in total cancer mortality with increasing dioxin exposure in three cohorts. Trend tests showed an increase in total cancer at cumulative TEq serum levels resulting from lifetime intake of 7 pg TEq kg^{-1} bw d^{-1}; a linear dose-response model fitted well to the combined data and predicted an ED(01) of 45 pg TEq kg^{-1} bw d^{-1} (95% CI, 20-300). Starr (2003) however examined the USEPA and Crump et al. (2003) meta-analyses that concluded "dioxin TEq exposures within roughly 3-fold of current background levels may be carcinogenic to humans" in the light of an alternative meta-analysis using an intercept-only model that predicted zero additional human cancer deaths from all exposures to DLCs, including those arising via dietary intake. He identified causes for the discrepancy in a) different selections for a dose metric, b) different assumptions regarding the elimination half-life for TCDD in humans and regarding the importance of the most recent (15 a) period of exposure, and c) extrapolations from TCDD to TEq exposures. He presented several arguments against such an extrapolation, in line with the IARC position that, unlike TCDD, other dioxins cannot be classified as to their human carcinogenicity.

Boroush and Gough (1994) noted that if dioxins are associated with a common tumour, the increase in cancer risk would go undetected against the already high background incidence, and that also cancers of lower incidence can be impossible to detect by cohort studies. They concluded that the failure of epidemiology to produce convincing evidence of cancer from dioxin probably results from the generally relatively low exposures in humans.

Since the no-threshold **dose-response** does not seem appropriate due e.g. to lacking genotoxicity, the linearized multistage model or its modifications are not strongly supported. Alternatives include two-stage (cf. above) and one-hit models. Among non-linear models, also those for hormesis (lower response at higher than at lower doses in some part of the dose-response curve) have been considered. Such response may result e.g. from detoxifying or compensating mechanisms setting in only at higher doses (see Rozman and Doull 2003). Indications for hormesis have been obtained for carcinogenesis (cf. above). Tuomisto et al. (2004a) reported a higher risk at lower than at higher doses of PCDD/Fs and dlPCBs in human populations for soft-tissue sarcoma.

Bonvalot et al. (1987) showed that estimates of the dose corresponding to 10^{-6} excess cancer (background) risk of TCDD in rats based on the key chronic data sets of NTP (1982) and Kociba et al. (1978) and using only those dose-response models not rejected by goodness-of-fit tests varied by a staggering 22 orders of magnitude, and by 3 orders of magnitude even using only validated one-hit and multistage models of carcinogenesis. They also showed that the no-threshold linearized models, generally considered the most conservative at low doses, appeared the least conservative. It was concluded that the impact of the various assumptions regarding dose-response functions was even more important than previously proposed.

Kayajanian (2002) argued that, contrary to Fingerhut et al. (1991), the dose-response is J-shaped as in the two major data sets that these authors "failed to reference or explain away", and concluded that even though incidence may increase at high exposures, this is preceded at lower exposures by a reduction. In fact, instead of J-shape a two-peak ('N-shape') curve seems plausible as the excess cancer response at zero (excess) dose may be assumed to be zero. Kayajanian (2002) concluded, using epidemiological data by Fingerhut (1991), Kociba et al. (1978) and Bertazzi et al. (2001), that dioxin is (1) a promoter blocker of certain cancers, including all those that "USEPA scientists claimed dioxin promoted"; (2) a promoter of some other cancers that "USEPA scientists failed to identify"; and (3) a net anticarcinogen. These analyses and discussions again illustrate the need to distinguish between tumour types and sites in evaluating the cancer risk of dioxins, and the needs and pitfalls in statistical analysis and dose-response models especially in low-dose extrapolation.

Against the above considerations, the report of Svensson et al. (1995b) of slightly elevated

incidence of stomach and squamous cell cancer (SIR=1.6, 95 % CI=1.0-2.4) and of myelomas (3.1, 1.2-6.4), among Baltic fishermen as compared with regional referents is regarded as suggestive only, even for fatty Baltic fish in general. A link specifically with DLCs is still harder to establish.

TCDD and some other DLCs have in some conditions been shown to be anticarcinogenic mainly against oestrogen-conditioned tumours (Gierthy et al. 1993, Holcomb and Safe 1994, Ramamoorthy et al. 1999, see also Cohen et al. 1979 and Oenga et al. 2004). There is on the other hand some evidence from human studies of a positive association between breast cancer and 1-*ortho* PCBs (e.g., Warner et al. 2002, Demers et al. 2002 and the references therein), but also much conflicting evidence; collectively these studies may be regarded as inconclusive (see Annex 8B). The same is true of associations with PCDD/Fs, 0-*ortho*-PCBs and other DLCs.

All in all, TCDD is a (non-genotoxic) carcinogen in many animals and tissues, particularly in rodents and probably also in humans. For many tumours the human evidence is inconclusive and the extrapolation from laboratory animals is difficult despite some similarities in underlying mechanisms. Other DLCs cannot yet be classified as to human carcinogenicity, but some are established animal carcinogens. The evidence for the carcinogenicity of PCBs seems weaker than that for PCDD/Fs, possibly in part due to poor specification of exposures to congeners belonging to different groups. The carcinogenic potency of TCDD does not seem very strong in humans, judging from epidemiological data and plausible dose-response relationships that are however constrained by great uncertainties in dose (and response) metrics, models of carcinogenesis and statistical treatment of the data.

4.2.7 Metabolic effects

TCDD and many other DLCs have been found to influence metabolism in many ways. Some of these effects such as **wasting** (loss of body weight) are among the hallmark effects especially in experimental animals. Metabolic effects have been reported also in humans, although tentatively.

Effects on vitamin homeostasis are found at low doses especially for **retinoid (vitamin A** and metabolites) in several animals including rodents (Håkansson and Hanberg 1989, van Birgelen et al. 1995a,b), wild mammals such as seals (see Simms and Ross 2000 and 4.3), and non-mammalian vertebrates (cf. Annex 8B). The close involvement of the retinoid system in the overall toxicity of DLCs has been supported by many studies (e.g., Fattore et al. 2000). Vitamin A is crucial e.g. for normal development of immune functions, reproduction, vision, skin and growth. It also plays a role in immunocompetence (e.g., Semba et a. 1992, 1993), which may be important considering immunosuppressive effects in Baltic fish consuming animals. On the other hand, also non-AhR mediated retinoid effects take place, caused e.g. by non-dlPCBs, and the normal variation and determinants of retinoid homeostasis are poorly known in many species (Simms and Ross 2000).

Effects on **vitamin K** states deserve attention as well. They are affected in rodents at doses of c. 100 ng kg^{-1} bw (Bouwman et al. 1999) and have potential significance for blood coagulation in neonates.

Effects of TCDD and DLCs on **lipid** metabolism including cholesterol and lipoprotein lipase are seen in experimental animals. Brewster and Matsumura (1989) showed that profound differences occur in lipid metabolism between various species in response to TCDD and these changes do not appear to be related to generalized toxicity. Such effects of DLCs have been suspected also in humans. These effects are important and complex partly because DLCs accumulate in fats, and changes in lipid metabolism may profoundly alter exposure levels.

DLCs affect **glucose** metabolism and neoglucogenesis. There has been particular interest in links between diabetes and dioxins, based on modest evidence from studies mainly in high-exposure groups (e.g., Calvert et al. 1999, Bertazzi et al. 2001, Annex 8B). The reasons and implications of such links are not clear, as there are many other causes of diabetes, including obesity and overall fat metabolism. Higher levels of DLCs in diabetics may be a consequence, not a cause of diabetes (e.g. Remillard and Bunce 2002, cf. Diliberto et al. 2002). However, as pointed out by Longnecker et al. (2001), even if diabetes would be the cause of higher blood levels of PCBs and not *vice versa*, subjects with increased blood

levels of DLCs could be at a higher risk for other disorders caused by these substances.

Among biochemical effects, those on **enzyme** functions are prominent. In particular, increased induction of hepatic Cytochrome P4501 (Cyp1) subfamilies and of AHH and drug-metabolizing monooxygenases such as EROD occur in most animals, often at low doses. Also effects on other enzymes including those involved in cell energy metabolism are common (cf. Annexes 8A, 8B, 8D). In addition, DLCs affect many other metabolic processes and their markers, such as the hallmark accumulation of porphyrins due to disturbances in heme biosynthesis (e.g., van Birgelen et al. 1996a).

In summary, metabolic effects from exposure to DLCs, some of them well established and non-specific such as wasting, are seen in several species. They are closely related to effects on growth and development. Effects on diabetes have been suspected but despite tentative epidemiological findings and experimental and mechanistic information have not been confirmed at low levels of exposure. Effects on lipid metabolism are seen in non-human animals and are possible in humans but their significance is unclear; they complicate dioxin effects by affecting fat stores. Effects on vitamin states are common, and effects on vitamin A are observed at relatively low doses. The same applies to induction of mixed-function oxidation enzymes and to effects on many other biochemical mechanisms in metabolism, but these are not necessarily associated with significant outcomes.

4.2.8 Cardiac and cardiovascular effects

TCDD and related compounds cause adverse cardiac and pulmonary effects in many animals such as rodents (Brewster et al. 1987, 1988b), monkeys, chickens and fish (cf. 4.3.2). In non-mammalian groups most cardiac effects are related to subcutaneous oedema. However, in chicken the heart muscle itself seems also to be a direct target of TCDD toxicity (see Walker et al. 1997) and also cardiac deformation occurs (e.g., Walker and Catron 2000, Sommer et al. 2005, cf. Annex 8D).

In **non-human** mammals, Riecke et al. (2002) reported dose-dependent increase of collagen deposition in cardiac tissue in marmoset monkeys after single doses roughly corresponding to average human body burdens (Annex 8B). Jokinen et al. (2003) found that 2-year exposure to TCDD or CB 126 at doses of 100 and 1000 pg g^{-1} bw d^{-1}, respectively, caused cardiovascular lesions in rats in a dose-related manner. There is also evidence for a role of PCBs in potentiating cardiovascular diseases through proinflammatory effects (Hennig et al. 2002). Hennig et al. (2005) presented data indicating that CB 77 as well as CB 153 induces inflammatory pathways in endothelial cells.

In **humans**, associations have been found between exposure to TCDD and other DLCs and impairment of cardiovascular health from high-exposure situations e.g. in Seveso (Bertazzi et al. 2001) and in some occupational settings (cf. Annex 8B). The latter commonly involve exposure to also other contaminants. In accidental exposures, psychological stress may have contributed (see below).

The risks of cardiac or pulmonary diseases due to DLCs in fish need to be assessed in relation to the benefits for cardiovascular health from consumption of fish (cf. 4.4.2, 5.4.4).

4.2.9 Psychosomatic and non-biological effects

It has been pointed out (e.g. Pesatori et al. 1998) that some of the apparent cardiac effects of dioxins in Seveso may be due largely to anxiety. On the basis of the Australian Evatt Royal Commission assessment of possible associations between health effects and exposure to dioxins (TCDD) in Vietnam veterans, Hall (1986) noted that although they had slightly higher rates of psychiatric disorders (among other disorders), these effects were unconnected with such exposure.

It is a matter of debate in how far such responses are actual dioxin effects or are caused by other factors, thus being indirectly related or secondary effects (if at all related). Psychosomatic effects are however real to those experiencing them, and should not be dismissed as imaginary. This would be unjustified also medically, as there are links and overlaps between somatic and psychological aspects. In general, the importance of risk perceptions and emotions needs to be realized in risk management.

On the other hand, the distinction also needs to be often made between primarily

Table 15. Summary evaluation of evidence for adverse effects in human populations linked with dioxin-like compounds, with particular reference to consumption of fatty fish from the Baltic Sea and low-to-medium-level exposure. Critical effects on the basis of relevance, severity, exposures and evidence are indicated. Supporting evidence implies information supporting the hypothesis of effects caused by DLCs. Cf. text and Annexes 8A, 8B.

Effect/condition	Populations studied[a]	Exposures specified and measured/inferred[b]	Effects or conditions studied and found[c]	Evidence for DLC-attributable human effects; effect or risk level[d]	Supporting evidence[e]		
					hi-D	non-hum	mech/ plaus.
Developmental							
Neurological/ cognitive development	SW **BS fish** eaters[1] NL children[2] Gt Lakes fish eaters[3] Gt L. fish eaters, old[4] Faroese children[5] US children, general[6]	perinatal fish, CBs perinatal CBs, D/Fs CBs/OCs, MeHg CBs/OCs, MeHg whale CBs, MeHg perinatal CBs, DDE	psychometric **motor function** learning, memory learning, memory subtle cognitive infant motor function	none (no effects found) modest; lactation protected inconclusive; expos. proxies suggestive; unspecif. expos. inconclusive (interactions) weak; lactation protected	 (x) (x) (x) 	 x x x (x) x	 (x) x (x) (x) (x)
Behavioural	NL children[7]	milk CBs, D/Fs	feminized play	weak/subtle, inconclusive		x	x
Bone and tooth development	SW BS fish eaters[8] FI children[9]	CBs, DLCs, DDE lactat. D/Fs (CBs)	bone densit/osteopor. **tooth devel**/mineral.	inconsistent; expo proxies suggestive	 (x)	x x	(x) x
Growth incl. fetus development	SW **BS fish** eaters[10] Gt Lakes fish eaters[11] NL children[12] FI children[13]	fish DLC etc DDE, fish DLC *in utero* CBs, D/Fs Imother's milk D/Fs	low birth weight birth weight, size birth weight birth weight	tentative, inconclusive inconclusive inconcl. (ΣPCBs correl) slightly lower, not in primipara		x x x	(x) (x) (x)
Reproduct. devel	NL girls[14]	DLCs/Bio-TEq	delayed puberty	tentative	x	x	x
Reproductive							
Fertility and fecundity	SW **BS fish** eaters[15] Gt Lakes fish eaters[16]	fish DLCs etc fish DLCs etc	time-to-pregn./fertility low fecund. (matern.)	inconclusive; fish protective? inconclusive/conflicting		x x	(x) (x)
Menstrual cycle	SW **BS fish** eaters[17] Gt Lakes fish eaters[18]	fish CBs, D/Fs etc fish CBs etc	slight shortening slight shortening	tentative; causes unclear tentative		x x	(x) (x)
Sex ratio (F/M) of offspring	SW **BS fish** eaters[19] FI general[13]	CBs, D/Fs, DDE etc D/Fs	**slightly higher F/M** no change found	weak, tentative - (child. to men not specified)	x x	(x)	(x,gen)
Sperm quality	SW **fishermen**[20] SW general popul.[21]	CBs, D/Fs, DDE etc CB 153	sperm motil., DNA sperm motility	slight, tentative; expo proxy tentative; expo proxy		x x	x x
Endometriosis	BE women[22] JPN women[23]	dlPCBs, D/Fs DLCs (Bio-TEq)	**peritoneal/deep endom.** nonspecified endom.	OR=3.3 (1.4-7.6), suggestive no associations		x (x)	x x
Sex hormones	Gt Lakes fish eaters[24] SW general popul.[25]	CBs, DDE CB 153	SHBG-bound testost. lower testosterone	suggestive (not free testost.)slight; expos. proxy	(x) (x)	x x	x x
Immunological							
Immuno-suppression	SW **BS fish** eaters[26] NL children[27]	CBs, D/Fs, DDE (co)CBs, D/Fs	NK, T cells cell/humoral markers	inconcl. (also link with D/Fs) supportive, inconclusive	(x) (x)	x x	(x) (x)
Host resistance	NL child/adolescents[28] BE adolescents[29] Inuit/Faroese child.[30]	CB, D/Fs (Bio-TEq) DLCs/Bio-TEq CBs, MeHg	airways infect. etc NK cells, IgA, IgEs ear etc infections	tentative tentative; infections unrelated tentative	x x x	x x x	(x) (x) x
Carcinogenic							
Stomach cancer	SW **BS fish** eaters[31]	CBs, D/Fs etc	small elevated RR	tentative; expos. proxies			(x)
Skin cancer	SW **BS fish** eaters[31]	CBs, D/Fs etc	small elevated RR	tentative; expos. proxies	(x)		(x)
Hormonal, metabolic and other effects							
Thyroid function	SW **BS fish** eaters[32] Gt L fish eaters[24] NL children[33] JPN children[34]	DLCs etc (non-spec.) CBs, DDE etc CBs, D/Fs	altered T3 altered T3, T4 **T3/T4 level, dysfunct** T3/T4 levels	slight, tentative (female only) tentative (mainly female) modest, inconcl., transient supportive	x x x x	x x x x	x x x x

References and explanations: *[a]NL=The Netherlands, SW=Swedish, BS=Baltic Sea, FI=Finnish, BE=Belgian, JPN=Japanese; [b]CBs=PCBs, D/Fs=PCDD/Fs, OC=organochlorides, MeHg=methyl mercury, DLC=dioxin-like compounds, TEq=TCDD equivalent; [c]SHBG=sex hormone binding globuline, NK=Natural killer cell, IgA=immunoglobulin A, IgE=immunoglobulin B, T3=triiodothyronine; T4=thyroxine; [d]OR=odds ratio; RR=risk ratio, nat=inherently; [e]gen=genetic basis Hi-D= high-dose; [1]Rylander & Hagmar 2000; [2]Huisman & al. 1995a, Koopman-Esseboom & al. 1996, Ilsen & al. 1996, Vreugdenhil & al. 2002a, 2004; [3]Jacobson & al. 1985-1992, Jacobson & Jacobson 1996; [4]Schantz & al. 2001; [5]Grandjean & al. 2001, cf. Despres & al. 2005 on lacking effects on neuromotor development among Canadian Inuits; [6]Gladen & al. 1988; [7]Vreugdenhil & al. 2002b; [8]Glynn & al. 2002b, Rylander & al. 2003; [9]Alaluusua & al. 1996a,b, 1999, 2002; [10]Rylander & al. 1995, 1995, 1996, 2000; [11]E.g., Dar & al. 1992, Karmaus & Zhu 2004; [12]Patandin & al. 1998; [13]Vartiainen & al. 1998; [14]Den Hond & al. 2002; [15]Axmon & al. 2000b, 2001, 2002; [16]Courval & al. 1999, Buck & al. 2000, McGuinness & al. 2001; [17]Axmon & al. 2004a; [18]Mendola & al. 1997; [19]Hagmar & al. in Brouwer & al. 1995, Tiido & al. 2005 (on X chromosome proportion shift); [20]Rignell-Hydbom & al. 2003, 2004, 2005a,b, cf. results in other high-dose cohorts, esp. Mocarelli & al. 2000; [21]Richthoff & al. 2003; [22]Heilier & al. 2005, cf. Pauwels & al. 2001, De Felip & al. 2004; [23]Tsukino & al. 2005; [24]Persky & al. 2001; [25]Richthoff & al. 2003; [26]Svensson & al. 1994, Hagmar & al. 1995; [27]ten Tusscher & al. 2003; [28]Weisglas-Kuperus & al. 2000; [29]Van Den Heuvel & al. 2002; [30]Dewailly & al. 2000, Heilmann & al. 2003, Dallaire & al. 2004; [31]Svensson & al. 1995b; [32]Hagmar & al. 2001b, cf. Hagmar & al. 2001a, Pavuk & al. 2003; [33]Pluim & al. 1992, 1993, Koopman-Esseboom & al. 1994, Ilsen & al. 1996; [34]Nagayama & al. 1998b, 2005a.*

psychological effects and those depending on biological processes. An effect has in some respects a different significance if it is caused even to persons unaware of it (e.g. infants), as compared to effects that are (even if partly) due to such awareness, or fear. The distinction is important also for causal inference: it can be difficult to know what the relative contribution of dioxin exposures is to effects, when complex socio-psychological mechanisms blend with biological phenomena. Some psychological effects may be seen as confounders and error sources, depending on study purpose and design (e.g., recall bias).

4.2.10 Summarizing evaluation of relevant human and experimental animal data

A summary of the evaluated reports of effects as a result of low or intermediate level exposures to DLCs is presented for humans (Table 15) and for experimental animals (Table 16).

DLCs exert a variety of adverse and other biological effects in many species and in both sexes, generally in patterns that have much resemblance. However, the effects are multiattribute and variable. Although some effects such as those on reproductive development in rodent male offspring and some tumours have been extensively characterized,

Table 16. Summary of important experimental studies of lowest adverse effect levels of body burdens and effective dose estimates for TCDD and dioxin-like compounds. Emphasis is on well-documented chronic low-dose studies involving distinct, consistent and severe or persistent endpoints with plausible biological basis, and pivotal studies underlying recent authoritative assessments or surpassing these. Cf. Annex 8B.

Effect (gender, M=male, F=female), pnd=postanatal day	Species/ strain	Dosing			LOAEL body burden, pg g^{-1} bw (critical period)		Effective dose and benchmark dose estimates[c], pg g^{-1} bw d^{-1}		
		Route[a]	Level/range pg TCDD g^{-1} bw	Timing[b], duration	Adult	Off-spring	ED_{01}	BMD_{01}	BMD_{10}
Reproductive development									
M spermatogenesis[d]	rat	scn	25+n5[e]	gd-14, -7, 0	33 (18+11+3)[f]				
M spermatogenesis[g]	rat/L-E[h]	orl	50	once, gd15	c. 30		10 (0.1)	60 (6)	na
M spermatogenesis[i]	rat/Holzman	orl	64	once, gd15	38 (from 50)		0.6 (0.3)	8 (4)	50 (30)
M spermatogen, pnd 63[j]	rat	orl	50-2000				**0.3 (0.1)**	**3 (1)**	**20 (9)**
M prostate wght, pnd 49[j]	rat	orl	50-2000				0.5 (0.1)	6 (1)	40 (10)
M anogenital morphol[k]	rat/Holzman	orl	13, 50, 200, 800	once, gd15	c. 30	**(5)**[l]			
F vaginal morphology[m]	rat/L-E	orl	50, 200, 800		(from 200)	(13-40)[l]			
Neurological development/behavioural									
F accelerated eye open[n]	rat/LE	orl	50, 2000, 800		(50 admin D)	**(5)l**			
Spatial learning[o]	rat/S-D	orl	25, 100	daily, gd6-10	(25 admin. D)				
F/M neuromotor[p]	rat/S-D	orl	20, 60,180	once, gd8	**20 (ED10=3)**		**0.3 (0.1)**		
F neuromotor[q]	rat/Holzman	orl	20, 60, 180	once, gd18	**20 (ED10=7)**		0.7 (0.6)		
Object learning[r]	monkey	orl	c. 0.15 ppt bw d^{-1}		(20?)				
Other developmental									
Teeth (3rd molar block)[s]	rat	orl	30	once, gd15	(30 admin. D)				
Immunological									
M offspring DTH[t]	rat/F344	orl gav	100, 3000, 1000,	once, gd14	100				
Lymphocyte changes[u]	marmoset	orl diet	0.3 ppt bw wk-1	wkly, 30 wk	9-10				
F splenic PLCs/10^6 cell[v]	B6C3F1 mice	orl	5-3000				2 (1)	10 (9)	70 (50)
Total thymic cells[w]	mouse	orl	100-5000				7 (0.7)	20 (5)	50 (20)

References and explanations: *[a]scn=subcutaneous; orl=oral; [b]gd=gestational day; [c]Mean and (in parentheses) lower 95 % confidence limits; ED01 estimates from USEPA 2000a, BMD estimates from Gaylor & Aylward 2004; [d]Faqi & al. 1998; [e]Loading dose + maintenance doses; [f](Pseudo)steady-state contribution from loading dose and first maintenance doses + contribution from last unequilibrated maintenance dose + contribution from feed, cf. COT 2001; [g]Gray & al. 1997a; [h]Long-Evans Hooded; [i]Mably & al. 1992a-c, maternal body burden estimated by SCF 2001; [j]Hamm & al. 2003; [k]Ohsako & al. 2001; [l]Estimated from data of Hurst & al. 2000b; [m]Sumida & al. 2005 (preliminary report); [n]Gray & al. 1997b; [o]Schantz & al. 1996a, no body burden given but 5x25 pg g^{-1} bw dose caused significant effect; [p]Hojo & al. 2002, based on estimated ED10 for male-female difference in various neuromotor measures; [q]Markowski & al. 2001 based on ED10 for altered operant responding in adult female offspring; [r]Schantz & Bowman 1989; [s]Kattainen & al. 2001; [t]Gehrs & al. 1997 (in original source, effect found at maternal body burden of 60 p g^{-1} bw), possibly irrelevant to humans due to differences in immune development (SCF 2001), DTH=delayed-type hypersensitivity; [u]Neubert & al. 1992b, regarded as an effect difficult to extrapolate to low doses and of uncertain significance; [v]Narasimhan & al. 1994; [w]Rhile & al. 1996.*

they involve uncertainties as to factors, forms and implications. The understanding of immune, neurological, behavioural and hormonal effects is still poorer, and precludes firm conclusions.

In the mechanistic and genetic basis of dioxin effects, in addition to AhR complexes, post-transcriptional processes e.g. in enzyme induction, growth factors and hormonal modulation are important. They add to the plausibility of emergent effects also in humans but, due to the complexity present, do not allow certainty in explanation and prediction of effects, particularly in generalization. The dose-response relationships are as yet very insufficiently characterized and quantifiable.

Some adverse effects have been reproduced at low levels of exposure, also internal. These doses may correspond to levels in humans that are near the background, despite the decline in exposure levels. However, interspecies and inter-period dose conversions are not straightforward. The critical effects have in many cases occurred after foetal and perinatal exposure, also during short early stages. Some effects have been transient, while others have persisted. Some of the effects in experimental animals such as disturbances in reproductive development are grave. Biochemical responses have been demonstrated in multiple systems, but the significance of many such responses is not clear.

Some of the evidence from experimental animal studies is summarized for effects deemed as critical based on exposure levels, adversity of effect and human studies (cf. Table 15, Annex 8D). In the case of effects from perinatal exposure, body burdens have been seldom been reported for the offspring. In many studies, no NOAELs have been established. In addition to LOAELs, estimates of effective and benchmark doses have been presented based on Gaylor and Aylward (2004) who compared ED estimates given by USEPA (2000a) with alternative benchmark dose estimates.

It can be seen that there is considerable variation in estimates of effective levels of exposure also within a type of effect and biological system, depending on the dose metric used, e.g. between ED and BMD estimates and the percentile fraction of population considered (e.g., BMD10 vs. BMD01 and similarly ED10 vs. ED01). In addition, the effective dose levels differ depending on whether central stastitics or lower confidence levels are used, due to the statictical variation present; the difference between e.g. mean and 95 % lower confidence limits varies from a factor of 2 to a factor of 100.

Notwithstanding these variations and uncertainties, it can be concluded from this limited comparison that the effects caused at lowest doses are those on reproductive development and on neurological (neuromotor) development in rat offspring after perinatal exposure (cf. 4.2.2). In other mammals less evidence is available, but the latter effects seem to be caused by low exposure levels also in monkeys, as are immune effects for which also mice are particularly sensitive.

The significance of such information on effects and on effective body burdens and doses are discussed in the following chapter, including discussion of their implications for the definition of acceptable dose or intake levels and other quantitative risk criteria in humans (5.5). This involves in particular the question of the applicability of alternative benchmarks in addition to or instead of the traditional NOAEL or LOAEL and uncertainty factor based approaches.

4.3 Adverse effects linked with dioxin-like compounds in Baltic Sea dependent non-human animals

4.3.1 General considerations and assessment approaches

The task of answering the question whether DLCs cause (or have caused, or may still cause) adverse effects and conditions in non-human animals consuming Baltic fish is in many respects linked with and parallel to, but in other respects different from, the above assessment of human health effects and the underlying information on effects in laboratory animals.

The '**dioxin-linked effect**' in the title does not imply that the effects would necessarily be proven to be caused by exposure to DLCs; in most cases such proof is not available and attainable. Nor can even a link in the form of statistical associations in many cases be established. Instead, by "dioxin-linked" is meant that such effects have in some connections been assumed or sometimes even shown to be associated with DLCs. This evidence will be summarized and discussed. Along with this, risk characterization

and attributability will be addressed. Also information on effect or no-effect levels of exposure is summarized. However, most of this evaluation, along with comparison of ambient and estimated effect levels, is presented in the chapter on risk and uncertainty characterization (cf. 5).

By a PEC/PNEC based approach, (predicted) environmental levels are related to (predicted) no-effect levels that are derived from experimental NOELs or LOELs using appropriate safety factors (EC 2003a), in combination with bioconcentration and dietary accumulation factors. This has been made for TCDD in aquatic environments to derive NECs for body burdens, food and water (Fig. 13). The approach is comparable with human health assessments based on experimental animals (see above), despite the additional consideration of food chain accumulation. Such risk assessment involves major uncertainties due to the description of assessment or safety factors (in extrapolating toxicity data) and of accumulation and other factors (in exposure assessment).

The paucity of information on effects on non-mammalian **species**, and also other mammals than the common laboratory animals, is a particular challenge for the assessment of ecotoxicological risks of DLCs (e.g., Brunström et al. 1995, see also Van den Berg et al. 1998).

Effect types are varied and need to be distuinguished. In addition to reproductive, developmental, immune and metabolic effects, **tumours** have been tentatively linked with exposure to DLCs in wildlife even in the Baltic Sea, such as in gray seals (Bergman 1999, Bäcklin et al. 2003, cf. below and Annex 8D), in flounder at high doses (Grinwis et al. 2000b) and in bottom-dwelling fish and rainbow trout (Williams et al. 1998). Leiomyomas have also been reported in seals from pristine waters (see Mattson et al. 1998) but have been more common in Baltic grey seals.

Because of the limitations of ascertaining and linking field observations to specific causes, **supportive evidence** must be used (cf. Hendriks 1995). In particular, population and community level effects need to be distinguished from individual-level effects, using also population models (e.g., Murata et al. 2002, Naito et al. 2004, Fig. 13). Hoffman et al. (1998) specifically pointed out that extrapolation of laboratory tests on eggs may underestimate risks, as exposures in the nature start already at fertilization, may also affect parenting, and are increased by the remaining yolk and forage. Other stressors also may increase risks in the field over those in the laboratory. In some cases however experimental animals (e.g. captive seals) may be more vulnerable than in nature.

The population-level significance of PECs much below lethal levels depends on effect **modes** and reproductive parameters (Hendriks and Enserink 1996) that are difficult to account for explicitly. An important limitation is the unavailability of data on exposure processes for many congeners, species and systems. Therefore, preferably data or data-based estimates of body burdens in the target species and tissues are used instead of such extrapolations, as is done also in human risk assessment.

There are several lines of evidence for the presence or absence of relationships between toxicity and DLCs in the Baltic Sea environment. These typically include:

- **Field studies** of disorders linked to DLCs in other species in the Baltic or elsewhere
- **Experimental** toxicological studies *in vivo* in (preferably closely) related species
- Observed **population variations** and developmental disturbances in the species under consideration and in other related species exposed to high levels of DLCs
- Studies of **dead animals** (due to various causes), and victim-control comparisons
- **Historical analogues** (both regarding effects and exposures)
- Studies and monitoring of the levels and **trends of DLCs** in animals or their feed
- **Bioassay-based** measurements of dioxin toxicity in tissues (exposure markers)
- *In vitro* studies, including xenografts and other semi-*in vivo* studies
- Mechanistic information, preferably from closely related species.

Generally an **ecoepidemiological** weight-of evidence approach is needed (Fox 1991, Mac and Edsall 1991, Wren 1991, Grasman and Fox 2001, Suter 1993, p. 332-40, Suter et al. 2002). Due to the many dimensions of generalization and the many factors in this inference (e.g. as to the weights to be placed and what extent should specifically Baltic related evidence be required), a single and uniform unequivocally best approach cannot be identified. The mode of inference is also always to some degree data-driven.

Another key challenge is the **variation** in wildlife populations. Gender, age and

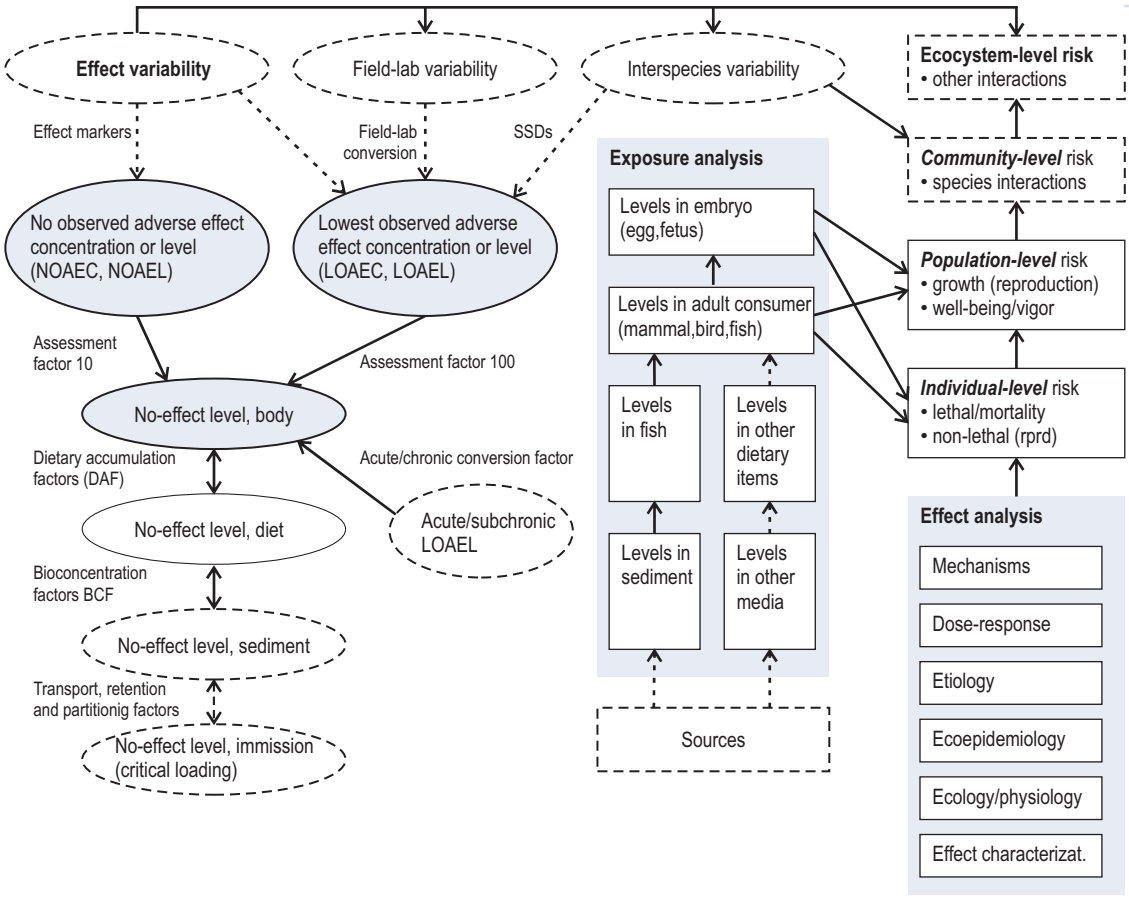


Fig. 13. Principal approaches to assessing ecotoxicological risks from dioxin-like compounds in fish. The left-hand procedure (modified and extended from Loonen et al. 1996) involves comparisons of no-effect concentrations in an aquatic food-chain with no-observed-effect or (lowest) effective concentrations and associated assessment factors, extending standard PEC/PNEC methodology to include bioconcentration and dietary accumulation (and immission stage); the right-hand procedure (modified and extended from Naito et al. 2004) also includes tissue distribution, dose-response modelling and population-level assessment. SSD=species sensitivity distribution.

reproductive status play a great role, for reproductive and developmental effects by definition. Distinct populations may have different habitats and habits and thus exposures, and also different sensitivity. One consequence is that it is difficult to know what population segments are relevant for assessment. Sometimes mature animals are considered more representative of exposures (e.g. Nyman et al. 2002). However, they may not reflect the specific effects on vulnerable juveniles. Representativeness may thus be sought also within such high-risk groups.

A similar difficulty is caused by the variation in physiological conditions. Starved animals typically exhibit high dioxin levels, but it is difficult to establish the causality between these conditions and exposures; also DLC accumulation with age is involved. Individuals in general and post-reproductive stage individuals in particular are not such a high priority in ecotoxicological as in human health risk assessment. However, the omission of highly exposed individuals may give an overly positive impression of the effects, as such mortality and morbidity goes easily unnoticed (cf. discussion of seals by Olsson 1974).

The variation in the properties and specifications of wild animals and in their condition is intimately related to the question of separating the effects of DLCs from those of **other agents** and factors causing such variations, i.e. to the question of cause-effect relationships. A common limitation is the complex and partly unknown mixtures in the field and in some experiments. Moreover, as noted e.g. by Nyman et al. (2003), particularly as the response to DLCs is often non-specific, it is difficult to distinguish it from responses to natural influences, such as nutritional status and stress. These authors stated rather pessimistically that it can be "never" excluded that the suggested causal relationship is due to co-correlation with other pollutants. This is a key issue also with Baltic fish dioxins, as in most cases other pollutants such as non-dlPCBs and

p,p′-DDE are contributing to and possibly even dominating any adverse effects. Nevertheless, supporting evidence from experimental studies regarding e.g. response mechanisms, and field evidence on the temporal development of exposure and adverse conditions can go some way to resolving such questions.

For critical **weight-of-evidence** assessments, toxicological data on non-human wild animals linked to exposure to PCDD/Fs and other DLCs need to be considered in an integrative fashion. They need to be related to other kinds of empirical (especially *in vivo*) data such as experimental studies of captive wildlife and studies in closely related laboratory animals, and acknowledging the limitations and difficulties in qualifying and interpreting much of the data.

In the following, anomalous conditions in Baltic Sea living and feeding wildlife and their suspected linkages with DLCs are summarized, based on a more detailed evaluation of evidence (Annex 8D). Some estimates of effect levels of doses, body burdens or dietary concentrations have been presented and discussed (cf. 3.4.2, 5.5.4). Information from semi-controlled studies in captive wild animals has been included, and supporting laboratory animal and mechanistic studies have been taken into account.

The analysis is based mainly on peer-reviewed literature and on previous ecotoxicological assessments. Information on Baltic Sea living species also from other geographical areas has been utilized. The possibilities and limitations in generalizing such information to the Baltic have been paid attention. These limitations are in some respects similar to the task of generalizing from laboratory or semi-field studies to field studies, although generalization in these cases involves additional issues (see above). The assessment is mainly concerned with effect profiles and indices and associated exposures. Additional characterization of effects, risks and uncertainties is included in the next chapter (5).

4.3.2 Effects and effective exposure levels in populations of Baltic Sea living and related species

Marine mammals

All Baltic marine mammals, ringed seal, grey seal, harbour seal and harbour porpoise, have during recent decades commonly displayed a variety of severe **pathological conditions**. Their profiles have varied between species, populations, regions and periods. These conditions have included reproductive and developmental disorders such as uterine occlusions in ringed and grey seals (Olsson et al. 1975, Helle et al., 1976), various lesions, and lowered immunocompetence as seen in epizootic disease outbreaks. Immunosuppression in harbour seals has been shown to be caused by consumption of Baltic Sea fish also experimentally (de Swart et al. 1994, 1996a, Ross et al. 1995, 1996b, 1997, Annex 8D).

Bergman and Olsson (1986) and Bergman et al. (1992a) described a disease complex, **hyperadrenocorticism**, in Baltic seals, indicative of metabolic disorders, immunosuppression and endocrine disruption. The prevalence of skull bone lesions increased after 1960 especially in Baltic grey seal (Bergman et al. 1992) but also harbour seals (Mortensen et al. 1992), and was considered part of the same disease complex (Bergman et al. 1992a, Olsson et al. 1994). Lind et al. (2003) found specifically that lower mineral density of trabecular bone coincided with the period of high exposure (1965-85), whereas that of cortical bone decreased only later. The role of contaminants in the disorders remained unclear. In grey seals the incidence of uterine leiomyomas had slowly decreased during 1990's, but the increased prevalence of colonic ulcers in young animals was of concern (Bergman 1999).

In some respects (e.g., development of bones and reproductive organs, immunosuppression) the adverse conditions in Baltic seals resemble effects known to be associated with PCDD/Fs and dlPCBs in laboratory animals (cf. 4.2 and Annex 8B). However, there are differences in pathologies (e.g., the absence of thymic atrophy and the presence of adrenal lesions in Baltic seals).

Many researchers have expressed the opinion that health impairment in Baltic seals has been caused at least partly by **DLCs or PCBs** (Helle et al. 1976, Ross et al. 1996c, Bergman et al. 2001). However, the possibility remains that other contaminants, including p,p′-DDE, dieldrin and chlordanes that were elevated in uterus-occluded specimens (Olsson et al. 1994), and nutritional, biotic and ecological factors, have also or alternatively been responsible (cf. Ross et al. 1996b). Wiberg et al. (2002) found that blubber levels of ΣPCBs correlated with kidney capillary wall thickening and uterine leiomyoma in Baltic female grey seals, but not with adrenal weight, uterine occlusions, other kidney lesions and

colonic ulcer; also chlordanes and DDTs were correlated. DDE-methyl sulphones are known to cause damage to adrenals in several species (Brandt et al. 1992) and p,p'-DDE and PCB methyl sulphones have been found in blubber and the latter in adrenals in Baltic seals (Haraguchi et al. 1992). This supports a role in these lesions for metabolites of p,p'-DDE and PCBs, also non-dioxinlike (cf. Fair and Becker 2000). Levin et al. (2005a) also found that lymphocyte proliferation markers in harbour seal pups correlated with PCBs rather than PCDD/Fs of TEqs. Likewise, tT3 levels in ribbon seals correlated with CB 180, not TEqs (Chiba et al. 2001). As put by Olsson et al. (1992b), "to conclusively evaluate which specific substances are hazardous for seals is not possible", on the basis of studies in wild populations.

Also in the previous reproductive disorders in Baltic ringed seals, non-dlPCBs, p,p'-DDE and other contaminants may play a role. The associations of reproductive disorders in seals were stronger with ΣDDTs than ΣPCBs in the data of Helle et al. (1976); the assumption of PCBs as a more likely cause was based largely on studies in Californian sea lions and minks. The levels of ΣPCBs were also higher in grey than ringed seals although occlusions were seen also in ringed seals (Olsson et al. 1994).

The experimental results of de Swart, Ross and co-workers on harbour seals (cf. Annex 8D) are important as these animals were fed diets of herring, allowing a comparative evaluation of immunotoxicity due to measured level of intake of also DLCs. Markers indicating immunosuppression in both NK cells and lymphocytes were in particular observed from this diet. It was judged by TEq levels in blubber that dlPCBs were mainly implicated. Ross et al. (1995) could not rule out an immunotoxic contribution from non-AhR binding chemicals. However, Ross et al. (1996a) noted that DLCs have the most immunotoxic potential of conceivable contaminants and suggested mono- and di-*ortho* PCBs were mainly responsible. The generalizability of these findings in captive seals to wild populations (cf. Storr-Hansen and Spliid 1993a) and their ecological implications are however unclear. They may have contributed to the observed disease outbreaks, but other causes are likely to have been present as well.

As discussed by Ross et al. (1996a), NK cells constitute a first line of defence against infections; specific T-cell responses then clear the infectant and maintain protection. Reduced activity of NK cells thus may lower the threshold for infection by viruses not encountered previously, T-cell responses lead to more serious infection. As both lines were dysfunctional due to Baltic fish diet, this could have contributed to the observed PDV viral disease outbreaks, even if not governing them. The observed seasonal changes in NK cell activity were also mentioned as a potentially relevant factor. Ross et al. (1997) pointed out that young individuals may be at particular risk; their studies did not cover the perinatal period. Van Loveren et al. (2000) added that long-lasting exposure occurs in the wild.

There are **supportive data** from rats fed Baltic herring (Table 17). In general, rats have been

Table 17. Comparative evaluation of immunotoxic, thyroid and retinoid (vitamin A related) effects in harbour seals and rodents (or cultured tissues) exposed experimentally to Baltic Sea fish (from Ross et al. 1996b and other sources). In parentheses, transient or weak effects.

Effect or other variable	**Captive juvenile harbour seals** exposed to Baltic herring diet[a]	Rats exposed perinatally to Baltic herring/oil[b]	Adult rats exposed to DLC mix equivalent to Baltic herring[c]	Mouse tissue culture exposed to Baltic salmon[e]
T-cell function *in vitro*	↓	(↓)		
T-cells in foetal thymus anlagen				(↓)
Delayed type hypersensitivity	↓			
Antibody response to ovalbumin	↓			
Antibody resp to RCMV[f] infection		↓	(↑)[b]	
Baseline NK cell[g] activity	↓			
Virus-associated NK cell[g] activity		↓		
Thymus cellularity		(↓)		
Thyroid levels	(↓)		↓ (TF4, FT4)	
Hepatic retinol			↓[d]	
Plasma retinol			(↓)	
Plasma retinyl palmitate			↓	
Dose, pg TEq g^{-1} d^{-1}	1-6	0.9	0.1 (0.2-0.3)	(not applicable)

References and explanations: *[a]Ross & al. 1996b, from several other studies of the group; [b]Ross & al. 1996b; [c]van der Plas & al. 2001; [d]Stern & al. 2002; [e]Håkansson & al. 1991; [f]RCMV=rat cytomegalovirus; [g]NK=Natural Killer cell.*

less susceptible to immune effects than harbour seals; most effects have been transient and weak (Ross et al. 1997). Collectively, these findings indicate that immunosuppression has occurred in multiple marine mammal species as a result of consumption of a diet consisting predominantly of Baltic herring, and DLCs are strongly suspected although not proven to have caused this. The potential implications also for humans consuming great amounts of such fish were speculated by Ross et al. (1996a, cf. 4.2).

Olsson et al. (1994) however concluded that the species hit by the PDV epizootic in 1988 did not contain higher contaminants of environmental contaminants. The most severe disease outbreak in seals also hit North Sea more severely than the Baltic. If DLCs had been a key cause, this would not have been expected, as TEqs in North Sea seals and their diet have generally been somewhat lower than in the Baltic (see below). Thus, also immunosuppression in seals may not be (or have been) caused only, or even predominantly, by DLCs.

Obstacles in attributing disorders to DLCs and in establishing effect levels have included the simultaneous occurrence of many contaminants and the difficulty of representative observation of free-ranging animals (Ross et al. 1996b). Also other factors cause variability in responses, including metabolism in some seals and porpoises e.g. for dlPCBs (cf. Kannan et al. 1995).

Decreased levels of vitamin A and, more commonly, **retinyl palmitate**, have been reported in seals exposed to Baltic herring (de Swart et al. 1994, Nyman et al. 2003). The latter authors also reported increased levels of vitamin E in all tissues in ringed and grey seals. They noted that variations in vitamin levels might be due to many factors including nutrition and feeding. Alterations in retinoid homeostasis may have been linked with immunotoxicity, as discussed by Simms and Ross (2000). Associations between retinyl palmitate and exposure levels in captive harbour seals suggest that dlPCBs and other DLCs in fish have contributed, but retinoids are also influenced by other organochlorides (de Swart et al. 1994).

Wild seal populations may be subject to still higher and more prolonged exposures than captive animals, also in more critical stages of development, and to additional stress factors (Van Loveren et al. 2000). Also for this reason the real effects e.g. in harbour seals may have been still more severe, although less easily noticeable. Olsson et al. (1994) pointed out that juvenile seals do not represent the general population but only the reproducing part. The under-representation of individuals hit by reproductive and other impairment may explain e.g. apparent constant concentrations and in general mask ongoing effects.

Marine mammal populations in the Baltic have generally **recovered**, but with temporal and regional variations. In e.g. the Archipelago Sea and Bothnian Sea fishermen already consider them a nuisance causing considerable loss of catch e.g. from traditional trap nets (Kahila 2005). Fluctuations especially in harbour seal populations in the S-W Baltic and Kattegat have been caused by recurring viral epizootics (see e.g. Helander and Bignert 1992). In ringed seal and possibly grey seal populations the reproductive rates have been estimated to be still below those in pristine natural conditions (ICES 2003a, cf. Annex 8D). Several lesions persist and some have increased, and the overall health status of these Baltic species is considered adverse. Harbour porpoise is a rare and endangered species in the Baltic due mainly to fishing, and fishing vessels are obliged to carry observers to protect it.

Few quantitative estimates of **effect levels** of DLCs in Baltic marine mammals have been produced. For harbour seals fed Baltic herring, de Swart et al. (1994) reported average intakes of 300 pg TEq d^{-1} per seal; this can be roughly converted to 6 pg TEq g^{-1} d^{-1} (cf. Ross et al. 1996b, 1-6 pg TEQ g^{-1} d^{-1}). Ross et al. (1996a) tentatively estimated a LOAEL body burden of c. 100 ppt TEq blubber fat for immunological alterations. However, it is not clear what the adversity and significance of some of the experimental effects (such as retinoid states) are in harbour seals. It has been evaluated e.g. by Falandysz et al. (2002b) that the levels in harbour seal do not pose a significant risk of adverse effects.

In summary, the etiology of the disorders in Baltic seals is as yet unclear. However, there is strong evidence implicating DLCs, especially dlPCBs, in many adverse conditions including immunotoxicity (Ross et al. 1996c, 2000). The role of PCBs in the epidemics of 1980's is still debated (O'Shea 2000a,b). The prevalence and severity of some disorders linked with DLCs have decreased concurrently with DLC levels, but the same goes to other candidate causes like p,p'-DDE. It is in any case likely that the diverse disorders in Baltic seals are multifactorial.

Terrestrial mammals

Mink (North American mink) in the Baltic Sea region is a nuisance introduced and invasive species. Fugitive from farms, it preys on coasts especially on sea birds, and is consequently actively abated. However, mink may act as a sentinel warning of risks from DLCs to other animals.

Tillitt et al. (1996) and Fox (2001) considered mink one of the most sensitive mammals to PCBs, also dlPCBs (see e.g. Aulerich et al. 1987). Reproductive disorders are among the critical effects. Also developmental toxicity and other effects, e.g. on vitamin A homeostasis, have been found after exposure to DLCs mainly in experiments with PCBs and farm mink (Annex 8D). In many cases it cannot be established to what congeners the observed effects should be attributed, but a body of evidence implicates 0-*ortho* and 1-*ortho* PCBs (e.g., Brunström et al. 1991b, 2001). Also other factors confound the assessment of some effects of DLCs. For instance, Käkelä et al. (2002a) found that hormonal (also seasonal) and dietary factors affect plasma IgG levels in minks so that possible changes due to PCBs in wild populations are difficult to detect. The influence of pharmacokinetic factors on effect levels of dlPCBs is illustrated by Heaton et al. (1995).

Effects of different fractions of PCBs on minks have been studied in a joint Swedish project. Kihlström et al. (1992) reported that minks had reduced litter size especially when fed fractions containing 1-*ortho* and 0-*ortho* PCBs, at a body burden of c. 20 ppm ΣPCBs lw; the effects were more pronounced if non-dlPCBs were included, with also elevated frequencies of early foetal death (Bäcklin and Bergman 1992). Bergman et al. (1992b) likewise found effects on liver histology from exposure to PCB mixtures and combined rather than single fractions although some effects were seen after exposure to 0-*ortho* and 1-*ortho* CBs. Edqvist et al. (1992) observed lowered cholesterol levels and Håkansson et al. (1992) lower vitamin A levels particularly after exposure to 0-*ortho* CBs. As levels of dlPCBs were not reported and were considered unreliable in the minks with most effects (Bergman et al. 1992c), quantitative assessment of these data is difficult.

The effects of Baltic fish and their DLCs on **wild** mink populations are not clear. It is however likely that peak exposures caused adverse effects also on population viability, as in the Great Lakes (Aulerich and Ringer 1977, Wren 1991, see also Fox 2001). Later on, subtle effects on mink condition probably have taken place. Nevertheless, there are grounds for concluding that risks to mink from the present lower exposure are not great. Populations of fugitive mink in the Baltic Sea region have increased strongly, in some areas already since late 1970's (Olsson 1986, ref. by Brunström et al. 2001). Mink, although very sensitive to DLCs, also seems better able to metabolize and excrete dlPCBs than otter and rats (Leonards 1998).

Farmed mink may have thus suffered adverse effects, including reproductive disorders, in earlier times when fed abundantly Baltic herring and sprat that contained levels of DLCs comparable to those in Great Lakes carp impairing mink health. It seems now subjected mainly to stress unrelated to DLCs.

Based on reproductive effects, Brunström et al. (2001) estimated a NOAEL and a LOAEL of 0.3 and 2 pg WHO-TEq g^{-1} bw d^{-1}, respectively (ca. 3 and 20 pg WHO-TEq g^{-1} ww feed). These figures are similar to the LOAEL obtained by Heaton et al. (1995) for mink fed carp, considering the differences in the TEFs used especially for CB 126 (e.g., Tillitt et al. 1991). These effect levels are well below the dietary TCDD level causing severe toxicity and increased neonate mortality (Aulerich et al. 1988, 2001, cf. Annex 8D). The effect levels are evaluated in relation to exposure levels in the chapter on risk characterization (cf. 5.5.4).

In **otter**, qualitatively similar effects from DLCs can be expected to have taken place as in the closely related minks, i.e. primarily reproductive disorders and related hormonal and metabolic effects.

The evidence for dioxin-attributable effects in otters in the Baltic Sea countries is scanty and inconclusive. However, it has been estimated (e.g., Leonards et al. 1998, cf. Brunström and Halldin 2000) that the accumulation of key dlPCBs, particularly CB 126, in otter exceeds that in mink several-fold. The lack of metabolic capability e.g. for 1-*ortho* PCBs increases the vulnerability of otter. It thus seems likely that otter declines also in the Baltic Sea region that were dramatic in 1960's to 1980's have been partly due to contaminants in diet, including dlPCBs and other DLCs in fish (see Sjöåsen et al. 1997, Roos et al. 2001).

The development of otter populations in the region is poorly known; they have recovered

e.g. in Latvia (Sjöåsen et al. 1997) and Northern Sweden but are still weak in Southern Sweden (Roos et al. 2001). Some authors have concluded that PCBs and DLCs do not any more pose a grave risk to otter populations in the region (Sjöåsen et al. 1997, Mason and Madsen 1993). Subtle stress may be present still.

The contribution of sea fish diet to toxic effects in otter is hard to distinguish. Most otters feed also on freshwater fish and amphibians and are thus not among the primary targets of DLCs specifically in Baltic Sea fish.

Birds

Adverse conditions in many species and populations of fish-consuming birds, especially reproductive disorders, have been linked with exposure to DLCs and related compounds, particularly PCBs (e.g., Gilbertson et al. 1991, Giesy et al. 1995, cf. Annex 8D). Some tentative associations have been found also for bird populations in the Baltic Sea area.

In some cases, other contaminants such as p,p'-DDE may have (had) greater importance than DLCs. It seems that only p,p'-DDE causes eggshell thinning at environmentally realistic doses (Peakall and Lincer 1996, cf. Annex 8D). However, DLCs have other effects on reproductive and other functions critical to population health. de Voogt et al. (2001) judged that other factors than p,p'-DDE are likely to have contributed to the population effects in fish-eating birds in NW Europe in 1970's-80's, and that PCBs are among the major causes. The evidence for the specific contributions of dlPCBs is yet inconclusive.

Many birds readily accumulate DLCs such as dlPCBs. On the other hand, birds are as a whole generally much less sensitive than mammals to most dlPCBs (Van den Berg et al. 1998). However, the species sensitivity varies, and in birds some qualitatively different mechanisms from those in mammals are operative e.g. in the expression of Cyp1a isoforms that may be associated with actions of DLCs, such as their multiple low-dose effects in chicken (Mahajan and Rifkin 1999). There are also ecological factors that render birds particularly vulnerable in some cases.

White-tailed sea eagle populations in the Baltic Sea area were near extinction in early 1970's. The mortality e.g. in Stockholm archipelago in 1965-66 was assumed to be related to PCBs (Jensen et al. 1969). Also eggshell thinning was reported in this species by Koivusaari et al. (1972); p,p'-DDE was considered the likely cause, due to its declining levels in 1970's (when PCB levels were still high) in association with improved reproduction (Koivusaari et al. 1980). Olsson et al. (2000b) found a significant negative correlation between the blood level of CB 118 and brood size, but not with productivity. These authors noted that CB 118 is correlated in white-tailed sea eggs with CB 126, and suggested that the levels of dlPCBs were possibly still impairing the reproduction of this species on the Baltic coast. However, they also concluded that if dlPCBs do impair reproduction of individual birds or pairs, the effect is not great enough to constitute a serious threat to the population since it is growing and reproducing well. Helander et al. (2002) judged that p,p'-DDE seemed to have been the most important factor, largely masking the effect of PCBs.

Koistinen et al. (1997b) concluded that DLCs still posed a considerable risk to Baltic white-tailed sea eagle reproduction based on hazard indices, i.e. ratios between body burdens of mainly dlPCBs and LOAELs or NOAELs in other species of birds, based on many assumptions, notably concerning extrapolation from LC50 to LOAEL. The indices suggested that adverse effect levels were exceeded by orders of magnitude. The discrepancy between this assessment and the observed recoveries of the populations and their reproduction may in part be explained by the fact that Koistinen et al. (1997b) used older exposure data; it may also be due to the use of mainly conservative assumptions increasing more readily than decreasing risks and, perhaps particularly, by p,p'-DDE masking effects of dlPCBs. Koistinen et al. (1997b) did not discuss the implications of exposures to and effects from other contaminants.

The populations of white-tailed sea eagle in the Baltic Sea have recovered during the previous decades, due to protection measures (winter-time feeding, artificial nests). Koistinen et al. (1997b) cited an unlocated meeting paper suggesting that the reproductive capacity was still suboptimal and the coastal population was supported by immigration. More recent evaluations are however optimistic. It has been estimated that reproductive performance of at least a large share of the sea eagle populations is now at or near natural background (Helander 2003). Such recoveries have taken place in both Sweden and Finland, in South and North.

This success is in part due to the sustained effort of surrogate feeding during winter, but also declines in p,p'-DDE and PCB levels have been invoked as causes (Helander 2003). It has been evaluated that e.g. the stock in the S-W Finnish archipelago is now so strong that it probably would survive even without surrogate feed (Högmander 2003). The estimated amount of nestlings e.g. in Finland in 2005, c. 250 (HS 11.7.2005), is higher than ever recorded.

All in all, even though dioxins and dlPCBs cannot with certainty be implicated, reproductive disorders have occured and subtler effects may still linger in Baltic white-tailed sea eagle populations.

There is some supportive but also conflicting evidence for adverse effects of DLCs in bald eagle. The correlations found by Elliott et al. (1996) pointed to TCDD/F as causes of Cyp1a induction although dlPCBs contributed more to TEqs; a NOAEL of 100 pg g^{-1} ww egg was proposed. Senthil Kumar et al. (2002b) found high levels of DLCs in dead bald eagles, mainly 0-*ortho* PCBs exceeding estimated effect levels in some other bird species, but of unclear etiological significance. Kennedy et al. (2003b) reported bald eagle hepatocytes were least sensitive of all bird species tested for Cyp1a induction by TCDD. However, extrapolation also across these species is difficult.

Although dependent on fish, **ospreys** are not so predominantly bound to the sea as is the case with sea eagle. Thus, even though coastal ospreys are still exposed to higher levels of DLCs and other contaminants than ospreys elsewhere, effects at a population level are not easily recognizable. As many ospreys migrate to Africa and feed on fish there, including different contaminants than those in the Baltic Sea, an assessment of risks to ospreys specifically associated with dioxins in the Baltic is difficult. Elliott et al. (2001a) found biochemical responses but no overt toxicity in osprey chicks in association with exposure to mainly CB 126, starting at levels of c. 130 pg TEq g^{-1} egg ww; involvement of p'p'-DDE was also suggested. Rattner et al. (2004) could not link the marginal productivity of ospreys to p,p'-DDE, ΣPCBs or AhR active PCBs, although the recovery of the population after DDT bans suggested earlier adverse effects (Annex 8D).

Double-crested cormorants have displayed disorders linked with DLCs in the Great Lakes, including reproductive disorders and impairments (Annex 8D). Powell et al. (1997a) found that the estimated LD50 for this species based on field studies of Tillitt et al. (1989) (cf. 5.5.4) is within the range of the LD50 predicted experimentally, concluding that the 20-fold higher levels of DLCs in the Great Lakes in 1960's and 1970's could account for much of the mortality observed then, although the species is 70-fold less sensitive to TCDD than chicken. However, Larson et al. (1996) found no association between EROD-based estimated TEq_{DFP} in eggs and the prevalence of reproductive failures or bill deformations in a small lake Michigan colony, proposing that individual variation in susceptibility may have concealed any association. They discussed that other factors including other contaminants, nutrition, diseases and gull predation may have contributed to observed adverse effects, noting also that similar bill deformations have been reported after vitamin D deficiency, i.e. rickets which has not been linked with DLCs, like vitamin A (and E and K) effects have.

In the Baltic species, **black cormorant**, no such emergent effects have been reported (cf. 3.4.2). For several years this species has increased explosively in the Baltic, now being among the most common birds also in the S-W Finnish archipelago and exerting much pressure on other species (Persson and Stenberg 2004). This increase may have been until recent decades inhibited and thus delayed by the still higher levels of DLCs. The modelling analysis of Hendriks and Enserink (1996) indicated that the Dutch black cormorant population suffered from heavy stress due to dlPCBs from late 50's to early 70's (at TEq body burdens of c. 2-10 ng g^{-1} ww), but the results depended heavily on population parameters, and other factors probably also played a role for population status. The levels measured in Southern Baltic black cormorants caught in 1992 were far below this range. This is consistent with the thriving population that has boomed since then.

The levels of dlPCBs, other PCBs and PCDD/Fs in eggs of **guillemots** in the Swedish population have declined (cf. 3.4.2). The population has had low reproductive rates but this has been associated mainly with other causes than DLCs or other contaminants, e.g. with mortality in fishing nets. Presently the population does not seem endangered.

De Roode et al. (2002) found morphological alterations in the bursa of Fabricius of guillemots

at a dose of 3 bird egg equivalents, but concluded that as the relative sensitivity of the Baltic guillemot remains unknown, no strong evidence supports that such effects will occur. Most malformations occurred at low doses in Baltic Sea embryos, and the overall malformation rate was rather similar in Baltic and North Sea embryos. The contaminants responsible for effects in any case remain to be established.

The ecotoxicological implications of the TEqs of c. 5 ng g^{-1} lw in **black guillemot** eggs in mid-1980's, due mainly to 1-*ortho* PCBs (Koistinen et al. 1995a), are unclear. It seems unlikely that this species is vulnerable to DLCs, also as the levels in its main food, eelpout, have not been very high in comparison with other species (Falandysz et al. 1996c).

Adverse reproductive, immunological, hormonal (thyroid and adrenal), behavioural and other conditions or their biochemical response markers in **herring gulls** have been often associated with contaminants in the Great Lakes (e.g., Grasman et al. 1996, 2000a, Lorenzen et al. 1999, Grasman and Fox 2001, McNabb and Fox 2003, see also Fox 2001, Annex 8D). In addition to DLCs, non-dlPCBs and p,p'-DDE have in some cases exhibited similar associations, and also other factors may have been responsible. However, e.g. porphyrigenic effects suggested the involvement of dlPCBs such as CB 105 and CB 118 (Kennedy et al. 1996). A tentative LOAEL of c. 1 ppb 'herring-TEq' ww egg can be derived from the correlation obtained by Lorenzen et al. (1999) with plasma corticosterone levels. These are based on TEFs by Kennedy et al. (1996) for herring gull embryos, being lower than WHO-TEFs for birds.

Adverse conditions have been reported in herring gulls also in the Baltic Sea region, but have differed from those in the Great Lakes. Levels of ΣPCBs and PCB-TEqs exceeding those found elsewhere to exert adverse effects have been measured, but in apparently healthy chicks (Hario et al. 2004, cf. Annex 8D). It is also difficult to link specifically DLCs in fish with conditions of herring gull as it is an omnivorous species feeding as a scavenger e.g. on garbage and eggs even in terrestrial coastal environments.

In lesser **black-backed gulls**, drastic population reductions have been observed in the Baltic during recent decades; extremely low reproduction rates and high mortality due especially to intestinal inflammation and degeneration have been estimated in Gulf of Finland. Based on ΣPCBs levels and other evidence, Hario et al. (2000) did not find strong support for a causal role of contaminants as many factors confounded associations and typical embryotoxicity was not seen. Hario et al. (2004) pointed out the high p,p'-DDE levels (from migration areas) as a possible explanation of disease and mortality. The TEq levels, being due mainly to 0-*ortho* PCBs, were higher in the disease-struck Gulf of Finland population (but not in Bothnian Bay population) than in healthy herring gull chicks, although not so clearly based on WHO TEFs or lipid weight. The TEq levels in lesser black-backed gulls exceeded LOAELs proposed for other birds; the unclear basis of many of these LOAELs and of interspecies extrapolation (and of TEFs) was not discussed.

Terns as long-range migrators are exposed to hazardous substances also in other regions than the Baltic. For PCDD/Fs and dlPCBs the contribution of faraway feeding areas is however less than e.g. for DDTs. Risks from dioxins to such Baltic Sea living birds should be tackled also in other regions (mainly Africa in the case of terns).

In North Sea **common tern** colonies, Bosveld et al. (2000) derived a LOAEL of 25 ng TEq g^{-1} liver lw for Cyp1a1 induction, caused by food levels of c. 600 pg TEq g^{-1} fish ww; also thyroid effects were seen at this exposure level. It was however judged that the species was not very sensitive, and no overt effects on reproduction or growth were expected. Murk et al. (1996) concluded that other factors and environmental changes were more important than DLCs for common tern reproduction, but their data suggested that DLCs, at levels of c. 10 ng g^{-1} egg, could have some adverse effects e.g. on egg laying date, incubation period, egg size and chick weight, and plasma vitamin A ratios and thyroid levels. The same seems likely for the Baltic, based on levels in small pelagic and coastal fish. This is supported by experimental data of Hoffman et al. (1998) who noted that levels of CB 126 causing adverse effects in common tern eggs were comparable to the highest levels in Great Lakes terns (cf. Annex 8D).

The scattered **Caspian tern** populations in the Baltic have exhibited low reproduction, but the causes of this and the role of DLCs are unclear, as exposures have not been studied. The reproduction and nesting behaviour of Caspian terns in the Great Lakes have been suspected of being compromised by DLCs (e.g., Tillitt et al. 1989, Mora et al. 1993, Yamashita et al. 1993,

Grasman and Fox 2001), as has that of Forster's terns (Kubiak et al. 1989). The latter authors pointed out that predation and other extrinsic factors may also cause aberrant parenting in terns. In Caspian terns, DLCs have also been linked with immunosuppression and deformities (Grasman et al. 1996, Fox 2001). In the Baltic Sea area, the reproductive success of Caspian tern has varied in time and geographically. Minks and parasite outbreaks are considered key threats at breeding sites, hunting and droughts in Africa (Persson and Stenberg 2004).

In **little terns**, Thyen et al. (2000) concluded based on the levels of PCBs and other organochlorines in Southern Baltic colonies that effects of recent contamination by these compounds on eggshell thickness seem improbable, but are likely to have taken place in 1960's and 1970's (cf. 3.4.2, Annex 7C). However, such effects also in terns may be due primarily to p,p'-DDE (see above and Annex 8D). No congener specific or TEq based data have been found on dioxins and related effects in Baltic terns. Also in tern populations the peak exposures have passed, and it seems unlikely that grave reproductive effects from dioxins are presently encountered.

In **Arctic terns**, no reproductive and developmental disorders in the Baltic have been reported, even during top exposure (Lemmetyinen and Rantamäki 1980). Importantly, these authors cited data (Hawksley 1957) showing that the frequency of failure at hatching could be 3- to 4-fold that in the Baltic Sea in 1960-70's in the Canadian Arctic in 1930-40's, i.e. before the accumulation of PCBs (and PCDD/Fs). It thus seems that reproductive disorders in this species (and possibly others) may occur due mainly to other causes, perhaps also natural. Such historical data have rarely been utilized in published studies and assessments of reproductive disturbances hypothesized to be caused by DLCs and other contaminants.

Eiders were found to be relatively insensitive to reproductive effects of DLCs (Brunström et al. 1990) and also to EROD induction (Annas et al. 2000), whereas Murk et al. (1994a) found that PCBs (Clophen A50 and CB 77) produced effects e.g. on retinol and thyroid states in eiders in Waddensea (slightly less contaminated than the Baltic) at a level of c. 20 ng TEq g^{-1} lw serum. They hypothesized that PCBs may have played a role in mass mortalities of eiders seen in the North Sea. Information on exposures or adverse conditions in the Baltic populations has not been found.

In summary, of wild bird species feeding on Baltic fish, there is most evidence for adverse affects of DLCs on white-tailed sea eagle that probably has been impaired especially by dlPCBs. Even in this species, other contaminants have probably played a role and grave effects e.g. on reproduction seem to have been passed. In other bird species, inconclusive findings have been presented.

Fish

Ecotoxicological assessment of risks from DLCs to fish is hampered by the lack of data on the toxicity of individual compounds in Baltic Sea living species other than rainbow trout (cf. Annex 8D). As discussed e.g. by van der Weiden et al. (1993), fish exhibit some differences in comparison with mammals in responses to TCDD, such as Cyp1 enzyme induction. Also the application of mammalian TEqs to fish should be viewed with caution. In particular, it is apparent that fish are much less sensitive to 1-*ortho* PCBs than mammals (Van den Berg et al. 1998), but some older TEqs do not account for this. The apparent relative susceptibility of various species also among fish may be due to both sensitivity and toxicokinetics, e.g. the greater half-life of TCDD in rainbow trout than in mirror carp (van der Weiden et al. 1993).

Of salmonids, especially **lake trout** has been found to be sensitive to DLCs (e.g., Guiney et al. 1997, Brown et al. 2004, cf. Annex 8D). The latter authors found exposure to 40 ng CB 126 g^{-1} bw affected T4 levels. Experimental data have been extrapolated to population declines and recoveries of lake trout in the Great Lakes. Cook et al. (2003) concluded that by 1940 the predicted sac fry mortality due to AhR-mediated toxicity alone explained the subsequent loss of the species in these lakes. Reduced survival associated with adverse effects and complicated by other environmental factors occurred after 1980 and contributed to reproductive failure of stocked trout despite declining TEqs. Present exposures in the Great Lakes are close to the most probable NOAEL, 5 pg WHO-TEq g^{-1} egg; mortality is estimated to take place already at c. 30 pg WHO-TEq g^{-1} egg (e.g., Wright and Tillitt 1999). In addition, Brown et al. (2004) found that in mature lake trout CB 126, at levels of 3 pg TEq g^{-1} muscle, transiently elevated plasma T4; a lower level of 0.1 pg TEq g^{-1} muscle induced enzymatic activity of UDPGT. These results may be generalizable to Baltic (brown) trout (cf. 5.5.4).

In the Baltic, associations between DLCs and the 'M74' reproductive disorder in **salmon** have been in focus; fry mortality has been 95 % in some hatcheries. Vuorinen et al. (1997, 1998a) found that of the agents studied, PCBs 77, 126, 169, TeCDF, and 2,3,7,8-PeCDFs had the strongest associations with fry mortality. They also affirmed that thiamine plays a role and acts as a remedy. However, Asplund et al. (1999) and Wiberg et al. (2002) found no association between M74 and levels of contaminants including PCBs (e.g. PCB 118), which might be expected to correlate with TEqs.

Several factors may contribute to thiamine deficiency and M74, including other contaminants, overall diet, food chain structure, life history and eutrophication (Bengtsson et al. 1999, Breitholz et al. 2001, cf. Annex 8D). The variations in M74 also limit explanation of the etiology of the syndrome and the specific roles of PCDD/Fs, dlPCBs and other DLCs.

Elevated fry mortality has also occurred in some Baltic **sea trout** stocks but much more seldom, and no M74-like syndrome has been seen (Landergren et al. 1999). Neither did the dioxin-like PCB 77 accelerate the breakdown of thiamine in the offspring of sea trout (Åkerman et al. 1998). However, some of the disorders reported such as oedema and blood vessel rupture and neurobehavioral disturbances resemble those found in other trout species, e.g. lake trout exposed to DLCs in the Great Lakes and experimentally (cf. Annex 8D).

Effects of DLCs in fish have been studied mainly in rainbow trout exposed to TCDD. Most studies involved injection in eggs or fry and have reported LD50 or ED50 values, not LOAELs or NOAELs, generally finding biochemical effects (on enzyme induction and EGF) at lowest doses (Annex 8D). The sensitivity varies by strain, the LOAEC and NOAEC for hemorrhage being for the Arlee strain only 15 and 3 pg TEq g^{-1} ww egg, respectively, i.e. 3-fold less than for lake trout (Wright and Tillitt 1999). Walter et al. (2000) and Giesy et al. (2002) found increased mortality in rainbow trout already at levels of 1 pg TCDD g^{-1} diet. No reports of adverse effects in Baltic rainbow trout in fish farms have been found. It seems likely that the levels in its feed (see e.g. Isosaari 2002b) are sufficiently low to protect it from overt toxicity. The recent viral outbreaks e.g. in Finnish rainbow trout farms may have mainly other causes than immunological impairment by DLCs.

Observed disorders in **other fish** (Annex 8D) have not been conclusively linked with DLCs. Whitefish seems little more sensitive than rainbow trout (Fisk et al. 1997) and effects in the Baltic Sea strains are thus unlikely. Elevated rates of mortality and deformations in Baltic **cod** have been found but contaminants seem an unlikely cause in comparison to other factors; also the peak recruitment during peak exposures (in late 1970's) is evidence against their role (Vallin et al. 1999).

The results of Grinwis et al. (2001) on European **flounder** suggest that the toxicity of CB 126 is greater than indicated by TEFs based on early life stage mortality in salmonids; the authors concluded that CB 126 may play a role in lymphocysts, skin ulcers and liver tumours seen in the field, but mentioned other possible causes. Such lesions have been found also in Baltic flounders (Lang et al. 1999, Bylund et al. 2000), but the causes and the potential roles of DLCs remain obscure. The lacking ability to metabolize dlPCBs (Murk et al. 1994b) and the inhibitory activity of PCBs on EROD induction in flounder (Besselink et al. 1998) influence the risks.

Renal lesions have been reported in **bream** and asp exposed to PCB contamination in a lake (Koponen et al. 2001) but it is unclear what the relevance is in the Baltic Sea. **Mirror carp** is a sensitive species to TCDD and related compounds (van der Weiden et al. 1992, 1994a); PeCDD was the most potent (van der Weiden et al. 1994b). TCDD has been found to be anti-estrogenic and potently reduce vitellogenin synthesis in carp *in vitro* by an AhR dependent mechanism unrelated to Cyp1a (Smeets et al. 1999b). Sakamoto et al. (2003) reported that PCDD/F discharges in a river were associated with reproductive parameters in female carp. However, the ecological significance of carp in the Baltic is not great, and it is not known whether effects are generalizable to other fish species.

Eel as a mainly littoral omnivorous predator on benthic fauna accumulates high levels of especially CB 126, CB 118 and 156 (e.g., de Boer et a. 1993). In Baltic Sea eel, 0-*ortho* PCBs, particularly CB 126, have contributed most to the $WHO\text{-}TEq_{DFP}$ (mammalian) reaching a level of >5 pg g^{-1} ww (Bjerselius et al. 2002b, cf. Annex 6B). Eel seems to be able to metabolize dlPCBs such as CB 77 and CB 126 (de Boer et al. 1993). Its sensitivity for Cyp1a1 induction by CB 77 is rather low (Schlezinger and Stegeman 2000),

although appreciable for EROD (Hewitt et al. 1998). It is likely that DLCs pose risks mainly to consumers of eel, not to the fish itself.

There are little data on the toxicity of DLC to other fish species of relevance, and results based on water concentrations (e.g., Helder 1980) are difficult to interpret. Kleeman et al. (1988) reported that the lethality of TCDD in yellow perch and in carp (LD50 of c. 4 ng g^{-1}) exceeded that in rainbow trout.

In summary, it can be neither confirmed nor excluded that DLCs have contributed to adverse effects in Baltic fish. This may have occurred earlier mainly in salmonids. Grave effects on reproduction are however not conclusively linked to DLCs.

4.3.3 Summarizing evaluation of dioxin-linked Baltic Sea wildlife effects

Many adverse effects in wildlife populations in the Baltic Sea and its surroundings have been tentatively linked with exposures to PCDD/Fs, dlPCBs and other DLCs. This is in part a result of the use of increasingly sensitive effect measures and markers, including responses at biochemical level, many of which are indications of responses that may or may not follow (Table 18).

Because of the bioaccumulation and other properties of DLCs, their effects in the Baltic Sea are mainly suspected in animals largely consuming fish, such as seals, minks and otters, and predatory and colonial fish-eating birds. Environmental and effect levels are compared in more detail below (5.5.4).

Many suspected linkages of emergent effects with DLCs are still inconclusive, and for some hypotheses contrary evidence has been produced. Such evidence has included the absence of overt toxicological or pathological conditions that could clearly be linked only to DLCs and not to other factors, as well as typically improved reproductive performance in populations of the species considered to be most highly exposed and vulnerable.

The concurrent exposure to other agents, some of them with similar temporal trends, fate and effect profiles, together with the great variation in biological systems, makes it difficult in most cases to establish definite effects and causes particularly in field studies. For many ecological effects tentatively ascribed to dioxins, basic data on the natural variation and the role of other factors are lacking, or are not fully utilized in making inference of effects.

In some cases, there is however mounting evidence of significant involvement of PCDD/Fs and still more often dlPCBs in adverse conditions, especially in the past. Such evidence is based e.g. on time orders and specificity of effects and exposures, data from other animals, dose-response relationships, supportive mechanistic and ecological information, and coherence of additional studies.

In addition to and already before overt reproductive failure or other severe population-level effects occur, other adverse effects are likely to set in. These include effects that do not normally affect reproductive rates but impair the overall condition of the population. In some cases, such effects may have an impact on a more infrequent basis e.g. through epidemics caused in part by immune disorders.

Confirmation of dioxin-attributable risks is difficult due to the lack of information on the development of exposure and adverse conditions. Ecological factors may play a considerable role in both, e.g. as modifiers of the fate of DLCs and of their effects, also through other stressors. Such factors may also prevent rejection of the hypothesis of dioxin-attributable effects. For instance, the absence of the M74 syndrome in salmon until 1974, despite high levels of PCBs in herring already before, may have been due to changes in the food chain and feeding habits, and a shift at that time to favour herring as a food source, due to its increased stocks, as speculated by Larsson et al. (1996).

As to the specific DLCs, dlPCBs have often been implicated as the key causes of adverse effects. The relative significance of dlPCBs varies by case; CB 126 in many cases dominates dioxin-type toxicity, but in some species and settings, other CBs such as CB 77 and CB 169 have been prominent as well. These (as well as PCDD/Fs) interact with other contaminants including non-dioxin-like PCBs in ways that are poorly known and hamper risk assessment.

As more sensitive and specific methods and markers are developed, and as the selection of species increases, dioxin-related responses may be found in still more wildlife species and populations and in still more settings. On the other hand, some of these effects (e.g. within enzyme induction or mRNA alterations) are subtle and transient, not necessarily adverse, difficult

Table 18. Summarizing evaluation of information on disorders in Baltic Sea wildlife that have been linked with dioxin-like compounds and PCBs, emphasizing weight of evidence. Cf. Chapter 5 and Annex 8D.

Species	Disorder(s) or adverse conditions suspected	Occurrence, development and variation of effect	DLC exposure information, etiology and confounders	Supporting(+) or conflicting(-) information	Overall evaluation of evidence as to dioxin and DLC effects
Ringed seal	uterus occlusion (sterility), renal lesions	70-80's peak (BB), improving but suboptimal; lesion incidence↑	tissue (co)CBs; other OCs (DDTs) and (biol.) stressors may dominate effects	+other (mar) mammals/ regions; (hormonal) basis of effects; metabolism	likely to have contributed to past declines and may still compromise health (suboptimal pregnancy)
Grey seal	colonic ulcers, skull lesions, adrenal/renal lesions, uterine leimyoma	ulcers increased, lesions persist	tissue CBs; other OCs/ stressors (e.g. worms) confound (immune effects ?)	+ other (mar) mammals/ regions; mechanistic basis; - metabolism	likely to have contributed to past and possibly to some present disorders, but stocks have recovered
Harbor seal	PDV epidemics, bone, skin lesions; experimental immunosuppression and biochemical responses	epidemics in –88, -02 (KAT, S-W BS); lesions in S-W BS	high DLC levels still; fish levels of concern	+ information from other regions; exp, toxicokin, mech information	contributing role of fish contaminants confirmed experimentally; field effects and share of DLCs unclear
Harbor porpoise	parasites, skin lesions, pneumonia, arthrosis, fibrosis, abscesses	frequent (S-W BS) but sparse; unclear variation in stick	D/Fs and CBs high in BS; high levels in stranded NS seals	+ sensitivity, effects in related mammals; - metabolic capability	may not have contributed significantly to adverse effects
Mink	rprd/devel impairment, VitA reduction	thriving (nuisance) population in BS archipelago	only ΣPCBs reported from BS area	+ Gt L etc studies, mechan info; - many (nat) factors	likely to have caused some adverse effects but these have not reduced populations
Otter	-rprd disorders, population declines	low rprd in 1970-80's	only ΣPCBs reported from BS area (inland)	+ mink analogue - p,p'-DDE important - also other food	have probably contributed to declines but mainly from freshwater food
White-tailed sea eagle	-infertility -failed nesting and embryo mortality	rprd disorders (in recent yrs strong recoveries)	strong accumulation; PECs>some PNECs but these are variable and unclear	+ effect levels in bald eagle - p,p'-DDE/other OCs shown to confound	(DL-)CBs have contributed to disorders; grave effect levels passed, stocks recover strong
Osprey	rprd impairment as seen in US in association with DLCs	fluctuation but no DLC-indicative disorder	no congener data; fish levels, inland exposure	- JPN data on tolerance at population level	unlikely to cause adverse effects anymore
Common tern	rprd impairment	NS (NL) colonies	below threshold for enzyme markers	+ other related species	not likely to suffer from overt adverse effects anymore
Little tern	rprd impairment in Gt L, NL common terns linked w/ coPCBs	popul declines in 1970-80's	little congener data; fish levels of concern	+ other regions, mechan info - other OCs/ factors	assessed to be probably contributing to population declines but these have past
Arctic tern	no rprd impairment or disorders reported	-	no congener data but PCBs not very high	-Lapland and old Canadian data	BS DLCs do not seem a cause for adverse effects
Caspian tern	rprd impairment in Gt L linked w/ coPCBs	low reproduction in BS populations	no representative congener data; fish levels of concern	+ other regions, mechanistic info - other OCs/ factors	may not have contributed to adverse effects as a key factor
Herring gull	rprd impairment in Gt L linked w/ coPCBs	no clear disorder reported in BS; no embryotoxicity	no representative congener data; also inland exposure	+ other regions, mechan info - other OCs/factors	may have contributed to some (subtle) effects in the past
Lesser black-backed gull	rprd impairment, mortality	popul fluctuations (causes unclear)	PCB	- p,p'-DDE regarded as more important	not likely to be of importance
Guillemot	emaciation associated w/ TEqs in NS	previous mortality in BS	declining D/F and PCB trends in eggs	- studies suggested fish nets as cause	do not seem important in BS, particularly presently
Eider	mortality and VitA, Th effects of CBs in NS	not reported in BS	no representative congener data from BS	- rprd tox in BS has been low	possible (subtle past) effects
Black cormorant	rprd/devel disorders in NS linked with DLCs	not reported in BS, popul grwth	no representative congener data	use of data on double-crested c. ?	no adverse effects likely as populations thrive
Salmon	rprd syndrome M74	common in BS	co-vary with some DLCs; fish levels of concern	diet/thiamine etc (nat) causes; +expo/mechan info	possible contributor but uncertain key cause of rprd disorders
Sea (brown) trout	rprd disorders based on Gt L trout evidence	disorders not observed in BS	little data; levels near those in salmon	+ experimental data show top sensitivity	may have suffered disorders during peak exposures
Burbot	rprd disorders	common in BB	high DLC levels in liver	-other (nat) causes	possible (subtle past) effects
Flounder	tumors and lesions	common in BS	PAH etc DLC levels unrelated to preval.	-nat causes (parasites), insensit.	probably not
Cod	rprd impairment	variable	high liver DLC levels	- many nat causes known	probably not
Herring	emaciation	variable	relatively high DLC levels (aged fish)	- many nat causes (e.g. diet)	probably not

Explanations: Gt L=Great Lakes; NS=North Sea, BS=Baltic Sea, BB=Bothnian Bay, KAT=Kattegat, SW=Swedish; D/F=PCDD/Fs, coCBs=coplanar PCBs; DLC=dioxin-like compounds, OC=organochlorides; rprd=reproductive; devel=developmental; VitA=vitamin A; PDV=Phocine Distemper Virus; M74=salmon reproductive syndrome; exp=experimental, nat=natural.

to interpret and of uncertain physiological and ecological significance. Some of them are more of the indicator and early warning type of responses (Table 18).

In most of the cases where adverse effects have been strongly linked with DLCs, mainly including reproductive disorders, the situation for these affected populations seems to have improved, at least in terms of reproduction, or to be improving. Examples are the increased viability of many seal populations, white-tailed sea eagle, black cormorant, mink and possibly otter. It is to be assumed that less drastic and notable effects are caused already at lower exposures, and may thus still continue.

4.3.4 Identification of key information and issues relevant for community and ecosystem effects

General points

In addition to effects on and risks to single-species populations that have been in focus above, DLCs in the Baltic may exert effects at the level of communities of several species and at the level of ecosystems incorporating also abiotic components. As pointed out e.g. by Downs and Ambrose (2001), a strong case has been made e.g. by McMichael et al. (1999) for considering health risks in terms of ecological relationships and collective experiences rather than solely in terms of the summation of individual exposures or characteristics.

Such effects are difficult to assess explicitly and in quantitative terms, partly as the interactions in general between the species in these communities and ecosystems are not well known in such terms, and partly as also the roles of DLCs in these complex interactive processes are hard to establish. In addition to DLCs potentially impacting the community and the ecosystem, these in turn influence DLCs (e.g. in its cycling).

It may be hypothesized that effects on higher levels of organization are not likely as long as exposure does not cause emergent effects in any species, i.e. the assemblage of single species observed might be a sufficient safeguard against higher level effects. In this case one has to assume however that the most sensitive sentinel species indeed have been observed, or at least those that have a sufficiently significant function in the community and ecosystem. This is not necessarily the case, as e.g. little is known of the risks from DLCs to and through some of the benthic fauna. However, these especially invertebrates are unlikely to be among the sensitive species themselves.

The interactions between species in Baltic **food webs** affect the community structure and function. As toxic effects of DLCs are mainly caused in top consumers, repercussions on associated or competing species and on lower trophic levels are conceivable (see some discussion of such effects in 4.3.1, and below).

Considering the importance of critical developmental stages of animals for exposure and susceptibility to DLCs, it is notable that many species at some stages in their life history assume varying positions in the food web. This may be related to changes in habitats, such as from benthos to the water column. Thus, ecological risks cannot be predicted, observed and explained only based on static and average life-history stages and spatial ranges.

Community and ecosystem effects are caused, modified and complicated by **humans**. Humans not only load the system with DLCs (and reduce these loads) and constitute objects to resultant risks. Humans also impact and interact with the sea in other respects, such as through eutrophication and, crucially for the present assessment, through fishing and aquaculture (cf. 2.3.3). The human dimensions in the system are particularly important from a management point of view(cf. 8.3, 8.4).

Humans play a role also as fishing predators and controllers. However, despite the long-term experience and some established systems, the level of control is not very high, but is constrained by both knowledge and by policy, organization and economy. For instance, some fish stocks are continuously endangered also by fishing pressures, and community-level assessment and steering of fisheries is only emerging.

Adding to this complexity, at higher organization levels the influence of **scale** on risks may become more pronounced. Community and ecosystem effects at the local and sub-regional scale are not as complex as those on the scale of the whole Baltic and its adjacent systems (such as the catchment, but also migration areas). Likewise, in structural terms, on the scale of biotopes and habitats the effects of DLCs differ from those on the scale of the whole ecosystem.

Community and ecosystem level effects may **amplify or attenuate** effects at lower levels of biological organization such as in populations. Also at population level, long-term effects may be

caused by collapses that may be (in part) due to earlier exposures to DLCs. For instance, Heide-Jørgensen et al. (1992) estimated that the skewed sex and age distributions in harbour seals in the Kattegat-Skagerrak area after the virus outbreak in 1988 could persist for decades although the high rate of reproduction was expected to enable the population to recover by 1995-96.

A notable feature in Baltic Sea community ecology, also intensified by human activities, is the introduction of **invasive species**. These may disrupt the community, even wiping out former key species. They interact with DLCs as well. Being opportunistic species in their ecology, they may e.g. have different tolerance also toward DLCs than the natural species. On the other hand, they may influence the cycling and biomagnification of DLCs, providing also new pathways of exposure.

Eutrophication is a key driver in the ecosystem. As discussed above (2, 3), the resultant oxygen deficit from consumption in bottom water has multiple effects on DLCs, both direct and indirect, e.g. by restructuring and reducing bottom fauna and by causing additional stress on the animals exposed to DLCs. Eutrophication also more directly influences the biomagnification of DLCs. However, a large part of the increased biomass e.g. in cyanobacteria blooms may not be readily recycled to fishes, but may instead bind some of the DLC pool and reduce its availability to vulnerable animals. Thus, the impacts of eutrophication on the risks from DLCs are difficult to assess.

The Baltic Sea is **vulnerable** ecologically to DLCs and other agents, due e.g. to its brackish water, cold climate and ice cover, frequent anoxic conditions, young evolutionary age and low biodiversity (cf. 5.4.3). It may be assumed that disruptions in some part of the community in such a setting may more easily lead to damage in the whole community and ecosystem. At higher levels of organization, also non-linear processes may be involved and become more pronounced in relative terms, thus adding to ecological risks also from (even indirectly) DLCs.

There is on the other hand also **resilience** and adaptive capacity in the communities and ecosystems. Even with low average diversity, and strong disequilibria due e.g. to human impacts, the system is self-organizing to a degree and is not open to wholly chaotic processes and collapses.

The decreased **trends** of exposure to PCDD/Fs and PCBs and most other DLCs in the Baltic have decreased also community and ecosystem risks, although in this case there may be longer lags involved in both impairment and recovery in the system (cf. 5.2.3).

Aspects of possible community level effects

Nisbet et al. (1997) among others have found that population dynamics will be influenced by the **recycling** of toxicants within the entire system and that a realistic model of population dynamics will in general require coupling to a model of toxicant recycling within the system. However, these authors have not explicitly dealt with the complexities involved in biomagnification. In the case of Baltic DLCs, their particular fate and effect characteristics and interacting with each other and with the environment (see below) add to the complexity.

An important aspect of community effects arises from the possibility of **top-down control** through effects on consumers cascading down to producers (Nisbet et al. 1997). In the Baltic, there are varying evidence and views of the importance of this control in relation to the bottom-up influences of autotrophic production and subsequent consumption by heterotrophs. Mollmann and Koster (1999) published evidence for top-down control of Baltic zooplankton by fish predation only in seasonal development (cf. Rice 2001). It is likely that both directions of control are operating, as concluded by Flinkman et al. (1998), also in turns as part of the fluctuation in the system.

Harvey et al. (2003) found support for the notion that **cod** exerted top-down control on sprat biomass, but had little influence on herring. Cod may thus limit and influence sprat, along with other factors including especially fishing, other species interactions and overall fluctuations in environmental conditions. Cod may thus indirectly act as a control even of macrozooplankton or other crustaceans, at least within some temporal and spatial boundaries. However, DLCs have not been found to impair Baltic cod that does not exhibit M74-like reproductive disorders (Breitholtz et al. 2001). It seems evident that cod stocks in the Baltic are controlled instead by other factors, including especially fishing pressure and environmental conditions, particularly oxygen deficiency and salt content, for which there is plenty of evidence (Aro 2000). Thus, any anthropogenic impacts and risks on and through cod, excluding the primary risks from exposure to DLCs in cod (liver) discussed already before, are probably due mainly to other factors than DLCs.

It has been regarded that **seals**, principally grey and ringed seals, have played a significant role in the regulation of Baltic fish stocks in early 1900's, their fish consumption at that time being estimated to c. 300000 t a^{-1} based on 2 kg fish d^{-1} per seal (Thurow 1997). The decline of seal stocks (as a result mainly of hunting) may have contributed to the increases and structural changes in fish stocks, although they are influenced by also other factors, largely fishing and possibly eutrophication (positively and negatively, depending e.g. on the stock, cf. Hansson 1985). The additional pressure on the seal populations by DLCs, especially dlPCBs, during the peak exposure period in 1960's and 1970's when reproductive disorders were common in these seals, may thus have affected not only seals but indirectly also fish stocks. DLCs seem however unlikely to have been a major factor in these species interactions. Correspondingly, the ongoing increases of seal stocks have also and perhaps predominantly other reasons than declines in contaminant levels.

The M74 disorder in Baltic **salmon** has not been clearly associated with DLCs (see 4.4.3). Moreover, salmon is not as important as cod as a limiting factor of pelagic fish stocks. Baltic sea trout, although presumably more sensitive than other Baltic fish including salmon, has not been struck by reproductive disturbances that might be associated with DLCs. The stock of sea trout is also so small that it is not a key species in the food web. However, levels of WHO-TEq (fish) in Baltic sea trout are still above the levels of CB 126 (in TEqs) found to cause transient thyroid effects in lake trout, and far above the levels causing enzyme induction (Brown et al. 2004). Thus, it is possible that DLCs have caused limited community-level risks through Baltic salmonids.

An important feedback loop is provided by **bacteria** that act both by degrading DLCs and decomposing animals containing DLCs, thus cycling them. They also play a key role in the development of the state of the sea more generally, e.g. in degrading the biomass increased by eutrophication, and causing anoxia. They thus influence DLC risks.

Many **other biological factors** contribute to and influence the community and ecological effects of DLCs. These factors include predation, competition, symbiotic relationships, parasitism and infections. For instance, the latter may hit animals weakened (also) by DLCs. On the other hand, such factors make it difficult to discern the specific effects due to DLCs.

Humans may contribute to community-level risks of DLCs on and through Baltic fish by exerting other pressures on stocks, especially through fishing but also eutrophication. Such pressures may make, or have made, the stocks more vulnerable to DLCs as well. However, such risks from DLCs are difficult to evaluate. As such mechanisms of risk formation are only partly influenced by DLCs, they do not seem pronounced.

Aspects of possible ecosystem level effects

As outlined e.g. by Downs and Ambrose (2001), in 'syntropic' ecotoxicology the stress from toxic substances is felt at multiple levels in the ecosystem, down to the molecular and cellular, along with other stressors like physical habitat disruption, invasive species, and over-harvesting. To this list must be added climate change and, in the case of the Baltic Sea in particular, eutrophication.

The **interacting stressors** exert impacts on the ecosystem through complex networks of stressor-response functions that include non-linearity, irregularity and stochasticity. These response functions thus allow and entail both escalating, chaotic processes and homeostatic processes. The latter act through evolutionary self-organization reflected in system resilience, e.g. in the form of recovery and repair. This amounts to a 'noisy clockwork' in the fluctuation of populations, such as the coastal cod, that cannot be captured by purely deterministic or purely stochastic descriptions (Bjørnstad and Grenfell 2001). As an extension (or a revision) of this model, it may be postulated that some of the noise is carried over to the subsequent levels of the ecosystem and of human ecology encompassing societies, while other and even qualitatively different noise is added as other chaotic and organizing processes and additional interactions come into play. It has been shown theoretically (McCann et al. 1998) and empirically that, considering nonlinearity and stochasticity, weak and moderate-strength trophic interactions are important in promoting community persistence and stability and, conversely, in upsetting them.

Physical destruction of habitats influences Baltic living and Baltic Sea dependent organisms greatly in many ways, causing stress and thus adding to and modifying (and masking) effects of DLCs. On the other hand, these and other physical disturbances may influence DLC directly, e.g. through increased cycling from sediments.

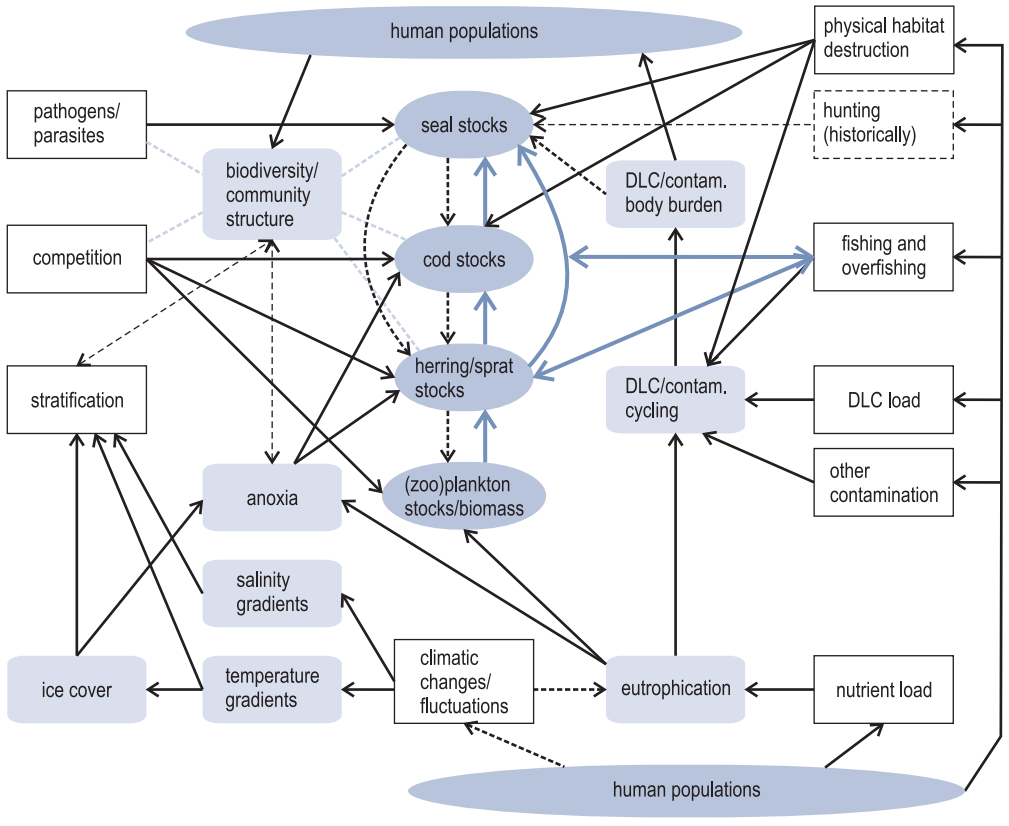


Fig. 14. Simplified influence diagram of ecological relationships affecting the cycling of dioxin-like compounds (DLCs) in pelagic communities in the Baltic Sea, with particular reference to food-chain interactions and human activities. Drivers and pressures have been indicated by quadratic frames, state variables and processes by rounded quadrats, and biological entities accumulating and suffering from DLCs by ovals. Note 1) the approximate grouping of pressures as anthropogenic and natural, but also their overlap and interplay; 2) the multiple roles of humans as causes and targets of risks; 3) the interactions between physical and other abiotic variables and processes with biotic.

Over-harvesting constitutes a key pressure on the ecosystem in coastal seas (Jackson et al. 2001). In the Baltic, especially over-fishing of cod is important ecologically (that of salmon economically and politically, and in general to a more limited extent).

In the benthic and littoral **habitats** different community and ecosystems and their interactions are present and play a role in the cycling and effects of DLCs; these are moreover linked to the pelagic system. For instance, as an example of the complexity present in this system, also these fish species in different (young) developmental stages are littoral.

An area of ecosystem effects that is closely linked to human health is the impact on **production animals** through feeding-stuff contaminated by DLCs (see SCAN 2000). This can be approached also on the level of individual species, such as concerning specifically the effects on the sensitive production animals, minks and chickens, of which at least the former is still fed Baltic herring and sprat (and vendace in Sweden). However, it is instructive to analyze these risks also in terms of human ecology, and of the fluxes and impacts in production systems. Some additional characterization of these effects and risks is provided below (5.2.4).

Climate change may influence the Baltic Sea ecosystem and technosystem, including those in its catchments, in many ways that also have a bearing on dioxin risks. Some of these changes may increase (some) risks, some decrease (other) risks. Climate change may have multi-directional effects on carbon cycling that is important as a carrier of dioxins, especially in the Northern part of the Baltic. Such changes are complex and uncertain, and their influences on dioxin risks are difficult to predict (or reconstruct), as they are multifactorial and vary e.g. by season and area, being connected with catchment characteristics and land use. The estimated annual load of TOC had a statistically significant increase in the catchments characterized by peatlands but not in those dominated by fields (Arvola et al. 2004); also some decreasing trends were seen in loads. The winter North Atlantic Oscillation index predicted rather well the load in March in the eight northernmost rivers.

The time scale of additional (and perhaps accelerating) climatic changes to take effect on dioxins may be decades. Some risks from dioxins and PCBs may by then have been additionally reduced. However, changes affecting the cycling of dioxins already in the sea (sediments) e.g. through increased storms or altered ecosystem structure may still be negative also in this regard.

Summarizing, a simplified influence diagram is produced of some of the key ecological relationships from the point of view of DLC effects and higher-level risks (Fig. 14). The focus is on the food chains and parts of the communities that have most importance for DLC bioaccumulation and also can have important feedbacks (including top-down control) in terms of biomass, i.e. the key species in the pelagic habitat, herring and sprat, cod, and seals.

4.4 Other biological effects of Baltic Sea fish, including beneficial health effects

4.4.1 Other adverse health effects of Baltic Sea fish

Baltic Sea fish contains many other harmful substances in addition to DLCs. These substances have potential impacts on the risks of DLCs. Their risks may be amplified or alleviated, either by direct synergistic or antagonistic interactions or by other means, e.g. through influences on the fate of dioxins and dlPCBs. Other substances thus need to be taken into account at least at a rudimentary and qualitative level in integrative assessment.

Multi-stressor interactions in risks go beyond biological interactions occurring on the level of AhR. For instance, even though the effects of some of the other fish ingredients are not dioxin-like in the sense of being AhR mediated, they may be rather similar regarding some endpoints in particular.

Effects of other substances (in fish) may need to be considered to decide whether fish can be consumed or used otherwise or not, and to what degrees and how. This is analogous with the comparative assessment of risk from dioxins and of benefits from other ingredients and qualities of fish (cf. 4.4.2, 5.4.4). In the present context it is appropriate to consider mainly such substances that are persistent and bioaccumulative, as they are closely associated with dioxins both in the fish matrices and in terms of other properties, and as they are relevant also in many POPs management decision contexts. Also for these compounds, only a limited description and discussion of effects is feasible in this connection, mainly in order to identify potential links with dioxin effects and risks.

Relevant other toxic substances in the Baltic that may interfere with or influence the risks from dioxins in fish include the following (for more detailed valuation, see 5, SPCFC 2005 and Annex 8):

- Di-*ortho* and poly-*ortho* PCBs have non-additive interactions with PCDD/Fs or dlPCBs in some systems and for some responses, mainly antagonizing DLCs (cf. Annex 8C)
- HCBz (as not included in DLCs)
- p,p'-DDE and related DDT metabolites
- Dieldrin and related chlorinated insecticides
- Chlordane, oxychlordane, nonachlordane, Mirex and related chlorinated compounds
- Toxaphene (Camphechlor) and related compounds
- Chlorinated phenols, guaiacols, catechols, anisols and other phenolic compounds
- Polybrominated diphenyl ethers (PBDEs)
- Methyl mercury (mainly locally and through complementary freshwater fish diet)
- TBT and related organotin compounds (for non-human animals)
- Non-dioxin-like PAHs (mainly for non-human animals)
- PFOS and related fluorinated hydrocarbons.

In general, the existence of such other contaminant classes makes the resolution of the risks due to DLCs difficult especially in the case of environmental exposures. These other contaminants also affect the magnitude and profile of risks from DLCs and have importance especially in risk comparisons (cf. 5.4). In the case of similar endpoints, the effects may often be additive. There are also other and potentially complex interactions between such other contaminants and DLCs.

While the levels of most of the above groups of POPs, like PCBs and PCDD/Fs, are declining in the Baltic due to phase-out, those of PBDEs have been found to increase (see e.g. Darnerud 2003).

4.4.2 Health benefits from fish consumption

General

Fish is a valuable source of fats, proteins, energy, vitamins and minerals. The benefits of fish consumption are variable as are their causes and co-factors, including type of fish and other dietary, life-style, genetic, biogenic and environmental factors. Health benefits are not solely based on the chemical ingredients, but depend also on the total impact of fish meals and diets. This makes assessment of them difficult despite the extensive nutritional and health research in this area. Due to the potential great significance of these effects in risk management, they are treated here in some detail (cf. Annex 8C).

Fish is of vital health significance also to many other animals, those in particular that are wholly dependent on it. From a management point of view, comparison of risks and benefits from dioxin-rich Baltic fish is meaningful mainly in terms of human health, as the reduction of human consumption has been focused on e.g. in the EU dioxin strategy (cf. 6, 8). However, any restrictions imposed on fisheries have indirect impacts, adverse and beneficial, also on non-human consumers of these fish, and still other risk management means may influence the benefits and risks to the health of other animals than humans.

Fish and particularly fatty sea fish such as salmon, rainbow trout and herring contain long-chain n-3 series polyunsaturated fatty acids (LC n-3 PUFAs) such as eicosapentaenoic acid (EPA) and docosahexaenoic acid (DHA), EPA being dominant in herring (SPCFC 2005). Other sources of n-3 PUFAs include vegetable oils that often have shorter chains, marine algae (Jacobs et al. 2004) and mother's milk (SPCFC 2005).

In Finland, c. 20 % of the total intake of n-3 PUFAs is obtained from fish (roughly as much as from meat). However, not only the amount of n-3-PUFAs is important but also the overall composition of fats. Meats contain much saturated, mono-unsaturated and polyunsaturated fatty acids (supplying c. 20, 30 and 30 % of the total intake of these by Finns, respectively) while fish contributes only 3, 5 and 7 %, respectively (Männistö et al. 2003).

In addition to overall n-3 PUFA content, the nutritional and physiological effects of fish are affected by their chain lengths, as well as by the content of n-6 PUFAs (Sargent 1997) and trans fatty acids (Dyerberg et al. 2004) and by their balances (see especially SPCFC 2005). In addition, not only the intake of fatty fish influences PUFA levels. In Sweden, mean and median levels of circulating n-3 PUFAs in a male cohort increased by c. 1/3 from 1991 to 2001 among moderate and non-consumers of fatty Baltic fish, but less among high consumers; the changes in n-6 PUFA levels were small in these groups (Hagmar et al. 2004b).

There are many general recommendations for sufficient intake of fish in general and of marine fish rich in essential PUFAs, and specific LC n-3 PUFAs, especially during pregnancy and lactation (see SPCFC 2005). However, few quantitative guidelines have been given for minimum intakes, as the suitable amounts depend e.g. on the case and on fat composition.

Cardiovascular health

Consumption of fish has been often found or thought to be beneficial for cardiovascular health. Since the early studies in Eskimos (Bang et al. 1971), this has been postulated to be due mainly to LC n-3 PUFAs in seafood. Mainly these effects and diets or supplements have been studied also in clinical interventions and randomized controlled trials. Reported beneficial effects include especially those on coronary heart disease, and also effects on risk factors like hypertension and arrhythmias (SPCFC 2005).

SPCFC (2005) summarized and discussed the evidence for cardiovascular health benefits from dietary consumption of fish, fish oil and LC n-3 PUFAs based e.g. on comprehensive reviews by Wang et al. (2004), Hooper et al. (2004) and SACN and COT (2004) of observational and intervention studies (see also Marckmann and Gronbaek 1999, Kromhout et al. 2001, 2003, Kris-Etherton et al. 2003, Marchioli et al. 2002, Harper and Jacobson 2003, He et al. 2004 and Annex 8C). Generally, the cardiac and cardiovascular health impacts of fish and fish ingredients have been studied more extensively and reliably than the adverse health effects of fish DLCs and other contaminants, using large cohorts, randomization and long follow-ups.

Many analysts have regarded the evidence as strongly supportive of the notion that n-3 PUFAs have considerable cardiovascular health benefits. For instance, Harris and Isley (2001) summarized the evidence as supporting the view

that already relatively small intakes of n-3 PUFAs are cardioprotective.

The consensus regarding these benefits is however not as clear-cut as often implied. Some large and well-designed studies could not demonstrate such benefits (e.g., Ascherio et al. 1995, see also the review e.g. of the Seven Countries and EURAMIC studies by Kris-Etherton et al. 2003). Also the above reviews arrived at different conclusions. Wang et al. (2004) on the basis of 39 studies (of 7464 reviewed) concluded "Overall, consumption of omega-3 fatty acids from fish or supplements of fish oil reduces all cause mortality and various cardiovascular disease outcomes.", while Hooper et al. (2004), based on 48 randomized controlled trials and 41 cohort studies, stated "it is not clear that dietary or supplemental omega 3 fats alter total mortality, combined cardiovascular events or cancers … There is no clear evidence that omega 3 fats differ in effectiveness according to fish or plant sources, dietary or supplemental sources, dose or presence of placebo." Such deviating conclusions depend e.g. on different study inclusion and evaluation criteria.

SPCFC (2005) concluded that while the evidence is inconclusive for effects on stroke, there is substantial evidence that fish consumption, preferably of fatty fish and alternatively fish oil or isolated LC n-3 PUFAs, benefits the cardiovascular system and is suited for secondary prevention in manifest coronary artery disease. Also SACN and COT (2004) concluded that trials provide evidence that increased fish or fish oil consumption decreases mortality among patients who have suffered a myocardial infarction. According to some data based estimates, fish consumption of 30-50 g d^{-1} is associated with a mortality risk reduction of c. 50 % (Kromhout et al. 1985, Marckmann and Gronbaeck 1999) König et al. (2005) obtained estimates of 20-30 % reduction in CHD and MI risk from modest fish consumption. Also Studer et al. (2005) in a systematic meta-analysis found that among antilipidemic agents n-3 PUFAs reduced mortality from key cardiac causes to <70 %, even more than common drugs. However, regarding the above differing evaluations, the certainty and generalizability of such quantitative estimates of lowered risk should be regarded with caution.

As discussed by Kris-Etherton et al. (2003), Wang et al. (2004) and SPCFC (2005), the beneficial effects may depend on

- Precise fat composition, e.g. the balance of n-3 and n-6 PUFAs and the distinction between animal and vegetable PUFAs
- Types of fish and method of preparation
- The form of the fish meals and diets
- Duration of use (and follow-up)
- Population in question, including age
- Cardiovascular disease risk status; benefits are greater in high-risk groups such as CVD patients, as seen in the studies of Ascherio et al. (1995) and Albert et al. (1998) that found no general health benefits

Ascherio et al. (1995) postulated that fish intake may be cardioprotective in small amounts or that other factors might modify the effect of fish or n-3 PUFAs. SACN and COT (2004) noted that the prospective epidemiological evidence suggests a plateau effect in high-risk populations, which is however not seen when fatty acid composition of blood has been analysed. SPCFC (2005) stressed in particular that few studies have considered the simultaneous effects of contaminants in fish, which is important for the present comparative risk-benefit assessment (cf. 5.4.4).

As to Baltic fish consumption, Svensson et al. (1995b) presented some evidence of lowered mortality risk from cardiovascular diseases in Swedish fishermen (12 % on East and slightly on West coast).

Arthritis and inflammatory diseases

PUFAs exert effects on a variety of inflammatory disorders that are also related to cardiovascular health (see above and SPCFC 2005). Cleland et al. (2003) concluded from collected evidence that beneficial anti-inflammatory effects of dietary fish oils have been demonstrated in randomized, double blind, placebo-controlled trials in rheumatoid arthritis. They pointed out that fish oils have protective clinical effects in occlusive cardiovascular disease (cf. above), for which patients with rheumatoid arthritis are at increased risk, suggesting important synergistic benefits from consumption of marine fish oils. They further lamented that the clinical use of anti-inflammatory fish oil has been poor.

Immune status and allergies

SPCFC (2005) evaluated the evidence that LC n-3 PUFA supplementation during pregnancy is associated with immunological parameters

of breast milk that might influence the infant's immune state and the development of allergic diseases. Abundant intake of LC n-3 PUFAs or fish has induced suppression of immune parameters, which may be beneficial for atherosclerotic and inflammatory diseases (see above) but detrimental for defence against pathogens. There is for instance evidence that intake of n-3 PUFAs has decreased NK cell activity in human lymphocytes (Yamashita et al. 1991, Svensson et al. 1994). Thus, the immunological impacts of fatty sea fish consumption in humans are complex and multi-dimensional.

Even though sensitization to fish allergens is not uncommon, the correlation to specific symptoms of food intolerance and to specific causes is weak. Fish consumption has been found to protect against some forms of allergy also in children; in particular, PUFAs have been thought effective against inflammation in the airways based on cohort studies (Nafstad et al. 2003, Annex 8C).

Reproductive health and gestational development

SACN and COT (2004) concluded there is some evidence that maternal LC n-3 PUFA intake produces beneficial effects on development during pregnancy, especially in lower birth weight populations, and this may be more relevant in populations with a lower fish intake. The potential importance in relation to the postulated reproductive toxicology of DLCs is considerable.

As detailed in SPCFC (2005), sufficient LC n-3 PUFA supply during pregnancy is important, as the foetus is dependent on the mother for its supply of these fatty acids that are (at least conditionally) essential for normal pregnancy and development. These PUFAs must be supplied *in utero* and through lactation, either endogenously synthesized or from diet.

Most studies have found beneficial effects from consumption during pregnancy of fish, particularly marine n-3 PUFAs, on birth weight (e.g., Olsen et al. 1993, 2000, 2002, Smuts et al. 2003, cf. SACN and COT 2004, SPCFC 2005). There is also evidence for a significant reduction in preterm deliveries by mothers who consumed fish at least once a week (Olsen and Secher 2002) and for increase in gestational length in several studies (see SPCFC 2005). The evidence for benefits in pregnancy-influenced hypertension is equivocal (SPCFC 2005).

Neurodevelopment and neurological and mental health

As summarized by SPCFC (2005), there is evidence in some but not all studies that a good supply of LC n-3 PUFAs enhances perinatal neurological development, as reflected e.g. in visual acuity, visual recognition and IQ. SACN and COT (2004) did not consider effects on cognitive development due to the paucity of good-quality data.

According to Uauy-Dagach and Mena (1995), there is clear evidence that dietary supplementation of n-3 PUFAs is essential for normal eye and brain development, including motor function, and may help regulate infant sleep. However, Malcolm et al. (2003) in a randomized intervention trial concluded that while DHA was essential in the development and function of the infant retina, maternal DHA status was not significantly associated with infant retinal sensitivity and no direct effect of maternal supplementation was observed.

Helland et al. (2003), noting that DHA and EPA are needed for normal growth and repair of nervous tissue, concluded that maternal intake of very-long-chain n-3 PUFAs (from cod liver oil) during pregnancy and lactation was favourable for later mental development of children as measured by IQ at 4 a (cf. Burdge 1998). Gustafsson et al. (2004) found that infant cognitive development was not explained by only PUFAs in breast milk but instead collectively by length of gestation (that may be increased by PUFAs, see above), duration of breast-feeding and the ratio DHA/AA (arachidonic acid) in colostrum. Williams et al. (2001) however provided evidence that the availability of DHA to the foetus may be decisive.

These studies are important with a view of the suspected neurological and neurobehavioral developmental effects, including motor development effects, of dioxins and dlPCBs during low-level prenatal exposure (see above). It may be possible that fish, especially fatty fish and fish oil, counteracts the potential adverse effects of PCDD/Fs and PCBs on pre- and perinatal neurological development. Reversely, contaminants including DLCs and non-dlPCBs may eliminate such benefits from fish, as may MeHg (Oken et al. 2005).

There are some studies suggesting that fish oil supplements rich in n-3 PUFAs have beneficial effects on mental health, e.g. by

relieving depression also in pregnant women and postpartum (Chiu et al. 2003, Su et al. 2003), but the reproducibility, magnitude, causes and co-factors and significance of these results are still unclear (see discussion comments to Chiu et al. 2003 and SPCFC 2005).

Metabolism

Fish, especially sea fish and its LC n-3 PUFAs, have considerable effects on lipid metabolism. The most consistent and important effect of fish or fish oil consumption is the rapid decrease in serum trigycerides which when elevated in hyperlipidaemia constitute a risk factor for myocardial infarction (see above, cf. Dyerberg 1978 and SPCFC 2005). The triglyceride lowering is effective only when intake of saturated fatty acids is reduced at the same time. Fish diet has induced a less atherogenic lipoprotein pattern (Li et al. 2004). Effects on cholesterol are less clear. EPA can, in contrast to DHA, cause an increase in LDL cholesterol.

Intake of fish oil PUFAs has been found in several studies to be associated with beneficial effects on carbohydrate metabolism, including effects on women with insulin-dependent diabetes mellitus and insulin sensitivity improvement (cf. Sidhu 2003 and Annex 8C).

Vitamin D and bone development

Fatty sea fish, especially liver, is rich on vitamin D. In Finland, c. 45 % of its total intake was obtained from fish in 2002 (Männistö et al. 2003). The sources of vitamin D are otherwise scarce especially for children. However, SPCFC (2005) estimated that generally in Europe, vitamin D and other vitamins are not as exclusively supplied by fish as is the case with LC n-3 PUFAs.

Vitamin D deficiency is associated with bone development, as seen e.g. in the historical incidence of rickets in children, prevailing throughout adulthood. As reviewed by Molgaard and Michaelsen (2003), prolonged vitamin D deficiency resulting in rickets is seen mainly during rapid growth. The authors emphasized that reports of increasing rates of rickets due to insufficient sunlight exposure and vitamin D intake are cause for serious concern.

Many countries recommend vitamin D supplementation during infancy to avoid rickets due to the low vitamin D content of human milk, and sea fish liver oil products have been traditionally used to supplement diets. Without fortification only certain foods such as fatty fish contain abundant vitamin D, and many children will depend on sun exposure to obtain it. Sunlight is scarce especially during winter in the North, and is also avoided by some, due e.g. to concerns for skin cancer and to life-style. Although the skin has a high capacity to synthesize vitamin D, if sun exposure is low this synthesis is insufficient especially in dark-skinned infants (Molgaard and Michelsen 2003). Deficiency of vitamin D from reduced consumption of fatty fish among such infants and their mothers, even among the Northern (Baltic Sea) population as a whole, may thus increase the risk of abnormal bone development, if vitamin D fortification is not ensured by other means.

It has been maintained in the Finnish justifications for derogation from the EU regulations of dioxin-laden fish that vitamin D from fatty sea fish would be particularly beneficial for elderly people (FMTI 2001). This argument may be related to the common deterioration of bones with age, especially osteoporosis. On the other hand, as shown in the studies cited above, vitamin D also at earlier stages of development is needed. This is important given the concerns for adverse effects of DLCs on bone development (cf. 4.2.2, Annex 8D).

Cancer

The evidence for a protective role of fish consumption against cancer is not clear, and there is a lack of properly designed studies (SPCFC 2005). Also SACN and COT (2004) did not consider the reports of benefits for the development of cancer due to the paucity and quality of data.

Some evidence has been obtained for beneficial effects of sea fish consumption and specifically PUFAs mainly on the sex hormone modified breast, prostate and endometrial cancer (Terry et al. 2003, cf. Rose 1997, Annex 8C). The risk reduction for endometrial cancer was particularly notable (RR 0.6, 95 % CI 0.5-0.8, p for trend 0.0002) in clinical studies (Terry et al. 2002). However, Terry et al. (2004) concluded that the epidemiological evidence is still inconclusive and that it is too early to make nutritional recommendations for fish consumption, as e.g. the specific modification of the various fatty acids and the stage of cancer may have to be taken into account. Beneficial effects on colorectal cancer

are also possible (Rose 1997, de Deckere 1999, cf. Sidhu 2003).

In the study of Svensson et al. (1995b) there were some indications of a possibly lowered risk of mortality from stomach and skin cancers among Baltic Sea fishermen, while the risk of mortality from myeloma, suspected to be promoted by PCDD/Fs, was increased. This evidence is suggestive at best.

The potential anticarcinogenic effects of fish, including fatty sea fish, may be important with a view of the relative health risks and benefits from DLCs in such fish. It may also be noted that particularly the impacts of such contaminants in farmed and wild salmon on cancer mortality have been debated (see Hites et al. 2004a and the ensuing discussion, and 5.4.4).

Other effects

- Selenium (Se), abundant in fish, is an essential nutrient and an antioxidant that may also have protective effects against cancer (e.g. Ganther 1999); however, its absorption and effects from fish diet are not clear (SPCFC 2005)
- Excessive intake of fatty fish and the fatty acids may lead to adverse effects, especially by lowered blood coagulation and thrombosis (through bleeding), in some cases to increases in cholesterol, and immunosuppression and oxidative damage when containing low vitamin E concentrations, and inhibition of linoleic acid desaturation (SPCFC 2005).

Summarizing evaluation

Taken together, the evidence points to important beneficial effects to human health from the consumption of fish in general and of fatty marine fish in particular. The effects on cardiovascular health among coronary patients are particularly well established. Some of the effects are attributable to n-3 PUFAs in fish oil, but also other fatty fish ingredients such as vitamins play a role in nutrition and health. The effects may vary depending on the population, e.g. its age and condition, and on the characteristics of fish or fish oil consumption.

It is of interest that many of the important potential or confirmed adverse human health effects and risks of DLCs at low doses, i.e. bone and neurological development and immune dysfunction, as well as the higher-dose carcinogenic, metabolic and possibly cardiac effects, are matched and probably in most cases exceeded by the health benefits of fatty fish in these same broad classes of effects or conditions (cf. 5.4.4).

5.1 General considerations

5.1.1 Qualities and characteristics of risks with particular reference to dioxin-like compounds

In a generalizing fashion, some key qualities, characteristics, dimensions and attributes of risks associated with DLCs can be discerned for the present discussion and assessment (Table 19). Some of these characteristics are shared between DLCs and other risk agents, while in many cases dioxin risks carry special connotations, and some qualities of them are particularly pronounced. It should be noted at the outset that these qualities and characteristics include more objective 'factual' and thus scientifically tractable (but not exhaustively) as well as more subjective aspects (cf. 5.1.2).

The consideration of these characteristics serves to illustrate the multi-dimensionality of risks in general and of risks from DLCs in particular. The continua from direct to indirect risks and risk factors are also notable. These characteristics in the various dimensions and in the subdivisions or risk attributes imply that dioxin risks cannot be easily captured and reduced e.g. to one measure only, such as ratios between present and adverse exposure levels, or between (human health) risks and offsetting (human health) benefits from bearing the risk.

Table 19. Dimensions, divisions and characteristics of risk in connection with dioxin-like compounds (modified from Assmuth and Louekari 2001).

Dimension	Subdivisions/explanations	Interpretations and specifications related to DLCs
Cause	attributability singularity directness naturalness	▪ causes and their relative **shares often unclear** due e.g. to confounders ▪ **multifactoriality** (multiple chemicals and other factors incl. physical, biological, social) ▪ **direct and indirect** causes (e.g., precursors) ▪ most persistent DLCs mainly **anthropogenic** but natural DLCs may play a role
Exposure	intentionality continuity target media/compartment route	▪ inadvertent by-products and accidental releases; **voluntary** exposure (diet) ▪ **continuous** and discontinuous exposures; **cumulated** long-term exposures ▪ ambient exposures - administered doses - delivered doses – critical **target tissue** doses ▪ **multi-media** (aquatic, terrestrial) ▪ **oral/diet** (fish/other) + inhalation/ (ambient/indoor/occupational, in some cases)
Target/receptors	organism class age class biological organiz level aggregation/extent societal sector social class	▪ health/well being **of humans and non-human** animals (wild and domestic) ▪ **young** development stages often at higher risk from DLCs, but in some respects even elderly ▪ molecular + **higher** (up to ecosystem level) ▪ individual and **population** risks ▪ **various sectors** (food, health, environmental, fisheries, trade etc) ▪ risks may vary e.g. according to **socio-economic** status (inequality)
Consequence	adversity reversibility specificity overall quality	▪ continuum from **subtle to grave** responses; **qualitative differences** (e.g., cancers) ▪ **transient to irreversible**; recovery/detoxification ▪ **various endpoints**; often no clear syndromes ▪ **health and other** risks (e.g. to property, cf. indirect risks)
Time	duration window immediacy past/future	▪ mainly **chronic** or sub-chronic effects ▪ critical **stages of development** and of effect initiation/promotion ▪ **lags** in both exposure and effect processes; up to trans-generational risks ▪ some effects and risks from DLCs have **passed** (decreased), others are only emerging
Spatial	local-regional-global	▪ local concentration vs. **long-range** transport (air and products); ▪ also receptor organisms **migrate** ▪ risks to citizens in various **countries**
Control	preventability regulatory status regime	▪ old DLCs already **dispersed vs. new agents** (including precursors, also planned) ▪ DLCs/precursors already (partially) **reduced vs. uncontrolled** ▪ **national to EU** level and beyond; various **sectors**
Benefit	directness content/interpretation	▪ no direct benefits from PXDD/s but from some others (e.g., PCBs, PCNs, PBBs) ▪ **harm-offsetting benefits** exist (e.g., fish diet) or can be created; loss of benefits as a counterveiling risk
Notion	subjective/objective knowledge emotion	▪ variable **perceptions**, psychological factors (even in measuring) ▪ **variable**; DLCs partly well-known, partly still poorly known 'new' risks for many persons ▪ DLC risks often associated with **dread** and outrage

In the following, characterization is made based on some generalizing analyses specifically for DLCs in Baltic fish, including come comparative evaluation of risks and benefits.

5.1.2 Perceptions of dioxin risks and their interactions with scientific characterizations of risks

Relevant general aspects of risk perception

A number of factors have been found to affect perceived risks (e.g., Slovic 1987) and may be assumed to affect those associated with DLCs. A thorough treatment of risk perception and communication is beyond the scope of the present work. However, for the purposes of risk characterization and risk management strategy analysis, some discussion is in place.

Factors influencing risk perception that pertain to risks themselves include the following **aspects of risk:**

- Anthropogenic (e.g. dioxin risks thus may be blamed on someone, cf. Douglas 1996 (1992))
- Voluntary (e.g. to dioxins in diets; such risks may be perceived as more acceptable than risks imposed by others, due also to blame and an 'outrage factor')
- Present or imminent, or only potential
- Controllable (perceived as smaller; with dioxins, this factor is not clear-cut)
- Known (dioxin risks are still rather poorly known and not commonplace to most people)
- Catastrophic (dioxin risks in some cases such as accidents and contamination episodes; however, most of them are continuous and not catastrophic in the sense of sudden risks)
- Dreaded (dioxins to many people, as 'supertoxics' of health concern; on the other hand, fears may decrease e.g. as people learn about and get used to dioxin risks)
- Local (dioxin risks are felt locally, e.g. in hotspots, and globally both in a geographical sense and in foods and exposure media of many kinds; this may heighten risk perception)
- Identified with a target (e.g., victims evoking sympathy such as children).

In addition, a host of **personal and social factors** influence risk perceptions, including age, gender, character, social class, education and upbringing, communication about the risk, culture, economic interests, prestige, and other bindings and stakes. Such personal and social factors interact with risk-related factors; thus, a generally risk-averse (cautious) person also in the case of food contaminants may easily accept some other risks. Risk views may differ e.g. according to the discipline and affiliation of those viewing the risks, not only because of sector interests but also their 'risk culture' and frame of reference, and between exprets and lay persons (e.g., Mertz et al. 1998). Perception is also modified by the context, the channel of presentation (e.g. media exposure), the presenter, the way of presentation, and psychological anchoring and other such relationships.

Individual and societal perceptions of food-related health risks are particularly multidimensional and socio-culturally conditioned and carry a strong emotional dimension. Social, psychological and cultural factors interact with nutritional, medical and technological factors and affect these perceptions in complex ways. This was shown e.g. by Dosman et al. (2001) who established that many socio-economic and demographic variables were strong predictors of an individual's risk perceptions, while only gender was robust across all the classes of health and food safety issues examined and across two time periods.

Perceptions of risks (and benefits) may also be approached from a psychoanalytical angle. Association, transference, denial, resistance, internalization, adaptation and compensation in relation to the mental challenges of risks are among the key processes. In all of these, the life history of the individual (as part of the community and its traditions) is crucial for developing risk views and attitudes. Denial plays an important role. It is present on many levels and among various groups, from those directly subject to the risk to those studying it and supposed to manage it, and may occur both as a response to perceptions of new risks or as a numbing effect to repeated warnings. Denial of risks may in some respect be seen as a healthy 'filter' of excessive risk warnings, the reaction to all of which might be impossible and even harmful, by creating unnecessary panic and also as a stress factor.

Hatcher (1982) studied how breast-feeding mothers denied the presence and possible effects of chemicals in their offspring, focusing on their

psychological defenses and reactions such as guilt or ambivalence in the nursing relationship and their efforts at mastering the situation; he also pointed out that these reactions may have widespread significance and application in risk communication and management. It can be assumed that some similar denial and psychological factors are involved in reactions to contamination of fish e.g. among those inclined to value fish economically, emotionally or symbolically and dependent on fishing. These reactions may consist of excluding the thought of risk, rationalizing it away, or defiant continuous consumption of fish not only as a calculated net benefit but as a more general attitude. It may also be part of a resistance to risk management (and risk advice) attempts. On the other hand, risks also from dioxins are sometimes exaggerated e.g. due to societal attention and a general risk-averse and health and food focused 'culture of fear' (cf. 9.2) that has both a social and an individual basis.

Perceptions and expressed views of dioxin risks

Dioxins have from early on presented a case of heated risk communication where controversy about scientific information has mixed with policy controversy and related debate both professionally and among the general public, particularly about the health risks involved.

Of particular relevance with the present case, intense risk communication has taken place about the contamination of fish by dioxins and related compounds in the Laurentian Great Lakes, although mainly caused by consumption of sport fish and not professional fisheries. This risk communication has focused on human health risks; only as a complementary dimension have ecotoxicological effects and effects on other non-human animals and ecosystems been in focus. In many such cases, the controversy and the risk perceptions have been largely about the sense of violation of basic rights, the difficulty to relate to uncertain risks, and associated fear and anxiety.

Dunn et al. (1994) concluded in a study of socio-psychological responses to a local PCB contamination case that local community context exerts an important influence on psychosocial effects of environmental contamination, and that the types of outcome measures employed and the timing of the research in the context of the site history were important factors for the ability to detect psychosocial effects of the PCB contamination and remediation.

Public perceptions are interacting with media representations in a multi-way process. On the conflict/consensus model of Tichenor and others, Griffin et al. (1995, 1999) proposed that mass mediated information signaling that agents are contaminating the local environment and posing health risks is conflict-generating and therefore will be controlled in the interest of community stability. The model may be expanded but in a modified (partly even revised) form to the case of Baltic fish dioxins that presents some similar traits, although being a less localized issue. There are notable differences in the Baltic fish dioxin 'storyline', including e.g. the juxtaposition of risks and benefits, of experts in various sectors, and of local, national and EU-wide communities. The perceptions and views of the public toward these risks may thus be assumed to be conditioned by a more complex set of factors, including both risk-inflating and risk-deflating factors. The public, the media, stakeholders and others develop their views and responses to the risks, e.g. how high risks (or risk/benefit ratios) would be acceptable, partly based on their general attitudes and affiliations. Also the mass media may both exaggerate risks (selling scares) and downplay them, even selling controversies about risk-benefit relationships and the healthiness of fish as news.

In the case of Baltic fish, some denial of risk suspicions and alarms (cf. above) may ensure that health benefits from fish are not too easily lost or reduced by over-reacting. However, such denial may also imply selectively ignoring or downplaying indications of adverse effects, e.g. among those intent on finding arguments for continuous fishing and consumption of fish. Those prone to react more promptly and strongly to risks may find such downplaying or denial of risk dangerous. The former persons may be unwilling or unable to grasp real risks and justified warnings, the latter unwilling and unable to take into account benefits associated with accepting the risks and counter-veiling risks of rash reactions to them, and generally to pause to think what would be the wisest thing to do.

The average attitude to and degree of 'filtering' dioxin risk information may depend to a considerable degree on the discipline or sector one works and has worked in. It seems probable that those knowledgeable about dioxin risks are on the average more concerned about them, whereas those not knowing them more easily

have a tendency to ignore or downplay them. Also the reverse may be true however: some dioxin experts may based on their expertise have a more laid-down attitude, and be able e.g. to sort out excessive fears from more justified ones, while less knowledgeable persons, both experts in other areas or laypersons, often exaggerate risks new to them. A self-serving risk inflation, frequently suspected or alleged by risk skeptics, may also be a factor, as may self-serving risk denial or deflation. In reality, there is no sharp dichotomy between such groups or 'camps'. It is conceivable that one and the same person in some situations and respects denies and in others acknowledges, or in some cases inflates and in others downplays a risk. Also variable relative acknowledgement of different risks is usual ('pet' risks versus others).

Consequently, many differing perceptions of the risks associated with Baltic fish dioxins may be identified. While the variation in risk perceptions and views is prominent between individuals and situations, it is postulated that group affiliation and professional background are generally important in shaping risk perceptions. No one group or person possesses the only legitimate view and truth about these risks, but only complementary (in some cases conflicting) facets of the same fundamentally indivisible reality. In a similar variable fashion, other persons and groups including stakeholders, politicians, media representatives and lay persons perceive and relate to Baltic fish dioxin risks. Stakeholders belong in some respects to professional groups but may not have specific expertise in dioxins (in other respects they may be highly knowledgeable experts); they are also particularly likely to perceive things conditioned by the perspectives and interests of the stakeholder group(s) they represent.

Solicited expert opinions about the characteristics of dioxin risks

Expert opinions about risks associated with DLCs have been produced in many of the scientific and technical reports addressing these compounds, including reviews. In addition, expert opinions have been presented e.g. by committees and other expert bodies. However, in these opinions the problem has been rarely characterized in a comparative manner, trying to put it in a broader context, or linked explicitly with risk management considerations. It can also be assumed that some of the researchers and other experts on DLCs may have bias in their views on these compounds (inherently also as a result of their focus and valuable specific expertise).

Table 20. Summary of expert opinions on characteristics of risks from PCDD/Fs and dlPCBs based on a questionnaire survey (Assmuth and Hildén 2002). Cf. 5.2.1, 5.4.2 and 8.

Risk characteristic	Opinions of respondents (number of mentions)
Key compounds	PCDDs (9), PCDFs (11), coPCBs (11), PBDD/Fs (7), PCNs (3), PCTs (3), dlPBBs (2)
Key sources	incineration (10), domestic burning (8), metal ind. (5), other ind. (4), chlorobiocides (5), traffic (2), other diffuse (2), ? (1)
Key receptor organisms	humans (11), wild mammals (6), fish (9), birds (3), domestic animals (3)
Key receptor groups	foetuses (11), brest-fed/infants (8), reproducing adults (8), juvenile (4), other/ adults in general (3)
Key effects	developmental (9), reproductive (7), neurological (7), immunological (4), tumours (6), hormonal/metabolic (5)
Dioxins compared to non-dlPCBs	> (7), = (3), < (1)
Dioxins compared to PAHs	> (3), = (4), < (3), ? (1)
Dioxins compared to PBDEs	> (4), = (3), < (2), ? (2)
Dioxins compared to toxaphenes	> (7), = (1), ? (2)
Dioxins compared to chloroparaffins	> (5), = (2), < (2), ? (2)
Dioxins compared to DDT/DDE	> (4), = (3), ? (2)

In connection with the present assessment, a questionnaire survey was distributed (mainly by email) to over 100 experts world-wide, inquiring about their views on the characteristics and particularly the management of dioxin risks (cf. 8.4.2). The replies are too few to allow a representative evaluation of expert opinions (Table 20). They are presented here merely as an illustration of the variety of opinions and as a starting point for characterizations based e.g. on published evidence and for further discussions, also in the following chapters.

It may be seen from this initial survey of expert opinions that particularly with regard to the relative risks caused by DLCs in comparison with some other classes of environmental contaminants there is considerable deviation in opinions. Also the opinions regarding the most important DLCs, their sources, receptors and effects vary.

5.2 Variations and qualities of risks associated with dioxin-like compounds in Baltic Sea fish

5.2.1 Risks due to different congeners and mixtures of dioxin-like compounds

Important congeners and congener groups

In the following, the characterization of risks by substance groups and congeners is based mainly on levels in key fish species for consumption by humans and non-human piscivore species. Additional characterizations of the various DLCs and the risks and risk factors associated with them are presented in the respective other sections, e.g. in connection with the spatial and temporal dimensions, variability and uncertainty in risks.

A specific consideration of the various DLC groups and individual DLCs is needed not only for direct assessment of their risks. In addition, the congener profiles serve as a means of identifying risks and risk factors for further assessment (e.g., Su and Christensen 1997, cf. 3.3.1). However, there are problems in this approach, as the profiles change due to differential processes along the life-cycle of the compounds, as shown e.g. for PCNs by Järnberg et al. (1997, cf. Nylund et al. 1992). Likewise, elucidation of the sources of PCDD/Fs in Gulf of Finland sediments has proven difficult because of the many variable processes along the trajectories of PCDD/Fs from different sources, and the source identification of PCBs based on their profiles is complicated by the complex and variable sources and fate processes even when applying model-based correction methods such as metabolic slopes.

Many evaluations of the relative risks due to **PCDD/Fs or dlPCBs** have concluded that the latter contribute greatly to overall dioxin toxicity and risks, based on exposure and relative toxicity. This applies to several species of Baltic fish (3.4.2, 4.3.2, cf. Annex 6B and SPCFC 2005). Such conclusions have been reached also for several Baltic fish consuming species including humans (e.g., Asplund et al. 1990, 1994, Järnberg et al. 1993, Olsson et al. 1994). A major contribution of PCBs to total dioxin risks has been found in also other aquatic systems, notably the Great Lakes (Zabel et al. 1996, Anderson et al. 1998, Kannan et al. 2001). According to Kannan et al. (1989), the relative bioconcentration and metabolic capacity of terrestrial and marine mammals suggest that the risk from dlPCBs increases from land to sea but the reverse is true for PCDD/Fs, and the bioaccumulation of CB 126 and CB 156 in carnivorous marine mammals in particular is a cause for concern.

This evaluation however more precisely depends on the system considered and on the measure of exposure and risk. For instance, while PCBs contribute in many cases half the dioxin toxicity in biota at the level of the whole Baltic, PCDD/Fs dominate in Gulf of Finland, in both sediments and other matrices (Isosaari et al. 2002c). Öberg et al. (2002b) showed that the liver retention of PCDD/Fs in rats fed Baltic fish oil was higher than that of dlPCBs, which may emphasize the relative toxicity of the former group. Kannan et al. (1995) on the other hand stated that risks from PCBs of the metabolic group III (including both 0-*ortho* and 1-*ortho* PCBs) due to their metabolism might be underestimated by TEFs.

Importantly, the TEF scheme applied can radically change the relative significance of the various compounds for total dioxin toxicity, especially in the case of PCBs. Depending on the TEFs and other premises, the risks caused by dlPCBs in relation to PCDD/Fs may thus be greater or smaller. The same goes for the individual dlPCBs and PCDD/Fs, as shown e.g. by Huestis et al. (1997) for Lake Ontario trout. Sometimes the TEFs used have not been clearly indicated, and often absolute concentrations have not been reported, making it impossible to recalculate TEqs.

Among **dlPCBs**, some distinction is needed between coplanar 0-*ortho* PCBs and 1-*ortho* PCBs. The former have often contributed much to WHO-TEq_{DFP} also in Baltic fish and fish consumers (cf. Table 8 and 3.4.2, cf. Grimvall et al. 1997, Kiviranta et al. 2002a). Many 1-*ortho* PCBs due to their higher levels also have high apparent dioxin toxicity, although their affinity for AhR is lower (e.g., Van den Berg et al. 1998, Emond et al. 2003a). The relative potency of dlPCBs varies also depending on the endpoint; for instance, Pan et al. (2004) pointed out that CB 169 potency data are much more variable than those for CB 126, and have shown that for immune effects CB 169 behaves differently than TCDD.

Mono-*ortho* PCBs can also have partial antagonistic effects with PCDD/Fs and coPCBs, as noted *in vivo* e.g. for immunotoxicity (e.g., Davis and Safe 1989; cf. below and Annex 8A).

By considering such risk-attenuating processes the relative significance of the 1-*ortho* PCBs may be lowered; also the risk due to PCDD/Fs may be lower (Safe 1993, 1997-98). The impact on risk would depend on how exclusively dioxin-type toxicity is focused on, in relation to other effects of PCBs such as neurotoxicity, and on the animals considered, e.g. birds being especially sensitive (in relative terms) to PCBs. In a management context, also the differing sources and reducibility of the risks from PCBs in comparison to PCDD/Fs need to be accounted for (cf. Alcock et al. 1998). As a general conclusion, it may however be said that dlPCBs and thus PCBs more generally comprise a key group of substances in the context of dioxin risks in Baltic fish.

Among the key 1-*ortho* PCBs in Baltic fish and its consumers, endpoints judged to be non-specific, such as effects on thyroid hormones and body and thymus weight, although indicative of developmental or metabolic disorders (wasting) and immune system effects, respectively, were not included in the re-evaluation of CB 156 by Van den Berg et al. (1998).

A congener of PCDD/Fs or dlPCBs that would involve the greatest risks in the Baltic and its fish cannot be defined unequivocally, e.g. by their levels and TEFs (cf. Assmuth 2003), as such a definition depends e.g. on the fish consuming organisms focused on. If body burdens and effects in humans are in focus, different characterizations of the risks also in terms of the key contributing substances are obtained from those for other mammals, birds or fish. Likewise, additional considerations of sources, environmental fate, toxicokinetics, effects profile and so forth affect such prioritization of congeners. In a long-term perspective the relative importance of the various DLCs will change, e.g. as generally the most persistent higher chlorinated congeners become more dominant.

As a summarizing characterization, the DLCs that may have particular importance in the Baltic Sea on the basis of their toxic properties (see 4, Annexes 1 and 8) and their levels in Baltic fish and other biota (see 3.4, Annex 6) include the following (cf. Table 8):

- **4-PeCDF**: A dominant contributor to WHO-TEqs in most fish and many fish consumers. Its elimination is slower than for most PCDD/Fs (3.4). Its relative toxicity may be greater than the present TEF, due to immune effects (see Van den Berg et al. 1998). It originates mainly from thermal sources via atmosphere and not from CPs or point sources (e.g., Jonsson et al. 1993). Vulykh and Shatalov (2001) confirmed its utility as an indicator by emission and transport modelling for the Baltic Sea area (see 3.3).
- **CB 126**: The dominant contributor to WHO-TEqs in many fish and most fish consumers; has properties in sources, fate and metabolites and toxic profile that emphasize its risks. CB 126 has been highlighted in many human and non-human risk assessments in the Baltic. It originates from PCB applications, entering the sea as diffuse loading.
- *PeCDD:* The second most important PCDD/F congener in terms of WHO-TEq contributions e.g. in Finnish fishermen; has kinetic and toxic properties that stress its importance
- *CB 156:* A key contributor to WHO-TEqs in human consumers of Baltic fish (cf. Asplund et al. 1994) and in marine mammals (Kannan et al. 1989); unclear kinetics and (dioxin) toxicity
- *CB 118:* A notable contributor to WHO-TEqs in many consumers of Baltic fish; an indicator of WHO-TEq_P and 1-*ortho* PCBs (Glynn et al. 2000a); unclear (dioxin) toxicity.

Other DLCs of importance in some systems and regards include the following (cf. Table 8, not necessarily in order of importance):

- TCDD: high potency, persistence and biomagnification, found in herring; well-studied
- 6-HxCDD: medium potency, high persistence and biomagnification, elevated in herring
- CB 105: notable contributor to TEqs e.g. in seals
- CB 169: low contribution to TEqs but particularly high persistence and biomagnification
- CB 114: notable contributor to human TEqs in some cases, e.g. Lithuania (Becher et al. 1995)
- CB 77: contribution to total TEq e.g. in salmon.

In addition, other classes of compounds that may be of direct importance for dioxin-type risks from Baltic Sea fish include especially the following (cf. 5.4.2):

- **Polyaromatic DLCs** may comprise even more of the total dioxin toxicity in Baltic Sea samples, as measured by *in vitro* Bio-TEq assays of separate fractions, than PCDD/Fs, dlPCBs or other diaromatic compounds (e.g., Brunström et al. 1992, Engwall et al. 1997a). It is not known what these polyaromates are, and what their sources, fate and effects in the Baltic might be. They may include PAHs such as BkF that are known to elicit relatively strong dioxin-type activity *in vivo* (Brunström et al. 1991a, cf. Annex 1) and are present at elevated levels in the Baltic. Some PAHs have been found to be as toxic as dlPCBs in eiders but much less toxic than dlPCBs in chickens (Brunström et al. 1990), which may indicate greater relative dioxin-like risks from PAHs in the former species. In general, polyaromatic DLCs add to the risks from PCDD/Fs and dlPCBs (cf. below). However, dioxin-like polyaromates may not be of comparable toxicological significance in food chains due e.g. to preferential transformation and elimination or to relatively lower toxicity *in vivo*, although they are recovered in such assays.
- **Dioxin-like OH- and MeSO2- metabolites** of dlPCBs: Most identified metabolites have been di-*ortho* derivatives (see 3.4). Of metabolites of dlPCBs, the 1-*ortho* 4-OH-CB 107 has been mostly detected in Baltic Sea countries (Guvenius et al. 2002, cf. Annex 6B). Its toxicological importance is not known. The main metabolites of 4-PeCDF and CB 126 in the Baltic Sea biota have not been confirmed.
- **PBDD/Fs and PBCDD/Fs** have potentially growing importance due to the increased uses of brominated compounds including PBDD/F precursors and the foreseeable increase also in exposures to PBDD/Fs. PBDD/Fs and PBCDD/Fs may also be approximately as toxic as corresponding PCDD/Fs (Birnbaum et al. 2003). However, these concerns cannot yet be substantiated (cf. 3.4.2, 3.5.3). The reported measurements in air and other matrices near sources (e.g. in waste incineration, cf. Annex 6A and IPCS 1998) have limited relevance in this regard. Model-based estimates of environmental releases and levels may be misleading as information is lacking on many relevant properties of these compounds. Also the relative potencies based on AhR binding and *in vitro* responses may differ from those *in vivo*, due e.g. to differing toxicokinetics and to greater susceptibility of these molecules to biochemical and physico-chemical attack and decay (Birnbaum et al. 2003). The same may hold for dlPBBs as compared to dlPCBs (cf. IPCS 1994a).
- **PCNs** have contributed in some cases significantly to overall dioxin toxicity in the Baltic and its biota through coplanar HxCNs and HpCNs, mainly CN 66 (e.g., Järnberg et al. 1993). Notably, in harbour porpoise liver the contribution of HxCNs to quasi-TEqs has been >50 %, exceeding that of CB 126 that dominated in other tissues (Ishaq et al. 2000). However, the total dioxin-like toxicity of PCNs in humans in the Baltic Sea region has been low (e.g., Lunden and Norén 1998). The levels of PCNs in key Baltic biota have also decreased (Järnberg et al. 1993, Norén and Meironyté 2000, 3.4); in mother's milk the drop of the quasi-TEq levels was 60 % between 1972 and 1992. The PCN congener patterns reflect various sources (Falandysz et al. 1996b, 2000b, Annex 6B). Bioaccumulation also varies; in some studies no marked biomagnification has been found (Lundgren et al. 2002b). Assessment of the specific risks of PCNs is constrained by the lack of analytical differentiation and of information on *in vivo* toxicity.

Mixture effects and cumulative risks of various compounds

Mixture effects of PCDD/Fs, PCBs and other DLCs are crucial in risk assessment. Mixture effects may be extended to interactions with yet other compounds occurring together with DLCs and with additional stressors and risk factors.

Much of the uncertainties in risks from DLCs and in their links with putative causes are related to interactive effects. They may be due to interactions in pharmacokinetics, in biological responses, or both. The aggregation or specification need regarding the interactions of various compounds also holds for the assessment of the behaviour of the various congeners in the environment.

Most information on interactive effects of DLCs comes from laboratory **rodent models** (Annex 8A) while there is relatively little information on fish and birds (see evaluation by Van den Berg et al. 1998). Mixture effects of DLCs can be approached by the following means:

- Congener-specific assessment especially when a dominant congener for overall toxicity is present
- TEFs aggregating all DLCs present, assuming additivity in effects of the individual DLCs
- In principle, also by other models (e.g. more elaborate and diversified SARs) of mixture effects
- Direct measurement of the compound effects e.g. by bioassays providing Bio-TEF values.

It is usually assumed that DLCs have **additive** (concentration or dose additive) effects. This is the foundation for the TEFs and for the calculation and quantitative assessment of dioxins as TEqs. The assumption is based on the premise that DLCs all elicit their effects through the Ah receptor and that their binding to the AhR obeys simple forms. This assumption has been shown to hold rather well for the relative bioactivity and toxicity of DLCs in a number of systems.

Many **limitations and caveats in the TEF** approach to mixture effects have also become evident, also after the revision of the system (Van den Berg et al 1998). Generally, the adequacy of the TEF approach depends on the level of imprecision allowed. It will also vary according to the framing and factors considered, e.g. toxicokinetics or toxicodynamics. TEFs may assume on one hand an unjustified singularity (with regard to the individual component congeners) and on the other hand an unjustified precision. It has been often stressed that TEFs are order-of-magnitude approximations of relative toxicities (Yoshida and Nakanishi 2003, cf. Zabel et al. 1995a).

TEFs vary between the animals and endpoints used as a basis of their definition. Discrepancies between assigned TEFs and relative *in vivo* potencies have been reported e.g. in non-human primates (van der Burght et all. 1999) and rodents (Golor et al. 2001). Also tissue-specific additive and nonadditive/antagonistic effects but no synergism were observed when doses were increased, suggesting that TEqs might overestimate risks (cf. Safe 1997-98, 2003).

Pharmacokinetics as a limitation of TEFs was stressed e.g. by van Leeuwen et al. (2000). The TEF concept was specifically criticized by Maruyama et al. (2004) for being based on data on administered doses instead of delivered doses or body burdens, and for ignoring the differences between half-lives between rodents and humans. Also the results of DeVito et al. (1998), supporting the presence of an inducible hepatic binding protein for 4PeDCF, suggest that pharmacokinetic differences between congeners are important for their relative potency. DeVito et al. (1997) proposed two sets of TEFs, one for estimating intake equivalents and the other for tissue equivalents that would account for pharmacokinetics. In a similar approach, McLachlan (1993) proposed 'exposure toxicity equivalents' accounting for environmental fate. This would further diversify the TEFs, now being specified only for the various key receptor groups (mammals, birds and fish). Hamm et al. (2003) on the other hand found that the slightly lowered degree of response to a PCDD/F mixture including 4-PeCDF and 0-*ortho* PCBs based on administered dose appeared to be due to decreased transfer of mixture components to the offspring, and that, therefore, the use of the WHO consensus TEFs reasonably predicts the developmental toxicity of this mixture of DLCs (see also Viluksela et al. 1998).

At a principle level, the TEFs may give a false sense of the precision and accuracy of effects and thus of risk estimates. When it is said that TEFs are "sufficient for risk assessment purposes" or that toxicities are "reasonably" conforming to TEFs, the verdict is entirely dependent on the meaning then implicitly attached to sufficiency or reasonableness.

Pohl and Holler (1995) concluded that the TEF concept is valid only if specific criteria are met (e.g., a broad database, consistency across endpoints, additivity of effects, common mechanism of action) and that in environmental exposure the total toxicity of DLCs is not necessarily the sum of the total individual congener toxicities. Neubert et al. (1992a) more specifically summarized the requirements for even considering the application of a TEF concept "from a scientific point of view", laying down strict criteria based on the prerequisites used routinely e.g. in pharmacology. They concluded that none of these conditions are met in the case of PCDD/Fs:

- The action of the congeners must be strictly additive in the relevant dose range
- The manifestations in different species must be identical over these dose ranges
- Dose-response curves for various endpoints for a given congener must run parallel
- The dose-response curves for a given endpoint must run parallel for the various congeners
- To extrapolate between species the kinetics must be identical, or differences accountable
- For human risk assessment, manifestations in the low dose range are of special interest, and must be identical with those observed at high doses.

A particular challenge to the TEF concept is the possibility of **non-additive effects**, synergistic or antagonistic. Synergism or antagonism as such may however be uninformative and ineffective, even misleading concepts (see Hertzberg and MacDonell 2002). Deviations from the additivity assumption and TEFs based on it may either increase or decrease risks. This can be endpoint-dependent. Brouwer et al. (1995) pointed out that exclusive use of the TEF approach might underestimate the risk of neurodevelopmental effects, as AhR independent mechanisms may be involved. Synergism or antagonism may also be dose-dependent. Van Birgelen et al. (1994) found that CB 126 antagonized TCDD effects on hepatic retinol and Cyp1a2 induction in female rats at high levels but evaluated that this may not be relevant at dose levels in general human populations.

The form of **mixture** effects of DLCs depends on the mixture. Nagao et al. (1993) reported that while a PCDD mixture conformed to I-TEFs in causing mouse cleft palate, the potencies of PCDF mixtures including 4-PeCDF were clearly overestimated by the I-TEF concept. Bol et al. (1989) found that mainly CB 77 had synergistic action on TCDD and 4-PeCDF in rainbow trout, and Zabel et al. (1995a) showed that the only pairs significantly deviating from additivity in rainbow trout sac fry mortality test were TCDD+CB 77 and TCDD+CB 126. This deviation may be important with a view of the abundance of CB 126 in the Baltic and its putative effects on salmonids. The TEF concept may be particularly poorly applicable to complex mixtures of DLCs. However, even among PCDD/Fs the effects of mixtures have been over-predicted by the relative potencies of individual congeners, sometimes 4-fold (Korner et al. 2002).

The additivity assumption does not hold equally in all **biological systems**. The increasing specification of various TEF values for different groups of animals, i.e. mammals, birds and fish (Van den Berg et al. 1998) represents an elaboration of the assumption of uniform additivity. However, additional specifications and developments in TEFs may be warranted. In non-mammalian animals, less information is available on the relative toxicity of the various DLCs. The high relative toxicity of dlPCBs in birds is an exception (cf. 4.3.2), and is reflected in the TEF values for PCBs.

The dependence of TEFs on the **endpoint** has been resolved by an expert weighing of hallmark *in vivo* effects such as wasting and thymic atrophy in standard laboratory animals (e.g., Goldstein and Safe 1989, Van den Berg et al. 1998). TEFs may however be very different if based on other effects such as specific immunotoxicity (cf. 4.2.4), or reproductive or developmental toxicity. Toyoshiba et al. (2004) demonstrated in rat Cyp1a binding and enzyme induction studies that for some responses the use of a single relative potency is not appropriate for comparing the dose-response behaviour of different DLCs; they concluded that the relative potencies were not consistent with WHO TEFs, responses failed to support dose additivity, and congeners had different dose-responses, suggesting that both the shape and magnitude of effects be considered in defining TEFs.

The application of a competitive binding model to cancer risk assessment of PCDD/F mixtures by Rao and Unger (1995) suggested that the standard TEF approaches tend to overestimate the combined total risks of higher chlorinated congeners with low toxicity but underestimate the risks of the more toxic congeners. However, Walker et al. (2005) found dose additivity for a mixture of DLCs supporting the use of TEFs.

The relative toxicity in *in vitro* bioassays may differ greatly from those in *in vivo* tests, as seen e.g. with CB 126 and TCDD in birds (Powell et al. 1996a) and with several PCBs in fish (Walker and Peterson 1991).

Uncertainties and specification needs in TEFs and mixture effects are particularly important for **PCBs** in Baltic fish assessment, due to the large share of PCBs of overall TEqs, to the inclusion of dlPCBs also in food risk management, and to the evidence for the pattern

of individual and mixture toxicity for this class of DLCs. DeVito et al. (1993) concluded from their mixture studies in the mouse that the TEFs then proposed for dlPCBs overestimated the potency of these compounds by factors of 10-1000.

Walker and Peterson (1991) found that the fish-specific TEFs of several 0-*ortho* and 1-*ortho* PCBs were 10- to 80-fold less than those based on mammalian cells. TEFs for PCBs have subsequently been defined based on their relative toxicity to the specific animal groups. Nevertheless, the presence of non-dioxinlike and weakly dioxinlike PCBs complicates assessment. Some 1-*ortho* PCBs, especially CB 118, and many di-*ortho* PCBs are present in levels orders of magnitude higher than those of dioxins and thus may block receptors and other binding sites (Windal et al. 2003b, Annex 8A). Harper et al. (1995a) concluded that the TEF approach overestimates the immunotoxicity of PCB mixtures, by up to 20-fold (for Aroclors 1242 and 1248). The relative toxicity of dlPCBs also depends on the animal group; it is lower in fish (e.g., Walker and Peterson 1991) but higher in birds (Van den Berg et al. 1998) in comparison to mammals.

Van Birgelen et al. (1996a), studying porphyria caused by PCDD/Fs, PBDD/Fs and PCBs, found higher relative potencies than those implicit in TEFs for 1-*ortho* PCBs and concluded these to have important implications for risk assessment of these compounds. Deviation from TEFs may be dependent on endpoints, notably those not associated as clearly with AhR see e.g. Kodavanti et al. 2001).

Di-*ortho* PCBs do bind to AhR but fail to elicit subsequent effects (Chen and Bunce 2004). As summarized by Haag-Grönlund et al. (1998, cf. Annex 8D), synergism has been reported for plasma thyroxin, immunotoxicity, altered hepatic foci, liver porphyrin, and liver Cyp1a1 induction between non-planar PCBs and planar PCBs or TCDD, while antagonism has been reported for hepatic Cyp1a1, Cyp2b and retinoid, immunotoxicity and teratogenicity. Although these authors found evidence only for weak antagonism and not synergism between CB 126 and CB 153 in several endpoints in rat liver, they noted that effects of non-dioxinlike PCBs by other mechanisms might add to adverse effects of DLCs. This is a principally important point related to the definition of 'dioxin effects'; the evidence for antagonism within specifically dioxin effects may therefore not provide a sufficient argument for lowering dioxin risk estimates.

In particular, the interactions of **4-PeCDF and CB 126** with other DLCs should be taken into account. Studies of these have produced variable results. De Jongh et al. (1993) reported that interactive effects on the hepatic deposition of PeCDD were observed in most of the mixed dose groups in the mouse, while for 4-PeCDF interactive effects were small or absent. On the other hand, Nagao et al. (1993) showed that the TEF of 0.5 for 4-PeCDF, proposed already by NATO/CCMS (1988a) and retained for mammals and fish by WHO in 1998 (Van den Berg et al. 1998), overestimated the cleft palate inducing potency of 4-PeCDF in mice about 2.5-fold, suggesting 0.2 as a TEF; the potencies of the PCDF mixtures studied were also clearly overestimated, unlike those of the PCDD mixtures that better conformed with WHO TEFs. Korner et al. (2002) concluded that I-TEFs overestimated the potency for EROD induction of the mixtures including 4-PeCDF in the concentration range tested in rats. It thus is possible that the risks associated with 4-PeDCF differ from those indicated by the present TEFs. This will essentially depend on which species and effects (*in vivo* and *in vitro*) are selected as the basis of TEFs, and how they will be weighted.

For **PBDD/Fs**, some differences in relative toxicity among the various congeners may be seen from those in corresponding PCDD/Fs. Birnbaum et al. (1991) reported that in mice, bromination decreased the teratogenic activity of TBDD relative to TCDD and of both 1- and 4-PeBDF relative to the chlorinated isomers, but substitution of bromines for chlorines increased the potency of TBDF relative to TCDF. Thus, in PBDD/Fs the relative significance of TBDF is emphasized.

Even after improvements, it is evident that the applicability of TEFs is not objectively resolvable but depends on notions of the meaning of 'scientific criteria'. Relaxed criteria may be allowed e.g. in the name of practicality in assessment and are not to be wholly dismissed, but are always to a considerable part ambiguous.

5.2.2 Variations and relations of risks among organisms

General

Who are subjected to risks is crucial consideration in risk management. In particular, high-risk species, populations and groups have to be identified and their specific risks characterized. This holds for both human and other risk groups.

Information is still lacking on the distribution of risks. For instance, we do not know in much detail the dioxin exposure in foetuses or children. These distributions may also be related to and conditioned by other distributions and differences between groups.

Risk variation between groups is caused not only by variation in exposures but also in susceptibilities to risks, i.e. factors in the effect stage. This may be extended to other stages of the event chain all the way to risk management and adaptation to risks.

Species, strain and other taxonomic differences and extrapolations

There are great differences between species, strains and other taxa in sensitivity to dioxins. This is reflected e.g. in the range of LD50 values of TCDD in laboratory rodents (e.g., McConnell 1989). Among rats, differences between strains are seen due e.g. to aberrant AhR in TCDD-resistant strains (Simanainen et al. 2004). Also salmonid strains differ greatly in their susceptibility to DLCs (Hornung et al. 1996a), due e.g. to genetic variation (Hansson et al. 2004), while intra-strain variation is less (Carvalho et al. 2004). Some of such differences are due to pharmacokinetics and not captured by differences in binding to AhR. The relative sensitivity of taxa also depends on effects. Some of the apparent variation in sensitivity is further due to artefacts, e.g. study design and characteristics and variability of test organisms beyond generalizable taxonomic differences.

Quantitative estimates of inter-species variability in sensitivity have been produced in ecotoxicology using species sensitivity distributions (SSDs). For human risk assessment, mainly the variability in extrapolation from rat to humans is relevant.

Differing evaluations have been offered of the significance of inter-species variability in dioxin assessment. These have often been linked with differing views of the relevance of adverse effects in experimental animals for humans, and been part of the debate of how precautionary policy should be adopted (cf. 6.1, 8.1.2, 9.3). Similar questions of the commensurability of information may be seen e.g. in the discussions about whether a 'weight-of-evidence' assessment should substitute a more traditional approach of human studies supported by (carefully selected) experimental animal studies, and what evidence and weights should in the affirmative case be applied. It seems that the evaluation of the similarity or dissimilarity of species largely depends on subjective criteria, and that uniform, detailed, objective and indisputable criteria are difficult to provide.

Birnbaum (1995a) stressed that although some species are outliers for some effects (being markedly less responsive than other species), they usually have similar susceptibility as other species for other effects. The overall conclusion was that dioxins produce similar effects in many species. Such an evaluation would have an impact on risk assessment especially by strengthening the case for extrapolating animal data to humans. In a similar vein, Grassman et al. (1998) summarized their integrated human and non-human animal assessment for dioxins: a) The reproductive, developmental, immunologic and carcinogenic responses seen in humans also occur in animal models; b) Most biochemical effects induced by dioxins in both animals and humans are mediated by the AhR; c) Animal dosing regimens can be varied to examine the range of exposures in humans; d) Dose metrics based on internal dose can be used to compare responses across species as these parameters take into account species differences in clearance; e) The biochemical responses in animal models show similarity to those observed in humans. In the authors' opinion, the similarity makes it possible to study the underlying mechanisms of dioxin toxicity relevant for humans, and, importantly, the relationship between simple biochemical and more complex responses which lead to adverse effects.

Lucier et al. (1990) and Lucier (1991), based on biochemical responses in human placentas in the Yu-Cheng cohort and in rat liver, suggested that humans are even more sensitive to such biochemical effects of these DLCs than rats, and that it is prudent to retain a conservative risk assessment. Working (1988) concluded from his

review that the human male is of relatively low fertility and thus may be generally at greater risk from reproductive toxicants than are males of common laboratory animal species, although the lack of knowledge of the physiological differences can prevent effective application of animal data to the assessment of human reproductive risk.

Other researchers have pointed out the differences between humans and other animals especially in more specific assessments of effects. Lawrence and Gobas (1997) showed that the relationship between the external dose of TCDD and resulting liver and adipose tissue levels in humans and various species of rodents vary over 700-fold, illustrating that humans and experimental animals differ considerably in their ability to convert external doses of dioxin to tissue concentrations; it was recommended that pharmacokinetic differences be considered explicitly (cf. DeVito et al. 1997, above). Neubert (1997-98) evaluated that humans seem to be a rather insensitive species e.g. in comparison with rats especially if pharmacokinetic differences are taken into account.

There is evidence for considerable inter-species and even inter-strain differences in dioxin carcinogenicity (Schramm et al. 2000). SCF (2000) concluded that rat based on body burden is more sensitive than humans to the carcinogenicity of TCDD but within an order of magnitude. Also these comparative evaluations may need to be qualified, e.g. in terms of tumour types (cf. 4.2.6). Hays et al. (1997) indicated by several dose metrics that humans are much less sensitive than rats to the carcinogenic effects of TCDD, and pointed out that prediction of cancer risks in human males based on hormonally influenced liver cancer in female rats is questionable.

Genetic studies have shed light on variation in sensitivity to DLCs. Korkalainen et al. (2001) found that the closest homolog of the AhR of the most sensitive species, guinea pig, was the human AhR. *In vitro* studies have often concluded that humans are likely to be less sensitive than rodents to TCDD (e.g., Okey et al. 1994, Harper et al. 2002, cf. Annex 8B). It can however always be argued that the generalizability of such differences to *in vivo* models is limited. It is thus difficult to arrive at definite conclusions about the significance of such findings for human health risk.

Interspecies differences in susceptibility are often assumed to depend on evolutionary similitude between the species. This relationship may not be straightforward, and e.g. intra-species (inter-strain) differences may override some of the differences between species. However, it may be assumed that, on the average, susceptibility and also effect profiles are more alike between closely related than more remote species. The closeness between humans and other primates was notably used as an argument e.g. by HCN (1996a) for emphasizing studies on effects in monkeys in risk assessment.

Species at most significant risk in the Baltic Sea

The species at most risk cannot be unequivocally defined, as this depends on both exposures and susceptibility, and as there is insufficient information on the relative vulnerability of Baltic populations or the generalizability (even in same species) from other regions or laboratories. However, some tentative comparative risk characterizations are given.

Czub and McLachlan (2004) made the point that due to the temperature dependency of the bioaccumulation of lipophilic compounds, poikilothermal organisms living in cold environments can generally attain particularly high concentration, posing a high risk to warm-blooded organisms praying on them (such as seals). Many groups and organisms living in the Baltic are in this regard relatively vulnerable to effects of DLCs (cf. below, regional variations in risks).

Seals have considerable capacity to metabolize dlPCBs (Boon et al. 1987, 1997). On the other hand, they are wholly restricted to life in the sea, and their diet predominantly consists of fish of higher DLC contents, such as herring, sprat, and whitefish. Therefore, the seals that are confined to the Baltic, i.e. ringed seal and to a great extent also grey seal and even the Swedish harbour seal populations, are in relative terms more exposed to risks from dioxins in Baltic Sea fish. In addition, their long life spans, late maturity and relatively low reproductive potential make marine mammals susceptible to DLCs (Fair and Becker 2000). Also between these species there are differences, e.g. because ringed seals display higher exposures due in part to different metabolism. However, all of them have exhibited disorders that have been linked with DLCs and PCBs in the Baltic.

Mustelids (otter and mink) are known to be highly sensitive to DLCs. They are exposed to DLCs also in other food items than Baltic Sea fish (cf. Annex 8D). In particular, the key mustelid

species of conservation concern, otter, also on the Baltic Sea coast consumes inland fish. In both coastal and inland waters it feeds much on eel (Bekker and Nolet 1990, ref. by Leonards et al. 1998). It has on the other hand been shown by Boon et al. (1997) that the ability of otters to metabolize key 1-*ortho* PCBs seems more limited than that of phocid seals, making otters vulnerable to these (weak) DLCs.

Although **human** health risks are the main concern of most people, and are a driver also for dioxins management e.g. in EU (and largely therefore are in focus also in the present assessment), humans generally do not seem to be at particular high risk from DLCs in Baltic fish, in comparison with some other animals. Although perhaps equally sensitive in some respects (see discussion above), humans are not exposed as exclusively to such DLCs as some other species.

Birds seem to be more sensitive than mammals to some effects of some DLCs, especially dlPCBs and 1-*ortho* PCBs, due to different metabolic and physiological characteristics. The risks from DLCs in Baltic fish to white-tailed sea eagle are reduced by the low-dioxin feed provided during the winter. Although the diet of the sea eagle also naturally includes non-fish items such as eider eggs, the dioxin load in this case comes also from the sea. Therefore, among birds a distinction between Baltic fish attributable and other risks cannot be easily made. Sea eagles were perhaps the most severely endangered Baltic bird species probably also due to DLCs and may still suffer effects from DLCs (cf. 5.5.4).

Compared to some other birds, herring gulls seem not sensitive, judged by EROD induction (e.g., Sanderson et al. 1998). In the Baltic no signs of the hallmark dioxin effect, chick oedema, have been reported in gulls, and the increased mortalities among herring gulls and lesser black-backed gulls may have been due to e.g. p,p'-DDE (Hario et al. 2004, see 4.3). Also common terns seem c. 10-fold less sensitive than double-crested cormorants (Giesy et al. 1994a), and 80-fold less sensitive than chickens to dioxins (Kennedy et al. 1996, Lorenzen et al. 1997a, cf. Brunström and Halldin 1998 and 5.5.4). The uncertainties in inter-species differences are however illustrated by the finding of Lorenzen et al. (1997a) that terns, relatively insensitive by some measures, seem to be much more sensitive to DLC mixtures than to individual compounds (cf. above, 5.2.1).

There are contrasting evaluations of the sensitivity of eiders to DLCs (Brunström et al. 1990, Brunström and Halldin 1998, Murk et al. 1994a, cf. 4.3.2). This may depend on endpoint; the former authors studied mortality and gross developmental disorders, the latter retinoid effects. Sensitivity may also depend on gender and stage in reproductive cycle, incubating females being particularly vulnerable due to fasting (Murk et al. 1994a). Effect levels in black cormorant have not been reported, but they do not seem sensitive (cf. 4.3.2). As indicated by the results of e.g. Sanderson and Bellward (1995), the closely related double-crested cormorant is not as sensitive to toxicity by DLCs as e.g. blue heron; however, in both species subtle effects were observable already at doses encountered in the environment (Sanderson et al. 1997).

Among **fish**, risks from DLCs are particularly great to salmonids due to their sensitivity. However, adverse effects have not been conclusively linked with DLCs in the Baltic. The M74 syndrome in salmon has also, and maybe primarily, other causes (cf. 4.3.2, Annex 8D). Levels of DLC in salmon are high considering effect levels in other species (cf. 3.4.2, Annex 6B), and the TEq levels in its diet are above those associated with reduced survival in rainbow trout; however, it is not known if salmon is as sensitive. Rainbow trout in which DLCs have most often been studied is a relevant species in the Baltic economically if not ecologically. As mentioned already (4.3.2), flounder and eel seem generally less sensitive that many salmonids to DLCs. In experimental studies, mirror carp has been shown to be relatively sensitive.

Such variability affects the characteristics and significance of risks, in addition to and in part irrespective of the absolute level of risk. The variation in both exposure and susceptibility contributes. For instance, the risks to seals may be considered more significant in a Baltic Sea fish context than the risks to otter and to other species, although the latter have been and may still be hit severely (e.g. as approximated by PEC/PNEC ratios), because for seals almost all of the risk comes from Baltic fish.

Inter-individual and inter-population variation

Inter-individual variation in responsiveness for risks may be particularly great in an outbred species like humans, although the average responsiveness in humans may be lower (e.g. Harper et al. 2002, KemI and IEM 2003, cf. above). As discussed by Hankinson (2005), also variation in coactivator expression may result in inter-

individual differences in response to AhR ligands. Inter-individual variation in sensitivity occurs in all taxa, e.g. in colonial fish-eating birds of Baltic Sea relevance (Sanderson et al. 1998). Also the data of Head et al. (2003) indicate that genetic variation between individuals may be more important for Cyp1a1 induction than variation in environmental conditions between nesting sites in herring gulls. However, usually inter-individual variability in wildlife and other non-human animals is not considered important, as the focus is on effects aggregated at population level.

Rosenberg et al. (2002) estimated that differences among individuals account for 95 % of the total **genetic variation in humans**, differences among major groups (such as ethnic) constituting only 3-5 % (still less in Europe). The inter-individual variation also in health risks from dioxins is thus a key factor. There is no superior or inferior race in terms of dioxin tolerance; instead, there are many individuals considerably more susceptible than the average person. However, Inoue et al. (2000) found race-related differences in the occurrence of genetic polymorphisms in Cyp1a1 and Cyp1b1 genes in Japanese and Caucasian populations that may, in part, cause differences in the occurrence of lung and breast cancers. Toide et al. (2003) showed that Cyp1b1 mRNA (not Cyp1a1) in leukocytes correlated with plasma PCDD/Fs especially in high responders. The estimate of Nebert et al. (1991) that 10 % of humans are particularly sensitive to AHH induction and the information on polymorphisms in dioxin-responsive genes (e.g. Watanabe et al. 2004) have relevance for population variation in risks from DLCs. In other connections, Dieter and Konietzka (1995) applied an uncertainty factor of 3-10 when extrapolating from the general population to a sensitive sub-population (cf. 5.5.1).

Inter-individual differences in **exposure** are due to variability in intake patterns, toxicokinetics, or both. Inter-individual differences in exposure scenarios are caused e.g. by occupation and lifestyle. Individual food preferences and habits play a key role. As to toxicokinetics, individual variation has in some cases and populations been relatively modest (e.g., in the TCDD elimination data used by Aylward et al. 2003b). However, diseases, nutritional deviations (e.g. malnutrition) and other physiological anomalies including genetically based variations especially in lipid metabolism may substantially alter TCDD behaviour and fate in the body. For instance, the results of Maruyama et al. (2002b) suggest that differences in body weight, gastrointestinal absorption and food intake behaviour partially explain the 24-fold (Max/Min) variation in tissue dioxin levels among the Japanese.

Variation in **responsiveness** to DLCs may depend on e.g. variations in AhR levels and enzyme activities. Pollenz et al. (1998) observed six-fold inter-individual variation in rat hepatic AhR levels. Masten et al. (1998) measured variation of Cyp1a1 in human populations with high occupational exposures and general population volunteers (cf. Annex 8A), suggesting that human individual variation in response to dioxin exceeds the standard 10-fold safety factor routinely used in risk assessment (cf. 5.5.1). The presenter of the paper (Walker, personal communication 1998) added that a safety factor >20 due to individual variation could be justified based on such data. The authors however pointed out that it is unclear whether (and how strongly) Cyp1a1 inducibility is related to altered susceptibility to dioxin-associated emergent health effects. This is a consideration mainly for risk policy and depends on the precaution and safety desired with respect to evidence. Bogaards et al. (2000) have likewise shown that the inter-individual variation in liver EROD and Cyp1a2 activity is 20-fold (see below). Also Lucier et al. (1990) found much inter-individual variation in AHH and EROD induction by DLCs in a highly exposed Taiwanese population. No data or theoretical grounds have been found that refute the empirically based assumption of human individual variability in responsiveness; it can also be postulated that although some factors may attenuate it in response sequelae, others may in contrast amplify it.

Genetic **polymorphisms** and their relationships with inter-individual variation in susceptibility to dioxins (Canton et al. 2003, cf. above and Annex 8A) provide additional insights, e.g. for identifying and marking high-risk groups and specific effects. It has been evaluated that the polymorphisms found so far have not been associated with important health outcomes, e.g. in cancer risk (Harper al. 2002). Also Watanabe et al. (2004) concluded that it is not yet known whether the polymorphism found in the AHRR gene has a direct effect on dioxin-related signal transductions involved in male infertility. However, some suggestive *in vivo* results have been provided also on the genetic basis of variability in human susceptibility to DLCs. For instance, Fujita et al. (2002) found that 46 % of patients with micropenis and 27% of controls were homozygous for

Pro185Ala polymorphism (p=0.03), concluding that homozygosity for this allele of AHRR may increase the susceptibility of a foetus to under- or demasculinizing effects of dioxin exposure *in utero*. It should in any case be realized that no single gene or even a combination of genetic factors will exhaustively explain the variability in the multi-factorial and multi-attribute effects of dioxins, and that genetic information even in combination with information on other factors, e.g. environmental and life-style related, will not elucidate the risks in a deterministic and complete manner.

Johnson and DeRosa (1999) in the appraisal of public health implications of Great Lakes contaminants stressed that the findings of neurodevelopmental and reproductive deficits are compelling and, although sometimes characterized as subtle, have profound implications particularly for the tails of the distributions in populations. The evaluation of the certainty of these effects and of their implications is still open to question. However, the emphasis on the most susceptible sub-populations seems justified by the inter-individual variation observed. These authors also pointed out that these are deficits that, once incurred, cannot be repaid.

Particular risk groups

A) Age

Young individuals, humans and non-humans, are the most commonly stated particular high-risk group with regard to risks from DLCs. This is a reflection of both the high relative exposure of the young developmental stages and their often-higher susceptibility. Also the growing general recognition of many particular and even qualitatively different characteristics in the physiology and overall biology of infants and other young developmental stages has contributed to the identification of this risk group. Such individuals are in a different and in many respects more vulnerable position also in terms of social characteristics and risk factors. Thus, many experts have stressed infants and children in addressing dioxin risks (e.g., Upton 1994, KemI & IEM 2003).

The young have some metabolic and adaptive abilities, while others are deficient (cf. Annex 7C). The balance between metabolizing, detoxifying or adaptive abilities and inabilities becomes important. The functions of the brown stool fat in the perinatal stages may specifically play a key role for what the net outcome of exposure to DLCs will be (Rozman et al. 1987).

Regularly the high relative exposures and conceivable greater susceptibility of young development stages also in humans have been invoked as arguments for a more precautionary risk management. However, this has often implied only a general and qualitative statement, and few exact and formal procedures for taking such greater risks into account have emerged. There are variable evaluations of the significance of such findings, and mixed signals have been given of the seriousness of associated risks, ranging from alarming to placating views. Pollitt (1999) judged that the steady-state body burden is not greatly increased by the short period of human breast-feeding, and thus the TDI may be sufficient to (safely) accommodate also the high intakes (per unit time body weight and time) by breast-fed babies; however, she also stated that it might be advisable to re-evaluate the toxicological database (on TCDD) to ensure that there are no implications for early postnatal development.

The extensive information on age-related aspects of dioxin risks in non-human animals may within limits be generalized to humans for specific developmental processes and specific actions of DLCs. Age-specific risks also depend on the effects considered and on individual factors, and possibly on the DLCs in question.

Research in non-human animals suggests that the developing **foetus** is a particularly highly exposed and susceptible target of dioxins (Annex 8B). There is evidence also from human populations at both high and even lower exposure levels for associations between perinatal exposure to DLCs and adverse or abnormal conditions among neonates, e.g. for birth weight, hormonal (especially thyroid) changes, neurological development and development of immunocompetence (cf. 4.2.4). These findings have been questioned based e.g. on poor specifications of exposure, treatment of confounders and natural variation, and study designs that have been deemed insufficient for establishing causality (e.g. Kimbrough et al. 2001, Kimbrough and Krouskas 2001). The epidemiological studies of effects in offspring of high Baltic fish consumers do not indicate pronounced risks, but have fundamental limitations in resolution.

The **critical time window** for some developmental effects is narrow, and for many

processes occurs very early; on the other hand, effects after early initiation may develop after great lags. The question has thus been posed (e.g. Birnbaum and Fenton 2002) whether we have been looking for dioxin effects in the wrong time perspective. Exposures at the embryonal stage are readily transferred over the placenta. Specifically, there is evidence also from human studies that *in utero* exposure may be more important for any adverse effects than lactational exposure. However, it might be assumed that marked adverse effects from short critical periods of development during gestational exposure would be captured in studies where prenatal exposure in a more aggregated fashion is used as a determinant.

Breast-feeding involves an additional load on children that may be considerable. Children at this age are also still sensitive. The almost total absorption from breast milk emphasizes this exposure (cf. Annex 7E). The load of DLCs in breast milk is high especially on primipara (Vartiainen et al. 1997b) and through long-time nursing such as is common in Finland (up to one year). Variations in exposure, depuration via lactation and half-life in the infant should also be considered. LaKind et al. (2000) showed that point estimates could result in overly conservative estimates of the contribution of breastfeeding to long-term body burdens in children.

Young individuals **after the infant stage** may still be at elevated risk due to specific exposure and susceptibility factors. These risks are difficult to distinguish from those in infants, as the later childhood risks are affected by early exposures and events, e.g. during the perinatal stage. Few long-term follow-up studies of effects of DLCs on children have yet been made, and these suffer from many methodological problems.

The individual variation in risks to the young due to particular exposures and to susceptibility e.g. due to genetic factors are difficult to elucidate and quantify. As pointed out by Gough (1991), any estimate of a heightened risk distribution among children as compared to adults leaves open the general question of whether the average child or the most exposed and vulnerable child (or some level in between these, e.g. 95 % percentile in the pdf for group-specific risk) should be protected.

According to Hoffmann and Krupnick (2004, cf. 5.4.4) children's health is demonstrably highly valued by most people, even to the level that parent's willingness to pay to reduce children's health risks is 2-fold higher than that to reduce their own health risk (see also Neumann and Greenwood 2002 e.g. on estimated costs of low birth weight).

Regarding risks in **later childhood**, many effects that have been (tentatively) found among infants have not been persistent but transient (cf. 4.2, Annex 8B, see also Stewart et al. 2003). Transient effects were seen even among the children in Seveso exposed to high doses of TCDD (Mocarelli et al. 1986). In the case of such effects, the risks to older children are smaller. Other risks such as those associated with effects on neurological tissues and functions may however be persisting. Boersma and Lanting (2000) concluded that prenatal exposure to PCBs has subtle negative effects on neurological and cognitive development of children up to school age, discernible in children to young mothers (see also Vreugdenhil et al. 2004 and 4.2). ten Tusscher et al. (2003) considered that mainly the associations between near-background levels of PCBs and negative effects on middle ear infections and lung function were persisting. In some of these studies, dlPCBs and other DLCs were not specified and effects could be due also to other PCBs (cf. Weisglas-Kuperus et al. 2000). Kimbrough et al. (2001) questioned such conclusions of clear and persistent moderate-dose effects attributable to DLCs (cf. Schantz et al. 2003).

Adolescents constitute a potential risk group, e.g. as reproductive development may be influenced. This applies not only to teenage girls whose fish consumption has been in focus e.g. in food advisories, partly perhaps due to educational reasons when targeting in general women of reproductive age (cf. 7.4.1, 8.2.1), but also to boys.

It seems that adolescents may be at heightened risk for some effects of DLCs. These effects include e.g. sex ratio shifts in offspring after exposure of young males that has been repeatedly observed (Mocarelli et al. 2000, cf. 4.2.3 and Annex 8B). This may apply also to some immune effects (e.g., Van Den Heuvel et al. 2002), although the evidence in humans is inconsistent. Evidence for such effects has been obtained at high exposures. Risks to those adolescents with high intakes of fish may likewise be elevated (e.g., Vrijens et al. 2002). Also among these, exposure to both DLCs and other adverse ingredients from other parts of the diet (as well as from general nutrition and other sources) may however

contribute more than DLCs to overall risks (cf. 5.4). The apparent lower consumption of fatty sea fish also among adolescents than among older persons provides some protection. On the other hand, this also may deprive them of health benefits of such fish, such as in the case of rickets (Molgaard and Michaelsen 2003).

Few studies of adverse effects among **children to heavy consumers of Baltic fish** have been published (cf. 4.2 and below). The reconstruction of the temporal development of exposures is a general difficulty, and is compounded by the fact that exposure estimates or classifications have often been based on reported fish consumption or, at best, on proxies for body burdens (cf. 3.5.3). Little data on the consumption of fatty Baltic fish by children of different ages have been available for these high-risk or other child populations. The levels of some immune markers have been altered among adult consumers of Baltic fish (Svensson et al. 1994), but boys at a later stage have not exhibited below-average health conditions. It may have offered and still offer some protection that fatty Baltic fish is often not consumed in great quantities by children. On the other hand, this reduces the benefits to growing children from fatty fish that may have compensated for adverse effects.

Elderly individuals can also be at particular risk depending on e.g. diet and lifestyle. Important factors here include the increased susceptibility to infections due to altered immune functions of the elderly as well as their life-long accumulated exposure. It may be that in some connections and respects the young have been too exclusively in focus. As described above, there are many general biological reasons for prioritizing risks to young; in addition, there are social and principle-level (including ethical) reasons that justify such an emphasis, e.g. the fact that the young can not grasp and avoid the risks, and that older people have already accomplished most of the valued things within their reach (including reproduction). However, there are dangers in excluding other age groups totally from consideration based on such a general emphasis. Schantz et al. (2001) stressed that their results on associations between consumption of PCB contaminated fish and neuropsychological impairments in adults may be important as fish consumption advisories focus heavily on protecting the pregnant woman and foetus or infant. Also Neubert (1997-98) and KemI and IEM (2003) pointed out the elderly as a risk groups due in part to their lower general condition.

A possible argument for targeting elderly persons within the context of the dynamics of dioxins in Baltic Sea fish is that, given the reduced emissions to dioxins, the age classes where population-level effects have occurred probably include those born and brought up in the period of top exposure, 60's and 70's, while fewer infants are now exposed to PCDD/Fs in Baltic Sea fish. Some of those effects may have been transient or, although persisting, are not identifiable anymore in the populations exposed, while others may yet become visible after a lag. This means that even with a perinatal exposure focus the possibility of later (also lagged) emergence of effects from past prenatal exposure among the middle-aged and elderly needs to be accounted for, along with other population dynamics of risks. A factor may also be that the relative amount of elderly people is strongly increasing, which is likely to further increase the population risk (=individual-level risk * population size) among the middle aged or (becoming) elderly persons. Notably, the benefits from (past) consumption of fatty fish are particularly great among elderly (cf. 5.4.4).

Among **non-human animals**, the young are at particular risk as among humans, due to high relative exposures (also through egg nutrition in birds and fish) and some higher sensitivity. Some of these risks are also more difficult to prevent, as many wild non-human Baltic Sea animals do not have easily available alternative diets to ensure protection of their young. Marked recoveries in the populations of many species that have been impaired by DLCs or potentially at risk indicate that presently the risks also to young individuals have considerably decreased. Some more subtle effects among young may however be still present, and some adverse effects may also be difficult to notice in the population, including weakened individuals.

B) Gender

Females in reproductive age constitute an important risk group both in themselves and as a mediator of dioxin risks to their offspring, e.g. through *in utero* and milk exposure. This applies to both human and non-human females.

Both genders in many species seem at risk from low-level exposure to dioxins, although in different ways. The sexually dimorphic character of effects has been noted especially

in reproductive and developmental (including neurodevelopmental) effects (cf. above).

In humans, it was found e.g. by Falk et al. (1999) that body burdens of PCDD/Fs and dlPCBs were higher in men than in women consuming fish from the Great Lakes; it was suggested that this was mainly related to the ability of women to eliminate contaminants in breast milk. As females in the Great Lakes area also have been less aware of food advisories and of risks related to fish consumption, targeted risk information strategies and measures have been developed (Ashizawa et al. 2005). Similar evaluations and recommendations have been produced in Europe regarding DLCs in fish, specifically pointing out the need to protect young female consumers of fatty Baltic fish, both at EU level (SPCFC 2005) and at national level e.g. in Sweden and in Finland (cf. 7.4.1, 8.3.2).

On the other hand, for some effects adolescent men or boys seem to be at higher risk, such as effects on the development of reproductive functions and systems, associated e.g. with male sex hormone modulation (cf. Mocarelli et al. 2000 and the discussion above). The concern for male effects is additionally based on the finding of such effects in experimental animals at the lowest dose levels commonly accepted as a basis for quantitative risk assessment (e.g., SCF 2001, cf. 4.2.2, 5.5.1).

Pesatori et al. (2003) presented the evaluation that on the basis of epidemiological studies in Seveso and other human populations as well as on animal and mechanistic data, it seems plausible that TCDD and thus possibly other DLCs exert gender-dependent effects, with evidence for some effects such as diabetes being found only or predominantly among females, others such as sex ratio alterations of offspring among males.

C) Fish consumption

High consumers of Baltic fish such as fishers and their families but also recreational fishers and their families and some other fish consumers are at higher risk with respect to DLCs than many other groups of people. The increased exposure to PCBs as a result of heavy intake of Baltic fish has been documented since early 1990's (e.g., Svensson et al. 1991, Asplund et al. 1994, Grimvall et al. 1997, Sjödin et al. 2000, cf. 3.5.3). Some studies analyzed congener-specifically both PCDD/Fs and dlPCBs (Kiviranta et al. 2002a), showing that the PCDD/F levels and also the congener patterns rather consistently reflect those in the fish.

Even in these groups the intakes of and risks from fish dioxins are only part of the overall dioxin intake and risk. The dynamics of fish consumption and of kinetics of DLC in tissues also affect body burdens and complicate the picture. Thus, Rylander et al. (1997) found that plasma CB 153 did not correlate with estimated consumption of Baltic fish, contrary to some of the other above studies, although it was related to growing up in a (East coast) fishing village; this was assumed to be due to the association between age and consumption of such fish. Moreover, along with PCDD/Fs and dlPCBs other contaminants are present in fish. Some of these add to the overall burden and risk of dioxins, but some such as non-dlPCBs may act antagonistically (cf. 4.2, 5.2.1, Annexes 8B, 8C). It is difficult due to these polyexposures to attribute any apparent risks specifically to dioxins (particularly in quantitative terms of attributability), especially until a Bio-TEq based and individual-level exposure measurements are made in the highly fish consuming study populations.

Studies of health conditions among human populations consuming above-average quantities of Baltic Sea fish, especially fatty fish such as herring, have in some cases given some slight indications of an elevated risk (cf. 4.2). These findings have included elevated mortality from skin and stomach cancer (Svensson et al. 1995b), effects on some markers of immunocompetence (e.g., Hagmar et al. 1995), reduced birth weight (Rylander et al. 1998), and weak effects on thyroid levels in women (Hagmar et al. 2001b). Most investigations of reproductive effects have produced negative or inconclusive evidence (see Axmon et al. 2004a); those of Rignell-Hydbom et al. (e.g., 2005a) suggest a slight association between sperm quality and PCBs (CB 153). By and large, the studies of human health effects in association with heavy consumption of Baltic fish, conducted in small or medium-sized cohorts, have not provided consistent, convincing and striking results.

In many cases, the precise exposure has not been defined but has been based on approximations. In some studies measurements of body burdens among subsets of the investigated cohorts have been used. However, also in these cases usually dlPCBs and other DLCs e.g. as TEqs

have not been specifically measured. The effects e.g. on immune responses and thyroid levels observed by Hagmar et al. (1995) and Hagmar et al. (2001b), respectively, may thus have been due to other contaminants and causes. A problem common to all these studies is the reconstruction of the temporal development of intakes and thus of cumulative and peak exposures.

These uncertainties apply to studies of human health effects from consumption of DLC-contaminated fish in other regions, too. For instance, high consumption of contaminated fish from the Great Lakes has been reported to constitute a risk factor for some health disorders such as low birth weight (e.g., Karmaus and Zhu 2004) and impaired intellectual development (cf. Annex 8B). However, Weisskopf et al. (2005) reported that p,p'-DDE, not PCBs, was associated with low birth weight in offspring to mothers consuming sport-caught fish; they also noted that beneficial effects of fish consumption and some unknown contaminants may have caused inconsistent results on effects of PCBs in other studies (cf. 5.4.2). As to effects on neurological development from perinatal exposure to fish contaminants, several limitations for drawing conclusions have been presented. Among the Inuit, seafood diet has been found to be a risk factor for recurring infections (e.g., Dallaire et al. 2004). Also associations with slight neurotoxicity have been reported (Grandjean et al. 2001) but are complicated by MeHg (see 5.4.2). Usually only PCBs have been studied in both regions, and also other contaminants shown to be associated with such effects have been present at high levels (e.g., Karmaus et al. 2001). In the case of Inuits the seafood diets have differed from those in the Baltic also in terms of general quality (e.g., due to consumption of marine mammals).

Yoshida et al. (2000) assessed that the margins of safety for human health risks from dioxins in Japan were particularly narrow for heavy consumers of fish, based on neurobehavioral effects on infants and foetuses, while the risks for reproductive dysfunctions and cancer were not found to be exceptionally high even among these consumers (the benchmark dose for reproductive effects was however higher than that derivable from some other rodent studies).

Not only risks but also benefits from dioxin-contaminated (fatty) fish vary between populations. Thus, also those at risk from reduced fish consumption need to be assessed. For instance, the particular groups of beneficiaries from such fish do not necessarily coincide with the particular risk groups. This issue has been discussed at some length below (5.4.4).

5.2.3 Temporal dimensions and variations of risks

General points

Resolution of the significance of risks and of patterns and dependencies in them is obscured by multi-dimensional temporal variation, both systematic and long-term and erratic and short-term, causing noise that prevents observation of signals. Much of the temporal variation can be aggregated to cumulative body burdens and chronic outcomes, due to the slow dynamics of DLCs in tissues. However, comparisons between systems and conditions may still be misleading because of temporal variation.

The temporal dimension of risks can be divided e.g. in the following aspects:

- Long-term **lags**
  - slow transport and cycling in the environment
  - accumulation in biota including humans
  - chronic and lagged toxic impacts
  - inter-generational effects
  - inertia in control systems along the risk chain
- Actual temporal **variations**
  - fluctuations
  - regular rhythms, e.g. diurnal and annual
  - sudden contamination episodes
  - specific time windows of exposure
  - transient events and processes, e.g. effects
  - time trends and historical comparisons of risks
- **Frequencies and durations** of events (c.f. rhythms, former point) within the subsequent main stages of risk chains:
  - emissions (continuous or intermittent)
  - exposures (e.g. fish consumption)
  - adverse effects (e.g. diseases)
  - abatement measures (technical and institutional).

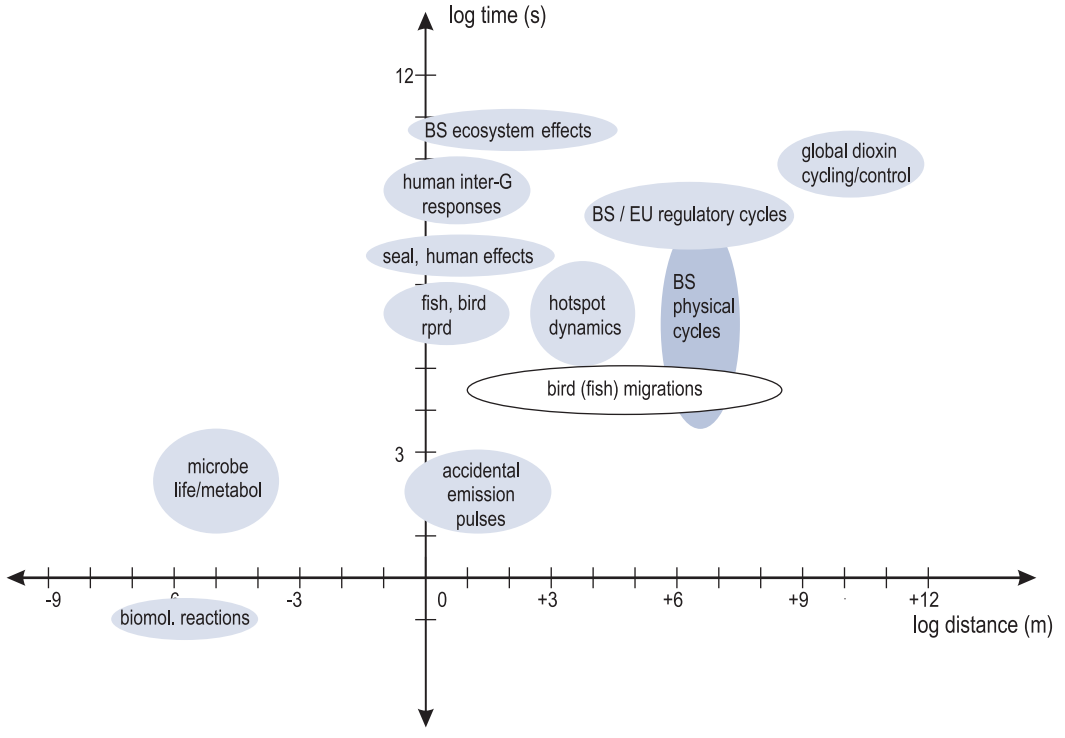


Fig. 15. Conceptualization of time-space continua in risks associated with dioxins in the Baltic Sea, based partly on Suter (1993).

Temporal variation in DLC risks thus occurs on many different **scales**, from seasonal (e.g. due to changes in fat content, cf. Rose and Startin 2003) to longer time scales (Fig. 15). Temporal variations in risks moreover occur at several stages, from environmental levels or intakes to alterations in later stages of the risk chains, such as in pharmacokinetics and effects.

The **long-term** aspect is particularly important. This is related to the defining characteristics of DLCs and their effects: persistence, accumulation and chronicity. Likewise, management requires a long-term approach. Long-term emphasis is also needed as a counterweight to the dominance of acute episodes that tend to drive the attention and action (cf. 8).

The temporal aspects interact with **geographical** variability and variation between receptors (Fig. 15). The levels of DLCs in biota and the environment are subject to largely irregular and poorly known variations (cf. 3), as are the human and non-human animals subject to risks (cf. 4). These variations may blur trends or cause trends that are apparent but not real. This variability is partly due to the dynamics of the system under study, partly due to the study process, including data handling and monitoring. Apparent changes and trends in levels of DLCs and subsequent risks are caused by changes in analytical methodology and calculation methods for TEqs, and the temporal development of measurement results can thus sometimes not be clearly reconstructed or evaluated from reports.

Some effects of DLCs that already have taken place or will take place are of **transient** nature. This has been noted in many occasions especially in biochemical responses and in alterations of hormone states but also other effects of DLCs (cf. 4). However, some sudden or transient events have significance in a longer term. For instance, the relative importance of the sudden release of and exposure to mother's milk in relation to the life-long risks to the infant may be accentuated due to particular windows of sensitivity during early development. There is also the possibility that an effect transient *per se* has initiated a sequence of events of more long-lasting (and irreversible) nature that may lead to later adverse effects.

Aylward and Hays (2002) reconstructed by using a simple pharmacokinetic model that the absorbed intake levels of TCDD must have decreased by >95% from 1972 to result in the observed decrease in human lipid levels, the bulk of the decrease occurring before 1980. They predicted that mean TCDD levels in the general (US) population will decrease to 0.5-1 ppt lipid by 2015, even if intake levels do not decrease further, and that although fewer data over a

shorter period are available for other PCDD/Fs, they indicate substantial decreases as well, with general population TEq lipid levels currently at least 4-fold lower than in 1970 and still decreasing. As estimated by Hays and Aylward (2003), the decline in body burdens in the general population is expected to continue for at least the next two decades even if average intake levels remain constant, due to the increase in the part of population that never experienced the top exposures. It is not clear whether the same will be true of consumers of Baltic fish.

Sweetman et al. (2000b), based in part on Alcock et al. (2000), found that peak levels of PCDD deposition occurred in late 1970s but those of PCDFs already in 1930's. Sweetman et al. (2000b) estimated, assuming steady decline, that by 2010 the intake of an average (UK) consumer over a 70-a lifespan would comply with the WHO lower guideline value of 1 pg I-TEQ kg^{-1} bw d^{-1}. However, considerable variation in the pace of the reduction was seen e.g. between high and low intakes and between age groups, being slower among children due to initial high perinatal burden. Intakes from breast milk for infants up to 10 mo would exceed the WHO guideline beyond 2020. The breast milk levels of 4-PeCDFs were estimated to be halved in 15 a, consistent with the observed 12 a in Swedish breast milk. Sweetman et al. (2000b) stressed that the development may differ between congeners. Declines in the Baltic are slower than in some other systems more directly dependent on airborne emissions, such as agricultural systems, but in relative and general terms the results may be applicable.

Development of risks from DLCs in Baltic Sea fish

Generally speaking, **exposures** to DLCs also in the Baltic and its fish have **decreased**, as seen in the levels of PCDD/Fs and dlPCB in biota. As detailed in 3.4 and Annex 6B, body burdens of TEqs have typically been decreased several-fold in both Baltic herring and other fish and in the important and representative animals consuming Baltic fish, including guillemots, seals, white-tailed sea eagles, and humans. Czub and McLachlan (2004) simulated the dynamics of the indicator CB 153, estimating that the peak exposure of infants in Baltic fish occurs c. 10 a after the peak in environmental levels (Fig. 16). In fact, the lag may be still longer depending on the age of reproduction and the onset of lactation. Changes in exposures are affected also by fish consumption; intakes of fatty Baltic fish may have decreased at accelerated pace by risk management and warnings, even with harmful effects (cf. 5.4.4).

The declines in exposure and risk levels **vary by region** and species. For instance, the comparison of Nyman et al. (2002) with the data of Helle (1976) from 1973 suggested slow declines in Bothnian Bay adults ringed seal PCBs, while in juvenile animals, greater declines have been reported (Blomqvist et al. 1992).

Declines in levels of some DLCs in Baltic Sea biota have **slowed down** and in some cases may have already levelled out (cf. 3.4.2). Olsson et al. (2005) showed that the decrease of PCDD/F-TEq levels e.g. in Baltic guillemots seems to have reached a plateau already in mid-1980's, and the decrease from recorded top levels was proportionally not so great as that of ΣPCBs. On the other hand, this implies that the risks due to CB 126 (and other dlPCBs) also are likely to have decreased particularly greatly, at rates nearing those for ΣPCBs (10-fold in guillemot).

The relative **speed of decrease** of the levels (especially body burdens) of PCDD/Fs and dlPCBs is of interest e.g. for the future development of the risks and of their profiles and determinants. With poor information on the relative persistence of the key DLCs in the key species of concern such as seals and humans, it is however difficult to come up with predictions. Among both PCDD/Fs and dlPCBs, congeners of variable persistence (and bioaccumulation) are found, so that the relative importance of e.g. PeCDD and CB 156 and 169 is likely to increase. There are indications from ringed seal in the Arctic that the importance of dlPCBs in relation to PCDD/Fs has decreased from 1999 to 2003 (Riget et al. 2005). In the Baltic, the requisite congener-specific and representative trend data of levels of PCDD/Fs and dlPCBs in these or other non-human animals are not available (cf. 3.4.2).

In **humans** in the Baltic Sea region, the body burdens of DLCs have decreased in the population, e.g. in the levels of WHO-TEq_{DFP} in Swedish mother's milk by ca. 10 % yearly, to ⅓ from 1972 to 1997 (Norén and Meironyté 2000). It is more uncertain what the development has been in heavy consumers of Baltic fish, as only a few and not necessarily comparable cross-sectional studies have been made (Svensson et al. 1994, Asplund et al. 1994, Becher et al. 1995, Kiviranta

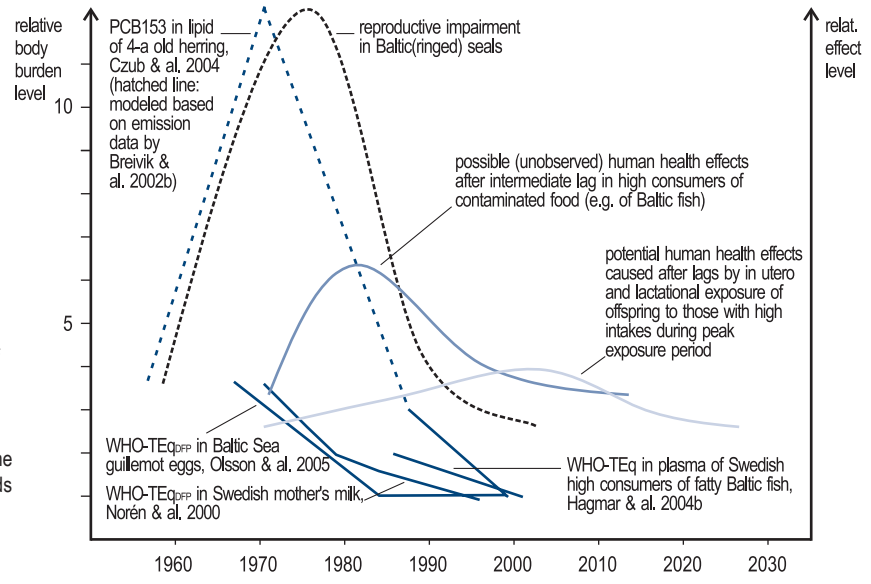


Fig. 16. Reconstructions of the temporal development of some representative estimates of body burdens of dioxin-like compounds and indicator PCBs in Baltic herring and its consumers (straight lines), and hypothetical courses of some biological responses to resultant exposure (curved lines). Body burdens are given as relative values (year 2000=1), and data representations have been simplified. Cf. text and Annexes. Note that quantitative levels and time courses of effects due specifically to dioxin-like compounds are difficult to establish and may fall within background variation.

et al. 2002a, cf. 3.5.3). In a cohort of Swedish men consuming variable amounts of fatty Baltic fish, the body burdens of CB 126 declined strongly from 1987 to 2002 while those of PCDD/Fs did not (Hagmar et al. 2004b). The time trends of DLC levels in these population segments reflect both the development of levels in herring and salmon, being key contributors of DLCs in such diets (see 3.4, esp. Lind et al. 2002 and SPCFC 2005), and congener-specific accumulation and retention.

Predictions on the basis of observed trends are constrained as the decline stage has been replaced by levels fluctuating around a plateau, for both PCDD/Fs (TEqs) and dlPCBs. The highest chlorinated PCBs and PCDD/Fs and the most toxic and metabolically stable congeners will be particularly persistent. Further declines may thus take long. It has been estimated on the basis of a simple mass balance model that the EOCl content in the Baltic will be halved in c. 50 a (Wulff et al. 1993); the relative persistence of DLCs is still greater. The response time of the Baltic Sea to decreased loading by PCBs has been estimated to be in the range of 20 a or more (Axelman et al. 2001, Wania et al. 2001), due e.g. to internal loading from sediments. Thus, the levels in Baltic Sea dependent biota may now be 'near' long-term (<100a) residual levels.

Declines in the levels of **ΣPCBs** in some systems have taken place already in early 1970's (e.g., Moilanen et al. 1982, Olsson and Reutergårdh 1986, Roos et al. 1998, Vuorinen et al. 1998b). In cod liver, Fromberg et al. (2005) reported clear declines, whereas Falandysz et al. (e.g., 1994d,f) earlier found that the levels in the Baltic had decreased only slowly. PCBs include congeners that are less persistent than dlPCBs, especially low-chlorinated congeners (Annex 3) that may thus induce more rapid removal of ΣPCBs than of dlPCBs. On the other hand, some of the highly chlorinated non-dlPCBs may be even more persistent and bioaccumulative than some dlPCBs. Based on the rather consistent declines in the levels of ΣPCBs in biota from early 1970's and on the correlations between these and dlPCBs it may be assumed that the time trends also in the levels of dlPCBs, and thus in PCB-TEqs and even total TEqs, have been declining.

The declines also in these exposures vary. Olsson and Reutergårdh (1986) noted that ΣPCBs had declined in herring from Southern Baltic but not in Bothnian Sea; their general evaluation however was that the Baltic seemed to have started to recover more quickly than the most gloomy prophecies gave reason to believe.

In **coastal** systems the rate of recovery from dioxin contamination can vary considerably. Hagen et al. (1997) reported that dioxin levels had declined in crustaceans and molluscs by 10-90 % and in sediment by 60 % in British Columbia coastal waters in 5 years after the reduction of emissions from pulp and paper mills. In a Southern Norwegian fjord, PCDD/F and dlPCB levels e.g. in cod liver declined in 1991-2001 c. 10 % yearly (Knutzen et al. 2003) but then levelled off; a tentative target level was estimated to be reached as late as in 2015-20.

The progress and variations of recovery are influenced along the risk chain by differences in relative cuts of emissions or immissions, by changes in the cycling of DLCs in particles

(depending e.g. on hydrodynamics and resuspension) and by changes in food chains (depending e.g. on biomagnification in the key species).

As related by Hansen (2003), the AMAP Report (2002) concluded that there are likely to be minor decreases in POPs in the tissues of Arctic human populations by 2010 and major decreases by 2030, depending however on prompt ratification and implementation of the Stockholm Convention and other multinational environmental agreements. An improved description of these dynamics and of their key drivers especially for 4-PeCDF and CB 126 would be important to enable predictions of the development of dioxin risks and for setting in and following up targeted abatement measures. In particular, combined descriptions are needed of the dynamics in exposures (intakes and body burdens) and in effects, in both human and non-human populations.

As to the temporal dimensions of biological **effects**, as discussed above, several factors can diminish dioxin risks further. On the other hand, some factors call for prudence in declaring the risks from dioxins as definitely diminished:

- There are **latencies** in some effects. This is true of cancer and some chronic non-cancer effects, especially on development. Such lags can be associated with temporal distributions of exposure. Eskenazi et al. (2003) for instance concluded that although little evidence was found for abnormal birth outcomes in Seveso, it remains possible that such effects are yet to be observed, as the most heavily exposed women were the youngest and the least likely to have yet had a pregnancy. Salvan et al. (2001) showed that by including a latency of 10 a and covariates related to duration of exposure to an analysis of the dose-response function of TCDD in occupational cohorts, a slightly but significantly elevated risk for all cancers was obtained already at doses 2-fold the background. However, their model is less reliable in the low-dose region and disregards variations in background exposure associated with age. Scheuplein and Bowers (1995) noted that even after incorporation of exposure duration, it was unlikely that the main epidemiological studies have adequate power to detect the common cancers caused by TCDD in rodents.
- There is a risk of **inter-generational** effects that may be realized only in later generation(s) after exposure. This has been noted in rodent studies of TCDD effects also at low perinatal exposures (Mably et al. 1992c). Indications for such second-generation effects in humans in association with perinatal exposure to dlPCBs were published for testicular cancer by Hardell et al. (2004).

There is furthermore a possibility of (relatively) **sudden** increases in risks and particularly exposures, due e.g. to the following causes:

- Pulses of additional DLCs into the Baltic, e.g. in **floods** like those in 1997 and 2002 in Central European rivers including Oder (Witt et al. 2001, Kowalewska et al. 2003). Such floods may speed up the flux of DLCs from the catchment that would under normal conditions take place later and may thus also increase the absolute fluxes as degradation in the catchment is reduced, as well as change the cycling of DLCs e.g. by increasing their availability. Sudden increases in emissions and discharges may take place through wars and other large-scale abnormal social or technological disasters and conditions.
- The DLCs stored in the sediments may be mobilized in exceptional **storms**. Assuming the storm frequency and severity will grow, e.g. as a result of climatic changes, more of the DLCs in the bottom may be mobilized. Sudden disturbance of sediments due to technological exploitation measures may also take place.
- Relatively small risks exist for seismic and volcanic upheavals in the Baltic region.
- The risk of large-scale catastrophic DLC mobilization by near-Earth object (NEO) impact is still smaller (a possibility of one every 50 Ma for an impact on the Baltic Sea surface with regional effects such as tsunamis and sea bottom slides, Stuart and Binzel 2004).
- There may also be sudden natural and human factors that will **reduce** influxes, exposures and risks. They may however not be so likely as sudden processes that would tend to remobilize the DLC pools.

5.2.4. Geographical dimensions and variations of risks

General considerations

Geographical variation in risks from DLCs also in the Baltic Sea exists in all **stages** of risk formation, from sources of DLCs onwards to emissions, immissions, cycling, exposures and effects (cf. Fig. 4). This may be taken further to encompass geographical variability in the subsequent stages of other impacts from DLCs (in fish), in risk management, and in counter-veiling risks from management. For instance, the simultaneous benefits from consumption of Baltic fish vary also regionally depending e.g. on the distribution of the populations experiencing these benefits.

The geographical variability in risks occurs at multiple **scales**: between the Baltic (and its catchment) and other seas or regions; within the sea between sub-basins; and so forth. On a more detailed scale, emissions from certain local sources such as some industries (e.g. metal industries) along the coast influence the geographical pattern of dioxin risks in the Baltic (see Näf et al. 1992). In particular, rivers discharging to the sea such as the river Kymijoki integrate impacts and risks from a large area but from the point of view of the sea constitute localized sources.

The **sources** of DLCs to the Baltic vary by country, region and locality, due e.g. to differences in chemicals and processes forming and emitting DLCs (cf. 3.2). The great emissions of PCDD/Fs in and from Poland are a case in point.

Initial geographical variations in immissions subsequently even out to some degree in the Baltic. This is a thermodynamic necessity that is caused by the interacting dispersal (resulting from entropy growth) and fugacity driven concentration in certain compartments such as sediments and notably biota. As explained above, variations in these processes are due partly to inherent properties of the DLC molecules and partly to external factors in their environment, such as the biological activities of the cycling and actively accumulating organisms.

Geographical differences in the **cycling** of DLCs in their turn contribute to and modify the pattern of risks. At the next stage, geographical differences in risks are caused by differences in intakes and **exposure** patterns (e.g., Lundstedt-Enkel et al. 2002). For instance, some species that are for some reasons particularly exposed to dioxins, such as ringed and harbour seals, are present in only part of the Baltic. Among humans, some risk groups may be distributed unevenly between the different geographical areas, e.g. due to regional differences in preferences for consumption of fatty Baltic fish.

There are important differences between **sub-areas** and basins of the Baltic due also to general ecological conditions such as salinity and stratification. For instance, the sustained high growth rate of herring in the Gulf of Bothnia has been considered to be due to such factors (Ojaveer and Lehtonen 2001), and further influence the cycling, subsequent exposure and risks of DLCs.

Effects of a given exposure may differ due to variability e.g. in the sensitivity of receptor organisms, and notably depending on the biological effect in question. This may be related to the greater stress in certain areas due e.g. to climatic and other external reasons; also particularly susceptible geographically restricted populations exist. Ecologically, these may not have significance for the overall well being of the meta-population. In human toxicology however, even such smaller geographical variations in risks may be relevant.

For assessment of geographical variations, the following factors for the various compartments or matrices have to be taken into consideration:

- In **sediments**, few analyses including both PCDD/Fs and dlPCBs have been made. Many analyses are from bottoms near coasts that are not representative of general sediment quality in the sea (cf. 3.4.1, Gustafsson et al. 2003). Moreover, the levels vary depending e.g. on the sedimentation and other characteristics of the bottom, and on carbon content that often has not been determined so normalization of the concentration data can not be made.
- In **fish**, mainly herring data are available. It has been estimated using geostatistical methods (semi-variograms) that the data on herring PCDD/Fs in Swedish Baltic coastal areas reflect conditions in an area within 100 km (Bignert et al. 2005). This however depends on the age of the stock, as particularly older herring may migrate and thus represent conditions in a wider area (e.g. extending from Gulf of Finland to Baltic Proper). Still, some 10 local stocks of spring-spawning herring with different ecology exist in the Baltic (Ojaveer and Lehtonen 2001, cf. Parmanne et al. 1997).
- In **other biota**, the resolution of geographical trends depends on the locality of the species. Those of most interest, humans, marine mammals and white-tailed sea eagles are relatively migratory. This causes aggregation of local variations especially for adults.

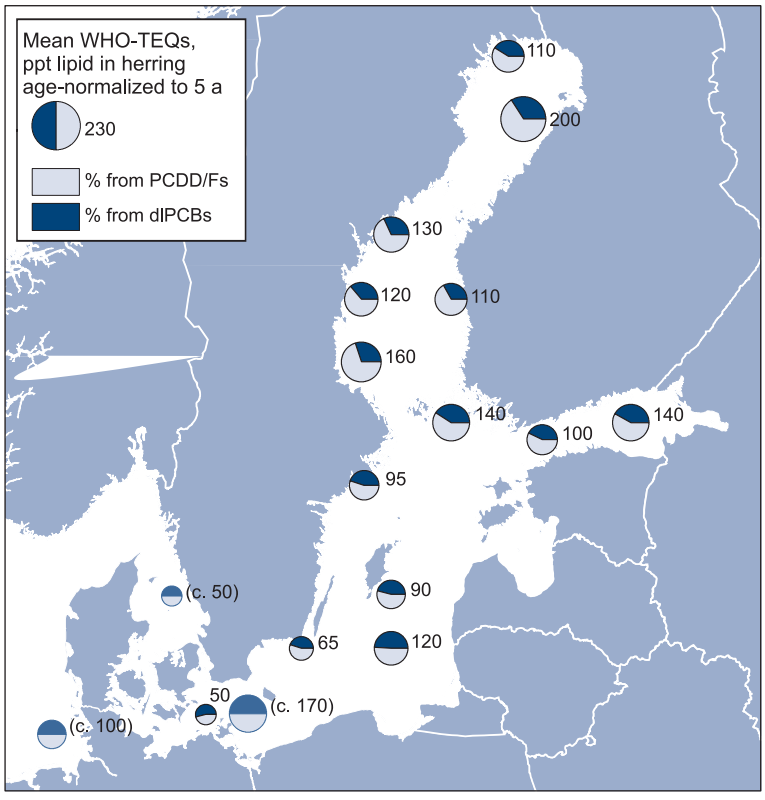


Fig. 17. Regional variation of total dioxin-like toxicity to mammals due to PCDD/Fs and PCBs in Baltic herring at the turn of the millennium, as illustrated by average levels of lipid-based WHO-TEq$_{DFP}$ concentrations, normalized to an age of 5 a, in the various stocks according to aggregated Swedish (SNFA 2005) and Finnish (Hallikainen et al. 2004) data. The reference value from North Sea is for herring of unreported age and location caught in 1996-97 (Karl et al. 2002). This value and those for Kattegat and Rügen (SNFA 2004) are estimated from PCDD/F levels only. Note that lipid and age normalized values give a different picture than non-normalized wet weight based values. For the procedure and interpretation, see text.

Variation within the Baltic Sea region

The geographical variations in the exposure to and risks from DLCs in Baltic fish can be illustrated by the mean levels in the various regional stocks of herring, as approximated by WHO-TEq$_{DFP}$ values in the most extensive and internally coherent sets of data, those by Hallikainen et al. (2004) and SNFA (2005, cf. Bignert et al. 2005). Data of sufficient representativeness, reliability and comparability are available only for this species (Fig. 17). In order to enable a comparison between different areas, only data on samples taken around the year 2000 are used. As the lipid contents and age of herring correlate strongly (although variably) with levels of DLCs, only lipid based concentrations are used and normalized to an age of 5 a on the basis of nonlinear potency regression equations derived separately for Gulf of Bothnia and Gulf of Finland-Baltic Proper: WHO-TEq$_{DFP}$ (ppt lw) = 16*Age [a]$^{1.3}$ (r^2=0.79) and 24*Age [a]$^{0.85}$ (r^2=0.48), respectively. The use of lipid-based data also reduces the variation due to season (spring or autumn herring having great difference in fat contents), tissue (muscle with or without skin fat in Swedish and Finnish data, respectively), and gender. Age rather than physiometric data can be assumed to be more distinct from lipid contents as an explaining variable. It may be noted that in the Swedish data the catch season and in the Finnish data the gender has not been reported, and that neither data have good coverage of herring stocks in Southern Baltic.

It can be concluded from this analysis that on lipid basis the WHO-TEq$_{DFP}$ levels in Baltic herring have been c. 100 ppt, being notably higher (twice that) only in Finnish Bothnian Bay coast and possibly in the South-Western part of Swedish Bothnian Sea coast, and lower (half of that) in Southern Baltic. It must be stressed that due to the many sources of variation and particularly the dependency on lipid contents (and other correlated variables), it is not as meaningful to compare wet weight based levels, as is sometimes done (see also Jones et al. 2001), perhaps due to the focus of the fish limit values on this unit of measurement.

Bignert et al. (2005) analyzed Swedish data on spring or summer herring in the Gulf of Bothnia and Northern Baltic Proper after weighting the levels (in WHO-TEq$_{DFP}$ ww) by inverse of distance (up to 75 km, being within the range for spatial autocorrelation in these data) to account for geographical variability. They preliminarily reported that the highest levels were found in Southern Bothnian Sea, but using lipid based values the area of higher levels extended to Northern Baltic Proper. However, the migration of herring, especially of older age of most interest for human consumption, makes it difficult to

analyze DLC contents on a scale below 100 km (cf. Parmanne et al. 1997, ICES 2005c).

In general, considering the overall variation in herring levels due to multiple causes, the above differences, even if reflecting real conditions, do not seem very significant. The uncertainties surrounding the TEq levels (being at least half-order of significance) additionally reduce the information content in demarcations between areas differing e.g. by only 50 % in TEq levels.

Resolution of gradients and trends in salmon is difficult due also to the migration of biota and to other sources of variation, notably age and condition in connection with spawning (cf. 3.4.2).

In Bothnian Bay where the levels of some DLCs in some biota are higher, possibly due to the higher loading per unit area in relation to biomass or to more efficient cycling of DLCs also from sediment (cf. 3.3.1), at the same time organisms are particularly vulnerable due to osmotic and climatic stress. In Southern Baltic, the more eutrophic and marine state of the sea may correspondingly in some respects decrease the risks in relation to those in other parts of the sea also partly regardless of exposures.

Hotspots have been identified on the basis of emissions or immissions. Conceptually however, emphasizing the actual outcomes of such local contamination, hotspots on the level of risks may be found also in other areas, due e.g. to particular local exposures and susceptibilities to dioxins. Thus, the location of hotspots based on DLC occurrence is not necessarily the same as the location of hotspots based also on factors in subsequent stages of the risk event chain.

A system of identifying and managing hotspots in the Baltic Sea area, established under HELCOM, has listed over 100 hotspots around the sea (Annex 3). The development of these hotspots is monitored and assessed with regard to abatement efforts, including a de-listing procedure. These hotspots are mainly based on major nutrient discharges and direct industrial discharges, with little consideration of toxic substances (apart e.g. from oil spills and some heavy metals), despite the importance of such substances; this may represent the inertia of such organizations in tackling 'new' challenges. It is notable that e.g. the key Finnish local area of dioxin contamination, representing an important emitter of PCDD/Fs to the Baltic as a whole, the river Kymijoki, has not been included in these HELCOM hotspots.

Differences in levels of DLCs and related compounds between the Baltic and other sea areas

The levels of PCDD/Fs in Baltic Sea biota were initially compared above (3.4.2) with those in the biota of other seas. It could be concluded that the Baltic is relatively strongly contaminated in some comparable species (cf. Annexes 6B and 7B). The relative contamination of the Baltic however varies depending on the studies, samples and congeners compared and on the expression of levels (fat or wet weight basis). Some of such differences or similarities may be due to factors such as age, reproductive stage, sex and nutrition, and may thus not allow a representative comparison of the contamination situation in these seas.

In general the PCDD/F level, as WHO-TEq_{DF} ww, in comparable age classes of Baltic **herring** increases toward East, being 5-fold greater in the Eastern part of SB than in Skagerrak (Karl and Ruoff 2004), and also toward North (Bothnian Sea and Bothnian Bay). de Swart et al. (1994) likewise found that the intake of DLCs by harbour seals fed Baltic herring was 10-fold higher than that in seals fed Atlantic herring; the same was true of dieldrin intakes. However, on lw basis (and after age normalization) the regional differences are much reduced, and levels of DLCs relatively close to those in the Baltic may be seen also in North Sea herring (Fig. 17, Table 21). Also the levels of key dlPCBs such as CB 126 in herring have been in some studies high in the North Sea in comparison with the Baltic (de Boer et al. 1993, cf. Jansson et al. 1993, Kiviranta et al. 2003).

In **harbour porpoise**, other nearby seas including the Norwegian coast have exhibited almost similar levels to those in the Baltic (Ross et al. 1996c, Berggren et al. 1999, Bruhn et al. 1999, cf. Karlson et al. 2001), while those in Arctic waters have been lower. Based on >200 samples, Fromberg et al. (2005) reported that the levels of ΣPCBs had declined similarly also in **cod** liver, a tissue representative of herring-based food-chains, from the North Sea and the Baltic Sea. Bignert et al. (1989) found that the WHO-TEq_{DF} levels in juvenile Baltic **ringed seals** exceeded those in Atlantic ringed seals 5-10-fold while the levels in Baltic common (harbour) seals exceeded those in the Atlantic only slightly (cf. Annex 6B).

A summarizing comparison of the levels of dioxins and dlPCBs in herring, salmon and harbour porpoise in the Baltic and in other marine areas is presented (Table 21). To facilitate comparisons, both

Table 21. A comparison of contamination by PCDD/Fs and dlPCBs in representative animals in Baltic and nearby coastal sea areas, based on data sets selected for maximum comparability as to sampling period, sampled specimens and analytical methods.

Marine area	Levels of dioxin toxicity (means, in parentheses ranges), in pg WHO-TEq (mammalian) g^{-1}				
	herring, muscle **lipids**	**herring**, muscle **wet** weight	wild **salmon**, muscle **lipids**	wild **salmon**, muscle **wet** weight	mature male **harbour porpoise**, blubber[c]
Baltic Sea	50-200[a,d]	3-10[a], 10-30[d]	200[d]	20[a], 20[d]	80[e], 200[c]
North Sea	80(30-100)[b]	8[d]	200[d]	20[d]	
Atlantic coast					100[c]

References: [a]Hallikainen & al. 2004, for 5-a old herring (cf. Fig. 17); [b]Smith & Gangoulli 2002, based on MAFF 1997, 1999; [c]Berggren & al. 1999, data for porpoises caught 1988-90, including PCBs 77, 126 and 169 (not CB 81), 1-ortho CB 118 (not 105, 114, 123, 156, 157, 167 and 189) and di-ortho CB 180 (not CB 170) among those PCBs given TEFs, significantly higher BS levels found only for TCDF and 4-PeCDF and ΣPCBs; [d]SCOOP 2000, SNFA 2003, 2004, 2005; [e]Falandysz & al. 2002b, data for porpoises caught in 1992.

lipid and wet weight based data are given (cf. Fig. 17). The data by Berggren et al. (1999) on harbour porpoises are included although not covering all dlPCBs, as they illuminate regional variations in dioxins and PCBs in a marine mammal, include statistical analysis of the significance of differences, and demonstrate the importance of the weakly dioxin-like 1-*ortho* CBs in this species.

Generally the PCDD/F and PCB levels, in terms of dioxin toxicity, have been higher in the Baltic than in other nearby seas. However, the Baltic contamination level was not always significantly higher. The levels in 1996 German Baltic herring were also lower than those in 1996-97 German North Sea herring on lipid basis (Karl et al. 2002). In salmon, the levels have been comparable in Baltic Sea and UK (North Sea) samples from approximately the same period (mid-90's). In the blubber of mature harbour porpoises the Baltic Sea levels were not much higher for TEqs or the key contributors CBs 180 and 118 in the data of Berggren et al. (1999), and were still lower in the study of Falandysz et al. (2002b, cf. Annex 6B).

Ross et al. (1995) reported c. 3-fold higher I-TEqs (70 % from 1-*ortho* PCBs) in the blubber of **harbour seals** fed Baltic or North Sea herring, i.e. a smaller difference than in dietary intakes (see above). Bruhn et al. (1999) found that the central statistics of the levels of ΣPCBs and of PCDD/Fs, also as TEqs, in mature harbour seals in the North Sea were equal to or even higher than those in the Baltic (cf. above 4.3.2).

Nyman et al. (2003) found 4- to 5-fold higher levels in the Baltic than in Svalbard **ringed seals**. However, levels were reported by AMAP (2002) for ringed seals in the Kara Sea that were within the range and near the average reported by Nyman et al. (2003). No congener-specific data have been reported in these sources to facilitate comparison of levels in the Baltic and the Arctic. Some of the data sources in the AMAP (2002) assessment are also not published, or their data characteristics can otherwise not be evaluated. On most of the species studied in the Arctic and occurring in the Baltic, such as herring gull, common guillemot, whitefish, Northern pike and ide, only unpublished sources were cited in AMAP (2002).

There is little comparable information on the geographical variation of **dlPCBs** in fish from the Baltic and nearby seas. Jensen et al. (1977a) reported based on samples from 51 locations on the Swedish coast that the highest levels of ΣPCBs in pike were found in Bothnian Bay, those in eel in Skagerrak. However, within the sub-regions considerable variation was present. It may be assumed that the distribution of dlPCBs and of WHO-TEq_P is similar. Lundgren (2003) reported that lowest concentrations of some (weakly) dioxin-like PCBs (CB 105, CB 118, CB 156) were found in the sediments and benthic species in Northern Gulf of Bothnia, and highest levels in the South, although this trend was not so clear for four-horned sculpins (*Oncocottus quadricornis*).

In Northern European marine **sediments**, some comparisons can be made specifically of dioxin contamination based on Bio-TEqs. However, Stronkhorst et al. (2002) demonstrated that on the Dutch North Sea coast the DR-CALUX response varied between 0.2 and 136 ng Bio-TEq kg^{-1} dw; c. 90 % of the samples failed the target value of 2 ng TEq kg^{-1} dw while c. 10 % and 3 % exceeded threshold values for offshore disposal of dredge spoil (25 and 50 ng TEq kg^{-1} dw, respectively). With such skewness in frequency distributions, spatial variations in sampling greatly affect the picture.

The PCB contamination situation of the Baltic can be tentatively compared with other seas also based on the levels of the non-dioxin-like indicator CB 52 in continental shelf sediments (Gustafsson et al. 2003b). On area basis, the sediments in the Baltic are much more contaminated than those north and south of main emission areas (30-60° N).

However, even considering the area of continental shelf, some other coastal or inland sea areas such as the Mediterranean, the Black Sea and North Atlantic including the North Sea are roughly in the same class, even worse. This was noted also by Konat and Kowalewska (2001) in their comparison of data on PCBs. A key difficulty in comparisons is the variation in concentration ranges and the selection of representative statistics. Due to the skewed distributions, in most cases median values would be most suited for comparisons, but have been seldom reported.

As to contaminated inland waters, the levels of PCDD/Fs in frequent fish consumers in the Great Lakes area (at 10-20 pg I-TEq g^{-1} lw, Falk et al. 1999) are considerably lower than those in heavy consumers of fatty Baltic fish (e.g., Kiviranta et al. 2002a).

Risks through animal production systems

As described above (3.3, 4.3.4), the risks from DLCs transmitted by animal production are directed both to humans consuming these animals, which have been the main concern and indeed a key driver in dioxin risk management, and to the production animals themselves (see SCAN 2000). The animal production systems may also contribute to the geographical distribution of DLCs and associated risks.

Including this route, it may be said in general that the absolute (and aggregated) human exposure will be greater than from direct consumption of Baltic fish. It is however difficult to estimate the risks from DLCs entering the production animal food chains from Baltic fish based feeding-stuffs, due to limitations in the data on the uses of such feeding-stuffs and their variations and on the production animals receiving and transmitting them (cf. 3.3.1). Relative exposures of humans may be smaller through a dilution effect, if the levels of DLCs in such foods produced by fish based feeding-stuffs will be lower than in Baltic fish itself. However, such dilution of and contamination by DLCs accumulated in Baltic fish in food production systems is problematic with regard to total fluxes, exposures, risks and management (cf. 8.3, 8.4).

Some initial evaluations of the associated ecotoxicological risks (to the production animals themselves) are presented below (5.5.4).

5.3 Uncertainties of risks

5.3.1 Types, qualities and characteristics of uncertainties in risks and benefits

General points

Many kinds of uncertainties (cf. 1.4.3) influence the assessment and management of risks from DLC. These uncertainties are not only a negative thing and something that could be used as an excuse for doing nothing. Upton (1994) proposed that agencies increasingly subject their dioxin risk assessments to formal uncertainty analysis.

The irregularities and uncertainties found regarding DLCs, in all stages of risk chains, although considerable and varied – even greater and more varied than often realized - do not mean there are no generalities to be found and only noisiness in data and lacking clarity of theory. It must be recognized that, as put by Rozman and Doull (2001), claiming that exceptions exist stifles any attempt at generalization, which eventually is needed to develop theory, which in turn is a precondition of a scientific discipline. Thus, generalizations (and simplifications) are needed in the face of all the variability, irregularity, complexity and uncertainty of risks.

The following points in uncertainties related to dioxin risks must be stressed:

- Uncertainties are not only caused by measurement variation (and, within these, only or even predominantly variation due to laboratory analysis), but also by models
- Still more fundamental uncertainties are involved in decision rules. Many of these uncertainties may be termed (or may be based on) ambiguities, representing in some respects a different level of uncertainty in comparison with measurements and physical (including biological) models
- There is uncertainty in several dimensions, including regional, temporal (on its importance and aspects, see especially Rozman and Doull 2001) and 'structural'
- Uncertainty analysis may be a means to direct inquiry and also other management actions, instead of only bringing forth confusing complexities and impediments.

Statements in the literature about the certainty and significance of dioxin risks

Uncertainties have been used both as an argument for declining from management decisions and for proceeding to them (cf. 8, 9). In both cases, bias in interpretations and results is possible. Biases are caused in (environmental, health and general) risk assessment by technical as well as procedural and related valuation factors (Assmuth and Louekari 2001, Ball 2002).

In the present connection, some general statements about uncertainties associated with dioxins risks are highlighted, focusing on statements of the characteristics and significance of human health risks due to population effects from low dose exposures. While some of these statements are directly included in original studies, others are presented in reviews of studies. Other statements and evaluations have been given and commented on elsewhere in the text.

Many affirmations have been made of effects on humans at background and medium exposures, including exposure from fish consumption (e.g., Koopman-Esseboom et al. 1994, 1996, Patandin et al. 1998, Boersma and Lanting 2000, cf. text below, and Annex 8B). A weakness in many of the studies underlying these assessments is the lack of specificity as to the exposure and poor distinction between DLCs (including dlPCBs) and other PCBs. Also the general problem of spurious co-correlation with yet other contaminants and factors remains.

A key source of uncertainty in both qualitative and quantitative terms is variability and irregularity. This is related to the question of how many factors of risk are included in the assessment. Portier et al. (1990) stressed that in addition to the variation in (cancer) risks of TCDD due to the variation in bioassay data, many other sources of variation exist and the overall spread in the estimates of safe exposure levels may be substantially increased by the inclusion of a large number of parameters in mechanistic models of cancer risk. The variability in risk estimates is further increased if incorporating other endpoints and still other factors beyond animal experiments, such as particularly factors determining real-life exposure (e.g. from fish intakes), factors related to biological variability in the (human) populations that are subject to assessment, and factors affecting concurrent benefits from the diet involving exposure and in other such risk-modifying factors.

Portier et al. (1990) also emphasized, importantly as a general methodological point of view, that additional variability is not a shortcoming of the use of more complex (and more inclusive) models and not only an argument against attempts to grasp this complexity. Instead, they concluded that this greater variability may represent a more reasonable estimate of the true population variability in risk, and proposed that it be included in order to estimate not just the mean safe level (under some boundary conditions and assumptions that may be misleading) but also the entire distribution of safe exposures in the population of concern.

Kimbrough and Krouskas (2001) evaluated the evidence for several key effects reportedly associated with low-dose exposure to dioxins; they likewise scrutinized the evidence for neurobehavioral development (Kimbrough et al. 2001) and for effects from PCBs (Kimbrough and Krouskas 2003). For all of these, they found that the evidence is inconclusive and that in almost no studies have such elementary factors been accounted like the normal variation range in the population, or extensive and rigorous analyses have been made of confounding factors (including e.g. breast-feeding) (cf. Annex 8B).

Neubert (1997-98) presented a critical, extensive and thorough evaluation of the evidence for human effects from 'dioxins', stressing the lack of sufficient quality data to draw any firm conclusions, either from epidemiological studies or from extrapolations (interpreted by the author to be inherently speculations) of experimental non-human animal data. His analysis was mainly concerned with studies and assessments of highly exposed persons, and it was explicitly stressed "it is not *at all* justified to … extrapolate from … the polyexposure of industrial workers to possible risks for the general population that exhibits comparatively low PXDD/F body burdens (e.g. via the food chain). Conclusions … on such a basis should be ignored with respect to medical risk assessment, as they violate *all basic rules* of pharmacology and toxicology" (emphasis added). The author concluded that 1) little unequivocal evidence exists for adverse human health effects from even high exposures to TCDD or particularly to other related substances, using strict (but routine) criteria in medical studies; 2) the occurrence of effects at lower exposures is still more unlikely (depending however on the possible existence of non-monotonous dose-response relationships i.e. hormesis) and are in any case still harder to prove, due to the low-level polyexposures and to other methodological difficulties. Another point of

interest is the categorical statement that "it is *always impossible* to directly and quantitatively predict the occurrence of substance-induced effects in humans solely from animal data"; the author further stated that while extrapolation from high to low doses is difficult, extrapolation from one species to another is "virtually impossible". This in essence refuted the standard risk assessment for environmental chemicals, but no alternative (in human studies) was explicated.

5.3.2 Quantification of uncertainties

As quantitative as well as qualitative aspects of uncertainty have been already treated in connection with exposures (3) and effects (4), the present discussion will be restricted to some general and specific points and to summarizing some of the quantitative information on relevant uncertainties.

There are great uncertainties regarding the **formation** rates and **emissions** of DLCs. As already discussed, these can crucially constrain predictions (and reconstructions) of the environmental levels of DLCs. However, if exposure and risk assessment can be based on body burdens and other measures further down the risk chain, these uncertainties in the source term do not matter so much. Nevertheless, they may play a role for identification of the key sources of risks and for quantitative assessment (see e.g. Luthardt et al. 2003).

The uncertainty associated with estimates of risk from **consumption of dioxin-contaminated fish** can be estimated e.g. by Monte Carlo sampling of assumed probability distributions of risk variables (Hart et al. 2003b). Such analyses are sensitive to assumptions regarding pdfs, independence of variables, sampling methods, and overall framing of the analysis. Holmes et al. (2003) showed that measurement uncertainty (of CB 126 levels) was the main contributor to overall uncertainty in exposure to PCDD/Fs and dlPCBs in salmon, and noted that sampling may also introduce substantial uncertainty if the number of samples is small (<10). The analysis of Hart et al. (2003b) was moreover limited to exposures and to TEFs as a proxy of uncertainty associated with toxicity.

Judd et al. (2004) found that for high-level consumers and individuals eating fish from relatively contaminated sites, PCB exposure from fish alone may exceed 1 pg TEq kg^{-1} d^{-1} ADI; the exposure for average consumers of commercial fish was expected to be far less but highly uncertain due to the dearth of congener specific PCB data for commercial fish. Among specific sources of uncertainty, these authors estimated that if data below LOD were substituted by LOD instead of 0, the estimated exposure and risk levels were considerably increased (exceeding the ADI by consumption rate of 30 instead of 65 g fish d^{-1}). Judd et al. (2003) also pointed out that insufficient analytical methods for PCDD/Fs and PCBs (dlPCBs and others) may lead to inappropriate fisheries management decisions, and that population fish consumption patterns should be taken into account.

The modelling of Aylward et al. (2003a,b, 2005, Annex 7) highlights the uncertainties due to **pharmacokinetics** of DLCs. These uncertainties may result in great discrepancies when back-calculating from present body burdens to previous peak exposure or cumulative exposure. Whether such uncertainties result in risk overestimation or underestimation will also depend on the direction of the inference, e.g. are safe levels of exposure derived from body burdens or, *vice versa*, acceptable effect and risk levels from some intake levels. Other factors in kinetics may also increase risks, e.g. preferential disposition in sensitive tissues or in sensitive periods for effects.

Shirai and Kissel (1996) provided a quantitative analysis of uncertainties in the elimination half-lives of PCBs in humans (cf. Annex 7). Only limited congener-specific analysis was possible and did not include 0-*ortho* PCBs for which human elimination data had not been reported, but some general conclusions deserve mention. The authors concluded that if the blood concentration of ΣPCBs representing a NOAEL of threshold level is <75 ppb (considered plausible for the protection of more sensitive subpopulations such as children) and long half-lives are assumed, the ostensible margin of safety may disappear altogether. On the other hand, in another case a NOAEL of up to one order of magnitude higher resulted if a plausible half-life was chosen instead of the assumption of no excretion in deriving a reference (safe) dose from maternal body burdens for neurotoxic effects in offspring.

As to **effects** assessment, the uncertainties in dose-response models and in related extrapolations are crucial (cf. 4.2.1, esp. Table 16). On the biochemical level, Kohn et al. (2001) predicted for enzyme induction by TCDD in the rat an ED01 of 1 pg g^{-1} d^{-1}, corresponding to a body burden of 35 pg g^{-1} bw; this was much higher than previously predicted, because of the sensitivity of modelled responses to assumptions.

Some analysis of the uncertainties in quantitative estimates of risks to human health from general exposure to dioxins has been produced for

Japanese subjects, using a pre-validated exposure model (Maruyama et al. 2002b). Maruyama et al. (2003) found that the observed range of tissue levels was 0.2-4 times the level calculated using standard parameters. The results suggested that differences in body weight, food intake and absorption may partly explain the great inter-individual variation in tissue levels of c. 24 for the general population. Maruyama et al. (2004) specifically estimated that the maximum liver concentrations in breast-fed infants was 1/5 of the level in maternal rat liver associated with alterations in reproductive organs in the next generation. However, the simulated values for infant body burdens differed from measured values in other populations, e.g. for CB 126 and 4-PeCDF.

A summary of quantitative estimates of uncertainties from data, models and other sources in risks and risk factors of dioxins has been presented (Table 22). Additional discussion is presented of uncertainties and of uncertainty or safety factors in defining acceptable risk, effect, exposure and intake levels and other such benchmarks (cf. 5.5).

The differing and partly confusing notions of mixture toxicity of dioxins (cf. 5.2.1) are

Table 22. Sources, magnitudes, characteristics and factors of variation, error and uncertainty relevant for assessment of health risks from dioxin-like compounds, with particular reference to the Baltic Sea and fish consumption by humans. Note that uncertainties in previous stages of assessment may not count if better estimates are obtained for subsequent stages.

Area	Uncertainty category	Uncertainty factors	Variation, error and uncertainty estimates	Example factors/qualitative aspects of importance	Typical implications for overall uncertainty
Sources and emissions	formation	rates	great variation (up to orders of magnitude)	precursors, reaction conditions	+++
	emissions	emission factor	often 100-, some 1000-fold; for some 10 %[a]	process	+++
		activity rate		type of activity	
Environmental levels	sampling	representative. reliability	30 % of total var in intakes[b]		+(+)
	chemical analysis	pre-treatment quantitation	60 % of total var in intakes (CB 126)	matrix effects congener	+
Transport and fate	environmental transport	phase partition. advective/diffus	50-70 % in river transport[c]; more for sed/air-water fluxes	process	++(+)
	fate of compound incl. decay/transform	mass balance model–data fit	40 % rel error/best fit in BS[d]; orders of magn. for decay	congener profile environ conditions	++(+)
Human exposure	intake amount	fish levels	c. 10 % precision required, reached in specific fish only[e]	species, age, tissue, area, time	+
		fish consumption	generally some 10 % but more for other populat/fish[f]	skewed PDFs, special groups/ children	+(+)
	tissue delivery	absorption disposition	some 10 % potentially more	age tissue	+
	metabolism and excretion	$t_{1/2}$ products	>100 % e.g. for humans[g]; 20-fold for non-TCDD[h]	age, dose, congener metabolite identity	+(+)
Effects	type	specification of effect	max. orders of magnitude (e.g. if pooling tumours)	affects dioxin-attributability	++(+)
	mixture	compound aggregation	500 % (half-order of magn)	TEFs (endpoint weights etc) intercations with non-DLCs	++
	interspecies variation	toxicokinetics sensitivity	several-fold several-fold	lab to human/field	++ ++ (some groups)
	inter-individual variation	toxicokinetics sensitivity	potent. up to 2000 %[i]	age, sex age, dose	++ ++
	dose-response and low-dose extrapolat	cancer non-cancer	orders of magnitude[j] several-fold/orders of magn.	species, individual, tumour type endpoint, species, individual	++(+)
	population effects	human, non-hum	depend on population sizes	susceptible populations	++(+)
	community effects	species interact.	?	community stability	++(+)
	ecosystem effects	biotic abiotic	?	adaptation/aggravation interactions (also social/tech)	++(+)
	health benefits from fatty fish	mortality	CVD, some 10 % (or more)[k]	age, condition (patient/not)	+
		morbidity	for neurodevel. several 10 %[l]	endpoint, age, individual etc	++
	risk/benefit relations	mortality-based morbidity-based	orders of magnitude (due especially to uncert. of risks)	quality adjustment of life[k]	+++ +++(+)

References and explanations: [a]E.g. Wenborn & al. 1999, Bergqvist & al. 2005, Luthardt & al. 2003; [b]Hart & al. 2003, Holmes & al. 2003; [c]Giri & al. 2001; [d]Su & al. 1997; [e]E.g. TWGIM 2004a; [f]Lind & al. 2002; [g]Caudill & al. 1992; [h]Wang & al. 1997; [i]Masten & al. 1998, Bogaards & al. 2000; [j]Bonvalot & al. 1989, Paustenbach & al. 1991; [k]CVD=cardiovascular disease; see e.g. Wang & al. 2004, SACN and COT 2004, SPCFC 2005; [l]Ponce & al. 2000.

exemplified by the level of precision attached to numeric values. For instance, a total potency of c. 30 % higher with the chick embryo TEFs than with the I-TEFs for human risk assessment (Bosveld et al. 1992) does not yet present a great difference compared to other sources of uncertainty in TEFs (cf. Zabel et al. 1995a). It has been stressed e.g. by van den Berg et al. (1998) that TEFs are order-of-magnitude estimates of the compound risk, and this is taken into account in some quantitative risk estimates (e.g., Yoshida and Nakanishi 2003). Yet, TEFs also frequently are presented and treated as more precise values, even to several decimal points. They thus assume on one hand an unjustified singularity (with regard to the individual component congeners) and on the other hand an unjustified precision.

Uncertainties of the validity of the TEF concept have been highlighted in studies where adverse effects in free-living populations have correlated with body burdens of individual PCDD/Fs but not of TEqs (e.g., Elliott et al. 1996). Thus, even though TEqs as a more integrated measure often are preferable to single congeners, this does not hold generally. On the other hand, TEFs and chemically determined TEqs by definition are limited to the compounds assigned TEFs and exclude the DLCs not accounted for, by chemical or (more inclusively but non-specifically) biological assays.

5.4 Risk comparisons

5.4.1 General considerations

Risks are often compared with and even contrasted to each other. A strict juxtaposition may be due to a tactical or rhetorical (or unconscious) attempt to downplay or exaggerate some of the risks. For instance, human health risks from environmental contaminants are commonly compared with the risk from smoking to emphasize the smallness of the former. However, the need to relate risks (as any entities, especially quantitative) to others is natural also in scientific analysis, e.g. in testing and predicting differences. It is essentially a question of resolving the significance, in statistical or general terms, of a risk with respect to a sufficiently similar reference or background risk. For instance, the relative risks from smoking and DLCs e.g. from seafood have been studied explicitly in some populations, finding EROD induction in placentas to be related to maternal smoking, not exposure to PCBs (Pereg et al. 2002).

Also in environmental policy and decision making, it has become commonplace to require that risks are put in 'perspective', especially for 'rational' risk reduction, and to this end essentially compared to each other to prioritize the risks to be reduced. This was a strong emphasis in the policy of the USEPA (and of the US government in general) from early 90's, to the point of requiring comparative risk and risk-benefit analyses as preconditions of passing laws involving federal regulation (e.g., North 1997). Such comparisons have become more common also in EU, and their need has been recognized in connection with food contaminants and dietary advice (Renwick et al. 2003).

It has on the other hand been often maintained that risks cannot be compared to each other in a straightforward and simplifying manner, as they may have different qualities; 'One can't compare apples and oranges'. This stance is also sometimes exaggerated to the point of dismissing any risk comparisons, and is used as an argument against them. Wilson and Crouch (1987) stressed that just as comparison of risks is an aid in understanding them, so is careful selection of the methods to express them. They also pointed out that sometimes it is useful to contrast risks to indicate the different ways they are treated by society, noting that the higher stringency against dietary exposure to TCDD, a poorly known carcinogen, than that of aflatoxin, a very well-know carcinogen, may be due to their qualitative differences, e.g. the fact that TCDD was present in chemicals used in (Vietnam) warfare.

The **commensurability of risks thus depends on the risks and on the context, purpose and manner of comparison.** Finkel (1995, 1996) argued that, even acknowledging the many constraints and uncertainties in comparing risks, despite the above saying, one *can* compare apples and oranges, pointing out that people do it all the time both literally and figuratively (often intuitively) in deciding what risks and benefits to focus on, to value or devalue and to reduce. Quantity at some point will override (or merge with) quality as the most rewarding outcome is evaluated.

We delve on this question because it is crucial in dioxin risk assessment and management

in the present context. A very simplistic and straightforward comparison of risks from dioxins in Baltic Sea fish to other risks, e.g. from (active) smoking, may in some respects and for some purposes be meaningless (even if also fish consumption would be considered voluntary). Comparisons with, say, the risks associated with the voluntary activity of hang-gliding, the imposed technological risk of a nuclear plant accident, the social risk of homicide or the natural disastrous risks of an asteroid strike may be still less relevant (cf. Assmuth 2000), and still lower on a hierarchy of criteria for risk commensurability proposed by Finkel (1996). On the other hand, commensurability (within human health risks) is better e.g. with risks that are caused by similarly acting other substances, other agents in diet, and similar endpoints. Thus, hierarchies in the criteria for risk comparisons can be discerned and, as they are not universally applicable, devised for a particular application.

Even when focusing on human health, the policy question has been evoked and debated especially in US of the justification for comparing 'body counts' and basing decisions on such measures of risk. It has been countered that also the **quality of life** can be included; various systems in risk-based decision analysis have been devised e.g. to compare quality-adjusted or disability-adjusted life-years (Barnard 1996, Ponce et al. 2000, Wong et al. 2003), representing an analogue to the attempt quoted above to integrate qualitative and quantitative considerations of commodities. The more fundamental question is whether a quantitative description and comparison at all is meaningful or appropriate; this has both epistemic and ethical connotations that cannot be discussed here at length (see e.g. Shrader-Frechette 1991). In terms of overall approach to risks, this issue is in some respects similar to the possibilities and limits of quantifying and particularly of monetizing impacts in cost-benefit analysis (cf. 8.4.5).

An analysis of risk comparability or commensurability can be (and should in management contexts be) extended to address **benefits** from accepting risks, or risks of losing the benefits, such as those from fatty fish (see below in more detail). In addition, the costs associated with risks or risk reduction measures often are an important consideration in management decisions.

The **variation and certainty** about the risks to be compared often is an important consideration. For instance, it was shown by Finkel (1995) that the simplistic contrasting of the herbicide Alar in fruit to the apparently greater health risk from aflatoxin in peanut butter, used as a generalizing argument for herbicide use, was misleading not only by disregarding the qualitative differences but also by ignoring the quantitative probability distributions of these risks and the associated possibility of their reversed order. The author concluded that the fact that environmental and health risks differ in unknown quantitative respects is at least as important a caution for risk comparison as the fact that risks differ in known qualitative ways.

5.4.2 Comparisons between various risks

Comparisons of dioxin-like compounds in different settings and from different sources

DLCs in fish and in the general diet: This is part of the assessment of fish-attributable dioxin risks. As has been related above, the contribution of fish to intakes of TEqs varies from high levels of c. 60-90 % depending of study and calculation method in Finland to lower levels of 35 % in Sweden and only c. 10 % in Germany (cf. Annex 7). However, these dioxin-attributable risk shares vary within the populations in these countries, being higher for high fish consumers. Young persons may consume less fish (also per body weight) but receive DLCs during pregnancy and lactation. Accounting for susceptibility, the risks vary also between individuals in these groups.

DLCs from anthropogenic and natural sources: The possible importance of natural DLCs also in human diet has been suggested by several researchers (Safe 1998, Hovander et al. 2002, Denison and Nagy 2003, cf. Annex 1). The important preliminary data of Connor et al. (2004) indicate that PCDD/Fs and PCBs comprised a small fraction of the total Bio-TEqs in human blood (depending however on sample pre-treatment). They presumed that most of the total dioxin-like activity was due to natural compounds, including AhR-agonizing endodioxins and other natural DLCs such as the metabolite I3C of ingredients in some cruciferous vegetables that have appreciable levels and AhR agonist potency. The authors interpreted the findings as conflicting with the assertion (driving much of dioxin risk assessment) that low level PCDD/F and PCB exposures cause significant human health effects.

Such an evaluation may be in part premature as it ignores e.g. the specific effects from PCDD/Fs or dlPCBs during critical development stages, the possibly differing levels (Sekizawa and Matsuda 2002) and behaviour of PCDD/Fs and dlPCBs in target tissues in comparison with endotoxins, the effects not captured by the CALUX bioassay, and differentially AhR-mediated (including differently potentiated) effects. Also Safe (1997-98) pointed out that endodioxins and exodioxins might not be readily compared (but see Safe 2003). Nevertheless, such findings do add weight to arguments questioning the toxicological significance of PCDD/Fs and PCBs, and extending dioxin risk assessments that are now almost exclusively focused on PCDD/Fs and dlPCBs.

The management implications of the co-existence of other DLCs depend on the dimensions of risk that are in focus and on the case. The additional risks due to such other compounds may be considered to aggravate the risks also due to dioxins, and justify stricter dioxin risk management policies. However, the question arises whether the other compounds should be focused on instead of dioxins, if the former are found or assumed to cause greater risks (of sufficiently similar characteristics). It is conceivable that there is a point where the risks from such other contaminants (and yet other, sufficiently comparable agents and causes) will be great enough to warrant a shift from dioxin control to other contaminants, agents and concerns. This will depend e.g. on the availability of options and resources for risk management. In cases where there are easy opportunities and synergies, both dioxins and other contaminants may feasibly be reduced; in other cases the risks from other contaminants are competing for scarce resources (cf. 7, 8).

Comparisons with other toxic substances, particularly POPs

A) Non-dioxin-like PCBs

These comparisons are important as di-*ortho* PCBs and still higher chlorinated (3-4 *ortho*) PCBs are almost without exception present in elevated levels together with dlPCBs and often also with PCDD/Fs, usually in concentrations even several orders of magnitude higher, and as PCDD/Fs and dlPCBs have interactions with di-*ortho* PCBs (see above). In particular, non-additive interactions between dlPCBs and other PCBs have been noted (e.g. Jensen and Sleight 1986, Sargent et al. 1991). There is no conclusive or uniform picture of these interactions; in some cases antagonism has been found, in others additivity, and in still others slight synergism (cf. 5.2.1 and Annex 8A). Different action mechanisms probably are involved, and also the effect profiles differ. It is thus difficult to say what the exact relationship and balance of the partly disparate risks from non-dioxin-like PCBs, dlPCBs and PCDD/Fs would be. It may depend e.g. on the species and group. This difficulty is rather similar to some of those in comparing the risks of various DLCs; in some respects the dlPCBs and non-dlPCBs can even be more readily compared than PCDD/Fs with some other DLCs. As to exposure, the di-*ortho* PCBs and dlPCBs may be regarded as rather similar, both groups including congeners of higher and more moderate persistence and bioaccumulation properties. However, there are notable differences between these groups in metabolism and toxicokinetics in some cases. There is thus a multi-dimensional continuum of risks in terms of commensurability. From a management point of view, the impediments to comparison may not be so crucial, as PCBs are managed as a group especially within source control: In reducing dlPCBs, also other PCBs are reduced, and vice versa (cf. 7, 8). This is reflected in the ongoing efforts e.g. in the EU to develop assessment and management of PCBs collectively, also in connection with dioxins.

In epidemiological studies, some indications of the relative risks of DLCs and non-dlPCBs have been obtained (cf. Annex 8B). The tentative association reported by Hardell et al. (2004) with testicular cancer was stronger for PCB-TEqs than for other aggregate measures of PCBs including non-dioxinlike congeners. In contrast, Guttes et al. (1998) found that the levels of non-dioxinlike or weakly coplanar PCBs were elevated in breast cancer tissues while those of dlPCBs were not. In other cases where only non-specified PCBs have been studied (e.g., Dallaire et al. 2004), associations between them and adverse effects may be due at least to a large part to non-dioxinlike congeners. Often it is very difficult to ascertain whether an association is due to non-dlPCBs or dlPCBs that are correlated with the former; this may e.g. be the case with the tentatively reported association between prostate cancer and the general marker PCB 153 (Björnfoth et al. 2005, cf. Wolff et al. 1997).

B) p,p′-DDE

This main metabolite of DDT, still present in high amounts also in Baltic fish consumers (e.g., Atuma et al. 1998a), may exert effects ascribed also or alternatively to DLCs. p,p′-DDE may thus mask effects of DLCs and cause erroneous claims of risk attributable to DLCs. On the other hand, in cases when DLCs exert similar effects as p,p′-DDE they add to each other's risks, and there is even the possibility of synergism. It is a matter of judgment in what situations and in what respects the presence of p,p′-DDE-attributable risks aggravates risks due to DLCs; this holds for mixture effects and multi-stressor risks also more generally.

In **humans**, p,p′-DDE has been associated e.g. with immune effects (Dewailly et al. 2000, Karmaus et al. 2001), with neurological impairment after perinatal exposure (Koldkjaer et al. 2004), also in fish (Weisskopf et al. 2005, cf. Hardell et al. 2002), and tentatively with bone density reduction (Glynn et al. 2000b). Liljegren et al. (1998) found an association with breast cancer for CB 77 but not for p,p′-DDE in a small cross-sectional case-control study; Laden et al. (2001) however found no association with either DDTs or PCBs. The effects of DLCs have not often been studied separately in conjunction with p,p′-DDE, but usually only total or main PCBs have been analyzed (e.g., Karmaus and Zhu 2004, Weisskopf et al. 2005). This and the common correlation between organochlorides makes it difficult to elucidate the relative importance of p,p′-DDE and DLCs or of their interactions, and may also have masked effects of dlPCBs. The evidence in many studies is not strong for any cause-effect relationships. Hauser et al. (2002, 2003a) initially reported associations with semen quality in a normal population for total or main PCBs in addition to p,p′-DDE, but in-depth studies (Hauser et al. 2003b) removed all associations.

In several evaluations of **ecological** effects in the Baltic, p,p′-DDE and other non-dioxinlike compounds have been regarded as more plausible factors than DLCs in observed disorders (e.g., Olsson et al. 1994, Helander et al. 2002, Hario et al. 2004). In some cases the evaluation of their relative significance has been reversed (Annex 8B, 8D). In yet other studies, both DLCs and DDEs have been (Gill and Elliott 2003) or neither of these has been associated with adverse effects (e.g. Elliott et al. 2001a,b). Grasman et al. (2000a) found that most haematological and immune variables in Great Lakes herring gulls correlated with p,p′-DDE but not with gull-specific TEqs; it was judged that also other contaminants and factors could be responsible. Supportive experimental and mechanistic information exists for some effects of p,p′-DDE, including oestrogenicity-related effects. It is of importance that p,p′-DDE is known to cause adrenal disorders, as these have been the key element in the syndrome seen in Baltic grey seals (Olsson et al. 1994).

C) Toxaphene

Toxaphene is a complex mixture of mainly polychlorinated bornanes that has been used extensively as insecticide. It is produced by chlorination of camphenes and is also named Camphechlor (SPCFC 2005). Accumulation of specific indicator congeners or total toxaphenes has been reported in several Baltic Sea species, including herring, sprat (also oil), salmon, cod (liver) and eel, and in fish consumers such as seals (see Annex 6B). In seals, the levels in the Baltic have been higher than in the Atlantic or Arctic Ocean (e.g., Vetter et al. 2001), but in fish of similar magnitude or even lower (e.g., Paasivirta et al. 1993, Alder et al. 1997, Oetjen and Karl 1998, Fromberg et al. 2000). Andersson and Wartanian (1992) found that the levels were significantly higher in Baltic seals suffering from pathological conditions than in healthy seals. Both the NOAEL for immune effects in monkeys of 0.1 $\mu g\ g^{-1}$ bw d^{-1} (Tryphonas et al. 2001) and the TDI derived from this by Brüschweiler et al. (2004, ref. by SPCFC 2005) using an uncertainty factor of 1000 may be (have been) exceeded for some Baltic herring consumers.

D) Chlordanes, heptachlor, nonachlor, aldrin, dieldrin, endrin, endosulfan, Mirex and other cyclodiene pesticides

These and related POPs including reaction products are ubiquitously present globally. Many of them have relatively strong and varied toxicity, including reproductive and neurological effects. In the Baltic, mainly chlordane-related compounds, dieldrin and Mirex have been found in herring and perch (Tarhanen et al. 1989, de Swart et al. 1994, Strandberg et al. 1998b-d). The biomagnification of chlordanes and dieldrin seemed particularly strong, the latter being present in herring at 10-fold higher levels than

in the North Sea, like dlPCBs (de Swart et al. 1994). Olsson et al. (1994) found that chlordanes, like ΣPCBs and p,p'-DDE, were present at higher levels in starved and unstarved seals with uterine occlusions than in normal seals. Some of the effects and risks caused by consumption of Baltic fish may be attributed to this group of compounds. However, the levels of many of these compounds have decreased.

E) Hexachlorobenzene

HCBz is still produced in EU and an important contaminant due to its persistence. It has been measured in surface water in the Baltic Marine Area (Wodarg et al. 2004) and occurs at elevated levels in fish and other biota (e.g., Jansson et al. 1993, Strandberg et al. 1998b). The levels have however declined e.g. in Belt Sea terns (Thyen et al. 2000) and in Swedes (Lundén and Norén 1998), depending on intake of fatty Baltic fish (Sjödin et al. 2000). HCBz has been proposed to be included in dioxins and be assigned a TEF value, due to dioxin-like bioactivity; in this case it could increase human milk TEqs by 10-60 % (van Birgelen 1998). Such a straightforward incorporation of HCBz in dioxin risk assessment has been rejected by SCF (2000) due to differences in structure and effects. Among Inuits the levels of HCBz (like those of p,p'-DDE) have been found to have a statistically significant association with otitis media in children perinatally exposed to contaminated seafood diet (Dewailly et al. 2000). Also in German children an association was obtained between otitis media and HCBz especially in combination with p,p'-DDE and PCB contamination (Karmaus et al. 2001). HCBz in Baltic fish thus still poses added risks to those of DLCs. HCBz may also confound some of the effects of the latter. For instance, in some persons occupationally exposed to PCBs, only HCBz was related to immunological dysfunctions (Daniel et al. 2001).

F) Polybrominated diphenyl ethers (PBDEs)

Increasing levels of PBDEs, especially PBDE 47, have been found in Baltic fish and biota. Their toxicological significance is as yet unclear (e.g., Darnerud 2003). Their dioxin-like potency seems low (e.g., Chen et al. 2001); they thus constitute an added risk of different character and a point of reference for dioxin assessment. It has however been preliminarily evaluated (SPCFC 2005) that the intakes of PBDEs from fish by general EU consumers amount on the average to only 1-2 % (maximally 3-6 %) of the tentative TWI (PTWI) levels, and thus pose a lower risk than PCDD/Fs and dlPCBs as well as toxaphene (Camphechlor). Jakobsson et al. (2005) provided data indicating that the relative contribution of consuming fatty Baltic fish to total serum levels of PBDEs had decreased from 1991 to 2001, as little difference could be seen between non-consumers and high-consumers. It is of interest that PBDEs have been shown to antagonize or inhibit TCDD-induced enzyme activity in various *in vitro* systems (Kuiper at al. 2004, Peters et al. 2005). On the other hand, they may aggravate other effects of DLCs.

G) Organotin compounds

TBT and related organotins (e.g. DTB, TeBT and TPhT, TBTO) are of concern in the Baltic due to their recent use in anti-fouling agents in sea-going vessels (Champ 2000) and to their effect profiles. Effects on reproductive development, e.g. sex differentiation in molluscs and other hormonally mediated abnormalities, have been in focus; effects on vertebrates are less documented. Organotins have some similar effects as PCDD/Fs and other DLCs; TPhT has also been found to cause aberrant mitosis *in vitro* synergistically with CB 118 (Jensen et al. 2000). Also synergistic effects of TBT and PCBs on fish have been found (Schmidt et al. 2005). Grinwis et al. (2000a) concluded that TBTO at environmental levels may cause immunotoxicity in flounder and possibly play a causal role e.g. in lymphocystic virus infections observed in the field. Smialowicz et al. (1990) reported TBTO causes immune alterations in rats. Aw et al. (1990) found that TBT initiated apoptosis in rat thymocytes in a manner similar to that of TCDD, providing a mechanism for the selective immunotoxicity of TBT *in vivo*. Kikuchi et al. (2001) showed that TBT activates caspase-3 in human T-lymphoblastic leukemia cells at doses below 200 nM, TCDD at 20 nM and CB 126 at higher doses. Due to their higher environmental levels in some (coastal) areas, adverse effects may be caused by TBT and related compounds, instead of or in addition to PCDD/Fs and DLCs. However, a comparative assessment of the risks from organotins and DLCs is as yet difficult.

H) Chlorinated paraffins

Jansson et al. (1993) found elevated concentrations of chlorinated paraffins in all the Baltic Sea biological samples analyzed. Also Granby and Spliid (1995) reported them from blue mussels in Danish Baltic waters. The implications of exposure to these compounds in the Baltic fish are poorly known. The results of Hallgren and Darnerud (2002) suggested that PBDE 47 and the mixture Witaclor 171P had synergistic effects on FT4 and EROD induction. Warngard et al. (1996) showed that chlorinated paraffins inhibit the intercellular gap junctional communication within 1 h, DLCs only in 48 h.

I) Methyl mercury

MeHg contamination and effects are commonly encountered in aquatic food chains (cf. 4.3.2, Annex 8). Despres et al. (2005) showed that Hg and Pb were associated with disorders in neuromotor development in Inuit children while PCBs were not. The results of Omara et al. (1997) suggested MeHg in fish may pose greater risks to immune functions than PCDD/Fs and dlPCBs. A study in Eastern (inland) Finland (Virtanen et al. 2005) indicated that exposure to MeHg is associated with elevated risk of cardiovascular diseases in middle-aged men.

MeHg accumulates to a higher degree to muscle than in fat, and has not been generally present at high levels in Baltic herring consumers (e.g., Perttilä et al. 1982). Hagmar et al. (1998) reported that organic Hg in erythrocytes did not correlate with intake of fatty Baltic fish in middle-aged females, while Svensson et al. (1995a) found Hg levels in fishermen were twice those in persons with low consumption of fatty Baltic fish. SPCFC (2005) concluded from limited data that although the levels are slightly higher in Baltic than non-Baltic herring, they are lower than in tuna and not of specific concern. It has also been pointed out (Weihe et al. 2003) that the elimination of MeHg from tissues is faster than that of PCBs; thus, even when present at equally or more alarming levels with regard to toxicity, they may decline faster with reduced intake. However, Frank et al. (1992) found up to 700 ppm MeHg (ww liver) in occluded Baltic grey seals. In hotspots such as those impacted by pulp and paper (Hg from slimicides) and by chloroalkali industries (Hg from electrodes), MeHg may particularly influence risks.

K) PFOS and related fluorinated hydrocarbons

High levels of perfluorooctanesulphonate (PFOS), up to 1000 ng g^{-1} ww, have been measured in the Baltic Sea region in guillemot eggs (Holmström et al. 2005) and in livers of ringed and grey seals and white-tailed sea eagles, but not in salmon (Kannan et al. 2002b). Notably, an increasing trend was discernible in specimens of white-tailed sea eagle from Germany and Poland during 1990's, and in guillemot eggs from Baltic Proper between 1968 to 2003 with a yearly increase in PFOS levels of c. 10 %, from 25 to 600 ng g^{-1} ww egg. The implications of PFOS and related compounds for risks from DLC or more generally cannot yet be assessed, as there is little information on long-term biological effects (Seacat et al. 2003). In cynomolgus monkeys, a NOAEL of 0.15 μg g^{-1} d^{-1} was reported; several effects occurred at 5-fold higher doses. In addition to hypolipidemia, peroxisomal proliferation has been found in rodents exposed to PFOS, and several studies have been made of possible implications for cancer risks (see discussion by Seacat et al. 2003).

Comparisons with other toxic agents

Sonnemann et al. (2002) calculated that the human years of life lost (YOLL) due to exposure to PCDD/F and many other toxic substances (in a local setting) were insignificant in comparison with those caused by small air particles (PM10). Such risks may be less commensurable than e.g. the above risks from other contaminants in Baltic fish, although the endpoints may be similar (such as cancer).

Differences in commensurability of risks may be caused e.g. by the voluntariness of exposure. Birnbaum (1994b) pointed out that in the case of medicine, one may be willing to take risks for the potential benefits of the treatments, while exposure to environmental contaminants is different, as people often do not have a choice or, even worse, do not know they are being exposed. Such differences are related to general aspects in risk comparisons.

There is thus no clear-cut divide in commensurability of risks. It depends on many dimensions and attributes of the risks and also on the context and purpose of comparisons. For ecological risks voluntariness is not relevant, as non-human organisms have limited ability to avoid risks, which is one justification of

managing these risks strictly. In some respects, also voluntary exposure may serve as a point of reference. There are also benefits from accepting exposure to dioxin-laden materials, especially fish due to their great health benefits (see below).

In the case of tobacco, a comparison may be warranted also because tobacco smoke causes additional exposure to PCDD/Fs, even at the level of 4 pg TEq kg^{-1} bw d^{-1} from 20 cigarettes a day (Muto and Takizawa 1989), a value close to TDI, and causes elevated serum levels especially in women (Fierens et al. 2005). Smoking on the other hand influences the pharmacokinetics and body burdens of DLCs (e.g. through lipid states); surprisingly, it has been found to lower the TEq levels in breast milk (Houweling et al. 2002). As the overall health risks associated with smoking are so great, it is natural to consider it both as a potential confounder of effects of DLCs and in other regards. Smoking may act also in the case of many risks from dioxins as a synergistic risk factor.

Comparisons with other health risks

For decisions some (even if crude) grasp of the significance of the putative risks in relation to others is needed; often this comparison is made only intuitively and narrowly, superficially, in an *ad hoc* processes and in late stages of deliberation.

Some comparison of risks is required in studies of effects in control groups (which often are small and poorly defined, cf. Neubert 1997-98). Likewise, the definition of normal ranges of variation for a health outcome or intermediary response, the lack of which was repeatedly emphasized by Kimbrough and Krouskas (2001) for dioxin health effects studies, may be seen as one dimension of comparing risks from one cause to the total risk of anomalies in that endpoint.

The comparisons of dioxin risks with risks from other agents are related to the resolution of background risks that aggregate the effects of all agents causing these effects in a population. This is a different aspect and type of background risk comparison than that involving the comparison of background exposure to DLCs with higher exposures to DLCs (see above). In epidemiology and in ecoepidemiology, the distinction of a specified (attributable) risk from such background risk for the given endpoint is difficult for many reasons. One of them in DLC risk assessment is the lack of measures of natural variability of the endpoint, which in the case of many effects of DLCs are non-specific and thus difficult to measure (cf. 5.2.2, 5.3.1, Kimbrough and Krouskas 2001, 2003, Kimbrough 2001). The prevalence or incidence of the endpoint also is an issue; if it is common, the potential added effect and excess risk of DLCs might be hard to distinguish, as shown for cancer (4.2).

It usually cannot be stated unequivocally whether DLCs exert a significant effect (cf. Northridge 1995), or whether the excess risk is minuscule in comparison with other and aggregated causes. In most cases, greater other causes of the same outcomes can be found. These causes include chemical (exogenous and endogenous, dietary and other, e.g. medicine-related), physical (continuous and accidental exposure and stress, also life-style related) and notably biological (e.g., infections, reproductive history). The relevant risk characterization issue for management then becomes whether the smaller but estimable risk that can be attributed to DLCs is commensurable enough with those from other causes to warrant some concern also for DLCs (cf. 8.2.1, 8.2.2).

5.4.3 Comparisons between risks in other regions

Some characteristics were already discussed in the geographical distribution of risks from Baltic fish dioxins within the Baltic Marine Area itself and in relation to other sea and land areas (5.2.4). It was shown that while the Baltic is generally and by some measures more contaminated by DLCs than other coastal seas, the difference is not great and is variable. It is in place here to outline a fuller picture of the risks in the Baltic Sea area as compared with risks from comparable agents in other regions. Such a comparison is possible mainly with regard to the risks from contaminated fish in the Great Lakes area, and in the case of human health risks from dietary intakes also with some other parts of the EU. The latter will be in focus here because of the relevance for EU policies.

A comparison between total dietary intakes (and roughly risks) of the regulated DLCs between various EU countries is relatively straightforward, despite the deficiencies in representativeness and reliability of the estimation procedure particularly for dlPCBs e.g. in SCOOP (2000, cf. Lind et al. 2002), when specifying fish as a total category only. A much

Table 23. Tentative semi-quantitative assessment of relative risks from dioxin-like compounds in Baltic Sea fish, other fish and other sources in some Baltic Sea countries and reference regions. Note the variation and uncertainty of the assessment due to different sub-populations of fish consumers (not indicated in all cases).

Country/ region	Relative risks from dioxins attributed to Baltic Sea fish[a]	Relative risks from dioxins attributed to other fish[a]	Average relative risk from total dioxins[a]	Share of the risk from Baltic fish from total risks[a]
Finland	2 (general population) 4 (high users/susceptible)	4 (most of the average intake is from fish)	1-2 due to clean rest of the diet	4
Sweden	as for Finland	3 (≤1/2 of intake from fish)	2	3
Denmark	2 (general population) 4 (some high users/suscept)	3	3 due to intakes from meat, dairy, eggs (but declining)	2
Poland and Baltic states	as for Denmark	3-4?	3 due to intakes from meat, dairy, eggs (but declining)	2
Central Eur	0 (even including Baltic Sea fish based fodder)	1-2 (many countries/users) 3-4 (high seafish use/suscept)	3 due to intakes from meat, dairy, eggs (but declining)	0-1

Explanations: Relative scale from lowest (0) to highest (4) risk.

cruder and more uncertain comparison, in many respects only qualitative, can be made between countries with respect to the risks attributable to DLCs specifically Baltic Sea (fatty) fish (cf. 3.5.3). The risks associated with DLCs to fish-feeding animals, including man, between the Baltic Sea and other sea areas can thus be compared in a qualitative or semi-quantitative fashion only (Table 23).

The complexities in comparisons of contaminant levels in various regions are exemplified by the results of Sormo et al. (2003) on PCBs in grey seals in the Baltic and the Atlantic. Well-fed pups from the Baltic had levels of PCBs and other compounds 2- to 10-fold higher than in the Atlantic, but the proportions of the PCBs with >6 Cl atoms were comparable or lower in the Baltic pups. As discussed e.g. by Nyman et al. (2002), comparability of body burdens in seals is poor in young animals that reflect intake from lactation. On the other hand, comparisons of adult females are hampered by differences in their reproductive stage.

Regional differences in DLC levels (cf. 3.4, 5.2.4) do not necessarily and clearly correspond with differences in toxic effects (cf. 4.2). While Murk et al. (1996) suggested DLCs had some relation to reproductive parameters in North Sea common terns, Bosveld et al. (2000) noted that although the TEqs were comparable with or even higher than those in the Great Lakes area, adverse reproductive effects were absent; it was suggested that variable interactions of DLCs and other contaminants such as non-dlPCBs may contribute to such discrepancies. The same may be true of other fish consumers. Thus, differences in DLC levels may give a misleading picture of actual differences in risks.

Some of the particular characteristics of the Baltic with regard to risk from DLCs in fish have been summarized in Table 24. Many of these characteristics and factors are related and partly overlapping. The direction of their influence on risks from DLCs can also in many cases not de defined simply. It can in any case be seen that many factors exist that both increase and decrease risks, and affect their qualities. In some cases, all factors have an influence in the same direction, in most cases not, especially if also indirect influences are taken into account. The exact or aggregate significance of these characteristics with regard to relative risks cannot easily be ascertained (cf. 8.4).

Table 24. The influence of some Baltic Sea factors on risks associated with dioxin-like compounds in its fish.

Characteristic	Direct influences on human health risk	Indirect influences on human health risks	Direct influences on ecotoxicolocal risks	Indirect influences on ecotociological risks
Young				<adaptat ↑
Shallow	>cycling ↑		>cycling ↑	
Atlantic climate		storms mix ↑		storms mix ↑
Medium retention		<recovery ↑		<recovery ↑
Semi-detached	<dilution ↑		<dilution ↑	
Brackish			nat. stress ↑	nat. stress ↑
Low diversity	<food chains ↓		<food chains ↓	<stability ↑
Cold		<degradat. ↑	nat. stress ↑	nat. stress ↑
Ice-clad	<mixing ↓		mixing ↓	anoxia ↑
Stratified	<dilution ↑		<dilution ↑	anoxia ↑
Eutrophicated	biomass dilutes DLC ↓	biomagn. (in some) ↑; may boost biodegrad. ↓	biomass dilutes DLCs ↓	anoxic stress ↑
Humus-rich	carry DLCs↑, bind them↓			
Industrialization	DLC load ↑		DLC load ↑	
Populated catchment	DLC load, exposure ↑	sludge recycling ↓	DLC load ↑	sludge recycling ↓
High-economy	manage. capacity ↓	recreational fishing ↑	traffic stress ↑; manage. capacity ↓	
Fish-rich (relatively)	biomagnif. ↑		biomagnif ↑	
Heavily fished	exposure ↑	manage. capacity ↓		discard, trawl stress ↑
EU-surrounded	manage. capacity ↓	uniform manage. ↑	manage. capacity ↓	
Protected	direct capacity ↓	indirect capacity ↓	direct capacity ↓	indirect capacity ↓
High traffic	emissions ↑			
Well researched		manage. capacity ↓		manage. capacity ↓
Educated inhabitants		manage. capacity ↓		manage. capacity ↓

Explanations: *↑=is likely to increase the risks, ↓=is likely to decrease the risks.*

5.4.4 Comparisons between health risks and benefits from human consumption of fatty sea fish

General considerations

As discussed above (5.4.1), comparisons of risks and benefits of consuming Baltic fatty fish present a crucial issue for risk characterization. This aspect of risks has been omitted in most assessments, except e.g. that by SPCFC (2005) for EFSA and to some extent that by SACN and COT (2004), but is increasingly referred to in debates about dioxin risk management; however, it is sometimes dealt with in a superficial, generalizing and unbalanced manner.

Among the many categories and facets of risk comparisons, those between risks and associated benefits were not among the highest-ranking candidates for commensurability in the detailed hierarchy of Finkel (1996), who further distinguished risk-benefit comparisons and comparisons between risks and costs. However, in the present case as in many other risk management situations, the comparison between risks and benefits (from consumption of Baltic fish) emerges as a central decision issue, despite its complexity and the limits of comparability.

The comparison of health benefits and risks from fish consumption is one aspect of a more general risk-benefit analysis, as also other benefits, and other risks and impacts, especially from fisheries than those directly related to human health, should be factored in. Within the more restricted area of health risks and benefits, both human and non-human animals may be addressed. However, the risk-benefit relationships within the effects of fish consumption on human health are of main importance. The comparison of even these risks and benefits has several facets and complexities. The qualities of risks and benefits and of uncertainties associated with them need to be better accounted for (cf. 8.3). The relative benefits and risks from many different alternatives need in particular to be considered. Indirect risks and benefits also require attention. These aspects are largely addressed in connection with risk management strategy analysis (8).

There are some differences between health benefits and risks also in terms of effect profiles, and these may have to be taken into account in closer evaluation of the relative significance of such risks and benefits for decision-making. Especially the aggregation of all kinds of effects, adverse and beneficial, to some measures of total health, presents difficulties even if the risks and benefits could be compared within each particular effect. 'Total health' is a rather ephemeral concept and much depends on the varying valuations of the different aspects and qualities of health. The different adverse or beneficial effects may also have fundamental incommensurability, as in the case of fatal or non-fatal effects (although both may be and are valued, even monetarized, in public health policy).

Fish is commonly considered by expert bodies to be an important component of a nutritious diet. The risk of consuming contaminated fish must thus be balanced with the beneficial nutritive effects of fish. SACN and COT (2004) and, at the EU level, SPCFC (2005) have produced comparative assessments of risks and benefits (to human health) from fish consumption. However, SACN and COT (2004) considered the evidence base insufficient to conduct a quantitative risk-benefit analysis with regard to high consumption of oily fish.

It has been explained above (4.4) that health risks and benefits from fatty fish display a rather similar profile in terms of biological response types. The immunological, developmental, reproductive, metabolic and even some carcinogenic risks from DLCs in Baltic fish, at a generalizing and endpoint-aggregating level, are matched and possibly exceeded in many cases and collectively (e.g. in terms of mortality from all causes) by benefits from consumption of fatty fish. This makes it meaningful to compare risks and benefits from Baltic Sea fish consumption. Notably, the key benefit from fish consumption at least in terms of mortality, that for cardiovascular health, is much more pronounced and certain than the indications of cardiac toxicity by DLCs (occurring at high levels in humans), although some findings of low-dose effects on lipid metabolism may challenge this evaluation. Thus, within this class of effects, comparison of risks and benefits is justified.

However, even within largely comparable endpoints, the analysis of associated morbidity, population effects and risks and indirect risks present problems, as will be discussed below. In any case, the interaction of DLCs and other fish ingredients may be complex, multidirectional and dependent on the case, on the endpoint and on the target system in question (cf. SPCFC 2005). Risk-benefit analysis thus requires as detailed specification of effects, cofactors and confounders as risk assessment itself requires.

As stressed by Wheatley and Paradis (1996) and Wheatley and Wheatley (2000), comparative assessment of risks and benefits from fish consumption needs to balance theoretical physical science based risk assessment, whereby health policies are defined essentially by extrapolation from LOAELs, with analysis of the total health impacts from fish consumption and alternatives. As emphasized by these authors, particularly in the case of indigenous people, these total impacts include indirect impacts through loss of health benefits but also disruption of traditional ways of life through restrictive dietary advice. However, even in other populations the risk also to health from scaring people away from healthy fish diets needs to be considered.

Previous analyses and discussions

Ponce et al. (2000) analyzed the risks and benefits of fish consumption that included comparison of risks associated with neurodevelopmental delay due to MeHg (assumed higher in children due to increased fish consumption by mothers), and risks associated with myocardial infarct fatality (higher from reduced fish consumption). It might be assumed that in the case of the Baltic, the neurodevelopmental effects of perinatal exposures to DLCs are to some degree interchangeable with those of MeHg. In any case, this analysis (among the few of its kind found) is in general terms highly instructive for consideration of risks and benefits also from Baltic fish dioxins. Notably, the comparison was made both for the entire population and, focusing on children's health, for women of childbearing age. The benefits for cardiac health are particularly important in the case of DLCs in fatty Baltic fish, as cardiovascular diseases are among the key causes of mortality and morbidity in all Baltic Sea countries, e.g. in Finland still despite many interventions and extensive efforts. Ponce et al. (2000) pointed out that even small shifts in the risk for cardiovascular disease incidence (and severity and thus mortality and morbidity) would have tremendous impacts on public health.

The conclusion by Ponce et al. (2000) was that for the entire population the net benefits from fish consumption exceeded the risks when quality-adjusted life year estimates were used; the positive health effect was strongly driven by health gains among the elderly. If however myocardial infarct fatality was assumed to be 'only' equal in severity to neurodevelopmental delay, the negative impacts from fish consumption exceeded the positive effects at high consumption and contaminant levels. When considering only women of child-bearing age and their children, the risks from fish consumption exceeded the benefits regardless of the weighting of health impacts, as the proportion of the population at risk for developmental delay was greater and as the risk for infarct (among women) was lower than in the entire population (see also Cohen et al. 2005). In addition, it was estimated that one would have to dislike a case of developmental delay 7 times more (considering the entire population) or 250 times less (considering women of child-bearing age) than myocardial infarct fatality to be ambivalent about whether to consume fish. It was also noted that fertility rates affect the relative risks and benefit.

As a response to the assessment of health risks from consumption of farmed salmon by Hites et al. (2004a), several replies were published, addressing the health benefits of fatty fish and pointing out dangers involved in reducing fish consumption. Some of the replies presented health benefits from fish as if the original article had ignored them. However, as pointed out by Hites et al. (2004b), they had explicitly emphasized the many health benefits from fish and the need to consider both risks and benefits. On the other hand, little criticism and discussion has been directed to some real weaknesses in the analysis of Hites et al. (2004a), such as the assumption of risk additivity across many classes of compounds acting also by different mechanisms, and the restriction of the assessment to cancer risks.

An initial quantitative human health risk-benefit analysis of fish consumption was included in the reply by Tuomisto et al. (2004b) who provided a comparison of the cancer mortality risk due to salmon contaminants with the cardiovascular mortality risk due to lowered salmon consumption, in the European Economic Area. The comparison was limited to reducing consumption of farmed salmon to <1 meal a month, as recommended by Hites et al. (2004a). It was estimated that (maximal) excess cancer mortality due to pollutants in farmed salmon would be c. 200 (90% CI 110-340) cases a^{-1}, but only 40 (2-110) yearly cancer deaths could be prevented by restrictive recommendations. On the other hand, including cardiovascular deaths, the recommendation would worsen the net health effect by causing c. 5000 (34-19000) deaths a^{-1}. That is, the mean estimated benefit from avoided death as a result of fatty fish consumption exceeded the mean estimate of preventable mortality risk over 100-fold. It was mentioned that none of the "scientific uncertainties" considered, such as contaminant levels in wild or farmed salmon, changed the result. Tuomisto et al. (2004b) also presented a quantitative risk-benefit assessment of the alternative of reducing contaminant concentrations of salmon feed, finding this to save an estimated 360 (90% CI -3200 – 4100) lives a^{-1}, mainly because of the possible increase in consumption of salmon (cf. 7.3.3, 8).

However, it is unclear what uncertainties were considered and how. For instance, it was mentioned that care was taken not to exaggerate benefits. Still, the benefit estimate was based only on the reviews by Harper and Jacobson (2001) and Din et al. (2004), not the more extensive evaluations of the data e.g. by Wang et al. (2004) that found considerable reservations as to the certainty and generalizability of dose-benefit relationships (cf. 4.4.2 and SPCFC 2005). Distributions of risks and benefits in the population, and the types of fish consumed also as factors influencing benefits, were likewise not fully addressed. Regardless of such constraints and reservations, the estimate of benefit/risk ratio of 100 (or even hundreds) from consuming Baltic fish has been repeatedly quoted in Finnish media, also in connection with the detection of PCBs and various other contaminants in the blood of the Finnish minister of the environment (as in many other European environmental ministers).

Leino et al. (2005) further analyzed the risks and benefits to human health in Finland from fish dioxins (including dlPCBs and aggregated by WHO-TEq_{DFP}) and n-3 fatty acids (based on data mainly from the national nutritional database Fineli). Both Baltic Sea fish and inland fish were included, specifying the species of main consumption. Also this analysis compared cancer risks (based on liner low-dose extrapolation) with avoided coronary heart disease mortality (based on Din et al. 2004 and Marckmann and Gronbaek

1999). A probabilistic uncertainty analysis was included using Monte Carlo simulation with median Latin hypercube sampling; it was not specified what distribution forms were assumed. The mean risk/benefit ratio varied from 19 to 440 between the species, exceeding 1 in all of them; the species with highest risk/benefit ratios were farmed salmon, vendace and burbot. It was concluded that banning commercial fishing of salmon and herring would save <4 cancer deaths a^{-1} but cause c. 90 (90 % CI 38-1000) deaths a^{-1} from coronary heart disease, decreasing the net benefit of consuming all domestic fish from 410 to 320 (32-790) avoided deaths a^{-1}.

These assessments carry some significant limitations from the point of view of risk-benefit analysis for integrated risk management. Some decision dimensions and factors have not been fully considered. First of all, the relevance and commensurability of the impacts and thus the validity of the net risk estimates may be questioned. When increasing commensurability e.g. by restricting the analysis to mortality (and only to cancer risk of DLCs), critical effects of DLCs are excluded (cf. Ponce 2000, above). Additional analysis might include comparisons of health benefits (not only from avoided cardiovascular mortality) with morbidity, also from the reproductive and developmental disorders that are possibly caused by DLCs in fish and often assumed to constitute their critical effects, although non-lethal. Disability or quality adjusted life years could be used.

Secondly, depending on risk and benefit distributions, the risk/benefit ratio may considerably deviate from mean value, as found in other probabilistic comparative risk analyses (e.g., Finkel 1995). The population risk in general requires attention. It is conceivable that the benefits from fatty fish are obtained by a larger population than the risks are, as cardiovascular disease is a more common outcome than some of those putatively linked with DLCs, and as the beneficial effect would encompass a larger share of fish consumers. However, also a fuller consideration of the distributions of risks and benefits, e.g. among high-risk and high-benefit populations such as pregnant mothers and their offspring, seems in place.

Thirdly and in connection with the previous point, there are important uncertainties that are not captured e.g. by simple analysis of the pdf's of some risk and benefit variables. All such additional uncertainties cannot be categorically dismissed as "political" (Tuomisto et al 2004b); they can be uncertainties associated with real impacts that have also a scientific dimension. They include uncertainties of framing effects (cf. above) and of models underlying the risk-benefit comparisons. For instance, dose-response functions may influence the outcome of the analysis. As explained above (4), there are indications of a plateau in the dose-response for some effects of DLCs (even suggesting hormesis or other non-monotonous functions). Also the benefits from PUFAs do not increase linearly and indefinitely, but reach a ceiling (cf. 4.4.2, SPCFC 2005, Annex 8C). The net effect of these along the dose scale seems however not easily tractable.

Finally and most importantly, risk-benefit comparisons often do not take into account what alternatives there are to fatty fish consumption, as was pointed out by Hites et al. (2004a). Thus, analyses of alternative food production systems (Tuomisto et al. 2004b, cf. above) could be augmented e.g. by analyses of alternative diets (Cohen et al. 2005). This is crucial as straightforward juxtaposition of risks and benefits from (fatty Baltic) fish may too easily presuppose that people automatically shift from consumption of such fish to more unhealthy fats and diets. This presupposition should however be questioned, also as such dietary behaviour may be influenced (cf. 3.5.1, 7.4.4 and 8.3-8.4). This aspect of risk-benefit comparison also has methodological significance. As such factors play a role for the health outcomes, they may circumvent some of the need for analyses of other factors and dimensions of risk-benefit relationships.

There are few empirical epidemiological studies where the relative effects from exposures to beneficial and harmful ingredients in fatty fish have been explicitly compared, and none in the Baltic Sea area. In connection with some studies, this issue has however been reflected and commented on. Axmon et al. (2002, 2004b) noted that sea fish may have compensated for any harmful effects of DLCs in Baltic fish on reproductive functions among Swedish East coast fishermen's wives. Svensson et al. (1994) on the other hand mentioned, having found a slight but significant negative association between some plasma dlPCBs and NK cell indices in consumers of Baltic fish, that an intake of PUFAs has been found to reduce NK cell activity in human lymphocytes. As discussed in 4.4, there is however also evidence for beneficial

effects of fish consumption on immunological development.

Lucas et al. (2004) found in studies of mothers and their newborns from the Canadian Arctic that gestational age and birth weight were higher in the third tertile of percentage of n-3 highly unsaturated fatty acids (HUFA, cf. PUFA) of total HUFAs, as compared with the first tertile. There was no evidence that contaminants had negative effects on gestational age or birth weight. Some support for a R/B ratio <1 was thus obtained. Lund et al. (2004) in their reply to the article by Hites et al. (2004a) pointed out that no increase in relative risk for cancer (specifically in liver and breast) could be seen among 65000 women with a higher self-reported consumption of (mainly) farmed salmon in Norway. This result is in line with the absence of such cancer risks from exposure to DLCs in fish found in other, albeit smaller studies and also with the lack of clear association more generally between exposure to dioxins and risks for (oestrogen-related) breast and liver cancer (cf. 4.2.6). These authors did not consider other health risks from DLCs in fish. Importantly, they concluded by a comment on risk communication and management, stressing that "unilateral promotion of very limited health hazards could affect the overall health of populations by urging people to give up a healthy diet, causing them to substitute fish with less healthy and perhaps less safe foods.". This real and highly significant but multi-faceted danger will be discussed in more depth below in Chapter 8 (and 7).

In summary, it can be said that the limited quantitative analyses so far suggest that, in general, the benefits to human health from consuming fatty Baltic fish exceed the corresponding health risks, at least as measured by mortality and quite possibly also by morbidity and other measures. Deviations from this generalization may be significant e.g. to some population segments, and are discussed below. It can also be concluded at this point that more many-sided risk-benefit analyses are needed to complement the picture from initial comparisons.

The importance and implications of specifications and distributions of risks and benefits

A more detailed definition of the receptors of risks and benefits and their distributions in time, space and population segments is needed for risk-benefit comparisons (cf. Ponce et al. 2000). In addition to capturing the full distribution of risks and benefits in the populations consuming fatty Baltic fish, some particular groups of high or low risks or benefits may be identified and focused on. In quantitative terms this may represent a 'reasonable worst case' approach.

Age group is a significant consideration, as for risks (cf. above). There is plenty of scientific information pointing to that particularly great health risks from dioxins are in general caused to early developmental stages in perinatal (especially prenatal) exposures, and to other young persons including adolescents of both sexes (4.3, Annex 8B). There are indications that children and pregnant women presently consume small amounts of fatty Baltic Sea fish (cf. 3.5.1). Thus, the risks from Baltic fish dioxins are in general not directed to those most susceptible. Exceptions include children consuming high amounts of fatty fish, and foetuses and lactating infants to mothers who have consumed large amounts (during past decades). From the perspective of risk-benefit analysis, the consumption of fatty (Baltic) fish among children may be even too small, considering both the potential loss of benefits for their healthy development from such fish, and the loss of habit of consuming fish that may result, also as fisheries and fish processing and marketing cycles wane (cf. Lund et al. above and 8.3).

Much of the health benefits from fatty fish consumption may be accrued to elderly persons that are a particular risk group for cardiovascular diseases. There are data (Daviglus et al. 1997) suggesting that the beneficial effects of fish consumption to myocardial infarct risk may decrease with age, but as discussed by Ponce et al. (2000), these estimates were not based on age-specific relative risks. Hites et al. (2004b) in their reply to Lund et al. (2004) pointed out that young people are not at risk for heart attacks, but their risk of cancer at older age is increased by exposure to [fish] contaminants.

The age distribution of cardiovascular health benefits is modified by the age distribution of risks. As discussed above, young developmental stages also in humans are likely to be particularly susceptible to adverse effects from DLCs e.g. on the development of immune and reproductive systems. On the other hand, many of the benefits from fatty fish consumption are accrued

also to young developmental stages. These benefits include, in addition to the avoidance of neurodevelopment delays (see Ponce et al. 2000 above, and 4.4), healthy pregnancy and early development in general (SACN and COT 2004, SPCFC 2005), and potentially also other aspects of development and well being in addition to lower risk for later-life cardiovascular disease.

As to the risks from DLCs mediated from contaminated fish to human infants through mother's milk, few studies have been made based on specific exposure and effect characterizations. The lacking conclusive results regarding effects of specifically DLCs on developmental disorders in breast-fed children from the Baltic Sea region and even from other areas such as the Great Lakes (cf. 4.2, Annex 8B) suggests however that great health impairment has not taken place, and that the beneficial effects of fatty fish diet and of mother's milk from fatty fish consumers have outweighed adverse effects.

It should be stressed that not only risks from DLCs (and other contaminants) in fatty fish but also benefits to foetal development are accrued already to young individuals; this may be disregarded if only cardiovascular health benefits are accounted for. It is in any case possible that the overall distribution of risks and benefits from consumption of dioxin-laden fatty Baltic Sea fish is even strongly asymmetric in terms of age groups, although the age-averaged R/B ratio would be considerably <1.

Results from laboratory animal studies have provided some support for such an assessment, despite limitations of inter-species comparisons (Desaulniers et al. 2003, see Annex 8B). These authors noted that three factors may contribute to overestimating the risk to humans from DLCs in mother's milk: a) declines in the levels of AhR agonizers in mother's milk (this does however not diminish the risks, including lagged risks, from earlier exposures); b) dosing was based on mean values in mother's milk that are in fact not normally distributed but left-skewed (the significance of this factor is unclear, as also means do provide a risk estimate; fuller information on the pdf's of an infant's and a rat pup's exposure would be preferable); c) health benefits from breast-feeding are expected to counteract adverse effects, e.g. neurological and immunological (this will depend e.g. on the correspondence of adverse and beneficial effects). Also other factors may be identified that can either decrease or increase human risk estimates, e.g. due to toxicokinetics. The importance of such factors is fundamentally conditional on the extrapolation of non-human to human data. In addition, the specific implications of fish-based exposure of infants (e.g. from heavy consumers of fatty Baltic fish) for risk distributions are not straightforward.

Ponce et al. (2000) used no discounting to adjust for future health impacts or to modify the 'quality-adjusted life year' scales, as measures of optimal health, for health impacts that occur later in life. They noted however that these assumptions will affect the distribution of risks and benefits, and that while the overall population may experience a health gain from fish consumption, younger members of the population may experience a net health deficit. Some characteristics of DLCs such as the importance of perinatal exposures and lagged kinetics and effects emphasize such a distribution of risks within age groups, in absolute terms (disregarding the temporal development of exposures). However, particularly in the case of PCDD/Fs and PCBs in Baltic fish (and even in fish from many other regions), such a possibility must be put in relation to the greatly decreased exposures and the passing of the peak exposures already 30 years ago. This development acts toward the opposite direction, allowing much greater protection to the young now than to the young of 30 years ago (i.e., now middle-aged). This may imply that the distribution of risk-benefit relationships and associated asymmetries are not as significant anymore.

The age factor is related more generally to time dimensions in risks and benefits (cf. 5.2.3), and in natural and anthropogenic processes changing risks. It can be envisioned that as dioxin risks decline (albeit slowly and with some fluctuation and variation), the benefits from consuming fatty fish may in principle be held constant or may even increase, if fatty fish consumption can continue at the present (or higher) level; this is not certain but depends e.g. on risk management strategies (cf. the discussion below on risk reducibility and 8.3). The benefits and risks and their relationships can depend e.g. on the development of other factors affecting the health conditions that are benefited by fatty fish; they can also depend on whether the use of clean fish oil based dietary supplements will increase as a surrogate to fresh fish and to other dietary items (see Ponce et al. 2000 and Melanson et al.

2005, and discussion on risk reducibility below), and on other dietary changes.

Comparison of risks and benefits should also take into account the **gender and reproductive** status of fish consumers. In general, women are particularly vulnerable to risks from DLCs not only in themselves but also because of the exposure of their offspring during pregnancy and lactation, i.e. inter-generationally transferred effects (cf. 2.4.2, 5.2.2). SACN and COT (2004) gave some consideration to this in their comparative assessment of risks and benefits, noting that DLCs in fish posed particular risks to women of childbearing age, but also particular benefits to the development of pregnancy and of the child. Due to the slow disequilibria in dioxin exposure, it was concluded that a woman who had not consistently exceeded the toxicological guideline values for fish consumption could increase her oily fish consumption throughout pregnancy and lactation without deleterious effects. On the other hand, prolonged exposure to and accumulation of DLCs by women and girls before pregnancies pose particular risks, potentially offsetting at least some of benefits from fish consumption for health in general and reproductive health in particular. As to consumption of fish from the Baltic, Axmon et al. (2002) in connection of their studies of the reproductive health of female consumers of fatty fish noted that the absence of adverse effects could be caused in part by the compensating beneficial effects of fish intake.

A significant aspect affecting risk-benefit analysis and risk management that is related to the cause of the risks is their **reducibility**. This includes the question of whether benefits can or cannot be obtained from alternative dietary items. There is some evidence for a greater benefit to cardiovascular health from consumption of fresh fatty fish than from fish oil supplements (c.f. 4.4, 7.4). Still, the possibility of substituting the fatty fish with other, more dioxin-free but equally or almost equally healthy dietary items and supplements presents an important management decision issue (e.g., Foran et al. 2005b, cf. 8). The reducibility criterion is important in light of recent debate on human health risks from fish dioxins (in farmed fish however, the general arguments have great relevance for the present risk-benefit assessment).

Studies have been made of the content of DLCs and other contaminants in fish oils (see Annex 6). In general, many fish oil products especially based on cod liver have been found to contain appreciable levels of dioxins, also some products from allegedly clean marine areas. This may imply that the relative net benefit obtainable from substituting consumption of fatty Baltic Sea fish with fish oil supplements is not great. Melanson et al. (2005) proposed that also in dietary and therapeutic recommendations, consumption of clean fish oil instead of whole fish be emphasized, to ensure both its health benefits and to avoid risks due to its contaminants.

The purity, availability and other characteristics of dietary supplements becomes an important question in relation to alternatives to consuming whole fatty fish or fish oil from the Baltic (cf. 7.4, 8). FSAI (2002) published data on PCDD/Fs and dlPCBs in fish oil capsules in Ireland, finding highly variable levels, but concluded that use of these products according to recommendations (or consumption of farmed salmon, commonly exceeding 4 pg WHO-TEq g^{-1}) does not threaten consumer health (see also SPCFC 2005).

In the present assessment and discussion, it is feasible to focus on a semi-quantitative comparative evaluation of risks and benefits from consumption of Baltic Sea fish by humans (Table 25), instead of e.g. attempting to estimate the magnitude of risks and benefits in commensurate

Table 25. Comparative assessment of human health impacts of ingredients in fatty Baltic Sea fish, including beneficial effects.

Effect type	Effect direction and strength, by effect factor[a]					Notes on risks (R) and benefits (B)
	DLCs	Other OXs	MeHg	n-3 PUFAs	Other ingredients	
cardiovascular	-		-	++		subtle/inconcl. high-D Rs, strong clear Bs (in patients)
neurological	-(-)	- (CBs)	(-)	+(+)	vitamin benefits	MeHg important locally; PUFAs benefit development
developmental	-(-)			+(+)	vitamin benefits	(non-neurological)
reproduction	-(-)	(-) (DDE)		+(+), (-)		inconclusive results in fish-eating BS cohorts
immune	-(-)			(+)		some persons allergic to fish; complex multi-D effects
metabolism	-			+	vitamin benefits	R and B in diabetes/lipid metabolism
tumors	(-)	(-)		+(+)		varies by tumour type; inconclusive Bs
hormonal	-	- (DDE)		+		

[a] *- =adverse, + =beneficial; (-)/(+)=weak effect, i.e. at high doses or subtle effects at all doses; -(-)/+(+)=moderate effect; --/++ =strong effect.*

Table 26. Distribution of human health risks and benefits from Baltic Sea fish among population segments.

Impact type	Distribution among age groups	Gender distribution	Other deviating segments of population	Geographical distribution	Distribution in time
Risks	▪ highest in prenatal exposure esp. for developmental effects ▪ high in children (breast-feeding may considerably compensate)	highest in females (also apart from risks mediated to fetus)	▪ high in high consumers of fatty fish ▪ high in those heavily exposed from other sources, e.g. smokers (but relative risk addition is smaller) ▪ high in fasting persons ▪ high in persons with certain anomalies (e.g. thyroid disorders) ▪ high in persons of unfavorable genetic disposition (polymorphisms)	▪ high in hotspots ▪ high in coastal fish consumers ▪ high in areas of high exposures from other sources (but relative risk addition is smaller) ▪ high in areas with greater susceptibility	▪ some risks have up to trans-generational lags (reproductive effects in offspring) ▪ some risks are transient (e.g., behavioral), some permanent (structural development)
Benefits	▪ for cardiac health, highest in elderly ▪ high in other age groups	for cardiac health, highest in men	▪ high in persons with excess risk of cardiac disease (coronary patients) ▪ high in persons with low vitamin D and other nutritional deficits ▪ high with favorable genetic disposit.	▪ high in areas with excess risk of cardiovascular disease ▪ high in areas of fish deficits and favorable genetic disposition	▪ some benefits lagged ▪ some benefits transient
Compound	B>R for all ages; special risks to young	B>R for both sexes; special female risks	B>R for all segments but precaution advisable for some of the high-risk groups	B>R for all areas but precaution is advisable for those with high risk	R/B decreases further if exposures decrease (in fish and generally)

measures. The main thing here is to identify some of the risky and beneficial influences on and characteristics of human health that are relevant in the context of consumption of Baltic Sea fish. In addition to such generalizing comparisons, it is important to outline the distribution of risks and benefits among various groups of people defined along some key dimensions (Table 26).

Breast-feeding as a parallel case of risk-benefit balancing

The relationships between adverse effects and benefits of breast-feeding are important from the point of view of Baltic fish dioxins, as some general issues in risk and risk-benefit assessment and in risk management are comparable between these two cases of human nutrition. Also the differences between them are instructive.

Risk-benefit comparisons in connection with contaminants in human milk and breast-feeding have been noted as an important topic on several occasions (e.g., Clench-Aas et al. 1992, Pohl and Hibbs 1996, Brouwer et al. 1998b, Shu et al. 1999). In general, it has been stressed that contaminants do not justify discouragement and reduction of breast-feeding.

Smith and Gangolli (2002) explicitly mentioned that it is accepted (e.g., SACN and COT 2004) that the advantages of consumption of PUFAs in oily fish by the average consumer and the benefits of breast-feeding outweigh the potential risks from intake of contaminants by these routes. Patandin et al. (1998) among others considered the possibility of reducing the nursing period due to the relatively high background level of contamination of mother's milk by PCDD/Fs and (0-*ortho*) PCBs in the Netherlands, but concluded there are numerous demonstrated advantages of the nursing itself on the general development and health of children. Tuomisto et al. (2004b) pointed out the fact that most authorities have continued advising mothers to breast-feed despite risks from contaminated milk, although in some cases suggesting that the duration of breast-feeding might be limited to e.g. 6 months. These authors also noted the lack of research on the relative benefits and risks from breast-feeding.

Hoover (1999) assessed the exposures to and risks from persistent organochlorines in Canadian breast milk probabilistically for cancer and non-cancer effects, weighing the risks quantitatively and qualitatively against the benefits of breast-feeding. It was concluded that current levels of the majority of contaminants identified in breast milk do not pose unacceptable risks to infants and that the well-documented benefits of breast-feeding, including psychological effects, appear to outweigh potential health risks from organochlorines. However, potentially significant risks were estimated for exposure to PCBs and PCDD/Fs although levels of POPs had been declining, and recommendations for restricted breast-feeding by high-intake populations were regarded as possibly justified. It should be taken into account, also as a policy argument, that without exposure to this contaminant the benefits from mother's milk might have been still greater (cf. 8). It was also pointed out that the risks of mortality from not breast-feeding estimated by Rogan and Ragan (1994) exceed even the

theoretical cancer risks estimated for infants exposed to potential carcinogens in breast milk. Follow-up work was suggested on physiologically based pharmacokinetic models with probabilistic inputs to predict dioxin exposure to infants and on coupling exposure estimates with a dose-response analysis accounting for uncertainty.

The debate is still ongoing among researchers on the evidence for over-riding health benefits from breast-feeding in comparison with the adverse effects from exposure to DLCs, other PCBs and other contaminants, e.g. p,p'-DDE. Ribas-Fito et al. (2003) found that the delay in mental and psychomotor development up to 1 a of age that was associated with breast-feeding was in the long term exceeded by the beneficial effects of breast-feeding. Importantly, Jacobson et al. (1999) found that the beneficial effect of breast-feeding on IQ was no longer significant after adjusting for maternal IQ, parenting skills and home observation, suggesting that the observed advantage of breast-feeding is related to genetic and social factors rather than to the nutritional benefits of breast-feeding on neurodevelopment. Thus, it may be concluded that also in the case of breast-feeding, the relative benefits and risks from such contaminated nutrition depend also on specification and resolution of the various types and factors of risks and benefits, and of their variability.

The relationships and differences between mother's milk and other foods, including fish, are largely related to risk perceptions and health and nutritional risk management policies and practices. Some of these aspects of this illuminating parallel will be dealt with below.

5.5 Tolerable intakes, allowable concentrations and other quantitative decision criteria

5.5.1 Basis and definition of measures of tolerable human intakes

General

In the present risk management of dioxins e.g. in EU the key quantitative criterion of acceptable risks is the human Tolerable Daily Intake (TDI) and similar measures, such as increasingly Tolerable Weekly Intakes (TWI) or even Tolerable Monthly Intakes (TMI) due to the emphasis put on long-term accumulation (SCF 2001, JECFA 2001, FAO/WHO 2002a). Because of this importance of the TDIs, they should be scrutinized as to their scientific and policy foundations and implications. A scrutiny of TDIs that are often accepted in a rather unquestioning manner (or criticized unanalytically) is necessary for a balanced risk assessment and management of dioxins. In particular, it needs to be clarified in how far TDIs are science-based or based on other premises, also non-explicated.

The TDI is in essence based on a) effective doses or body burdens, and subsequently derived no-effect doses, in animal experiments fulfilling certain criteria, b) conversion of these measures to corresponding human doses, body burdens or intakes, and c) application of various safety factors because of the uncertainties involved in the process. Some consensus procedures have evolved for this, based e.g. on WHO (1993a). However, the process is not guided in detail, and also more non-formal, informal and even silent decision criteria, partly convention based and partly case-specific, are applied e.g. as to the quality of data, the relevance of effects, and the appropriate safety factors. It should be noted at the outset that the procedure is not strictly scientific (even allowing that nothing fully is, and that the notions of 'scientific' are to be questioned), but involves considerable judgment based on pragmatic considerations and valuations, more or less transparent.

The TDI concept is evolving, e.g. to include other dose measures than intakes, such as body burdens that shift the focus closer to critical tissue doses and avoid some of the uncertainties in dioxin kinetics in the body and related limitations of interspecies extrapolation. The development of TDIs is explicitly indicated in the concept by the additional specification of temporary TDIs ('t-TDIs') or provisional TMIs (JECFA 2001). In the most recent SCF (2001) opinion and re-evaluation of dioxin risk assessment, this reservation has been removed, implying that a more permanent TDI could be definable. The criteria for this basic distinction between types of TDI are however not clear and the implied conclusion is in some respects not easily defensible, as will be explained below.

In the following, the SCF (2001, 2002) evaluations and proposals for a TDI (or TWI) for dioxins are used as a case, justified e.g. by their central role in EU dioxin risk management. These

TDIs will be compared with some other (including older national) evaluations and proposals, related to some pertinent discussions of TDIs, and analyzed also more generally, even if in short, within the context of quantitative decision criteria. Many of the criticisms offered apply to both the TDIs as stated by SCF and to other TDIs and quantitative benchmarking approaches to dioxin risks, and are thus not to be construed as indicating exceptional shortcomings of the SCF opinions on the matter. On the contrary, it can be said in general that in some respects (such as in toxicological and pharmacokinetic assessment) these opinions are thorough and well founded, their important limitations, weaknesses and non-clarities being found elsewhere.

Database and pivotal study selection

The selection of the studies, mainly animal experiments, to be used as a basis of quantitative and qualitative toxicological assessments and definitions of a human TDI is an initial critical step affecting the conduct and outcome of these assessments.

SCF first (2000) qualified as pivotal low-dose studies those by Mably et al. (1992c) and Gray et al. (1997a) on reproductive development in male offspring of Holzman and Long-Evans rats, respectively, and those by Schantz and Bowman (1989) and Rier et al. (1993) on neurobehavioral development and reproductive effects (endometriosis) in Rhesus monkeys. It also discussed the study of Gehrs et al. (1997) on immune effects in rat male offspring. Of these, only the Gray et al. (1997a) and Mably et al. (1992c) studies were retained in SCF (2001), while new studies by Faqi et al. (1998) and Ohsako et al. (2001) on reproduction in Wistar rat male offspring and Holzman rat male offspring, respectively, were included.

Among the criteria for evaluating and selecting the studies accepted as a basis of TDIs that may be identified in the SCF (2002, 2001) opinions (Annex 8C/Table 3) many seem logical and defensible, even necessary for quality control. Also other, unstated criteria exist for selection of pivotal studies, and may be justified e.g. on the basis of the general practice in toxicology. Specifically, GLP and other methodological guidelines and evaluation criteria influence the choice of acceptable low-dose studies. Replication and confirmation of studies may also be considered as a requirement especially in the case of dioxins for which non-reproducible 'candidate pivotal' studies have been published (cf. 4.2).

However, among the above selection criteria for studies, some seem problematic and ambiguous. Foremost among these are the following (numbered as in Annex 8C/Table 3):

2) The '**clear adversity**' of effects is difficult to establish objectively, also among organ or organism level emergent effects. This difficulty, related to the multi-dimensionality of effects and risks, should on the other hand not be construed as an artificially inflated obstacle to assessment; the variation in endpoints and stages of response could instead be utilized as a source of additional information (e.g. using effect biomarkers in an extended weight-of-evidence approach). The adversity of some low-dose and potentially pivotal effects is not well established; this regards e.g. neurobehavioral effects in rats, as compared with organismal developmental effects such as those on spermatogenesis. However, e.g. Markowski et al. (2001) regarded the neurobehavioral effects as significant developmental outcomes, and noted that the effect levels approach or even fall below background human body burdens (cf. 4.2).

3) The '**clear relevance**' to humans of animal effects is also difficult to define unequivocally (cf. 4.1, 4.2). Its definition hinges in part on a somewhat subjective general and effect-specific evaluation of interspecies comparability that is based also on supportive information e.g. about conserved effect mechanisms or about the physiology of the species compared. This difficulty may also be exaggerated. For instance, even if a behavioural anomaly in monkeys does not occur in a similar form in humans due e.g. to differences in overall behaviour, or the relevance of a subtle behavioural effect is not the same (as emphasized by SCF 2001), due e.g. to social compensation mechanisms, the effect in itself may have a generalizable neurological and general biological basis and may be an indication of broadly similar and significant risk also in humans. It is at any rate evident that particularly in neurobehavioral respects monkeys are more closely related to humans than rats. Regardless of these considerations, in as far as the human relevance is accepted as a key criterion, the decision of SCF (2001) to limit itself to rat studies contradicts such a line of thought. It is interesting that because of the difficulty of interspecies extrapolation and because of the need for data from human-like experimental animals, the

previous Dutch health risk assessment for dioxins (HCN 1996a) stressed the need to utilize the information on low-dose effects in monkeys.

4) Criticism is in place of the statement of SCF (2001, p. 14) that "In view of … the firmer basis for extrapolation from the **pivotal rodent** studies that is now available, … decided not to include the study (on monkeys) as a pivotal study …". This is problematic on principal grounds, suggesting that those species that have already been studied (for reasons that may be not relevant to human health) could be continuously prioritized if only some added support e.g. for extrapolation is accrued. Such a criterion of study selection would imply that additional information from more species be unexploited. More generally, this carries the threat of a self-imposed limitation in knowledge and view, and may also influence knowledge production by over-emphasis on established knowledge, in a vicious circle. It seems preferable that also within definitions of TDI, possibilities be actively sought to utilize more fully information on additional species (and other entities) without bias toward favouring traditional ones. There is a trade-off between specificity and reliability of information on one hand and coverage of variation and human relevance on the other (the latter can also be seen as attributes of high-quality information in a wider sense); the SCF (2001) stance tends to push the balance toward the former (cf. point 2).

The comment of UBA (unpublished 2002) questioned the decision to limit the knowledge base of the assessment to studies in only one species, rats, and in fewer strains, sexes and development stages. As a general methodology, the deliberate limitation to one species differs from the accepted approach in ecotoxicological risk assessment as practiced e.g. in EU chemicals management, where a basic requirement and a strong determinant of safety factors is the availability of data on several species representing several groups (EC 2003a). This approach may not be directly applicable to human toxicology, but points to a more information-inclusive direction. The use of data on other taxa (including humans) does not have to be made in a similar manner, and may e.g. account for differential uncertainties and make use of the increased knowledge of similarities and differences in metabolism and in effect types and mechanisms between species.

Dose conversion and pharmacokinetics

To derive no-effect doses that would be equivalent to human long-term exposure (Estimated Human Daily Intake values), SCF (2001) in its revised opinion utilized additional data on pharmacokinetics of TCDD in rats, obtained by Hurst et al. (2000a,b). Basically, SCF has relied on the single-compartment (whole-body) pseudo-first order kinetic model of dioxin depuration. The conversion between acute and sub-chronic maternal doses in the rat using the power function model fit by SCF (2001) to the data of Hurst (2000a,b) was shown to be robust, and also the conversions between maternal and offspring steady-state doses seemed reasonably supported by the correlations (the statistical significance of the correlations was not given). Nevertheless, the bounds of these empirical models could be better acknowledged (cf. SACN and COT 2004). They may depend e.g. on body fat stores (as stated by SCF 2001) and dose levels (Carrier et al. 1995a,b). Notably, the models may not apply to non-TCDD DLCs (cf. Brewster and Birnbaum 1987, Abraham et al. 1989) and humans (Van der Molen et al. 2000).

In particular, the models were derived for the rat, and their applicability to humans may be questioned (cf. Annexes 7E and 8B). The distribution model, the approximating power function model and the associated dam-foetus and exposure duration conversions may not be consistent with the demonstrations of improved dose estimates for humans obtained e.g. by using more complex physiologically based generic mammalian models (Carrier et al. 1995a,b), by considering age dependencies in depuration and pharmacokinetics (Van der Molen et al. 2000) and by specifying trans-placental pharmacokinetics between mother and foetus (Kreuzer et al. 1997), or other models and data that might account for some of the above factors.

The SCF (2001) at the outset of the TDI definition laid down the paradigm that the uncertainty factor should account for the possible **inter-species** differences between experimental animals and humans in pharmacokinetics and toxicodynamics to TCDD, and to the potential inter-individual variation in susceptibility in humans. They then stated (p. 15) that an inter-species pharmacokinetics based uncertainty factor was not required since the default pharmacokinetic factor was replaced by actual data in calculating the body burdens used to scale doses across species. This seems somewhat questionable, as

the data utilized are largely limited to rats, and do not cover extensively and explicitly dissimilarities between rats and humans. Little scaling of data across species and little consideration of the factors and uncertainties involved has been applied (see also SCF 2000).

For **inter-individual variation** in pharmacokinetics a default safety factor of 3.2 was assigned as recommended by WHO (1993a). This was backed up by the observed difference in TCDD half-life between humans giving a data based factor of 1.5 (ratio of 95 % CL and mean). Since these data came from occupationally exposed males and may not adequately cover females, the default safety factor was chosen, i.e. in this case stricter than that indicated by the data. It should however be added that these occupational data not only exclude gender differences but also variation in pharmacokinetics between age groups and between persons of variable uptake and metabolism through genetic disposition, nutrition or health. For instance, the occupational data may be biased also due to the healthy worker effect, conspicuous in epidemiology and shown also for dioxins (e.g., Vena et al. 1998) and possibly applicable to nutritional and metabolic anomalies. If accounted for, data on such variability might on the contrary increase the safety factor over the default.

Toxicodynamics and LOAEL-NOAEL extrapolation

With regard to toxicodynamics and the sensitivity of different animal species, the SCF (2001) states (p. 16): "studies of AhR binding affinity and adverse responses directly dependent on AhR suggest that humans are less sensitive to TCDD than responsive rodent strains". This seems a rather one-sided generalization, as there are data suggesting equal or even reversed sensitivity between humans and rodents for some responses (Lucier 1991, cf. 5.3.2). On the other hand, it has not been shown convincingly that "primates and also humans appear to be more sensitive than rodents for functional developmental effects caused by *in utero* and lactational exposure to PCBs and PCDD/Fs" (e.g., Brouwer et al. 1995).

The statistical and conceptual constraints and problems of the NOAEL have been emphasized, and alternatives to NOAELs including benchmark doses (BMD) have been searched (e.g., Sand et al. 2002, Dekkers et al. 2001, Renwick et al. 2003, cf. 4.1, 4.2.10, Annex 8D). NOAEL is an arbitrary measure of insignificant risk level as it depends on study design (dosing regime). The NOAEL may also utilize the data sub-optimally. For instance, McGrath et al. (1996) demonstrated for immune and enzymatic effects of TCDD that a modified benchmark dose approach using the Sigmoid-Emax function applied to dose-response data by nonlinear regression provided a significantly improved fit, and that the confidence in the estimation of an ED10 was especially improved through the use of multiple datasets.

Brouwer et al. (1995) further pointed out that experimental NOAELs are not necessary when LOAELs are already at levels that correspond to levels within the range of current human background body burden. This line of reasoning may be seen in relation to the definition of TDIs, margins of exposure or other such benchmarks of exposure and risk levels (see below).

Safety factors

Safety factors are the decisive element in TDI derivation, in addition to the selection of pivotal studies (and are dependent on the relevance and quality of these studies). The importance of safety factors can be illustrated by the fact that in the definition of TDIs for dioxins they have varied by orders of magnitude in authoritative assessments, e.g. from 250 in Ahlborg et al. (1994) over 100 in HCN (1996a) to 10 in WHO (1998) and 3.2 in SCF (2001).

IPCS (2000) recommended that when a default value is replaced by a value based on quantitative chemical specific data the value should be termed an 'adjustment factor', and the term uncertainty factor retained for the default values shown in the risk assessment scheme above. Thus the "safety factor" would be the product of chemical-specific adjustment factors (for aspects where data are available) and default uncertainty factors (for the remaining aspects where data are not available).

General guidance for safety factor designations has been offered by the WHO (1993a), suggesting that safety factors be defined based on:

- Inter-individual differences in pharmacokinetics and pharmacodynamics
- Inter-species differences in pharmacokinetics and toxicodynamics
- Extrapolation from LOAEL to NOAEL
- Extrapolation from (sub)acute to (sub)chronic toxicity.

Most importantly, the SCF (2001) concluded that no safety factors are needed due to inter-**individual variation in susceptibility**, without further justification or comment. This is a remarkable statement, considering e.g. the standard guidance for safety factor designations and the empirical data on inter-individual variation also *in vivo* in human populations as approximated by biochemical effect markers (see above 5.3.2). This is in contrast with the assessment e.g. of ATSDR (1998) that included an inter-individual variability of sensitivity in humans comparable to that in experimental animals, i.e. a factor of 10. Also ECETOC (1995) has explicitly stated that the partial safety factor for inter-individual variation due also to susceptibility always has to be above unity. This decision of SCF (2001) to drop a safety factor based on inter-individual variation was severely criticized by UBA (unpublished 2002) (Table 27). It has not been possible to locate and scrutinize comments of other parties on this draft or to the discussion on safety factors in TDI.

The German claims are in some respects justified or can be supported, e.g. the inter-individual variation may be substantial (see above), primates are justifiably human-like, and generally other non-rat species are desirable. However, some claims and arguments seem selective and biased, e.g. HCBz inclusion can be debated; reference to Burleson mouse study omits conflicting later results by Nohara et al. (2002); and epidemiological studies, while generally desirable as checks, do neither confirm non-effect nor effect, and yet may add weight to more stringent or more lenient risk criteria.

A compound safety factor of 10 was applied in the earlier WHO evaluation of TDIs for PCDD/Fs (see e.g. van Leeuwen et al. 2000). Specifically with regard to safety considerations in immunotoxicology, Richter-Reichhelm et al. (2002) concluded that despite some differences in immunocompetence at birth and after lactation, and differences in the time frame for maturation of the immune system, reaction types are thought to be common, comparable and similar in human childhood and early adolescence and the postnatal lifetime of laboratory rodents. The authors also felt strongly that regulatory steps urgently need to be initiated to incorporate some relevant aspects into existing test guidelines for testing developmental immunotoxicity.

Additional insights in uncertainty evaluation are provided by analysis of the justifications and arguments for and against uncertainty factors used in the derivation. These arguments combine qualitative and quantitative aspects, and are in many cases related to fundamental epistemic and policy principles and decision rules, i.e. uncertainties at a high level (Table 27).

Table 27. Characterizations of the treatment of some key uncertainties in relevant authoritative international and national assessments of human health risks from dioxins and in proposals for human tolerable daily intakes.

Assessment	Pivotal effects and studies	Uncertainty factors (UF) used in deriving risk criteria	TDI/TWI/TMI1 pg g bw^{-1} $time^{-1}$
The Netherlands 1996[2]	Rhesus monkey neurobehavior (Bowman89) and immunosuppression (Neubert92b) + human infant neurodevel (Huisman95a), hormone status (Koopman95), immune status (Weisglas95)	2 from LOAEL-NOAEL extrapolation x 5 from interspecies variation x 10 from inter-individual variation x=100	1
WHO 1998[3]	Rhesus endometriosis & neurobehav (Ries93, Schanz89) rat male offspring rprd (Gray97a, Mably92a-c) and immune effects (Gehrs97)	e.g. pharmacokinetic differences accounted for but not very extensively or explicitly	1-4
EU/SCF 2000[4]	approximately as above (but with different body burden estimation and dose conversion)		1-3 (7-21 TWI)
EU/SCF 2001[5]	Rat male offspring rprd (Gray97a, Mably92a-c, Faqi98, Ohsako01; Rhesus and rat immune effects omitted)	10 overall, unspecified	2 (14 TWI)
USEPA 2000[6]	including carcinogenicity in humans, possibly not a critical endpoint(s); assessment is debatable	not applicable (cf. page 190)	not applicable (cf. page 190)
UK 2001 (2004)[7]	Rat rprd devel (Faqi98); lacking confirmation and consistency with other studies noted Tumors (various studies)	LOAEL-NOAEL extrapolation 3 x inter-indiv variation in toxicokinetics 3.2 (+half order of magnitude due to TEFs and uncertainties due to rat kinetics etc. noted)	2, rprd/devel (8, cancer)
Germany 2002[8] (unofficial)	more endpoints, species, dosing ages/periods and dioxin-like agents (HCBz) requested	more (empirical) UFs required for pharmacokinetics, LOAEL-NOAEL, dose and animal-human extrapolation, individual variation, additional DLCs	1
Japan 2004[9]	Rat female rprd development (Gray 97b)	10 LOAEL-NOAEL extrapolation	4

References and notes: *TWI/TWI/TMI=tolerable daily/weekly/monthly intake; [2]HCN 1996a, [3]WHO 1998; [4]SCF 2000; [5]SCF 2001; [6]USEPA 2000a; [7]COT 2001,SACN & COT 2004; [8]UBA unpubl. 2002, comment on SCF 2001; [9]Sumida & al. 2005.*

Summarizing evaluation of the basis of TDIs

The 100-fold increase in apparent confidence as to dioxin risks that is displayed in authoritative health risk assessments of dioxins during the past decade may only partly be explained by improved scientific information unequivocally dispelling uncertainties during this relatively short time. It seems rather likely that as new evidence has accumulated on low-dose effects of dioxins, standard and also some data-based safety factors would have led to impracticably and alarmingly low guideline values for intakes, even below present intakes for much of the population, and thus pressures have amounted to justify lowered safety factors.

Information on only one model species (rat) and one endpoint (disorders in reproductive development of offspring) has increasingly replaced a more multi-dimensional effect assessment. This may be partly justified by that this receptor and endpoint are the best known and the most sensitive identified (depending on the criteria for study selection with regard to both the quality and relevance of information), but leaves the assessment vulnerable to important limitations.

Such a critical evaluation of the present TDIs and the process of their definition should not be taken to imply that TDIs should (have had to) be necessarily lowered. It only seeks to scrutinize the arguments used and their potential weaknesses and gaps in TDI definition, also to identify possibilities for alternative interpretations and evaluations. That is, the criticism avoids taking a stance to the matter *a priori*. We are instead concerned with the argumentation (or the lack thereof) in the previous official assessments, and how the scientific evidence has and has not been used therein.

There are conceivable reasons for not lowering TDIs, as well as for lowering TDIs. Reasons for a more lenient TDI might include e.g. human population evidence (and strict criteria for its quality) and the lack of noticeable severe effects even from previous higher population exposures, i.e. further reducing the safety factor from interspecies extrapolation; further pharmacokinetic considerations including lower cumulative distributed doses; interactions (also antagonistic) and associated developments in TEFs; alternatives to NOAELs; more restrictive definitions of adversity. Even the additional consideration of variability does not have to necessarily lead to lowered TDIs, especially if a more explicit (and theoretically better founded) probabilistic approach be adopted instead of the use of repeated safety factors, as the combined probability distributions would narrow the spread in risk estimates based on repeated worst-case assumptions. For instance, if heterogeneity of human exposures or susceptibility to dioxins e.g. due to polymorphisms would be shown to partly cancel out each other, this could have (lowering) impacts on safety factors and TDIs.

Thus, it can be conceivable on scientific grounds (although politically and pragmatically problematic) that TDIs for dioxins would even be increased instead of being decreased; this is largely dependent on evaluation of the evidence and the overall rationales in deriving a TDI. Also a decision to exclude DLCs from TDI in effect implies a more lenient TDI for those included; correspondingly, apparent risks are increased if additional DLCs (such as dlPCBs) are included. Such decisions are partly political and pragmatic, and their political and scientific dimensions should be more clearly disentangled. Such decisions should preferably also be based on more transparent inference where the reasons for adopting some methodologies and procedures and for omitting some factors and arguments and emphasizing others would be given, instead of being silent on key issues e.g. related to human variability. In particular, such decisions (on TDIs) should be based on assessments and inference more fully acknowledging and more explicitly indicating their limitations (cf. 8).

As an overall evaluation and a methodological commentary, dioxin health risk and intake criteria (and thus the present key drivers for EU risk management) are not indisputably derivable from scientific facts. They involve value judgment, conventions and pragmatism, and are subject to political decisions, which should be acknowledged as such (also to allow clearer role for and more intelligent use of scientific arguments and more productive debate).

5.5.2 Translating tolerable human intakes to allowable fish concentrations

No clear and detailed methodology has been published in connection with the EC (2002a) recommendations for defining how allowable levels (maximum, action and target levels, cf. Annex 11) of dioxins in fish and fish products, including feeding-stuffs, are related to TDIs.

The basis for maximum or action levels in fish is affected not only by the assumptions and judgments in deriving TDI or TWI values, many of which involve great uncertainties, are unsubstantiated by scientific facts presently or even permanently, and are essentially value-based (see above). The basis of such allowable levels additionally depends on assumptions and judgments concerning many other factors previous to intakes in the risk chain. Importantly, the limit values in fish (and in fish-based foods and feeding-stuffs) also depend on **fish consumption** and its relation to exposure from **other sources**, and associated uncertainties.

Key factors in fish-based dioxin exposures that have not been clearly addressed but ought to be so when defining allowable fish action levels include the following:

- What fish species (wild and farmed) and qualities (e.g. sizes of herring) are consumed
- How frequently and what quantities of fish per meal are consumed
- Whether the fish is consumed directly by humans or as feed to non-human animals producing food for human consumption, and what transfer models from these animals to humans are applied in the latter case
- What is the consumption of other dietary items, including fish from other waters.

Baeyens et al. (2002) estimated on the basis of levels of PCBs and WHO-TEq_p/ΣPCBs ratios in North Sea fish that, for an adult of 75 kg, 10-30 g fish d^{-1} would suffice to provide an exposure in excess of a TDI of 1 pg WHO-TEq_p kg^{-1} d^{-1}, if 20 % of the diet consists of fish. These authors did not include PCDD/Fs and did not consider the composition of diet, variations in fish amounts consumed or in body weights; neither did they evaluate the relationships with alternative TDIs.

As shown above (3.4), all these factors **vary** considerably between regions, population segments, ages and also sexes. An additional factor or class of factors that may need to be specified in definitions of allowable fish contents is for what group of consumers they are intended. Specifically, a distinction may have to be made between adults and children in quantitative analysis of allowable levels of intakes and consumption, e.g. as children have a body weight several fold lower (besides having potentially greater susceptibility). In principle, allowable levels of consumption (and intakes) might be derived also for breast-fed children and foetuses. However, these would have to be tied to the body burdens and subsequently consumption levels of pregnant and nursing mothers, which involves still greater uncertainties due to the complexity of related toxicokinetics and the lack of data and models to take them into account.

The **time** dimension is highly important in this connection. A fixed level of allowable fish concentration supposes that fish of a certain (e.g. just unacceptable) dioxin content would give rise to a certain (just unacceptable) TDI. Hereby uniform time-averaged fish consumption is assumed, regardless of the benchmark. However, the risk from consumption of fish exceeding e.g. 4 ng TEq g^{-1} ww is dependent on how often and how much of it is consumed over the long term. Those consuming seldom and little may during those short periods safely eat fish containing even orders of magnitude higher levels (SACN and COT 2004); conversely, regular high consumers may need to worry for fish containing even lower dioxin levels (at least if health benefits from fish are ignored and if these subjects belong to high-risk groups on the basis of their susceptibility, such as pregnancy). With dioxins the long-term average exposure is still more important than with many other intake media. The time dimension is also associated with the need to take the various age groups and exposure periods into account.

With strict and fixed levels of allowable fish concentrations counter-intuitive effects may result. It can be difficult for people to understand and accept that a fish may be risky under some circumstances and risk-free under others, and that the limits are not only (educated) guesses like those of TDIs, but also disregard many additional factors of variation e.g. in fish consumption.

This can be illustrated by some example calculations. A maximum tolerable level of consumption of medium-sized Baltic herring would be every other week for adult (60 kg) or every other month for a young child (10 kg) based on the lowest TDI of 1 pg TEq kg^{-1} d^{-1} and an average content of 5 and 1 pg TEq g^{-1} ww in herring and

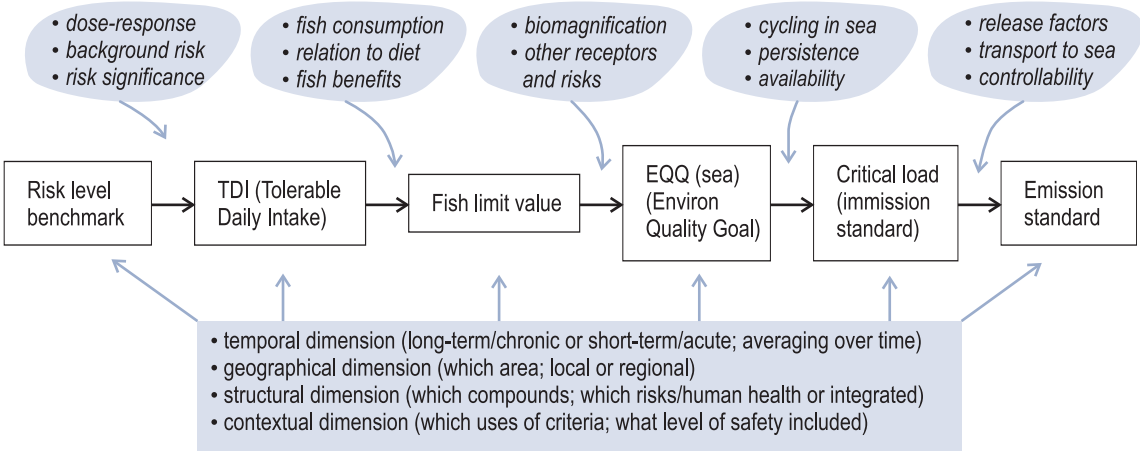


Fig. 18. A simplified risk-based 'upstream' process of deriving quantitative human health risk management criteria for dioxins in fish, and of crucial factors and associated questions and models.

other foods, respectively. Such calculations of the consumption of (fatty) Baltic fish that corresponds to tolerable intakes, in average or more or less realistic worst case exposures, have however limited utility in defining risk management goals for DLCs in Baltic fish. This is due to the uncertainty regarding the consumption of other fish and other foodstuffs, even if the TDI level would be fixed. Nevertheless, the calculations demonstrate the deficiencies in the definitions of allowable fish levels. For instance, only very crude calculations are possible regarding body burdens in the critical young developmental stages. A fundamental limitation of such calculations in a more comprehensive nutritional and health context, and a reason not to pursue them further, is that they disregard the relative benefits from various foods.

The dependency of health based quality criteria for environmental matrices on fundamental models, assumptions and decision rules involving considerable value judgments can also be seen in the great variety in sediment benchmarks for PCDD/Fs that have been stipulated or proposed by various regulatory bodies (Wenning et al. 2004). It cannot be claimed that any of such benchmarks, or yet others, are definitely best.

5.5.3 Other human health risk criteria and factors

In principle, many quantitative criteria may be applied as decision supports. These include, following the chain of dioxin risk formation and subsequent stages of assessment (Fig. 18):

- (Allowable emissions)
- Allowable loads on an environment
- Allowable **body burdens** (accounting for pharmacokinetics more directly)
- Allowable **tissue doses** (e.g. in foetus or critical organs, cf. the TTDs defined by ATSDR 1998)
- Allowable **effect levels** (if effects can be conclusively related to dioxin exposure)
- Allowable **population risk** (accounting for populations affected)
- Allowable **risk/benefit** ratios or relationships
- Specifying risk benchmarks to various age **groups and lengths of exposure** (see ATSDR guideline values for dioxins in soil, DeRosa et al. 1999a), and specifying such criteria in terms of target and action or limit values.

Among other quantitative criteria for risk, the **reference dose** (RfD) approach as used for USEPA to derive still safe (almost unsafe) exposure levels for humans is a common metric. RfD is in principle and in many details similar to the traditional procedure for deriving effect and no-effect levels in defining TDIs or tolerable (lifetime) average daily doses (ADDs). Greene et al. (2003), on the basis of e.g. USEPA (2000a) and JECFA (2001) assessments reviewed the human and experimental animal data for non-cancer effects of dioxins and identified a NOAEL of 13 ng kg^{-1} (maternal body burden) as the most pertinent for deriving a RfD for humans. A RfD of 1-10 pg/kg d^{-1} (TEq) was found to be consistent with the objectives of this risk criterion. They concluded that maintaining a lifetime average daily dose below this level should prevent non-cancer effects in 'virtually all persons', and that this value is consistent with the

JECFA recommendation of 70 pg kg^{-1} mo^{-1} (ca. 2 pg TEq kg^{-1} bw d^{-1}). The experimental animal NOAEL used as point of departure is below that selected by SCF (2001).

The selection of **critical endpoint** has crucial influence on quantitative risk benchmarks. Gastel (2001) argued that the differences in dose-response between dioxin-induced genes suggests that even with the identification of a sensitive marker, a safety factor of 10 still needs to be employed to ensure that the most sensitive marker has been found. This may be regarded as a LOAEL-NOAEL extrapolation based on biochemical responses. He thus derived from the LOAELs of 5 ppt in rat liver a NOAEL of 0.5 ppt liver, assumed to be equivalent with 5-50 fg kg^{-1} d^{-1} in human consumption. This is in the range of USEPA (2000) reassessment and the '**BESTNOEL**' of 15 fg kg^{-1} d^{-1} (Gastel and Sutter 1995), and is significantly below the current exposure of the general population. Whether such a NOEL and benchmark dose estimate is justified is open to debate. As argued by Gastel and Sutter (1995), it is at least based on empirical evidence instead of arbitrary default extrapolations. However, application of even such empirically based benchmark definition approaches is fundamentally a matter of value judgments concerning what effect indices are considered adverse (NOAELs, instead of only NOELs), and what levels of safety are preferred, considering sources of variability in effects.

As discussed by Hays and Aylward (2003), the USEPA (2000a) opinion on RfDs and generally safe levels of exposure to dioxins, i.e. that such levels cannot be established, departs radically from those of many other organizations. These authors deduced from USEPA (2000a) that safety factors in the range of 250-10000 would need to be applied to the identified non-cancer 'point of departure' benchmark to derive a RfD. They also noted that the conclusion based on USEPA's assessment that safe levels of exposure are 2-3 orders of magnitude below current background intakes is that no degree of control of dioxin sources will be sufficient. This inference can however be questioned, in the absence of specific stances of USEPA (2000a) concerning control strategies and their sufficiency.

Gaylor and Aylward (2004) demonstrated the great differences between estimates of effective dose levels in pivotal studies of experimental **animals**, depending on the models and estimations procedures applied (cf. 4.2.10). In a generalizing fashion, the hierarchy of reasoning in such estimations specifically of effective doses may be divided e.g. in the following interacting steps:

- Definition of the **general model** for estimation
- Specification of the **dose-response** model
- Selection of the **data** to be used, including dose metrics and conversions, and associated criteria for their relevance and quality
- Selection of the **percentile in the population** (e.g., ED50, ED10 or ED01)
- Selection of the appropriate **statistic** and measure of statistical confidence (e.g., lower 95 % limit).

It could be noted specifically in connection with the alternative estimations by Gaylor and Aylward (2004) that the uncertainties surrounding e.g. the conversions from maternal to offspring doses were considerably less than those associated with the selection of the basic estimation procedure, of the percentile of the dose metric, and of the confidence level, all of which in some cases introduced 100-fold variation in effective dose estimates (cf. 5.3.2).

The uncertainties and ambiguities of just how poisonous DLCs are contain a great amount of expert and value judgments, conventions and even political contents, and can not be resolved on the basis of some exact scientific procedure. On the other hand, there are overlaps and interactions between the 'scientific' and 'political' grounds for defining a significantly effective dose and subsequently of an acceptable dose, intake etc. For instance, some theoretical and empirical justifications of a scientific nature can be found for the decision whether a ED50, ED10 or ED01 is more appropriate in a given situation, partly regardless of the additional value-laden considerations e.g. of how precautionary and conservative benchmarks ought to be applied (cf. 8).

Additional factors or aspects of relevance in defining human health risk criteria include the following:

- The **intakes** of dioxins by humans in the Baltic Sea countries, coming almost wholly from diet, have **declined**. A levelling off of the declines may however occur at least for some parts of the population, as the rates of decline for also non-fish foods decrease. Such a development has been noted also in TEq levels in Great Lakes trout (Huestis et al. 1997).
- Despite these declines, the general population exposure due to dietary intakes of PCDD/Fs and dlPCBs (e.g. as average

daily intakes, ADIs), are very **near** the present recommended **TDI** values for long-term human intake. The recommended TDI values, based on animal data, involve considerable uncertainty and may not be directly applicable to humans; some uncertainty factors may advocate lower, some on the other hand higher TDIs.

- The ADI/TDI ratio is particularly unfavourable in **foetuses and breast-fed** infants. These measures of risk are not necessarily comparable to those for adults. Foetuses and probably also infants are in some respects still more susceptible and sensitive than older persons to some effects of DLCs; on the other hand, other factors may reduce the relative risks. In any case, the elevated risk accrued by infants comes from heavy exposure during a relatively short time.
- Also in some **high consumers** of dioxin-rich diet such as highly contaminated fish will ADI/TDI levels still exceed unity, and are reflected in body burdens of some of such persons, being at the level of the high-end exposures measured in Seveso
- According to recent Swedish studies (cf. 3.5), in some **10 % of the population** the ratio of average daily intake to tolerable daily intake (ADI/TDI) is >1
- The **total dioxin intakes** of particularly Finns, despite their high intakes from fish including Baltic Sea fish, are lower than in many other European countries, while they are closer to European averages in Sweden and Denmark
- Intakes of **other DLCs** by populations in the Baltic Sea countries are poorly known, with some exceptions (such as the putative DLCs, PCDEs).

5.5.4 Ecotoxicological risk criteria

General

Some generic ecotoxicological benchmarks have been proposed for PCDD/Fs or dlPCBs (or DLCs collectively) based on TEqs. These proposals vary in terms of their area of application, such as the exposure media, and their basis of derivation; they also vary greatly in the quantitative cut-off point proposed (see e.g. CCME 1999, and Wenning et al. 2004 for review of benchmarks for dioxins).

Wenning et al. (2004) evaluated that there is not 'sufficient' data to develop acceptable tissue concentrations for representative organisms, particularly estuarine and marine organisms for which virtually no PCDD/F tissue residue-toxicity relationships have been established. This however depends on the goals set for benchmarks and on the uncertainties one is willing to accept. In principle, tissue residue based benchmarks are more appropriate than basing benchmarks on biota-sediment accumulation factors (BSAFs) or on equilibrium partitioning that involve more indirect measures of exposure. All these benchmarks moreover share uncertainties of dose-response relationships.

Benchmarks for non-human wild animals consuming fish based on fish tissue residue levels have been proposed e.g. by USEPA (1998b) and CCME (1999), invoking a guideline value of fish tissue concentration for mammal piscivores of c. 0.8 pg TEq g^{-1} ww (Wenning et al. 2004). This is below the limit or target values for human foods in the EU recommendations for food (4 pg TEq g^{-1} ww, cf. above). The TEq levels in many Baltic fish species and specimens exceed these proposed guideline levels, sometimes by two orders of magnitude. The latter source also has proposed a tissue residue guideline of c. 5 pg g^{-1} ww for birds. However, birds have not been shown to be less sensitive or better able to eliminate DLCs, and the share of fish in the diet varies in both mammals and birds from infrequent usage to exclusive reliance on fish.

In addition to tissue residue based benchmarks, LOAELs, NOAELs or PNECs have been estimated for individual species regarding the dietary levels of DLCs, mostly as TEqs.

Ecotoxicological benchmarks for wildlife and associated risk evaluations and characterizations

In the following indicative quantitative ecotoxicological risk assessment and characterization, measured levels of DLCs (as TEqs) in body burdens or diets of relevant Baltic Sea species are compared with LOAELs, NOAELs or other threshold values of body burdens or dietary levels for various endpoints or responses in these species (or closely related species). In the absence of body burden data, extrapolations can be made from other tissues, environmental levels and from other species based on statistical relationships, in an approach similar to that of Cook et al. (2003) for retrospective Lake Ontario trout effect assessment (Table 28).

These risk indices are tentative and in some cases may contradict observed population growth and well-being, and overestimate risks, while in other cases they may disregard and underestimate risks. The comparability of exposure and effects data is limited due to differences in the coverage of DLCs (especially dlPCBs), in TEFs applied, in species or strain studied, in their age, sex and general condition, in the tissues or media analyzed (such as egg or adult tissue). Estimation of risk benchmarks on the basis of dietary levels (which often have only been reported) is particularly uncertain, as for many animals the necessary basic physiological and kinetic data for conversion between diet values and tissue levels do not exist. Some effect or no-effect levels have been deduced from correlations and other information in field studies, and may not be specific to DLCs but reflect other factors.

While some of these difficulties may be alleviated e.g. by resorting to indicator compounds or surrogate species and to using empirical correlations between these and other above variables, in many cases meaningful quantitative risk indices cannot be produced. Such benchmarks are thus to be treated as rough estimates of effect or threshold levels. In some cases, particularly when derived through extensive extrapolations, they may be seen as order-of-magnitude approximations, although the dose-response curves are in many cases steep. The following points should especially be kept in mind:

- TEFs are assumed, but involve considerable uncertainties (see discussion above)
- In some cases benchmark exposure and effect levels (in TEqs) have been given for dlPCBs; this presents difficulties as comparability with PCDD/F-based TEqs is limited
- Dose conversions between matrices can not always be made, and dietary levels have to be used
- Extrapolation from other species has in some cases been used but should be viewed with caution due to interspecies differences in sensitivity (even on equal body burden basis)
- The calculation basis (wet, lipid or even dry weight) varies, and conversions are not always possible due e.g. to lacking data on lipid or water contents in the original sources
- Many different benchmarks have been given (for fish and birds)
- In many cases deviating values have been given
- Some variability is due to the system studied and measurement methods.

As an example of ecotoxicological benchmarks and of methodological issues and limitations in their basis and application, Traas et al. (2001) defined sediment quality criteria for dlPCBs based on effects on otter. Using uncertainty analyses, they evaluated that the acceptable sediment level depends considerably on the endpoint or response selected, and on the variation in exposure and response measures. In addition to the effective concentration benchmark chosen, i.e. the cut-off point estimate in the sensitivity distribution (e.g., EC1 or EC50), a cut-off point estimate needs to be chosen in the probability distribution for exposures, depending e.g. on degradation and biomagnification (biota-sediment accumulation). These choices led to estimates of toxicological sediment quality criteria for otters (based on litter size) ranging from 0.4 to 90 pg TEQ g^{-1} OC. The authors further noted a striking difference in NOAEL and other effects measures for males and females. These analyses illustrate that multiple factors and uncertainties affect the derivation of ecological (and other) environmental quality criteria. Thus, a single figure that would be unequivocally 'right' and generally applicable cannot be found. It may be preferable to try to identify the factors involved and their implications for informed decisions regarding such criteria.

Notwithstanding these obstacles and limitations, this basic assessment of ecotoxicological risks based on observed and no-effect or effective body burdens (B_{obs}/B_{NOAEL} or B_{obs}/B_{LOAEL}, or generally $B_{obs}/B_{benchmark}$ or similarly diets, may be summarized as follows:

- Among fish-dependent species in the Baltic Sea region, B_{obs} values exceeding B_{NOAEL} or B_{LOAEL} have been encountered but for **several species data do not allow** such assessment
- In many cases B_{NOAEL} or B_{LOAEL} has been exceeded by B_{obs} during **earlier periods of higher exposure**, but in some populations the risks by such indications continue to be of concern
- Although there are scanty data on frequency distributions, it can be assumed that in most cases no-effect or lowest effect levels have been **exceeded by upper-range** concentrations, not by typical levels, as the generally skewed distributions cause average values above medians
- With several species, $B_{obs}/B_{benchmark} > 1$ has been obtained using estimates of no-effect

Table 28. Summary evaluation of ecotoxicological risks from dioxin-like compounds in fish-consuming Baltic Sea animals by comparison of body burden or diet levels and effect or no-effect levels for various endpoints and TEqs, estimated from experimental and field studies. Those data for different species, endpoints or compound groups are in parentheses. The species with likely adverse exposure levels or possibly or subtly adverse levels have been indicated by bold and italics, repectively. Note the concentration units and variable inclusion of DLC in TEqs. Cf. text and Annex 8D.

Species	Body burdens (diet levels), pg WHO-TEq_{DFP} g^{-1} (ppt) lw unless stated otherwise		Empirical benchmark doses for effects, pg WHO-TEq_{DFP} g^{-1} (ppt) lw if not stated otherwise		Other benchmark, notes on application
	Median (mean)	95th % (max)	NOAEL, effect	LOAEL, effect	
Ringed seal	c. **200** plasma[x], blubber[w]	800 TEq_{DF}[s]		c. **100** liver, retinyl palmitate[x]	dose-response ot retinoid effect?
Grey seal	c. *100*, plasma[x]; c. 200, blubber[w]	1500 TEq_{DF}[s]		(*100-200*, ringed and harbor seal)	apply to grey seal?
Harbor seal	**200 blubber**, fed BS herring[2] (100 TEq_P)[14]			**200 blubber,** immune[2], retinol[23]; 400, lymphocytes[3]	
Common (harbor) porpoise	200[k], 80[6], 90[o] ww blubber[0] (50 ppm ΣPCBs lw[k])		~3-50 ppm ΣPCBs ww/lw blubber[4]	~3-50 ppm ΣPCBs ww/lw blubber[4]	applicability of ΣPCBs based benchmarks originally proposed for Amer./>Arctic mar mammals ?
Mink	(**herring/sprat 8-10**, all BS fish, 5 ww)	(all BS **fish 30 ww**)	0.08 pg g^{-1} bw d^{-1} and 0.3 ww diet, rprd[m] (0.7 ww fish[a])	2 pg g^{-1} bw d^{-1} and **10 ww diet**[m]; 20 ww feed, rprd[b] (1-4 ww diet (fish)[a])	EC50 160 ww, kit survivalp (40 pg g^{-1} bw, AHH-TEF based EC50, kit survival)[p]
Otter	400-*4000* muscle[20]			*2000 EC1*, 5000 EC90, liver/ plasma, vitA[f]; 5000 liver, immunotox, thyroid, vitA[f]	corresponding diet (fish) level 10 (EC1) and 30 (EC90)[f] mink bechmarks applicable?
White-tailed sea eagle	**200** BioTEq ww egg[u]; **300-6000** ww egg[21] (3000 lw egg)[1]	**400** BioTEq ww egg[u] (7000 lw egg)[1]	(100 ww egg, Cyp1a1)[v]	(**200** ww egg, Cyp1a1[h]; 210, survival)[v]	applicability of bald eagle data or LC50 data in other birds?
Herring gull	*20000* TEq_P, *lw liver*[16]	30000 TEq_P, lw liver[16]		*800 ww liver*, porphyria[r] ; 1000 ww egg, corticoster.[24]	porphyria "subclinical" (Fox01); 560 ww egg, hatching LD_{19}[g]
Lesser black-backed gull	20000 TEq_P, lw liver[16]	30000 TEq_P, lw liver[16]		(1 pg g^{-1} d^{-1}, 20-50 ww egg, embryo mortal)[j]	
Caspian tern	?		(2-20 pg ml-1 ww blood)[g]		750 ww egg, hatching LD_{50}[q]
Common tern	?		25000 lw liver, 600 ww diet, Cyp1a1, tT4[12]		
Black cormorant	120 ww, 2700 lw[6]		(300 ww egg[19])	2000-10000 ww egg[10] (100 ww egg[o]; 1000 ww egg, EROD, double-crested c.[11])	double-crested cormorant data applicable?
Eider	?			c. 20000, retinol, thyroid[t]	
Chicken	0.3 ww egg[28] (? from BS fish)	(? from BS fish)		6 ww egg, cardiac deform[26]	(150 ww egg LD50)[27]
Salmon	c. 10 ww (3 ppt ww diet[13])	c. 30 ww, (6 ww diet[13])		(6 ww diet, 0.9 ww liver[e], rainbow trout)	critical tissue levels uncertain; rainbow data applic. to salmon?
Sea trout	2-5 ww muscle[l]	*(20-30-40 ww egg)*[n]	(5 ww egg[c] , lake trout)[d]	*20 ww egg*, fry survival[22]	lake trout data apply to sea trout?
Rainbow trout	c. 50 ww (whole fish fed BS herring)[15]		(<38 pg l^{-1}, growth)[5]	1-6 ww diet, 0.9 ww liver[e,8]	levels lower (5 ppt ww whole fish) in rainbow fed dry-feed

References: *[a]Tillitt et al. 1996 (unclear whether lw or ww basis is used); [b]Brunström et al. 2001, approximately similar as the value given by Heaton & al. 1995 based on lower TEFs esp. for CB 126, seem, cf. Aulerich et al. 1988; [c]Cook et al. 2003; [d]Guiney & al. 2000; [e]Walter & al. 2000; [f]Murk & al. 1998; [g]Giesy & al. 1994a, threshold estimated for both Caspian terns and chicken and pheasants despite their differences; [h]Elliott & al. 1996, for bald eagle Cyp1a1 induction; [i]Bosveld & al. 2000, c. dietary LOAEL in Feyk et al. 2000; [j]Hoffman & al. 1996, threshold value for embryo mortality in chicken, cf. Giesy et al. 1995; [k]Berggren & al. 1999, comparable with the PCB-TEq levels of 90-300 pg g^{-1} ww blubber in female porpoises in SBS reported by Falandysz & al. 1994e and 2002b; [l]Bjerselius & al. 2003; [m]NOAEL of 0.3, threshold dose of 2 and LOAEL of c. 10 ppt ww diet, and 0.08, 0.4 and 3 pg TEq g^{-1} bw d^{-1} for maternal toxicity, respectively, Heaton & al. 1995, Tillitt & al. 1996; [n]Walker & al. 1994 (cf. Comber & al. 2003) for maternal, waterborne and egg injection, respectively; [o]Ishaq & al. 2000; [p]Leonards & al. 1995; [q]Giesy & al. 1994a for double-crested cormorant; [r]Fox & al. 1998; [s]Koistinen & al. 1990; [t]Murk & al. 1994a; [u]Koistinen & al. 1997b, the TEqs differ from the WHO-TEqs for birds calculated from reported PCB levels; [v]Elliott & al. 1996 for bald eagle; [w]Koistinen & al. 1997a; [x]Nyman & al. 2003 (Bio-TEqs); [z]From dose-response curve of Shaw & al. 2003; [1]Helander & al. 2002, the TEqs differ from the WHO-TEqs for birds calculated based on reported PCB levels and can not be converted to ww based values; [2]Ross & al. 1995 (mean level in blubber of immunosuppressed animals; a LOAEL is difficult to define); [3]Ross 1996a; [4]Wagemann & Muir 1984, de March & al. 1998, ref. Berggren & al. 1999; [5]Mehrle & al. 1988; [6]Falandysz et al. 2002b; [8]Giesy & al. 2002; [10]Hendriks & Enserink 1996, based on levels in 1950's-70's (extrapolated from CB 153 and ΣPCBs) assessed to have caused adverse population effects; [11]Powell & al. 1997a for double-crested cormorant; [12]Bosveld & al. 2000; [13]Based on mean TEq values for the main diet species sprat (Karlsson & al. 1999) in GF (Hallikainen & al. 2004) and BP (Bjerselius & al. 2003); [14]Storr-Hansen & al. 1993a; [15]Isosaari & al. 2002b; WHO-TEqs for fish have been used; [16]Hario & al. 2004, conversion to WHO-TEq for birds in ww not possible based on the reported lw based or Safe-TEq based data; [18]Murk & al. 1996; [19]Powell & al. 1997b for double-crested cormorant based on field-collected cormorant extract; [20]Based on levels of ΣPCBs of 2-20 ppm lw muscle in otters in the BS region in 1980's (Sjöåsen & al. 1997) and ΣPCBs/TEqP ratio of 4700 from Leonards & al. 1998; [21]Tarhanen & al. 1989, based on samples in 1984-85, and including analyses of only PCDD/Fs and 0-ortho PCBs; [22]Guiney & al. 1997; [23]Simms & al 2000, based on relationship in weaned pups in Washington State; [24]Lorenzen & al. 1999, based on herring gull TEFs by Kennedy & al. 1996, being c. 2-fold lower e.g. for CB 126 than WHO-TEFs (Van den Berg et al. 1998) for birds; [25]Feyk & al. 2000; [26]Cheung & al. 1981; [27]Powell & al. 1996a; [28]Lind & al. 2002.*

levels (or lowest effective levels) that are based on **responses of unclear toxicological** and general biological significance, e.g. subtle and transient biochemical changes

- Among **marine mammals**, in Baltic harbour and ringed seals and by extrapolation also in grey seals the $B_{obs}/B_{benchmark}$ ratios indicate that toxic effects probably have been caused by DLCs, including even grave effects (as suggested by the reproductive, immune and endocrine effects), and at least more subtle effects may still be caused although populations have recovered.
- Among **terrestrial mammals**, wild mink and possibly otter seem to have been endangered by DLCs in fish; however, this evaluation depends on the assumptions regarding intake of Baltic fish in comparison with other dietary items (as with humans)
- Among **birds**, ambient levels may have exceeded toxic levels in white-tailed sea eagle and also some other species such as gulls. It should be noted that avian TEFs are higher than mammalian TEFs especially for TCDF, 4-PeCDF and 4-HxCDD, but lower for CB 156 and CB 118. Regardless, populations are thriving for most species potentially threatened.
- Among **fish**, WHO-TEqs (for fish) in Baltic sea trout are still above the levels of CB 126 (in TEqs) found to cause transient thyroid effects in lake trout, and far above the levels causing UDPGT enzyme induction (Brown et al. 2004). The TEFs are particularly uncertain for 1-*ortho* PCBs.

Ecotoxicological benchmarks and evaluations for domestic animals

Considering the Baltic Sea and its fish, mariculture as a food production system is of particular importance as a mediator and a potential target of DLCs. Rainbow trout is the key species produced, although in Kattegat salmon farming takes place and may even increase. Rainbow trout itself may not be subject to overt toxicological risks as its feed is relatively clean (TEq levels below the above dietary effect level, cf. Isosaari et al. 2002b). However, if fed exclusively Baltic herring, levels of c. 50 pg $WHO\text{-}TEq_{DFP}$ (fish) g^{-1} ww in whole fish result in rainbow trout. This is near the levels found to cause lowered survival in rainbow trout, and is mainly contributed by 4-PeCDF, not by dlPCBs as in the case of mammalian $WHO\text{-}TEq_{DFP}$s in fish for human consumption.

For **humans consuming rainbow trout**, it can be estimated based on levels in trout and on consumption amounts (cf. 3.5.1, cf. Isosaari et al 2004, Tuomisto et al. 2004b) that the intakes and risks are lower than those by consumption of other fatty fish including Atlantic salmon. Should the use of herring and sprat in rainbow trout feed change, the risks to human and other consumers would correspondingly change.

Among terrestrial animal production systems, pig and poultry as users of fish-based feeding-stuffs are of potential importance from a human health as well as from ecotoxicological point of view also in the case of Baltic fish. Of these, **poultry** is of particular concern, as chicken is the most sensitive avian species to DLCs and may thus also be a target of dioxin toxicity. The lowest reported LOAEL of 6 ppt TCDD ww egg (Cheung et al. 1981) is for cardiac deformation and thus a relevant effect. The mean levels in eggs are 20-fold lower (Lind et al. 2002). If however chicken were fed predominantly by Baltic herring or sprat or feeding-stuffs consisting largely of herring or sprat oil or meal, adverse effects could arise (or have arisen when levels in Baltic fish were still higher).

Farm **mink** also is a sensitive species to DLCs, and is presently a key target of Baltic herring, sprat and even vendace (in Sweden). It has been found that PCBs including dlPCBs and possibly PCDD/Fs in contaminated fish-based feed have caused adverse effects in farm minks, especially in previous decades in the Great Lakes (cf. 4.3.2). Some experimental studies (Brunström et al. 2001) have defined dietary NOAELs and LOAELs as low as 3 and 20 pg WHO-TEq g^{-1} ww feed, being within the range still encountered in Baltic fish (cf. Annex 8D). Thus, it seems possible that toxicological risks are caused by DLCs in those minks receiving a large proportion of fatty Baltic fish (especially large herring and sprat) or feeding-stuffs based on such fish in their feed. However, health impairment and especially reproductive disorders in mink depend also on many other factors, and the specific roles of DLCs and other contaminants are difficult to ascertain. Widespread and grave reproductive impairment has not been reported in minks fed Baltic fish.

Part B: Risk management analysis

“*eration is the process of trying to figure out what one do and is thus essentially evaluative. …To act for a ries with it the possibility that the reason for which ot be a good one, or at least not good enough. And ether or not it is good enough is one that necessarily question of whether or not it really is one's reason for l. Just as it is essential to benefits that they can be false they may be desires for things not worth having, so it reasons for acting that they may not be good reasons.*”

F. Schueler: Reasons & purposes – Human rationality and the planation of action, Clarendon Press, Oxford 2003, p. 164-165

6.1 Introduction and conceptualization

6.1.1 Policy contexts and contents

The environmental problems of the Baltic Sea cross political and jurisdictional boundaries. Simultaneously, many global environmental conventions and initiatives also concern the Baltic area. The multiple scales and dimensions of ecosystem management involve a broad range of interests and organizations. Traditionally different countries have had their own policies and regulatory activities on dioxins.

Relevant policy activities for dioxins in the Baltic and its fish include chemical policies as such but also e.g. marine protection, fisheries, food and health policies. Many international initiatives in the environmental chemicals area address POPs that include several groups of hazardous substances. Some initiatives, e.g. on PCBs, have not been labelled as POPs-related.

In the 2003 edition of the Master List of Actions on the Reduction and/or Elimination of the Releases of Persistent Organic Pollutants prepared by UNEP Chemicals, 47 countries around the world reported having restricted dioxin and furans emissions by establishing release standards. All other Baltic coastal states except Russian Federation were included in these countries. In the Baltic context, the cooperative ecosystem management approach has mainly been executed by HELCOM activities. With the European enlargement the influence of the EU regulation on the Baltic area has increased.

Many policies dealt with within this chapter are adaptable. Adaptive management sees policy and decision-making and implementation as an iterative process rather than a one-time exercise, and emphasizes the role of learning from successive management choices. Adaptability is a common idea in international policies and e.g. within the POPs Convention it enables parties to agree on activities for substances accepted as harmful while the negotiations and evidence reviews on more debated substances can continue (e.g., Eckley 2002). Likewise, the EU strategy for PCDD/Fs and PCBs involves adaptive steps (cf. Annex 12). Indeed, all policies, to be successful, may need to be adaptive in some sense. However, the contents and levels of adaptation vary, and in some areas a traditional and more rigid one-time management prevails. A parallel in fisheries management is incremental (as opposed to total) planning (Hildén 1997b).

6.1.2 Policy approaches and principles

There are different approaches to legitimizing policies. A general requirement for different states and institutions to cooperate with each other is that various regulatory and scientific organizations need to share some level of consensus about what constitutes sufficient evidence of hazard. Criteria for identification, however, can vary substantially across institutions. Substances may be evaluated differently e.g. depending on whether there is more concern about preventing potential public health problems or minimizing the economic impact of false positive identifications. An example of criteria for the harmfulness of dioxins is the current discussion in several policy arenas on the inclusion of other than toxic effects to risk assessment.

One of the factors affecting the relevant policy principles is the nature of policy action. Some initiatives work as a coordinating and indirectly or 'softly' harmonizing framework for different countries or levels of regime and between different sectors, e.g. in devising and reporting their separate action plans. Other initiatives set legally binding regulatory principles and activities.

As described in Part A, the problem in dioxin risk assessment is to evaluate the actual human health and ecotoxicological effect of the variable concentrations in the environment and of cumulative exposures to these mixtures. Consequently, the scientific community disagrees on the risk of the present environmental concentrations of dioxins. While there is a certain agreement that the risk of cancer is not very high, the risk of other effects such as developmental disorders might remain high (cf. 4, 5). Thus many

dioxin-related policies apply a precautionary approach (cf. 7.2-7.4).

Precaution is a principle for making decisions based on available evidence, when the evidence is considered highly uncertain. The precautionary principle is often restricted to cases of irreversibility of effect. It might also be applied to mitigate a harm that is ultimately reversible – if reversing the damage could be more costly that preventing it. There are also cases in which there are no uncertainties, for example when it is known that future generations will be harmed in some way (Ashford 2002).

The precautionary principle is in sharp political focus today partly because (1) the nature of scientific uncertainty is changing and (2) there is increasing pressure to base governmental action on more "rational" schemes, such as cost-benefit analysis and quantitative risk assessment that are difficult to combine with straightforward precautionary action. There is plenty of literature on policy issues relating to precautionary principle (e.g. Raffensperger and Tickner 1999, Harremoës et al. 2002, Haag and Kaupenjohann 2001, Ledoux and Turner 2002), notably in connection with long-term effects and risks of POPs in Northern environments (e.g., Godduhn and Duffy 2003, Eckley and Selin 2004, Eckley 2002).

Degnbol et al. (2003) emphasized that the precautionary principle has often been 'taken-for-granted', agreeable to all parties in principle but in vague terms, and then interpreted according to one's interests, notably in the fisheries area. Gray and Bewers (1996) stressed the need for a more scientifically based definition of the principle instead of acting on unsubstantiated perceptions, but did not specify the "nature of risk" legitimating application of the principle to chemicals. They also stressed the impracticality of having a series of definitions that apply individually to only one aspect of environmental damage, but did not consider the extension to and integration with other kinds of damage. Specifically in marine resource management, Johnston and VanderZwaag (2000) highlighted problems associated with the downgrading of scientific evidence and the dominance of adversarial groups, but mentioned only 'ethicists' (in favor of precaution), not others lobbying or arguing selectively. In the present connection it is particularly important that exaggerated and one-sided precaution may lead to even worse counter-veiling risks e.g. due to harmful surrogate products or processes, or to the loss of benefits associated with the risk such as health benefits from dioxin-laden fish (see general discussion by Tuomisto 2004).

On the level of knowledge acquisition processes, it has also been noted that an emphasis on precaution may alter established practices in production and use of scientific information, even eroding well-tried modes of inference (Tuomisto 2004, Weed 2004a,b, see also Weed 2002), and may create disincentives for research (Grandjean 2004 and unpublished 2003). On the other hand, precautionary principle does not exclude the use of scientific evidence.

6.1.3 Policy levels and actors

There are several global and regional policies and efforts in managing hazardous substances, many of which include dioxins. Within regional policies and actors or regimes, several levels need to be distinguished in the present work, from EU and other supra-regional regimes such as POP Convention regions, to sub-regional e.g. Nordic or Baltic Sea. The global and regional policies interact with national level policies; the latter typically implement and specify the general global and regional policies, but often also conversely influence them e.g. when national and sub-regional such as Nordic activity initiates global and supra-regional such as EU policies.

Global initiatives include a range of scientific and technical programs and policymaking under different inter-governmental organizations, e.g. in UNEP, WHO, FAO, IMO and ICES, and in inter-organizational international programs, e.g. IPCS.

Among regional forums and regimes, Helsinki Commission (HELCOM) for the protection of the Baltic Sea has a specific role, as it involves all Baltic seaboard countries including the Russian Federation (and some observer countries that occupy land in Baltic Sea catchment area), and is engaged in close official and expert cooperation in many areas including hazardous substances (e.g. HELCOM 2002b), also within land-based activities, and within integration of marine protection with fisheries. The role of HELCOM and other regional marine protection organizations is undergoing reform in connection with developing EU mandates and initiatives in this area, especially in the Baltic Sea due to EU enlargement.

Other regional forums that deal with hazardous substances in the marine environment include the corresponding Convention for the Protection of the Marine Environment of the North-East Atlantic ("OSPAR Convention"), the Convention on Long-Range Transboundary Air Pollution (LRTAP) managed by UNECE, and the EU (Selin and VanDeveer 2002, cf. below).

The policy initiatives discussed in this chapter emphasize the management of dioxins as a global environmental problem or the management activities directly affecting the Baltic Sea (Table 29). Instruments outside this analysis include e.g. the EU Marine Strategy as well as the EU Chemicals Policy and the related regulatory system (REACH), both in preparation. It is thus not clear what their implications for dioxin management will be; they are referred to only in general terms, and discussed in chapter 8.

In addition, also some other existing policies and instruments in EU, in other international domains, and nationally are relevant for the management of dioxins even in Baltic Sea fish. Some are specific and dioxin-focused while others have a more general scope; also the time frames differ. Even some strategies that extend outside the jurisdictions in charge of Baltic Sea matters may be relevant, such as the global strategies for controlling waste transfers. Moreover, some strategies wholly outside the Baltic Sea jurisdictions such as the recent US food dioxin strategy are important as procedural examples that have in some respects covered insufficiencies in EU strategies and related assessments (e.g. accounting more explicitly for uncertainties and analyzing management goals and means).

Specifically in the food area, the Codex Alimentarius sets out a code of good practice also for animal feeding to avoid food contamination, e.g. covering the sourcing, storage, production and delivery and traceability of feeding-stuffs (see e.g. Sijtsma and Doring 2002). These authors state e.g. that feed materials from increased risk for dioxin contamination, such as fish oil (also from the North Sea) are used by the food industry only after chemical analysis.

6.2 International policies on POPs at the global level

At the international level there are two legally binding instruments, the Protocol to the regional UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP) on POPs[1] in 1998 and the multilateral environmental treaty on POPs (The Stockholm Convention)[2] in 2001. The latter entered into force on 17 May 2004. These instruments include a list of the 16 (LRTAP) and the 12 (Stockholm Convention) prioritized POPs. Both instruments contain regular reviews and other procedures for including additional chemicals, and include provisions for the reduction of unintentionally produced POPs.

UNECE Protocol obliges Parties to reduce their emissions of dioxins and furans below their levels in 1990 (or an alternative year between 1985 and 1995). It also establishes specific limit values for the incineration of municipal, medical and hazardous waste, all of which have been important sources of PCDD/Fs, and still are in many present incineration processes. UNECE Protocol takes into account the application of the precautionary approach, as it is described in principle 15 of the Rio Declaration on Environment and Development[3].

The Stockholm Convention identifies sources and reduces releases of unintentionally produced POPs with continuous minimization. The objective is ultimate elimination where feasible. The main tool is the National Action Plans, which should cover source inventories and release estimates as well as plans for release reductions. The most severe control with regard to by-products is to promote and require the use of best available techniques for new sources within their major source categories. Stockpiles management e.g. for PCBs and pesticides is also an important part of the Convention both in identification, monitoring and assessment and in management including capacity building. This will include some waste management operations, but additional measures may be needed in this area e.g. for control of emissions of dioxin-type POPs not included in stockpiles

[1] *http://www.unece.org/env/lrtap*

[2] *http://www.pops.int*

[3] *In 1992 United Nations Conference on Environment and Development (UNCED), countries agreed by consensus to the Rio Declaration on Environment and Development which binds them to implementing the precautionary principle to protect the environment. This approach states that "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as reason for postponing cost-effective measures to prevent environmental degradation" (Principle 15). During the 1990s, the precautionary principle has become widely recognized and reflected in many international policy statements and legally binding agreements. However, it has usually not been specified.*

Table 29. Policies and instruments relevant to the management of PCDD/Fs and PCBs in Baltic Sea fish, including their sources.

Policy statement or regime/forum	Year	geogr scale	Legislative status and other significance
the Protocol to the regional UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP) on POPs[4],	1998	Global	Legally binding
The Stockholm Convention	2001 (in force 2004)	Global (+ regional, national)	Legally binding
EC on POPs 1) Commission Proposal for a Regulation of the European Parliament and of the Council on persistent organic pollutants; 2) Commission Proposal for a Council Decision concerning the conclusion, on behalf of the European Community, of the Stockholm Convention on Persistent Organic Pollutants; 3) Commission Proposal for a Council Decision concerning the conclusion, on behalf of the European Community, of the 1998 Protocol to the 1979 Convention on Long Range Transboundary Air Pollution on Persistent Organic Pollutants		EU	Legal proposals
the Communication on a Community Strategy for Dioxins, Furans and Polychlorinated Biphenyls (COM(2001) 593 final)	2001	EU	Includes legally binding limit and action values
EU Interim strategy for PBTs and VPVBs	2001	EU	None (as of now)
The European environment and health strategy (SCALE) of DG-SANCO, DG-ENV and DG-RES (EC 2003, 2004c)	2003	EU	No legal status; includes a pilot project on dioxins (monitoring) in the Baltic Sea region, linked to management strategy development
EU strategy for endocrine disrupters	1999	EU	Some proposed e.g. on labelling but mainly non-binding; addresses many DLCs
HELCOM's strategies for the general protection of the Baltic Sea	several	Baltic Sea	No legal status; recommendations unanimously accepted by signatories, incl. land-based and monitoring strategies
The HELCOM objective with regard to hazardous substances	1998	Baltic region	No legal status
EU Common Fisheries Policy	2003	EU	Includes legally binding instruments
ICES/IBSFC fisheries strategies for the various stocks	several	Baltic Sea	Binding for catches
Nordic Council of Minister's Nordic product-oriented environmental strategy	2001 (1998)	Nordic states / NCM	No legal status
Nordic strategy on sustainable development	2001	Nordic states / NCM	No legal status
Arctic environment protection strategy; Action Plan for elimination of pollution from the Arctic (1999)	several since 1991	Arctic Council (some BS states)	No legal staatus; includes project on phase-out of PCB use and management of PCB-contaminated wastes in Russian Federation

Explanations: VPVB=very persistent-very bioaccumulative

(expect for dioxin-like PCBs and some other DLCs). In addition, risk reduction by preventing formation of by-products such as dioxins e.g. by process design, selection of raw materials and use of non-precursor substances and even by control of some activities giving rise to formation are included (see below) and could be an important category of options to be utilized and further developed also for dioxins.

The Convention includes several general obligations to promote and facilitate public information, awareness and education measures and monitoring of POPs. The specific reference to information exchange in the Convention was supported by the realization that research, testing, monitoring and demonstration data was being generated and numerous publications, data sets and databases were produced but national agencies responsible for management decisions were unable to access them. An important part of these materials seemed to disappear in a black hole, never becoming available to those requiring them at the time when they are needed (Keita-Ouane 2003).

Several other international agreements also call for the establishment of monitoring activities to verify the effective implementation of the conventions and the decrease of environmental levels of pollutants. Some monitoring activities are already in place but, as different methodologies are used, the data is often difficult to compare. UNEP Chemicals has, therefore, launched the project "Global Network for Monitoring of Chemicals in the Environment"[5], to create an electronic forum and working group on the harmonization of methodologies and analyses

[4] *http://www.unece.org/env/lrtap*
[5] *http://www.chem.unep.ch/gmn/default.htm*

of chemicals in the environment (Keita-Ouane 2003). This network operates on several regions, of which one includes The Baltic Sea and its catchment along with much of Eastern Europe and Black Sea catchment.

The Stockholm Convention includes four explicit references to precaution, but according to Willis (2002) the spirit of precaution flows through the treaty. Considering the control measures of unintentionally produced POPs it is stated that "in determining best available technique (BAT), special consideration should be given, generally or in specific cases, to the following factors, bearing in mind the likely costs and benefits of a measure and consideration of precaution and prevention…" (Article 5, Annex C). The spirit of precaution is to be also found in a "pollution prevention hierarchy" established for by-products e.g. promoting the development and requiring the use of substitute or modified materials, products and processes to prevent the formation and release of the chemicals listed in Annex C of the Convention.

6.3 European Community policies on dioxins, furans and PCBs and in other relevant areas

6.3.1 European chemical policies

European Union (EU) has activities in different regimes that are either directly related to dioxins or dioxins are a part of a larger regime. The EU has established a number of specific regulations on dioxins, e.g. industrial safety requirements and maximum permissible concentrations for air emissions (cf. Annex 11). This regulation is relevant for the policy work on POPs. Another regime deals more specifically with dioxins and human exposures. Notably, in 2001 the EC adopted a Communication on a Community Strategy for Dioxins, Furans and Polychlorinated Biphenyls (EC 2001; COM(2001)593 final). The strategy attempts to adopt an integrated approach aiming to assess the current state of environment and the ecosystem, and in the short-term to reduce human exposure to dioxins and PCB's and in the long term to maintain human exposure at safe levels (for more detail, cf. below and Annex 11).

With regard to the environment, there are two general types of legislation in the EU (Geiser and Tickner 2003):

- Legislation regulated by Article 95 of the Maastricht Treaty, covering "things that move." This is the internal market regulation – in this case, countries can only go beyond EU regulation if they can demonstrate the need. To achieve derogations (exemptions) from European-wide policy, countries have to prove local conditions that warrant extra protection, such as a sensitive aquifer.
- Legislation regulated by Article 175 of the Maastricht Treaty covering environmental protection. While this article includes all environmental protection policies, it generally covers things that are fixed, such as production facilities. In this case, countries can go beyond EU law – i.e., in banning emissions of a chemical, permitting, etc – but must respect the internal market.

Chemicals policy generally falls under Article 95 since chemicals are marketed and traded among Member States and the new chemicals legislation will probably be developed under this article. Also the dioxin regulation setting the maximum limit aiming at human exposure reduction (Directive 1999/29/EC) falls under Article 95, while the release control measures (e.g. the IPPC Directive 96/61/EC) that cover also dioxin sources are under Article 175.

In general, there is a major change in chemicals regulation at the European level. Centralization of a decentralized chemicals management system seems to increase (Geiser and Tickner 2003). Yet, according to Neyer (2000), EU's regulatory philosophy is neither hierarchical enforcement nor centralized decision-making. Instead, e.g. scientific expertise is to be used to legitimate regulation (Joerges 1996). Jordan and Jeppesen (2000) claim that the impact of the subsidiarity principle (which in general means that a larger organization or one on higher level of governance should not exercise functions which can be carried out efficiently or adequately by a smaller one or one on lower level of governance, e.g. EU and its member states) upon the process of European political integration is open to question. In the detailed aspects of environmental policy development, only the more insignificant aspects have been returned to the member state level, whereas e.g. health as a more important question has remained at the EU level.

A focus of chemicals regulation in the EU and even more so in US has been on carcinogenic substances, but this is a process with variation and now in transition. Other kinds of chronic effects have been also traditionally in focus, such as reproductive disorders in the context of 'CMR' substances (that are carcinogenic, mutagenic or reproductive toxicants).

Endocrine disrupting chemicals are a more recent critical topic in risk assessment and management. The EC adopted a strategy on endocrine disrupters in December 1999. It addresses the identification, testing, assessment and prioritization of potential endocrine disrupters, and also some areas of risk management, information and communication, distinguishing between near-term actions (based on existing regulations) and long-term actions. In the framework of the European Environment and Health Strategy, endocrine disrupting effects are included in the first cycle (2004–2010) of the strategy. The aim is to establish an understanding of the link between environmental factors and endocrine disrupting effects. In a review of the existing knowledge on health effects of dioxins and PCBs (EC 2004b), it was concluded that they still are a hotly debated area, with developmental effects in children as the most relevant health effect.

Bar et al. (2000) claim that European environmental policy stands out for its flexibility that is due to the fact that it has always had to consider different requirements in terms of nature and substance. Its flexibility is also due to the separately evolved legislations, administrations, economies, societies and cultures that have not yet become very integrated, especially in the dynamic field of environmental protection. This necessity of flexible responses to different conditions is met by the establishment of permanent exemptions or temporary transition periods within the EC Treaty or in individual secondary legislation (directives or regulations). The newer European policies also lean more to a risk adverse and precautious approach (e.g. ban on beef hormones and the resistance to genetically modified organisms, Geiser and Tickner 2003). The precautionary principle is mentioned in the environmental chapter of the EC Treaty, together with the principle of prevention (Article 174).

6.3.2 European policies on POPs

The EU policy on POPs is to considerable part overlapping with those on chemicals in general. However, it also has independent areas and is of great importance as a general framework for the management of dioxins and PCBs, being important classes of POPs. Therefore, policies and strategies for POPs will be specifically dealt with in the present work.

In EU (as in US and elsewhere), policies on POPs have also included an emphasis on PBT substances that has in some respects other characteristics than POPs policies (e.g. through the greater emphasis on toxicity in the former case). The European Community has signed both of the above international instruments on POPs, together with all Member States. The Community has taken activities in order to ratify and implement these agreements (EC 2003b1, b2). Therefore the EC has adopted three legislative proposals[6]:

1) Commission Proposal for a Regulation of the European Parliament and of the Council on persistent organic pollutants;
2) Commission Proposal for a Council Decision concerning the conclusion, on behalf of the European Community, of the Stockholm Convention on Persistent Organic Pollutants;
3) Commission Proposal for a Council Decision concerning the conclusion, on behalf of the European Community, of the 1998 Protocol to the 1979 Convention on Long Range Transboundary Air Pollution on Persistent Organic Pollutants

The Community legislation includes many instruments relevant to unintentionally produced POPs. The main release control measures are defined in Directive 96/61/EC (the IPPC Directive) which covers the major stationary sources (industrial sources, energy and waste management) of by-product POPs.

The EPER, a Community-wide inventory of the principal emissions and respective sources, was established by Commission Decision 2000/479/E and it covers all unintentionally produced POPs, except PCBs. The list of pollutants to be registered will be expanded due to other international obligations, and also PCBs will be included.

[6] *http://europa.eu.int/comm/environment/pops/index_en.htm*

The Waste incineration Directive (2000/76/EC) deals with an important source of by-product POPs. In addition, the Directive on Large Combustion Plants (Directive 2001/80/EC) is relevant for POPs emissions. The Council Directive 96/82/EC (the so-called Seveso II Directive) on the control of major-accident hazards is applicable on the prevention of accidental releases of dangerous substances.

Although there is Community legislation on release control measures, there are no emission reduction targets at Community level, and the current release inventories do not cover all sources of POPs (EC 2003)[7].

6.3.3 European policies on health and food safety

General features

The European Commission published the White paper on food safety (COM (1999)719) (EC 2000a). As the paper made public, under intense pressure from consumer concerns due largely to food dioxin scandals, plans for "a huge shake-up in the way food safety issues are treated in the EU" (ENDS Daily 12.1.2000), it also revealed planned measures to include laws to set maximum levels of dioxins and PCBs in foods and feed-stuffs. The Commission's guiding principle is to apply an integrated approach covering all sectors and stages of the food chain, including e.g. feed production, food processing, storage, and distribution. The precautionary principle is included in food safety, stating "where appropriate, the precautionary principle will be applied in risk management decisions" (EC 2000a).

In 2001 the Communication on a Community Strategy for Dioxins, Furans and Polychlorinated Biphenyls (COM(2001)593 final) (EC 2001) was adopted with the overall objective to reduce human intake levels below the levels recommended by the SCF (2000). This quantitative objective is 14 picograms WHO-TEq per kg bodyweight per week (2 pg per day). The strategy includes also the objectives of reducing environmental effects and finding out more about dioxins and PCBs and the risks they pose.

The Community strategy for dioxins and PCBs consists of three elements. The first is the legislative action and limit values that entered into force in 1 July 2002. The two others are actions identified during 5- and 10-year periods. The short-to-medium-term (5 years) actions cover hazard identification, risk assessment, risk management, research, communication to the public and co-operation with third countries and international organizations. The long-term actions (10 years) cover data collection, monitoring and surveillance. The intent is to present a comprehensive picture of the environmental dioxin/PCB problem and an understanding of existing trends, which will permit further policy making and evaluation.

Action and limit values

The main elements in the strategy are the legal maximum levels in food and feeding-stuffs, supplemented by "action levels" to provide early warning of higher than desirable levels and "target levels" to be achieved over time (Annex 11). These targets must be achieved in order to reduce exposure levels, to which a large part of the European population is subjected, to lower than the tolerable intake levels established by the SCF (2001). The aim is to provide additional protection to European consumers from the long-term effects of dioxin consumption.

Council Directive 1999/29/EC on the undesirable substances and products in animal nutrition merges the numerous amendments to Council Directive 74/63/EEC on the undesirable substances in animal nutrition. This Directive includes maximum limits for heavy metals such as arsenic, lead, mercury and cadmium, as well as for dioxin, aflatoxin, certain pesticides, and botanical impurities in certain feed materials and feeding-stuffs.

In response to dioxin contamination, the Commission presented a proposal for a European Parliament and Council Directive in 17 Dec 1999 to replace Directive 1999/29/EC. Following the comments of the Parliament, an amended proposal was presented by the Commission on 19 Dec 2000. A Common Position on this Proposal was politically agreed in the Council on 19 June 2001 and formally adopted on 17 Sep 2001. It foresees the prohibition of dilution of contaminated feed materials with other feed materials, adding rules for maximum limits of undesirable substances in feed additives, deleting any possibility of derogation to the provisions of

[7] *Proposal for a regulation of the European Parliament and of the Council on the persistent organic pollutants and amending Directives 79/117/EEC and 96/59/EC, Brussels 12.6.2003 COM (2003) 333 final*

the Directive and introducing thresholds on the presence of undesirable substances that trigger intervention by the competent authorities if exceeded (cf. below).

The proposed amendment of Directive 1999/29/EC (CD 1999/29/EC) on the undesirable substances and products in animal nutrition proposes dioxin limit values. A concurrent Commission Recommendation will contain corresponding action values. The limit and action values for dioxins in feeding-stuff are introduced (Annex 11).

The three components of the proposed legislative measures are (e.g., Joas et al. 2001, 2002):

1) The establishment of maximum limits (=limit values): The proposed maximum limit means that the product such as fish oil or fish meal, with a contamination level above the corresponding maximum limit, will not be used for a production of feeding-stuff (e.g. fish oil with a contamination level above 6 ng kg^{-1} or fish meal with a contamination level above 1,25 ng kg^{-1} whole weight). The proposal is restricted to dioxins because on the basis of the current data it seems inappropriate to include dioxin-like PCBs. It has been planned that the maximum limits will be reviewed in the light of new data particularly to include dioxin-like PCBs in the levels to be set. A further review is planned before the end of 2006 to significantly reduce the maximum levels.
2) The establishment of action values: The action values are a tool of early warning for higher than desirable levels and trigger a proactive approach from competent actors to identify sources and pathways of contamination and to take measures to reduce the contamination. The exceeding of action values will not have direct consequences on the marketing or use of the feeding-stuff concerned as long as maximum limits are followed.
3) The establishment of target levels: Target levels would be levels to be achieved over time in order to bring human exposure to dioxins and dioxin-like PCBs below the recommended tolerable weekly intake (TWI) (see above). The current proposal does not contain target levels.

5-year-activities

A key goal for action within five years on the basis of the Community dioxin strategy is the completion of a series of dioxin and PCB emission inventories. Some industry-related measures are considered as well. The inventories are focused on emissions to land and water, but also some emissions to air are reviewed. Some of this is done in connection with the work based on the Stockholm and LRTAP Conventions e.g. at the MSC-E, some in specific tasks such as in the pilot project under SCALE (European environment and health strategy) on dioxins e.g. in the Baltic Sea region, or the further EU actions on dioxin emission inventories and management (see below).

Certain industrial air emissions will be targeted for further monitoring, with the possibility of limits being introduced within the framework of the EU's integrated pollution prevention and control directive. These include emissions from hospital waste incinerators, iron ore sintering, electric arc furnaces as well as zinc recovery processes in the non-ferrous metals industry. Other possible industry-related measures include introduction of subsidies to curb illegal dumping of equipment containing PCBs and limits for dioxins and PCB's in sewage sludge.

Development of environmental indicators and bio-indicators is planned, as is research into dioxin-like PCBs, which are now suspected of posing a much larger threat to human health than previously thought. Non-industrial sources will come under the spotlight, with emphasis on emissions from domestic solid fuel combustion. A public education campaign about dioxin emissions from backyard burning is also on the cards (ENDS Daily 25.10.2001).

Accordingly, there have been several studies and other preparatory actions financed by the European Commission (Annex 11). These have included The European dioxin emission inventory, Stage II (LUA-NRW, see Quass et al. 2004a) that identified the need for further investigation or actions on specific sources. According to this inventory, the Commission should therefore take action in the following sectors:

- Hospital waste incinerators
- Iron ore sintering
- Electric arc furnaces
- Non-ferrous metal industry
- Miscellaneous sources such as secondary smelter for non-ferrous metals (aluminium,

copper), iron foundries (cupola furnaces), cement production
- Non-industrial emission sources concerning the domestic solid fuel combustion, domestic waste burning, burning of animal carcasses.

Additional work funded by the EU has been ongoing on inventories of data on environmental emissions and levels of dioxins in the candidate (new and accession) countries (Anon. 2004, cf. Annex 11). Also this work has focused on monitoring, and the recommendations produced have mainly dealt with the coverage and quality of measuring environmental levels and exposures. However, some information, evaluations and proposals have also been provided in the area of emissions, being more directly linked with management.

In the EU dioxin strategy little detailed suggestions or concrete policy principles have been presented in the key area of dioxin risk management, prevention of formation (cf. Annex 11); also some of the formulations in the emission control area are vague. No sea-based management options have been mentioned. Most attention in post-sea management has been given to management by means of fish and feeding-stuff concentration levels. Additional evaluation of the EU strategy will be provided below (8).

European strategy for Environment and Health

Also the new Community strategy for Environment and Health (COM(2003)338 final) addresses dioxins and PCBs. "A European Environment and Health Strategy" was launched by the European Commission in June 2003 (EC 2003c). The overall aim of the strategy is to reduce the disease burden caused by environmental factors in Europe. One of the pilot projects is on integrated dioxin and PCB monitoring in the Baltic Region[8]. The project aims to develop a methodology for integrated environment and health monitoring and response.

The synthesis of baseline reports (EC 2004a) presented an analysis of integrated environment and health monitoring of dioxins and PCBs in the Baltic Region. The most important problem is considered to be the general lack of communication, coordination and cooperation between environmental, food and health researchers and authorities. Suitable international and national mechanisms need to be established in order that a unified approach can be taken. The objective of integrated monitoring in not only to express risk. Monitoring includes much of the official control systems. Integrated monitoring should also translate to integrated risk management leading to better tracing and linking, targeted risk management, and quicker problem solving.

6.3.4 European fisheries and marine policies

Common fisheries policy

The Common Fisheries Policy (CFP) of the EU is in principle an important sector policy also from the perspective of managing Baltic Sea dioxins (EC 2002b). It may be seen as one of the policies where a stronger EU coordination will develop for the Baltic Sea also due to the inclusion of new member states on the seaboard. The CFP emphasizes structural changes in fisheries, including development of fleet structure. Within environmental conservation and natural resource use areas, mainly the sustainable productivity of the fish stocks is emphasized.

The importance of the CFP in the present management case is diminished or made less clear by the initial stage of its development and implementation, and by the lack of explicit considerations for bioaccumulating contaminants in fish as a contributing and even driving factor in fisheries management. The CFP is however likely to have increasing importance also indirectly through overall development of EU fisheries.

The Commission proposes in its communication on the reform of the CFP (EC 2002b) a multi-annual framework including activities that are implicitly relevant for the Baltic Sea dioxin problem. These include e.g. protection of non-commercial fish species, marine mammals and sea-birds, and 'discard ban trials' by which some fishing vessels would be motivated to retain their entire catch by economic incentives.

The CFP faces many great challenges also in more traditional areas, including the resolution of national interests and conflicts under the pressures of diminishing stocks and declining economies, a broader participation of stakeholders, and the structural changes in fishing fleets. It will not be an easy task to include dioxins as an additional boundary condition for the dynamically

[8] *http://europa.eu.int/comm/environment/health/index_en.htm*

developing fisheries management, particularly in the Baltic involving many new member states.

The coordination of the CFP with other international fisheries policies and procedures will be important but contribute to the lags involved in its development. In particular, coordination with BSMFC (Baltic Sea Marine Fisheries Commision) under ICES (International Commission for the Exploration of the Seas) will be crucial. This has some parallels and resemblance with the tasks of coordinating HELCOM and other Baltic Sea based structures and activities with EU policies and programs. Also coordination with national fisheries policies needs to be developed within EU. This seems a challenging tasks, given the pressures on fish stocks e.g. by competing fisheries, and the disagreements already within EU and BSMFC on sustainable TACs.

Application of the precautionary principle in CFP is mainly associated with the increasing emphasis on sustainability of stocks and on consideration of uncertainties in their states and in predictions and analyses (see Degnbol et al. 2003).

Marine policies

Marine policy in the EU is in part established based also on many legally binding international conventions, but includes areas that are still under preparation (see Annex 11). In particular, an ecosystem approach based marine resource management is still not extensively included in the maritime instruments. Also Coastal Zone Management policies and instruments are still under development.

The ecosystem approach in marine policy is of particular importance regarding Baltic Sea dioxins, as a holistic human ecological approach is adopted in this connection, and as operationalizations of the precautionary principle, including considerations of uncertainty management, are important dimensions in the development process.

At a global level, policies for the protection as well as for the use of the marine environment and resources are developed and implemented within UN organizations mainly under ICES, focusing on fisheries management and associated issues in marine ecology and environment; FAO, likewise engaged in fisheries and aquaculture matters; UNEP, with a strong emphasis on marine protection; and IMO, focusing traditionally on marine trade but increasingly covering environmental and resource related issues. Among the latter, the definition of Particularly Sensitive Sea Areas (PSSA) deserves mention.

6.4 Baltic Sea cooperation on dioxin-like compounds and related substances, and other regional policies

One of the most important means in protecting the Baltic has been the Convention on the Protection of the Marine Environment of the Baltic Sea Area (the Helsinki Convention), agreed upon and set up in 1970, concluded in 1992 and entered into force in 2000. The governing body of the Convention is the Helsinki Commission – Baltic Marine Environment Commission – also known as HELCOM.

Some HELCOM recommendations for the management of hazardous substances and specifically on dioxins were established in early 1980's (Recomm. 13/4 & 16/8). The first major HELCOM objective on hazardous substances set in 1988 Ministerial Declaration on the Protection of the Marine Environment of the Baltic Sea Area with intention of 50 % reduction of the total discharge in 1987. To explain this goal a list of 47 priority substances – including dioxins - were identified for this reduction goal and the list was supplemented with a list for national programs to achieve the goal.

As this general 50 % goal was not reached (HELCOM 1998b), more detailed targets were specified in 1998 and HELCOM Recommendation 19/5[9] was adopted. Its aim is to prevent pollution of the Convention Area by continuously reducing discharges, emissions and losses of hazardous substances towards the target of their cessation by the year 2020, with the ultimate aim of achieving concentrations in the marine environment near background values for naturally occurring substances and close to zero for man-made synthetic substances. A Project Team on Hazardous Substances was created under the HELCOM Land-based Pollution Group in 1998 with the purpose of acting towards the implementation of Recommendation 19/5. The Project Team consists of members from all Parties and representatives from three NGO's: the European Chemical Industry

[9] http://www.helcom.fi/recommendations/rec19_5.html

Council (CEFIC), the European ChlorAlkali Industry (EuroChlor), and the World Wide Fund for Nature (WWF).

The guiding principles of the strategy are the application of a) the precautionary principle, b) the polluter pays principle, and c) best available technology and best environmental practice. Parties to Baltic cooperation may, for example, deem a substance to be hazardous even if it does not fully meet all the criteria for toxicity, persistence and bioaccumulation if there are other grounds for concern, such as suggestions of endocrine disruptive functions or damage to immune systems (Selin and VanDeveer 2002).

Recommendation 19/5 lists in Appendix 2 some 280 hazardous substances as potential substances of concern to be considered by HELCOM. A further list of hazardous substances selected for first action was produced. Dioxins were added to the current list of 36 substances in 2001. Dioxins are the only by-product substance group in the list.

HELCOM has prepared a guidance document to support policy makers in their selection and application of appropriate instruments to achieve cessation of emissions, losses and discharges of dioxins. The detailed proposals are following (HELCOM 2002b):

- The knowledge concerning dioxin/furan sources, emissions and concentrations in the environment has to be improved. Especially country specific information about potential point sources of halogenated hydrocarbons should be searched for in co-operation.
- Activities related to local and regional risk assessments and risk reduction measures should be planned, drafted and even to some extent decided without unnecessary delay. The HELCOM strategy for risk management and preliminary decisions on Recommendations for risk reduction measures may be specified later when the improved research data is made available.
- Measures aiming at a broad reduction of emissions of halogenated hydrocarbons from waste treatment and from combustion processes in industry should be promoted by HELCOM using its available means.
- The measures necessitated by EU legislation and obligations or recommendations, foreseen as follow-up actions according to the Community Strategy for Dioxins, Furans and PCB's, should be taken into account when formulating HELCOM Recommendations concerning the legislation or practice in EU applicant countries as soon as possible. Russian Federation should be recommended to apply the same standards for risk reduction.
- Other new international conventions of relevance to dioxins and furans (UNECE POP, Stockholm Convention) should be ratified by the HELCOM countries without delay.
- Pollution load compilations (PLCs) by HELCOM should in future include data on the amounts of persistent organic compounds including dioxins and furans.

In addition, the following HELCOM activities or initiatives can be mentioned:

- Inventories of PCBs stockpiles and emissions and recommendations of controls (cf. Annex 3)
- Evaluation and control of land-based activities
- The list of hotspots under the Joint Comprehensive Program (JCP) (cf. Annex 3)
- The strategy on monitoring and assessment (MONAS).

There are also other regional policies and instruments of relevance for Baltic Sea dioxins risk management. Some of these may be listed and described in short:

- Nordic policies in other related areas, such as health, food and fisheries, in addition to environmental protection. A key actor as an official multilateral inter-governmental body is the Nordic Council of Ministers (NCM). It promotes coordination both between the Nordic countries (including Finland, Sweden and Denmark of Baltic Sea countries and Norway that also has ratified the Helsinki Convention) and between the various sectors. In practical cooperation, the various standing Working Groups of NMC (including those supporting the present assessment, particularly the Chemicals Groups) play an important role. In addition, inter-sector coordination has been emphasized, e.g. in connection with sustainability initiatives and more general development of Nordic cooperation, and alignment with the work of the enlargened EU.
- Regional policies and instruments of EU, e.g. in connection with cooperation programmes for accession countries in Central and

Eastern Europe and former Soviet republics (including Russian Federation), and with the strategic initiatives in the Northern dimension. In relation to the latter, also the individual Nordic countries in the Baltic Sea area have developed regional international cooperation for protection and utilization of the Baltic, in addition to national policies, strategies and programmes for the Baltic Sea (see next section)

- OSPAR Convention and its commission (OSPARCOM), as a 'sister' organization of HELCOM for the protection of the North-East Atlantic including the North Sea
- Regional instruments of global policies and instruments, including e.g. the LRTAP Convention under UNECE (cf. the developing regional activities and bodies under the POPs Convention, below).

6.5 National policies on dioxins and related substances

The Nordic States (Denmark, Norway and Sweden, see Geiser and Tickner 2003) have used a variety of voluntary and mandatory policy tools – such as education, procurement, lists of chemicals of concern, eco-labelling, research and development on safer substitutes, and chemical phase-out requirements - to encourage companies using chemicals to reduce their reliance on harmful substances and to develop safer substitutes.

All of the Nordic policies have a major focus on product-based risks, as officials in these countries believe that products – not simply point sources – are important pollutant sources. Government officials in each of these countries also believe they must develop policies to stimulate industrial innovation in safer technologies and products (though the extent to which government works with industry differs among countries).

The chemical management policies in each of these three countries are more developed (using many policy tools and widely implemented) and restrictive (strongly regulatory and focused on restricting problem chemicals) than the more general policies of the broader EU (Geiser and Tickner 2003). It may be pointed out however that there is variation in the relative status of development and strictness of chemicals management between the Nordic states and the rest of EU.

Sweden has traditionally been at the forefront of POPs management. Several initiatives have been made and actions taken at a national plane, e.g. in connection with the recent work on environmental goals including total phase-out of PBTs by 2020. This is related to the so-called generation goal, one among the environmental goals (e.g., Rikskansliet 2000) and internationally to the Aarhus process and the Stockholm Convention. Sweden has also often proactively restricted chemical products that are feared to give rise to DLCs. This policy has a long history, from bans and restrictions of chlorinated pesticides and PCBs over chlorophenol use restrictions to the more recent attempts to limit brominated flame retardants (Table 30). No specific action plan on dioxins has yet been produced however. In managing dioxins in Baltic fish, Sweden like Finland has adopted a deviant policy with respect to the EU recommendations, referring e.g. to the benefits of fish and fisheries (see 1.2 and below).

Some of the mechanisms being used by the Danish government to decrease the use of undesirable chemicals include action plans on problematic classes of chemicals. For chemicals found to pose particular risks the government prepares action plans outlining problems with the chemical, goals and steps for reducing hazards, and costs of implementation. An action plan has been developed by the Ministry of the Environment and the Ministry for Food, Agriculture and Fisheries on reducing dioxin emissions.

The white paper on environmental policy (Anon. 1996-97) stated the ambitious goals of Norwegian chemicals policy. These include an elimination or reduction of releases, by certain deadlines, of dangerous substances on a national priority list. The list includes emissions to be substantially reduced by 2010, including dioxins and furans. An action plan on hazardous substances was subsequently laid out (MoE 1999). A sharpening of the zero discharge goals for hazardous chemicals (an impossibility with PCDD/Fs and also some other inadvertently formed DLCs) was published in a later white paper (Anon. 2002-03). The white papers also proposed the institution of criteria on health and environmental hazards, ensuring that all of the most dangerous substances are encompassed by the national goals.

In 2002, the Finnish government adopted a national protection program for the Baltic Sea (Anon. 2002). One of the areas is reduction of

Table 30. The status of regulations and actions in the management of PCDD/Fs and PCBs in Baltic Sea countries (based mainly on UNEP 2003).

Country	Regulation for reduction of dioxin and furan risks	National Action Plans
Denmark	Tolerable Daily Intake (TDI) 5 pg I-TEq/kg bw. (Danish Guidelines)	Danish Action Plan on dioxins, objective is to obtain more knowledge about Danish emissions and to implement measures to minimise emissions (1999 – not specified)
Estonia	▪ Integrated Pollution Prevention and Control Act (Oct 10, 2001, based on EU IPPC Directive 96/61/EC), emission limit values in air and water ▪ Food control (based on Council Regulation (EC) No. 2375/2001) – Baltic Sea fish	-
Finland	▪ Regulatory control on major sources (plant permits) ▪ Waste containing dioxins or furans must be disposed of in such a way that waste does not exhibit the characteristics of persistent organic substances. If this is not possible or the PCDD/F content is low, the waste may be disposed of in some other environmentally sound manner (Government Decree on POPs 735/2002)	-
Germany	▪ Ban of production and use of substances known to be contaminated with PCDD/Fs; i.e. PCBs (18/07/1989) and PCP (12/12/1989) ▪ Ban of production and use of substances known to form PCDD/Fs; i.e. PCBs, halogenated scavengers in gasoline fuels (17/01/1992) ▪ Ordinance on Prohibition of Certain Chemicals under the Chemical Act, setting limits for PCDD/Fs in substances, preparations and articles on the market ▪ Off-gas concentration of waste incinerator plants and crematories must not exceed 0.1 ng I-TEq/m^3 ▪ The content of PCDD/Fs in sewage sludge to be used on agricultural land limited to 100 ng I-TEq/kg d.w. ▪ Ordinance on the Prohibition of Certain Chemicals (14/10/1993) - Ordinance on Ban of Halogenated Scavengers (17/01/1992) ▪ Directive 2000/76/EC of 04.12.2000 on incineration of waste with a limit value of 0.1 ng TEq/m^3 ▪ Ordinance on the Incinerators for Waste and Similar Combustible Materials 23/11/1990) under the Federal Immission Control Act (14/05/1990) with a limit value of 0.1 ng TEq/m^3 for all industrial plants	Program to find out the consequences of the reassessment of PCDD/F dioxin-like PCBs for the requirements from industrial plants, objectives of emissions of dioxin-like PCBs from industrial plants (with programs for measurements) and review of requirements for their limitations, if needed (2002-2004)
Latvia	▪ The 30-min average air emission limit value for Furans is 0,01 ng TEq/m^3. Regulations of the Cabinet of Ministers No. 219 "on Air Quality" (15.9.1999) ▪ Furans and Dioxins total emission limit values for incineration plants 0.1 ng TEq/m^3; waste co-incineration plants in cement kilns 0.1 ng TEq/m^3; discharges of waste water from cleaning of exhaust gases from waste incineration plants 0.3 ng TEq/m^3. Regulations of the Cabinet of Ministers No. 323 On requirements for incineration of waste and for operation of waste incineration plants (17.7.2001)	-
Lithuania	▪ Basic technological requirements for waste incineration and limited values of pollutants in ambient air. ▪ Maximum permissible concentration of chemicals (including dioxins and furans) polluting air of residential areas ▪ Basic requirements for waste incineration and limited values of pollutants from wastes incineration plant.	Project to Assist the Republic of Lithuania to Transpose EU Requirements in the Water Sector Standards Project), e.g. assisting Lithuania in transposing Dangerous Substances Directives (76/464/EEC and daughter directives)
Poland	PCDD/Fs are subject to emission fees for air pollution, and emissions of them have to be based on inventories. There are emission limits for PCDD/Fsfrom incineration of municipal waste hazardous waste and waste fuel oils Some substances from this group were placed on the list of hazardous chemical substances (1996)	Construction of the installation for recovery of HCl from waste containing chloro-organic compounds to prevent generation of dioxins. The installation is to comply with EU standards of waste generation and emissions to water bodies and to the atmosphere. (1999 – completion of the installation construction)
Russian Federat.	No information on regulation	▪ Federal Target Program for "protection of the environment and population from dioxins and dioxin-like toxic substances. No info on status. With UNEP ▪ Evaluation of PCDD/F emissions with focus on Northern regions, to identify and quantify sources, quantify environmental releases and prioritize sources for reduction measures. The project includes training in sampling and analysis from Western to Russian laboratories.
Sweden	▪ Operating permits for waste incineration plants include emission limits. A step-wise reduction since mid 1980's. The emission to air from waste incineration plants is estimated to have been reduced from 50-100 to c. 1 g TEq/yr. Other industries have also substantially reduced their dioxin emissions to air and water. ▪ The National Food Administration has issued dietary recommendations on fish consumption based on the dioxin and PCB content in fish from certain areas.	

discharges of harmful substances including dioxin. The program points out the problems related to dioxin charges, but maintains that there is still a need for further research before efficient measures can be determined. The program emphasizes the need to enhance research and monitoring (Degnbol et al. 2003). In the POPs area, presently the development of a National Action Plan (NAP) is underway, as in most other Baltic Sea countries.

Chemicals statutes, policies and actions have been particularly prominent in Germany at both federal and state (Länder) level (Table 30). Also Germany has stressed the prevention of risks and the substitution of harmful chemicals (e.g., Stolzenberg 2000). The implementation has varied depending e.g. on the legislative, economic, technical and other conditions, also influenced by the strong position of the chemicals industry. Especially within dioxin management Germany has adopted a proactive position, in part due to some local cases of heavy dioxin contamination (e.g. in Hamburg), and has produced several detailed inventories, proposals, plans and criteria and other provisions for management.

When the European Community dioxin strategy was adapted both Finland and Sweden opposed the restrictions on fisheries. As a result, Sweden and Finland were allowed to delay full application of imminent EU limits on dioxins in food until 2006. In Sweden the political attitudes towards the derogation varied (DN 28.11.01). The other Baltic Sea states in EU, Germany and Denmark, accepted the limit values as such, although at least in Denmark limit values were criticized for being too stringent (Copenhagen Post Online News 13.8. 1999).

On the basis of high dioxin concentrations in fish, the Finnish and the Swedish national public health and food safety authorities have given recommendations on fish consumption. Because fish is healthy food, the authorities have not given recommended a reduction in overall consumption of fish. In order to reduce the dioxin intake it is recommended to consume different types of fish – fish from lakes, marine fish from outside of the Baltic Sea as well as Baltic Sea fish. It is recommended not to eat only Baltic Sea fish and when eating Baltic Sea fish it is better to eat young fish. Finnish recommendations have been less detailed than the recommendations in Sweden (Degnbol et al. 2003). Due to the derogations, the Finnish recommendations on fish consumption were reviewed and publicized in 2004.

Domestic sale and consumption of Baltic Sea fish will be allowed until at least 2006. Herring with higher dioxin concentration than the EU limit of 4 pg/g can be sold in Finland and Sweden as well as outside EU, but not exported to other EU countries. Under the agreement, all exports will be banned and both countries will be required to warn consumers of health risks and to report annually on dioxin concentrations. The deal also requires Sweden and Finland to abide by new rules on dioxin content in animal feed. These are less stringent.

The fish products are to be given a label indicating how much of the toxic dioxins they contain. An ellipse-shaped symbol on the side of the jar or can of fish is to be an indication that the dioxin content does not exceed the maximum limits set by the European Union. A rectangular symbol indicates that the level could be higher. The Baltic herring under 17 cm is included to those fish products allowed to use of the ellipse-shaped symbol. Closer records are also to be kept to keep better track of the fish sold in Finnish markets. The new rules on fish hygiene took effect on July 1, 2002, and the Ministry of Trade and Industry has drafted a report to European Commission on the tracking of the fish sold in Finland, and the enforcement of health standards. The system was required after Finland got special permission from the EU to use fish caught in the Baltic Sea, whose dioxin content exceeds the maximum levels set by the EU.

In their argumentation for derogation, both countries emphasized the cultural importance of certain species such as herring, salmon and whitefish roe in national dishes. They also argued that the north's dark winters meant citizens needed to consume the fish in order to maintain adequate levels of vitamin D. (ENDS Daily 26.11.2001). The countries are moreover concerned about protecting the livelihood of their fishermen.

Recently, the Commission has made a proposal that would allow continuing derogation from the Community recommendation concerning maximum levels of PCDD/Fs in fish, and marketing Baltic fish for human consumption on certain terms. This proposal is being treated in WTO (Rajakangas, Finnish Min Trade Ind, personal communication Jul 2005).

6.6 Interests of non-governmental organizations in the Baltic Sea fish and dioxin issue

Non-governmental organizations (NGOs) comprise a heterogeneous group ranging from environmental and consumer groups to organizations representing industry and workers. NGOs thus may have very different

ideas, motives, profiles, principles, agendas and activities. Although some basic characteristics can be discerned, e.g. according to the general interests of the NGO, they vary widely according to the issue and case. It cannot be stated simplistically that e.g. the environmental and consumer NGOs consistently form an opposing camp to those of industry in debates about DLCs. Within chemicals control such a traditional polarity can often be discerned, the environmental and consumer NGOs being on the average for stricter control of DLCs and potentially dioxin-forming chemicals, the NGOs affiliated with chemicals industry questioning risk reduction e.g. through extensive and rapid bans of halogen products. However, the positions of NGOs cannot be generalized.

In particular, in the case of food contamination by dioxins, food and feeding-stuff industry and the respective organizations have understandably been very concerned due to the dioxin scandals and in general to perceived consumer fears (cf. Lok and Powell 2000). Food industry has consequently actively promoted the improvement of quality assurance of the food production cycles and the development and implementation of the maximum levels for foods and feeding-stuffs (Sijtsma and Doring 2002). In contrast, the NGOs representing fisheries and particularly of fishermen e.g. in Finland and Sweden have been very concerned about the collapse of Baltic fisheries as a result of these maximum levels and other restrictions on fisheries, and have presented criticism against categorical implementation of such limits.

Environmental NGOs have been concerned with the dioxin issue in the Baltic Sea whereas the consumers' organizations have not taken as much part in the debate in these countries. In the following short discussion, the focus is on WWF as an influential environmental NGO especially within marine protection. It may however be noted that NGOs differ greatly in their orientation, characteristics and activities. For instance, consumer NGOs have different concerns for contaminants in fish than environmental NGOs.

On the international level, Greenpeace International and WWF have permanent programs on toxic chemicals. In connection with Stockholm Convention negotiations in 1999, Greenpeace campaigned delegates to agree to eliminate dioxins at their sources, rather than to concentrate on expensive schemes to reduce dioxin emissions into the environment. As a part of the campaign, Greenpeace published the report "POPs in the Baltic"[10].

A WWF-UK report on dioxins and dioxin-like PCBs in the EU (Lyons 1999) recommended Member States to set and adopt a EU-wide TDI, as well as Maximum Admissible Concentrations or at least guideline levels in specified foodstuffs for dioxins and dioxin-like PCB's; this corresponds largely to the management approach in the subsequent EU dioxin strategy. The report also recommended setting a food contamination monitoring system to ensure compliance, which also has been a prominent and extensive part of the EU strategy. These regulatory measures seek to impose a common standard of human health protection. Therefore it may be assumed that these international environmental NGOs would support the recent regulatory activities in EU. However, WWF-UK (2002) later published the complaint that its warnings had been suppressed in the UK and EU chemicals policy. Specifically, WWF-UK (2005) issued a warning that consuming fish from the Baltic may be unsafe.

Both of the above organizations have also expressed concerns on endocrine disruption by proposing the elimination of exposure to endocrine disrupters to protect wildlife and humans. This has been e.g. WWF's initial proposal for the new EU chemicals legislation.

In the study by Degnbol et al. (2003), WWF-Finland was interested in the state of the Baltic Sea. The high concentration of dioxin in fish was seen as an indicator of a polluted sea. According to WWF–Finland, the food safety question is important, but from the environmental point of view not as central as two other aspects of the state of the marine environment. First, how does dioxin as a bioaccumulating substance affect the marine ecosystem? Second, what are the impacts on herring populations? Therefore, the concentration of dioxin in herring was seen as relevant both for the conservation of other species, some of which may be endangered, and for the health of herring populations as such.

There is some variation among the national WWF organizations in the relative weight of DLCs in the concerns for fish and fisheries. WWF-Sweden (undated) has produced material stressing the value of fish as healthy food,

[10] *http://archive.greenpeace.org/toxics/reports/popsbaltic.pdf*

focusing on the conservation status of various fish species and on the environmental effects of fisheries instead of the contaminants they contain; contaminants were not mentioned as a purchase criterion even for herring or salmon in this consumer guide for fish. Such an orientation may be understood e.g. in relation to the traditional emphasis of WWF (and many other environmental NGO's) on environmental and species conservation. On the other hand, WWF-Sweden (2005) has also strongly emphasized the threat from toxic chemicals including DLCs to Baltic Sea fish and through it to humans and wildlife. This has been linked to international activities to promote a more precautionary chemicals policy e.g. within the REACH system (WWF 2005). Risks from contaminants are addressed by national organizations in the Baltic Sea countries in WWF's Baltic Ecoregion Action Programme along with other concerns such as eutrophication, fisheries and transport.

During its long engagement with dioxins and demands for elimination of emissions (see e.g. Allsopp et al. 1994), Greenpeace has campaigned also in the Baltic Sea area against chemical and other industries that have been perceived as particular sources of dioxins. Actions have taken place e.g. in Denmark where the blockade of the alleged largest single dioxin source, a metal industry, was ended by the police (Anon. 1999) and in Russian Federation where Greenpeace has taken the Petersburg water and sewage company Vodokanal to court because of the feared contamination from waste incineration (Kovalev 2004).

6.7 Implications of the policies and multi-forum activities

There is not a single cooperative action on Baltic Sea and hazardous substances but a number of policy activities affecting the Baltic dioxin problem (Table 30). The policy-making and management of dioxin risk is incorporated within a complex web of institutions, with different institutions focusing on different aspects of the dioxin risk issue. The countries around the Baltic Sea have adopted different regulations on dioxin, so the geographical coverage varies, as does the legislative status of different policies.

Most of the Baltic Sea countries are involved in the above central policy regimes and activities. They all are contracting parties to HELCOM, Russian Federation being the only contracting party outside EU. The Stockholm Convention has not been signed by Estonia and it is not ratified by European Community, Latvia, Lithuania, Poland and Russian Federation. The UNECE Protocol on POPs is not signed by Estonia and Russian Federation and it is not ratified by European Community, Latvia, Lithuania and Poland. Nevertheless, the persistence of dioxin compounds and the increasing importance of airborne emissions change also the relevance of geography for the Baltic Sea dioxin problem. As the earlier waterborne point sources are being controlled, the importance of the distant emission sources increases.

The problem of coherence (different rules, e.g. different standards caused by the implementation of precautionary principle) arises not only between states, but also between different interrelated regimes and institutions. At the global level, there are several institutions dealing with dioxins. It is crucial to pay attention to mechanisms and arrangements ensuring coherence between different regimes. Therefore, institutional and geographic clustering of the relevant institutions and multilateral environmental agreement should be an important element of the future dioxin risk management. According to Selin and VanDeveer (2002), HELCOM activities e.g. show regulatory overlaps with the Stockholm Convention.

HELCOM has concluded that as long as EU measures do not apply to all of the countries of the Baltic region, it is useful to have HELCOM actions on dioxins and furans. HELCOM activities should be compatible with EU actions. The most urgent need according to the HELCOM evaluation seems to be filling knowledge gaps concerning emissions and concentrations in the environment including biota (HELCOM 2002a).

With EU enlargement, it is likely that HELCOM overlaps with EU activities will be of particular importance (Selin and VanDeveer 2002). EU Directives are legally binding while the HELCOM Recommendations are not. This means that states that are members to both generally have a stronger incentive to implement EU Directives first. The non-binding status of HELCOM regulations has been stressed e.g. by Swedish environmental authorities as a limitation in protecting the sea. Although the HELCOM recommendations have been dubbed "soft law" and thus non-binding,

none of the contracting parties have refused to implement the recommendations due their non-binding nature. The recommendations are seen to carry a highly powerful moral and political significance (Andersson, unpublished 2002). HELCOM cooperation on hazardous substances exhibits many positive signs of high compliance of substantive obligation (Selin and VanDeveer 2002). Yet, as for instance some marine animals continue to exhibit disorders, suggesting that also hazardous substances including PCBs and dioxins are still causing problems (HELCOM, 2001b, 2002b) and as their levels have no longer clearly decreased, it seems fair to assume that the current level of HELCOM activities on dioxins is not sufficient to solve the Baltic dioxin problem.

If HELCOM Recommendations and EU Directives are more or less identical, the key added value of HELCOM after Estonia, Latvia, Lithuania and Poland become EU members may be that HELCOM also includes the Russian Federation. In such a case, the future importance of HELCOM is thereby to a large extent dependent on that HELCOM Recommendations either go further than EU Directives in terms of the hazardous substances that are controlled and the reductions that are necessary, or that they cover technical, scientific and policy aspects of Baltic Sea cooperation that fall outside the scope of the EU. It is noteworthy that in HELCOM guidance document on dioxins (HELCOM 2002b) it was stated that the data concerning dioxins in the Russian Federation is very vaguely reported (Selin and VanDeveer 2002).

HELCOM recently completed a study assessing the compatibility of HELCOM Recommendations and EU and OSPARCOM requirements (HELCOM 2001a). The report also looked at possible ways to rationalize HELCOM reporting requirements with those of the EU and OSPARCOM, in an attempt to reduce the burden on state officials by streamlining reporting formats and standardizing many informational demands. The report offers a detailed analysis of obligations under the three multilateral fora, making numerous recommendations regarding changes that HELCOM Parties might make to bring HELCOM recommendations more into line with EU and OSPARCOM (without lowering any HELCOM standards). The parties accepted a number of these recommendations at the 2002 HELCOM meeting. Seven of the twelve Recommendations adopted at the meeting -- HELCOM Recommendations 23/6-23/12 -- were largely consistent with the proposal of the harmonization report. Other suggested revisions have been under review by HELCOM (Selin and VanDeveer 2002).

Coordination between international fora is also relevant for reporting purposes. Different fora often have their own separate reporting requirements. In order to avoid that states are not subject to widely diverging reporting requirements that will strain sparse resources, efforts should be taken to ensure that reporting requirements are to the fullest extent possible harmonized across forums. Since the late 1990s, the EEA, EC, HELCOM and OSPARCOM participants have also shown greater interest in standardizing reporting requirements, monitoring systems and data gathering and calibration procedures, with the hope of simultaneously improving data quality and availability, and reducing the administrative burden on state officials (e.g., HELCOM 1998b, 2001a,b). Also the development of risk assessment procedures has been in focus (Fairman et al. 1998). In addition, harmonization and cooperation between HELCOM and OSPARCOM has continued and been also given new impetus due the EU enlargement and the development of EU marine strategies and marine protection. Also other standardizing efforts have been made e.g. under the Stockholm Convention, where a standardized toolkit for identification and quantification of dioxin and furan emissions was released (UNEP 2005).

While it is positive that more international forums are paying increased attention to hazardous substances including dioxins and related POPs, this development requires more attention to coordination between forums to avoid costly overlapping, or even conflicting and counterproductive policy actions in separate forums (Krueger and Selin 2002). In the future, Baltic cooperation needs to connect more to other fora and investigate possibilities for coordination and cooperation. Priorities may differ across fora, due to differences in economic development; agricultural and industrial production; and climatic, geographic and social conditions, but some issues and experiences will have validity across forums (Selin and VanDeveer 2002).

6.8 Framing the Baltic Sea dioxin issue

POPs issues have been framed as questions of long-range transport, persistence, bioaccumulation, and toxicity, which emerged out of science-policy interaction e.g. in the LRTAP assessments. This framing remained as the dominant one even as more parties became involved at global level and as other dimensions of the issue (e.g. existing stockpiles and technical assistance) emerged as further concerns (Eckley and Selin 2003).

The dominant theme on dioxins is and has been the issue of human health. As the main pathway of human exposure to dioxins is via food, the limit or target value for human intake has been a main tool for deciding the acceptable risk, thus serving as a basis for many current policies. When the limit value of dioxin is not exceeded, the fish is healthy source of nutrition in many ways. If the limit value is exceeded, there are potential harmful effects from dioxins, but the health benefits of fish oil remain. Agreeing on the significance of these impacts is complicated when discussing the effects of fish in nutrition, as the risks and benefits are distributed among different population groups. The nature of dioxins as a boundary object is crystallized in the debate over the limit or acceptable values.

The effects of dioxins on the Baltic ecosystem are a less discussed although acknowledged dimension. As the limit and action values set by EU are intended to protect human health, there is insufficient knowledge if this level will also protect the wildlife. Yet, if the dioxin levels in the Baltic are considered a significant problem for human health, also other species should be studied and taken into account in risk assessment and management. The Great Lakes guidelines have been proposed as an applicable approach to protect non-human species, but none of the current EU dioxin-related policies properly address this dimension.

While the integration of fisheries and environmental management has not dealt extensively with dioxin and other contaminant issues, links between the Baltic dioxin problem and fisheries management are recognized. Conventionally fisheries management looks at the state of stocks trying to optimize their yield within the constraints of the fishing effort. A reformed EU Common Fisheries Policy emphasizes the ecosystem approach that assumes e.g. that one does not examine only one species, e.g. Baltic herring, but looks at the ecosystem at large, including species interactions, environmental effects, public health, eutrophication and other such processes and entities. In such an approach Baltic Sea dioxin problems and fisheries management have a dynamic relationship as they both affect each other.

Although there has been only limited regulation concerning the dioxin concentration of commercial products such as cosmetic products, the idea of fish as a commercial commodity is established in EU regulation. This sets certain standards for food sold in the EU market in order to ensure safe products for consumer.

Degnbol et al. (2003) concluded that the dioxin problem does not include any conflicts between interests when it comes to fisheries and environmental stakeholders: all stakeholders want to reduce dioxin concentrations in fish. The objective is the same whether high dioxin concentrations are seen as ecosystem problem or as a food safety problem. But on a more concrete and specific level, the objectives for human health and ecosystem protection may differ in some respects. Constraints in aligning and uniting the objectives of various groups and areas were implied also by the study made for the European Parliament on conflict potential for fisheries caused by the EU strategy and proposed regulations of fish dioxins (Joas et al. 2001). Nor has the delicate balancing of risks and benefits of fatty Baltic Sea fish been pointed out much yet in the public discussions on the matter, although it has been discussed among experts.

Agreement may be more easily reached with respect to management actions that are in the interest of most groups, e.g. dioxin prevention and emission control (as these reduce both human and ecological risks in approximately equal proportions expect when congener profiles and emission localities play a role), but also this may apply to a general level only. When more concrete discussions start on the relative merits and costs of various measures to prevent dioxin formation, on prevention versus other management approaches such as extensive and expensive hotspot cleanup, on the reduction of herring fishing versus promotion of herring consumption, or on compensation for loss of value e.g. to fisheries, difficult choices will emerge between conflicting views and interests.

In a summarizing fashion, the following concrete initiatives and activities may finally be mentioned in particular within the various areas

of multi-actor policy and strategy coordination that have relevance for addressing the risks from DLCs in Baltic Sea fish:

- The work under UNEP-Chemicals, regionally and nationally on implementation of the POPs Convention also for the Baltic Sea, e.g. through National Implementation Plans
- The development and coordination of a European chemicals policy, especially in connection with the institution and implementation of the REACH system
- The coordination of the development and implementation of BAT and BEP criteria for EU enterprise branches
- The designation of the Baltic Sea as a Particularly Sensitive Sea Area (PSSA) in 2004 by the IMO, despite indications from Russian Federation to block the approval of such a designation (see press release from WWF-Sweden 2.4.2004, www.wwf.se)
- Development of food safety activities in EU and their coordination with other areas such as human health and the environment e.g. under EFSA
- The ongoing work in both EU, HELCOM and elsewhere (such as in ICES) on the development of marine protection policies and programmes for the Baltic Sea
- The increasing coordination between HELCOM and OSPARCOM (in collaboration with EU) also on policies and activities related to hazardous substances.
- In the Nordic Council of Ministers, increased interaction between the various sectors, working groups and other bodies, and improved collaboration with other actors (such as EU, HELCOM and ICES) in order to both avoid duplication and create new cross-cutting initiatives that could make the work on Baltic dioxins and related matters more efficient.

7 Management options and technological measures

7.1 Defining and evaluating options along risk chains and at various levels

Many options exist for managing risks associated with dioxin-like compounds. They include technical and other such as regulatory and information related options. It is important for efficient management that options and measures are identified and evaluated in a comprehensive and systematic manner.

It has been stated in many connections (e.g., van Leeuwen and Younes 1998), that "all efforts" should be made to reduce dioxin emissions and exposures. However, it is impossible to make all efforts conceivable and apply alls means technically available, at least to similar extent. Some of the measures available have important limitations and obstacles; some may be readily combined while others may be mutually exclusive. Some may be set in immediately, others only later, also depending on previous measures. For some measures, opportunities for additional risk reductions are exhausted. In general, the characteristics and impacts of options thus need to be scrutinized and compared.

Some areas of management and some classes of measures and options have been given little attention in strategies due to their inherent limitations in scope and approach (cf. 6, 8). For instance, the EU strategy (EC 2001, 2002a) and the strategy of the IOM (2003) are focused on exposure reduction by measures within food quality regulation and food production (and thus on human health), while some dioxin and PCBs strategies are oriented toward emission control. Also some of these are general and non-evaluative, mainly referring to existing regulations, norms and BAT approaches.

Important **areas of dioxin risk management** along risk chains (Fig. 19) have thus been left with little consideration, both more generally and especially with regard to the Baltic Sea. Such less considered areas of management, at least in the Baltic Sea region, include

- **Prevention** of dioxin formation, e.g. in connection with precursors in chemical products
- Influencing dioxin transport and fate within the **catchment** or other donor areas
- Steering of dioxin **cycling** in the sea e.g. by fisheries-based ecosystem management
- Reducing risks to **non-human** receptor organisms
- Alleviation of, adjustment to and **compensation** for risks and impacts (including socio-economic)
- **Utilization of the benefits** associated with dioxin-laden fish
- Targeted measures **specified** e.g. in terms of geographical area and risk group
- **Information** measures, expect for diet advisories (e.g. Lind et al. 2002, Darnerud et al. 2003).

More extensive emphasis on management measures represents an **opportunity-oriented** approach to dioxin risks that may be efficient in circumventing risks and problems in a more long-reaching and comprehensive manner. Increased and improved emphasis on management opportunities is in line with many relevant general policies in EU (and in Baltic Sea states) and with specific procedures such as the STOA (Scientific and Technological Options Assessment) applied by the European Commission and the European Parliament.

Although dioxin risks have been in focus for decades and considerable R&D work on management measures has also been conducted, most of the scientific **studies** and assessments have been centered on the occurrence and fate and the effects of dioxins. The studies and analyses of management measures have moreover usually been specific and have not evaluated measures, or their relationships with risks, in a comparative and comprehensive perspective. Some aspects of management measures are implicitly included in assessments of the problem and risks. In particular, identification, quantification and characterization of dioxin sources is directly linked with management opportunities.

Innovation is needed in developing emission and exposure reduction and other risk management options, including alternative products and processes. In this case innovation may be more straightforward than e.g. when developing ways

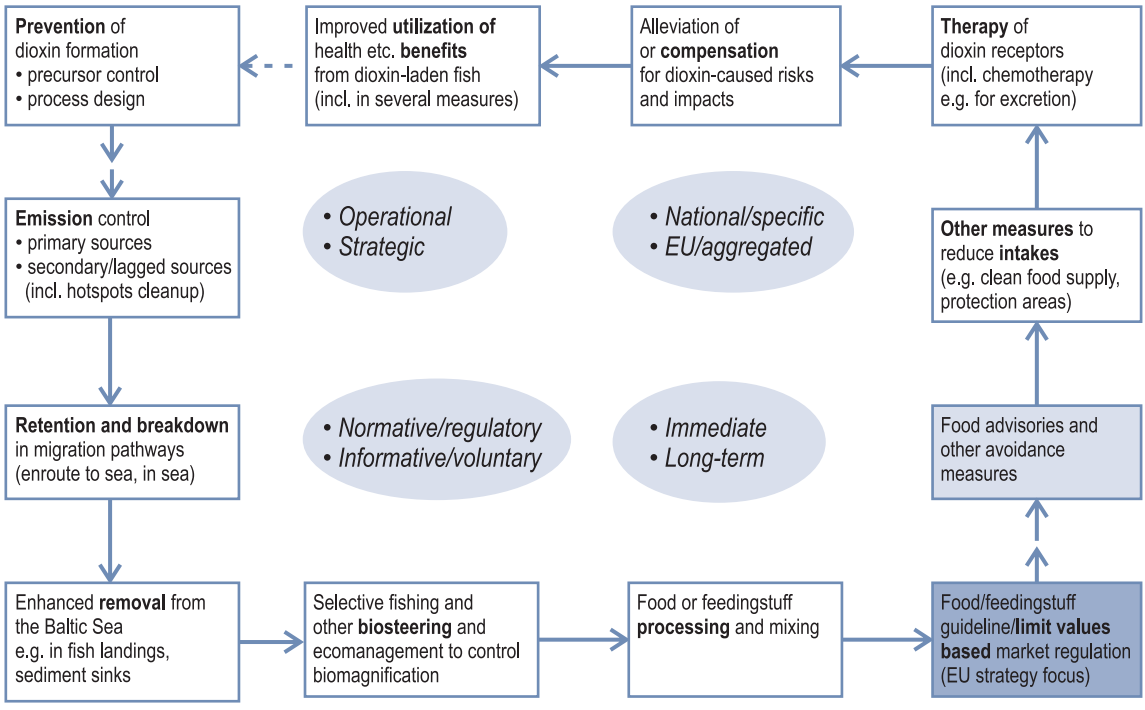


Fig. 19. Main areas of management options for dioxin-like compounds in Baltic Sea fish, structured according to the subsequent stages in the risk chain. Note that this does not necessarily represent an order of management, and that all options involve several levels and dimensions, shown in center.

to reduce the needs for a product or process, as the producers and other actors already possess much of the know-how and other prerequisites for developing alternatives, and less diffusion of knowledge and less innovation is needed. However, this direct linkage between the present producer of dioxins, precursors or co-factors on one hand and their alternatives on the other hand does also involves limitations and challenges, for instance as these actors may be bound by their present solutions.

Methodologically, analysis of management options can be seen as part of an **iterative process** that can inform knowledge production. In principle, some of the work on elucidating effects and risks may thus be reducible. On the other hand, such a general emphasis on options involves uncertainties and threats, e.g. if not linked with fate and effects information, implying the need for careful deliberation.

There are many ways to classify risk management measures relevant to dioxins, e.g. that used for BAT and BEP under the Stockholm Convention (EC and EU 2002). In the present connection, measures may generally be divided e.g. along the following dimensions:

- *Preventive or curative* measures, i.e. based on stage of risk formation and chain of events in management (cf. Fig. 19).
- *Normative, economic, technological and information* steering measures, and hybrids
- *Long-term, intermediate-term, short-term* and hybrid measures
- Measures in *Baltic Sea* countries and *elsewhere*, and local and global measures
- Measures on specific *classes of compounds* such as PCDD/Cs or dlPCBs, or broadly.

In the following, management measures are divided mainly along the first dimension, i.e. in measures directed to sources of DLCs before they enter the sea, to DLCs in the sea, and to DLCs after they have exited the sea. Existing classifications are utilized especially in prevention and control of emissions. In addition, the above other dimensions will be utilized. There is no strict separation between means according to the risk formation stage. For instance, some technologies are applied in the control of primary emissions as well as in cleanup of contaminated sediments or treatment of fish products and rejects.

An outline of measures is presented. In some areas, specific measures are not readily identifiable and have not been studied or tried, and only some general remarks have been provided. Some initial discussion is included in connection with the description of measures and in a summarizing evaluation; more general strategic aspects will be

dealt with in Chapter 8. Some emphasis will be put on areas of dioxin risk management that have been little studied and applied (cf. Annex 10).

The extent of peer-reviewed scientific **literature** in this field, despite its importance, is much more limited than in the fields addressed in the previous chapters, due in part to the applied and pragmatic nature of research in management technologies for dioxins and related compounds. More use has therefore been made of publications at dioxin symposia, being in many cases preliminary reports and limited extended abstracts, and in technical documents.

Within **source control**, emphasis is placed on prevention in stages preceding the processes of dioxin formation at the facility level. This is done to avoid duplication of the common focus on technological measures within facilities, and also because of the overall orientation to management issues at the policy level. Prevention of PXDD/F formation in products and processes is treated separately, and some attention is given also to prevention through the processes that are indirectly (but potentially highly importantly) contributing to dioxin risks, such as the needs for some of the formation processes.

7.2 Measures before immissions of dioxin-like compounds to the sea

7.2.1 Prevention of the formation of dioxin-like compounds

General considerations

Prevention of the formation of PXDD/Fs and even other DLCs is an area of risk management that holds much promise. For instance, by reduction of material flows and by promotion of low-waste technologies and low-transport solutions, the formation of PXDD/Fs in many processes can be avoided in the first place. A crucial advantage is that the subsequent formation of additional toxic compounds or materials and the need to treat them can be reduced.

Dioxin prevention includes many levels and may be accomplished at many stages:

- Prevention by influencing production and consumption **patterns including societal** needs (general or high-level prevention)
- Prevention by reducing the use of dioxin-forming **products, materials and processes** (general or specific technical prevention, particularly product-oriented prevention)
- Prevention **in facility-level** technical processes, e.g. by focusing on primary instead of secondary measures in thermal processes (specific technical prevention, see e.g. Pandelova et al. 2003 on the use of inhibitors in co-combustion)
- Prevention of subsequent PCDD/F formation **in treatment of precursors** ('downstream prevention', see especially Weber 2004).

Prevention of the formation of DLCs can also be divided according to the compound class:

- PCDD/Fs (the traditional and most common target of prevention)
- Other PXDD/Fs (including particularly PBCDD/Fs and PBDD/Fs)
- Prevention of other inadvertently formed DLCs (e.g., dlPAHs)
- DLCs advertently produced (e.g., dlPBBs); these are also produced inadvertently e.g. in combustion.

An important strategy consideration in prevention of DLCs is further the origin of the product or process causing their formation in terms of geographical **location and regime** of jurisdiction. Some DLC sources reside outside Baltic Sea countries and even outside EU, in countries producing potentially PXDD/F-forming materials (such as BFRs). In a global economy and with progressive development in the control of DLCs closer to the Baltic, such sources can become more important in relative terms for overall dioxin risks, and will present particular challenges for risk management, both technical and knowledge-related (e.g. regarding product chains and flows) as well as political.

Some preventive measures may have also **other benefits** and grounds, e.g. by reducing other dioxin risk factors or by generally benefiting the environment, health and safety. For instance, dioxins along with many other harmful substances are avoided by more modest levels of activities that cause their formation. A related issue is that such measures are often not considered for the purpose of dioxin management, as the relevant actors see dioxins as minor concerns, and as the relevant mechanisms do otherwise not facilitate their inclusion. The question hereby arises how far other benefits and purposes should be considered (cf. 8).

Reduction of the need for dioxin-forming products and processes

Little attention in research and analysis and in management strategy development has been directed to prevention at the level of **societal needs** in comparison with technological aspects of prevention. This may be related to the fact that environmental problems and their solutions are still not sufficiently understood as essentially depending on (and in turn influencing) social processes. Also the political nature of such a level of prevention can create obstacles for its consideration. However, e.g. in connection with the work on technology evaluation under the Stockholm Convention such more fundamental preventive strategies and options have been addressed in the form of root cause analyses within socio-economic evaluation of POPs abatement technologies (Annex 11).

Reducing the need for dioxin-forming products or processes may target activities that require materials (such as organochlorides and other organohalides) which are prone to form dioxins, or activities that directly cause dioxin emissions. In both cases, this dimension of dioxin prevention entails **low-waste economy**, sustainable consumption and cleaner production. Reduction of root causes of dioxin-forming processes can also include alterations in societal structures, both in social and physical sense, e.g. as to urban development. Because of the indirect connection with dioxin formation, information based and economic instruments may be important in this area.

These measures presuppose a more **fundamental questioning** of the necessity of present ways of life and production, thus having in some respects a character of political rather than technical choices. This may be both an asset and a constraint. Reduction of the need for products and processes that induce PCDD/Fs, and a general dematerialization of society, may offer attractive and efficient ways to manage risks at a fundamental level and to achieve other desirable goals, but may also clash with reigning systems of and interests in economic growth. They may also in some respects be harder to 'sell' to decision makers and others that are reluctant to consider and implement deep-going changes, in comparison with more limited technical 'easy fixes' e.g. in the form risk prevention in facility-level process control, although the former might potentially be more important for risk reduction.

There are also more genuine **knowledge related impediments** for this type of dioxin risk prevention. As measures are devised that address processes far 'upstream' in preceding formation, it often is difficult to know whether such measures will in fact reduce dioxins or will also affect dioxins in other ways, even increasing other risks they cause (and, at worst, increasing the net risks). For instance, some recycling processes increase dioxin formation, especially when chlorinated organic materials are involved. Likewise, the phase-out of some potentially PCDD/F-forming products or processes may induce the use of other products and processes that still form PXDD/s (at some rate), in addition to having some other disadvantages, along with other benefits. Thus, intuitively attractive measures that seem to be able to reduce some of the risks from DLCs (and to have general environmental and health merits) may have unanticipated drawbacks also for these risks, and need to be carefully scrutinized. This is related to the general difficulty of identifying causes for developments, including both successes and failures, especially in the case of indirect influences (cf. 8).

Despite these uncertainties, it may be said in a generalizing and simplifying fashion that the formation of PCDD/Fs can be on the average benefited by economies and technologies that involve

- Less consumption and material flows
- Specifically, less use of organohalogen compounds (and, with still more generalization, halogens)
- Less combustion (in industrial and waste treatment processes and other connections like traffic)
- Improved functional and structural fire safety by design and material choices (see KemI 2003).

Prevention by product use regulation and steering, and by alternative products and processes

General

Regulation of uses of dioxin-forming products is a common approach to reducing associated risks. Product use regulation is often combined with substitute products and processes (see above).

Use regulation can involve total bans, partial or conditional bans or restrictions of variable extent and content. It may apply to all uses or to some, e.g. downstream uses. Several levels (e.g., international or national, general and specific) are relevant. In addition to regulating uses directly, also indirect measures steering and influencing them may be included.

The **substitution** of dioxin-containing or dioxin-forming products and substances constitutes an area of risk management that can be approached within a defined technological domain and utilizing the abilities of enterprises and other actors engaged in the production and use of commodities. Substitution of substances is also for other reasons naturally considered when looking for alternatives. It is consequently included or considered in many risk management strategies and schemes, e.g. in the risk reduction strategies under existing (and proposed new) EU chemicals policies.

The **imperfect knowledge and** imperfect systems of **application** create obstacles also regarding substitutes. For instance, some halogenated biocides have emerged as safer alternatives for persistent and non-selective classic biocides such as DDT. The PCDD/F forming potential is an additional consideration that may not be alone decisive even for environmental safety. Likewise, chlorinated solvents, some of which may form PCDD/Fs, were introduced in part as safer (e.g., less explosive) substitutes for traditional aromatic solvents. Some halogenated substances may also have important more indirect benefits for society even within health. Such benefits will constraint efforts to achieve dioxin risk reduction by simple and cross-the-board dechlorination or dehalogenation solutions (cf. 8).

In product use regulation, a key issue is **how broad** options are defined. In evaluating options, their impacts in terms of risks and also of benefits (including socio-economic and safety related) may thus need to be considered (Yosie 1996). In addition to PCDD/Fs, also other DLCs and their precursors can be addressed. Within risk management policy and technology, even other compounds (such as all POPs) may in some cases be appropriately included in considerations and actions.

In the present connection of dioxin prevention, **precursors** of PXDD/Fs are of main importance. The following hierarchy of precursors can be distinguished (cf. 2.2.2, Annex 3):

- Strong precursors such as some chlorophenols and chlorinated phenoxy phenyl esters
- Compounds where Cl or other halogens are substituents in a phenyl ring
- Other aromatic halogenated compounds
- Other halogenated organic compounds (aliphatic compounds and other cyclic compounds)
- Organohalogen compounds with halogens only as anion in salts (typically chlorides)
- Inorganic halogenides.

This hierarchy is only indicative, as the dioxin-forming potential of compounds is not defined solely by their structure but also by their fate during the life cycle. Substances having lower inherent dioxin-forming potential (such as chlorinated aliphatic compounds) may cause greater formation of PXDD/F if commonly subject to thermal processes, as opposed e.g. to environmental degradation, as is the case for many precursor pesticides and herbicides dispersed in soil.

Important chemical categories

Initial screening of the IUCLID database for High Production Volume **(HPV) chemicals** in EU that possess Cl or Br especially in aromatic structure resulted in hundreds of chemicals (cf. Annex 3). Use and restriction data mainly for Sweden, Denmark and Finland were used in prioritization (Table 31). The significance of many of these substances as dioxin precursors is not clear. On the other hand several other chemicals exist that are potential precursors. These data (mostly for c. 2000) may also not reflect the present situation. This is true particularly of Baltic Sea countries where many potential PXDD/F precursors have not been produced. Several precursors produced and used in EU are already in phase-out stage, or will be reduced by measures already taken. In many cases, Sweden, Denmark and Finland (based also on Nordic cooperation) have restricted production and use earlier and more completely than other EU member states have. The pace and completeness of phase-out however varies. In Baltic Sea countries outside of or recently included in EU, production and use patterns may be different. For some HPV chemicals much of the EU production is exported

outside EU. For many others imports from other regions imply an influx of PXDD/Fs or precursors that is difficult to control.

The **dioxin-forming potential** of the HPV chemicals is poorly known except for a few precursors such as chlorophenols, chlorobenzenes and some chlorinated herbicides or other biocides. Even within these classes the potential varies. Also chloranil and compounds based on it are noted potential sources of PCDD/Fs, and the same may be true of some other chloronitrogen compounds. On the other hand, organochlorine compounds where Cl is present only in a salt and even inorganic chlorides may form PCDD/Fs e.g. in incineration (cf. 2.2.2, Annex 3). Standard EU risk assessments for biocides, or other existing chemicals, do not address risks associated with formation of PCDD/Fs e.g. upon combustion. Evaluation of the dioxin-forming potential must thus be continued, preferably in connection with evaluations of uses and possible alternatives, in order to focus on those precursors with greatest potential risk and also greatest potential opportunities for risk prevention and reduction.

Some of the potential PXDD/F precursors are multiple **use purpose** chemicals, e.g. some chlorinated biocides, benzenes and aliphates. Some are used mainly in closed systems as intermediates in chemical synthesis and are efficiently recycled and treated, while others are deliberately used in dispersed applications (e.g. herbicides and some consumer chemicals). Data on production and use of many precursors are not readily available.

Along with phase-outs of many halogenated chemicals, also emission prevention and control measures have increased. However, the **implementation of regulatory measures** such as bans and reductions of varying coverage (production, marketing, import, use) is unclear for many dioxin precursors, both on EU level and in Baltic Sea countries. The mere notification, listing, prioritization and even regulatory action in the form of risk reduction strategy development does not yet guarantee actual reduction of uses and emissions; on the other hand, some precursors which are not regulated may have been subject to considerable voluntary reductions (Table 31).

PeCP is an important precursor due to its high potential to form PCDD/Fs (cf. Annex 3). It was banned in Sweden in 1978, in Denmark in early 1980's, and in Finland and Germany in late 1980's, and subsequently phased out. PeCP use still continues in EU, due to claims for particular needs in some materials and structures. Uses are scheduled to ebb out by 2008 (OSPARCOM 2001). Less dioxin-forming biocides have emerged as substitutes. These include 2,4,6-TCP, produced and used in EU but not in Baltic Sea countries. Notably, also alternative processes have replaced PeCP, such as thermo-mechanical wood preservation, the use of naturally resistant wood, and planning of structures for durability, in a process of adaptation and innovation to cope with the loss of a dioxin precursor.

The risks from PeCP use have been assessed, both for EU (ERM 1997) and OSPARCOM (2001) and also by Euro Chlor (1999); the latter did not consider PCDD/Fs. Risk reduction strategies have also been evaluated (ERM 1998). CSTEE (1998) commented on the "risk aspects" of these assessments, concluding they cannot be considered adequate due to an unsatisfactory literature review and many inconsistencies between reported data and conclusions. However, CSTEE did not provide alternative assessments, and especially did not comment on the analyses of ERM regarding risk reduction or on risk management in other respects, including opportunities for replacement of PeCP.

OSPARCOM (2001) evaluated previous and possible additional risk reduction measures for PeCP, including cessation of use, control under the Biocides Directive (98/8/EC), emission limit values, economic instruments, ban on import of treated material and products, labelling, information campaigns, and clean-ups. Based on effectiveness, practicability, economic impact and monitorability, it was concluded that while accelerated phasing out seemed the most effective and practical option, it may be unrealistic (with reference to negotiations on Directive 1999/51/EC); permits and emission limit values for producer and user facilities were regarded as the most appropriate way forward. The grounds for this were not explicated, and also the relationships with other instruments such as import bans and economic and information steering remained unclear. Further measures on PeCP may be judged also in relation to the development and significance of resultant dioxin risks.

Of **chlorobenzenes**, mono- and dihlorobenzenes and 1,2,4-TCBz are on the IUCLID list of HPV chemicals, and additionally many derivatives are produced in high volumes in EU (cf. Annex 3). Their specific dioxin-forming rates are not clear. HCBz is already in the phase-

out stage. 1,2,4-TCBz, used e.g. as a carrier in dyes, has been assessed under EU's Existing Substances regulations (by Denmark), and risk reduction strategy development is underway. The regulatory impacts assessment by DEFRA (2002) indicates that 1,2,4-TCBz is among the priority substances for which the costs from phase-out, depending on measures, could be significant. PeCBz is on the EU list of priority substances, and subject to measures under the POPs Convention also in EU (see EC 2003b). PeCBz and TeCBzs are inadvertently produced from some chlorinated chemicals, partly in same sources as PCDD/Fs. Thus, some measures such as restriction of uncontrolled burning and control of metal industry emissions (see below) will reduce both these precursors and PCDD/Fs themselves (Environment Canada 2005).

Among **pesticides**, lindane is being phased out in EU although some minor uses remain. It is however present e.g. in many imported materials (DEFRA 2002). Pesticide use in Northern Baltic Sea countries is small in European comparison, and chlorinated pesticides have been generally phased out. Of chlorinated pesticides still used in Baltic Sea countries, e.g. chlorpyrifos may be mentioned.

Among **herbicides**, 2,4-D and some other chlorinated phenoxy phenol acids such as dichlorprop and mecoprop and derivatives are produced in EU and used in Baltic Sea countries, in decreased amounts. Although less dioxin-forming than 2,4,5-T, 2,4-D may cause some PCDD/Fs in thermal reactions. The dioxin-forming potential of the high-volume MCPA may be still lower. Among other chloroaromatic herbicides fluazinam, fluroxipyr and metazachlor are used. In general, use of plant protection products has been reduced, e.g. in Sweden from 5 to <2 t active substance a^{-1} from early 1980's to early 1990's (SNV 2005), that of formulations causing greater environmental hazards even more.

Among **biocides**, mainly propiconazol and tabuconazol are used in Denmark (Lassen et al. 2004), the former also in Sweden (KemI 2005) and Finland. These contain Cl on a benzene ring, but their dioxin-forming potential is unknown. In addition, other biocides such as dichlofluanid that is used e.g. in Finland for wood preservation contain Cl in other structures. By far the greatest use amounts are those of the inorganic hypochlorite that has been found to cause formation of PCDD/F in chlorobleaching (Rappe et al. 1989c) and also considered a potential source of PCDD/Fs in association with PeCP impurities (Ohlsson, ref. by Engwall et al. 1999). The rate of PCDD/F formation from hypochlorite may depend greatly on production and use processes, including other substances present. Rappe et al. (1990) found that tall oil (resin) based soft soups without major Cl ingredients contained PCDD/F levels 100-fold higher that hypochlorite bleaches.

Several **chlorinated solvents** produced and used in great quantities in EU and also Baltic Sea countries may contain or form PCDD/Fs. These include 1,2-dichloroethane that is used in PVC production, and tetrachloroethene (cf. Annex 3). However, all chlorinated (organic) chemicals may form PCDD/Fs in some circumstances. Hexachlorobutadiene is also a precursor priority chemical but is reportedly not produced or used in EU (DEFRA 2002). In general, use of chlorinated solvents has been much reduced e.g. through more extensive recycling and regeneration, in addition to substitution by other solvents. Vinyl chloride production has not decreased, and is of particular importance due to the great amounts produced for PVC manufacture.

PVC has been targeted in dioxin risk management in many connections. PVC has some technical properties that may be difficult to achieve with alternative materials (Shibata et al. 2003), and may justify some uses particularly if risk management incorporating e.g. safety considerations is practiced. Thus, reduction of PVC use may require additional analysis and planning. The efficiency and overall grounds of PVC use reduction may depend e.g. on how PCDD/F-related risks to human health and the environment are prioritized in relation to the benefits from PVC also to these objects of protection, and on how fast and with what consequences alternatives may be developed. For instance, if considering the risks associated with the formation of PAHs from combustion, common alternatives to PVC such as PE, PP and other plastics seem less attractive (cf. Takasuga et al. 2003). Due to such choices, the reduction of unnecessary uses of plastics in general may become a viable option (cf. above). This however requires and agreeing on what uses and needs are truly unnecessary and how to implement policies and strategies that act accordingly. The difficulty even in this case of alternative products, strategies and actions (PVC, other plastics, or less plastics in general) is to balance a process of careful deliberation and anticipation of all key consequences with one based more on 'gut feeling' and on the hope that less riskier alternatives will emerge.

Table 31. Implemented and possible new risk management measures for chemicals produced or used in EU that are or may be potential precursors of PXDD/Fs. The list and data, mainly from IUCLID, www.mst.dk, www.kemi.se, SYKE, www.eurochlor.org and SPIN (2005), are not exhaustive. Cf. text, Annex 3.

Chemical	Production in EU		Uses, BS countries (EU)		Dioxin-form pot	Regulation/prior measures		Implementation	Pot new measures
	amounts, kt a^{-1}	BS countries (others)	amounts, kt a^{-1}	purposes, patterns		EU (other intl)	main BS countries		of prevention ?=to be considered
MCBz	70 (-93)	D etc (not BS)	FI 1	nitro-CB prod etc	< (indirect)			emissions to 15%	?
1,2-DCBz	21 (-88)		FI0.003 DK0.001	nitro-CB prod etc	≈(pyrolys)		S ban		extended replacem
1,3-DCBz		as MCBz	DK (etc)	chem synth	≈(pyrolys)				extended replacem
1,4-DCBz	26 (-94)	D (not BS)	FI0.1 (EU15)	pestic+dye synth	≈(pyrolys)	PL1 (no restr yet)	S ban		extended replacem
1,2,4-TCBz		D (not BS)	S DK FI	by-prod	≈(pyrolys)	PL2, RRS			reduction
PeCBz		discont'd		quintozen prod	≈(pyrolys)	PL	S etc		extended replacem
HCBz	<	D (not BS)	S DK	fungicide etc	≈	(partial)	S FI DK bans	prod & use ↓	?
Cl-nitroBzs		D (not BS)	DK		?				possible substit.
chlorotoluenes		D (not BS)	S		?				possible substit.
2/4-chloraniline		D (not BS)	in mater?	dye prod	≈?		SW		extended replacem
chloranil			in mater?	dye prod	≈?				reduction
2,3/3,4-DCA					≈	PL1, RRS			multiple RR
4-CP	3		S DK	further synth	≈		S (CPs)	emissions ↓	?
246-TCP		(UK)	S (FI)	biocide	≈ (<245-)		S (CPs)		further restr.
PeCP; Na-			S0.1; 1	fungicides	>	reductions	ban (all BS)		further restr.
lindane		(F, IT)	(DK S)	pesticide	≈	PL, restr	ban (all BS)	phased out	?
2,4-D		(prev SW, D)	DK	phenoxyherbic	<		DK restr		further restr.
MCPA			FI S0.4	phenoxyherbic	<		DK, FI restr	uses ↓	further replacem.
dichlorprop			DK (↓)	phenoxyherbic	≈?		S ban, DK, FI		?
mecoprop			↓	phenoxyherbic	≈?		S, DK, FI		?
diuron			DK0.07	biocide	?	PL	S ban		further restr.
chlorpyrifos			DK1000	insecticide	?	PL	S <restr		further restr.
propiconazol			DK≥6	fungic (wood)	?		(FI)		further replacem.?
prochloraz			(S0.01)	fungicide	?		(F)		further replacem.?
fluazinam			(S0.03)	fungicide	?				restr/replacem.
fluroxipyr			(S0.03)	herbicide	?		(S)		extend RR
flamprop-M			prev. SW	herbicide	?		S ban		extend RR
metazachlor			(S0.03)	herbicide	≈? (anilid)				restr/replacem.
vinyl chloride	5200	S, NO, D 8,	S60 FI	PVC prod	≈ (therm)		S (FI imp)		replacem. as appropr
chloroethane			S10	ind	<				?
DCM	140	D (etc)	FI1 S DK	solvent, pharm	<	EU (HEL)	S (FI)	recycling ↑	?
chloroform	240	D (+etc)	S0.2 FI0.1	synth of HCFC	<	PL2, restr	S (FI restr)	solvent use ↓	?
CCl_4	60 (-96)		DK0.03	prod of CFCs	<	bans, restr	S DK FI	most uses phased out	?
1,2-DCEa			FI2	VCM prod etc	<	PL, restr	S ban FI restr		further replacem.
1,1,1-TCEa	<?		DK0.8	CFC prod etc	<	(restr)	(FI ban)	phase-outs	?
1,1,2-TCEa			DK	solvent prod	<	(restr)	(FI restr)	solvent use ↓	?
1,1,2,2-TeCEa			FI	ind	?	(restr)	S (FI)		further reduct ?
1,1-DCEe			S 1	ind	?	(restr)	FI restr		further reduct ?
TCEe	110 (-95)		S DK FI 0.5	degreas, synth	<	PL1	S	recycled	?
TeCEe	160 (-94)		S DK FI 0.5	clean, synth	<	PL1, RRS	S	use ↓	?
C_{10-17} Cl-alkane			DK50 S0.2	var	<?	PL			?
Cl-acetic acid			S4 FI	var	<				extend risk reduct
hypochlorite			DK2000	biocide	<	PL2			safe use
Cl	12000	P 5, S 2, FI 2		bleach, synth	</≈ (prod)	PL3 (HEL)		tech shifts	cont'd safe prod/use
1,2-DBEa				var	</≈ (pyrolys)	PL	S FI		?
Bis-PeBDE			(S DK FI)	flame retard	</≈ (pyrolys)	PL1 (restr)	(S, FI ban)		alt fire saf
HBCD				flame retard	</≈ (pyrolys)	PL2 (restr)			alt fire saf
OBDE			(S DK)	flame retard	</≈ (pyrolys)	(restr)	S (FI)		alt fire saf
dekaBDE				flame retard	</≈ (pyrolys)	restr	S, FI restr		alt fire saf

Abbreviations (cf. list of abbreviations): CBz= chlorobenzene; Cl-nitroBz= chloronitrobenzene; DCA= dichloroaniline; DCM= diclorometane; DCEa/e, TCEa/e, TeCEa/e= di-, tri- and tertachloroethane/ -ethene; DBEa= dibromoethane; TBP= tribromophenol; PeBDE/OBDE/dekaBDE= penta/octa/decabromodiphenylether; HBCD= hexabromocyclododecane; VCM= vinylchloride monomer; CFC= chlorofluorohydrocarbon; PL= priority list (EU); RRS= risk reduction strategy (EU); S= Sweden; P= Poland; NO= Norway; BS= Baltic Sea (catchment); HEL= HELCOM.

Brominated flame retardants (BFRs) were introduced in part as environmentally friendly substitutes for chlorinated diphenyl ethers (PCDEs), until it was realized that also these alternative substances cause risks, even by forming PBDD/Fs. Muir and Alaee (2002) estimated that the cost-benefit ratio of alternatives to BFRs favors their substitution, referring to a calculated cost of 170 M$ for replacing all BFRs and 24 M$ for PeBDEs in the Americas. These authors judged that this is vanishing small compared to the value of the products to be treated, and also to the health costs of assumed IQ losses and thyroid disorders due to PBDEs and related compounds; they also claimed there is no evidence that alternatives could not provide the same protection to life, and that obstacles are mainly technical and institutional. However, indirect benefits and impacts and relative efficiencies of BFRs and alternatives seem uncertain. There is thus a residual risk that substitutes will introduce new risks either in themselves or by other mechanisms (e.g. lower than claimed life-saving potential). It also seems that the assumptions underlying BFD-attributable health costs are controversial.

A parallel for such prevention of dioxin risks is the risk reduction strategy and activities in Germany for chlorinated paraffins (CP), being the first entry from the EU Existing Substances program priority list. This has involved e.g. cessation of production forcing development of alternatives; industry lists of undesired substances and requests to vendors for CP-free metal working fluids; and appropriate waste management (Stolzenberg 2000). As a summarizing evaluation, it was concluded that after crossing a threshold of initial investment efforts, the substitution process turned out to be self-preserving, irreversible and causing net benefits (cf. 8). However, in the case of dioxins, the challenge is to efficiently utilize precursor substitution.

Prevention of dioxins in waste recycling and waste management outside incineration

Within prevention of PCDD/F formation in waste management the focus has been on preventing the *de novo* formation in incineration plants (see below). However, waste management (including recovery and recycling) and material flow steering in general offer plenty of options, although also constraints, for preventing the formation of dioxins and DLCs.

In waste incineration both reduction and increase of the formation of PXDD/Fs is possible, as in other areas of waste management. Many recycling processes, notably those involving catalyzing metals in metal recycling, have caused considerable PCDD/F emissions (cf. 3.2 and Annexes 3 and 4). It is possible to utilize optimized recovery and recycling processes for prevention of dioxins and (simultaneously or alternatively) for separation and treatment of them. The general problem is largely that many recycling systems were designed based on other criteria and without explicit consideration of dioxin formation and the needs and possibilities to modify the process to avoid this.

Schlummer et al. (2002) reported the development and testing of a polymer recycling process (for PVC and ABS plastics) that is able to eliminate additives and contaminants, including PBDD/Fs, from the polymers. Also removal of BFRs may be possible to prevent PBDD/F formation. Likewise, removal of chlorinated dioxins and some of their precursors in plastics and in other materials might be technically possible, to reduce the formation or transfer of DLCs or both. Such processes may be seen as pretreatment steps before incineration, thus avoiding their thermal formation.

Prevention in combustion and other thermal processes

Prevention of the formation of PCDD/Fs and other DLCs can be accomplished in technological processes that cause such formation and emissions. Prevention in dioxin-emitting processes involves a more specific, technical and concrete level of prevention than the largely policy or strategy level prevention measures oriented at root causes of PCDD/F formation. Both types and levels of prevention are important and may complement each other, even though it may be emphasized in general that by preventing at the level of root causes, the need for further technical-level prevention can be removed. In this sense, root cause prevention is primary and may be more broadly effective.

On the other hand, the technical level of prevention can in some cases provide an operationalisation of the overall and broad prevention policy, instead of being the 'backup' alternative if e.g. waste prevention has not been yet possible. An advantage of preventing dioxins in processes emitting them is the more

certain and direct control over such well-defined systems (such as in specific facilities). Also regulation of this level of prevention may be more straightforward, e.g. through extension, modification and implementation of existing emission control regulations.

Prevention of dioxins in technological processes involves the following key sub-areas

- **Fundamental alterations** in industrial processes e.g. with regard to their technological principles
- Control of **raw materials** and other material inputs in the processes, such as the quality of wastes or fuels to be combusted
- Control of **process conditions**, such as the '3t' (time, temperature, turbulence) in incineration and other combustion systems
- **Inhibition** of formation at various process stages, from primary to *de novo* synthesis (e.g. in flue ashes), using e.g. inhibiting substances (e.g. Shibata et al. 2003, Yasuhara et al. 2005).

Prevention in such technological processes emitting dioxins is by definition an integral part of emission control technology, even though the preventive approach can be rather different from end-of-pipe controls. In any case, e.g. in waste management and industrial facilities both stages of emission control need to be and are also in practice usually treated in an integrated fashion. Therefore, prevention in this sense is treated in more detail below in connection with emission control.

Prevention in treatment of dioxin precursors

In addition to reducing the production and use of precursors of PXDD/Fs, treatment of those **precursors already produced and stored** is important. Increased attention is paid to destruction and treatment of POPs, including many chlorinated dioxin-forming compounds, based especially on the POPs Convention. Prevention of dioxin formation within treatment of POPs stockpiles may be termed 'downstream' prevention. It will be dealt with mainly in connection with remediation of contaminated sites and treatment of stockpiles (cf. 7.2.2). Only some initial remarks are made here.

Technical treatment of dioxin precursors in wastes and stockpiles has been in some cases considered in a limited fashion, focusing on the precursors themselves and not fully evaluating the possibilities for the **control of PXDD/F** formation. Weber (2004) in a critical evaluation of destruction technologies for POPs from the point of view of PCDD/F formation stressed that high concentrations of PCDD/Fs have been shown to result e.g. from some of the non-combustion treatment technologies ranked highest by UNEP and UNIDO. Thus, some of the acclaimed options proposed for the destruction of PCDD/F precursors like PCBs may instead cause additional formation of PCDD/Fs. Such processes underline the need for looking at dioxin formation, fate, destruction and treatment in a comprehensive manner, from cradle to grave and including precursors along the risk chain, and accounting for the cycles, links and feedbacks in these processes.

The formation of PCDD/Fs from PCBs presents some particular aspects, as dlPCBs are included and as their control also for other reasons is a priority e.g. in POPs abatement.

7.2.2 Control of land-based emissions

General

Emission control options are evaluated here separately from prevention in processes causing emissions, to emphasize the latter. Processes must however often also be viewed as a whole, including preventive and other measures down to end-of-pipe controls. Emission control can and should in many cases be extended to reducing emissions from environmental sources beyond the facility originally emitting them (see below).

The importance of controlling and reducing releases has been acknowledged in the general context of source control (e.g., Verstraete 2002). In many national programmes on PCDD/Fs, emissions have been in focus, and considerable reductions have been achieved throughout Europe and also in Baltic Sea countries, especially in direct emissions to air (Quass et al. 2004a, c).

Some areas of emission control are well established on a commercial scale. In others technologies are still at a laboratory or pilot scale. The design, development, up-scaling, demonstration, testing, and commercialization of emerging technologies is a long process and the potential of a technology in relation to existing technologies is not always easily assessed.

In addition to primary emission control, releases from waste materials need to be

controlled. Fly ash from incineration presents considerable problems of containment or destruction of PCDD/Fs.

Primary emissions

Plenty of information is available on measures for controlling and reducing dioxin emissions from waste incineration and, increasingly, industrial processes. Some comparative evaluations have been produced in connection with international environmental instruments, especially the Stockholm Convention (e.g., UNEP 2004b-f). However, many of these evaluations address mainly treatment of PCBs in environmental matrices. As shown by Weber (2004), some of such evaluations and recommendations ignore the potential for PCDD/F formation.

Within emission control, air pollution abatement technologies have dominated. There is an increasing need for emission control in other areas. In particular, control measures for **solids** resulting from air (and water) purification are needed. This is closely related to the control of secondary emissions e.g. from waste management and also from environmental sources (see below).

There are synergies but also competitive or conflicting relationships with other emission control and non-environmental technologies and solutions. Some intuitive abatement measures may not be effective but on the contrary increase emissions. For instance, Marklund et al. (1992) reported that higher PCDD/F levels were found after an additional air pollution control system than a fabric filter, due to *de novo* formation.

In the present work, control technologies are described at a rudimentary level for the purposes of strategy evaluation. In primary emission control, the following specific points and areas should in particular be noted (cf. Table 32, Annex 10):

- **Thermal** technologies provide robust and efficient means of destruction of PXDD/Fs, PCBs and other DLCs. On the other hand, PCDD/F formation in thermal treatment of dioxins and their precursors needs to be prevented (see e.g. Weber 2004). Control of thermal reaction conditions in both incineration and in other types of combustion (e.g. of fuels) as well as in other, e.g. industrial thermal processes, is a key to success. The use of inhibitors is also growing.
- **Non-incineration physico-chemical** methods have been under intensive development. Both chemical de-halogenation by base-catalyzed and non-basic methods and photolytic degradation e.g. supported by catalysts (Wu et al. 2004) have been shown to be efficient for destruction of PCDD/Fs and dlPCBs. However, most technologies for which results have been published have not yet developed to full-scale commercial-level applications. Some of these technologies have been studied mainly in connection with treatment of stockpiles and contaminated sites (cf. below).
- Technologies for **inhibition** of dioxin formation in combustion and other (thermal) processes also in small-scale units and outside waste incineration are under development. Due to the extent and scale of the units, sufficiently simple and cost-effective improvements are sometimes difficult to achieve through equipment renovation. A key approach to abatement thus seems to be control of fuel and operating conditions.
- Along with improving scavenging of PCDD/Fs and other DLCs in gaseous, liquid and solid streams, the treatment of **residues**, including residues from pollution control and in ash, is an increasing need. For fly ash, full-scale treatment technologies are evolving also in low temperatures (e.g., Behnisch et al. 2002b, see Annex 12)
- Within **municipal solid waste incineration** there are still no emission limit values at EU level, but national standards and EU standards for hazardous waste incinerators are applied as deemed appropriate (e.g. Francois et al. 2000). Considerable development of control technologies has taken place (e.g., Baeyens and Goyens 2000, Kim et al. 2001). In Western Baltic Sea countries, reductions of these emissions have taken place; in Eastern countries more potential for emission reductions exist. Prevention of dioxin formation can be accomplished by plant design, operation conditions, inhibitors, integration with NOx control and catalytic breakdown (e.g., Raghunathan and Gullett 1996, Bonte et al. 2002, cf. Annex 10). Under unfavorable conditions some emission control technologies can act as precursors

Table 32. Summary of established and potential reduction measures for dioxin emission in major source categories, emphasizing industrial and waste management processes. The categories and measures estimated to be of greatest emission reduction potential are highlighted. From various sources. Cf. Annex 10.

Source category	Established in West EU Baltic Sea states – preventive	Established measures in Western EU Baltic Sea states – curative	Potential further measures based on BAT – preventive	Potential further measures based on BAT – curative
Iron and steel / sintering	none specifically for dioxin prevention	▪ belt dust extraction (electrostatic filters) ▪ hall dedusting and filtering	operating conditions	▪ wet filters ▪ adsorbents in electr. filter ▪ fabric filters
Iron and steel / coking	▪ charge hole control/sealing ▪ large furnaces ▪ water immersion at standpipes	▪ displacement gas extraction	▪ elastic seals on coke charge doors ▪ air cooling instead of quenching	extraction of emissions at coke removal doors
Iron and steel / blast furnaces		gas dedusting by separators, cyclones and filters	furnace designs operating conditions	not considered cost-efficient or necessary but may emerge
Iron and steel / oxygen steel	primary off-gas reuse	dedusting esp. of secondary gases, fabric/electrostatic filters	operating conditions, e.g. gas control	not considered cost-efficient or necessary but may emerge
Iron and steel / electric furnaces	scrap input control	dust emission control in primary and secondary off-gases (fabric filters)	▪ scrap shredding ▪ diversion of contaminated scrap	adsorbents (possibly)
Iron and steel / casting foundries	primary gas reuse	waste gas dedusting (fabric filters)	▪ plants for waste-heat recovery (incl. cold-blast, rotary furnaces)	▪ optimising filters ▪ sorbents, secondary filters
Non-ferrous metal and aluminium / secondary Al production	-grinding and grading -rotary drum furnace smelter	off-gas cleaning together by dry sorption (hydrated lime) and fabric filter	▪ salt use reduction (hearth-type instead of rotary drum furnaces) ▪ burner modification ▪ O_2 injection in gas-air burners	▪ novel adsorbers (C+lime) ▪ second filters (1 & 2 stages) ▪ use of Al remelting or sintering tech to scrap Al
Copper production (esp. secondary)	input material quality control	off-gas dedusting, fabric filters	▪ avoidance of contaminated feed materials ▪ O_2 injection	▪ adsorbents ▪ use of Al remelting or sintering tech
Secondary zinc production		off-gas dedusting by fabric filters (in roller drum quenching)	roller drum and hot briquetting improvements	use of adsorbents in other metal processes
Power stations	no specific dioxin reduction measures	off-gas dedusting in chimney (electrostatic or fabric filters)	▪ replacing solid fuels by natural gas ▪ fuel throughput control	
Industrial firing places		off-gas dedusting in chimney (electrostatic, fabric filters, cyclones)	▪ avoidance of PVC-coated or PCP-treated wood and chipboard ▪ replacing solid fuels by natural gas	
Small-scale firing places	Firing optimization and control (electronic ignition, temperature control, boiler design etc)	none for fluegas cleaning	▪ reducing heating (insulation) ▪ replacing old firing places ▪ avoidance of PVC-coated wood ▪ other heat sources ▪ use of dry seasoned wood ▪ optimised heating elements ▪ improved degree of use	
Waste incineration (municipal, hazardous, medical)	▪ materials selection ▪ process control (time. temperature, turbulence)	▪ post-combustion controls: ▪ electrostatic precipitators ▪ dry scrubbers ▪ fabric filters ▪ wet scrubbers ▪ AC/lime addition ▪ catalytic reduction e.g. with NO_x ▪ condensation device/collection	▪ improved incineration control ▪ improved waste selection ▪ materials processing (briquettes/RDF) ▪ alternative thermal treatment systems e.g. pyrolysis and tunnel reactor-gasification	▪ activated coke/C adsorption ▪ reagent (inhibitor) addition, e.g. lime+coke, urea ▪ fluegas temperature control ▪ catalytic TiO_2/H_2O_2 oxidation ▪ high-yield washers ▪ particle deposit reduction and removal (hi-temperature)
Co-incineration of waste (coal, industrial, cement works, furnaces)	▪ slight retrofitting of existing firing places ▪ incineration control ▪ waste quality control (low-Cl)	flue-gas dedusting (electrostatic and/ or fabric filters)	raw material selection operating conditions	
Landfills	▪ waste control (non-haz, inert) ▪ sequential filling/capping ▪ gas collection and flaring ▪ acid digestion control ▪ leachate prevention	-leachate collection and recycling/ treatment	▪ improved waste control (low-Cl) ▪ replacing flare with gas motor ▪ gas, condensate and flare treatment ▪ leachate treatment	
Crematories		fluegas dedusting (fabric filters)		novel filters (e.g. biofilter)
Traffic	unleaded (non-scavenger) petrol	catalytic exhaust cleaning (new vehicles)	▪ reduced use ▪ low-emission drive mode	catalytic cleaning (all vehicles) by Dir 94/12/EC (EURO II)
Pulp and paper	▪ elementary Cl free (ECF) or total Cl free (TCF) bleaching processes ▪ black liqeur incineration	▪ basic-level incineration and flue-gas control in incineration of black liqueur, other liquids, solid residues and sludges	▪ extended ECF/TCF (e.g. O3) ▪ closed processes incl. waters ▪ reduced need for bleached prod. ▪ improved incineration (cf. above)	▪ improved flue-gas treatment ▪ improved effluent treatment ▪ reprocessing of bleached pulp
Mineral production (esp. clay bricks)	porosity promotion in tunnel kilns (carbonisating agents)	single dust separation	carbonisation gas recycling to firing zone	thermal or catalytic afterburning
Dye industry	hydroquinone instead of chloranil based dyes		hydroquinone instead of chloranil treated dyes (extended)	
Textile industry	▪ phaseout of PCP treated textiles		▪ hydroquinone instead of chloranil ▪ phaseout of PCP treated textiles	
Dry cleaning	▪ use of low-Cl solvents	▪ recycling and treatment of solvents	▪ cleaning of low-dioxin textiles (global control of production)	

of dioxins, thereby increasing instead of decreasing their emissions (e.g. Abad et al. 2002a).

- **Refuse-derived fuel** (RDF) combustion has been regarded as an improvement over traditional solid waste incineration also in terms of PCDD/F emissions. Low emissions have been achieved, also in co-combustion of RDF with wood, peat and other biofuels in both large and small-scale facilities (e.g., Vesterinen and Flyktman 1996, Hedman et al. 2005). This however depends on fuel quality and on combustion and emission control technology (Ragazzi and Sibisi 2003).
- Controls for emissions of PCDD/Fs and DLCs from **open burning** lag behind those for incineration facilities (e.g. Entec UK Ltd 2003, Neurath 2003). In general, waste composition, e.g. the content of chlorides, catalyzing metals and moisture, and burning conditions are key factors; the latter is related to the quality of ovens and hearths and improvements in such equipment.
- For control of PCDD/F emissions in **iron** and steel industry improved technologies are available. Emissions from sintering plants and electric arc furnaces can be influenced e.g. by the Cl content of the mix (Fischer et al. 2003, cf. Kim et al. 2003a,b). Emission reductions of up to 95 % from sinters have been achieved in only 2 years by massive efforts (François et al. 2002). In secondary steel smelters, Cl-containing impurities are a key to PCDD/F abatement (Detzel et al. 1998).
- **Non-ferrous metal** industry is more heterogeneous than iron industry, and thus presents continuous great challenges in PCDD/F abatement. Especially at secondary smelters, in addition to the quality of the scrap, the control of temperature profile, flue gas circuit and dust is essential, as demonstrated for Cu anode production unit by François et al. (2002). Adsorbents (e.g. with lime) can moreover be injected to complement fabric filters.
- **Pulp and paper** industry is important mainly in those countries and plants still using elemental Cl and old bleaching technologies. Additional abatement measures include the control of incineration of solid, sludge and liquid (e.g. spent liquor) wastes.

Secondary emissions

A considerable pool of PCDD/Fs, PCBs including dlPCB and other phased-out DLCs exists in products, materials and wastes and dispersed in the environment, causing secondary emissions. The relative importance of these will grow as primary emissions are increasingly controlled. The secondary emissions are partly overlapping with stockpile and hotspot control (see below).

Along with considerable potential, there are constraints in dioxin risk management in this area. The DLCs dispersed in the environment, even in as yet concentrated hotspots on land, may be difficult to remove and treat, both for technical and other reasons. On the other hand, also DLCs dispersed in products and wastes present challenges. Options depend on the availability of technological and societal systems for treating products containing or forming DLCs. Extensive and dispersed use (especially of consumer products intended for dispersed use) makes it difficult to capture them for appropriate treatment regarding DLCs.

Notable opportunities for prevention and control of secondary emissions include the following:

- Collection and appropriate treatment of PCBs in **construction sealants**, flooring and self-insulating glass. These represent a major remaining pool of PCBs (estimated at c. 100-500, 25 and 25 t, respectively, in Sweden), and include some dlPCBs. As they are present in concentrated form, removal can in principle be made efficiently. There are however many obstacles both in terms of information, economy and technology. The replacement costs have been estimated at several billion SEK (hundreds of M€), depending mainly on PCB sealant amounts. Additional costs are caused by treatment at hazardous waste incineration facilities. On the other hand, replacement of sealants is a necessary maintenance action at some point regardless of PCBs, and the additional cost due to PCBs is difficult to estimate. Removal of PCBs may thus be particularly feasible in connection with renovation of old buildings (SNV and Boverket 2002).
- Extended collection and treatment of **PCBs in other pools** such as capacitors and transformers in small electric appliances (cf. stockpiles management, below)

- Extended collection and treatment of other **DLCs in products and materials**, including e.g. remaining products containing PCNs and PBBs
- Increased collection and treatment of **products and materials containing PXDD/F precursors**, e.g. on the basis of management systems for hazardous waste streams (cf. stockpiles treatment, below). Some of this may take place within treatment of substances being phased out for other reasons, such as chlorinated solvents with ozone destruction potential based on the Montreal Convention.
- Collection and treatment of **leachates** and **waste gases** from waste disposal sites that contain DLCs, including fly and bottom ash disposal sites. Containment and treatment of emissions from ash disposal sites pose particular challenges as they contain considerable PXDD/F pools. For leachates, treatment on-site or off-site (in wastewater purification plants) may be applied. In the latter, DLCs are however dispersed in other sewage and sludges, and difficult to capture for efficient dedicated treatment e.g. of oily fractions that contain much of the DLCs. Treatment of gas emissions is applicable to those DLCs that have greater volatility or are emitted in sorbed form in gas and gas-transported particle streams. Treatment of this flux in practice entails e.g. high-level incineration of landfill gas condensates.
- For hazardous waste disposal sites, multi-barrier **high-level disposal technologies** are available. Many older disposal sites for hazardous wastes containing DLCs are of lower standard, approaching that of ordinary landfills and requiring additional control (cf. previous point)
- Collection and treatment of emissions from **recycling facilities** handling DLC-contaminated materials or involving processes where PXDD/Fs are formed, such as metals, waste oil and spent solvent recycling facilities.

Remediation of hotspots and management of stockpiles

General

Control of emissions from products containing or forming DLCs during all their use stages is in line with the emphasis in many recent developments, including the POPs Convention and EU procedures for existing substances, and in general with increased application of life-cycle management approaches. With PCBs, PCNs and many dioxin precursors including chlorinated pesticides, herbicides and other biocides, this particularly involves the management of stockpiles.

The emissions from waste management or other later stages in dioxin or PCB life-cycle extend the area of emission control from primary processes e.g. in industrial and other facilities to emissions already released but not too far dispersed to the environment. Important areas of emission control therefore are the cleanup or other controls at dioxin or PCB contaminated areas, either on land or in the sea (in sediments). Some risk management technology evaluations have emphasized this area of activity (e.g., UNEP 2004f). It has intimate connections with emission control technology in other, more closed and technological systems e.g. in facilities.

Treatment technologies for stockpiles and environmental hotspots have close similarities but also differences. Many options are also applied (and developed) for process and waste streams.

Stockpiles

Stockpiles include closed storages, uncontrolled dumps and adjacent environmental release areas, and may be defined to encompass even more generally pools in some products. Technological guidance has been produced for their treatment especially under the Stockholm Convention (UNEP 2000, 2004c-f). Some of these sources are repetitive and not based on clearly documented information that can be evaluated critically in detail. The continuous development of abatement technologies also makes it difficult to evaluate their applicability, efficiency and feasibility.

The control of stockpiles of products and materials containing DLCs is in some respects similar to and overlapping with environmental cleanup. However, in other respects stockpiles present particular problems, e.g. regarding measures on storage containers, structures and facilities. Stockpiles differ from environmental DLCs also through their generally higher concentrations. This can be an asset in treatment. Moreover, stockpiles most commonly include

PCBs and other chemical products containing PCDD/Fs, such as chlorinated pesticides and herbicides, and thus have different congener profiles than PCDD/Fs in many waste streams e.g. from thermal processes.

Stockpile management thus involves, in addition to removal and final treatment, sheltering, safeguarding, separating (with a view of avoiding unwanted reactions), repacking and pre-treating the products containing DLCs. Reduction of the volume contaminated is not as crucial as with DLCs dispersed in the environment. Some of these operations may be performed on site, some off-site. Considerable emissions of PCBs may arise from cleanup projects and storage sites also to air (Mills et al. 2002), and should be considered and controlled.

In addition to the types of stockpiles traditionally focused on, i.e. PCBs and biocides containing or producing PCDD/Fs (e.g., Pignatello and Huang 1993, UNEP 2004c-f), extensions are needed to include other DLCs, and in the scope of management to include prevention. This introduces great challenges with new types of stockpiles such as those containing brominated flame retardants and industrial chemicals potentially forming dioxins.

In the Baltic Sea region, the key stockpiles include the following:

- Obsolete stocks of chlorophenoxyphenyl herbicides, esp. 2,4,5-T and mixtures/ derivatives
- Obsolete stocks of chlorophenols, especially Ky-5 mixtures and PeCP brands
- Obsolete stocks of other commonly used chlorinated biocides such as lindane
- Stockpiles of PCBs both from electric and electronic appliances and other uses
- Stockpiles and storages of industrial chemicals potentially forming DLCs, e.g. chlorobenzenes
- Waste storage sites of industrial facilities that have typically caused considerable accumulation of DLCs, e.g. pulp and paper, metal and organohalogen production plants.

Control of such stockpiles is influenced and constrained by climatic and other natural factors (e.g., freezing, snow and ice cover, acidic and humic soils) and by social and technological factors (e.g., sparse inhabitation, inadequate infrastructure and low technological capacity in some areas).

An evaluation of treatment technologies for stockpiles of particularly obsolete pesticides in Central and Eastern Europe has been made by DANCEE (2004). Based on technology screening, this evaluation focused on container-based incineration (CBI) in hazardous waste treatment facilities, base-catalyzed dechlorination, gas-phase chemical reduction (GPCR), and cement kiln incineration. A variety of technological, environmental and economic criteria were considered and applied. It was concluded that GPCR and CBI were the most cost-effective and available technologies.

Hotspots remediation

Treatment of dioxin- and PCB-contaminated sites can be accomplished by several thermal, other physico-chemical and biological methods and by combinations of methods e.g. in treatment trains. Treatment may involve removal, separation, concentration and other pretreatment; destruction and transformation; solidification or stabilization; encapsulation or isolation of the contaminants. These methods can be applied *in situ*, on site or off-site. Many of the available technologies are restricted to seriously polluted sites, as stressed e.g. for dioxins in soil by Hashimoto et al. (2004). This is true also of contaminated sediment sites. Notable improvements have been made in both thermal treatment and chemical destruction (cf. Annex 10).

Methods for treatment of DLCs in organic liquids and wastewaters, sludges and soils are generally speaking more advanced than those for **sediments**. The costs can also be less favourable in treatment of sediments, due e.g. to the need for extensive dredging operations (FRTR 2004, cf. Khan et al. 2004). More information is available on treatment of PCBs than PCDD/F-contaminated materials (e.g., USEPA 2003a,b, NRC 2001, UNEP 1998, 2000, 2001, 2004f). On the other hand, the information sources addressing treatment of PCBs usually do not specify dlPCBs. Most documentation on full-scale treatment of dioxins in the environment originates from the US. The paucity of documents from European RTD and application in this area is reflected e.g. in the open scientific literature and the CORDIS databases.

In general, cleanup technologies for sites contaminated by PCDD/Fs and dlPCBs, although tried in laboratory, pilot and even

Table 33. Development status of demonstrated dioxin and PCBs treatment technologies applicable to environmental matrices, emphasizing full-scale applications, sediment treatment and destruction (based mainly on OTA 1991, NRC 2001, USEPA 2003a,b, FRTR 2004). Cf. Khan et al. 2004 and Annex 10.

Destruction technology type	Devel status[1]		Application to matrices		Cost data (or estimates), $ t^{-1} waste[2]
	D/Fs	CBs	Sedim	Soil/sludge etc	(for PCDD/Fs in solids or semi-solids); notes
Thermal					
Rotary kiln incineration, stationary	A[3]	A	(+ dehydr)	+ (solids)	900-1500
Rotary kiln incineration, mobile	A	A		+ (soil/solids)	400-600 (160-300 in XTRAC & triple-dryer trials)
Liquid injection incineration	B[4]	A		(+, pumpable)	(>500)
Fluidized bed incineration	B	A		+ (modified)	as for rotary kilns
Circulating bed incineration (mobile)	B	A	(+)	+ (high-airflow)	ca. 250
Advanced electric reactor/fluid wall pyrolys	C	C		(+, free-flowing)	(ca. 400-600)
Infrared incineration (mobile/semi-mobile)	C[3]	B		+	350
Plasma arc pyrolysis	D	C	(-, not thicker than motor oil)		NA (Westinghouse or other trials)
Supercritical water oxidation (mobile)	D	C		(+, slurry pumping)	(>500-<900; NA for solids-based slurry)
Thermal desorpt/Anaer Thermal Process	-	B	(+ w/ soil)	+	250-280
In-situ thermal desorption/oxidation	-	C-D		+	100-250 based on RT/TerraTherm field test)
In situ vitrification	B	C	(+ dehydr)	+	250 (1100 in New Bedford Harbour field test)
Ex situ vitrification (glass furnace)	B-C	B-C	+	+	40 based on Minergy/Hazen bench-scale test)
Non-thermal					
Chemical dechlorination/APEG(-PLUS)	C	A	(-)	+ (in situ/slurry)	ca. 300 *in situ*, 100 slurry; 100-800+ APEG-PLUS
Base-catalyzed decomposition	D	D	(+, bench)	+	(100-250)
Basic extraction/Solvated Electron Tech	(C)	C	+		700
Thermal-gas phase reduction dechlorin.	C-E	C	+	(+, aqueous)	c. 350-500; 600 for N Bedford Harbour w/ PCDD/Fs
Thermal desorption/UV destruction	(C)	C		+	NA
Solvent extraction to incineration/destruct	-	D		+	NA (remote cold location trial)
Bioremediation	D	C	+ (<DRE)	(+, met PCBs limits)	(160 slurry/aerat; generally considered inadequate)

Explanations: [1]A=operating system has been built, tested, permitted and used on a site cleanup; B=system has been built and tested but not permitted or used on a site cleanup; C=pilot plant has been built and tested with waste material; D=laboratory or bench-scale tests have been completed; [2]With standard USEPA-specified Destruction/Removal Efficiency of 99,9999 %; [3]More experience in Europe; [4]Sea-based system.

field scale, are constrained by the **efficiency** of removal and destruction, including the control of environmental releases and human exposures, by energy needs (especially for thermal processes), by infrastructures and know-how, and therefore by costs. Also the social acceptance and administrative procedures for site cleanups may constitute great obstacles.

In particular, the **costs** of remediation depend on the destruction or removal efficiency (DRE); if the standard USEPA remedial goal of 99,9999 % DRE (for PCDD/Fs and PCBs) is to be achieved, excessive costs of large-scale remediation may result. If a lower DRE is accepted, other technological solutions may suffice and costs may be much lower. Secondly, the inclusiveness of costs is an important factor. Full costs including e.g. dredging (or other removal), transport and storage, treatment and disposal of residues as well as planning, development, testing, monitoring, documenting and administration of solutions may exceed several-fold the costs of destruction operations only.

With **thermal treatment**, the formation of toxic DLCs constitutes an important limitation. However, several thermal systems and combined systems utilizing multi-stage desorption and destruction and even vitrification techniques and (intermediate-temperature) supercritical water extraction have been shown to be able to treat PCDD/Fs or PCBs efficiently in full scale (Table 33, cf. de Percin 1995, Kasai et al. 2000, Yak et al. 2000). Some methods have been successfully applied also to sediments and at costs much below those incurred at hazardous waste incineration facilities.

Chemical treatment (extraction, destruction or stabilization) provides notable abatement options (Table 33). Alkaline dechlorination has been especially applied (desRosiers 1989, Peterson and Milicic 1992). It has been modified (e.g., Oku et al. 1995) and tried also for treatment of sediments (Chen et al. 1997, cf. Annex 10, e.g. Table

10A4). Moreover, photolytic processes (Isosaari et al. 2001, Poster et al. 2003) and even radiolytic processes (Gray and Hilarides 1995) have some potential, depending on contaminants, matrix and conditions. However, as stressed by Weber (2004), several non-combustion technologies including base catalyzed and other reductive dehalogenation processes as well as oxidative treatment technologies such as supercritical water oxidation involve the possibility of PCDD/F formation. This author noted that for almost all POPs destruction technologies an evaluation of PCDD/F formation in dependence of operation conditions is missing and sound evaluations are available only for the high-temperature destruction in incineration.

Bioremediation is attractive in principle as a nature-emulating solution to DLC treatment. Bioremediation has been mainly tried for anaerobic dehalogenation of PCB mixtures. Some progress has been made with treatment of sediment PCBs (Ye et al. 1992, Pagano et al. 1995, cf. Tiedje et al. 1993/94 and Annex 10). It has been applied also *in situ* (Deweerd and Bedard 1999) and on destruction of PCB 126 and other dlPCBs (Mousa et al. 1998). However, few full-scale systems have evolved, and reduction efficiencies are unclear and generally low (Table 33). Efficient bioremediation systems are still less developed for PCDD/Fs (Habe et al. 2002), especially highly chlorinated congeners (e.g. Du et al. 2001). Specialized bacteria able to degrade highly halogenated dioxins have been isolated (Bunge et al. 2003) but their potential seems limited, even by use of bioaugmentation (e.g., Barkovskii and Adriaens al. 1996, 1998). Reduction of PCDD/Fs (TEqs) of up to 70 % in contaminated soil has however been reported (Souta et al. 2004, Ha et al. 2004, Annex 10). Some promise has been shown by white-rot fungus (Yadav et al. 1995, Takada et al. 1996) and activated sludge based systems (Kao et al. 2001, cf. Wittich 1998).

Within management of PCDD/F and PCBs emissions to aquatic environments, **sediment cleanup**, removal or safeguarding (solidification/stabilization or isolation) has been in focus. The NRC (2001) strategy for sediment PCB management is a notable example of such technology evaluations. Also the Great Lakes initiative and the ensuing strategies for POPs abatement include extensive sediment management elements (see Annex 12). At a strategic level for the protection of the Baltic Sea, a key question is what the relationship of these controls is with land-based emission abatement and cleanups. It can be argued that technically, economically and environmentally, large-scale sediment cleanup is exceedingly difficult and can thus be prohibitively costly. This was pointed out also in the 2002 strategy for the Great Lakes (USEPA 2004 (2002)), noting that only one of 43 Areas Of Concern had yet been remediated, and that the cost of the projected operations, 200 M$, could be orders of magnitude higher. A particular challenge with sediment remediation is the need to limit the transport of contaminated materials and sediments in general. This may require great efforts and special technologies and yet cause considerable disadvantages, even additional and new risks, and thus offset benefits of remediation.

The **time** dimension of remedial solutions is important, both for impacts and technical performance as well as for costs and overall management. With isolation and stabilization solutions, the long-term certainty presents a key problem (e.g., OTA 1991, USEPA 2003b).

Initial listing has been produced of hotspots around the Baltic Sea that may be relevant with regard to emissions and control of DLCs, especially PCDD/Fs (Annex 3F). This listing is partly based on the HELCOM list of hotspots. These are mainly concerned with municipal and industrial discharges of nutrients and oxygen consuming organics, and have included dioxin hotpots only marginally (e.g. metal and pulp and paper industries). Therefore, this list has been used only selectively and augmented by additional information specifically on hotspots that are known or strongly suspected to contain dioxins. These include many chloralkali factories and sawmills. Abatement measures for such sites also with a view of emissions to the Baltic should be specifically developed and applied after careful deliberation.

Dioxin hotspots exist both on the coast and in inland areas of the Baltic Sea catchment. The listings produced in the present work have been focused on the former, as they can have more direct emissions to the sea, in some cases already having emitted considerable loads to marine sediments. However, much of the cumulative pool of PCDD/Fs and dlPCBs in the Baltic Sea catchment are in soils, wastes and products (Bergqvist et al. 2005) and thus in inland hotspots, including industrial areas and landfills. These pools and emissions are easier to treat or contain than those on the coast and especially

those already dispersed in sediments. Therefore, for management strategies especially in the long term, also inland hotspots as well as inland point sources need to be included.

7.2.3 Interception of transport to the sea

Several measures may be and are taken between the emissions of dioxins from the various facilities and processes of formation and their entrance in the Baltic Sea, in various branches of material flows. The scale of such measures vary; some may be directed to extensive processes in the ambient environment, while others are undertaken in closed or semi-closed technological systems such as waste-water purification plants.

Many of the potential options in this area have been established to control unwanted fluxes of solids and other materials to watercourses (and other recipients) with regard to other substances such as nutrients and other contaminants. Their application to dioxin risk management may require some modification; the foci of the processes may also differ. Nevertheless, dioxin control and the control of associated other substances and materials may have sufficient shared interests to allow efficient abatement solutions also with regard to costs. As with many other solutions, e.g. those in emissions control, it is in many cases largely a matter of raising awareness of the needs and possibilities to take into account dioxins in designing and operating systems that are primarily intended for other purposes.

Scavenging dioxins in **wastewater** treatment is an important area of control measures, as much of the dioxin fluxes from the catchment, also of those originally airborne, are deposited on surfaces connected with sewers for wastewaters or storm water (cf. 3.2). Moreover, the reduction of dioxin emissions from industrial and waste facilities will cause their disposal (in ash, slag and sludge) in waste deposits that are increasingly directed to sewers as well. Thus, the relative importance of the flows of dioxins in sewerage systems is likely to increase. As storm water is usually and increasingly separately diverted in Baltic Sea countries, the treatment of settled solids from these sewers, normally allowing and achieving less efficient retention than in wastewater treatment, is of importance.

The wastewater treatment process used affects the control of dioxins. Increased solids removal will increase dioxin removal, but also the type of process (active sludge or combined biological-chemical precipitation, and auxiliary processes for N removal) has an influence e.g. through flocculation and binding in biomolecules. The BAT of wastewater treatment generally contributes to improved dioxin retention in sewage (cf. Eduljee et al. 1997), and the upgrading of wastewater treatment plants especially in new member states will be important to stall the total dioxin fluxes to the Baltic. Also the sludge treatment processes (anaerobic digestion, aerobic composting, or others) influence dioxin retention, and some of these are difficult to optimize for this purpose. For instance, anaerobic and semi-anaerobic waste treatment processes may induce formation of DLCs, including OCDD (Klimm et al. 1998), and composting has been reported to form dioxins (Krauss et al. 1994).

Preliminary results have been published (Nakamiya et al. 2000) from laboratory-scale trials of PCDD/F removal, supposedly by biodegradation, in an activated sludge process, giving surprisingly high removal (99,9 % of TEq), e.g. compared to the reductive halogenation of TCDD in activated sludge microcosms (maximal reduction of 86 %, Kao et al. 2001). The removal rate remained high in continuous culture, suggesting the method has potential in large-scale application. This might have the added value of utilizing existing treatment systems. The process requirements and modifications needed in sewage purification plants and other constraints have however not been elucidated, and no later information has been found.

In a wider dioxin cycling and balance perspective, although dioxins are scavenged in wastewater treatment, their subsequent recycling in **sludges** to agricultural, horticultural and other such areas (including urban greenery areas, roadside cultivations and silviculture) will again involve the possibility to either bioaccumulation in terrestrial food-chains (up to humans) or transport e.g. in runoff to the Baltic Sea (e.g., Fiedler 1996).

DLCs in **landfills** (including disposal sites for fly-ash) are an emission source especially in the long term (Assmuth 1995, Bergqvist et al. 2005). Some progress has been made in DLC removal from landfill leachate, especially by alkaline chemical dehalogenation (Tiernan et al. 1989) and also by coagulation (Toji and Kusuda 2002). Leachates are usually also treated in wastewater purification plants (see above).

Dioxins already dispersed in inland **watercourses** may be scavenged by sedimentation basins constructed for other purposes (such as solids and nutrient retention) and through careful disposition of dredge spoils from such basins and from particularly dioxin-rich sediments.

Land-based direct discharges of DLCs to the sea, especially from major industrial point sources but also municipal point sources and diffuse sources, need to be better controlled. More extensive and efficient treatment and prevention of wastewater discharges is called for in particular in those Baltic Sea countries that have recently joined or are outside EU. Among different types of discharges, those from coastal plants of metals, chemical and petrochemical (including oil harbors) and pulp and paper industries may be noted. Interception of DLC emissions is in practice dependent on and linked with the removal of suspended solids, as explained above in connection with wastewater treatment.

7.3 Measures in the sea

7.3.1 Control of emissions and influxes of dioxin-like compounds in the sea

Emissions

Control of anthropogenic emissions into the open sea involves mainly restriction of emissions from **shipping** (cf. land-based direct emissions, above). Shipping has potential importance mainly through accidental releases in connection with transport of dioxin precursors and carriers, particularly mineral oil and halogenated chemicals, instead of regular emissions from vessels. The risks from oil shipping have increased and are projected to continue increasing due mainly to growing Russian shipping. Management of risks from oil and chemical spills has a high priority in national and international (e.g. HELCOM) activities to protect the Baltic. However, additional measures, enforcement and steering instruments are needed (cf. 8.4).

Sediment control

Reduction of resuspension of dioxins from Baltic Sea sediments is a measure of risk reduction that is intuitively promising and has attracted some interest. Reduced resuspension may be desirable also for other reasons to reduce adverse impacts of the sediment on the Baltic marine ecosystem (and adverse impacts on sediment ecosystems themselves). Specific measures to control this dioxin source, based on sediment-impacting and resuspending agents, processes and factors (cf. 2), include the following:

- Restricting bottom trawling (cf. Hansson, oral communication 2003)
- Reducing dredging of waterways for shipping, e.g. restricting it to areas that are not heavily dioxin contaminated
- Reducing dredging for use of sand/gravel and possibly extracting metal-rich nodules in sediments
- Restricting ship traffic causing excessive wave action and bottom disturbance in archipelagos
- Reducing and steering the installation of bottom cables
- Restricting or controlling and steering pipeline construction, e.g. that planned for transport of Russian gas through the Baltic to Central and Western Europe.

7.3.2 Reduction of the pools of dioxin-like compounds in the sea

There are several possibilities to reduce the pools of DLCs in the Baltic and subsequent risks to exposed organisms, e.g. fish consumers. In particular, fisheries offer options. However, also other means are conceivable, by both biological and other techniques. Potential measures include the following:

- Concentration and removal in fish catches by '**cleanup fishing**' has been proposed as a means of removing PCBs from the Baltic by Hansson (unpublished 2003) and Mackenzie et al. (2004), supported by total PCB budget estimates based on data for 1980's and early 1990's (cf. 3.4); sprat was calculated to contain the bulk of the removable pool in the major economy fish species. While it is uncertain whether fish catch represents as large a share of the pools or fluxes of PCDD/Fs or even dlPCBs (cf. 3.3), this load reduction may not be insignificant. It is one of the means to reduce these compounds that is technically and economically rather easily implemented, as it can be combined with continuous fishing using equipment and practices already in place. Continuous

fishing in the dioxin-rich stocks, even though not consumed by humans or production animals, may be desirable also for the control of dioxin cycling through population structure (see below). Constraints of this option include the needs and difficulties in treating and disposing (and preferably utilizing) the 'junk fish' biomass in a manner that is both efficient and does not cause harmful environmental and other impacts (e.g., composting, other processing).

- Restriction of the recycling of **fish discards** back to the sea has been proposed especially for cod livers (that can no more be sold for human consumption) as they may contain a not insignificant amount of DLCs (Mackenzie et al. 2004).
- **Degradation** of DLCs in marine sediments by direct measures is constrained by low biodegradation rates in nature and by the efforts needed to greatly increase them *in situ*. Laboratory trials of bioremediation in sediments have usually achieved modest reductions especially of the highly chlorinated 2,3,7,8-congeners (e.g., Kao et al. 2001, cf. 7.2.2). Their hydrophobicity, persistence and toxicity limits biodegradation (see e.g. Beurskens et al. 1995). Soil bioremediation systems (see above and Annex 10) may not be readily applicable to sediments especially *in situ*. These constraints may be more prohibitive in marine sediments than in lake and river sediments that can be more efficiently contained in both *in situ* and *ex situ* (on-site or off-site) solutions (see also evaluation of sediment PCB management by NRC 2001).
- Removal of dioxins by **dredging** and subsequent treatment of the spoil, especially in maintenance dredging but also solely for dioxin removal in contaminated hotspots, provides a cleanup option. However, it is difficult to remove DLCs once they have entered the sea, due to dispersal and other marine conditions. The feasibility of this option may be limited to dredging operations taking place also for other reasons and to some coastal hotspots (cf. above and Annex 10).

7.3.3 Biological steering of the accumulation of dioxin-like compounds in marine food chains

Selective fishing and other fisheries management measures

While the potential of fisheries in removing dioxins and PCBs from the Baltic has been proposed in some connections (see above), the use of fishing to control bioaccumulation of dioxins has received little attention as a risk management approach for Baltic Sea dioxins (or elsewhere). Fisheries are centrally involved in the formation and abatement of dioxin risks in other respects, as carriers and suppliers of dioxins, targets of regulatory restrictions, and indirect targets of diet advisories.

It has been held that the present knowledge is too superficial and limited to enable planning for alterations in food webs, e.g. for fisheries management purposes (Ojaveer and Lehtonen 2001). Altering fish stocks and their population structure is however inherent in fishing. Such measures are used also to ensure sustainable stocks, increasingly taking into account interactions of species, e.g. under ICES. Biological steering of aquatic systems has moreover been considered and tried for purposes other than fish production, such as to control nutrient cycling and diminish eutrophication and to restore aquatic ecosystems, mainly in lakes; it has also been proposed as a management approach to the trophic state of the Baltic. In principle, fisheries management may similarly be utilized to decimate dioxins in the Baltic. Specifically, e.g. the following measures of fisheries-based biological steering may be considered for reduction of dioxin risks through fish, mainly to human health:

- Biomagnification control e.g. by adjusting cod stocks and predation on herring
- Catch size selection e.g. by fishing small herring and sprat and fishing gear specifications
- Area selection and designation of high or low intensity fishing areas, and geographical restrictions
- Catch time selection e.g. by reducing spawn time and spawn area fishing.

Mariculture

An important category of management options is the control of DLCs in mariculture. As described elsewhere (3.4, Annex 7), PCDD/Fs and dlPCBs

accumulate in Baltic rainbow trout fed Baltic herring (Isosaari et al. 2002b). The same is seen in Atlantic salmon (consumed in large quantities also in Baltic Sea countries) given feed based on fish oil from the Baltic (Isosaari et al. 2004). On the other hand, common dry feed brands used in Baltic Sea aquaculture contain relatively low levels of PCDD/Fs and dlPCBs, and have not resulted in elevated levels in the produce (Isosaari et al. 2002b). This provides the option to reduce contamination of Baltic fish-based food production systems by adjusting feed types.

Jacobs et al. (2002a,b) pointed out that vegetable oil based feeding-stuffs (with n-3 and n-6 fatty acids) offer also other advantages than lower contaminant levels over fish oil based aquaculture feeds, referring to the evidence that such vegetable feeds can accommodate more successful seawater adaptation and natural-like diets than diets based on marine fish, and could thus offer economical high-energy feeds while at the same time reducing contamination of human food chains. In principle, such an approach may be seen as a parallel to the increasing use of dietary supplements based on vegetable instead of fish oils in human nutrition (see below). Friesen et al. (2005) who studied the use of flaxseed oil to reduce PCB loading in farmed sablefish (a fatty species) reported that the WHO-TEq_p level could be reduced only 30-40 % as compared with anchovy-based feed. However, if compared with feeds fortified by herring-based fish oils, the relative reduction would be greater (depending on the levels in the final feed and on the conversion efficiency of the cultivation system).

Bethune et al. (2005) reported that although a rapeseed oil diet resulted in a WHO-TEq level in farmed salmon 3-fold lower than a fish oil based diet, patients who consumed 700 g per week for 6 weeks of rapeseed oil fed salmon had similar or even slightly higher plasma levels of PCDD/Fs and PCBs; circulating plasma levels of PCDD/Fs and 0-*ortho* and 1-*ortho* PCBs were reduced c. 40 % by fish oil based diet. This reflects the fact that short-term changes in DLC intakes are insignificant for body burdens. On the other hand, the rapeseed oil based salmon diet resulted in slightly lower plasma levels of n-3 PUFAs in the patients (markedly lower PUFA levels in salmon fillet). Thus, considerable benefits were not provided to these patients by rapeseed oil based salmon culture methods either through PUFAs or lower contaminant levels (this contrasts with the evaluation of the authors, see also Seierstad et al. 2005). The targeted short-term use of fatty fish based diets in high-risk groups such as coronary patients, considered to offer the greatest and most certain health benefits from consumption of fatty fish (cf. 4.4, 5), may thus not require fish cultured with very low-contaminated feeding-stuffs. For longer-term exposures the needs for and impacts of fish diets are different, also regarding DLCs. As stated by Bethune et al. (2005), the margin of safety of such patients may be lower also for DLCs. Nevertheless, also in such populations the efficiency, effectiveness and other characteristics and impacts of alternative fish culture and consumption schemes require additional analyses (cf. 8).

There are additional considerations in the needs and impacts of (sea) food production systems utilizing such alternative feeding-stuffs. It is not well known what the effect on contaminant burdens of consumers or their intakes of beneficial fish ingredients would be from consumption of such fish. Although these considerations are related to management measures in the sea, they have more to do with impacts of alternative food production and nutritional systems, and will therefore be discussed below (8).

Eutrophication control

Important links exist between trophic state and dioxin levels. Eutrophication plays a fundamental role in the ecology and economy of the Baltic, and should be considered as a factor and partly a boundary condition also in dioxin management.

It has been sometimes implied that as dioxins are present in higher concentrations in smaller biomass, the reduction of the trophic state and the eutrophicating nutrient load of the sea could lead to still higher dioxin levels in fish and thus increase risks from fish consumption. Consequently, continuous eutrophication might be considered as a means to reduce risks from dioxins based on 'fat dilution' in biomass (cf. Berglund et al. 2001). It also seems that community nutrient status may beneficially affect the magnitude of some toxic effects and the rate of recovery (Lozano and Pratt 1994).

However, the logic behind such measures to control DLC risks rests on simplifying and potentially misleading assumptions of the process of eutrophication and its variations as well as of its linkages with dioxin bioaccumulation (cf.

2.3.2, 3.3.1, Hildén and Assmuth 2003a,b). The relationships between trophic state and dioxin biomagnification are complex. It is not likely that eutrophication in the presently dominant form of cyanobacteria blooms will lead to or be directly linked with higher biomass (and fat) and lower dioxin levels in the key economic species, herring. There are some indications to the contrary, i.e. eutrophication may instead counteract herring growth and fat content and thus increase dioxin levels in them (ICES 2005c). The herring stocks and their condition may depend to a much greater extent on other species and on still other dynamic factors, including the population structure that is controlled in part by fishing (although not yet to minimize dioxin levels). Berglund et al. (2001) concluded that biomass dilution was an unlikely cause of the observed PCB dynamics in lake plankton, particulate settling being a more plausible explaining mechanism. Likewise, regional variation in the ecosystem dynamics should be accounted for. In general, eutrophication influences DLC bioaccumulation, fate and risks in various ways and directions. These need to be taken into account in simultaneous control of dioxin risks and eutrophication (cf. 8.4.3)

7.4 Measures on fluxes of dioxin-like compounds after catches from the sea

7.4.1 Reducing intakes by food advisories and other means of information steering

Food advisories are a key measure in the management of human health risks from fish contaminants, including dioxins, notably in the Baltic Sea countries like Finland and Sweden where this management task has been approached largely by such means, instead of bans on trade of contaminated fish. Fish food advisories are used as a management tool also in other contaminated watercourse areas, such as the Great Lakes where dioxins, PCBs and other POPs have likewise been the main concern. Food advisories have been traditionally used to respond to risks from mercury contamination in fish, notably in Sweden and Finland, as well as in other areas of food safety.

All Nordic countries have generally and actively recommended dietary consumption of fish, based on its health benefits. This has been made even when additional (targeted) considerations and recommendations for avoidance of contaminants in fish have been included in dietary advice. In addition to Sweden and Finland, Danish dietary guidelines have among the seven basic recommendations advised to eat fish and fish products often (Haraldsdottir 1999). Such recommendations have also been given by NCM (1996). In Swedish advisories attention has been paid to toxicological risks from fish consumption; these advisories have since 1995 specified roughly the catch area (Baltic Sea, other sea or inland), in addition to fish species and part of the fish (Lind et al. 2002, Darnerud et al. 2003).

In other countries, the advice in UK by SACN and COT (2004) is particularly notable, being based on extensive evaluations of both the nutritional evidence (for benefits from fish) and on toxicological information. Although a quantitative risk-benefit analysis of fish consumption was not considered possible (cf. 5.3), the previous recommendation of consuming at least two portions of fish weekly, one of them oily, was generally upheld and amended. Women of reproductive age and girls were advised to aim at consuming 1-2 portions of oily fish a week, while the recommendation for men and boys was 1-4 portions a week. Specifically with regard to the time dimension, it was advised that a woman who had not consistently exceeded the guideline range previously could increase her oily fish consumption throughout pregnancy and lactation above the guideline range without detrimental effects, because of the lag in accumulation of DLCs in the body.

Renwick et al. (2003) distinguished and systematized qualitative advice in reducing risks from chemicals in food and diet, to complement quantitative advice e.g. in the form of risk estimates, margins of exposure and TDIs. Some of these forms of qualitative advice may include e.g. the following aspects in the case of dioxins in fish:

- *Use-specific approval:* e.g., recommendation of fish consumption despite contamination for particular health reasons
- *Avoidance of certain foods:* an important form of advice (see below in more detail)
- *Avoidance of certain processes:* e.g., cooking advice (cf. below 7.4.3)

- *Modification of production processes:* this may be seen as advice not to consumers (cf. 7.4.3)
- *Reduction of intakes:* related to avoidance of certain foods (cf. above), and implies e.g. advice of the frequency of consumption
- Reduction of intakes by *sectors of the population:* e.g., age and gender specific advisories.

Targeted advisories may be provided for children and kindergarten or school personnel, girls, young women, other women of childbearing age, fishers, mother's milk use and other animal fat (see e.g. Ponce et al. 2000, Darnerud et al. 2003, Ashizawa et al. 2005).

Factors that may be specified in dietary advice regarding DLCs in fish include the following:

- Fish **species:** advice can be given to avoid consumption of Baltic herring and salmon; in a more positive vein, a mixed fish diet including several species can be recommended
- Fish **sizes:** An important specification to discourage especially the use of large herring accumulating high concentrations of DLCs, as has been recommended in some connections
- Catch **areas:** the use of fish from particularly contaminated areas can be discouraged; also the mixed consumption of fish from various areas including less contaminated areas may be encouraged, in accordance with the present advisories in Sweden and Finland
- Fish **parts:** the use of herring skin fat, the anterior parts of salmon (and trout) and the livers of fish (e.g. of burbot in addition to cod) and other tissues with high dioxin contents can be discouraged
- **Frequency of consumption:** quantitative recommendations have been issued in Sweden and in Finland on how often (e.g. no more than once a week) fatty Baltic fish ought to be consumed by particular risk groups such as women who are pregnant or in reproductive age.

Ponce et al. (2000) discussed that in order to balance risks and benefits from fish consumption and to account for the asymmetric distribution of risks and benefits in the population, a plausible alternative to population-wide fish consumption advice would be to target fish intake advisories to women of childbearing age. This has been made already for some time by food authorities in Sweden, and ever more explicitly also in Finnish advisories. In addition, complementary generic information may at some level be considered. For instance, the healthiness of fish in general, and the hazardousness of dioxins in general, may be communicated to the public at large.

Regarding the long-time prominence of such information steering, little studies have been made of the reception, efficacy and conduct of such advisories. Tilden et al. (1997) concluded on the basis of a survey of >8000 adult residents in the Great Lakes area that despite extensive education efforts only half of the sport fish consumers were aware of advice on fish use and preparation, and even fewer changed their behavior on the basis of the advice. Awareness was lower among women, indicating the need for targeted information. The receptiveness of citizens in Baltic Sea countries may be different but limitations for dietary advisories do exist. Connelly and Knuth (1998) found that multiple formats are required to meet the needs of a significant percent of anglers for fish-related risk information. Advisories should be geared to abilities of the target audience: for many audiences a combination of qualitative and quantitative information and a combination of diagrams and text may be most effective; for most audiences a cajoling rather than commanding tone better provides them with the information needed to make a decision about fish consumption.

7.4.2 Reducing intakes by regulating fish marketing and by associated product labelling

Regulation of the quality of marketed food and feeding-stuffs is a key instrument in the present EU strategy on dioxins and PCBs and involves rather detailed prescriptions particularly concerning limit values of PCDD/Fs in fish and fish-based products, and monitoring and reporting (cf. 6, 8.3). These measures have not been thoroughly analyzed in relation to alternatives. Impacts of these measures have been mainly assessed from the point of view of their legal basis, and certain socio-economic implications and conflict potential (cf. 8.3).

Information to consumers in the form of **labelling** and other documentation may be desired in connection with both market regulation and diet advisories or other information steering measures. Labelling may concern various properties of fish, notably catch site. 'Ecolabelling' of fish in the Baltic Sea area has been considered mainly for the

purpose of reducing the risks to threatened fish stocks and for minimization of unwanted effects on the (marine) environment from fisheries and aquaculture. Labelling of fish according to its estimated contamination levels has been applied in the Great Lakes area where fish have been designated to 'green', 'yellow' and 'red' categories (cf. Annex 11). Labelling has also been proposed for identification of farmed salmon (based on the relatively high levels of PCDD/Fs, PCBs and other organochlorides in farmed oceanic salmon) and of the country or region of origin (Hites et al. 2004a). Foran et al. (2004a) likewise noted that such labelling would be important for consumers to enable informed purchases of marketed salmon.

Labelling may be developed and specified further, e.g. as to the scope (possibly including dlPCBs), targets, timing, regions, accompanying measures and other respects. Some labelling and other information on the quality and characteristics of fish may be a natural complement to diet advisories.

7.4.3 Isolation and treatment of dioxin-like compounds in fish and fish products

Technical-scale processing options

DLCs can be removed from fish fats, either by removing parts known to be fatty or by treating the whole fish. The latter option is most important in the case of fish not used as such for human consumption.

The evaluation of Joas et al. (2001) pointed to the availability of technology to cost-efficiently remove dioxins mainly from fish **oil**. Fish oil comprises most of the load of DLCs e.g. in aquaculture feeding-stuffs (SPCFC 2005) and removal of these compounds may thus significantly reduce the risks to subsequent uses. However, SPCFC (2005) considered that the efficiency and feasibility of these procedures still need to be demonstrated. Fernandes et al. (2003) on the other hand proposed that the apparent decline in DLC levels in fish oil based dietary supplement products (see 7.4.4) probably reflects purification in manufacture rather than declining levels in original oil.

Information has been presented on treatment of fish oil mainly by activated carbon (AC) using low-temperature stripping or short path distillation (Breivik and Thorstad 2004, De Kock et al. 2004, ref. by SPCFC 2005). De Meulenaer et al. (2003) published results on contaminated fish oil from the Baltic demonstrating that AC could reduce 80-100 % of PCDD/Fs and coplanar PCBs. The removal efficiency toward 1-*ortho* PCBs was only c. 30 %, and seems to constrain the possibilities to reduce risks from total TEqs in fish oil e.g. to feed production. Eppe et al. (2005) likewise showed that PCBs contributed most to the residual WHO-TEq_{DFP} levels in fish oil. The process could be optimized to reach levels below EU limit values (for food), using AC contents of 0.25 %, temperatures of 50 °C, and treatment times of 20 min.

Joas et al. (2001) evaluated based on industry consultation that the removal of DLCs from fish **meal** is presently not economically feasible. Thus, dioxin removal from whole fatty fish products and particularly fishmeal remains as a longer-term opportunity (cf. SPCFC 2005).

Cooking and preparation

Fish retailers, consumers or cooks process the fish consumed in various ways that influence the exposure and risk. Thus, cooking and preparing methods have relevance for risk reduction.

Several possibilities to reduce exposure to DLCs and other contaminants in fish have been explored (Zabik et al. 1996, Zabik and Zabik 1995, 1999, cf. Annex 10). These include:

- Trimming of fish, especially by removing skin (including fat, particularly in herring) and body fat including belly flap (see e.g. Kris-Etherton et al. 2003, Foran et al. 2005a)
- Cooking methods and temperatures, e.g. by using grills in cooking
- Smoking that leads to particularly high reductions (up to 50 %) in PCB levels (in lake trout), but causes PAH formation (Zabik et al. 1996).

The applicability of these options depends on both technical abilities and equipment and on information and training to those preparing food, including retailers, cooks and consumers. Such measures may also rather naturally be combined with advisories for the use of fish, e.g. with regard to the part of the fish to be selected.

Diversion of dioxin-contaminated fish to insensitive uses

Instead of using the most contaminated fatty Baltic fish to human consumption, it can be

reserved to other beneficial uses, such as fodder and feeding-stuff, either as such or in processed form. This applies mainly to large-size herring and sprat. Vartiainen et al. (1997c) mentioned this as an option for reducing the toxicological risks in Baltic herring to human consumers.

This option would mean that part of the catch would be sold at lower price than can be obtained by selling it to human consumption. Thus, there are economic consequences in implementing such a risk reduction option (cf. 8.3). Baltic herring was given to non-human animals, mainly fur animals, already in 1970's due to low consumer demand or to deficiencies in the market chain, even though this was not prompted (at least to the degree presently) by fears for dioxins in herring.

The risks from Baltic fish dioxins through production animals fed on such fish can be further reduced by selecting the animals to be given Baltic fish. As minks and chickens are sensitive to DLCs (cf. 4.3) and other animals used as human food (such as pigs) would cause accumulation to humans, fur foxes may be a noteworthy alternative (cf. Vartiainen et al. 1997c), mainly in Finland and Sweden where fur animal farming is common. This would require appropriate treatment of the residues (excreta and carcasses) from these animals.

Dilution of dioxin-rich fish to obtain high- and low-dioxin raw materials

As discussed by Joas et al. (2001), this option is questionable both in principle and on practical grounds.

7.4.4 Exposure reduction by surrogate and supplemented diets and other means of protection

Surrogate diets for humans

Human exposures to dioxins can be reduced by shifting to another diet, including other fish. This in effect is the result of diet advisories recommending lower consumption of certain foods. However, also actual surrogate diets may be used. The provision of surrogates may increase the efficiency and health benefit of avoiding contaminated fish; they may also ensure that no harmful side effects of lower fish consumption are caused.

Surrogates are related to policy considerations in risk-benefit assessment, and to the acceptance and adoption of risk reduction measures by consumers (cf. 8). Because dietary habits and their determinants and effects are complex, avoidance of fatty Baltic fish consumption may lead to impoverished diets and to the substitution of dioxin-laden fish in diet with less dioxin-laden but in other respects unhealthier dietary items. This shift in diet may be assumed to take place more automatically than actually is the case. Instead, such harmful dietary changes may be prevented or at least alleviated e.g. by advice and also by other incentives and steering instruments and measures. Therefore, targeted dietary surrogates may be used; some of them potentially offer the possibility to both reduce dioxin intake and ensure the health benefits from fatty fish.

Huisman et al. (1995b) concluded that decrease of exposure to PCBs and dioxins of the fetus and the neonate probably requires long-term reduction of intakes, but that in Dutch conditions, substitution of normal cheese by low-fat cheese and the use of vegetable oils instead of fish oils in the preparation of foodstuffs by the food industry could reduce the intake of PCBs and dioxins. They cited data showing that, due to the development of world market prices, the contribution of fish oils to industrial oils e.g. in the Netherlands had already decreased from c. 18 % to 5 %, concluding that a total abolition of the use of contaminated fish oils in industrial oils thus appeared feasible, as it was calculated to allow a 22 % reduction of PCB and dioxin intakes. These authors also stated that the beneficial effects of fish PUFAs disappear during the oil preparation process due to hydrogenation (cf. above, contaminant removal from fish oil), but did not provide any information to substantiate this statement.

The risk-benefit analysis by Foran et al. (2005b) suggested that although the risk from contaminants is partially offset by health benefits from salmon consumption, young children, women of child-bearing age, pregnant women, and nursing mothers not at significant risk for sudden cardiac death but concerned with health impairments such as cognitive and behavioral effects can minimize contaminant exposure by choosing the least contaminated wild salmon or by selecting other sources of n-3 PUFAs.

Information has not been found on how much fish oils and particularly oils from Baltic

fish are used in the industrial preparation of foodstuffs sold and consumed in the Baltic Sea countries, thus adding to the exposure of the high fish consumers in these countries. Baltic sprat is reportedly processed e.g. in Baltic States and Russia to fish oil that may be used also in feeding food production animals.

Surrogates to other fish predators

A successful surrogate diet for non-human animals exposed to DLCs in Baltic fish is the feeding of white-tailed sea eagles by clean food such as pig carcasses from ice in winter-time. This has been considered an important factor in the recovery of the sea eagle populations in Sweden and Finland (including Åland). Such supplementary diets may offer complementary risk management measures for animals that are naturally feeding also on other foodstuffs than exclusively fish, such as white-tailed sea eagle. For seals such surrogate diet options are not available.

Surrogate feeding-stuffs

The substitution of feeding-stuffs contaminated by DLCs by cleaner feeding-stuffs has been increasingly considered and practiced in animal husbandry, due to the concerns of risks to food chains and food production systems associated with these compounds.

In mariculture interest has been directed to the possibility of shifting from contaminated fish-based feeding-stuffs produced from either cleaner fish such as Pacific fish or from alternative feeding-stuffs based on vegetable oils (Berntssen et al. 2003, Bell et al. 2004, ref. by SPCFC 2005). Feedings-stuffs based on vegetable materials have been viewed in a positive light by Jacobs et al. (2004, cf. above). However, as detailed by SPCFC (2005), these alternative feeding-stuffs have considerable limitations in terms of altered and in some respects suboptimal fat composition and other nutritional qualities. Some of such limitations may apply to other nutritional uses of alternative feedings-stuffs, e.g. in terrestrial food production systems. Therefore, also the continued use of fish-based feedings-stuffs and even relatively dioxin-rich feeding-stuffs (such as those produced from fatty Baltic fish) may be considered as an option. The relative health benefits of consuming fish fed vegetable or fish-based feeding-stuffs, or other impacts of such alternative food production systems, are not clear (cf. 8).

Risk reduction by other measures for protection of receptor organisms

Risks form dioxins in Baltic Sea fish may be reduced especially for non-human sea-living animals by protection measures. These may particularly include

- Protection areas for seals (e.g., HELCOM 2002a, ICES 2003a,b)
- Protection areas for waterfowl (e.g., RAMSAR convention wetland areas)
- Protection areas for fisheries e.g. at spawning.

Marine Protected Areas constitute an important instrument of risk management and sustainable use of marine living resources (Degnbol et al. 2002). However, in the case of long-range transported pollutants such as PCDD/Fs and PCBs, they offer only limited protection and can be considered as an auxiliary measure at best.

7.4.5 Exposure reduction by increasing excretion through altered general diet

In principle, dioxin exposures may be reduced by affecting the general diet, especially fat, and thereby the fat stores, in order to increase dioxin excretion and clearance from the body. Such risk reduction approaches might be seen as a natural extension of the attempts to reduce dioxin intakes by dietary changes. Reduced dioxin intake in many cases is necessarily accompanied by some form and measure of reduced intake of fats, especially animal fats high on food chain and with high dioxin contents. Dioxin clearance takes place naturally during lactation, although then transferred to the "top consumer in the food-chain, the baby" (Koppe 1995).

Excretion-focused remedies have important constraints, as pointed out e.g. by Birnbaum (1994a) with reference to the results of Schnare et al. (1984). Some constraints can be deduced already from the kinetics of dioxin in the body (cf. 3.5.2, Carrier et al. 1995a,b). The studies of the development of dioxin body burdens after dietary intervention (Weihe et al. 2003, Darnerud et al. 2003) also indicate that it is difficult to significantly and rapidly reduce the body dioxin stores either by intake reduction or by de-fatting

or other such diets. In general, once fat and dioxins have set in, it is hard to get rid of them.

Pluim et al. (1994b) studied the possibilities to diminish the intake of PCDD/Fs and PCBs by the baby using two diets tested in breast-feeding women for their ability to reduce concentrations of dioxins in human milk: a low-fat/high-carbohydrate/low-dioxin diet (ca. 20% of energy intake derived from fat) and a high fat/low-carbohydrate/low-dioxin diet. Despite significant influences of these diets on the fatty acid profiles, no influence on the dioxin concentrations in breast milk could be found. The conclusion was that short-term dietary measures would not reduce dioxin concentration in human milk. This may also be seen as an argument generally for prevention instead of remedy.

Radical changes in diet, be they sudden (and thus generally inefficient for dioxins) or prolonged, carry the risk of causing other adverse health effects instead, as the general condition and potentially metabolism is changed. Obesity in itself is a very important (and increasing) risk factor of many adverse health conditions and should be reduced, and these reductions offer synergies with reduction of toxicological risks from DLCs. However, the relationship between obesity and dioxin exposure is complex. Carrier et al. (1995b) demonstrated by model simulations that reduced fat stores e.g. through fasting may increase the level of circulating (serum) dioxins during a transition period before they are metabolized by the liver. That is, fat stores act to some degree also a buffer (and in some sense a diluter) against dioxin exposure in other, potentially more critical tissues. Pelletier et al. (2002) showed that such mobilization of DLCs in body fat may be associated with thyroid effects.

There is always a balancing act in dietary advice and interventions between risks and benefits, and a need to adjust such interventions to the needs and preconditions of the individual subject based on expert guidance. Drastic dietary changes and even liposuction are presently a fashion and it is conceivable (and indeed already indicated) that e.g. strong detoxification diets may become more common, involving some counter-veiling new health risks. These considerations are in some respects similar to those with extreme vegetarian diets. Thus, in addition to eating low on the food chain to avoid bioaccumulating toxicants, beneficial and necessary ingredients from higher levels are also needed for many individuals.

On the basis of studies among Inuits (Deutch et al. 2003), cessation of smoking seems to provide a means to lower the body burden of POPs. If valid for dioxins, this would be a potentially important risk management approach as it simultaneously would reduce a major cause of health impairment within and beyond the effects of DLCs, and as smoking is unnecessary. On the other hand, elimination of PCDD/Fs seems to be faster in smokers at least at high exposures (Flesch-Janys et al. 1996), thus constraining such strategies to abate exposures and risks.

7.4.6 Therapies of adverse effects

Once adverse effects of DLCs have set in, it may be possible to treat or alleviate them by therapeutic means. These may target various processes from kinetics to receptor binding and subsequent responses, even to adaptation and biological compensation. Therapies may be directed to various species of domestic and wild animals and to various groups, including those that are at particular risk and at the same time particularly amenable to treatment for biological and social or practical reasons.

Generally, it may be held that therapies as a means of risk management are too passive and end-of-the-chain. The applicability of many therapies may be restricted to higher levels of exposure causing gross dioxin toxicity, and thus may have limited ability to reduce population risks, also from dioxins in Baltic fish. However, some therapies can have more general relevance. If other risk reduction measures are found to be ineffective, therapy can be a practicable complementary strategy and class of measures. Some therapies may also be part of prevention, e.g. by measures that generally promote health of humans or other animals exposed to DLCs.

The reduction of uptake or enterohepatic recirculation of DLCs including PCBs and PBBs has been found to be limited (Schnare et al. 1984, Schnare and Robinson 1986). On the other hand, accelerated excretion has been accomplished by various therapeutic agents. Geusau et al. (1999) showed that Olestra reduces the elimination half-life of TCDD from c. 7 to 1-2 a in highly exposed persons. Also ginseng ingredients (Moon et al. 2005), medicinal carbon (Kamimura et al. 1988) and chlorophyllin-chitosan (Kitamura 2005) have increased excretion of PCDD/Fs in humans and other mammals exposed to high levels; ginseng

also has other protective effects (e.g., Hwang et al. 2004). However, psychological resistance to detoxifying treatment may be a key obstacle (Schnare et al. 1982).

Some therapeutic agents may be developed on the basis of their antagonist activity against DLCs. For instance the competitive AhR antagonist resveratrol, a wine ingredient, has been considered to possess adequate potency and non-toxicity to warrant clinical testing as a prophylactic agent against DLC-induced pathology (Casper et al. 1999), exemplifying the importance of total nutrition for health risks form DLCs and also the potential of specific dietary ingredients (or other biogenic agents) for therapies. However, such agents may not offer sufficient remedies and may also carry other risks.

Sakurai et al. (2004) reported that a cholesterol-lowering drug, colestimide, can decrease the blood dioxin level of humans; Mori et al. (2005) found it could (slightly) reduce body burdens of PCBs. Since this drug, capable of altering the kinetics of dioxins and reducing exposure, is beneficial also in other ways, it may be more feasibly used. The potential effect on cardiovascular health is also notable, both as a benefit generally and in relation to fatty sea fish. Such an agent may provide one way to add to the health benefits of fatty fish and simultaneously reduce risks from DLCs. However, alternatively such drugs may reduce some of the need for consumption of such fish in the first place, acting in a way as surrogates for the healthy ingredients in fatty fish. Thus, it is not clear what the decision alternatives are, how important a win-win situation such an agent offers, and how strong arguments it provides for reducing or increasing consumption of fatty fish. This would depend on the relative weight given to the reduction of cardiovascular risk on one hand and risks from dioxins on the other. Such therapy may in any case not be so important an option as to override some other benefits from and grounds for consuming Baltic fish (such as cultural and social fish diet related grounds).

An important factor in favour of dioxin therapies is the rapid development of knowledge and tools in biomedicine and biophysics, structural and clinical chemistry and pharmacology, e.g. by the use of advanced structure-activity models, that help design treatments. This development is intimately linked with the ongoing studies of the biochemical and physiological basis of dioxin action mechanisms that will also allow tailored specific treatments. For instance, Thomae et al. (2005) showed *in vitro* data suggesting that the transforming growth factor 3 that restores fusion of palatal shelves, or stimulators of this signaling pathway, hold potential as antidotes of dioxin-induced deformations and that such activities may extend to additional toxicological endpoints. The practical application of such findings is not in near-term, and they should not be exaggerated in dioxin risk management (cf. above). However, they may have other significance, also for therapies, beyond immediate effects and risks of PCDD/Fs.

7.4.7 Post-effect options including adaptation, compensation and maximization of health benefits

Adaptation to risks

DLCs will load the Baltic for a long time to come, and will cause risks despite restriction of exposures both of non-human animals and humans. Therefore, adaptation to these risks emerges as a notable approach to these risks. In lay terms, one needs to 'live with' such lingering past legacies, even if also attempting to control them and to diminish risks.

In the ecosystem some adaptation develops naturally, directly to toxicants and indirectly by other mechanisms to cope with their suppressive and adverse effects. In human populations, adaptation takes place by particularly varied mechanisms, including medical, technological, socio-cultural and other means and processes. The borderline between risk avoidance and adaptation is not sharp.

Compensation for risks

Compensation for risks may be understood both in terms of biological compensatory mechanisms and as economic compensation. DLCs in Baltic fish cause biological risks and impacts and socio-economic impacts; both of these may involve losses that should and could be compensated for. Compensation for risks may be divisible in support to those bearing health risks (in proven cases) and compensating for those bearing economic risks.

The losses to fishermen and fisheries are of prime importance, particularly if implementation

of regulatory or even non-regulatory e.g. fish consumption advisory measures and developments will reduce the income for fishers, fisheries and the fisheries industry and associated other industries as a whole. This is not a technical but mainly administrative, legal and socio-economic risk management area, and will be discussed in more detail below (cf. 8).

Maximization of fatty fish health benefits

Options in this area include e.g. the following:

- Increasing herring use overall and relying on positive risk-benefit balance
- Increasing herring use selectively e.g. among elderly but reducing use among young
- Increasing dietary consumption of fatty fish and fish oil to lower plasma lipids and lipoproteins in treatment of hypertriglyceridemia (e.g., Phillipson et al. 1985, cf. SPCFC 2005).

7.5 Synthesizing evaluation of risk management options

7.5.1 General and technological considerations

As shown above, there are many options for reducing and otherwise managing risks from DLCs in Baltic fish. Several important options are available also before DLCs reach the sea, e.g. in the catchment and during formation of PXDD/Fs. On the other hand, risk management and intervention options in the **risk chain** are extended from the present focus - intake control - to effects and beyond. Risk management not only signifies reducing fluxes or pools of DLCs; it also includes prevention of, adaptation to and compensation for risks. These categories of options involve technological and other (e.g. institutional and economic) dimensions.

Several **levels and scales** can be discerned in technological management options. They include both 'soft', immaterial, and 'hard', concrete material aspects. Their spatial scale varies from EU-wide or catchment-wide to local options. The scale of technological options is further divided in lab or bench, pilot, field demonstration and commercial scale. Also in the case of DLCs, commercial systems are rare in comparison with those realized only in smaller scale.

Many technologies and options are as yet only **emerging**. This makes it difficult to assess their potential. For instance, even though the promise of biological treatment of materials contaminated by PCDD/Fs (and dlPCBs) has been around for decades, no efficient and reliable commercial-scale systems have materialized. This may be due to inherent limitations of such options; nevertheless, breakthroughs are still possible. Intensive development is underway also in treatment of DLCs in fish oil, prompted by the concerns and regulations for feeding-stuff dioxins. As a whole, it can be expected that many technologies will develop to greater cost-efficiency, given sufficient incentive. However, this development is variable, often takes place in spurts and requires sustained support, may also run into inherent and permanent limitations, thus reaching plateaus and dead-ends, and is generally dependent on the contexts for development and application (see e.g. DANCEE 2004).

Technological options have variable **applicability** in terms of agents (DLCs), matrices or environmental compartments, receptor organisms, risk stages, application sectors, and so forth. For instance, some treatment or prevention technologies are applicable only to PCDD/Fs, others to DLCs and POPs more generally. Some technologies may be applied along extensive parts of the risk chain, from prevention and emission control (including hotspot cleanup) to treatment of contaminated fish. In particular, options available in other regions may not be directly applicable to the Baltic Sea and its catchment. The specific characteristics, contexts and capacities of the various countries affect the appropriate technological solutions, e.g. in stockpile treatment (DANCEE 2004). Different approaches may thus be needed, especially between Eastern and Western seaboard countries before the harmonization that is possible due e.g. to EU integration takes place.

Within **prevention** of dioxins, options are dependent largely on production and handling technologies and procedures for precursors, including institutional, economic and other soft aspects. Influencing the need for precursors in the first place, though low-halogen and generally low-waste technology, and development of production and consumption processes e.g. to include more efficient recycling, are

complementary keys to progress. The former seems likely to offer opportunities despite the difficulties caused by indirect impacts, as this 'root cause' level has seldom been even considered in dioxin risk management. The latter seems also promising, since in many cases PXDD/Fs have not been fully taken into account in designing chemical production and treatment systems, the focus having been on the precursors themselves. PXDD/F formation is also not normally accounted for in EU's assessment and management systems for existing chemicals, pesticides, herbicides and biocides. Thus, there is expected potential for such minimization of PXDD/F formation as would also be compatible with other goals and constraints of the chemical industries in question and even with the goals and constraints of broader fields of technological, economic and societal activity. However, not all restriction of (potential) dioxin precursors is efficient and warranted, and some is already taking place. In general, high-risk and easily replaceable uses of precursors can be prioritized for substitution.

In **emission control**, in addition to applying the advanced technologies available for point sources, the abatement of diffuse and secondary emissions including those from wastes is a key area, the importance of which will grow along with shifts in risk patterns and sources of DLCs. In this area, control technologies for point sources such as incineration, metal industries and other thermal processes can be utilized, and technologies for other pollutants (such as PAHs and particles) can be integrated. On the other hand, regular BAT and BEP procedures in many cases require development to enable simultaneous abatement of PXDD/Fs. This can be accomplished e.g. by the processes developed under the POPs Convention and other policy instruments. Stockpile and hotspot treatment constitutes an additional area of risk reduction. In general, it is advisable to prevent dioxin emissions by removing and treating them in concentrated form, on land and in the catchment rather than after dispersal especially to the sea.

In the sea, some other options than excavation-based cleanups have potential for managing risks from DLCs. Some are based on partly established technology, such as removal of DLCs in fish catch. Its application requires relatively modest technological additions and modifications. The key challenge in these areas is to increase the knowledge and organizational abilities for taking DLCs into account. Other options directed to the sea such as steering of bioaccumulation in (fish-based) food chains are as yet untried, but hold promise in the longer term within multi-objective fisheries management. Restriction of DLC mobilization in maintenance dredging represents an intermediate case, as it is based on established technology but requires considerable modification in order to control DLC dispersal in spoils and to treat and dispose them appropriately.

Several options are available for **post-sea** reduction of exposures of humans and, to a more limited extent, other fish consuming species. These options include regulatory, information-based, technological and economic measures. The development of diet advisories is of particular importance, as will discussed in the next Chapter (cf. 8), but also this option involves limitations and uncertainties.

In general, efficient and reliable full-scale and practicable **destruction** of PCDD/Fs (and dlPCBs) in emissions, wastes and products, stockpiles and the environment still relies on thermal technologies: 'The ring forged in fire is to be destroyed by fire". However, this generalizing evaluation has to be modified by the development of physico-chemical treatment technologies that have demonstrated comparable cost-effectiveness and other properties. In containment and disposal, advanced established technologies are available and can be applied to DLCs, e.g. by pre-treatment and dedicated controls, but at considerable cost in many cases. The permanence (and reversibility or improvability) of some of these solutions may also be questioned.

7.5.2 Environmental and health considerations

The environmental impacts and efficiency in terms of risk reduction vary greatly among management options. As risks are multi-dimensional, some options may decrease some risks (such as risks to human health) but simultaneously increase others (such as ecological risks).

The impacts of many options on environment and health are difficult to foresee. The impacts and performance of preventive measures are particularly intractable. Even when the reduction in emissions can be estimated at some level of confidence, the dynamics of the impacts on the sea and on dioxin levels in the

fish are uncertain, due e.g. to the lack of fate and transport information on many specific congeners. Still greater is the difficulty to predict (and reconstruct) the impacts on health, considering their multi-dimensionality, the simultaneous other factors, and also the offsetting benefits from fish consumption. The multi-directional and total impacts of some important management strategies and options have been dealt with in some detail in the following chapter (8.3).

In relation to technologies and practical management of DLCs, occupational health effects and hazards need to be taken into account. These have been in focus e.g. in UNEP evaluations and guidance for treatment technologies for PCBs, PCDD/Fs and other POPs. Even so, in some cases such additional risks from risk reduction have not been sufficiently accounted for, especially in the past.

7.5.3 Economic and regulatory considerations

There is limited information on economic impacts and aspects of dioxin risk management options. This will be discussed in more general terms in the following chapter (8.4). In the present connection, costs for various technological options are initially considered. However, cost data are scanty, being mainly limited to some control measures for primary emissions (to air from stationary point sources) and to cleanup of contaminated sites or treatment of DLC-containing wastes. Even in these categories, cost estimates are constrained by the definitions and rationales used in estimation, e.g. the variable inclusion of indirect costs. In connection with this, the direct costs to operators (investment and running costs) have seldom been complemented by estimates of costs incurred to other parties.

A particularly important point in the present connection is that there is inertia in the dynamics of the management system, and management measures have already been agreed on and taken. Therefore, without (marked) additional spending, some risk reduction will take place, and the 'Business As Usual' alternative does not mean the risks are fixed to the present level. This is shown by the calculations of dioxin abatement costs in UK (Entec UK Ltd 2003), being the only published relatively comprehensive estimate of (air emission) abatement costs for dioxins that has been located. It illustrates that risk reduction beyond the measures that are already in the pipeline will be possible, but the marginal costs for additional risk reduction will be high.

As to substitution of dioxin precursors, some similar considerations may be noted. The production and use of many DLCs and (potential) dioxin precursors has already been considerably reduced, especially in Western Baltic Sea countries but also more widely, e.g. in EU. This has happened in many cases due also or even primarily to other reasons than PXDD/Fs. However, the other reasons for reduction indicate that multiple benefits and also multiple management processes may be utilized to reduce dioxin risks also. There will subsequently be some lagged decrease in exposures to and risks from dioxins based already on the measures taken. The costs for additional dioxin risk reduction are in general likely to be higher in relative terms. The concurrent loss of benefits, e.g. from some vital chlorinated biocides or industrial chemicals, may correspondingly be higher and may more easily exceed the benefits from dioxin risk reduction.

The **development** of costs, also unit costs, and cost structures, along with technology development, can have a major impact on the long-term costs. For instance, by strong regulatory or other incentives, new dioxin control technologies with much greater cost-effectiveness may be developed. These changes cannot be presently estimated expect for some categories of options, as shown by Entec UK Ltd (2003).

7.5.4 Summarizing evaluation

A summarizing comparative evaluation of risk management options for DLCs in Baltic fish is presented with an emphasis on technological factors (Table 34).

Table 34. Comparative evaluation of main classes of risk reduction and management options for dioxin-like compounds in the Baltic Sea, with particular reference to fish and to technological characteristics of options (+, ++ = favourable, -, -- = unfavourable, +/- = multiple; nr=not relevant; uncertain impacts or characteristics are in parentheses, and notes in table cells refer to the last mentioned impact). Cf. text, 8.3 and Annex 10.

Option (class)	Availability/ devel. stage	Dioxin RR effectiven.	Other health effect	Other environ impacts	Fish dioxin RR speed	Applicability/ comprehensive.	Compatibility/ flexibility	Robustness/ sustainability	Direct costs	Infrastruct. needs	Regul. base	Social acceptability	Overall/ notes
Prevention of dioxin formation													
Social needs/root causes	(+)	(++)	(++)	++	- -	++	++	-	++	++	+/-	+/-	++, omitted
Products/chemicals (incl. precursors)	++	++	++	+	-	+(+)	++	+	(-/+)	+	+	++/+ case var	++, promise
Technological processes/facilities	++	+	+	+/-	+/-	+/-	+	++	+/-	-	+	+	+
Emission control													
Land-based stationary primary	++	+/++	+	+/-	+/-	+/-	+/-	++	-	-	++	++	focused on now
Land-based diffuse primary	+	(+)	(+)	+/-	-	+/-	+	+	-/+	-	+	+	+, potential
Land-based second. (wastes/products)	++/-	++/+	+/-	+/-	-	+	+	+	+	+/-	+/-	++/-	+, potential
Stockpiles/hotspots cleanup on land	++/-	+/-	+/-	+/- -	+/-	+/-	+/-	+	-/- -	+/-	+/-	+/- -	+/-
Intercept/contain before sea	-/+	(+)	(+)	+/-	+/-	-	+/-	+/-	+/-	-/+	+/-	+/-	+/-
Control of direct discharges	+	+/-	+/-	+	+	-	+/-	+/-	+/-	+/-	+	++	+
Measures in sea													
Control of fluxes/ transport	-	(+/-)	(+/-)	(+/- -)	-	+/-	+/-	-	-	-	+/-	-	+, dredg. limits
Hotspot cleanup	-	-	+/-	-/- - (dispers)	-/+ (local)	-	-	-	- -	-	-	-	-, problematic.
Steering bioaccumul (fisheries manage)	-(+)	(++/-)	(++/+)	(++/-)	(+/-)	+/-	+	-	+/-	+	+/-	+	+, potential
Removal of pools	-(+)	(+/-)	(+/-)	(+/-)	(-)	+	+	-	-/+	+/-	-/+ (fish)	+	some potential
Exposure reduction													
Food advisories etc information	++	(+/-)	++	(+)	+/-	++/- (not ecol)	++/+	+	+	+	+/++	++	+, trad, natural
Fish market regulat (& product labelling)	+	++	-/- -	(+/- -)	++	++/-	+/-	+	+/- admin	+	++/+ (derog)	+/-	+/-, side effects
Other protection of receptors	+/-	(+) ecol	(+/-)	+/-	(+)	+/-	+/-	+/-	+/-	+/-	+/-	+	
Treating fish/fish products	+/-	+	+	+/- (refuse)	+/-	-	+/-	-	-/+ (oils)	- (tech)	+	+	evolving
Surrogate diets (incl. supplements)	+/-	+/++	+/- -	(nr)	+	-	(+/-)	+/-	+/-	+/-	+/-	+/-	potential, some risk
Increased excretion	(+)	(+/-)	(+/-)	(nr)	+	-	(+/-)	-	-	+/-	(+/-)	-	marginal
Alleviation of effects													
Therapies	+/-	(+/-)	+/-	(nr)	+	+/-	(+)	-	-	- (med)	+/-	+/-	marginal, reactive
Adaptation	+/-	(+/-)	(+/-)	(nr)	+/- (grad)	+/-	+	+	+	+	-	+/-	+, important
Compensation	-	- (per defin)	+/-	+/-	+	-	+/-	-	-/- -	+/-	-/+ (partly)	+/-	institut. limits
Maximization of benefits from fish	+/-	+/-	+ (per defin)	(nr)	+/-	-	+	+	+	+	-/+ (indir)	+	

8 Management strategies and their characteristics and impacts

8.1 General policy considerations

8.1.1 Basic styles of governance in risk management

Regulatory or voluntary management or governance

The management of risks in many areas, including environmental, health, food and fisheries, has often relied on regulations. This is the approach also to dioxin risks in EU and its member states, notably in the EU strategy for dioxins and PCBs in food and feeding-stuffs. A preference for regulations may be partly due to the nature of the union, and as these instruments are perceived as the primary ones.

Other management approaches and instruments have also attracted attention and have been subject to increasing development and application. These include information based and voluntary measures. The question of regulatory or non-regulatory risk management, or their intermediates and combinations, thus is a key divide in approaching the issue of dioxins also in Baltic fish (Fig. 20).

These choices are related to the fundamental question of what role a normative (i.e. legally based) approach to risk management plays in relation to other measures, including voluntary approaches. This will be discussed in more detail below, especially in connection with information steering approaches utilizing dietary advisories, as an alternative or complement to regulatory steering. It is pointed out initially that voluntary management is accepted on particular grounds even with some marketed foods and commodities carrying considerable health risks. In these cases, a mix of steering instruments have developed to achieve management of health and other risks, reflecting multiplicity in policy principles.

Centralization and subsidiarity

The coupling of EU-wide and other levels in governance and management of risks from dioxins in the Baltic Sea and its fish impacts strategic choices and their impacts. These other levels include especially the national level, to some extent also local, regional and global levels (Fig. 20, cf. 6).

A key choice in risk management approach also in the case of Baltic Sea fish dioxins is that between centralized governance, now increasingly at the EU level, and a bottom-up approach placing more reliance on participation of those regulated and allowing more variation due to their circumstances. The latter approach may avoid segregation between regulators and the regulated, and emphasize the multiple, overlapping and complementary roles of actor groups.

EU itself has increasingly been seen as a multi-level actor and governance regime (cf. 6). Consequently, the need for participatory and generally more varied approaches to governance has been recognized, to complement or substitute for centralized governance, e.g. in relation to fisheries (Degnbol et al. 2002). The interplay between integration, centralization and harmonization on one hand and differentiation and subsidiarity on the other is also constantly developing, and is linked with the overall development of an extending EU. The strive toward centralization and harmonization is inherently strong in the EU, a project largely about common markets.

The appropriate level of centralization varies according to the management situations. In the field of environmental protection, international harmonization is often particularly important because of the essentially international character of the problems. POPs with long-range transport are a good example. The same applies to fisheries, not only due to the shared market for products but also to the fact that marine fish reside and are fished in an internationally shared area such as the Baltic, as reflected in the detailed international procedures and cooperative structures created for this. Thus, Baltic fish dioxins as marine POPs are doubly a trans-boundary issue.

Also within POPs and fisheries management the combination of an international (increasingly EU-wide) governance with sub-EU levels is important and problematic. For instance, with the

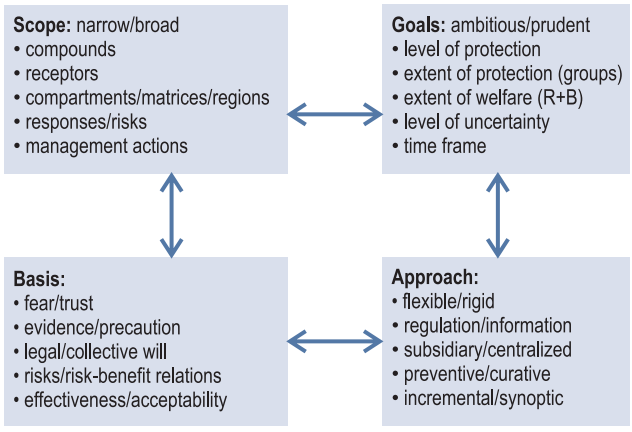


Fig. 20. Summary of key aspects of risk management strategies used in structuring the analysis of strategic options and their impacts. Some divisions describing the aspects have been indicated.

CFP, national and regional conditions need to be taken into account but present great challenges (as seen e.g. in Baltic cod and salmon fisheries). National conditions are increasingly addressed also as the POPs Convention enters the national implementation stage. With Baltic fish dioxins, the coordination of national and international policies has turned out to be tricky e.g. with respect to the Finnish and Swedish derogations from the EU strategy.

In fisheries management, attention has been paid to complementing centralized regulatory management by means of participatory approaches and co-management (e.g., McCay and Jentoft 1996, Degnbol et al. 2002). This has in part reflected the changing situations and roles of fisheries. Jentoft (1998) stressed the multi-level representation and rationality of communities e.g. in questioning the objectivity of established fisheries science. Likewise, in health care increasing weight has been put on participation. This has been associated with recent environmental health risk concerns and initiatives (e.g. SCALE), but also with other attention to the diverse needs and possibilities of the various groups in society. However, also in health care always a strong element of harmonization and central steering is present, e.g. to ensure uniformity of requirements and efficiency of measures.

In managing risks from DLCs from the Baltic Sea and its fish this balancing constitutes a central policy and strategy issue. It is a fundamental question also behind derogations from EU-wide quality criteria for fish: should they be allowed and on what conditions? The difficulties in this are related to the integration of a harmonized approach with consideration of national and other regional interests. More generally, it is a question of relating uniformity to plurality, e.g. with regard to sector and actor interests, and thus of democracy.

A centralized regulatory management presents problems as particular conditions of a region need to be taken into account. The central government is inevitably distanced from and unfamiliar with the specific needs and conditions of the region; the regional and national level actors on their part may not be as able to see broader European dimensions and interests. It may be increasingly important to account for regional aspects, as the centralized EU governance has frequently been experienced as alien to the common citizen, and resistance has surfaced to a style of governance not allowing for real participation and influence or for consideration of the concerns of those impacted.

It is possible that much of the criticism also towards the EU dioxin strategy can be interpreted in terms of a general resistance and suspicion towards EU governance. For instance in Finland and Sweden, reactions to alleged meaningless interference by the EC with local fisheries matters have been frequently expressed in media, also by politicians in EU-critical connections. Regulation of fisheries and fish dioxins may thus become more general symbols for what is perceived as misinformed and excessive EU regulation. Conversely, it is possible that some of the motivation behind uniform enforcement of the EU dioxin strategy is due to a more general unwillingness to allow deviations from a homogeneous market and policy, not only to factual health concerns.

Part of the criticism against the EU dioxin strategy and recommendation may also be related to other concerns, e.g. on the part of fishers that are under other (even unrelated) pressures as well. This however is largely a different issue that will not be discussed here.

Restrictive or inclusive involvement of sectors

Management strategies have boundaries also in administrative, sector and actor group sense, depending on their contexts and goals. The crossing of these borderlines, either by involving other sectors up front or by linking with them, is a key challenge particularly as integrated management is striven at (Fig. 20).

The present EU strategy on dioxins and PCBs has been portrayed as a comprehensive one, but is not comprehensive even in terms of sectors (cf. 8.3). This is partly related to the focus on human health and food. It is understandable that other environmental issues have not yet been so highly included in its implementation and development. The links with fisheries seem a more unnatural omission, particularly from the point of view of dioxins in the Baltic but also more generally. Management of DLCs in the context also of human health is largely a fish and fisheries issue, but relatively little of the know-how and interests in this area have yet been involved (cf. 8.4).

Level of detail

The level of detail in management is important as some of the criticisms of EU governance, notably in the fisheries area, have particularly concerned the unnecessary detail in regulation that moreover has sometimes been perceived as insufficiently based on 'facts on the ground'.

More detail is naturally required when dealing with specific regulation matters on a more restricted geographical scale, i.e. local or national level; if the central government attempts to have a strong regulatory presence also the local level, it will need to regulate in details.

The level of detail is closely related to the above aspects in management styles. If the overall approach to management is a regulatory and normative one, it is likely to involve more detailed controls than if voluntary measures are emphasized. However, there is a detailed dimension also in some of the non-regulatory management styles; e.g., diet advisories include considerable detail, even more so than regulatory fish quality management, as will be discussed below. In this case the detail may be more easily facilitated, as a flexible approach to management is adopted instead of a normatively binding one. The distinguishing quality thus is not the level of detail *per se*, but whether details are attended to within flexible or rigid management approaches.

If a high level of detail is combined with high uniformity in regulation, particularly inflexible and inefficient (and unpopular) steering can result. For instance, recent proposals by the EC to restrict the use of coastal fishing gear that were based on generalized prescriptions among the various catch species prompted widespread criticism in Finland. They were regarded as yet another example of unnecessarily and even harmfully detailed regulation; they were also ridiculed as a showcase of unfamiliarity of the EC with reality ("inability to tell perch apart from cod"). In such cases, both unfounded detail and lack of detail (unfounded generalizations) may come under attack.

The EU dioxin strategy, while generalizing in some respects, especially through the emphasis on harmonization and through the lacking consideration of particular regional conditions and of dioxin prevention, is on the other hand very detailed in other respects, such as in the specifications for the different target or limit values for the various food and feed items.

Combining styles of governance

One may discern a co-evolution but also potential conflicts of different styles of governance in dealing with the risks and associated impacts caused by DLCs in Baltic fish. An important issue then becomes what combinations of management approaches are or may be adopted, and how they might efficiently complement each other.

A market regulation based approach has been sometimes contrasted with co-management in fisheries regulation but they can also coexist. Similarly, for dioxin management a co-existence of different management styles and instruments can be envisaged, including e.g. combinations of normative regulation and information steering, and of centralization and subsidiarity.

8.1.2 Evidence-based and precautionary management

General considerations

The **definitions** of the precautionary principle vary beyond those general definitions offered in the Brundtland report and the Rio declaration and further in the EU statements on the principle. The need for more explicit interpretations and improved operationalisations of the principle has been stressed also in fisheries management, e.g. in resolving the interests of various interested parties

and the functions of science in advising decisions (Degnbol et al. 2002).

A key challenge for applying the precautionary principle is the very **uncertainty** invoked as an argument for the principle. As noted before, uncertainty involves many levels, and is related to both the probabilities and consequences of risks. For instance, Renn et al. (1998) pointed out in the context of sustainability of regions that the degree of allowable probability of "serious" injury becomes a crucial question in applying the general principle of immediately preventing actions that have "high probability" of such injury to future generations (quotation marks included).

The idea of evidence-based policy-making relies heavily on the **rationality** of decisions, and on the possibility of scientific proof. This can be an overly positivist point of view and unattainable in the case of highly uncertain risks (cf. 9). A key question then becomes: should science therefore be sidestepped, or should one attempt still harder to pursue scientific evidence, although realizing the limitations of this pursuit, and also expanding the limits of science? Science and the precautionary principle need not be mutually exclusive. However, as the latter is seen as an alternative to traditional science-based management, there are inevitable problems in their relationship (Grandjean et al. 2004, Weed 2004a, Durodié 2005). They may include the erosion of well-tried scientific procedures and even the perpetuation of a culture of fear and random alerts and actions. Some of these problems may be surmountable while some are more fundamental and potentially destructive.

Houck (2003), admitting the limitations of a proactive approach to environmental problems, argued against a return to "scientific management" and "good science" (and the lure of money, e.g. in the use of only economic arguments), as this in his opinion has been tried and has failed. He stressed that the request for scientific evidence and peer review can act like a knife that cuts only one-way: against environmental protection, in the name of "good science", pointing to the example of dioxins. However, he did not recognize the perils inherent in straightforward risk reduction, e.g. due to limited and asymmetric precaution, and in abandoning the requirements for scientific information and reflection, e.g. by institutionalizing a policy whereby also anecdotal evidence is accepted as a ground for decisions that become vulnerable to manipulation (cf. Durodié 2003a, 2005). Houck (2003) thus did not consider that also precaution may be misleading and misused, and 'cut only one-way': for alleged (environmental) protection as defined by those able to influence policies.

The level and **kind of precaution** thus need to be defined. There is a risk of missing justified opportunities for precautionary action if awaiting scientific proofs and comprehensive rational plans. There is on the other hand a risk for misplaced, wasteful, and even counter-productive and hazardous actions in the name of precaution. The precautionary principle has to be refined, operationalised, and integrated with other decision and policy-making approaches and principles. In general terms, the dilemma is to strike a balance between 'paralysis by analysis' and panic action. Precaution prompts both action, e.g. banning chemicals and jumping into alternatives, and inactivity in other respects, e.g. lacking innovativeness and even suspicion a priori towards innovations being perceived as dangerous. These have to be sorted out for successful precaution. For instance, there are certain obstacles to and, conversely, opportunities for decisions that can be tracked down to their scale and scope.

Specific considerations related to dioxin-like compounds

Prinz et al. (1993) proposed definitions by distinguishing precaution from prevention and intervention, defining the former to mean "all efforts should be taken to minimize to input of PCDD/F into the environment from the very beginning on". This is still rather ambiguous and also differs from definitions emphasizing the criterion that action is taken in lieu of scientific knowledge. Further definitions are needed e.g. as to what kinds of risk and impacts are treated in a 'precautionary' manner, such as also indirect risks and impacts from risk management, and as to what "all efforts" entail.

Dioxins and PCBs have been used as an example of the need for a precautionary approach e.g. in the analysis of Harremoës et al. (2002) for EEA. However, the risks of such POPs have been diminished (cf. 5). Thus, lingering fears may be excessive and management actions out of proportion to the actual risks. The question more specifically becomes: what kind and level of knowledge, under what circumstances and depending on what factors, justify what precautionary actions on DLCs. Not only are there uncertainties of the toxicological risks (especially of non-TCDD DLCs), but also of the means and consequences of actions, including their unwanted

(and even unforeseen) harmful effects such as the loss of health benefits from fish containing PCDD/Fs.

The list of **criteria** for precautionary actions offered by Rogers (2003) may be applied to the case of Baltic Sea dioxins e.g. in the following ways (cf. below):

- They should be *proportional to the chosen level of protection:* With DLCs in fish this depends on both the benchmarks for dioxin risks and on those for health benefits of fish
- They should be *non-discriminatory* in their application: This is attempted e.g. through the stipulation of EU-wide norms for food dioxin levels; however this can cause asymmetry and discrimination in other respects (e.g. between various foods including fish) and in general ignores those case- and region-specific factors that would be justified even in an uniform application
- They should be *consistent* with similar measures that have been previously taken: This is important for both scientific or analytical and policy reasons, but full consistency can not be required due e.g. to the development of management systems, ambition levels and options
- They should be based on an *examination of the potential benefits and costs* of action or lack of action: This may be disputed as the 'spirit' of the precautionary principle is proactiveness with less consideration of consequences that may not be foreseen; however, as a general principle of attentiveness to consequences it is recommendable and important also with fish dioxin risks
- They should be *subject to review*, in the light of new scientific data: This has some inherent contradiction with the criterion of consistency (see above) but is highly important also in the case of fish dioxins, as scientific research both provides the indices and evaluation criteria for action and may help analyze action itself, e.g. technically and in a policy and decision context.

In defining the appropriate level of precaution, different compounds may need to be discerned. For instance, Norstrom (2002) noted that it could have been predicted by structure-property relationships alone that PBDEs would be persistent and bioaccumulative already before the onset of their use in 1970's and 1980's. However, PBDEs are used, unlike PCBs, for directly life-saving (and indirectly even PCDD/F-preventing) purposes, in fire safety (cf. KemI 2003). It is also not clear whether PBDEs will turn out to pose problems and risks comparable to those of PCDD/Fs and PCBs (cf. 5.4.2). The precise interpretations and implications of precaution may thus be different for PBDEs or some other potentially dioxin-like or dioxin-forming compounds than for e.g. PCBs, depending on their use purposes, properties and other decision factors, even though the general argument and also some specific precaution-warranting characteristics of such classes of compounds may be alike.

Different decision situations and management opportunities need to be systematically explored. For some risk management decisions or actions on DLCs, the prevailing uncertainties do not matter as much as for some others, and action can be taken. For instance, some management opportunities may be more readily available and feasible with little risk of creating more problems and risks, while others may have identifiable and considerable counter-veiling risks. Synergies with other areas of (risk) management may also crucially influence when and how precaution is adequate. The opportunities may further be better or worse with regard to some general policy principles, such as participation, sustainability and equity and so forth.

8.2 Defining strategic issues in dioxin management

8.2.1 Framing

Inclusion of dioxin-like compounds and their precursors and reaction products

The definitions of 'dioxins' have important implications for risk management. DLCs are not limited to PCDD/Fs, or to combined PCDD/Fs and dlPCBs. Such a scope may mislead management by ignoring other compounds of importance. This is not the case when PCDD/Fs and dlPCBs dominate. However, a more inclusive definition seems justified in principle on both theoretical and practical grounds. In some areas an extension is already taking place (e.g. with PCBs, bromodioxins and Bio-TEqs). A precedent is that originally only TCDD was accounted for, not all PCDD/Fs.

There are particular reasons for including corresponding **brominated** and bromochlorinated compounds. Also other classes of compounds have potential significance in terms of dioxin-like toxicity, including some PAHs and reaction products including dlPCB metabolites.

The relation with **precursors** (e.g., ERM 1997, Anon. 2000, cf. 7.2.1) is an important framing consideration as comprehensive and preventive management is sought, even if such compounds are not definable as DLCs. Some substances such as PCBs and PCDE and PBDEs are both precursors and potential DLCs. PBDEs are notable as their levels have been increasing also in Baltic biota (cf. 5.4.2).

Inclusive consideration of 'new' DLCs may be warranted because of **management processes**. Existing processes for management of PCBs may be utilized to better deal with both dlPCBs and other PCBs. Care is needed to retain an adequate balance between integration and differentiation, so that PCBs are not lumped together indiscriminately or treated too separately in cases where they have interactions. Co-evolution of management for dlPCBs, other PCBs and all PCBs can be envisaged.

Extension of the scope in dioxin definitions has **complex repercussions**. Some of them are related to legal or administrative practices that inherently lag behind scientific knowledge. The stance toward this lag depends on the degree of conservativeness adopted. On one hand, there is some justification for sticking e.g. to only PCDD/Fs; one the other, this may mislead management. In a general perspective, artificial definitions and strict delimitations do not work at least as the sole premises, as facts accumulate that require changes. However, more inclusive definitions can be introduced in a gradual process. New concepts thus inform decisions to variable extent. Some potential 'new' DLCs may not ever become (or have to become) formally included in decision making on dioxins, instead forming part of the general background e.g. for POPs and addressed on an *ad hoc* basis only.

Many actors need to focus on those substances included in present regulations e.g. on TDIs and food limit values. The consequences of already these regulations should not be overlooked. The achievements with PCDD/Fs may also have indirect importance for control of other candidate dioxins. However, in other cases the management of PCDD/Fs can involve competition with management of 'new' DLCs that might require different strategies e.g. still more focused on prevention. This is related to the general question of priorities in the risk formation stages targeted; the inclusion of compounds is only one aspect of selecting efficient scope, or efficient combinations of scopes and approaches.

A reliance on e.g. POPs management mechanisms that are of rather different character as to specificity and context, style and pace of management may lose opportunities for efficient control of specifically DLCs. There is also a risk of too narrow focus on some (even if often dominant) DLCs, i.e. PCDD/Fs or combined PCDD/Fs and dlPCBs. That is, narrowness in the 'chemical universe' may be disadvantageous for dealing with both those substances included and with those excluded.

The appropriate inclusiveness of dioxin definitions depends on the **decision situation**. There are cases in which one may (have to) proceed on the basis of a more limited definition, due to e.g. a legal mandate; there are even cases where this is sufficient more permanently. As a general position it seems however that sufficiently comprehensive dioxin risk management in many contexts especially in future will require the consideration, also explicitly, of a growing amount of DLCs.

A key decision on dioxin definition in connection with the EU strategy on dioxins and PCBs in food and feeding-stuffs concerns the **inclusion of dlPCBs**. This has been estimated to increase the apparent TEq levels in feed materials of fish origin by 5-fold (SCAN 2000, cf. Verstraete 2002). Their implications will depend e.g. on the TDIs and on the derivation of allowable food and feeding-stuff levels from them. Also this extension of scope is thus not clear-cut, and alternatives may be identified.

The definition of dioxins by **biological activity** presents in some respects still more significant conceptual changes and challenges. Especially when (if) the impact on health is a starting point, bioassay-based definitions of dioxins are in place. Response-based definitions may offer a means to more inclusive risk identification and efficient management that avoids some of the limitations of chemical analysis of specific DLCs. Bio-TEqs may thus be one toxicologically well-founded and cost-efficient way out of the dilemma of an increasing and ever more complex amount of DLCs to be reckoned with. While sensitivity may be comparable, the specificity of bioassays is inherently lower as they do not distinguish between DLCs (cf. Annex 2). For some purposes additional confirmation of the compounds present will thus

often be needed. The risk of 'false positives' with bioassays requires qualifications however: the identification of elevated Bio-TEq levels may be a false positive with regard to PCDD/Fs and dlPCBs, but a 'correct positive' with regard to other DLCs and total effects. Conversely, chemical analyses of PCDD/Fs and dlPCBs may in this sense cause 'false negatives', excluding other DLCs at the outset. Bio-TEqs and regular TEqs thus complement each other. The EC already facilitates the use of bioassays as complements of chemical analysis (Annex 11).

Inclusion of risks and impacts and their receptors

Among **risk types**, different levels (e.g. biochemical and emergent), different severities and different qualities (e.g. transient or irreversible) of risks need to be specified, as they essentially affect the risk management policy goals and strategic approaches (cf. 5).

The **toxicological effects** driving dioxin risk management are associated with reproductive development. Also cancer risks still play a role, although commonly estimated to be relatively low. This may be partly due to regulatory traditions, partly to a 'cancer premium' in the valuations of people (Hammitt and Liu 2004). Despite improved prognosis, the mortality risk in the case of cancer may justify such a premium. Crettaz et al. (2002) in assessing the disability adjusted life years (DALYs) for various tumors as a result of PCDD/Fs estimated that death was the dominating factor in DALYs for most tumors, morbidities contributing little. However, consideration of other endpoints like developmental, immune, neurological and behavioral effects may alter such quantitative assessments and thus management of dioxin risks (in part as PCBs are more extensively included).

A crucial consideration is in how far **benefits** from accepting risks and risks from losing these benefits are to be included. Whether the most efficient and justified strategy in the case of Baltic fish is market regulation based on limit values in fish, or some other approach, is particularly heavily influenced by the health benefits from fatty fish. These benefits quite likely exceed the risks from DLCs (cf. 4.5, 5.4.4). However, the question arises whether restricted consumption of fatty Baltic fish actually leads to loss of healthy fish ingredients. It should also be specified what kinds of benefits are incurred, to whom, where and when. The benefits seem to be more certain than the adverse effects or risks from exposure to DLCs in fish, but the certitude of the former and this difference in uncertainty may also be exaggerated.

The health benefits from fatty fish do not affect only the EU approach to managing the risks from dioxins in food and feeding-stuffs. Risk-benefit considerations are equally crucial in diet advisories. However, in essence the argument against too extensive and strict reduction of exposure to dioxin-rich fish e.g. by market bans has been that they are likely to make consumers switch to unhealthy diets. Dellinger (2004) emphasized this danger in the case of tribal fishing in the Great Lakes, pointing out that the adverse health consequences due to further departure from traditional fish diet make switching to a market-based diet a risky venture. The health risk from reduced consumption of fish because of its contaminant contents was also stressed by Egeland and Middaugh (1997) and for the Baltic by Tuomisto et al. (2004a), and was initially analyzed by SPCFC (2005). Whether such unwanted consequences will be realized in the case of the Baltic, or may be prevented or alleviated e.g. by diet advice supporting healthier alternatives to herring, may depend on which populations are addressed, and what means of risk management are used (see below). Dellinger (2004) presented the optimistic evaluation that the Great Lakes intervention strategy focused on fish diet advisories can help substantially limit exposure while maintaining fish as an important dietary source of valuable ingredients (cf. 7.4.1).

As pointed out by Wheatley and Paradis (1996, 1998) and Wheatley and Wheatley (2000), also the broader **socio-cultural impacts** of altered diets need to be considered. These authors stressed that this is particularly important in the case of indigenous people whose social cohesion and way of life may largely depend on consumption of fish and other seafood. In such cases, restrictive dietary advice that is by narrow definition 'safe' may lead to severe unwanted consequences not only through the loss of beneficial ingredients in fish but also through a socio-cultural disruption. Such broader and largely indirect impacts may be highly relevant also in the case of other populations and the Baltic Sea. These impacts are largely associated with unwanted and harmful consequences of scaring people away from healthy and socio-culturally well-established and important dietary habits and ways of life.

In as far as significant risks to **non-human** animals are found to be caused by dioxins (cf. 4.3, 5), management may essentially require preventive measures, adding more weight to such general management approaches. Also in-sea and post-sea measures may be practicable and feasible to protect

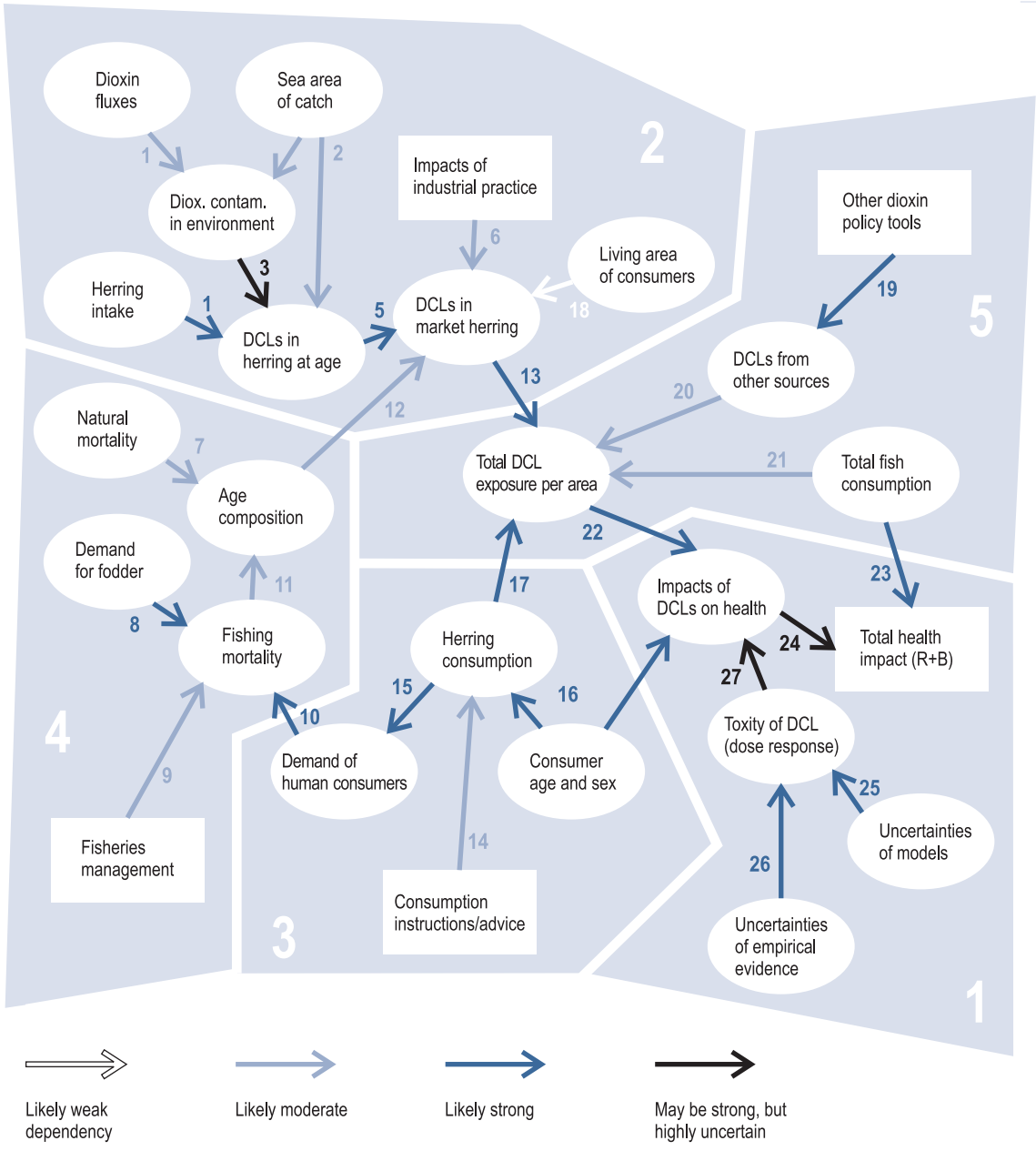


Fig. 21. Structure of a probabilistic management model of dioxins in Baltic Sea herring. Bayesian net notation has been used. Ellipses denote probabilistic variables, arrows conditional dependencies, squares control (decision) variables. The strength of arrows indicates first views about the strength of dependencies. (Modified from Kuikka et al., unpublished proposal 2004).

and to improve recovery of non-human animals, but will differ from those geared to human health risk management (cf. 7.4.4).

In human health risk management, it is assumed that all **human groups** are included. There is often particular concern for the young (e.g., EC 2004a). However, also other age groups are important. As to children, a difficulty is the lack of information on toxicokinetics and sensitivity, including critical windows of exposure (cf. 3.5, 4). In addition, the distribution of risk management options such as information across population groups needs to be considered (see below).

Inclusion of system parts: Description of the fisheries based health risk management systems

Dioxins in the Baltic, its fish, its catchment and adjacent areas comprise a wide and complex set of systems. These can be analyzed and managed on many levels using various frames and approaches. One may focus on human health and therein on food and feeding-stuff production systems. Fish and **fisheries** are actually part of these, but have not been taken into account in previous strategies. Therefore, it is of particular importance to analyze the fisheries system in more detail (Fig. 21).

There are many sub-areas and many interactions and dependencies, strong and weak and more or less likely, in the management system for Baltic herring fisheries. The herring fisheries are further linked to other systems such as other fisheries, fodder production, general nutrition and health care. An analytical simplification and delimitation of the management system that is in some respects arbitrary thus has to be made, and such contexts and links are treated as externalities at a less explicit level.

The **geographical scope** of risks and management is related to the levels of governance (cf. 8.1). The management of DLCs in the Baltic is influenced by and influences risks in other regions. Also the management systems within Baltic Sea fisheries are linked with those elsewhere, particularly Swedish and Danish North Sea and Atlantic fisheries. It is problematic if these areas of activity are separated too strictly. This may happen e.g. if Danish (or Swedish) fisheries and food policy try to secure a separation from Baltic fisheries perceived also by consumers to be much more contaminated than those in the Atlantic. Degnbol et al. (2002) emphasized the importance of scale both in natural and societal systems in integrating fisheries and environmental (and other) policies, and the particular needs and opportunities in this based on the Nordic and Baltic Sea experiences.

An important question is the inclusion of **animals fed** by Baltic fish. They are objects (especially minks) and food-chain transmitters (pigs and poultry) of toxicological risks in this fish, in addition to being nourished by it. The DLCs in their excreta and carcasses also cycle in the system.

Even without a detailed analysis of the processes in these systems, it is evident that the outcomes of management actions are complex and may be substantial. For the present purposes, this realization is sufficient (cf. 8.3). It can be seen that several factors in many sub-systems affect the risks, impacts and uncertainties associated with dioxins in Baltic herring, and thus influence management choices.

8.2.2 Goal setting

General

The **level of protection** has to be defined in setting goals for health and environmental risk management. As risks are multi-dimensional, no single quantitative measure alone may sufficiently capture the goals. A 'high level of protection' is a central principle in EU environmental and precautionary policy. However, it should be specified what organisms, in what respects and against what are to be protected, before such general statements of goals are properly translated into practice.

Goals may be **formulated** qualitatively or quantitatively, and in terms of general outcomes or of risk reduction. Goals may be substantive (e.g. a risk reduction rate) or procedural and thus indirect. Goals can be generic or specific, as with objectives for a particular region, group and food category. Important strategic goals may further include the speed by which a goal should (and could) be reached, and the rate of progress. This is related to both the outcome and the means. Empirical data indicate that expert opinions on such aspects of goals vary greatly (cf. 8.4.2).

Quantitative risk goals for human health are set by means of TDIs. Quantitative goals may also be expressed by margins of safety for exposure, on the implicit assumption that some exposure level represents a benchmark for an appropriate target effect level (cf. 5.4). Ecotoxicological risk goals can similarly be set at various points of risk chains, and are constrained by the same general limitations and uncertainties. All such goals may thus include measures of variation. However, differences in expert opinions regarding quantitative goals (cf. 8.4.2) also reflect more fundamental valuations.

The goals essentially depend on what they mean, i.e. their **function and implications**. If the function and general character of goals is long-term targets, it may be justified to set them ambitiously. If on the other hand a goal would trigger fast and binding action it can be counter-productive if not well founded and realistic.

From a risk management policy point of view, e.g. the following risk-related factors are further to be taken into account in goal setting:

- *Targets* of the risks or protection (e.g., humans or non-human animals; specific groups)
- *Dimensions* of risks (time, geographical and other)
- *Causes* of risks and the causality of effects (especially the attributability to dioxins)
- *Significance* of the risks (as perceived by those deciding on or influencing goals)
- *Benefits* associated with bearing the risks (related to the dimensions and targets of risks)

- *Possibilities* of controlling the risks.
- *Contextual* factors, such as in what regimes and based on what regulations risks are managed.

The inclusion of new substances

Risk management is to a great extent dependent on which compounds are included and how they are assessed, as discussed above. This applies also to goal setting. It was noted (cf. 3, 4) that TEqs in many tissues were 5-fold higher when applying early proposals for TEFs including di-*ortho* PCBs. On the other hand, inclusion of dlPCBs in the regulatory limit values for food would again cause an apparent increase of risks (in the case of fatty Baltic fish, often a doubling). Even after such revisions, the basis for TDIs and other quantitative risk management criteria is not clear-cut (cf. 5.4).

The inclusion of new substances may greatly increase the apparent risk (as defined by TEqs). This may also increase counter-veiling risks if vital livelihoods would be affected, e.g. not only Baltic fisheries but EU food production more generally. The interpretation of TDIs and other management criteria and goals may change in such a situation, and additional decision criteria may emerge. It can be envisaged that risk-benefit ratios of a foodstuff will be formally accounted for; a step in this direction can be discerned e.g. in the SPCFC (2005) assessment of risks and benefits from contaminants in fish. The question then becomes one of balancing inclusiveness of risks with consistency of goals. Adjustments in guideline values and advisory levels that by their nature are more flexible may be more easily possible than adjustments in regulatory action limits (cf. 8.3).

The implications of risk levels and uncertainties

As detailed above, the risks from dioxins in Baltic Sea fish are in general not *demonstrably* great in the sense that they would *certainly and consistently* cause *severe emergent* health impairment in humans or non-human animals, and be comparable in magnitude, extent and certainty with the risks from many other key causes of health impairment. Simultaneous health benefits from fatty fish are also considerable. Such comparative evaluations of risks and of benefits are relevant also for goal setting. They depend on the criteria set for a) risk significance e.g. in terms of the relationship between potential and emergent effects and between subtle and severe risks, and in terms of population risks, b) commensurability of risks and of benefits, and c) 'demonstration' and 'certainty and consistence'.

Even a smaller relative share of the Baltic fish in the DLC-caused risks may be important if the total risk is high enough; the additional risk caused by Baltic fish may 'break the camel's back'. Thus, the fact that the risks from DLCs are smaller than some other risks may not necessarily imply that the former could not be a priority, depending on some risk characteristics (such as reducibility). However, this argument does not allow dioxin risk management regardless of the context of other risks. The implications of other risks may also differ from case to case.

The level of risks associated with DLCs in Baltic fish has gone down (5.2.3). The key question regarding quantitative risk management goals then becomes: how much lower levels of risk, exposure and (fish) concentrations are still required, considering that no clear human health effects from even several-fold higher exposures have been found? A greatly and rapidly reduced level from the present may be inappropriate, if causing other offsetting risks to health, and other harmful effects.

Developments with respect to background and **reference values** can have important policy implications, in addition to those for inventories and monitoring (Green et al. 2000, Jones et al. 2000). The background level is not easily definable even for exposures, as it varies in time and space, among food and other matrices and between receptor groups. It is also influenced by natural formation of DLCs, including PCDD/Fs (e.g. Hoekstra et al. 1999). Thus, it is not possible to set a goal of zero emissions or zero dioxins in food, not even in the long term. However, most of the dioxin fluxes and pools still are caused by anthropogenic emissions (cf. 3). They will also continue to be so for a long time in the Baltic, as the accumulated pool in the sea keeps cycling also to food chains, and as additional anthropogenic emissions after lags reach the sea from the pool retained in the catchment.

In principle, management goals could be set in terms of the magnitude of effects and risks. However, effects of dioxins are caused also by other agents. Thus, the management goals with respect to effect levels may need to be stated as excess risk (or risk range), e.g. a probability of

10^{-4}–10^{-6} lifetime excess mortality risk (the latter representing a background or negligible risk).

Implications of the targets of protection

Most existing strategies for dioxin management target humans. There are attempts to develop broader environmental strategies for dioxins and PCBs (e.g., Van Tongelen 2002). Risk management with regard to **domestic and farmed animals** will be important in this connection as well, also as part of an integrated approach to food production systems and food chains (cf. 8.4).

Different conclusions of risk management needs and goals can be produced depending on whether **human health** or also **other concerns** are in focus. For instance, Ross (2004) concluded that because PCB exposures from environmental sources do not pose a significant health risk, little benefit to public health could result from continued remediation of PCB sources. The conclusion might be affected by the inclusion of other organisms. Likewise, the evaluation of the urgency and goals of further reductions in DLC levels in the Baltic and also choices regarding where and how they be best reduced will depend in part on the relative weight given to human and non-human animals. For instance, when reducing risks to humans or to otter the focus of the policies and decisions may be different than in the case of seals, e.g. as the relative importance of land-based activities in comparison to sea-based activities may be greater (even if they have importance also for seal protection on a preventive strategy).

An important point is whether specific quantitative management criteria would be provided for **children** or the **whole lifetime**, as presently. The latter approach is justified by the long-term behaviour of DLCs, the former by the concern for adverse effects at critical developmental stages. These objectives are not exclusive. Patandin et al. (1998) pointed out that as most of the health risks have been found to be related to *in utero* exposure, strategies should be directed toward reducing the intake of DLCs at all ages, instead of focusing only on breast-feeding. However, it is difficult to extend quantitative goals for short-period *in utero* exposure. As regards lactation, it carries great and overriding health benefits. Also for this reason it seems problematic to introduce some specific intake benchmark for the breast-feeding period, or recommendations to limit this period. The observed reductions in intakes and risks and the projected continuous (even if slower) reductions are an additional factor when considering specific management goals for young developmental stages.

Implications of integrating health and other benefits from fish and fisheries

If **benefits** from bearing risks caused by DLCs and, conversely, risks from losing benefits due to management measures are considered, management goals may radically change (cf. 8.2.1). Also in the case of Baltic fish, goals that may be justifiable when only accounting for toxicological risks from dioxins may not be appropriate when also risks and impacts from reduced or altered fishing and the beneficial health effects of fish consumption are included.

The inclusion of benefits means that also goals need to be defined more broadly. What seemed to be a relatively uniform goal and straightforward task of reducing the feared toxicological risks (to humans) from dioxins in fish, as in any food, turns into a more complicated goal and task of protecting and enhancing (human) health in a holistic perspective, considering effects of various dietary items. The benefits from fisheries do not only imply the need to alleviate, as a secondary objective, the socio-economic losses to and through fisheries (cf. Joas et al. 2001). Instead, the benefits of fish become a more important decision factor. This leads to multi-criteria and multi-objective decision-making.

While the consideration of health benefits along with health risks from fish has been mentioned in some assessments (e.g., Johnson et al. 1996, SACN and COT 2004, Tuomisto et al. 2004b, Leino et al. 2005, SPCFC 2005, cf. 5.4.4, 7.4.1), the implications for risk management have been little addressed. An objective of continuous consumption of also contaminated (Baltic Sea originating) fatty sea fish does not follow automatically from the goal of holistic health care, as benefits may be obtained also from other fish and dietary ingredients. On the other hand, fish consumption has also other health benefits (cf. 7). Many additional management goal considerations are thus introduced.

Goals in terms of causes of risks and risk stages

The definition of management goals is affected by what are the **causes** of observed developments. It has been pointed out that reductions in dioxin

levels may have occurred more by accident than by design, e.g. as a result of changes in heating and other such diffuse and secondary sources (Jones et al. 2000), instead of being due to reductions of primary point source emissions. In this case management goals are more challenging and a relaxed attitude to risks is not so easily affordable. However, there is also evidence for risk reductions due to reduced production and use of precursor chemicals (cf. 7.2.1).

Goals can be set for **dioxin formation** in various stages, and these can be prioritized (cf. 7). Goals can be set for prevention of dioxin formation already by influencing material choices and technological and societal activities, and for precursors that only indirectly and more remotely form PXDD/Fs.

The overall approach in terms of the relative weights given to a **fish oriented** or a **source or sea oriented** strategy is related to goal setting. If for instance a comprehensive and preventive approach to dioxin risk management is adopted, more extensively addressing precursors, non-human receptors and beneficial and indirect impacts, then less reliance on the present regulation based on fish concentration levels may become natural.

Different management conclusions regarding Baltic fish may be produced if focusing on direct human consumption of fish or if also accounting for **indirect effects** through food production. This is related to framing of risk management with respect to compartments, organisms and sectors (see above and 8.4). The impacts of this dimension of risk are not clear. In general, it may increase the risks also to human health and thus add weight to restrictions of uses of fatty Baltic fish. However, this depends on the precise transfer of DLCs from Baltic fish and from other sources in and through the food production systems in question, and on the interventions and controls in these systems.

Implications of the time factor for goals

The time frames may be different for different goals. It may not be possible to lower dioxin intake as soon as it is possible to reduce emissions to the same degree, due to the lags in dioxin cycling. The inertia in management systems, e.g. regulations, infrastructures, scientific advice and know-how, also requires a long-term approach. It further acts as a counterweight to the dominance of acute contamination episodes and scares that tend to constitute key drivers ('management through crises').

Strategies and goals can be defined that will reduce risks immediately, such as through intake reduction, or more gradually, such as through environmental remediation, emission control or even prevention of dioxin formation. In intake reduction, some measures such as marketing restrictions have fast effects while others such as diet advisories will influence intakes more slowly (cf. below).

The generally decreased exposures to DLC also in Baltic biota may influence management goals. There are indications of plateaus in dioxin and PCB levels having been reached in some parts of the Baltic Sea system. Additional risks may also surface and counteract the general declining trend, just as unanticipated risk-reducing factors may surface.

Risk management policy conclusions

Goal setting for risk management must consider the framing of risks in terms of substances (DLCs), matrices (foodstuffs), exposed organisms (humans and others) and impacts (including beneficial). These considerations reflect the need to address DLCs in fish in relation to other concerns. The distributions and qualities of risks may also need to be taken into account, including their time dimension. Goal setting is in practice also influenced by management means in addition to risks.

It may be unnecessary and even disadvantageous to set goals in the form of concentrations in marketable fish at a very strictly protective level, as average exposures and also risks are considerably below peak levels, as no dramatic harmful effects have yet been proven in humans and also other animals have considerably recovered, and as fatty fish carries many benefits also to human health.

The above considerations are reflected in the definition of quantitative risk goals or benchmarks, specifically for human health (cf. above, Fig. 18). In this conceptualization, it could be seen that a host of factors influence what guidelines or standards are to be set. There is no sharp bright line in either human body burdens or intakes of dioxins, and still less so in fish levels, to discretely separate a certainly harmful and safe risk level for human health. Such risk management criteria or goals are approximations and consensus values which may be lowered or

increased depending on the safety and certainty desired and the requirements for proof of dioxin-linked effects (cf. 5.4).

8.2.3 The choice of steering instruments

It has been often declared that all possible measures should be taken to further reduce the risks from dioxins (e.g., Koopman-Esseboom et al. 1996, van Leeuwen and Younes 1998). Upon closer inspection it is clear that also on a multi-frontier strategy it would be unfeasible and even impossible to make "every effort". This may be precluded also by normative constraints. Efforts and resources may be wasted if ill focused. Efforts in some areas may preclude others, either directly and absolutely or indirectly or partially. Measures in many cases compete for resources that might be best used in other areas and to reduce other risks, dioxin-related or others (e.g. other causes to the effects for which also dioxins have been suspected). Some efforts will even be counterproductive and on the contrary increase risks instead of reducing them; this is a real possibility and threat if in addition to the risks directly and primarily ascribed to dioxins also the indirect risks of management actions are factored in, including the loss of benefits from fish.

In choosing steering instruments, the **contexts and procedures** for risk management need to be taken into account (cf. 6). It has been pointed out by Andresen (1996) in a comparative evaluation of international environmental commitments in Northern seas that for the Baltic more planning than implementation has taken place. This is in part due to the status of HELCOM instruments, being recommendations only and not legally binding.

It is inevitable and desirable that action is taken in many areas by a broad **selection of means**. Measures specifically targeting dioxins or fish or the Baltic as well as more general measures and instruments are needed. Extensively preventive measures need to be combined with prevention within exposure reduction. Some things are best made on EU level, some on other levels. However, a very undiscriminating approach to risks is problematic. There is a danger of wasting resources if e.g. all DLCs are handled similarly or if hotspot cleanup activities are rushed into on large scale, without due consideration of their consequences and contexts (cf. NRC 2001). In addition, an incremental approach allowing the development, experimentation and application of various measures also differentially may be natural instead of a synoptic one based on a 'total plan' that may never be achievable (cf. Fig. 20, Hildén 1997b). Thus, a balance is needed between focus and breadth of management approaches and measures.

Choices of steering instruments and their combinations are needed along the whole risk **chain**. The choices depend e.g. on what is possible (cf. 7) and on how much and how soon risks - and what risks - are to be reduced. Within prevention of dioxin formation, multiple steering instruments are available, as prevention is achieved both on the level of root causes and of chemicals policy and on the level of steering operations in facilities with dioxin-forming processes. Within reduction of human exposures, the choice with regard to Baltic fish dioxins is basically between regulation of fish consumption and information steering based on dietary advisories.

The **integration** of dioxin management **with other processes** and instruments, e.g. in prevention of industrial emissions, presents important opportunities and also constraints (cf. 8.4). For instance, phase-out of dioxin precursors may have other important advantages and disadvantages, also (at least indirectly) to human health and the environment.

The present EU strategy is essentially based on **normative regulatory steering** (cf. 6, 8.1). Also in the EU approach, other steering instruments, including economic, can be discerned as more indirect means of managing dioxin risks. An information based steering dimension is included e.g. in monitoring, and such an approach is even allowed as a primary strategy on the basis of the derogations granted to Sweden and Finland in fish dioxin risk management. Nevertheless, as a whole the EU approach can be considered primarily regulatory and focused on quality of foods and feeding-stuffs. Such a management approach assumes that the decision whether fish can be eaten cannot be left to the consumer. This is different from many other commodities posing health risks.

There are also cases of dietary consumption where toxicity is caused in a subgroup of people but the decision to avoid the risk is left to the consumer. A difference between dioxins in food and such other dietary items partly lies in that PCDD/Fs are inadvertent by-products, while the hazardous substances in some other commodities are inserted on purpose or formed naturally. Another relevant difference may be

that DLCs in food are not only hazardous to the first consumer on the basis of immediate effects but will continue cycling, potentially causing further risks (cf. SCAN (2000). Therefore, also the steering instruments will differ from those of other substances.

Risk management for the protection of **non-human animals** cannot be achieved by either fish consumption advisories or fish marketing restrictions. Reduction of ecotoxicological risks will require an approach that is directed more to the sources, emissions and immissions of dioxins, and to measures controlling their cycling in the sea. Within post-sea and other subsequent risk reduction options, protective and compensatory mechanisms emerge as options for non-human animals, e.g. in the form of protection areas and provision of alternative food sources (cf. 7).

8.3 Assessing qualities and impacts of strategies

8.3.1 Evaluation criteria

General

Many aspects of risk management strategies (and other management strategies) need to be considered in evaluating them. These may be divided e.g. according to the interacting main facets or dimensions of strategies and their sub-categories (Fig. 21).

Many evaluation criteria are also conceivable with regard to management strategies for DLCs in Baltic fish. It is expected that there is great variation in views among experts, managers, stakeholders and others regarding evaluation criteria and strategies in general; some empirical evidence has been obtained for such variation in views on dioxin risk management (cf. 8.4.2).

The difficulties in evaluating the pros and cons of management approaches to dioxin risks from Baltic fish are due e.g. to the following factors:

- There are many different kinds of management approaches and strategies, e.g. in terms of their scope and level (EU-wide or local hotspot management, and so forth)
- Many of the various alternative options are not known at all or are known very incompletely; for instance, it is unclear what the potential as well as the limitations and the wider socio-economic impacts of surrogate diets to Baltic herring might be, in terms of risk-benefit ratios
- The consequences of management strategies are not well defined and known. They include indirect impacts such as benefits through a learning process; socio-political benefits of trust building; benefits for reduction of other risks, e.g. by abatement of other harmful substances; or counter-veiling risks e.g. through losses of health benefits and other values of fish. Consequently, the effectiveness of strategies and instruments may require multiple analyses (cf. Andresen 1996).
- The risks, benefits and management burdens are to some degree distributed asymmetrically, e.g. with regard to species and groups (including future generations)
- The inherent and fundamental subjectivity and context dependency of evaluation
- The existence of many different and partly incommensurate (and non-quantifiable) goals.

Evaluation of strategies

For initial evaluation of management strategies for dioxins in Baltic fish and more generally, the criteria used by Hildén et al. (2002) may be applied in a modified selection:

- **Relevance:** This fundamental criterion is related to the framing and scope of strategies, and to the integration or interaction with other strategies or, more generally, other areas of policy instruments and sectors. On the other hand, relevance also is dependent on in how far the issues within a scope are addressed in a meaningful (e.g., not only superficial and formal) manner. The key general question here is, do the goals and concept of the strategy cover key problems in the intended fields of policy, in this case including environmental, health, food, fisheries and marine policies.
- **Impacts:** These may be divided e.g. into health, ecological, socio-economic, cultural, technological and systemic. They include direct and indirect impacts. Importantly, intended (and anticipated) and unintended

impacts are distinguished, i.e. whether identified impacts are clearly due to the strategy or to other factors.

- **Effectiveness:** To what degree do the achieved (or foreseen) impacts and outcomes correspond to the set goals and intended outcomes?
- **Efficiency:** Do the results justify the resources used? This is a cost-results criterion in which benefits are not valued in monetary terms. Another possibility is to use the cost-effectiveness criterion: Could the results be (have been) achieved with fewer resources?
- **Acceptability:** This is essentially related to the acceptance of the various stakeholders and actors, including consumers. On another level, acceptability can be evaluated with regard to the legal basis of management, i.e. in how far it fulfils the requirements and boundaries set up by existing laws, regulations and other normative prescriptions.
- **Transparency:** How has the strategy been conceived, and its foundations and foreseen impacts evaluated and documented? This is related to knowledge, participation and trust
- **Equity:** Whom does the strategy involve and what is the distribution of its impacts among groups?
- **Flexibility:** How can the strategy cope with changing conditions, and in how far can it be modified to account for additional considerations? These may include additional sectors, more detailed e.g. regional, food and risk group specific factors, or considerations that change with time. Flexibility is also related to how easily the strategy can be combined with other strategies.
- **Predictability:** In how far can the impacts, outputs and administration of the strategy be foreseen? Is it thus possible especially for those affected to be prepared and take into account the strategy? This is related to transparency.
- **Sustainability:** In this connection, this is related to the continuity, contingency and long-term durability of the strategy, not to sustainable use of resources (cf. environmental impacts).

Additional factors that were prioritized by experts in a questionnaire survey on dioxin risks and their management are related to the above criteria in the following ways:

- Preventiveness: Can be considered to be part of effectiveness (and indirectly efficiency)
- Monitorability: This is a condition of transparency and of evaluation of impacts and of effectiveness
- Innovation incentives: This is related to broader impacts and also to flexibility and sustainability
- Stakeholder involvement: This is explicitly included in equity
- Well-established normative basis
- Coverage: Essentially included in relevance
- Ease and speed of implementation: Can be seen as sub-categories of effectiveness.

Only some of the above criteria are discussed in detail below and applied to Baltic fish DLCs. The EU strategy for dioxins and PCBs in food and feeding-stuffs and its alternatives are primarily evaluated. This is justified, as the present management discussions and activities also in the case of Baltic fish revolve around this strategy as a EU-wide regulatory instrument. Mainly such alternatives are discussed and compared with the EU strategy that address the reduction of human exposures to dioxins in fish and fisheries products use stages. However, also other strategies with different scope and focus, such as strategies in dioxin prevention and emission control, will be discussed at a more general level and including some specific examples of their characteristics, impacts and potentials.

Procedures and criteria for general assessment of management options have been developed in EU (e.g., Joas et al. 2001). Some guidance has also been provided for how to evaluate risk management options, particularly for risks from dietary DLCs. FAO and WHO (1997) mainly discuss identification and then directly selection of options (cf. Annex 11). The procedures for developing risk reduction strategies under the Existing Substances Regulations provide frameworks for options evaluation in qualitative terms, focusing on advantages and disadvantages of alternative substances. The guidance in USPCCRARM (1997) is mainly concerned with economic aspects.

An important general aspect in all strategies is their flexibility and adaptability. This has been discussed in connection with dietary guidelines by Beaton (2003) who noted that the public and professionals have come to expect new guidelines approximately every 5 years. Revisions at such a pace may be important for adaptive management,

but may in some respects be problematic with a view of the consistency, predictability, equity and thus also efficacy of policies and strategies.

8.3.2 Interpretations and evaluations of some relevant present strategies

EU strategy for dioxins and PCBs particularly in food and feeding-stuffs

Relevance and framing

The EU strategy on dioxins and PCBs (EC 2001) and the specific recommendations regarding these compounds in food and feeding-stuffs (EC 2002a), can be considered to be narrowly focused. The following limitations in them deserve particular mention:

- They include initially *only 2,3,7,8-PCDD/Fs* at an operational and detailed level; dlPCBs are to be added based on additional assessments and development of procedures, while other DLCs such as the corresponding brominated compounds and dioxin-like PAHs are not considered at all
- They address *only risks* from dioxins and PCBs in food, *not benefits* from food such as fish
- They address mainly (food toxicant related) *human health* risks, much less ecotoxicologial or other ecological risks, or still other risks (e.g. to safety and economy), although increased consideration of environmental risks has been stated as a development need
- They cover very insufficiently *prevention* of dioxin formation (see Annex 11). There is little description also of emission control, and it is on a rather vague level, mainly describing voluntary and information measures although particularly in this area there are regulatory instruments in place; their key links and coordination with the strategy are not described. It has been explained that the impact of 'environmental' measures on dioxin levels in the different feed materials cannot yet be accurately foreseen, which does give some justification for such a delimitation of the strategy. However, also within food and nutrition few measures apart from those based on food and feeding-stuff quality regulation have been included (cf. below). Many post-production and post-supply management options are omitted, including diet advisories even as an auxiliary approach despite their broad application and great importance.
- They include limited consideration of *related* environmental, fisheries and marine *policies* and strategies; that is, the dioxin strategy and recommendations are not explicitly linked with such other policies and strategies, at a level that would e.g. recognize coordination needs
- They do not address particular *regional* aspects e.g. of the Baltic and its fisheries; to some extent this is natural in a community-wide strategy ('narrowness in breadth' due to lacking specificity), but issues in aligning community and regional policies are not even identified (cf. SPCFC 2005).

Specifically, the recommendations for dioxins and PCBs in food and feedings-stuffs are limited by their focus on management by regulation of allowable dioxin levels in food and feed, giving little consideration to other areas and types of management (e.g., to connections between dioxin levels in various foods and feeding-stuffs and dioxin intakes and other fluxes). This is an inherent limitation of recommendations that purportedly address food and feed quality. However, the above other aspects and links should be identified and explicitly mentioned also in connection with such recommendations for truly comprehensive risk management even within the food safety area.

The strategy and associated recommendation for dioxins in food and feeding-stuffs are thus not comprehensive and can not be claimed to be surely the most relevant and adequate management approach, for Baltic fish or for food or feeding-stuffs more generally. The above limitations carry the risk that the present strategy may not lead to selection of goals and instruments that are the most efficient and justified ones, e.g. adequately covering all relevant areas and issues. Thus, its impacts and consequences, overall characteristics, development needs and alternatives need to be evaluated more extensively and closely.

Impacts

A) General

The EU strategy for dioxins and PCBs and associated recommendations for these compounds in food and feeding-stuffs have various impacts in the case of the Baltic Sea: on the human health risks, on environment (an unintended type of

impact), social and economic conditions, trade and industry, political processes, information and technology, as well as in other areas.

The impacts of the present strategy are as yet limited from the point of view of Baltic, as Finland and Sweden have been granted derogations. The impacts would be much extended, should the strategy be implemented also in these countries. Some of the impacts are not restricted to the Baltic; for instance, impacts on Swedish and Danish Baltic fisheries will have repercussions on (and are conditioned by) fishing in the North Sea and thus on processes and developments of a larger region.

The impacts of the EU strategy are felt on many levels, local to EU-wide, and in the form of both concrete and detailed as well as more general changes. They vary according to who and what sectors are affected. In addition to immediate and direct impacts e.g. on dioxin intakes after fish marketing restrictions, the strategy has long-term and indirect impacts in all the above areas, including structural influences. This makes prediction and analysis of impacts difficult, but at the same time underlines the need to at least identify possible impacts and their links, co-factors and key characteristics.

The impacts of the strategy have not been hitherto extensively and transparently explored even for health or for ecological, technical and socio-economic aspects. Generally, it can be assumed that its limitations, likely impacts and perhaps particularly the lack of risk-benefit considerations may lead to unfocused and inefficient regulation, e.g. excessive in some areas and lacking in others. On the other hand, the strategy has grounds and justification e.g. in certainly ensuring reduction of incremental exposures to DLCs from foods and feeding-stuffs with higher DLC contents, such as Baltic fish. The balance of its pros and cons and the resultant needs for alternative or modified strategies are not obvious, and should be evaluated in a broader context, in more detail and more systematically.

The consequences of the strategy cannot be predicted in detail, and more work should be done on trying to identify possible outcomes and their implications. It is for instance unclear how regulation of food quality will prompt measures toward dioxin sources, as the impacts of alternative courses of action are not explicated and as the coupling with prevention and emission control is weak.

A significant category of impacts is associated with changes in dietary habits as a result of restricted marketing and consumption of fatty Baltic fish. While such changes may be precisely sought based on toxicological health risk assessment and on a policy and strategy that is mainly concerned with these risks, they may have important unwanted (as well as some wanted) consequences also in other regards. Altered dietary habits have health and nutritional effects but also broader socio-cultural impacts (not easily separable). Some of such indirect socio-cultural impacts are not specific to the EU regulatory steering strategy, but can arise also in an information steering approach (see below).

B) Health impacts

Strict limitations of Baltic fish marketing and thus of consumption and fishing are problematic for health reasons, despite the reduction of risks from dioxin exposure, especially when health benefits from fish are factored in. There is a considerable risk of losing the benefits from additional dioxin intake reduction if simultaneously cutting nutritional benefits from fish.

Limitation of Baltic fish supply to the market would reduce the intakes of DLCs in fish from the present levels, thus reducing the risks caused by these compounds. The reduction of the human health risk from DLCs in fish would be near complete, but the reduction in total risk on the average low (cf. 3.4, 5.4), as most DLC intake for most people around the Baltic comes from other foods. The benefits from this would vary according to the segments of population, and the relative benefits would depend essentially on the intakes from such fish in relation to the intakes from other food.

The reduction of health risks caused by DLCs can be accomplished extensively on the basis of the EU strategy in the sense that also the use of fish in feeding-stuffs is controlled and the associated flux to food production chains is reduced. Alternative and modified management approaches (cf. 8.4) will need to give particular consideration to risk management in this area.

A key factor in the development of unintended health consequences (related to benefits from fish) is in how far fatty sea fish can be replaced by dietary supplements such as clean fish, clean fish oil and even vegetable-based fatty acid products (Melanson et al. 2005, SPCFC 2005, Foran et al. 2005a,b). There is evidence from studies especially in cardiac health effects (cf. 4.4.2) that the

benefits from such supplements are smaller than from consuming fish itself, while their utility is high e.g. in the case of vitamin D deficiency. Some fish oil based dietary supplements also from areas outside the Baltic have contained considerable levels of dioxins and dlPCBs (FSAI 2002, Jacobs et al. 2003, cf. Hites et al. 2004a), and may thus not offer a good alternative to fatty sea fish from the Baltic. This depends e.g. on the region, species, tissue and methods of production of the oil. Oil from dioxin-rich fatty fish may be industrially cleaned to ensure safer use of the catch (cf. 7). However, even if developed and extended, removal of dioxins from fish lipids impairs the nutritional value (and desirability and therefore consumption) of fish. Therefore, a key alternative is other dietary ingredients.

It is not clear whether the assumption holds that people instead of consuming fish would actually and automatically shift to more unhealthy fats. This assumption lies at the heart of the claim that reducing consumption of fatty Baltic fish would cause counter-veiling health risks (especially to cardiovascular health), and therefore requires careful scrutiny. There are other alternative fat and protein sources such as low-dioxin fatty fish, also farmed rainbow trout from the Baltic fed clean feeding-stuffs, and fats and proteins in more vegetarian diets (e.g. from legumes). There are also some indications of changes in dietary habits (in Denmark) where reduced fish consumption was not accompanied by increased meat consumption; instead, the consumption of other food categories was increased (Fagt et al. 2002). However, these alternative diets may be less healthy than fish. Fish consumption has also other health benefits than those from PUFAs. This would strengthen the argument for consumption of fatty fish, even from the Baltic, instead of surrogates.

People can be educated and supported to shift to healthy alternative fat and protein sources instead of saturated animal fats, much in the same way as they are educated and advised also to avoid dioxin-laden fish. There are indications that such information steering approach can be effective (cf. 7, 9). Thus, this argument for supporting diet advisories as a risk management alternative to fish marketing restrictions might possibly also be used as a support for alleviating the harmful effects of such restrictions, and therefore indirectly for the restrictions as well; i.e., the possible success or failure of diet advisories should not be used selectively only to back up one strategy and not the other.

Nevertheless, presently it seems reasonable to assume that most people would after dropping e.g. Baltic herring or salmon from their plate move to consuming more animal fats or other generally unhealthier food categories, and that it would be difficult to turn such fish eaters e.g. to vegetarians on a sufficient scale. It can be suspected that such attempts would meet with considerable cultural and social resistance and would have important limitations. Such attempts would additionally cause subsequent other health problems and risks particularly to some sub-populations for which a diet based on vegetarian proteins and other ingredients is unsuitable, also as such protein sources might be insufficient for key groups like growing children. The beneficial effects of fatty sea fish are not only based on the high content of LC n-3 PUFAs, but the overall fat composition and other ingredients.

Cardiovascular health improvement is only one although likely the most marked health benefit from consumption of fatty fish (cf. 4.4.2). In this respect, the benefit/risk ratio (e.g. in terms of disability adjusted life years, DALYs) is likely to be high for older persons, while it can be much lower for young persons and developing fetuses. As to other health effects such as those on development, the benefit/risk ratio for consumption of wild fatty Baltic fish may also be low; the risk for adverse effects may even exceed that for loss of benefits for some population segments (cf. 5.3.4). A quantitative assessment is precluded by the lack of dose-response, population risk and DALY data for the various kinds of effects and sub-populations (cf. Cohen et al. 2005). For some other health benefits from fish, alternatives to fatty (dioxin-laden) fish may also have considerable disadvantages.

Due to the ongoing decline of Baltic herring fisheries in general, it may be that herring is already such a marginal and under-valued product that the herring fisheries will continue to dwindle. This will further reduce the overall health risk from dioxins, although price declines may on the other hand have the contrary effect of inducing poorer and less well-advised persons to hazard a highly herring-based diet as long as such fish are available. On the other hand, the loss of health benefits from such fish will depend on the time course in shifts to other diets.

C) Ecological impacts

Altered fisheries and particularly a reduced herring fishery have potentially important ecological effects. The population structures of herring and dependent other species would

change. The anticipated immediate effect on herring stock from relieved fishing pressure would be a deceleration in the growth rate of herring and sprat and increased dominance of older age classes. This would in turn probably increase the average dioxin level in the herring biomass (see Harvey et al. 2003) and the intake at least by non-human herring consumers. It is uncertain whether such changes could reverse the recovery trends of the populations vulnerable to DLCs. It is possible that fisheries restrictions on the basis of DLCs would primarily have other than toxicological effects on the ecosystem.

Increased shift to Baltic sprat by the Baltic fleet would cause alterations affecting the whole community and ecosystem. Such consequences for the ecosystem will be mediated by the fishing effort and will depend on the interplay of TACs, on possibilities and constraints for using sprat (of various age classes and dioxin contents) especially as fodder, and on the resultant development of sprat fisheries economy. Thus, there is an intimate interplay between ecological, socio-economic and technological-systemic (and unintended health) impacts on several levels, also regionally, and involving many variables. Other kinds of ecological impacts of declines in herring and sprat fisheries are possible, including adverse developments (from the point of view also of humans), e.g. on plankton due to altered predation and on other fish consumers due to changes in herring and sprat stocks (cf. 4.3.4, Hansson et al. 1990). Additional studies are underway of the effects of altered fishing patterns on dioxin biomagnification through population structure changes.

In response to the proposed EU strategy, Finland expressed concern for the impacts of decreased fishing on the ecological state of the Baltic. The end of its herring fisheries or even other fisheries may change the ecosystem. Whether deleteriously or not is a value judgment, also as it has not been proven that the (expected) higher levels of DLCs would cause harmful effects. In evaluating the possibilities for ecological and other impacts from altered fisheries as a consequence of the EU dioxin strategy, also interactions between various fleets and their operations and between the various stocks (and thus human-ecosystem interactions) need to be taken into consideration. It is conceivable that if herring fisheries (in part through reduced demand and prices) decline, this fleet shifts increasingly to sprat. This would help maintain open-sea fishing for clupeids and could also be utilized for cleanup fishing. However, also the feasibility of sprat fishing is affected by the implementation of regulations on the acceptable quality of fish-based feedings-stuffs in EU, including its traditionally sprat-fishing new Baltic member states (see below).

D) Socio-economic consequences

The socio-economic risks and impacts from reduced fishing and fish consumption are wide reaching (regionally), and not limited to industrial and nutritional economy and employment. These impacts are coupled with cultural roles of fish consumption and fisheries that are hard to recreate, once lost.

The scientific and technological policy options assessment for the European Parliament (Joas et al. 2001) considered the effects on the fisheries industry resulting from the European Commission proposals on dioxin content of fish, fish oil and fish meal (cf. 6). To evaluate these effects, the concept of socio-economic conflict potential was used. High conflict potential means a risk for jobs and turnover due to dioxin levels exceeding the limit value that would prohibit free marketing of the fish or fish products. According to the assessment, the conflict potential for Danish, Finnish and Swedish fisheries is the highest, and considerable effects on turnover and jobs may be expected (Table 35).

Denmark is the most important producer of industrial fish in EU. More than 200 000 t a^{-1} of catches (i.e. 16 % of the total catches of herring and sprat) shows a high conflict potential. The share of the catch and the amount of jobs with high conflict potential are less than those in Sweden and Finland, as Denmark has large Atlantic fisheries. However, potential impacts on **turnover** are considerable, especially if lower conflict potentials are included. This has consequences for the Danish fisheries industry due largely to foreseen restrictions on Baltic fisheries. In Finland, 100 000 t a^{-1} of the relevant catches of sprat and herring show a high conflict potential. This is in absolute amounts less than in Denmark or Sweden but, being solely from the Baltic, includes all Finnish industrial catches. An important consideration in this connection is that the Danish North Sea and Atlantic fisheries may benefit from increased certainty of risk management under the EU strategy.

The potential impact on **employment** would be regionally significant. Finnish herring and sprat fisheries are labour-intensive per unit catch, and therefore the Finnish fisheries industry is likely to be particularly affected by the proposed limit values.

Nearly 90% of the Swedish industrial herring and sprat fisheries are also based on the Baltic and have a high conflict potential. The estimated impact on jobs is even greater than in Finland, and a great potential impact on turnover is also foreseen. Significant consequences can thus be expected for Swedish fishing industry, too. On the other hand, the above-mentioned potential positive impacts on the Danish (and the North Sea based part of the Swedish) fisheries and related industries also in terms of employment should be taken into account.

In addition to the direct impacts estimated by Joas et al. (2001), a still larger part of the catches, fisheries and economies may be affected indirectly e.g. through a generalized suspicion of the quality of fish, through lacking support for fisheries and other such effects.

Within fish processing **industry**, in the EU roughly 1000 **jobs** depend on the production of fish oil and fishmeal, 46 % thereof in Denmark. Joas et al. (2001) estimated a conflict potential for the c. 1000 jobs in the production of fish meal and fish oil, assuming no increased imports. It may be roughly estimated that most of the jobs in this branch having a high conflict potential are in those Danish industries that are based on Baltic herring and sprat, as these have the greatest average dioxins levels of EU fisheries. No assessment was made on the possible conflict potentials for and impacts on the processing industry in the field of fishmeal and fish oil, due to considerable uncertainties.

A loss of demand for raw material exerts a **negative feedback** on the fishing industry, which may finally lead to additional job losses and turnover reduction (cf. above). Reduced production of fish meal and fish oil will also impact the aquaculture industry as it may be difficult to satisfy the demand for fish feed. As discussed by Collins et al. (1998), there is a level of contaminants in the fish that triggers a switch in price, but how rapidly depends on information to consumers. This level varies among the various fisheries and consumer groups, and also depends on what kinds of risk perceptions they form on the basis of both this information and other factors.

The job losses due to the limit values are of **broader social significance**. The employment effects are distributed mainly to sparsely populated areas and rural areas suffering population losses due to outbound movement (migration loss). The Finnish response to the proposed strategy was also concerned with individual fishermen, who are and would still more be bearing the consequences of the regulation while neither responsible for the contamination nor able to influence the situation.

Table 35. Estimated social-economic conflict potential due to the EU recommendations for maximum levels of toxic PCDD/Fs in food and feeding-stuff (EC 2002a) for the fisheries industry in Denmark, Finland, Sweden and whole EU, in 1000 t a^{-1} and M€ a^{-1} (from Joas et al. 2001). Most figures have been rounded to one signifying digit. Note that the corresponding conflict potential and associated socio-economic impacts would be approximately doubled if including dlPCBs, other conditions being equal.

Member state		Total	Low conflict potential	Medium conflict potential	High conflict potential
Denmark	Catches	1400	1000	200	200
	Jobs	1400	1000	200	200
	Turnover	100	70	10	20
Finland	Catches	100	0	0	100
	Jobs	400	0	0	400
	Turnover	7	0	0	7
Sweden	Catches	300	40	1	260
	Jobs	400	50	0	350
	Turnover	20	3	0	20
Whole EU	Catches	3000	2000	300	600
	Jobs	5000	3000	500	1000
	Turnover	200	100	20	50

Herring and sprat fisheries in the Baltic Sea, and other major Baltic fisheries potentially limited by proposed limit values for dioxins and foreseen regulations on PCBs, have **cultural and general** social importance that extends far beyond the living conditions and economies of rural areas. They represent a way of life geared to varied traditional and novel harvesting and uses of renewable natural resources.

Many **other socio-economic impacts** are also likely (cf. Joas et al. 2001). These include structural changes in the fish processing, lagged effects on investment in this sector, and impacts on regional development (mainly in coastal areas with important fisheries).

It is possible that the social and technological conditions are already so far impaired, for reasons also unrelated to dioxins, that Baltic herring fishing has become marginalized. However, a retained and refined ability to influence dioxin fluxes may be valuable (se below and 7). Also the cultural including educational significance of herring fisheries needs to be acknowledged in its development.

E) Technological and capacity impacts

A crucial area that has received little attention is the technological and systemic impacts on fisheries from extending the EU strategy (e.g. to new substances, or to the countries now granted derogations). Such

impacts are linked to socioeconomic and ecological impacts. Technological impacts have a particular role as mediators between human and ecosystem processes and resources.

The decline and, after some breakpoint of feasibility, collapse of Baltic herring fisheries would **impair capabilities for fishing** out dioxins and managing dioxin biomagnification by focused and adaptive fisheries management (cf. 7). This is a consequence of the EU strategy's perspective on fish and fisheries only (at least as explicitly stated) as carriers of risks, not as vehicles of solutions.

Another consequence would be a **shift to fodder fish and fishmeal** (from young sprat and possibly young herring) and North Sea herring. This will be affected by the market for fodder fish, and thus also by the regulatory conditions for its uses in relation to its quality. A shift to North Sea fishing would be possible mainly for the Swedish and Danish fleets that already operate largely in these parts. Such changes in fisheries will influence ecological as well as socio-economic systems and conditions.

An important long-term systemic consequence related to the socio-economic impacts and processes from the collapse of herring fishing for human consumption also in Finland and Sweden would be that **investments in herring processing** capacity, e.g. filleting machinery, would no longer be feasible. Such a hiatus in the processing chain of herring that would make future recovery of this industry, even in an improved situation, very uncertain and difficult. Thus, there are indirect structural changes and thresholds and other non-linear impacts both within the technological and socio-economic spheres, jointly influencing in turn the supply of herring for human consumption and thus the intended and unintended (also harmful) health effects from reduced consumption.

Due to high PCDD/F levels, landings and marketing of Baltic **salmon** were prohibited in Denmark in 2004, but the ban was lifted in 2005. Also in Latvia, marketing bans were imposed based on WHO-TEq values exceeding the EU limit value in 5 of 9 salmon (ICES 2005b).

Restriction of commercial Baltic salmon fisheries in Sweden and Finland, in accordance with the regulations in the EU dioxin strategy and the recent prohibition in Denmark would cause additional changes in the fisheries structure. Because of the high dioxin contents in salmon, this fishery could be severely affected. Salmon fishing is already restricted as to catch areas, times and equipment. The recreational salmon fishing would probably benefit, but at the same time the restrictions would jeopardize a significant part of the total salmon fishery. The problems with a strict fish market regulation based management are exemplified by the Danish prohibition in 2004 which led to difficulties for the salmon market in the Baltic as a whole, as the Danish Bornholm is a key marketplace for salmon for human consumption (ICES 2005b).

F) Notes on other evaluation criteria especially within policy-level impacts

- **Transparency** is stressed in many connections in EU, notably in the Food Law (Annex 11C). The foundations of the EU strategy for dioxins and PCBs in food and feeding-stuffs are not wholly transparent, e.g. as to the methodology used in defining the maximum and target levels in foods and feeding-stuffs, or in defining efficient approaches and measures in risk management in general.
- Greater **trustworthiness** may have been sought by the EU dioxin strategy and recommendations, e.g. through 'ground rules' for managing dioxins in food. However, this trustworthiness is dependent on the consequences, including side effects and disadvantages, of the strategy.
- As to **flexibility and predictability**, changes in a market regulation based approach might erode trust and allow too much irregularity in regulation. On the other hand, these factors do not have to exclude additional considerations and measures of risks in separate management schemes.
- A limit value focused approach has some utility and benefits for **monitoring** systems. Nevertheless, this should not be exaggerated. It does e.g. not make much sense to stare at decimal points in fish levels when the total uncertainties in risks are much higher. This is true particularly when accounting for additional uncertainties in concurrent counter-veiling risks and other risk management impacts that are presently ignored in setting risk criteria and benchmarks.
- The limit value based management approach is likely to have '**back-stream**' effects. It may e.g. force reductions in the introduction of dioxins to the system as it creates pressures among the actors. Some of these effects may be intended and justified. However, also unwanted effects may be caused. There is e.g.

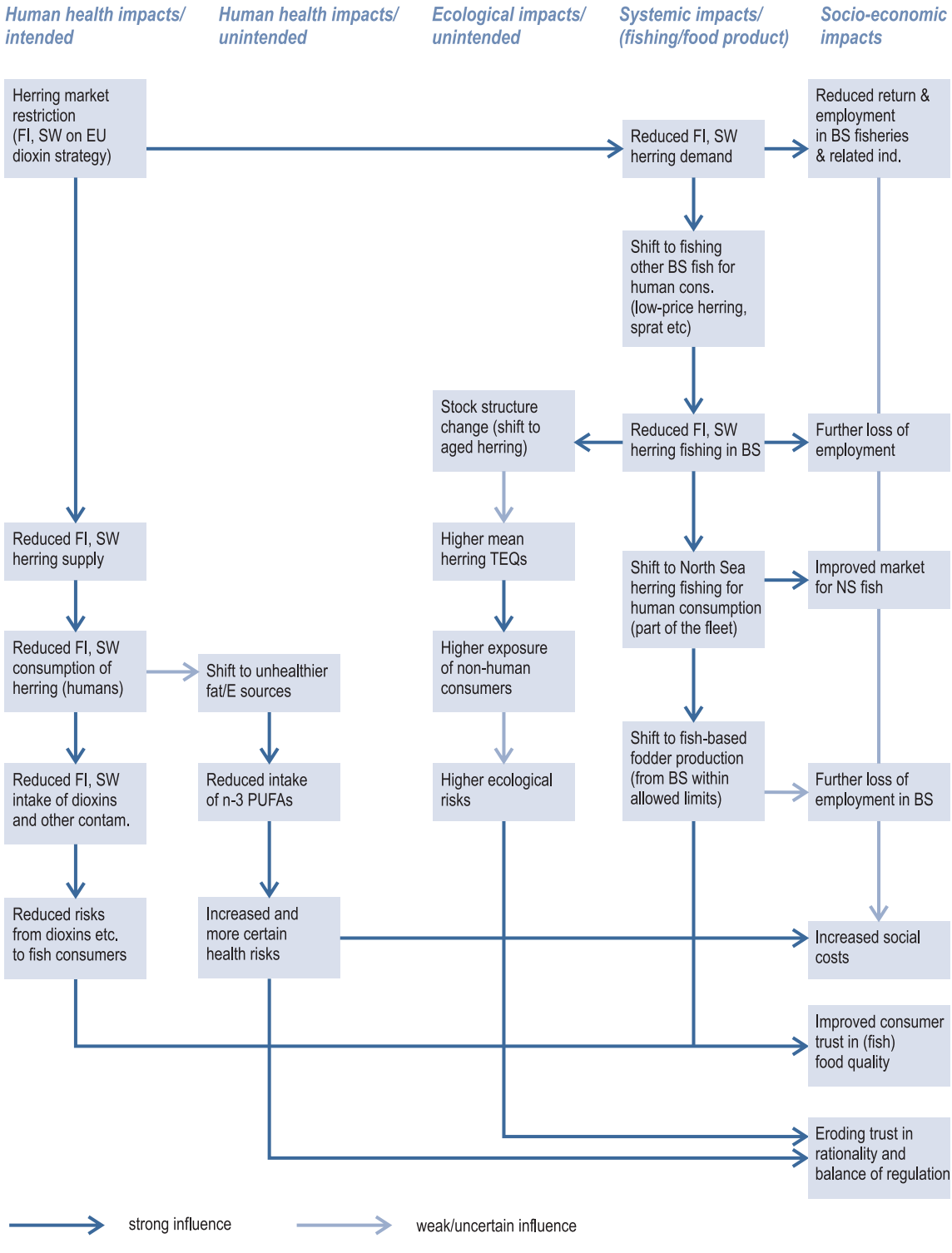


Fig. 22. Simplified influence diagram of certain, very likely (solid arrows) and possible (hatched arrows) intended and unintended impacts at multiple levels from implementation of the EU dioxin strategy in the Baltic Sea, with particular reference to herring fisheries.

the risk that such limits will force upstream actions in some sector or area more than in others regardless of where the greatest needs and gains for risk management are.

Some important anticipated consequences of the EU strategy within main categories of impacts can be illustrated schematically (Fig. 22). This is a simplified initial assessment of possible outcomes. Additional or alternative consequences can be envisaged. Some of the outcomes can be prevented, modified, alleviated or compensated by auxiliary measures, e.g. subsidies and other instruments. Already this limited analysis serves to illustrate that management of risks in Baltic fish is not as simple

and straightforward as conceived in the EU strategy, but involves many important impacts.

It can be anticipated that the EU regulatory risk management by market restrictions pending only on fish dioxin levels has consequences on several levels, including many potentially important unintended consequences. Some of these impacts may be regarded as positive, depending on whose perspective is in question and what the criteria are for positive impacts also on health. The EU strategy may also cause considerable problems even for the stated primary objective of health protection, as health benefits from fish are considered. In addition, possibilities or even high likelihoods of other kinds of problematic impacts are identified. These include ecological, socio-economic or cultural and systemic or technological impacts, the latter mediating many consequences as drivers.

Management of risks in such a complex setting by single-criterion regulation (of fish dioxin levels) is poorly founded, narrow in scope, potentially inefficient and unsafe, and may backfire. The goal of harmonized clean food market here threatens to clash with important health, ecological and social concerns, although being superficially health-protective.

Present alternative strategies and approaches

Identification of the main alternative strategic approaches

The main present alternative to the EU regulation of food and feeding-stuff dioxins in the case of the Baltic fish dioxins particularly within human exposure reduction in the food supply and consumption stage is that practiced in Sweden and Finland on the basis of derogations from the EU regulations. It essentially involves diet advisories coupled with other information-related activities (Lind et al. 2002).

This strategy leaves the decision of exposure to the consumer, based on information about risks.

This approach has not been laid down in such detail as e.g. the EU dioxin strategy, but is presented mainly in the national fish consumption advisory documents. However, the approach is partly based on extensive and prolonged previous management practices. It also has important parallels elsewhere, notably in the US both regionally, especially in the Great Lakes area (USEPA 2004 (2002), and nationally, especially in the form of the recommended strategy of the IOM (2003) on reduction of exposures to DLCs in the food supply. These strategic approaches display differences but also many important similarities. Conceptually and in terms of many principle-level, procedural and policy characteristics they represent the main alternative to the management approach in the EU strategy.

IOM (2003) presented a strategy to decrease exposure to dioxins and DLCs in the food supply. Being only a recommendation and a framework for further development, it is intended as a non-binding strategy, unlike the EU strategy that is more directly tied to regulation. However, the IOM strategy is authoritative and involves broad connections with government. Follow-up work on implementation and development is ongoing in an interagency group. This strategy deserves attention as representing information steering approach to management of DLCs in the food supply. In some respects, it is more comprehensive than the EU strategy, e.g. in covering additional DLC and in addressing variations, uncertainties and particular risk groups more explicitly (see Annex 12).

In addition to these main approaches to risk reduction in the food supply stage, other strategies exist in the Baltic Sea area, EU and elsewhere in other related areas such as emission control (cf. 6). These have been variably linked with food supply oriented strategies.

Evaluation of the characteristics and impacts of the management approach

The strategy on food dioxins adopted in Sweden and Finland has been based essentially on diet advisories, in Sweden already since 1995 (Lind et al. 2002). These constitute rather detailed, targeted, varied and well-established information steering instruments. The public confidence and trust in these advisories and the representative expert bodies is an important consideration. There is a risk that a market regulation focused management approach will erode the trust in the judgment and the capabilities of the consumers and the advisory organizations to deal with the risks (cf. above).

Also a diet advisory-based risk management strategy involves limitations and problems, and should not be promoted regardless of cost and risk. The coverage and effectiveness of risk reduction present problems. For instance, some people cannot be reached or are not influenced by advice.

If Baltic Sea fish above the EU limit values would be allowed on the market, it is unlikely that it would be more popular in other countries than Finland and Sweden, thus reducing the risk of high exposure in other markets. The provenience of some fish products is hard to establish; in particular

imported salmon may be sold as Baltic salmon and *vice versa*.

As a key difference from the EU strategy, the voluntary information based management approach excludes market control of fish and products. However, this need not be decisive for health protection.

The information-based approach is able to account for regional and other **specific conditions** and factors. Diet advisories are inherently capable of incorporating more variables at a much more detailed level than a regulatory strategy for fish and feeding-stuff quality control, as they do not have to be strictly prescriptive. Such an approach can capture risks both from recreationally fished and other fish. Advice on the frequency of fish consumption, selection of fish species and ages as well as catch areas is already given in both Sweden and Finland, and particular risk groups in terms of sex and age (children and women in reproductive age) have been accounted for. Similarly, there is long-time experience of devising and implementing advisories for consumption of contaminated fish in North America (see e.g. the methodology of USEPA 2000b, 2002c). However, there is still uncertainty as to the effectiveness and performance of such information.

The information-based approach can address **high-risk groups and situations** e.g. through parenting advice, day-care and schools. This may be a particularly efficient way to reduce risks through the personnel in charge of day-care and school meal planning in countries like Finland and Sweden having an extensive and well-developed school meal system. Thus, education and advice in this connection would not be needed directly to the high-risk young group which would require a more challenging risk and risk-benefit education task. A similar exposure reduction approach was proposed in the IOM (2003) dioxin strategy in connection with offerings of skimmed milk in school meals.

This approach is inherently **participatory** as it is based on information and advice to and interaction directly with consumers and other key actor groups. Partly therefore, such a strategy may be widely comprehensible and acceptable, despite the intricacies in the choices and details it entails, e.g. in selection of fish qualities and sizing the consumption.

Perhaps particularly, incorporation and **balancing of risks and benefits** in a many-sided manner can be easily facilitated, as the strategy is about people making such informed dietary risk management choices. Thus, the scope of such as strategy is much more relevant in terms of coverage of risks and impacts and for the task of holistic health care. Some of the health benefits can be secured while avoiding risks by adjusting the consumption of fatty sea fish in relation to other sources of PUFAs according to the distribution of risk-benefit ratios for different effects among different subgroups of populations (Foran et al. 2005b). This may be done perhaps more easily on the basis of information steering approaches involving dietary advisories.

An information-based strategy is easily compatible with other areas of management and with changing evaluations of risks and benefits.

An important consideration is whether the **efficiency and comprehensiveness** of such a more information-based management are sufficient. The effects of diet advisories for fish consumption on consumer behaviour and actual fish intakes have been studied little in the Baltic Sea countries. There are constraints and uncertainties in influencing consumer behaviour by diet advisories, due e.g. to educational, cultural and language barriers, as shown in the Great Lakes area (cf. Annexes 10, 12). It has been related by Lind (oral communication 2002) that even among Swedish medical students the awareness of fish consumption advisories has been found to be rather low. However, some contrary evidence pointing to the efficiency of risk reduction approaches has been published. Lind et al. (2002) mentioned that specific diet advisories for fish may have contributed to the notably lower consumption of fatty Baltic Sea fish by younger (<25 a) Swedish women as compared to older (>50 a) women. It is difficult to interpret such findings and to ascertain the relative contributions of advisories and other factors influencing dietary habits without additional empirical information.

The ethical question arises whether the freedom of a consumer to expose himself or herself to dioxins may violate the **rights of others**, including their offspring. However, it is normally assumed also in the case of non-regulated exposures that custodians are responsible for the protection of their children or protégées and are entitled to expose them to a multitude of harmful agents and depriving them of some beneficial agents (cf. 8.1.1). Historical and cultural reasons contribute to variations in the acceptance of food-related risks.

The validity and generalizability of such differences as arguments for an asymmetric risk reduction approach in the present case is questionable. These reference risks pose the legitimate question of why fish, even wild fatty Baltic fish rich in dioxins, should be regulated so

strictly in comparison to the above other foods that also include dioxins and related or other harmful compounds, given the beneficial health effects of such fish. The differences between fish dioxins and these other cases do not seem radical and valid as health arguments. They may thus not fundamentally preclude an alternative management approach allowing more consumer freedom and having potential other benefits (cf. 8.3). It is not clear whether or how such or other differences between dioxins and other areas of food safety (or other areas of health risk management) have been considered in formulating dioxin strategies.

The ability to choose between fish presupposes that some information is available on dioxin levels and that it is comprehended (not the case e.g. among small children, see Ponce et al. 2000). This information may be in the form of direct measurements of dioxin contents, but is in most cases indirect and approximating information based e.g. on the species, the size of fish, the part of fish, and possibly the catch area and other specifications. The information needed depends on the requirements of the decision maker (such as the consumer in a free choice based management approach).

Dietary advice may also lead to a perception that fish even more generally is unhealthy and thus decrease fish consumption overall, contrary to advice. Such experiences were made after the advisories discouraging the consumption of only some fish species from some regions due to mercury, also in Finland (cf. Wheatley and Wheatley 2000). Thus, the general recommendation common in nutrition to consume a varied diet is important (cf. Kris-Etherton et al. 2003). Care has also to be taken not to present too alarming advice, as some people may react strongly and even in unwanted ways.

Information must be given on which factors influence average and probable dioxin levels. Several possible strategies and levels may be conceived for such communication also in an information steering approach; e.g., a division in red, yellow and green signs for various fish based on their likely contaminant levels may be used, as in the Great Lakes states (cf. USEPA 2002d). Also labelling of fish (or in practice fish containers or vessels) may be justified regardless of whether there is actual prohibition of selling fish probably exceeding limit values, to indicate its provenience. However, this information on fish is only part of the information needed for informed and efficient risk management and avoidance decisions; also factors such as particular susceptibilities have to be considered, e.g. the possibly higher risk to infants. This is (and can increasingly be) accounted for in diet advisories (cf. 7).

An alternative, more varied and flexible management approach, e.g. one more source-oriented and focused on diet advisories and other information steering at post-sea stages, may become a rational choice if uncertainties in risks are taken into account. A variable accuracy of quantification in practical risk management may be acceptable and even necessary, due e.g. to the inherent variation in the accuracy of regulatory instruments.

Summarizing evaluation of the main alternative strategies of reducing human exposures

The initial comparative analysis of the EU and alternative approach to managing risks associated with dioxins and DLCs in Baltic fish is summarized (Table 36). The selection of impacts and evaluation criteria are not exhaustive. The evaluation of the pros and cons in various dimensions is inevitably somewhat subjective, but has been based as far as possible on facts and likely consequences identified in the previous strategy analysis. Arguments both in favor of and against have been sought, and every attempt has been made to view them and associated uncertainties in an unbiased manner.

The following summary points can be made regarding the above strategic approaches:

- **Health risks:** The fish market regulation approach on the basis of the EU strategy, although minimizing risks from dioxins, seems risky as this benefit is probably exceeded by the simultaneous loss of health benefits from fatty fish.
- **Ecological risks:** These favor alternative strategies allowing fishing of herring and sprat; the question becomes whether also other forms of herring fisheries than those serving human consumptions could be made feasible to ensure the capability to reduce ecological risks by fishing
- **Socio-economic impacts**, some beneficial but probably mostly unfavorable, will result from cessation of Baltic herring fishery. Alternative approaches to risk management offer possibilities to avoid or better alleviate unwanted impacts. Also in response to the EU strategy, instead of the present fisheries other livelihoods may develop, even fish-based (e.g. fodder and recreational fisheries). However, their importance would be less than that in the present open-sea fisheries.

Table 36. Summarizing evaluation of two principal alternative approaches to management of risks mainly to human health associated with dioxin-like compounds in Baltic Sea fish, focusing on the intake stage.

Objective or other characteristic	Risk management strategy focused on fish quality regulation (implementation of EU's present strategy for dioxins and PCBs in food and feed)	Risk management strategy focused on information based exposure reduction (diet advisories)	Notes, alternative or combined/ converging strategies, conclusions
Human health protection	-- significant reduction of health benefits from fatty fish (if many people shift to unhealthier foods) + certain added reduction of risks from DLC intake (in heavy fish consumers) + control of feed-based health risks = **-/±**	++ health benefits from fish ensured + health protection likely ensured (assuming effective advisories) - some risks to high fish consuming vulnerable persons remain = **+/±**	*benefits probably exceed risks & are more certain (esp. if omitting diet choices) *distribution of R&Bs a factor *key policy criterion
Ecological impacts and risks	- may increase risk to vulnerable non-human fish consumers through population and community changes, despite immission reduction = -	+ decreases ecological risks if combined with immission reduction; otherwise no change = **+/±**	*important criterion *consistent but slight change in foreseen impacts
Economic impacts	- conflicts with economic interests of fisheries sector - some administrative/control costs - costs from loss of net health benefits + benefits from surely safe feed industry + some indirect economic benefits by uniform trade = **-/±**	+ preserving fisheries economy + low administrative/control costs + preservation of health benefits - losses from feed contamination - if food safety concern remains, losses may > those from regulation = **+/±**	*important criterion *many economic impacts cannot be quantified *losses & benefits distributed asymmetrically (BS, N Sea)
Efficiency	**±** efficient but in narrow sense only	**±** may be broadly efficient	*cf. other criteria
Flexibility	**-/±** regulations rather rigid instruments	**+/±** info steering inherently flexible (e.g. advisory adjustments)	rigidity entails some constancy & predictability
Social impacts	- unemployment in sea fish-based sectors (esp. medium term) (± social trust in regulation) = **-/±**	+ employment in fisheries (± social trust in personal risk management) = **+/±**	employment has broad also indirect significance esp. in coastal regions
Participatory character	- none directly with consumers; actors basically assumed to follow regulations + some participation from industry = **-/±**	+ inherently broad through advisories, information and related activities	*also education a factor *empirical evidence from Gt Lakes (& Scandinavia)
Precautionary principle	+ precautionary with regard to dioxins - not precautionary with regard to fish benefits and other impacts = **±**	+ precautionary for preservation of fish benefits and other impacts - less precautionary for dioxins = **±**	*counter-veiling risks to be addressed for broad, functionable precaution
Sustainability	- threatens sustainable herring fisheries + may increase sustainable use of fish with other measures = **±**	+ ensures sustainable (herring) fisheries esp. if processing to feed less valued or also ensured = **+**	*related to long-term efficiency and precaution
Reg impacts, subsidiarity	- centralizes dioxin risk management - cannot account for details well = **-**	+ regional and special concerns more easily addressed = **+**	*inherent differences in alternatives
Transparency	± may or may not be transparent	+ presupposed by advice	
Compatibility with other sectors	- not easily compatible with fisheries or pro-fish nutritional & health policies + compatibility through regulation = **-**	+ highly compatible with fisheries management e.g. under CFP + compatible with other areas = **+**	*important as integrated policies are sought
General political and principle-level impacts	- enhances uniformity and rigidity + assumes responsibility of consumers, including those not reached by advice + supports non-fish food interests = **±**	+ enhances pluralism, adaptivity (esp. of fisheries and consumers) - puts responsibility for risk management on consumer = **±**	*uniformity and rigidity desirable for federalism & normative governance
Cultural impacts	-- wipes out fishing and nutritional traditions ± may build trust in regulations but also erode it as drawbacks become known - may strengthen a culture of fear = **-**	+ preserves fish/fisheries cultures + promotes a novel risk awareness - may be vulnerable to health fears without strong risk communication = **+**	*fishing and seafood cultures wide-reaching, linked with health & environmental awareness
Technology impacts	-- reduces open-sea fishing capacity (dioxin-controlling and other fisheries not for food are possible on subsidies)	+ dioxin control fisheries possible + relatively easy use of some other post-sea management options = **±**	*in both, prevent etc tech *spin-off to (econ) effectiv *innovation a factor
Overall evaluation	-	+	depending on weighing, pending on uncertainties

Explanations: in all categories, a summarizing comparative evaluation of the alternatives (with respect to each other, on scale +; +/±; ±; -/±; -) has been produced as a composite of the specific pros and cons (that are given in ++/+/-/--) and has been shown in bold.

- **Technological and systemic impacts:** The implementation of the EU strategy would curb the Baltic herring fisheries and potentially other fisheries as well, along with the processing chains, infrastructures and capacities. The alternative strategy could better retain and also develop these capacities for management that would facilitate dioxin control, even though also in this case the herring fisheries are constrained by the potentially declining human consumption and demand.
- **Cultural impacts:** A thriving fishing and fish utilization culture with considerable importance in Baltic Sea countries, and a large part of the coastal regional cultures, would be lost if particularly herring fishing for human consumption would end. In the alternative

strategy, such a culture would be retained. In addition, a participatory and many-sided risk-benefit balancing consumer risk management culture could develop in connection with fish consumption advice and education.

- **Policy principles:** A key question is to what extent people can be let *choose themselves* what (fish) to eat, and to what extent are *interventions in food supply* needed (cf. 8.1.1). Some subjects like small children cannot choose; to protect them a regulatory approach may be preferred. However, also mother's milk is exempted from food regulations and recommended by authorities. The difference from fatty fish may be mainly that human milk is not marketed. As to precaution, sustainability, efficiency, consistency, equity and transparency, the EU strategy is not clearly superior. It serves the principle of free market, but this should not be confused with health arguments.

8.4 Issues in developing, modifying and complementing strategies

8.4.1 Needs for new and adapted strategies

An adaptive evolution of management seems natural. Novel or modified management strategies for Baltic Sea fish dioxins can thus be conceived. The above and other existing strategies may be utilized. Also the EU strategy and its implementation are under development. Its links and coordination with other strategies may become important, including diet advisory approaches facilitating risk-benefit and other multi-objective considerations.

New management strategies may be realized at both general policy level and at operational level. They may, regardless of the level and scope in terms of sectors, be broader or more specific, including partial or sub-strategies (such as national strategies within the realms of fish marketing only).

New management approaches are influenced by and called for e.g. by the following factors:

- A fuller consideration of **general policy** aspects; in particular, the balancing of centralization and harmonization with subsidiarity, including consideration of the fundamental approaches to management such as information-based steering in addition to or in partial replacement of norms
- A more many-sided and in-depth evaluation of **risks and impacts**, including those from additional DLCs and pre-dioxins, and the risks associated with the loss of health benefits from fatty fish
- A more extensive and robust consideration of the **early stages and root causes** of the risks from DLCs, including control of their formation and emissions
- A fuller consideration of the management **options** and their relative merits (cf. 8), as well as generally of the opportunities available and feasible regardless of the above uncertainties
- A more extensive and explicit consideration of the **interests and possibilities** of various sectors and their strategies and policies, including the fisheries sectors.

8.4.2 Factors to be considered in strategy development

General

Some of the factors that need to be considered when developing strategies for managing risks from dioxins and associated other risks and impacts may be defined on the basis of general analyses of policy instruments, for environmental and resource management and even more generally (Fig. 22, 23).

Many of the factors mentioned earlier in connection with evaluation of (present) strategies are also relevant in conceiving and developing 'new' strategies and in revising, modifying and adapting existing ones. Particular emphasis is put here on questions related to the interactions between various strategies in different sectors, on different scales of governance and on different scope and specificity. This essentially involves questions of strategy comprehensiveness, convergence and coordination.

Convergences and intermediates of strategies

Because both the EU strategy and present alternative strategies have limitations, deficiencies and uncertainties in the case of managing risks associated with fish DLCs in Baltic fish,

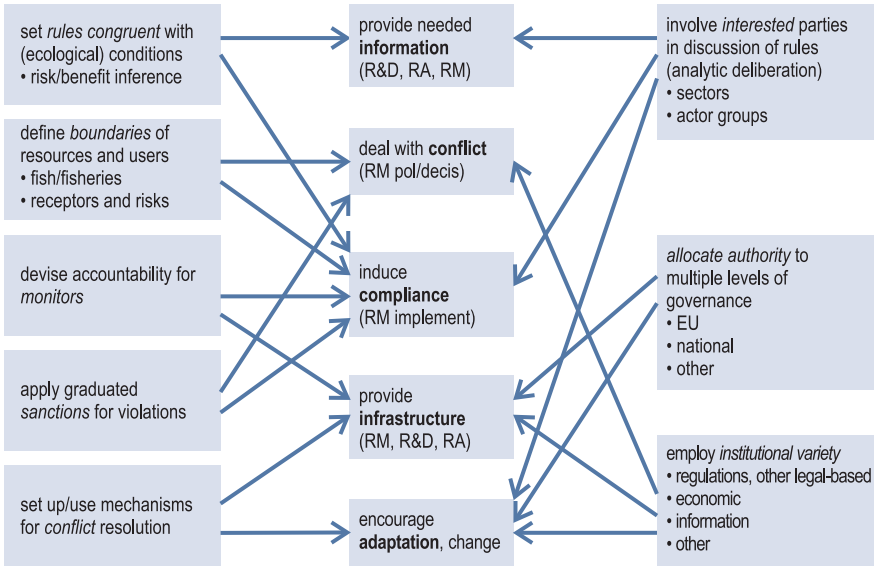


Fig. 23. General principles of robust governance of environmental resources and the governance requirements they help meet, as applied to Baltic Sea fish dioxins. Modified from Dietz et al. 2003.

the modification and intermediates of these and still other approaches could form a useful basis for further management. Despite their differences, e.g. the EU strategy and the present Finnish and Swedish approach are not wholly incompatible, and some convergence between them is possible. Both of these approaches are in a state of evolution, the EU strategy intensively due e.g. to the expressed goal of including other management areas (mainly within environmental risks beyond consumer health) and other DLCs.

It may be possible to solve some of these convergence issues using sector synergies. Solutions may also generally require tailored, more subsidiary and more information-based management approaches that take better into account the multiple and variable health impacts of fatty fish and alternative diets.

In developing the EU strategy and combining it with other strategies and approaches, it seems in any case highly important that in addition to the (conceivably also very dioxin-averse) environmental risk management area, also the broader health care and nutritional issues as well as fisheries issues are taken into account at a deeper and more concrete level than hitherto. Otherwise no real comprehensiveness and integration can be achieved, and balanced risk management or management in general (including also counter-veiling risks and other impacts) cannot be secured.

The *scope* of new or modified strategies presents a key issue, especially as coordination and integration with other strategies and sectors is seen as important (see below). With regard to the scope in more limited sense, within the present approach (focused on fish market regulation), an issue is how management of *wild* fish is related to that of *farmed* fish. These have some difference in terms of both risks as well as benefits, technology, ecological foundations, markets and the overall socio-economic and legal foundations of management, and so forth. For instance, fish farming and mariculture enables more control at least in a technological sense. On the other hand, the dioxin levels and the consumption amounts of both of these classes of fish need to be considered. This again calls for additional in-depth analyses of opportunities and their consequences.

Solicited expert opinions on dioxin risk management strategies

An e-mail questionnaire was sent to experts worldwide on risks and management of dioxins and like compounds (cf. 5.1.3, Assmuth and Hildén 2002). Results of this non-representative tentative survey (16 replies) are described in more detail in Annex 13. They pertain to dioxins in general, not specifically to dioxins in fish or even food (or in the Baltic). For the present work, the results may be summarized and commented on as follows:

Quantitative goals were considered necessary especially for emissions, by many also for food and feed and intakes, and by rather many also for environmental levels; only few respondents mentioned proposed goals to be set for body burdens including blood, and only one for risks. The priorities may reflect e.g. the present emphasis and the development stage of dioxin management but also some persisting

qualities due e.g. to practicability and inherent relevance for management. It was most commonly considered that such goals should cover all DLCs. The need for specified goals was recognized mainly with regard to different effects and target organisms. Opinions on the preferable basis of goals were divided: most respondents favoured dose-response models, and also epidemiological data and benchmark doses were preferred over NOAEL/LOAELs or BAT. According to most respondents, uncertainties should be accounted for by safety factors, but many of them mentioned statistical criteria. Interestingly, few respondents had an opinion on the appropriate goal in terms of risk levels; most often 50 % of the present level or 'as low as reasonably achievable' was preferred. The realistic timescale for reaching the goals was overwhelmingly regarded to be 20 a.

The most important **characteristics** of strategies based on ranking by respondents, in descending order of importance, were: Effectiveness, preventiveness, transparency, ease of implementation and stakeholder involvement.

In prioritizing strategic **approaches**, the following were considered most important (in decreasing order of importance): Emphasis on source control; focus on greatest risks; focus on greatest risk reduction opportunities (the two last mentioned being ranked equally high); multi-frontier action.

As to the **general means** of management, the opinions had considerable spread: while most respondents considered a necessary and sufficient effort to be c. twice that presently, several respondents preferred an effort 10-fold the present and also the present level; one also considered half the present effort sufficient. In this connection, most respondents regarded that the dominant relationship between the management of dioxin risks and other risks is synergistic, but several also regarded that the relationship is dominated by competition.

In evaluating different **categories** of management measures, prevention and source control were ranked clearly highest. Interestingly, occupational hygiene was ranked next in terms of overall dioxin risk management, above e.g. alterations in food and feed production, and limitations of intake in general (i.e. mainly from diet). Also supervision of actions and monitoring of their successes was ranked high. Remediation or cleanup was ranked low, even below maximization of benefits and compensation for risks e.g. from dioxin-laden food.

The greatest **obstacles** to appropriate dioxin management were considered to be (in decreasing order of importance): lack of knowledge, lack of technology, lacking enforcement of status and regulations, communication, and lack of funds. Some considered distorted perceptions of risks highly important (but none clarified what kinds of distortions are dominant). It may be noted also here that the obstacles are not separate, but lack of funds and technology may be conditioned by regulations.

It is thus evident e.g. that views of management alternatives and even the understanding of their meaning are highly variable. Experts differ widely in their evaluations of the significance and characteristics of the risks posed by DLCs (cf. 5.1.3), and even more on what should be done. Nevertheless, some consistency of opinions can be seen, e.g. regarding the overwhelming support for prevention and source control. Some of this is self-evident; the more precise contents of such evaluations however may display great variation (e.g., as to implementation of general approaches).

The roles of risk communication

The variable perceptions of risks (Slovic 1987, cf. 5.1.2) constitute a key issue in risk management. They are expressed in and influenced by communication about risks. Risk communication takes place in many forms and among many parties. In an open society based on democratic principles, communication about risks is fundamental for the formulation of acceptable policies and decisions concerning risks. In EU and notably in Nordic countries, increasing transparency and participation of policy and decision making in general and regarding risks e.g. to consumers have been emphasized.

There are limits to the abilities of those involved – also experts - to have an insight in and to communicate about risks (cf. 7.4.1). Dealing with complex and uncertain risks, the key challenge becomes to choose the message and the means to convey it appropriately, e.g. combining sufficient clarity with sufficient balance and avoiding both over- and understatement of risks and uncertainties. Also the lack of unequivocal scientific truths of risks and the dependency on value judgments need to be communicated. This reflects the degrees and types of precaution adopted.

8.4.3 Strategy integration and coordination areas and issues

General

In developing new strategies for Baltic Sea fish dioxin management, an important task will be improved integration of the human health and food oriented present EU strategy with environmental strategies, policies, programs and other instruments. As detailed above (6, 7), these include

- Instruments specifically addressing dioxins (e.g. for various emission sources)
- Those addressing emissions in general (e.g. in connection with BAT and EMAS)
- Those addressing POPs more generally (e.g. under LRTAP/EMEP and Stockholm Convention)
- Those addressing chemicals more generally, e.g. under the REACH system
- Those addressing fisheries management
- Those addressing marine protection (especially the developing Marine Policy)
- Those addressing feeding-stuffs and food production in general
- Those addressing environmental health and public health in general.

Integration with the first mentioned areas seems a priority because the EC already has stressed the needs and possibilities to complement the present dioxin strategy with strategies and actions within the environmental sector (EC 2001 and Van Birgelen, personal communication 2002).

Relationships with some other strategies

General strategies for POPs and other chemicals

Under the **POPs Convention** (cf. 6), the development of National Implementation Plans (NIPs) and National Action Plans is underway, including some guidance for strategy development. These plans may become a key instrument for risk management of DLCs also in the case of the Baltic. They may include technical and some economic, regulatory and information related elements.

In connection with regional POPs assessments, some options analysis and management strategy development is done. However, there still seems to be a lack of policy and decision analytical elements, such as socio-economic analyses (Albaiges, personal communication 2002). The regional structure under the POPs Convention involving the Baltic Sea extends the Black Sea, and thus may not efficiently address the specific issues in the Baltic.

Under **HELCOM** (as under OSPARCOM), in addition to general strategies on hazardous substances, some activities on POPs management and also specifically work on DLCs have been created (cf. 6). The non-binding character of the Helsinki Convention involves some limitations. The information steering functions and coordination with EU marine policies have been emphasized. HELCOM in any case provides a cooperative mechanism that is in some areas rather well established.

The **replacement of PCBs** by less hazardous substances is a generally accepted and ongoing process that also addresses alternatives to substances that have been intentionally produced. This is a very different management process from those addressing dioxins; in this case, the preventive considerations must include various precursors such as chlorophenols (already much reduced) as well as chlorobenzenes (reduced to a more limited degree and by less regulatory steering).

Existing chemical strategies and related control systems such as those associated with the Existing Substances, Plant Protection Products, and Biocides regulations of EU are important for dioxin risk management also in the Baltic especially as precursors are dealt with. These strategies are undergoing evolution and consolidation especially within the REACH system.

A **new European chemicals policy** has been under preparation. Legislation on an ambitious control system (Registration, Evaluation and Authorization of CHemicals, **REACH**), giving increased responsibility to industry e.g. in providing requisite information, has been proposed by the EC and accepted in modified form by the European Parliament. Although REACH addresses chemical products and not unintentionally produced substances such as dioxins, it does have important links with DLCs as well, e.g. through the overall procedures for assessment and control. For instance, the associated development of risk assessment, including extended treatment of marine environments, PBT substances, secondary

contamination through biomagnification, and waste stages, will be of importance also for management of DLCs. However, REACH is no panacea for PXDD/F prevention.

Integration is a challenge also within REACH, e.g. across chemical product classes (such as plant protection products and biocides containing or forming DLCs), between human and ecotoxicological concerns (also as these are often administratively separated), and between assessment and management. To this is added the challenge of integrating the complex legislation and practice with other areas of dioxin risk management, e.g. in food and health (as attempted in the SCALE initiative) and further with fisheries and marine protection.

Within EU chemicals risk management, the development of risk reduction strategies is done especially in connection with alternative products, including assessments of their advantages and disadvantages. Such considerations are important for many dioxin precursors. Thus, specific strategies may be developed e.g. for the phase-out and management of PeCP, for chlorobenzenes as well as products such as PVC (cf. 7.2.1). The latter has been subject to great efforts especially in the US where some state programs on dioxin risk reduction have specifically targeted PVC uses and incineration (e.g., in hospitals), despite criticism from chemical industry (see e.g. Belliveau 2003).

BAT and product steering strategies

In addition to general strategies for preventing and controlling dioxin formation and releases in chemicals control, improved focused strategies are needed for key branches. These include metals industry (e.g., Buekens et al. 2002a,b, François et al. 2000, 2002), textile industry (Fiedler 2003), chemical industry including biocides, and waste management also beyond incineration. These areas of management should be better coordinated, e.g. under the IPPC and particularly in developing **POPs-related** BAT and BEP systems to account for PXDD/F formation and treatment. Development of dioxin control strategies in waste management may be linked to EU waste policies and to instruments on hazardous wastes, such as Seveso II Directive and Basel Convention.

Specific **source-oriented** strategies are interesting to complement and potentially even to replace some of the fish-focused management strategies (cf. 9). Source-oriented strategies may include both emission control e.g. within an industrial branch and formation of dioxins in the first place. Such source-oriented strategies may be extended to more general preventive strategies, e.g. addressing broad classes of products containing potentially dioxin forming substances, as well as chlorine and bromine economies as a whole, e.g. based on PLCA approaches.

Food and feeding-stuff strategies

The integration of dioxin risk management with the food and feeding-stuff areas is in some respects less problematic than the inclusion e.g. of fisheries perspectives. There already exist integrated instruments addressing both materials, notably those included in the EU recommendations for dioxins and PCBs. Also the overall food safety policies and instruments have developed rapidly both in EU and other international organizations as well as nationally.

Development of risk management within the feeding-stuff area will require more emphasis on extensive food production systems. This may facilitate a more preventive approach to management. Also other areas of animal nutrition, such as veterinary (e.g. in reducing toxicity to production animals themselves) in addition to human health aspects, will be important in this connection.

Consideration of waste flows, recycling and materials management is needed to reduce the influx of DLCs in feeding-stuff and subsequently food production systems (SCAN 2000, Joas et al. 2002). Joas et al. (2003) proposed the following measures in this area:

- Establishment of a positive list of wastes that are used in feeding-stuffs
- Implementation of declaration duties for these wastes
- Development and implementation of unequivocal identification procedures for these wastes
- Separate handling of materials destined to feeding-stuffs and material containing POPs
- License and quality assurance for suppliers to feeding-stuff industry
- Coordinated inspection programmes.

These and other institutional and organizational measures essentially complement the technical

options in risk management (cf. 7) and build up their basis. Such measures need also be further developed within fish-based feeding-stuff and food industry.

The development of alternatives is underway to reduce contamination of the food production chain by DLCs. The instruments used in association with the Codex Alimentarus are e.g. being extended to cover animal feeding and dioxins issues (Sijtsma and Doring 2002).

From a Baltic Sea point of view, alternative mariculture systems are important (cf. 7.3). Vegetable oil based fish feeding systems have been in particular suggested and tried. However, the health, ecological, socio-economic and systemic and other impacts of also these alternative systems are not well known.

Part of the management problem in animal production is the lacking knowledge of and control over indirect exposures, risks and impacts through DLCs in fodder. However, this may not be comparable to the problem of feeding-stuffs contaminated e.g. by wastes, as in the dioxin and PCBs scandals. Also the levels of PCDD/Fs and PCBs in the food produced have declined, partly as a result of decreased levels in fish and other feeding-stuffs. The net benefits from fish-based feeding-stuffs, both to the health of the food production animals and to humans consuming them, have correspondingly increased. Multi-dimensional comparative risk-benefit analyses are to be extended from direct consumption of Baltic fish to cover the indirect risks and impacts through food production and consumption systems (cf. 5.4.4.).

Fisheries strategies

Perhaps most of all, new management strategies for Baltic fish dioxins need to be better integrated with management of the fish and fisheries themselves. This seems self-evident, as management of fish dioxins is about managing fisheries. Presently however there still seems to be an unfortunate division of these areas of management and advice. This is reflected e.g. in the limited consideration of dioxins when producing fisheries advice in the IBSFC expert bodies. Dioxins are presently not explicitly, broadly and systematically taken into account in the key assessments providing advice to Baltic Sea fisheries management by ICES WGBFAS. Dioxins have been considered cursorily among additional factors in the fisheries assessment for salmon (ICES 2004a, 2005c), but not in those for herring and sprat (ICES 2003c, 2004b, 2005b). Even the discussions of dioxins in salmon have not been linked with other areas and procedures of fisheries assessment, on the level of e.g. stock estimation or TAC specifications that are increasingly decided on within the CFP if EU. Also in general evaluations of fisheries management strategies, no mention is made of contamination, under ecosystem objectives or elsewhere (e.g., ICES 2005a), or resultant impairment of product quality, although they clearly constitute key constraints for Baltic (as well as some other) fisheries. In general, the Baltic fisheries management system, based on approval by all participatory states, is a rather formal mechanism where the inclusion of additional objectives and instruments and related assessment elements can be difficult.

On the other hand, fishing has not been considered in detail in some of the assessments of risks associated with fish consumption, either. Lack of integration between these management sectors and areas of expertise is particularly notable in the limited consideration of fisheries issues and management opportunities in the EU strategy for food, and in the frequent absence of fisheries scientists from meetings where Baltic fish dioxins and their management are discussed (mainly within food and human health areas). SPCFC (2005) assessments have started to fill this gap.

Such divisions, detachments and gaps parallel those between fisheries management and marine protection. However, in this area, work is underway on new joint procedures such as Ecosystems Approach Based Fisheries Management (e.g., HELCOM & IBSFC 2002). The Baltic presents in some respects a good ground for integration of fisheries and environmental policies, due e.g. to the traditionally relatively small differences between the Nordic countries in terms of regulatory culture and to the relatively low fences between sectors and levels of government (Degnbol et al. 2002).

Due to narrow foci and to the mismatch of sector policies and strategies, there is a risk that important fisheries (or dioxins) issues and opportunities are ignored, and strategies and decisions are formulated that do not work in real (holistic) life but instead amount even to additional problems. Better integration in this regard will require both the development at the EU level of linkages between health and

food related strategies with the CFP as well as the development, for the specific needs of the Baltic, of fisheries and food and health sector communication and cooperation. This should involve on one hand the key official bodies for the region, i.e. ICES/IBSFC, and on the other hand the various regional organizations in the food and health area (e.g. under Nordic Council of Ministers). The coordination between inter-sector regional processes and corresponding inter-sector EU processes will be important.

Marine strategies

The EC has prepared a strategy to protect and conserve the marine environment. While the strategy is still under development, the EC communication (COM(2002)539 final, cf. Annex 11), proposes a series of overall and sector objectives for the protection and sustainable use of the marine environment, in addition to actions of various kinds to attain them, along the lines advocated in the sixth environment action program. The fundamental objectives identified are to halt the decline that is in progress and to protect the marine environment in future, by incorporating these objectives into the existing policies and frameworks.

Marine policies and strategies emphasize an **ecosystem approach** including a comprehensive consideration of human-ecosystem interactions, also socio-economic drivers and impacts. Such an approach in fisheries management is under development also by HELCOM and ICES around common issues between environmental protection and fisheries. In this connection also dioxins have been noted, but only as a monitoring and research need.

At least the following processes seem to have sufficiently well established and important links with dioxins and their management to be mentioned in short:

- **Eutrophication:** As discussed above, the impacts of eutrophication on DLCs are complex and cannot be reduced to simple biomass dilution. Thus, abatement of eutrophication and DLCs can not be considered to generally compete but instead have also synergies. Eutrophication in some respects presents greater problems than DLCs that have already considerably been reduced; by contrast, no improvement in eutrophication may be achieved for decades due to long retention times (Wulff and Niemi 1992).
- Protection of the Baltic against **oil pollution**: Combatting and preventing oil pollution of the Baltic has some linkages with dioxin control, technical and institutional. The funding mechanisms for oil spill abatement may be mentioned. Mineral oil also acts as a carrier of DLCs and an additional stress factor that may accentuate the need for DLC risk management. Potentially, oil and gas can cause of dioxin mobilization from sediments (e.g. through gas pipelines),
- Utilization of **inorganic marine resources** involves some linkages with dioxin risk management. The ability of international mechanisms to control these developments even under strengthened marine environmental protection and in an enlarged EU are uncertain because e.g. of the interests of Russian Federation.

8.4.4 Cost considerations

General considerations and methodological remarks

Risk-benefit or cost-benefit information may in principle help resolve what the relative advantages and disadvantages might be of alternative courses of actions, considering the risks from reduced benefits associated with accepting a risk, and other counter-veiling risks of risk management. However, the limits of such information should be scrutinized (Ball 2002), and it should preferably be complemented by other considerations.

For all strategic alternatives and management options, a difficult question is how to delimit the cost estimates, for several reasons including the following:

- Some costs may be incurred for *other reasons than dioxin risks only*. This applies to both information related measures such as dietary advice and to technological measures e.g. in source control, and to general and specific options
- There are *indirect as well as direct costs*; the former may include costs incurred from counter-veiling risks of management actions, e.g. by loss of benefits, and from subsequently managing these, as well as indirect costs from impacts e.g. on fisheries

- The cost structure of a management option often undergoes *changes* e.g. in the course of innovations, expanded use, and regulatory requirements. Costs may also otherwise be distributed differently in time. Improvements may be achieved after lags by actions and investments already made (Entec Uk Ltd 2003). Therefore, present cost data for an option may be misleading and preclude cost (and benefit) comparisons, and the distinction of the costs (and other impacts) of short-term and long-term measures is not clear-cut (cf. the greatly differing estimates of cost per life-year saved by emergency and long-term measures in dioxin control in waste incineration by Kishimoto et al. 2003).
- The *costs to operators and other* primary economic agents (also when summed) may differ from those to society at large, and the shares and relationships of these may be difficult to elucidate
- There may be different costs (in absolute and relative terms) and cost structures in *different spheres* of management, e.g. in different countries and sectors.

The unavailability of cost information for risk reduction measures and for risks themselves affects both cost-efficiency estimation and particularly cost-benefit analyses. The latter fundamentally assumes that all important risks, impacts and benefits can be valued in economic terms. Cost-benefit, as well as risk-benefit, analysis may also be biased with respect to investing and spending in the present to prevent future harm, depending on discounting of cost and benefit over time.

Because of the difficulty in quantitative assessment, it does not seem to make sense to come up with estimates of risk-benefit ratios. They should in general not be considered as decisive arguments. Instead, the management options and their implications need to be evaluated in a more comprehensive, multi-dimensional manner including qualitative aspects of risks or disadvantages and benefits or advantages.

For such reasons, it is appropriate in the present connection to discuss cost considerations mainly at a policy principle level and in terms of knowledge production. Further work in this area is warranted for development of management strategies.

Considerations for Baltic fish dioxins and dioxin management in general

There is very incomplete and uncertain information on costs of dioxin risk management. The data and estimates that do exist are mainly for well-known technological processes of emissions control in waste incineration, industry and energy production (e.g., Detzel et al. 1998). Even among these, cost data for some source categories are much better than for most others (e.g., municipal solid waste incineration in comparison to industry, and sinter plants in comparison to other industrial processes), and usually relate predominantly to air pollution. There is also increasing information on costs of large-scale cleanup of dioxin-contaminated sites, including sediment sites (cf. 7, Annex 10).

The costs of dioxin abatement are related to other management areas (see above). If risk reduction goals or other goals, such as intake, immission or emission goals (which may be seen as successive proxies for risk goals), are set very strictly and ambitiously, it is exceedingly difficult and costly to reach them. There may be steep thresholds in the availability of options, for technological and economic as well as social and political reasons. Moreover, at some point, overly ambitious goals become counterproductive and may even cause harmful repercussions in the system. This is related to the question of balancing realism with proactiveness, and to the inclusiveness of costs to be considered, i.e. the framing issue. Action limits for dioxins in food and feeding-stuffs that would effectively prevent herring consumption and eliminate its health benefits are an example of this.

On the other hand, cost is often seen as a factor that limits abatement more severely and more permanently than actually might be the case. For instance, by setting ambitious management goals, innovation may be prompted that will lower unit costs in the future. Likewise, although management goals such as emission limits may seem overly ambitious or even unattainable in some region and system that lacks the necessary resources, infrastructure and other prerequisites, such as with upgrading of Polish waste incinerators to meet EU dioxin emission standards (Grochowalski 1998), the implementation of such goals may be justified by the need for uniformity and by the gradual development of such prerequisites; it may also be aided by support systems such as those in place

within EU. These may override the previous lack of economic transfer and support mechanisms for Baltic Sea protection (Markowska and Żylic 1999). A key issue herewith is what costs, to whom and how (e.g. under EU programmes) are accounted for.

Within environmental cleanup technology as well as in primary emission control, dioxin destruction or treatment cost data have in many cases been presented by the developer or vendor of the technology, and are not necessarily reliable and inclusive. In addition, there are detailed cost data for some other sources such as traffic but not assigned to dioxin control specifically but to technological control in general, e.g. for other air pollution related reasons. For some other specific options, cost estimates may additionally be relatively straightforward to produce. However, in most cases they are bound to be 'guesstimates' only.

Cost-benefit and risk-benefit considerations are particularly important in the context of dioxins in Baltic fish, as the need to balance risks and benefits from dioxin-laden fish has become an argument in scientific and policy discussions, e.g. in justifications for Finnish and Swedish derogations and national views of risk management (cf. 5.4.4, 6). Little analyses have been published of such costs; even comparative assessments of the health outcomes themselves from reduced fish consumption have been tentative and partial (see e.g. Tuomisto et al. 2004b). Also the cost-effectiveness analysis of dioxin abatement in waste incineration by Kishimoto et al. (2000) only estimated health costs based on cancer mortality.

The risks from DLCs in Baltic fish (and elsewhere) are not one-dimensional, involving e.g. only human health, and the quantified (and even monetized) impacts are on many levels. Therefore, no simple description, commensuralization and optimization of cost and benefit functions is possible. Single cost-benefit or risk-benefit and other such calculations can provide pieces of the decision basis. Collectively, they can also be part of a multi-dimensional and multi-objective exploration (on the basis of multi-actor communication) of the various dimensions and frontiers in management option and impact assessment, and finally in the selection of an effective or at least commonly acceptable and fruitful frontier.

In a situation like Baltic fish dioxins management with its extent, complexities and uncertainties, cost-effectiveness analysis, also by novel methods, may provide an intermediate approach to gauging the economic dimension of strategies, and a viable alternative to risk-benefit approach (see e.g. Kishimoto et al. 2000, 2003 for dioxin control in waste incineration and Neumann 2004 for health care).

8.4.5 Liability issues and economic instruments

A challenge is the unclear definition of risk and impact chains and thus of **liabilities**. Most dioxins have been formed inadvertently, and keep cycling in the environment in a highly complex and diffuse system. In such a situation, the polluter pays principle does not hold as readily as e.g. in the case of one-time chemical risks from intentional products, and of conscious discharges. Therefore, the liability may to a major part be reasonably assumed by the society as whole, if a need is perceived to allocate liabilities for risks and management burdens. Nevertheless, the application of institutional measures in addition to and as a support of technical measures in the prevention of additional dioxins and dioxin risks seems an important area of development.

Factors to consider in defining and allocating liabilities include:

- Are the risks and impacts *intentional*, and do liabilities apply to dioxins and to releases that have not been permitted or explicitly forbidden through regulation
- Are *presently ongoing* emissions from facilities more liable in a legal sense than past emissions that may have already diffused to the environment
- *Uncertainties* of the relative share of various source categories, and requirements for proof, that in the area of environmental legislation may not be as strict as in some other areas of jurisprudence, allowing a lower degree of probability for causes
- Liabilities for compensation of economic *losses to fishers and fisheries* may appropriately be borne by the society, in this case EU (especially if imposing market restrictions), instead of national instruments such as the laws (e.g. in Finland in early 1980's) on compensation to fishermen based on MeHg consumption declines, or instead of the fishers.

Institutional options potentially include also **taxes** targeted on halogen industry, either collectively and in a non-specified form or in a more focused manner. These are examples of economic (but regulation based) management measures that may in principle be efficient, also introducing a preventive element in steering, but may have considerable political and also practical obstacles. The above liability issue should be taken into consideration. It may however be argued that if other grounds are deemed to exist, no fundamental hindrance is necessarily caused by the present regulations, policies and practices; if such a management approach be jointly regarded as efficient and reasonable, the regulations and policies and practices perhaps should be changed to allow such new approaches.

The following mechanisms may be relevant as precedents or examples also for EU legislation and regulations regarding compensation and economic instruments for dioxin-related risks:

- In **US** an extensive legislation involving comprehensive and joint liability (CERCLA, SARA) also for past conduct and emissions and based partly on taxes collected from industry has been in place already for 20 years and reauthorized many times, to reduce risks associated with chemically contaminated sites, also including DLCs
- In Europe, the **Seveso directive**, created on the basis of and with explicit reference to accidental dioxin releases, lays down some principles and procedures regarding liability and compensation for environmental damage. Its applicability to past releases (including diffuse emissions) is more limited than that based on the above US legislation. The same applies to other European (also national) legislation on environmental damage.
- The developing regulations concerning **product liability**, although mainly addressing ongoing production, may have some relevance indirectly for liability and compensation procedures; they are an example of a general and broadly accepted strive toward liability that is extended to cover the whole life-cycle of products
- Some provisions in the **POPs area** based e.g. on the principles in the Stockholm Convention regarding liability may apply, e.g. for liabilities and compensation (also support for capacity building) within stockpiles including dlPCBs
- Within **health care**, some principles and procedures for establishing sufficient causation in order to enact compensatory and other regulatory action have been established and tested; however, e.g. the precedent of tobacco and disease suggests that these may not apply also to the reduction of environmental health risks by dioxins e.g. in Baltic Sea fish.

8.4.6 Local hotspot management and geographically broad options

In addition to strategies targeting the whole Baltic (and its bordering countries, catchment or still broader impacting regions), Baltic Sea dioxins and related compounds can be and are being managed at a more restricted geographical scale. This applies to both localized emissions sources such as industrial areas and to areas of the external environment already contaminated by DLCs.

There is no sharp borderline between hotspots management and more extensive management. This is related to the definition of a contaminated site: in many cases, they include several separate hotpots (also with different contamination profiles), and in some cases they constitute extensive areas in the sea or its catchments, such as in estuaries like that of River Kymijoki. Despite the gradients in contamination, most of the DLC load in such cases may be located in adjacent recipient, and thus management strategies need to consider this larger contamination as well. Moreover, also in more restricted hotspots the background contamination has to be taken into account in the assessment of site-specific risks and in the development and application of site-specific management measures.

The NRC (2001) strategy for management of **PCBs in sediments** considers the broader socio-economic aspects of risks and impacts, and ways to account for them in development and implementation of management. The value and use of fish and fisheries, their cultural importance (also from the point of view of the traditional cultures and economies of native populations), and the issues associated with losses (economic and health-related) from reduced fishing and fish consumption have been emphasized in this strategy, along with the more regular primary human health and ecotoxicological concerns and evaluation of technical management options.

Importantly, it was concluded that no generic solutions exist, as the solutions are largely site-specific, and that careful analysis and planning is the key to efficient, acceptable and long-term management solutions. The principle-level problems of the long-term security of isolation and other such *in situ* solutions were stressed herein, although it was also noted that such approaches in most cases are an important part of the toolbox of technical measures (in some part of the contaminated area).

Attention to the risks and impacts, including indirect, of the alternative options (cf. NRC 2001) is particularly important to prevent the formation of additional risks e.g. from the dispersal of contaminants during dredging and treatment. Such analysis needs to be extended to the control of subsequent emissions, exposures and risks at disposal sites for the materials removed and treated.

On the basis of the above treatment cost data and the information provided by Verta et al. (2003, 2004, accepted), it may be roughly calculated that cleanup of the greatest known dioxin hotspot directly loading the Baltic, River Kymijoki, in the most contaminated stretch comprising c. 100000 m^3 sediment (50 000 t, at 50 % water content) and 2 kg TEq (20 % of the TEq load in the river) at a unit cost of 500 $ (or €) t^{-1} (allowing basic-level dredging/safeguarding, demonstrably efficient thermal-chemical destruction, and basic-level treatment of residues), would cause total costs of the order of 25 M€. With higher level and extent of cleanup (e.g., precision dredging or silt screen, incineration, and off-site secure landfill disposal) the costs could easily be 100 M€. On the other hand, if treatable e.g. by *ex-situ* vitrification, unit costs could be 10-fold less and the total costs (including removal, transport and disposal/reuse) perhaps 5 M€. The potentially removable amount of dioxins is significant, at 4 kg WHO-TEq_{DF} representing c. 20-fold the yearly atmospheric load on the Baltic and perhaps 10 % of the pool in the sea (Verta et al. 2003, unpublished 2005). Extending cleanup to the 2 Mt of contaminated sediment in the whole river might increase the costs to 0,2-4 G€.

A cleanup spending of 100 M€ is not uncommon in the U.S. (FRTR 2004), e.g. on the SARA financing and liability-sharing mechanism, and may become possible also in EU. The unit costs could also come down due to technology development and greater cleanup markets. It may however be questioned whether such spending is worth the outcome, considering the magnitude, uncertainty and trends of the risks from these sediments, the risk of increased disadvantages due to remediation e.g. from sediment dispersal (see Malve et al. 2003), and alternative management options, both in the particular (local) case and more generally, e.g. within prevention of further formation of PCDD/Fs (and other DLCs). However, if such spending would e.g. be a decisive factor in allowing the continuation of herring fishing (depending on the regulatory process), it might be justified.

The issue of **land or sea focused** hotspots management has additionally to be resolved in setting priorities. The coastal and marine sediment hotspots readily emit DLCs to the sea and thus constitute an immediate risk. However, their control by intensive technological measures involving containment, concentration and destruction or other treatment is a formidable challenge in terms of technology and costs, and may also cause new problems. The hotspots in the catchment are in many cases more easily treatable and manageable. Most of the DLCs in the sea have already dispersed to remote sediments and are out of reach for technological control (cf. 7). Thus, a general strategy would be to favor land-based hotspot measures over extensive sediment cleanups, if these have no particular justification e.g. through simultaneous other benefits (such as in maintenance dredging) or if particular technological opportunities arise for most feasible and safe cleanup operations (cf. Annex 10).

It needs in general to be questioned how efficient a risk management strategy involving extensive remediation of dioxin- and PCB-contaminated sites would be, also in the Baltic Sea catchment, with regard to the impacts of such a strategy. Focusing on human health, Ross (2004) concluded that for PCBs there is merit to weighing all options and public health implications before deciding on a path of action, and that a weight-of-evidence evaluation suggests that removal of PCBs from the environment, no matter what the costs, makes little sense. He specifically pointed to a cost estimate of 400 billion USD based on USEPA estimates of the amount of PCB wastes to be remediated (500 Mt) and of unit costs for disposal (250 USD t^{-1}) and incineration (500 USD t^{-1}). This is one of many criticisms of environmental cleanup programmes and their poor benefit-cost ratios. However, evaluations of the economic impacts of cleanups ought to

account for also other impacts and justifications for such activities (see e.g. Wernstedt et al. 1999, Bonano et al. 2000), warranting use of multi-criteria decision methods (Efroymson et al. 2004). For instance, the land-use compatibility and public involvement discussed by Wernstedt et al. (1999) may be extended to cleanup of contaminated sediments by consideration of the normative basis and procedures for planning and managing of watercourses, but involves additional complexities in the case of Baltic Sea dioxins, legal and administrative as well as technical.

Part C: Synthesis and conclusions

"...und wenn wir nach diesen sorgfältigst gesiebten und geordneten Schlussfolgerungen uns für die Gegenwart und Zukunft ein wenig einzurichten suchen – so ist das alles unsicher und vielleicht nur ein Spiel des Verstandes, denn vielleicht bestehen diese Gesetze, die wir hier zu erraten suchen, überhaupt nicht."

– Jur. Dr. Franz Kafka: Zur Frage der Gesetze (posthum.)

"First, do no harm"

- Hippocrates, c. 460-380 BC (attributed)

Synthesizing discussion and recommendations

9

9.1 Knowledge of risks and of their management, emphasizing framing, causality and science-policy links

General considerations of knowledge about risks

There are multiple limitations of truth claims about the risks associated with dioxin-like compounds in the Baltic Sea. Some of these limitations are caused by physical factors, such as the inaccessibility of some object of study, some by psychological or social factors, and yet others by our imperfect understanding of the phenomena at play. These categories of limitations are not wholly separate.

Limitations of truth claims about risks depend not only on measurable facts, but also on views and values. Some of the limitations are thus of operational and provisional character, others more fundamental and permanent. Limitations apply to both predictions and to reconstructions or explanations of risks. Importantly, truth claims are made, and refuted, also concerning risk management, e.g. the means available to reduce risks and the impacts of these means.

The limitations and weaknesses of truth claims extend to claims about uncertainties. This is crucial with dioxin risks. Uncertainties are at times downplayed due e.g. to the fact that all sources of uncertainty have not been yet identified, but are also exaggerated as the knowledge already available is not fully utilized. Uncertainties may be used as an argument for excessive precaution, but also for endless research.

Empirical truths about risks are limited internally by the reliability of data, and externally by their generalizability. For instance, statements about dioxins in the Baltic or their effects are vulnerable to criticism as there is always a sizeable possibility that the data hint at a generality that is due to chance. In principle, such generalities can be elucidated by confirmation of the observations and experiments, and by getting a hold of the variations and irregularities e.g. through improved specification. However, in most cases one cannot readily know the general significance of an empirical measurement.

A classical positivist scientific ideal may not be sufficient in dealing with dioxin risks. No amount of additional measurement of risk-related entities will wholly (in some respects, even centrally) resolve confusions and disagreements about these risks. And yet, many act upon such an assumption, invoking more research or monitoring, or improved technical measures, as self-evident remedies to questions about risks and ways to deal with them. This reflects a narrow and naïve picture of the ability of science and technology. Others in turn reject scientific and evidence-based decision-making (see critique by Durodié 2003a, c, and below).

There are still considerable needs for measurements and experiments. Some of them are needed for pragmatic reasons e.g. to check compliance with norms. Importantly, data serve to strengthen or weaken models of risks. As knowledge is imperfect, one cannot easily foresee the value of data. Thus, one should be prudent also about normative statements of and prescriptions for data production. Nevertheless, more attention should be paid to the kind, extent and precision of data needed and used.

Models of risks or management are limited by their boundary conditions, structures, assumptions and even general logic. They can help simplify the complexity in risks and treat them in a structured and efficient manner, even allowing some predictions. However, the verifiability of models can be shown to be questionable in principle. Their primary value is heuristic, in revealing key uncertainties and how to reduce them (Oreskes et al. 1994). In this iterative process, there is uncertainty also beyond that revealed by models of factors already identified (see Kohn et al. 1994).

Both data and models often mislead their users to believe in them more than is warranted. They may suggest the sufficiency or superiority of quantification and of formal descriptions of realities that are only partly tractable by such approaches. This can be strengthened by, and in turn strengthen, a belief that risks and uncertainties can be reduced in a highly

controlled and organized way. Preoccupation with quantification may also interact with a belief that risk assessment and management is mainly a technical matter, e.g. that some 'magical number' (or probability distribution) of exposure or risk (or risk-benefit ratio) exists that is true, permanent, general and decisive.

The crucial question related to the 'risks from dioxins in the Baltic fish' is what this "from" entails: What are the risks due to dioxins (and which dioxins), what to other causes (cf. 4.1.2). There is a gap between experimental studies where the effects and associated risks can be relatively clearly attributed to the DLC tested, and field studies where a multitude of confounders are present and the study can not be controlled and replicated. Even many of the effects thought to be specific to dioxins are more general responses to stress (e.g., Matsumura 2003, cf. Neubert 1997-98).

The limitations of truth claims about dioxin risks are aggravated by their complexity and ambiguity. These risks emphasize the 'accidental character' of multiple 'relevant circumstances' that prevent the establishment of 'broad and simple uniformities', to paraphrase the motto (cf. 2) by Bertrand Russell (1948). Specificity and realism need to be balanced by generalization and simplification. Bjørnstad and Grenfell (2001) noted that to understand any system, we need to appreciate its idiosyncrasies; but to encompass broad patters, we need to extract generalities. This amounts to the challenge to simultaneously accommodate and transcend the details (of natural history in the case of these authors, but also of human social history).

At a still more basic level, the general views of dioxin risks and their management, conditioned not only by 'scientific facts' and models but also by social and mental factors, cause limitations for our ability to grasp and to deal with these risks. This is related to the artificiality of the division between facts and values, shown by Putnam (2003). Thus, knowledge about dioxin risks is only partly objective and scientific. This depends on the kind and the level of knowledge. To quote Niels Bohr: "The opposite of a trivial truth is false; the opposite of a great truth is also true." This is especially true of extensive, complex and variable and ambigous systems and phenomena.

In addition to knowledge, also political, economic, psychological, social and cultural factors influence the management of risks from Baltic dioxins. They add to the complexity, ambiguity and unpredictability of these risks, e.g. as they become boundary objects or 'new risks' in the sense of Beck (e.g. 1995) that transgress traditional definitions (and controls) of risks. They also add to the relativity of dioxin risks. Some of this dimension is captured by the old saying in medicine: It is often more difficult to come up with a therapy than a diagnosis, but still more difficult to get the therapy accepted.

In summary, what the dioxin risks are and what is best to do about them can be known only imperfectly and in relative terms. Increased modesty in truth claims as well as in approaches to formulating these is in place. A fuller acknowledgement of this ignorance is important; it also needs to be analyzed in order to resolve its implications for further inquiry and for action (cf. Hildén 1997b). Simultaneously, a new openness and ambition is in place: one needs to probe more varied paths to understanding the risks, even if only to find the incomprehensibility of truths and outcomes, i.e. balancing optimism and pessimism about knowledge and its meaning.

Interactions between research, monitoring, assessment and policy

For managing risks from Baltic fish dioxins, information is needed from many disciplines, also about the management system itself. This information is obtained from research, monitoring, testing, technology development and other areas. Assessment and evaluation, variably science-based, lie in between these areas of knowledge production and management. Management in turn influences these areas of knowledge production, including science.

Knowledge evaluation is implicit in many studies of dioxins, e.g. in the discussions on further research needs. However, evaluations and suggestions are often of a qualitative, narrative and *ad hoc* type and do not often question more fundamentally the primacy for research in this area.

A better grasp of the systems to be managed, natural and social, is imperative in developing new management strategies, e.g. for dioxins in Baltic fish. Studies in dioxin sources, fate and effects and in control technologies produce important information for management. In order to better link science and policy, research in these links, i.e. human and social systems, is particularly important.

Because of a separation of science and management, it may be suspected that there is an inability to focus in a more analytical fashion on the opportunities to manage risks. Ever more numerous and detailed studies of the natural scientific aspects of the problems and risks are being made, while management policies, decisions and actions are few and too detached from scientific deliberation.

The limits of knowledge about dioxin risks (cf. 9.1) imply that in addition to attempting to quantify and predict them, interpretation of them emerges as a viable objective. This is related to the need to complement natural scientific and technical approaches to risks and their control with analyses of social aspects. The importance of continuous interpretation in the latter area is coupled with lack of firm and generally applicable criteria for trueness. However, this relativism has strengths, if the aim is not so much to define absolute truths about risks but to facilitate discussion about their meaning and about how to deal with them and why.

The scientific basis of EU dioxin and related strategies

The food oriented part of the EU strategy on dioxins and PCBs, or the strategy as a whole, have not been based on explicit assessments and evaluations of the strategic issues themselves. Instead, the focus has been on inventories of emissions and on monitoring of environmental levels (see e.g. Van Tongelen 2002). The principles and choices in goals and means, their characteristics and implications and their links with other areas of policy have thus not been subject to critical investigations.

The assessments of the EU scientific committees have been focused on the problem instead of its implications and solutions. The same applies to activities on dioxins in accession countries (Anon. 2003, 2004). Also in connection with the EU environment and health strategy the focus in dioxin pilot projects has been on monitoring and on research recommendations (TWGIM 2004a, 2004b). Little studies have been proposed regarding management (cf. IOM 2003). Only some assessments of specific technological solutions, and initial analyses of the risk management system and of specific socio-economic issues also related to fish dioxins (Joas et al. 2001, 2002, 2003) have been published.

The scientific basis of the EU strategies has thus been weak especially in that little research and scientific analyses have been made in the strategic, policy and decision aspects (or related communication issues) – the essence of strategies. This may in part be explained by the fact that the dioxin strategy and the specific recommendations have been developed within existing legislative frames and other constraints, and as a fast response. However, it may also be due to lacking appreciation of the kinds of research and assessment needed to answer the management questions, and to better formulate these questions.

Although further research is needed (cf. Table 37), no single result from research will definitely clinch the argumentation. Even the compound weight of evidence will always be too small to conclusively prove the role of dioxins in health conditions of Baltic Sea populations. On the other hand, it probably will not be possible to exclude harmful effects in some populations, e.g. in heavy consumers of fatty fish, or even severe and irreversible effects in some individuals. As shown above, such verdicts are to a considerable degree subjective and relative; there are no unequivocal criteria particularly at a detailed level (see Weed 2002, 2004a). The same applies to TDIs and other formal decision criteria.

Better balance in research and monitoring is to be found between uncertainties and efforts to reduce them. Some studies have explicitly addressed questions of uncertainty and reduction of uncertainty in dioxin monitoring (e.g., Smith et al. 2002, Judd et al. 2003, cf. 5.3.2). Regarding dioxin risks, as put by Bellett (1990), "there is no statistical algorithm that will convert a controversial and confused situation in a simple objective truth that absolves society from making choices that involve value judgments." While scientific knowledge can dispel some of the confusion, these value judgments and other grounds for policies, decisions and actions need to be better appreciated, distinguished and balanced, also by explicitly studying them and their relationships.

Table 37. Key policy development and decision making needs, knowledge needs and opportunities regarding dioxin-like compounds in Baltic fish and environment, emphasizing knowledge use in integrated assessment and evaluation of strategic options. The needs and opportunities are not exhaustive and are partly overlapping. Cf. Table 34 and text.

Area/stage of risk formation	Policy development and decision-making needs	Additional knowledge production and assessment needs	Opportunities for analysis
Sources and emissions	▪ comprehensive and efficient prevention incl. processes & prod ▪ extended emission control ▪ improved BS action on sources	▪ dioxin-form potential & activity data ▪ diffuse and secondary sources ▪ cost data for controls ▪ utility of steering instruments	▪ root cause analyses ▪ precursor life-cycle assessment ▪ extended BAT evaluations ▪ RBA and SEA for identification of feasible controls
Transport, fate and exposure	▪ blocking key fluxes to sea ▪ sediment protection ▪ bioaccumul control (sea/land) ▪ long-term total expo reduction ▪ added intake cuts securing fish benefits	▪ catchment dynamics ▪ sediment remobilization & sinks ▪ dispersal and cycling in food prod ▪ relations of BS fish and total expo ▪ intakes in young devel stages ▪ kinetics in fish/consumers	▪ fugacity, SPR & budget models for fate prediction ▪ flux modelling & data (in priority gaps) ▪ bioenergetic, fat-based etc models of biokinetics ▪ fish use variation and diet analyses ▪ anal of levels in key media/tissues (model-based) ▪ exposure scenario analyses
Effects and impacts	▪ criteria for acceptable effects ▪ clarifying impacts of policies ▪ alleviation of adverse impacts ▪ balancing risks and benefits	▪ ascertaining attributable effects ▪ effect variability and factors ▪ health in hi-risk groups (human/etc) ▪ mixture effects (DLCs etc) ▪ risk-benefit relationships	▪ epidemiol & mechanism-based (meta)analyses ▪ perinatal develop tox data, dose-response anal ▪ chem-biol interaction models (e.g. focused SAR) ▪ comparative effect and risk analyses ▪ extended RBA (+uncertainty and variation anal)
Perception and communication	▪ reconcile risk views/allow plurality ▪ multi-actor/-sector communication ▪ risk and benefit information	▪ characteristics and factors of risk perceptions and views ▪ risk evaluation & prioritization ▪ factors of fish consumer behavior ▪ efficiency of risk/benefit information	▪ mapping perceived risks of DLCs and other agents ▪ opinion analyses, identification of disagreements ▪ preference analyses for risk management ▪ risk and benefit communication, risk dialogues
Integrated measures of implementation	▪ definition of goals (multiple levels) ▪ integrated R/B based manage ▪ life-cycle control strategies ▪ adaptation & compensation pol ▪ coordination (sector and intl)	▪ compatibility of goals ▪ strategy evaluation ▪ properties of tech and other means ▪ cost information (risks and benefits) ▪ obstacles and synergies in coord	▪ multicriteria/multiobjective policy/decision analyses ▪ extended options analyses ▪ social cost estimation ▪ experimental interventions evaluation
Follow-up	▪ policy implementation ▪ effects on risks ▪ coordinat/cooperat (sectors etc)	▪ definition of metrics and indicators ▪ impacts of actions	▪ risks and use-based metrics ▪ uncertainty analysis guidance ▪ evaluation for adaptation

Explanations: *(cf. List of abbreviations): R/B=risk/benefit, RBA=risk-benefit analysis, SEA=socio-economic assessment, SPR/SAR= structure-property/activity relationship*

9.2 Variability, socio-cultural contexts and implications of risk perceptions

There is great variation in perceptions of risks in general and of dioxin risks in particular (cf. 5.1.2). Part of this is the variation in perceptions of the relative importance of these and other risks. Likewise, perceptions of health benefits from fish vary. The processes of perception, their variations and their causes have great significance for resolution of risks, for risk communication and for management.

The basic views and values influencing risk perceptions are dependent on cultural context (Douglas 1996 (1992)) and on viewer. They interact with knowledge-based views. The perceptions and views about dioxin risks are thus a mixture of scientific influences (some only indirectly derived from science) and non-scientific influences such as fears and desires. Evaluation of research is inevitably modified by the perceptions and opinions of those making or transmitting these evaluations.

In general, different people, disciplines and sectors view the dioxin problem from different angles, and may miss the whole picture and crucial other aspects in it. This can be illustrated by the proverbial 'blind men and an elephant' or, in this connection, a seal (Fig. 24). The men and women engaged in Baltic fish dioxins have not been blind, only with limitations (or priorities) in vision e.g. due to their backgrounds and settings. Attempts have also been made to look at the problem and its solutions more broadly, e.g. regarding also benefits of fish; this however seldom has broadly included ecotoxicological, ecological, fisheries biology or socio-economic or technological considerations.

Dioxins can be regarded as a 'boundary object' (cf. 1.1 and 6) also in the sense that they unite people in their various concerns. This has the positive effect of collective learning and cooperation, building capacities to tackle even other risks and problems. However, this also poses challenges and obstacles in taking into

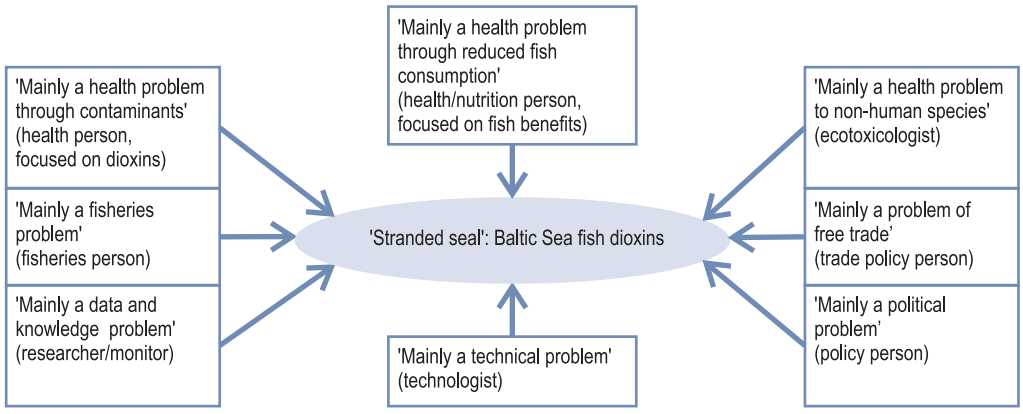


Fig. 24. 'Blind men and the seal': Conceptualization of different perceptions of and perspectives on the Baltic fish dioxin problem.

account the views of others. Moreover, dioxins have become a standard cause to invoke and a symbol of contaminants and other evils that may be too broadly generalizing. For instance, there is much less interest in Toxaphene, long-time contaminants of potentially equal human health risks to those of PCDD/Fs and PCBs in Baltic fish (see SPCFC 2005).

Unfounded risk exaggeration is common, not least regarding food contaminants and other health issues. Even a 'culture of fear' has been discerned, perpetuated e.g. by mass media but also others with an interest in issuing warnings and reassurance (Glassner 1999, Durodié 2003c, Siegel 2005). Fears may be generalized and transferred to other objects, and misplaced in time, arising only after the cause for warning already has passed, in a belated reaction. This may be the case with some of the concerns for dioxins and other POPs, of which often those long since phased out are addressed.

Dioxins and dioxin risks have many characteristics that increase perceptions of their seriousness. Many people thus think PCDD/Fs and PCBs are a recent and increasing problem. However, dioxin risks may gradually become more commonly perceived as 'past' risks that people get used to. A growing tendency can already be discerned that questions the seriousness of dioxin risks, referring to decreased exposures and to other greater risks and, notably, to greater health benefits from other fish ingredients. This 'storyline' might even lead to one-sided and premature perceptions of relief. The suggestion or assertion that dioxin risks are not to worry about may also be seen in some cases as a placating reaction to the undeniably often exaggerated fears of the public. Both alarmist and placating reactions may to some extent be a means of some parts of the administration or of a discipline to maintain organization and to guard its legitimacy and to invoke respectability.

Finkel (1989) has exposed some of the fallacies and biases in the claims that risks are commonly exaggerated and should be 'put into perspective'. However, there is also sound skepticism with regard to risk claims, and there exist many sound arguments for revising dioxin risk estimates and particularly for placing them in a wider context with other concerns. Therefore, there is need for caution and skepticism against hypothetical fears and narrow risk-aversion as well as against unfounded optimism.

There are thus many shades in the dioxin risk debate. It is a complex dialectical process. In particular, as the debate regarding dioxins in Baltic fish is embedded in the interactions between the national and EU level and between sectors, and becomes part of the overall political arena and 'power play', warnings of dioxin risks may be issued or rejected for reasons unrelated to the risks themselves. The rejection of risk claims may likewise be influenced by an apologetic attitude toward fisheries and local cultural traditions.

9.3 Evaluation of risks

Despite the uncertainty about the risks from Baltic fish dioxins, we can say that they are both greater and smaller than commonly thought. This seemingly contradictory statement is justified because they are more multi-faceted than regularly perceived. Therefore they may be in some respects – and to some observers – great or significant, to others small or insignificant (cf. 9.2).

This variety in statements about risks is present also in basic definitions of dioxins and of risks. The risks caused by 'dioxins proper' and even by similar dibenzofurans have on the average decreased also among the human populations around the Baltic. Nevertheless, risks due to dlPCBs, and other less-known DLCs complicate and potentially add to the risks of the 'standard dioxins'. DLCs further cause a variety of effects and corresponding risks both on humans and on non-human animals, and views on these risks thus depend on how effects and receptor organisms are weighed.

A key issue is the attributability of risks to DLCs. Many of the adverse effects and risks ascribed to dioxins (or DLCs) may be possibly or even more likely caused by other factors - other contaminants, and yet other e.g. nutritional, other environmental or host factors. Still, diseases have often been linked with and suspected to be caused by dioxins. This has typically been made based on correlations and on generalizations of mechanistic information that may have been over-interpreted as proving the cause.

A growing amount of disorders have indeed been caused by dioxins – or by TCDD and by extrapolation by other DLCs – in some animals, often at doses and in conditions that do not correspond to those in free-living populations, sometimes in environmentally more relevant scenarios. In some cases, causality has been gradually established, but its strength has been shown to be less than originally thought. In addition, aggregating dioxins or dlPCBs has brought new complications (along with opportunities) in proving a role for the group of compounds originally suspected.

No conclusive proof, by established (but variable and equivocal) epidemiological criteria, has been found for human health impairment due to DLCs in Baltic Sea fish, particularly for irreversible and grave emergent effects. Nevertheless, it has been commonly regarded that given the variability in risks, some effects may be caused by DLCs in part of the population even at present exposures. In high-exposure groups such as high consumers of fatty fish a relatively greater fraction is at risk. Even in these cases there is great uncertainty due in part to lacking studies and resolution, but also due to the extrapolation e.g. from non-human animals that often dominates the inference. The risks and adverse effects are moreover likely to be balanced or compensated for by health benefits of such fish.

The verdict on ecological effects also depends largely on the frames of reference and the criteria set for significance and proof. Especially in ecological field studies of DLCs, proof is difficult to obtain. Many species and populations have not been studied or are even observable. While this may increase the risks in hypothetical sense for some populations and settings, it lowers the possibilities for establishing causation and thus evidence-based risks. The exposure levels of many species in the Baltic have exceeded benchmarks for biological effects, but these are often tied to subtle effects and are uncertain due to the many assumptions used, especially regarding causation. Individual-level responses may also not translate into population-level adversities.

Exposures to most DLCs in the Baltic biota have generally decreased several-fold. The temporal development in risks is not the same due to latencies, and adverse conditions may thus still develop. However, by and large it can be stated with some confidence that the worst is over. The non-human animal populations that may have been harmed in part by DLCs have also considerably recovered. This is no guarantee that adverse effects could not still be caused, but does give powerful indications of reductions in dioxin risks. In this respect dioxin risks from Baltic fish are smaller than commonly perceived.

It may seem puzzling that dioxins are considered presently to pose so great risks also to consumer health, when even the earlier high exposures have not caused clear adverse effects in the human populations studied. A partial explanation is the limitations of the studies made. Another explanation is growing awareness of risks caused by dioxins and PCBs in food and feeding-stuffs but even more generally. Thus, signals and fears of earlier risks from dioxins and of risks in other areas may carry over (cf. 9.2).

In a general evaluation, the improvements that have taken place have to be put in relation to the possible lack of continuous improvement. The relative significance of these factors is essentially a question of optimism and pessimism, i.e. in how far the past improvement gives cause for relief, or whether despite it still stricter, more extensive and immediate management goals and actions are considered justified.

9.4 In search of balanced precaution

Better safe than sorry, it is said. However, excessive precaution, unfounded fear, automatic suspicions and narrow focus can prevent people from looking at risks associated with dioxins more objectively, comprehensively and in a balanced manner, taking into account alternative explanations and factors. There may be an over-reaction to fish dioxin risks especially when not considering benefits from fish. As shown in this assessment, actions may also be rushed into that are not efficient but could instead cause new problems and risks.

Sometimes such claims and actions may turn out right in hindsight, e.g. preventing or at least alleviating risks, or speeding up their abatement. This is not guaranteed, e.g. when applying a standard toxic centered precautionary approach that ignores other concerns even when connected. Mistakes and unnecessary actions can be made, along with missing actions that would have been needed. Moreover, even though suspicions would happen to prove correct, they may represent a faulty logic or process. The end result is not the only criterion that would justify any means.

If conclusive proof were required of dioxin-attributable risks, management actions would be late and insufficient. With dioxins, conclusive proof may never be obtained of adverse effects. On the other hand, if further action would be taken on the basis of remote suspicions and generalizations and by consideration of dioxin risks only, the result could be unnecessary and even directly harmful action.

There is an inherent tension between precaution in the sense of non-reflected reaction and based e.g. mainly on the perceptions of some sector (or some group, such as poison-fearing consumers), and precaution of more thoughtful kind: based on broad and balanced information, evaluation, planning and multi-partner negotiation. Reconciliation of this tension emerges as a key task in both research (in the natural and social scientific aspects) and in policy and practice.

In general, one needs to distinguish between 'false alarms' and 'true warnings', both of which have been common (Mazur 2004). It was argued by Pacala et al. (2003) that also the alarm over environmental false alarms has been false. These authors discussed cost-benefit relationships of precaution, pointing out e.g. that most anthropogenic hazards had caused considerable harm by the time their effects were even understood, let alone regulated, that most false alarms are discarded before they engender serious costs, and that mitigation costs are often far less than initially projected because of induced technological change. It was claimed that the marginal benefits dwarf the highly uncertain marginal costs in most cases. However, little data and analyses were provided to support this, and the multiplicity of interactions and consequences and the possibility of off-setting other risks and impacts were not properly accounted for. With dioxins and PCBs this problem takes on additional complexities associated with indirect costs due to loss of benefits and with lagged risks - and lagged fears due to earlier greater risks.

Precaution, i.e. careful foresight of unwanted consequences, is needed also before measures are taken that may reduce, even irreversibly, health or ecological quality. Precaution is not a monopoly of chemicals control or environmental policy although it has prominently developed in these areas. Precaution is equally necessary (and problematic) in other areas such as general health care and natural resource use. If this is not realized, a 'risk myopia' narrowly focusing e.g. only on the toxicity of dioxins, and thus a distorted view of risks and decision characteristics, easily results.

Some claims of dioxin risks have turned out to be true, in some sense, and have led to needed action or have justified previous action. This applies e.g. to Hormoslyr, a Swedish herbicide that was found to contain TCDD and has been hailed as a showcase of pay-off from a ban based explicitly on political precaution. Other suspicions and findings have carried other benefits, including indirect ones such as a more general insight in and devotion to environmental protection. Thus, the alarmist approach to dioxin risks should not be branded too easily and opportunistically as misled and exaggerated, either. However, we have come to a situation that is in some ways new. Exposures to PCDD/Fs and PCBs have decreased to a fraction of the peak levels. This is a strong reason why added strong and cross-the-board precautionary action on dioxins may imply overdoing risk management.

A thought experiment (cf. Kuhn 1966 (1977)) may illustrate the perils of unheeded precaution in the present case. If marketing of Baltic herring had been banned in early 1970's when the

DLC levels were peaking, on the basis of some powerful precautionary action employing the present EU action level, the herring fishery would have collapsed. In hindsight, that could have been regrettable, as much of the health benefits from this food could have been lost due to shifts to unhealthier diets, and as the benefits probably exceeded on the whole population level the risks from DLCs even at their higher levels, assuming they now exceed the risks by at least an order of magnitude (Leino et al. 2005). If this would have been the case, it could be considered lucky that such precaution did not carry the day. The benefit/risk ratio is now still higher. It therefore seems likely that such regulation today, even with lower ambient DLC levels and less herring affected and with uncertainties regarding dietary substitutes and consequences, would also be detrimental to human health. Thus, in this case a slow reaction to the problem by regulation of food consumption seems to have been sufficient, as other risk abatement actions helped reduce the exposures and risks without such side effects and counter-veiling risks.

A shift in approach to dioxin risks is influenced by an extension of concern to the consequences of management actions. With dioxins and PCBs in fish we have to deal with the complicated problem that reduction of exposures by the immediate response (shunning fish) will expose us to new and potentially worse risks. In broadening perspectives on risks, it is also legitimate to ask whether the attention should be refocused on some other (environmental health) problems.

Likewise in prevention of PXDD/F formation, superficially a clear-cut priority area of risk reduction, difficult choices need to be made regarding products or processes involving halogens. Often it is assumed that one simply bans toxics or their precursors, and gradually the alternative products and systems in society will be developed and the benefits of proactive and prompt action reaped. However, many substances potentially forming PXDD/Fs already have been or are being reduced, and have also grounds for some usage, even environmental and health related; their alternatives may moreover be harmful. In the name of precaution, action may be rushed on such products without awaiting proof or at least informed assessments that the suspects and the actions are the right ones. Also the control of dioxin precursors thus needs to be addressed in a wider context (cf. Durodié 2003c).

It needs to be asked what lessons can be and have been drawn from DLCs e.g. in the Baltic. Their precautionary lesson is double-edged. On one hand, it may help prevent additional (also similar) risks. Many spin-off benefits from research and other activities on Baltic dioxins can moreover be seen. On the other hand, they may teach a lesson of exaggerated reactions to risks only remotely and more slightly reminiscent of those that gave the lesson, while these have already been diminished. Precaution may be associated with 'phantom risks' – reflections of risks that were originally greater.

It may be justified to look upon dioxin risks as a success story – a sad one perhaps, because of the concerns and losses it has caused, and one possibly due in part to chance, not design. However, one may also learn from it that caution is needed to avoid pushing the frontier too far. This is a real danger because of the inertia in the societal system – and the stigma that dioxins now carry, conceivably as one of the symbols of the modern version of the risk-blaming process (Douglas 1996 (1992)).

We thus need to thoughtfully evaluate the multi-dimensional choices we face in balancing the risks, benefits, costs and impacts of fish dioxin reduction, as well as of dioxin prevention, and develop scientific and pragmatic procedures to aid in these evaluations and in the responses to their results. In doing this, we need the inclination, courage and sustained effort to question old truths – and to look for new ones. This amounts to careful navigation between and reconciliation of extremes, e.g. excessive and insufficient precaution (Fig. 25).

9.5 Conclusions on approaches to risks and management of dioxin-like compounds in Baltic fish

9.5.1 Improved consideration of multiple risks and impacts from dioxin-like compounds and of fish

For efficient management of risks associated with dioxin-like compounds in Baltic fish, these risks are to be considered in a manner that is more extensive and many-sided than previously and

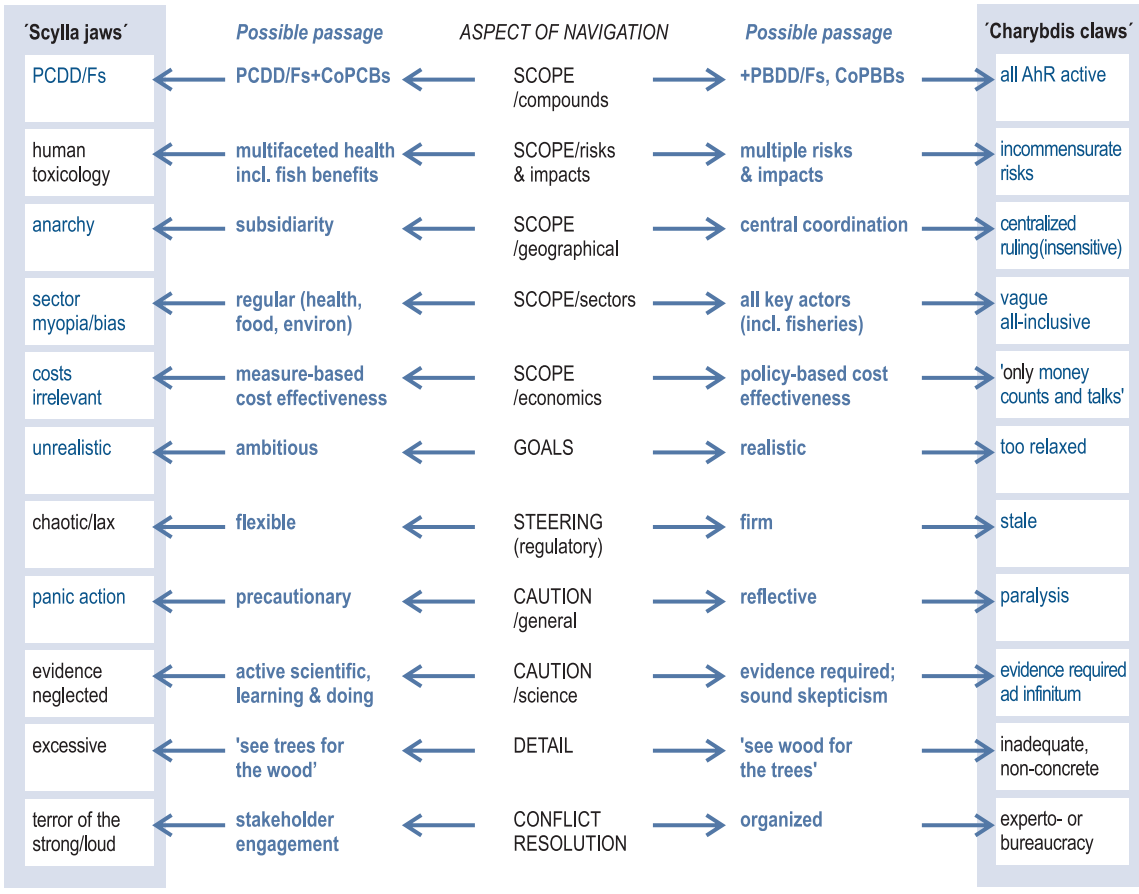


Fig. 25. Dire straits in the Baltic Sea dioxin Odyssey, with particular reference to framing of issues and dealing with uncertainty and precaution (modified from Assmuth and Hildén 2002).

more fully reflects the multi-dimensionality and complexity of risks and associated uncertainties. A narrow focus may be misleading and detrimental (see Fig. 25). In particular, when not only risks but also management is analyzed, a broader outlook is necessary to facilitate other considerations and interests.

Separation and narrowness in perspectives and approaches are to some extent understandable and in some cases and respects even necessary (Fig. 25). Dioxins have after all been by definition the focus of the studies and strategies addressing them. Therefore, it is natural that other aspects of management have not been given so much thought in these connections. Instruments have been devised and set in to address predominantly the 'risk side' of food including fish, not the 'benefit side'. The EU dioxin strategy and its application to (Baltic) fish are also to be extended and developed. In particular, the multiple benefits from consumption of fatty fish are to be accounted for.

However, even the assessment of benefits needs to be augmented and nuanced. The distributions of benefits and risks e.g. among population groups and the uncertainties in benefits need to be considered. In particular, the dietary alternatives are important to simultaneously avoid risks and secure benefits. The health benefits from fatty fish are thus no omnipotent patent argument for its unlimited consumption. Instead, multiple consequences from and conditions for actions need to be considered. Nevertheless, the health benefits from fish do seem to increase the attractiveness of information-based steering approaches instead of management by market regulation based on fish dioxin levels.

In addition, a still broader array of consequences and factors need to be considered. They involve multiple and complex risks and benefits, direct and indirect, including ecological and social. Toxicological risks from dioxins and other agents are only one part of the full picture, although important; so are the benefits from fish consumption. If risks and consequences are considered in isolation, the chances are that misleading conclusions will result.

Human health is the focus of most people and organizations in dioxin risk management. This involves considerable limitations. In general, ecotoxicological risks from Baltic fish dioxins may be regarded as more severe than their human health risks. The differences in risks between humans and non-human animals may be seen as grounds to refocus the current assessments and strategies. The difference between human and non-human risks on the other hand does not matter in all areas of management, e.g. prevention of dioxin formation and of emissions and immissions to the sea.

Also the toxicological risks associated with dioxin-like compounds need to be addressed in a more multi-dimensional way. These extensions in consideration include the following:

- *Other dioxin-like compounds*, such as some polyaromatic compounds and potentially PBDD/Fs and mixed bromo-chloro-dioxins, depending e.g. on the courses of their emissions
- *Additional contaminants in fish* that may add to or, alternatively, attenuate the risks from dioxins, such as non-dlPCBs and, as compounds of growing concern, brominated flame retardants
- *Community and ecosystem level* risks and impacts, including both risk-increasing and risk-attenuating factors, especially as the emergence of lower-level effects is considered
- *Precursors* of dioxins, both as root causes and potential solutions of the dioxin problem and as causes of risks (more or less commensurable with those of dioxins) in their own right
- Toxicological risks caused by dioxins *outside* the sea and direct human consumption of fish.

Even considering these aspects, the magnitude of the risks from DLCs in Baltic fish may not increase greatly. These additional aspects and even other classes of toxic compounds also may not obviate the overriding importance of health benefits from fish (see also SPCFC 2005). However, additional facets of the risks and new issues in management may emerge (cf. 5).

Uncertainties need to be addressed on multiple levels. A fuller account of uncertainties does not only add to management difficulty but can instead help find priorities and solutions. By considering the broader picture of risks and the relative uncertainties in them it may be possible to conclude that some uncertainties do not probably matter that much (see e.g. Ahmed et al. 1993). These may include e.g. uncertainties of sources and compounds that are known to have already much decreased in comparison to others. Likewise, when realizing the uncertainties in TEqs, TDIs and limit values, the required precision in some areas may be relaxed. However, this does not mean that important uncertainties can be identified certainly and objectively. Partly therefore, there is justification also for additional research in dioxins and their effects (cf. 9.1).

A more comprehensive analysis of the multiple risks, benefits, impacts and uncertainties associated with DLCs in Baltic fish will require communication and collaboration between various activities and actors on many levels (cf. Fig. 26). The fisheries sector needs to be better integrated with others, and risk-benefit analyses need to be intensified also in the food and health sectors.

9.5.2 Improved consideration of whole risk chains and of alternative actions

The focus in dealing with dioxins has been on exposures and direct toxic effects. As discussed above (8), the EU dioxin strategy (EC 2001) has concentrated on reduction of human exposures mainly through food and feeding-stuff quality standards. Such responses to the problem are inherently limited.

Part of the risk chains for human health have been considered more extensively within the food production system, including feed cycles and recycled materials. Emission control, including prevention of PXDD/F formation in facilities, has also been subject to many EU activities (cf. 7). This has been successful; a challenge is to feasibly further reduce secondary emissions. Many of the reductions in subsequent stages are already in the pipeline.

Relatively little activities have been identified in the area of product-oriented prevention of dioxin formation. This aspect has been implicit in some regulatory actions on halogenated chemicals, especially chlorinated biocides and some industrial chemicals. Some dioxin-forming chemicals are still produced or used in EU. An area of dioxin prevention that has potential importance are the root causes for their formation, including underlying societal needs. More systematic evaluation of the dioxin-forming

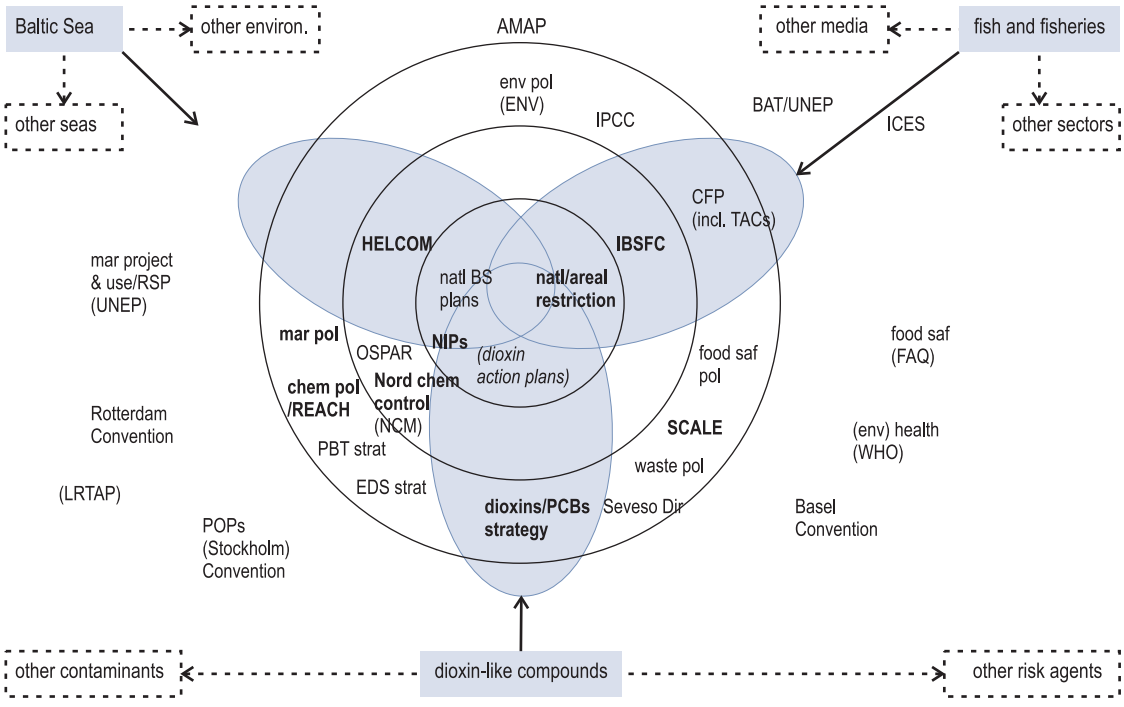


Fig. 26. Important existing or potential policy instruments, processes and actors on various levels of governance (from centre: national, regional, EU and global) in dealing with risks from dioxin-like compounds in Baltic Sea fish and with associated management issues. For abbreviations, see list of abbreviations. Key instruments and actors, indicating areas of communication, cooperation and coordination, have been shown. Note the differing specificity of the instruments and their direct or indirect roles in relation to the Baltic Sea, fish and fisheries, and dioxins (indicated by ovals).

potential of products and processes could be made. In particular, additional analyses of socio-economic, technological and other prerequisites and impacts of alternative prevention measures might increase the efficiency of steering. Multi-dimensional consideration needs to be given to the consequences of alternatives, including pros and cons due to PCDD/Fs management actions but also other factors (cf. 9.4).

In downstream stages along risk chains there also are considerable gaps. While the regulation and control of dioxins and PCBs in foods and feeding-stuffs have been given much emphasis, other measures in the food production system of main interest here, fish and specifically Baltic fish, have until now received little attention. The EU production systems for fish largely in aquaculture have been addressed (SPCFC 2005), but the specific problems, solutions and factors in the Baltic fisheries based on both farmed and particularly wild fish need additional consideration.

Within measures directed to reduction of human exposures, several approaches are conceivable. We have distinguished the key alternative strategic approaches of the EU market regulation based management and the alternative diet advisory based approach, relied on e.g. in Finland and Sweden, and subjected these to initial comparative evaluations. Also intermediates, combinations and modifications of strategies are conceivable, and in both approaches more specific measures e.g. within information and regulation may be needed and achievable. Regardless of the specific details in these, they do justify the conclusions that both approaches have pros and cons, that the choice depends largely on the criteria defined and the valuations given, and that the EU approach in any case cannot be claimed to be superior. On the contrary, as shown in some detail, it involves considerable problems and risks especially when emphasizing health in a more integrated way, including benefits from foods like fish. These evaluations also illustrate the kinds of questions arising when trying to conceive and compare management approaches based on consideration of their multiple impacts.

Many risk management opportunities exist also in other stages of risk chains. Above all, in future analyses and in developing risk management, it is considered important that broad assessments of many options in the various stages of risk chains are made (Table 37).

9.5.3 Improved consideration of Baltic Sea processes and factors in relation to other scales

For assessing and managing risks from dioxin-like compounds in Baltic fish and in the sea generally, its properties and processes need to be better taken into account. This is related to the regional aspect and to balancing of specification and generalization. The consideration of Baltic Sea processes may also be seen as a special aspect of multi-dimensional assessment of dioxin risks and of the need to broaden the scope to a more comprehensive outlook from toxic compounds and their direct risks.

Many processes and developments take place in the Baltic Sea and in adjacent systems. Many of these, notably social and political developments but also natural processes or natural-human processes, are not predictable, are beyond the scope of this assessment, or have connections with dioxins in fish (or elsewhere) that are not strong or clear.

The Baltic is in many respects particularly hard pressured by and sensitive to DLCs and also other stressors. The loading has been relatively high, the retention of persistent compounds in the system likewise, and many features such as osmotic and climatic factors render the sea vulnerable (cf. 5.4.3).

The ecological processes of the sea that affect and are affected by dioxins require additional consideration, and add to the complexity in managing risks and impacts. In particular, eutrophication and the development and management of fisheries are key processes that are inextricably intertwined with the fate and effects of dioxins and their management.

In a Baltic Sea connection, opening up to other areas of concern implies some things in particular:

- Better integration with fisheries management, including consideration of the value of the resource
- Better consideration of eutrophication as the key process of change in the Baltic
- Better consideration of interactions between the sea and land areas
- Better consideration of indirect impacts, also socio-economic, technological and systemic
- Better consideration of the particular problems and solutions of the actors and their cooperation, e.g. between the old and new EU member states and between EU and HELCOM

Eutrophication has still often been considered separately from DLCs. In terms of cycling of substances in the sea and its food chains, the link has repeatedly been made. However, at the level of policies and management it has remained less developed. This is reflected e.g. in simplistic juxtaposition of eutrophication control and reduction of dioxin exposures. A more nuanced approach is needed. Eutrophication presents an instructive reference for dioxin risks also because some grounds and systems for assessment and management have already developed in this area that may be utilized for dioxin management. This at the same times introduces the need to refocus the eutrophication related fields of marine protection and management on the specific problems with dioxins and other POPs.

Baltic fisheries are the vehicle transmitting the risks from Baltic fish dioxins to humans. However, the processes in this transmittal have seldom been scrutinized. In particular, fisheries have hitherto not been regarded as a key area in solutions to fish dioxin risks, with few exceptions (e.g., Mackenzie et al. 2004). Neither have the impacts of dioxins and dioxin management on fisheries and associated chains of industrial and general societal activity been much addressed. The fisheries sector is a subject of regulations as well as an actor implementing these regulations. This creates capabilities to deal with fish dioxins, but also some bias in evaluating associated problems and solutions.

To take on board these issues will require more detailed, systematic and integrative consideration of the assessment and steering systems than has been possible in the present work. However, some aspects in this integration have been identified, some discussion of associated issues has been made, and directions for future work have been outlined. As exemplified by fisheries, not only natural but also technological and social processes need to be considered in dealing with dioxin risks.

Background, objectives and approaches

This report has been produced in the Nordic Council of Ministers project Risk and management strategies of dioxins in Baltic Sea fish – an interdisciplinary assessment, for Chemicals Group (administrator), Sea and Air Group and Environment and Fisheries Group. A SYKE team and a network of experts have contributed. The overall aim is to analyze and improve the support for discussions and decisions concerning risks associated with dioxins in Baltic fish, especially in connection with the EU strategies for dioxins and PCBs e.g. in food. The approaches have included document studies and expert opinion analyses. Comparative evaluations are made for characterizing risks and strategies. Annexes are reported separately.

A. Risk assessment

1. In framing risks, focus and breadth are balanced in several dimensions. Along with 2,3,7,8-chlorodibenzo-p-dioxins and furans (PCDD/Fs), also dioxin-like PCBs (dlPCBs), other dioxin-like compounds (DLCs) and their precursors are considered. Human and non-human risks are both included. Various kinds of risks are addressed, including indirect risks and impacts. Conditions in the Baltic Sea and in linked systems (also on land) are considered.
2. Sources of DLCs and the contribution of sources to immissions to the Baltic are still uncertain especially for diffuse and secondary sources and non-PCDD/F compounds. However, emissions to land, sediments, and wastes dwarf the direct emissions to air. Key PCDD/F sources have included chlorobiocides, metal, forest and some other industries, and combustion. Many organohalogens have dioxin-forming potential. PCBs and most PCDD/Fs are anthropogenic. The PCDD/F load to the Baltic, as TCDD equivalents (WHO-TEq_{DF}), seems highest from Poland. The sources of dlPCBs include stockpiles and wastes of PCB products and combustion. A part of the load comes from outside the catchment and EU.
3. The load to the sea in atmospheric deposition is c. 400 g WHO-TEq_{DF} a^{-1} and some 10 % of this in PCB-TEqs (WHO-TEq_P). Runoff and adsorption from gas phase add to these. The DLCs cycle in the sea especially in organic carbon. The accumulated pool in the catchment enters the sea after lags due to recycling in solids. Sedimentation may exceed influxes but from shallow bottoms much DLCs reenter the water phase. Sediments thus constitute a key secondary source. Degradation removes only part of the load and forms also toxic products. PCDD/Fs and especially dlPCBs bioaccumulate in food chains in fat. The efflux in fish catch is c. 1 % of the external load of WHO-TEq_{DF} and some more of WHO-TEq_P. Eutrophication affects DLCs in complex ways, not only diluting them in key food chains.
4. The Baltic is still heavily contaminated by DLCs. Their contents are high especially in aged herring, salmon and other fatty fish. Levels in Baltic biota have however decreased to a fraction of peak levels, although with variations and fluctuations. Further declines will take place at increasingly slow average rates. Some DLCs from Baltic fish are spread in fish-based feeding-stuffs (see A.15). The Baltic dioxin and PCB levels are generally much higher than in most other seas, but in some biota are at or near those in the North Sea.
5. Dioxin and PCB intakes by citizens of Baltic Sea countries have decreased (cf. A.4). The intake distributions are skewed, exceeding e.g. 2 pg WHO-TEq_{DFP} kg^{-1} d^{-1} for c. 12 % of adult Swedes. In Finland the share of fish of total WHO-TEq intake is higher (60-90 %) than in other Baltic Sea countries; yet, the mean total intake by Finns is not higher, due to cleaner other foods. Intakes are still high in populations consuming much wild fatty Baltic fish. Consumption of Baltic fish is poorly known especially in children. Fetuses and breast-fed infants receive

proportionally greater loads. Body burdens of PCDD/Fs and dlPCBs in the general population have considerably decreased, but are still high e.g. in persons who have consumed great amounts of fatty Baltic Sea fish. Key DLCs (see A.10) accumulate more strongly than TCDD. Also other DLCs are found in humans.

6. The biological effects of DLCs are multi-dimensional, affected by many factors and well known only for TCDD. The effects are mediated largely by the aryl hydrocarbon receptor (AhR) but other mechanisms are involved especially for PCBs. Many effects are shared by several animal groups and arise at low doses, but toxicity differs greatly also within a species. Effects on development, reproduction and immune functions are crucial. Partly related to these, hormonal and neurological effects are notable. TCDD is a multi-site tumorigen in many animals, evidently also humans, but the cancer risk due to DLCs is not particularly high. Some non-additive mixture effects occur between PCDD/Fs and dlPCBs. In addition, non-dioxinlike PCBs in particular antagonize some effects of DLCs but may add to others. There is no straightforward way to account for such factors. The effects of DLCs in the Baltic are influenced by its traits, e.g. high vulnerability. Other stressors compound and aggravate the effects; yet other factors cause adaptation and compensation.
7. There is inconclusive proof of adverse human health effects from exposure to Baltic fish PCDD/F or dlPCBs. The possible effects can be assumed to have decreased from peak exposures in 1970's. The worst risks are thus over. Effects from peak exposures have gone unnoticed, in part due to limited studies. Some studies have found links between consumption of fatty Baltic fish and effects like sex ratio shifts among offspring and immune alterations, but such evidence is tentative and the roles of DLCs unclear. Benefits from fish may have compensated for possible adverse effects (see A.12). High consumers of fatty wild Baltic Sea fish are still continuously at elevated risk.
8. In some non-human animals such as some fish-eating seabirds and especially seals, past disorders were probably caused by PCDD/Fs and especially dlPCBs in Baltic fish. However, in many of these cases other contaminants and factors may have been important. Based on extrapolations, exposures remain at a level of concern in some species. Most susceptible stocks have recovered, but some are below natural reproductive capacity. There is a continuous possibility that DLCs impair animal well being. However, PCDD/Fs and dlPCBs do not presently pose a proven grave ecotoxicological risk (see A.9). Community and ecosystem effects are hard to establish and are affected by many factors.
9. Risk evaluation depends on the criteria set for proof of effects attributable to DLCs, and on associated causal inference. By some traditional criteria, clear population effects from DLCs cannot be proven, due e.g. to the presence of many potential cofactors. Biochemical, transient, high-dose and single-dose and *in vitro* effects may not be more generally relevant. The generalizability of experimental non-human animal data and models becomes a key issue, unsolvable merely on scientific grounds. Risk evaluation also depends on effect significance and adversity, not objectively and unequivocally definable.
10. There are many variations in risks: geographical, temporal, taxonomic, individual, and between various DLCs and effects. Local hotspots also in the catchment can disperse DLCs. Temporal variations in risks involve irregular changes, blurring trends. Risks from DLCs in the Baltic will linger for long. The greatest dioxin-type risks are generally due to 4-PeCDF and CB 126, but also other PCBs are important in consumers of Baltic fish, and unidentified polyaromatic DLCs may even dominate total dioxin-type effect in bioassays. Humans are not clearly less sensitive but many other animals are exposed to Baltic fish to greater extent. Exposure patterns and biological factors including genetic polymorphisms affect risk variations. Risks to young are greatest but also other age groups are at risk for some effects. Elderly sea fishers, their spouses, their 30 to 40 year old children and others who consumed lots of the most contaminated fish (in 1970's) constitute particular risk groups.

11. Risks caused by Baltic fish dioxins can in some cases be compared with other risks, e.g. other organohalogens in Baltic fish and DLCs in other foods. Epidemiological risk is related to background risk for the endpoint and population in question, uniting all causes. In such comparisons dioxin risks often pale as greater and more certain other risk factors such as biological or life-style related can be found for the adverse effects. Yet, fish dioxin risks may qualitatively differ from reference risks e.g. as to effect profiles, time dimensions and reducibility. This may emphasize or de-emphasize dioxin risks and alter risk-benefit relationships. Multi-dimensional comparisons can guide to efficient solutions.
12. The health risks from dioxins in fatty Baltic fish are matched and probably exceeded by health benefits of consuming such fish and fish oil. Benefits seem particularly great for cardiovascular health in high-risk groups and possibly for early development. Despite higher DLC levels, the net health risk of fatty Baltic fish may thus be lower than that of some alternative foods. Risk-benefit relationships can however not be reduced to simple and one-dimensional risk/benefit ratios. More attention is to be given to variations and uncertainties in risks and benefits e.g. in age groups (the benefit-risk ratio of consuming wild fatty Baltic fish being high particularly among the elderly with elevated risk for cardiac mortality, and lower among the young susceptible to developmental disorders from dioxins and other contaminants), to qualitative differences of risks, e.g. morbidity, and to possible dietary choices (see B.7).
13. The basis of Tolerable Daily Intakes (TDIs) is partly unclear. The safety factors in their derivation could be increased e.g. if accounting more fully for individual variation or congener kinetics. There are thus arguments for a TDI below the present 1 pg WHO-TEq kg^{-1} d^{-1}. On the other hand, many factors can lower safety factors and increase TDIs (see A.8). Adjustments in TEFs can also increase or decrease risks depending on the case. Quantitative criteria for ecotoxicological risks can likewise not be unambiguously defined. Such benchmarks depend on the rationales and assumptions used. Thus, it cannot be stated definitely if dioxin risks exceed safe levels. Such uncertainties and also non-scientific aspects of critical risk measures should be openly acknowledged and systematically analyzed.
14. Psychological and possibly psychosomatic effects are caused by dioxins in fish, and cannot be categorically separated from 'actual' biological responses. Both exaggerated reactions to dioxins in fish and denial of related risks take place. Reactions to risk management are varied and may be significant. They are influenced by individual sensitivity and conditioned by socio-cultural and situational factors. They operate among experts and others, and present challenges to risk inquiry and communication (see B.15).
15. Indirect risks may be caused by DLCs e.g. in sprat through fish-based fodder used in food production. However, the EU recommendations for dioxins in fish (see B.2) have reduced the risks, and the benefits, from such uses of Baltic fish (cf. A.12, B.5). Indirect risks are also associated with other socio-economic impacts of DLCs and of abatement measures that have repercussions on natural and social processes. Thus, the indirect risks from Baltic fish dioxins do not only increase total risks but also make them more complex.
16. Uncertainties of risks prevail despite research, monitoring and assessment of PCDD/F and (to lesser degree) other DLCs in the Baltic and in general. Uncertainties are caused by variability in data, but often more by models and decision rules, which represent uncertainties or ambiguities of a higher order and are also of a qualitative nature. Risk management involves great uncertainties both of technical means and of socio-economic aspects (see B). The coverage, precision and accuracy of dioxin measurements and models should be conditioned by their uses, and illusory precision resisted. Uncertainties are not to be exaggerated; instead, they can guide risk assessment and management (see B.12, B.13).

B. Risk management

1. Many policies and strategies are relevant for managing risks from DLCs in Baltic fish.

These strategies have variable purpose, scope, regime and characteristics. They include instruments focused on dioxins, Baltic Sea or fisheries. Many important instruments e.g. in EU and within the POPs Convention are continuously evolving. Their implications for Baltic fish dioxins cannot be evaluated in concrete terms (see e.g. B.16).

2. The EU strategy for PCDD/F and PCBs (EC 2001) is general and focused on human health risks, does not address regional issues and includes little on prevention and emission control. The Community recommendation on these compounds in feeding-stuffs and foods (EC 2002a) has a market regulation bias, is still focused on PCDD/Fs and does not incorporate health benefits of various foods. This management approach can thus not be regarded as the most adequate one for Baltic fish or generally. It could lead to unfocused, piecemeal and inefficient regulation (see B.5).
3. The acceptable levels of dioxins in foods and feeding-stuffs are still more unclear than TDIs (see A.13). The conversion of TDIs to acceptable levels should account for how fish is used in comparison with other foods and feeding-stuffs, and for variations in use. The models used to derive acceptable levels in the EU recommendations have not been explicated. Inclusion of dlPCBs would increase the apparent risks. If on the other hand the health benefits of different foods would be included, a certain consumption of fish exceeding maximum levels could be justified. The basis of ecotoxicological quality criteria for ambient DLCs is also weaker than for those in intakes. Generally, fish dioxin contents need to be evaluated more extensively and in relation to other food, health and environmental criteria.
4. The key present alternative *de facto* strategy to managing risks from dioxins e.g. in Baltic fish to human health is diet advisories. Also this approach has limitations and problems. The resulting risk reduction is uncertain. On the other hand, the information-based approach is able to account for regional and specific conditions and factors. It can address high-risk groups and may be easily accepted. It is flexibly compatible with other areas of management and changing evaluations of risks. Balancing of risks and benefits is relatively easy.
5. Both the EU approach and alternatives have positive and negative impacts in the Baltic fish context. Limitations of fish marketing by strict application of the EU recommendation are problematic. There is a risk of losing benefits from dioxin intake reduction by cutting health benefits from fish (see A.12). Collapsed Baltic fishing reduces employment mainly in Sweden and Finland. Important cultural and systemic roles of fish consumption and fisheries are hard to recreate once lost. Decline of fisheries may increase ecotoxicological risks by altering DLC cycling. The EU approach has justification mainly in the more certain reduction of exposures. However, management by essentially a single objective (minimized human toxicity) and criterion (fish dioxin levels) is narrow and potentially inefficient.
6. As to principles, a key question is to what extent can people be let choose what to eat, and what interventions in food supply are needed. This is related to subsidiarity, flexibility and normative governance. Some subjects like small children cannot choose. However, such subjects are legitimately exposed to also many other harmful agents, even in foods. On the other hand, realization of free choice may presuppose e.g. information on the catch area.
7. The crucial question becomes: can risks from dioxin-laden fish be avoided while ensuring its benefits, i.e. are there good and sufficiently certain alternatives to dioxin-rich fish diet. Reduced consumption of fatty wild Baltic fish may not necessarily lead to unhealthy diets. Alternatives include low-dioxin fish and fish products from the Baltic and elsewhere, and other healthy fats. Their potential may be realized e.g. by incentives and advice. Such alternatives however also have limitations and uncertainties. The benefits from fatty Baltic fish are thus relative, and make no omnipotent argument for their consumption.
8. A desirable strategy for DLCs in the Baltic could focus more on preventing dioxin formation. This involves products and processes, down to root causes like societal needs. Chemical regulations and many other instruments can be used. They require

improved implementation for DLCs. The formation of dioxins should be taken into account in halogenide production, use and treatment. Many dioxin precursors are used and produced in great volumes also in EU but not prioritized for control. A preventive approach is able to comprehensively and efficiently reduce risks. However, risk-benefit relationships should be analyzed in more depth, as all alternatives to dioxin precursors are not risk-free.

9. In emission control, key land-based sources of DLCs (see A.2) are a priority to efficiently reduce long-term risks. Many easy options are being exhausted, but emission reduction does proceed based on the instruments already prepared. The means include application of existing technologies and development of new ones e.g. for metal industry and combustion, including treatment of pollution control residues. There are opportunities within technology improvement especially in Poland. Measures can be desirable for also other reasons than control of DLCs. Careful consideration of all impacts is needed in cleanup of hotspots. Cleanup is generally more cost-effective on land than in sea. Cleanup in River Kymijoki could remove 4 kg WHO-TEq (ca. 10 % of the pool in the sea) for 5-100 M€, depending on technologies used and costs included, but impacts e.g. on fish TEq levels are uncertain.
10. Measures in the sea need to be considered more extensively, as part of marine strategies and fisheries management. In particular, fisheries may be a key part of solving dioxin problems (see B.5). Opportunities include cleanup fishing, restriction of sea discards, selective fishing in young age classes for control of bioaccumulation, and quality control of farmed fish. Dredging controls are warranted by the importance of sediments as a source. Protection areas and measures are an option for some animal species endangered also by DLCs.
11. Options to reduce risks in use of fish, apart from fish market regulation or diet advisories, can be linked with other parts of DLC management. Diversion of dioxin-contaminated fish to less sensitive uses and appropriate later treatment remain a need. Fish oils can be cleaned of PCDD/Fs and some PCBs. Clean surrogate foods may be given to more species (see B.7). Pharmacokinetic control and therapies have limited applicability in some cases. Health benefits of fatty fish may be enhanced (see A.12). Compensation for losses is not a substitute for fisheries and involves problems. Liability issues are to be addressed taking into account inadvertent pollution and retroactivity, and other economic steering instruments.
12. The merits and drawbacks of alternative management approaches are not unequivocally definable. Recommendations should thus be subject to further discussions, experimentation and evaluation. It can yet be concluded that many factors favor a liberal approach to market supply of fatty Baltic fish within a more comprehensive management. Positive interventions can provide alternatives to restrictions. Controls on fisheries, mariculture and fish (product) trade need not be bound to strict regulation of dioxin levels. Such a multi-frontier approach may best ride out the remaining risk in Baltic fish, while ensuring their sustainable use and improving coordination with other management areas and general dioxin strategies. A combination of options is needed in all strategies, e.g. due to lags associated with preventive measures. Options can become more efficient and feasible by innovation.
13. Dioxin risk management hinges on interpretations of the precautionary principle. Balance is needed to avoid both belated measures and panic actions. Dioxins and PCBs are a major argument for this principle, but it should not be applied uncritically and selectively. If exaggerated it can lead to wasted resources and worsening health risks. This is likely to have been the result even when the DLC levels were higher, if a toxic-fixated precaution would have lowered herring consumption. On the other hand, reflection and lack of proof should not be used as excuses for inactivity, e.g. with other poisons that are still increasing. Appropriate precaution can be found in measures that are easy to implement, multi-benefit, inherently low-risk and widely acceptable (see B.8, B.12).
14. Knowledge is insufficient for some decisions but not for all. The consequences of various strategies cannot be known in

detail. There is no one optimal solution, as there are many objectives and uncertain outcomes. However, efficient combinations of measures and processes of deliberation and action may be identified. The need for information on alternatives or improvements to the EU strategy in the Baltic case is in any case evident. The knowledge needed is increasingly multi-disciplinary and varied. Quantification has limitations but offers valuable insights. Multi-dimensional analyses accounting for qualitative aspects are essential complements. A key lesson is the need to suspect certainties.

15. Risk communication is needed between researchers and managers; different sectors; levels of government; stakeholder groups; and professionals and laypersons. Many people are confused and irrational about risks. Situations vary and communication is thus needed in many forms and languages. Ideally, those engaged should be open and clear about their arguments and valuations. Equally important is to communicate risks without inflating or deflating them. Shocking and one-sided warnings against and unconditional recommendations for consumption of Baltic fish cannot be justified.
16. Other Baltic Sea processes should be tied to dioxin risk management. Their links can constrain management or offer opportunities. Many Baltic Sea activities are still focused on other problems than dioxins. On the other hand, so much work on DLCs is underway that coordination is difficult. Co-management with other contaminants is justified. Links with eutrophication and oil spill abatement and health strategies are notable. Coordination and consolidation of risk management should take place horizontally between various sectors, actors and interest groups, as well as vertically between dioxin-focused and other actions. Integration of dioxin risk management with fisheries management is particularly important. Coordination is needed also between national and regional and EU and global levels.

These assessments can be summarized in the following main points:

- Dioxins and like compounds in Baltic fish cause multi-faceted risks that are partly poorly known but have generally decreased to a small part of the top levels in 1970's
- The greatest risks are due to developmental, reproductive and possibly immune disorders caused by dioxin-like furans and PCBs that accumulate in fatty fish
- It is not proven that these compounds have or have not caused human health impairment due to consumption of fish
- Grave effects have probably been caused in part by dlPCBs in some animals feeding mainly on Baltic fish, but have decreased since 1980's; more subtle effects may persist
- Health benefits from Baltic fish dioxins likely exceed several-fold their health risks
- Limitation of fish consumption based only on dioxin contents is poorly justified
- Risks can be managed in many ways; most efficient abatement is based on preventing the formation of dioxins and on many-sided dietary advice to high-risk groups
- Baltic fish dioxin risks emphasize the need to complement precaution with scientific evidence and many-sided deliberation, and the need for broad cooperation.

Bakgrund, målsättning och metodik

Denna rapport har producerats i Nordiska Ministerrådets projekt 'Risker och riskhanteringsstrategier av dioxiner i östersjöfisk – en tvärfacklig analys' med finansiering av Kemikaliegruppen, Hav- och luftgruppen och Fiskerigruppen. En forskargrupp i SYKE samt ett brett nätverk av experter har bidragit. Den övergripande målsättningen har varit att förbättra underlag för diskussioner och beslut om hantering av risker som förknippas med dioxiner och liknande föreningar i Östersjön och dess fisk, speciellt i samband med EU's reglering av dioxiner och PCB-ämnen t.ex. i livsmedel. Bedömningen bygger på litteraturstudier, andra dokumentanalyser samt analyser av sakkunnigas åsikter. En jämförande utvärdering har använts för att karakterisera risker och osäkerhet samt strategier. Bilagor rapporteras skilt.

A. Riskbedömning

1. Analysen av risker beaktar flera dimensioner. Förutom 2,3,7,8-klorodibenzo-p-dioxiner och -furaner (PCDD/F) behandlas även dioxinlika PCB-föreningar (dlPCB) och andra dioxinliknande föreningar (DLF) samt prekursorer som kan leda till uppkomst av dioxiner. Risker för både människan och andra arter beaktas. Flere olika slag av risker behandlas, inklusive indirekta risker och effekter. Förhållanden och processer både i Östersjön och närliggande områden behandlas.
2. Dioxinkällorna samt deras andel av utsläppen till Östersjön är fortfarande till stora delar okända eller osäkra särskilt beträffande diffusa och sekundära källor samt andra än PCDD/F-ämnen. Utsläppen till mark, sediment och avfall överträffar dock vida de direkta luftutsläppen. Viktiga källor omfattar klorerade biocider och herbicider, metall-, skogs- och vissa kemiska industrier, samt förbränning. Många organohalogener som produceras i stora mängder kan ge upphov till dioxiner. PCDD/F- och PCB-ämnen är huvudsakligen antropogena. Polen står för de största utsläppen av PCDD/F som TCDD-ekvivalenter (WHO-TEqDF) till Östersjön. Källorna av dlPCB omfattar PCB-produkter och –avfall samt förbränning. En del av belastningen på Östersjön härstammar från avlägsna områden utanför östersjöländer och EU.
3. Den nuvarande belastningen på Östersjön i nedfall från luften består av 400 g WHO-TEq_{DF} a^{-1} dioxiner och 10 % av detta i dlPCB (WHO-TEq_P). Därtill kommer belastning i avrinning och adsorption från gasfas. Belastningen cirkulerar i havet främst bunden till organiskt kol. Dioxinreserven i tillrinningsområdet når havet efter omlopp i fast substans. I havet sedimenteras största delen av dioxinerna, men på grunda bottnar kommer en stor del tillbaka i omlopp. Bottensediment utgör en central sekundär källa. Nedbrytning avlägsnar bara en del av belastningen och orsakar även toxiska produkter. PCDD/F och speciellt dlPCB bioackumuleras i näringskedjorna i fetter. Fisket avlägsnar 1 % av den externa belastningen av WHO-TEq_{DF} och av PCB-ämnen i litet större utsträckning. Eutrofieringen påverkar dessa ämnen på komplicerade sätt, inte enbart genom att späda ut dem i biomassan.
4. Östersjön är fortfarande förorenad av PCDD/F- och PCB-ämnen. Halterna är höga speciellt i gammal strömming, lax och andra feta fiskar. Halterna har generellt minskat till en bråkdel av toppnivåerna, även om minskningen har skett med variationer och fluktuationer. Idag sker minskningen i allt långsammare takt och i vissa fall har den redan avstannat. Östersjöns dioxin- och PCB-halter är generellt sett flerfalt högre än i de flesta kustvatten, men i vissa östersjödjur är halterna på samma nivå som i t.ex. Nordsjön.
5. Befolkningens intag av PCDD/F och PCB har minskat i östersjöländerna. Fördelningen är dock ojämn och överskrider t.ex. 2 pg WHO-TEq_{DFP} kg^{-1} d^{-1} för 12 % av vuxna svenskar. I Finland är andelen av fisk i det totala intaget av WHO-TEq högre (60-90

%) än i andra östersjöländer, men ändå är medelintaget hos finländare inte högre än på annat håll, tack vare att de andra livsmedlen är renare. Intaget är fortfarande högt i populationer som konsumerar mycket vild fet östersjöfisk. Barnens intag är dåligt känt. En del dioxiner från östersjöfisk sprids genom foder (se A.15). Halterna av PCDD/F och dlPCB i kroppen har minskat betydligt i östersjöländer men är fortfarande höga särskilt i storkonsumenter av fet östersjöfisk. Foster och ammade barn mottar proportionellt sett en större belastning. Nyckelkongenerer (se A.10) ackumulerar i högre grad än TCDD. Även andra dioxinliknande ämnen förekommer i människor.

6. De biologiska effekterna av dioxiner är mångdimensionella, påverkas av många faktorer och är välkända bara för TCDD. Effekterna förmedlas huvudsakligen via arylkolväte-receptoren AhR men även andra mekanismer är involverade speciellt med PCB-ämnen. Många effekter är liknande bland flere grupper av djur och uppstår vid låga doser, men stor variation förekommer i fråga om toxicitet även inom samma art. Effekterna på utveckling och fortplantning samt immunfunktioner är centrala. Hormonella och neurologiska effekter är likaså viktiga. TCDD orsakar tumörer av flere slag i många djur, uppenbarligen även i människan, men risken för cancer som följd av exponering för dioxiner är dock inte särskilt hög. Vissa icke-additiva effekter uppstår i blandningar av PCDD/F och dlPCB. Därtill har särskilt icke-dioxinlika PCB-ämnen konstaterats vara antagonister som minskar vissa effekter av dioxiner, men kan öka andra. Det finns inget entydigt sätt att beakta dessa. Effekterna i Östersjöns miljö påverkas av dess särdrag, bl.a. dess stora sårbarhet. Andra stressfaktorer ökar och komplicerar dioxineffekterna i Östersjön. Ytterligare faktorer kan bidra till minskade effekter.
7. Det finns inga entydiga bevis för skadliga hälsoeffekter på människan som en följd av exponering för dioxiner och dlPCB i östersjöfisk. De möjliga skadliga effekterna kan antas ha minskat från 70-talets exponeringstopp. Således är de värsta riskerna över. Effekter av exponeringstoppen kunde inte heller konstateras pga. bristande undersökningar. Vissa undersökningar har indikerat ett samband mellan konsumtion av (fet) östersjöfisk och olika störningar eller anomalier, t.ex. ökad andel flickor hos avkomman samt immunologiska förändringar, men rollen av dioxiner och dlPCB är oklar även i dessa fall. Nyttiga hälsoeffekter av fet havsfisk kan ha kompenserat för eventuella skador. Storkonsumenter av fet vild östersjöfisk är dock fortfarande utsatta för förhöjd risk.
8. I vissa andra djur såsom havsörnar, en del andra fiskätande sjöfågelarter samt särskilt sälar har tidigare observerade skador sannolikt orsakats av PCDD/F och speciellt av dlPCB i östersjöfisk. I en del av dessa fall kan dock andra miljögifter ha spelat en viktig roll. På basen av extrapolationer från andra djur och laboratorieförsök är exponeringsnivån fortfarande oroväckande hög för en del arter. De flesta utsatta bestånd har återhämtat sig, medan vissa fortfarande har en fortplantningskapacitet som är under det normala. Det är möjligt att dioxinliknande ämnen i östersjöfisk fortsättningsvis skadar vilda djur, men PCDD/F och dlPCB förorsakar inte bevisligen allvarliga ekotoxikologiska risker (se A.9). Effekterna på samfunds- och ekosystemnivå är svåra att definiera och påverkas av många faktorer.
9. Evaluering av riskerna beror främst på vilka kriterier effekter av PCDD/F och dlPCB och kausala samband förutsätts uppfylla. Flera traditionella kriterier leder till slutsatsen att inga signifikanta effekter av dioxiner kan bevisas i fritt levande populationer, särskilt pga. mångfalden av potentiellt verksamma faktorer. Biokemiska, övergående, högdosrelaterade och in vitro effekter kan ha begränsad generell betydelse. Att generalisera resultat från djurförsök till andra arter är en nyckelfråga som inte kan lösas enbart på vetenskapliga grunder. Riskevalueringen beror även på skadligheten och betydelsen av effekterna som inte heller kan definieras objektivt och entydigt.
10. Riskerna varierar i tid och rum, mellan arter och individer samt mellan effekter och ämnen. Halterna i vissa fiskar är högst i Bottenviken; i Finska Viken ökas riskerna i någon mån av PCDD/F-belastningen från Kymmene Älv. Lokala förorenade områden

kan sprida ut dioxiner och PCB. Tidsmässiga variationer omfattar oregelbundna förändringar som försvårar urskiljning av trender. Riskerna av dioxiner i Östersjön kommer att finnas kvar länge. De största riskerna av dioxintyp orsakas av 4-PeCDF och CB 126 men även andra PCB-ämnen samt eventuellt polyaromatiska ämnen är viktiga. Människan är inte nödvändigtvis mindre känslig, men många andra arter exponeras i högre grad för dioxiner i östersjöfisk. Exponeringsmönster och biologiska faktorer påverkar riskvariationer. Riskerna för unga är störst, men även andra åldersgrupper är utsatta för vissa effekter. Äldre havsfiskare, deras makor, deras 30 till 40 år gamla barn samt andra som konsumerat mycket av den mest förorenade fisken (på 70-talet) utgör särskilda riskgrupper.

11. Riskerna förorsakade av dioxiner i östersjöfisk kan till en viss grad jämföras med andra risker. En möjlig jämförelse är andra föroreningar i östersjöfisk och dioxiner i andra livsmedel. Epidemiologiskt sett relateras riskerna till bakgrundsrisken för ifrågavarande population och effekt, och omfattar alla orsakande faktorer. I jämförelser ter sig risken från dioxiner även i östersjöfisk ofta liten då andra större och säkrare riskfaktorer för skador finns, såsom biologiska och livsstilsfaktorer. Dioxin- och referensriskerna kan dock kvalitativt skilja sig betydligt från varandra t.ex. beträffande effektprofiler, tidsdimensioner och reducerbarhet. Detta kan betona eller minska betydelsen av dioxinrisker i förhållande till andra risker samt påverka risk-nyttoförhållanden. Mångdimensionella jämförelser kan ge vägledning då man söker effektiva lösningar.
12. Hälsoriskerna av dioxiner i östersjöfisk överträffas sannolikt av hälsonyttan av att konsumera sådan fisk och fiskolja. Nyttan är särskilt stor för kärl- och hjärthälsa bland högriskgrupper, samt möjligen för tidig utveckling. Trots de fortfarande relativt sett höga halterna av PCDD/F och dlPCB i östersjöfisk verkar nettorisken av sådan fisk vara mindre för hälsan än risken av vissa alternativa matvaror. Risk- och nyttoförhållanden är emellertid komplicerade och kan inte helt reduceras till enkla och endimensionella risk/nyttokvoter. Uppmärksamhet bör riktas på variationer och osäkerheter i risker och nyttor, bl.a. i olika åldersgrupper (nytto/risk-kvoten vid konsumtion av vild fet östersjöfisk är speciellt hög bland äldre personer med ökad risk för hjärtinfarkt, och lägre speciellt hos unga individer som är känsliga för utvecklingsstörningar av dioxiner och andra föroreningar), på kvalitativa skillnader mellan risker, och speciellt på vilka kostval och åtgärder som är möjliga (se B.7). Förutom hälsonyttor har fisken i Östersjön även socioekonomiska, kulturella och ekologiska värden.
13. Riktvärden för tolererbara dagliga intag (TDI) bygger på oklara grunder. De säkerhetsfaktorer som tillämpas kunde ökas om t.ex. individuell variation och skillnader i kongenerernas kinetik beaktades i högre grad. Det finns därför argument för TDI-riktvärden som skulle ligga även under den nuvarande lägsta nivån av 1 pg WHO-TEq kg^{-1} d^{-1}. Å andra sidan kunde många faktorer sänka säkerhetsfaktorer och höja TDI-värden (se A.8). Kvantitativa kriterier för ekotoxikologiska risker kan inte heller definieras entydigt. Dylika måttstockar beror på vilka principer och antaganden som tillämpas. Det kan således inte fastställas definitivt om dioxinriskerna överstiger säkra nivåer. Dessa osäkerheter om kritiska riskmått och även anknutna icke-vetenskapliga aspekter borde godkännas öppet och analyseras systematiskt.
14. Dioxiner i östersjöfisk orsakar psykologiska och möjligen psykosomatiska effekter som inte lätt kan skiljas från deras 'egentliga' biologiska effekter. Både överdrivna reaktioner och förnekande av risker sker. Reaktioner på riskhantering är likaså varierande och kan vara betydelsefulla. De påverkas av individuell känslighet och sociokulturella och situationsbundna faktorer. Dessa faktorer påverkar forskare och experter såväl som andra, och innebär utmaningar för riskstudier och –kommunikation (se B.15).
15. Indirekta risker orsakas av dioxiner i Östersjöns fisk eventuellt genom fiskbaserat foder i livsmedelsproduktionssystem. Dessa risker är svåra att bedöma bl.a. pga. otydliga transportvägar och följder av sådana dioxiner. Riskhantering i enlighet med EUs rekommendationer för fiskens

dioxinhalter (se B.2) har dock betydligt minskat de risker, och den nytta, som orsakas av sådant bruk av östersjöfisk (jfr. A.12, B.5). Indirekta risker förknippas även med andra socioekonomiska effekter av dioxiner och de återverkningar dessa har på ekosystem och samhällen. De indirekta riskerna av DLC i östersjöfisk medför inte enbart en ökning av risker utan gör dem mer komplexa.

16. Osäkerheter om risker består trots forskningen och övervakningen av PCDD/F och (i mindre grad) andra DLF. Osäkerheter orsakas av mätdata men oftast främst av modeller och beslutsregler, som leder till osäkerhet eller otydlighet av högre grad, och som även är av kvalitativ natur. Riskhantering omfattar osäkerheter både om tekniska möjligheter och samhälleliga aspekter. Täckning och precision av dioxinmätningar och –modeller borde avgöras av deras användningsändamål, och illusorisk precision undvikas. Osäkerheter bör inte överdrivas; de kan istället styra bedömning och hantering av risker (jfr. B.13).

B. Riskhantering

1. Många politiska linjedragningar, strategier och program är relevanta för hanteringen av risker förorsakade av dioxiner i Östersjön. De har varierande målsättning, omfattning och karaktär, och omfattar instrument som fokuseras på PCDD/F eller PCB, på Östersjön eller på fiske. Många av dem utvecklas kontinuerligt, t.ex. under POP-avtalet samt inom EU. Följderna för dioxiner i östersjöfisk kan inte ännu evalueras mera konkret (se nedan, t.ex. B.16).
2. EUs strategi för dioxiner, furaner och PCB (EC 2001) är allmänt upplagd och fokuserad på risker för människans hälsa, beaktar inte regionala frågor, samt innehåller litet om förhindrande av dioxiners uppkomst och emissionskontroll. Den kompletterande rekommendationen för dioxiner och PCB i foder och livsmedel (EC 2002a) har en ensidig inriktning på marknadsreglering, fokuserar fortfarande på PCDD/F och betraktar inte explicit hälsonyttor av olika livsmedel. Detta försök att hantera risker är således inte det mest adekvata för östersjöfisk eller för livsmedel och foder överhuvudtaget. Strategin kunde leda till dåligt fokuserad, inskränkt och ineffektiv reglering.
3. De tillåtna halterna av PCDD/F i livsmedel och foder är ännu oklarare än TDI-riktvärdena (se A.13). En omräkning av TDI-värden till tillåtna halter i fisk och fiskbaserat foder borde beakta i vilka mänger och hur fisk används i jämförelse med andra livsmedel och annat foder, samt variationer i användningen. De beräkningar som ligger bakom EUs rekommendationer för maximala PCDD/F-halter har inte presenterats explicit. Om dlPCBs räknas med höjs de synbara riskerna. Om å andra sidan hälsonyttan av olika livsmedel beaktas, kunde en viss konsumtion av fisk med halter över de maximigränser som kastats fram i EU vara berättigad. Ekotoxikologiska kvalitetskriterier för dioxiner i miljön har likaså oklara grunder. Generellt sett borde de tillåtna halterna evalueras mera ingående i förhållande till andra kriterier för livsmedel, miljö och hälsa.
4. Den huvudsakliga alternativa de facto strategin för hantering av risker från dioxiner för människans hälsa utgörs av kostråd och -upplysning. Även dessa har begränsningar och innebär problem som bör beaktas. Riskminskningen är osäker; t.ex. når eller påverkar råd inte alla. Å andra sidan kan en informationsbaserad linje ta hänsyn till regionala och andra specifika förhållanden och faktorer och beakta högriskgrupper och –situationer, och accepteras lättare. Den kan flexibelt kombineras med andra åtgärder, och lätt beakta nya evalueringar. Det är särskilt viktigt att risker och nyttor kan balanseras.
5. Både EUs riskhanteringslinje såväl som dess alternativ kan ha positiva och negativa följder i fallet östersjöfisk. Begränsningar av marknadsföring av östersjöfisk genom strikt tillämpning av EU's rekommendationer är problematiska. Det finns en stor risk att man förlorar nyttan med minskat dioxin- och PCB-intag genom att samtidigt minska hälsonyttorna av fiskkonsumtion (se A.12, B.3). Reducerat östersjöfiske drabbar sysselsättning speciellt i Sverige och Finland. Viktiga kulturella och systemfunktioner av fiskkonsumtion och fiske är svåra att återskapa om de förloras.

Reducerat fiske kan öka ekotoxikologiska risker genom att ändra på omloppet av dioxiner. EUs linje berättigas främst av en säkrare minskning av exponering. Hantering av komplexa risker med bara ett mål, att minimera humantoxicitet, och en typ av kriterier, halter i fisk, är dock snäv, potentiellt ineffektiv och skadlig, och kan få många oönskade konsekvenser.

6. En central principiell fråga är huruvida människor kan tillåtas välja vad de konsumerar, och huruvida interventioner i livsmedelsutbud behövs. Frågan är anknuten till subsidiaritet, flexibilitet och normativitet i regleringen. Vissa människor som t.ex. småbarn kan inte välja. Sådana människor är dock legitimt exponerade för andra skadliga ämnen och faktorer t.ex. i modersmjölk och även vissa marknadsförda livsmedel. Förverkligandet av frivillighetsprincipen förutsätter bl.a. uppgifter om fiskens ursprungsområde samt annan information och kostråd.
7. Då hälsonyttan av fet havsfisk sannolikt överträffar hälsoriskerna kunde konsumtion av sådan fisk även från Östersjön ökas. Detta kan vara omöjligt pga. konsumenternas dioxinrädslor. En nyckelfråga blir: kan risker förorsakade av dioxinhaltig fisk undvikas medan hälsonyttan av fiskkonsumtion upprätthålls, dvs. finns det goda och tillräckligt säkra alternativ till dioxinrik diet. Minskad konsumtion av fet vild östersjöfisk leder inte automatiskt till ohälsosammare dieter. Alternativen omfattar rena feta fiskar och fiskprodukter från Östersjön och annanstans ifrån, samt andra hälsosamma fetter. Deras potential kan ökas bl.a. med incentiv och kostråd. Även dessa har dock begränsningar och är osäkra, och minskad fiskkonsumtion är svår att återställa. De stora hälsonyttorna av fet östersjöfisk är således relativa och utgör inget ovedersägligt argument för konsumtion.
8. En önskvärd strategi för dioxiner och PCB-ämnen i Östersjön skulle i varje fall fokusera mera på att förebygga uppkomsten av dioxiner. Detta omfattar produkter och processer ända till grundorsaker inklusive konsumtions- och produktionsmönster och samhälleliga behov. Uppkomsten av PCDD/F kan minskas bl.a. med kemikaliereglering och andra instrument. Dessa kräver förbättrad implementering med avseende på dioxiner och PCBn. Dioxinuppkomst bör beaktas vid all produktion och behandling av halogenider. Många prekursorer används och produceras i stora mängder men prioriteras inte för kontroll. En preventiv linje kan effektivt och genomgående reducera risker. Risk-nyttoförhållanden bör dock analyseras ingående, då alla alternativ till prekursorer inte är riskfria.
9. Inom emissionskontrollen bör sådana källor prioriteras som effektivt, snabbt och långsiktigt kan reducera riskerna. Många lätta alternativ är uttömda men risker minskas dock även med åtgärder som redan befinner sig i implementeringsskedet. Möjligheterna för minskning av dioxinutsläpp omfattar bredare och effektivare tillämpning av befintliga tekniker och utveckling av nya t.ex. för metallindustrier och förbränning, inklusive behandling av restprodukter från föroreningskontroll. Det finns möjligheter till detta i synnerhet i Polen. Åtgärder kan vara önskvärda även av andra orsaker. Vid rengöring av dioxin- och PCB-förorenade områden behövs mångsidig bedömning. Rengöringen är allmänt sett mera kostnadseffektiv på land än i vatten. Rengöring i Kymmene Älv kunde avlägsna 4 kg WHO-TEq (ca. 10 % av mängden i havet) för 5-100 M€, beroende på tekniker och kostnader som inkluderas, men följderna t.ex. för halterna i fisk är osäkra.
10. Åtgärder i havsmiljön bör utnyttjas i högre grad vid utvecklandet av marinstrategier och fiskerier. I synnerhet utgör strömmings- och annat fiske en väsentlig del av dioxinproblemets lösning (jfr. B.5). Möjligheterna omfattar 'reningsfiske' med avsikt att avlägsna dioxiner, begränsning av dumpning av fiskrester i havet, selektivt fiske av unga årsklasser för kontroll av ackumulering, samt kvalitetskontroll av odlad östersjöfisk. Begränsning och styrning av muddring krävs då kustsediment utgör en viktig källa. Skyddsområden och -åtgärder utgör en möjlighet för vissa djur som hotas av dioxiner och PCBn.
11. Åtgärder för att minska riskerna vid utnyttjandet av fisk, förutom marknadsreglering eller kostråd, kan anknytas till andra delar av dioxinomlopp och -hantering. Styrning av förorenad fisk till mindre känsligt bruk samt säker

efterbehandling av sådan fisk behövs. Fiskolja kan renas från PCDD/F och vissa PCB-ämnen. Rena livsmedel kan ges åt flere arter. Kontroll av kinetik samt terapier är tillämpbara i vissa fall. Hälsonyttorna av fet havsfisk kan ökas bl.a. genom att utveckla dess användning för kärl- och hjärthälsa (se A.12). Riskhanteringen omfattar kompensation för förluster, som dock inte ersätter fiske och innebär problem (jfr. B.5). Ansvarsfrågor bör lösas med beaktande av bl.a. oavsiktlighet och retroaktivitet samt användbarheten av ekonomiska styrmedel.

12. Fördelar, nackdelar och följder av alternativa strategier kan ej definieras entydigt. Rekommendationer bör därför utvecklas genom ytterligare diskussioner, försök och bedömningar. Det kan dock konstateras att många faktorer talar för en liberal inställning till utbud av fet östersjöfisk, inom en mera helhetsmässig riskhantering. Positiva interventioner kan erbjuda alternativ till restriktioner. Viss styrning av fiskhandel kan införas utan en omfattande och strikt reglering av dioxinhalter, t.ex. inom delar av handel där risker och nyttor lätt kan balanseras. Uppföljningen utgör en viktig komponent (jfr. A.16). Sådan bredfrontstrategi kan minska de resterande dioxinriskerna i Östersjön medan bärkraftig användning av dess resurser säkras. En dylik strategi kan även bidra till utveckling av EU-strategier för dioxiner och PCB i livsmedel och foder. Det är viktigt att nya lösningar prövas. Många medel kan bli mera effektiva genom innovation. Kombinationer av olika åtgärder behövs i alla strategier. Åtgärder som minskar risker ger resultat på lång sikt och bör kompletteras med åtgärder som minskar exponeringen.
13. Dioxinriskhanteringen påverkas av hur försiktighetsprincipen tolkas. Balans behövs för att undvika både försenade åtgärder och panikreaktioner. Dioxiner utgör ett viktigt argument för principen men den borde inte tillämpas okritiskt och snärt. Överdriven kan den leda till slöseri och även större hälsorisker, t.ex. genom att fiskkonsumtionens hälsonyttor går förlorade. Detta skulle sannolikt ha varit följden även då halterna var högre, om en giftfixerad försiktighet hade lett till nedsatt strömmingskonsumtion. Riskerna av dioxiner och PCBn i östersjöfisk har redan minskat till en liten del tack vare de åtgärder som genomförts. Å andra sidan bör överläggning och avsaknad av bevis inte användas som ursäkt för passivitet med välgrundade fortsatta satsningar, även på andra gifter vars halter i fisk ännu stiger. Lämplig försiktighet bör sökas i åtgärder som är lättgenomförda, effektiva, mångsidigt nyttiga, ofarliga och brett acceptabla (se B.8, B.12).
14. Kunskapen är otillräcklig för vissa beslut men inte för alla. Konsekvenserna av olika strategier kan inte uppskattas i detalj i förväg. Det finns ingen entydig optimal lösning, då det finns många målsättningar och osäkra utgångar. Däremot finns det vettiga och effektiva kombinationer av åtgärder och processer för överläggning och utveckling. Behovet av kunskap om alternativ till eller förbättringar i EUs strategier är i varje fall uppenbart. Den kunskap som behövs är mångsidigare och i högre grad tvärfacklig. Kvantitativa bedömningar har begränsningar men ger viktiga insikter. Mångdimensionell risk-, nytto-, konsekvens- och beslutsanalys som beaktar även kvalitativa aspekter utgör ett väsentligt komplement till de kvantitativa bedömningarna.
15. Riskkommunikation behövs mellan forskare och beslutsfattare, olika sektorer, nivåer av administration, stater, intressegrupper samt fackmänniskor och lekmän. Många är förbryllade av och irrationella inför risker och osäkerheter. Kommunikations situationerna varierar även stort. Därför behövs kommunikation på många plan, sätt och språk. Idealet är att olika aktörer uttrycker öppet och klart sina argument och värderingar. Lika viktigt är att kommunicera utan att över- eller underskatta risker och osäkerheter även om saker måste förenklas. Information om halter i fisk förutsätter omsorg för att undvika missuppfattningar. Chockerade och ensidiga varningar eller ovillkorliga och generaliserande rekommendationer för konsumtion av östersjöfisk är inte berättigade.
16. Andra östersjöprocesser måste beaktas och anknytas till dioxinriskhantering. Synergier och konflikter mellan olika processer kan begränsa hanteringen

eller erbjuda möjligheter. Många östersjöprocesser är ännu fokuserade på andra problem än dioxiner. Å andra sidan sker det redan så mycket på dioxinfronten att koordineringen är svår. Sambanden mellan dioxiner och eutrofieringskontroll, fiske och oljebekämpning är viktiga. En del föroreningar kan åtgärdas samfällt. Eutrofiering kan inte anses vara positiv ens för dioxinrisker (se A.3). Koordinering och konsolidering av riskhantering bör ske horisontellt mellan olika sektorer, aktörer och intressegrupper samt vertikalt mellan dioxinfokuserade och andra, även allmänna instrument. Integrationen mellan dioxinhanteringen och regleringen av fisket är särskilt viktig. Koordinationsbehovet finns också mellan nationella och regionala nivåer och mellan EU- och globala processer.

Dessa bedömningar kan sammanfattas i följande huvudpunkter:

- Dioxiner och liknande ämnen i östersjöfisk orsakar många och delvis dåligt kända risker som dock har minskat till en liten del av toppnivåerna på 1970-talet
- De största riskerna är förknippade med utvecklings- och reproduktionsstörningar orsakade av dioxinlika furaner och PCB-föreningar
- Det finns inga entydiga belägg för eller mot att dessa ämnen skulle ha gett befolkningen hälsoskador på grund av fiskkonsumtion
- Grava skador har sannolikt orsakats delvis av dioxinlika PCBn i vissa djur vars föda huvudsakligen består av fisk; dessa skador har minskat redan på 1980-talet, men lindrigare skadliga effekter kan bestå
- Hälsonyttorna av östersjöfisk överträffar sannolikt flerfalt hälsoriskerna
- Tanken att begränsa fiskkonsumtion enbart på basen av dioxinhalter är dåligt underbyggd
- Risker kan hanteras på många sätt; de effektivaste åtgärderna bygger på att uppkomsten av dessa ämnen minskas samt på mångsidiga kostråd till riskgrupper
- Dioxinerna i Östersjön understryker behovet av att komplettera försiktighetsprincipen

Tausta, tavoitteet ja lähestymistapa

Raportti on laadittu Pohjoismaiden Ministerineuvoston hankkeessa Risker och riskehanteringstrategier av dioxiner i östersjöfisk – en tvärfacklig analys, rahoittajina Kemikaaliryhmä, Meri- ja ilmaryhmä ja Ympäristö- ja kalastusryhmä. Tutkijaryhmä SYKEssä ja kansainvälinen asiantuntijaverkosto on osallistunut työhön. Päätavoite on ollut arvioida ja parantaa tietopohjaa keskustelulle ja päätöksille koskien riskejä, joita aiheutuu dioksiineista ja niiden kaltaisista aineista Itämeren kalassa, erityisesti EUn dioksiini- ja PCB-strategian ja –säädösten yhteydessä. Arviointimenetemät ovat sisältäneet kirjallisuus- ja dokumenttitutkimuksia sekä asiantuntijanäkemysten analyysejä. Riskien ja strategioiden luonnehdintaa ja vertailevaa arviointia on painotettu. Erikseen raportoidaan liitteitä.

A. Riskinarviointi

1. Riskejä tarkastellaan useissa ulottuvuuksissa. 2,3,7,8-klorodibentso-p-dioksiinien ja –furaanien (PCDD/F) ohella tarkastellaan dioksiinien kaltaisia PCB-yhdisteitä (dlPCB), muita dioksiinien kaltaisia yhdisteitä ja niitä synnyttäviä prekursoreita. Sekä ihmiseen että muihin eläimiin kohdistuvia riskejä käsitellään. Erilaisia riskejä arvioidaan, ml. epäsuoria riskejä ja vaikutuksia. Itämeren ohella tarkastellaan Itämereen liittyviä muita alueita.
2. Dioksiinien lähteet ja niiden osuudet päästöissä Itämereen ovat yhä huonosti selvillä. Tämä koskee erityisesti hajapäästöjä, sekundaarisia lähteitä ja muita kuin PCDD/F-yhdisteitä. Silti on selvää, että päästöt maaperään, lietteisiin ja jätteisiin ylittävät moninkertaisesti suorat ilmapäästöt. Keskeiset päästölähteet sisältävät kloorattuja biosidejä ja herbisidejä, metalli-, metsä- ja eräitä muita teollisuuslaitoksia sekä polttoprosesseja. Monet suurina määrinä tuotetut organohalogenidit synnyttävät dioksiineja. PCDD/F- ja PCB-aineet ovat pääosin ihmisen synnyttämiä. Puolan PCDD/F-kokonaispäästö Itämereen TCDD-toksisuusekvivalentteina (WHO-TEqDF) arvioidaan suurimmaksi. DlPCB-lähteet sisältävät PCB-tuotteiden varannot ja jätteet sekä polton. Pieni osa kuormituksesta tulee kaukaa Itämeren rantavaltioiden ja jopa EUn ulkopuolelta.
3. Kuormitus ilmalaskeumana Itämereen on noin 400 g WHO-TEq$_{DF}$ PCDD/F-yhdisteistä ja 10 % siitä PCB-yhdisteistä (WHO-TEqP) vuodessa. Lisäksi tulee kuormitusta valunnassa ja adsorptiossa kaasufaasista. Valuma-alueelle kertynyt varanto pääsee mereen viipeiden jälkeen. Meressä kuormitus sedimentoituu, mutta matalikoilta suuri osa pääsee takaisin kiertoon. Pohjaliete muodostaa siten keskeisen sekundaarisen dioksiinilähteen. Hajoaminen poistaa vain osan kuormituksesta ja synnyttää myös toksisia aineita. PCDD/F ja erityisesti dlPCB bioakkumuloituvat ravintoketjuissa rasvohin. Kalastus poistaa n. 1 % ulkoisesta WHO-TEq$_{DF}$ –kuormasta ja vähän enemmän WHO-TEq$_{P}$ -kuormasta. Rehevöityminen vaikuttaa monin tavoin eikä vain laimenna dioksiineja biomassaan.
4. Itämeren PCDD/F- ja PCB-yhdisteiden pitoisuudet ovat yhä korkeita varsinkin vanhassa silakassa, lohessa ja muissa rasvaisissa kaloissa. Pitoisuudet ovat laskeneet murto-osaan huipputasoista, joskin vaihdellen. Jatkossa lasku on hitaampaa ja on eräissä tapauksissa pysähtynyt. Dioksiineja itämerikalasta leviää myös siitä valmistetun rehun kautta (vrt. A.15). Itämeren PCDD/F- ja PCB-tasot ovat yleisesti ottaen moninkertaisesti korkeampia kuin muissa rannikkomerissä, mutta ovat eräissä eläinlajeissa Pohjanmeren tasolla tai lähellä sitä. Dioksiinitoksisuudesta (TEq) vain pieni johtuu arvioinnin pohjana olevasta TCDDsta tai muista varsinaisista dioksiineista, suurin osa furaaneista ja dlPCB-aineista.
5. Väestön PCDD/F- ja PCB-yhdisteiden saanti on vähentynyt itämerimaissa. Saanti jakautuu kuitenkin epätasaisesti; esim. 12 % aikuisista ruotsalaisista ylittää tason 2 pg WHO-TEq$_{DFP}$ kg^{-1} d^{-1}. Suomessa kalan osuus näiden aineiden kokonaissaannista, jopa 90

%, on korkeampi kuin muissa itämerimaissa, mutta silti keskimääräinen saanti ei ole korkeampi johtuen puhtaammasta muusta ravinnosta. Saanti on yhä korkea ihmisillä jotka kuluttavat paljon rasvaista Itämeren luonnonkalaa. Saanti on pienempi nuorten aikuisten keskuudessa, ja huonosti tunnettu erityisesti lapsissa. Sikiöt ja rintaruokitut lapset saavat suhteellisesti suuren kuormituksen. Pitoisuudet kudoksissa ovat vähentyneet keskimäärin väestössä. Avainkongeneerit (ks. A.10) kertyvät elimistöön voimakkaammin kuin TCDD. Myös muita dioksiinien kaltaisia aineita on havaittu väestössä.

6. Dioksiinien biologiset vaikutukset ovat monitahoisia ja niitä muovaavat monet tekijät. Vain TCDDn vaikutukset tunnetaan hyvin. Vaikutukset välittyvät pääosin AhR-reseptorin kautta, mutta varsinkin PCB-aineilla on myös muita mekanismeja. Monet vaikutukset ovat samanlaisia eri eläimissä ja aiheutuvat pienillä annoksilla, mutta vaihtelu annos-vastefunktioissa on suurta saman lajinkin sisällä. Vaikutukset kehitykseen, lisääntymiseen ja immuunitoimintoihin ovat keskeisiä. Myös eräät hormonaaliset ja neurologiset vaikutukset ovat tärkeitä. TCDD aiheuttaa kasvaimia useissa eläimissä, ilmeisesti myös ihmisessä, mutta syöpäriski altistuksesta dioksiineille ei kuitenkaan ole erityisen suuri. PCDD/F- ja dlPCB-seoksissa aiheutuu eräitä ei-additiivisia yhteisvaikutuksia. Varsinkin muut PCB-aineet voivat vähentää dioksiinien eräitä vaikutuksia, mutta lisätä toisia. Ei ole yksiselitteistä tapaa ottaa näitä tekijöitä huomioon. Vaikutukset Itämeressä riippuvat sen piirteistä. Muut stressitekijät lisäävät ja sekoittavat dioksiinien vaikutuksia. Niitä vähentävät toisaalta adaptaatio- ja kompensaatiomekanismit.
7. Ei ole selvää näyttöä siitä, että altistus PCDD/F- ja dlPCB-yhdisteille itämerikalassa aiheuttaisi haitallisia vaikutuksia ihmisen terveyteen. Mahdollisten haittavaikutusten voidaan olettaa vähentyneen 70-luvun altistushuipusta. Pahimmat riskit ovat siten ohi. Vaikutuksia huippualtistuksestakaan ei ole alueella todettu, johtuen osin tutkimusten puutteista. Eräissä tutkimuksissa on havaittu yhteyksiä rasvaisen itämerikalan kulutuksen ja erilaisten häiriöiden tai anomalioiden kuten jälkeläisten sukupuolijakauman ja immunologisten muutosten välillä, mutta PCDD/F- ja dlPCB-yhdisteiden rooli on epäselvä. Tällaisen kalan terveyshyödyt (ks. A.12) ovat saattaneet kompensoida haittoja. Itämeren rasvaisen luonnonkalan suurkuluttajat altistuvat silti yhä kohonneille riskeille.
8. Eräille muille eläinlajeille kuten merikotkille, eräille muille vesilinnuille ja varsinkin hylkeille on aiemmin todennäköisesti aiheutunut haittoja PCDD/F- ja varsinkin dlPCB-yhdisteistä itämerikalassa. Eräät haitoista ovat olleet vakavia. Joissakin tapauksissa muut ympäristömyrkyt ovat tosin voineet olla tärkeitä syitä. Perustuen ekstrapolointiin muista lajeista ja laboratoriokokeista altistus on yhä huolestuttava eräissä lajeissa. Useimmat dioksiinien uhkaamat kannat ovat toipuneet, mutta eräiden lisääntymiskyky on yhä alentunut. Dioksiinien kaltaiset aineet itämerikalassa saattavat yhä vahingoittaa siitä riippuvaisia villieläimiä. PCDD/F- ja dlPCB-yhdisteet eivät kuitenkaan enää todistettavasti aiheuta vakavia ekotoksikologisia riskejä (ks. A.9). Elioyhteisö- ja ekosysteemivaikutuksia on vaikea määritellä, sillä ne riippuvat monista tekijöistä.
9. Riskien arviointi riippuu olennaisesti siitä, millaisia kriteerejä asetetaan todisteille nimenomaan PCDD/F- ja dlPCB-aineiden aiheuttamista vaikutuksista. Monilla kriteereillä ei voida todistaa merkitseviä vaikutuksia dioksiinien kaltaisista aineista populaatiotasolla, johtuen erityisesti useista samanaikaisesti vaikuttavista tekijöistä. Biokemialliset, ohimenevät, soluviljelmissä havaittavat sekkä korkeilla annoksilla ja kerta-annoksilla aiheutuvat vaikutukset eivät ehkä ole yleisemmin relevantteja. Eläinkoetulosten yleistäminen muihin lajeihin on avainkysymys, joka ei ole ratkaistavissa vain tieteellisillä perusteilla. Riskien arviointi riippuu myös vaikutusten haitallisuudesta ja merkityksestä, joita ei voida määritellä objektiivisesti ja yksiselitteisesti.
10. Riskit vaihtelevat ajallisesti, alueellisesti, lajien ja yksilöiden välillä sekä aineiden ja vaikutusten välillä. PCDD/F-pitoisuudet eräissä kaloissa ovat erityisen korkeita Pohjanlahdessa; Suomenlahdessa riskejä lisää Kymijoen kuormitus. Paikalliset saastuneet alueet voivat levittää dioksiineja. Epäsäännölliset ajalliset vaihtelut vaikeuttavat

trendien toteamista. Dioksiiniriskit tulevat jatkumaan pitkään Itämeressä. 4-PeCDF ja PCB 126 aiheuttavat usein suurimmat dioksiinityyppiset riskit, mutta muutkin PCB-yhdisteet ja mahdolisesti polyaromaattiset aineet ovat tärkeitä. Ihminen ei ehkä ole vähemmän herkkä, mutta monet muut lajit altistuvat enemmän Itämeren dioksiineille. Biologiset tekijät vaikuttavat riskivaihteluihin. Nuoriin kohdistuvat riskit ovat yleisesti ottaen suurimmat. Vanhemmat merikalastajat, heidän puolisonsa, heidän 30-40-vuotiaat lapsensa ja muut jotka ovat syöneet paljon saastuneinta kalaa (70-luvulla) ovat erityisiä riskiryhmiä.

11. Itämerikalan dioksiinien riskejä voidaan vertailla joihinkin muihin riskeihin, esim. kalan muihin myrkyllisiin aineisiin ja dioksiineihin muussa ravinnossa. Epidemiologisia riskejä suhteutetaan ko. vaikutuksen taustariskiin ko. populaatiossa, jolloin tarkastellaan kaikkia vaikuttavia tekijöitä. Näissä vertailuissa itämerikalan dioksiinien aiheuttamat riskit tuntuvat usein pieniltä, sillä suurempia ja varmempia riskitekijöitä, esim. biologisia ja elintapoihin liittyviä, on helppo löytää terveyshaitoille. Dioksiiniriskit ja vertailuriskit voivat kuitenkin poiketa toisistaan merkittävästi esim. vaikutustyyppien, aikaulottuvuuden ja hallintamahdollisuuksien osalta. Nämä erot voivat korostaa tai vähentää dioksiiniriskejä suhteessa muihin riskeihin sekä vaikuttaa riski-hyötysuhteisiin. Moniulotteiset riskivertailut voivat auttaa löytämään tehokkaita ratkaisuja.
12. Itämerikalan dioksiinien aiheuttamien terveysriskien vastapainoksi niistä todennäköisesti saadaan suurempia terveyshyötyjä. Rasvaisen merikalan kulutus on erityisen hyödyllistä sydän- ja verisuoniterveydelle riskiryhmissä, ja mahdollisesti varhaiskehitykselle. Itämerikalan nettoriski terveydelle voidaankin arvioida pienemmäksi kuin eräiden vaihtoehtoisten ravintoaineiden, huolimatta tämän kalan korkeammista dioksiini- ja PCB-pitoisuuksista. Riskit ja hyödyt myös ihmisen terveydelle ovat kuitenkin monitahoisia, ja niitä ei voi täysin pelkistää yksinkertaisiin ja yksiulotteisiin riski/hyötysuhteisiin. On kiinnitettävä huomiota myös riskien ja hyötyjen vaihteluun ja epävarmuuteen mm. eri ikäryhmissä (hyöty/riski-suhde villin rasvaisen itämerikalan kulutuksessa on erityisen suuri vanhemmilla henkilöillä joiden sydäninfarktiriski on korkea, ja pienempi erityisesti nuorilla jotka ovat herkkiä dioksiinien ja muiden saasteiden aiheuttamille kehityshäiriöille), laadullisiin eroihin riskeissä, ja erityisesti käytettävissä oleviin ravintovaihtoehtoihin ja toimiin (ks. B.7).
13. Perusteet siedettävien päivittäisten saantimäärien (TDI) ohjearvoille ovat osin epäselvät. Tässä käytettäviä turvakertoimia voitaisiin suurentaa jos esim. yksilöllinen vaihtelu ja kongeneerien kinetiikan erot otettaisiin pidemmälle huomioon. Voidaankin perustella saantiohjearvoja jotka olisivat esim. alle tason 1 pg WHO-TEq_{DFP} kg^{-1} d^{-1}. Monet tekijät voivat toisaalta pienentää turvakertoimia ja nostaa ohjearvoja (ks. A.8). Kriteerejä ekotoksikologisille riskeille ei myöskään voida määritellä yksiselitteisesti. Sellaiset mittapuut riippuvat käytettävistä periaatteista ja oletuksista. Siksi ei voida ehdottomasti todeta, ylittävätkö dioksiiniriskit turvallisen tason. Riskimittareiden epävarmuus ja ei-tieteelliset aspektit tulee tunnustaa avoimesti ja analysoida systemaattisesti.
14. Dioksiinit itämerikalassa aiheuttavat huomattavia psykologisia ja mahdollisesti psykosomaattisia vaikutuksia, joita ei voi helposti ja kategorisesti erottaa biologisista terveysvaikutuksista. Psykologisiin vaikutuksiin sisältyy sekä riskien liioiteltuja pelkoja että riskien kieltämistä. Reaktiot riskienhallintatoimiin ovat vaihtelevia ja voivat olla merkittäviä. Niihin vaikuttavat yksilölliset piirteet kuten herkkyys sekä sosiokulttuuriset ja tilanteesta riippuvat tekijät. Ne vaikuttavat tutkijoihin ja asiantuntijoihin kuin muihinkin ihmisiin, ja antavat haasteita tutkimukselle ja viestinnälle riskeistä (ks. B.15).
15. Itämerikalan dioksiineista aiheutuu epäsuoria riskejä mm. kun niitä joutuu rehuun ja ravinnontuotantoketjuihin. Näitä riskejä on vaikea arvioida mm. koska dioksiinien kulkureitit ja vaikutuskohteet ovat epäselviä. Kuitenkin EUn suositusten mukaisten kalan laatukriteerien johdosta (ks. B.2) nämä riskit, ja samalla kalan käyttöön liittyvät hyödyt, ovat vähentyneet (vrt. A.12, B.3). Epäsuoria riskejä liittyy myös dioksiinien muihin sosioekonomisiin vaikutuksiin ja näiden seurauksiin ekosysteemeissä ja yh-

teiskunnissa. Epäsuorat riskit ja vaikutukset eivät siten pelkästään lisää itämerikalan dioksiinien riskejä vaan tekevät ne monitahoisemmiksi ja monimutkaisemmiksi.

16. Epävarmuus riskeistä jatkuu huolimatta PCDD/F-yhdisteiden (ja vähemmässä määrin muiden niiden kaltaisten yhdisteiden) tutkimuksesta ja seurannasta. Epävarmuutta aiheutuu havaintoaineistoista mutta yleensä vielä enemmän malleista ja päätössäännöistä, ml. laadullista ja korkeamman tason epävarmuutta. Riskienhallinta sisältää epävarmuustekijöitä sekä teknisistä toimista että yhteiskunnallisista kysymyksistä (ks. B). Mittausten ja mallien kattavuus ja tarkkuus tulisi paremmin ratkaista tiedon käyttötarkoituksen mukaan, välttäen näennäistä tarkkuutta. Epävarmuutta ei toisaalta tule liioitella; se voi päinvastoin ohjata riskien arviointia ja hallintaa (ks. B.13).

B. Riskienhallinta

1. Monet strategiat ja ohjelmat ovat relevantteja Itämeren ja sen kalojen dioksiinien ja niiden kaltaisten yhdisteiden hallinnassa. Näillä strategioilla ja ohjelmilla on vaihtelevat tavoitteet, laajuus, soveltamisala ja luonne. Ne sisältävät instrumentteja jotka kohdentuvat PCDD/F- tai PCB-aineisiin, Itämereen tai kalastukseen. Monet strategiat, esim. globaalin POP-sopimuksen yhteydessä ja EUssa, ovat yhä kehittyviä. Niiden merkitystä dioksiineille itämerikalassa ei voida vielä arvioida kovin konkreettisesti (vrt. mm. B.18).
2. EUn strategia dioksiineille, furaaneille ja PCB-yhdisteille (EC 2001) on yleisluonteinen, keskittyy ihmisen terveyteen, ei ota huomioon alueellisia erityiskysymyksiä, ja ei sisällä juuri mitään dioksiinien ehkäisemisestä ja varsin vähän myös päästöjen vähentämisestä. Strategiaa täydentävät suositukset dioksiineista ja PCB-aineista ravinnossa ja rehussa (EC 2002a) keskittyvät yksipuolisesti markkinasäätelyyn, rajoittuvat yhä PCDD/F-aineisiin, ja eivät ota eksplisiittisesti huomioon elintarvikkeiden erilaisia terveyshyötyjä. Suositusten seurauksia ei ole tarkasteltu aiemmin perusteellisesti. Nämä riskienhallintayritykset eivät siksi ehkä ole tarkoituksenmukaisimmat itämerikalalle tai yleisemmin ravinnolle ja rehulle. Ne voisivat johtaa rajoittuneeseen, huonosti fokusoituun ja tehottomaan säätelyyn.
3. Sallitut PCDD/F-pitoisuudet elintarvikkeissa ja rehussa ovat keskeisiä riskienhallintakriteerejä, mutta vielä epäselvempiä kuin saannin ohjearvot (ks. A.13). Saantiohjearvojen muuntamisessa sallituiksi pitoisuuksiksi kalassa ja kalatuotteissa tulisi ottaa huomioon miten paljon ja miten kalaa käytetään verrattuna muihin elintarvikkeisiin ja rehuihin, sekä käytön vaihtelu. Tähän liittyviä laskelmia ei ole esitetty pohjaksi EUn suosituksille dioksiinien pitoisuusrajoista. Jos dlPCB-aineet lasketaan mukaan, lisääntyvät näennäiset riskit suuresti. Toisaalta jos terveyshyödyt eri elintarvikkeista otettaisiin huomioon, voisi olla turvallista syödä kalaa jonka pitoisuudet ylittävät EUn esittämät pitoisuusrajat. Ekotoksikologiset laatukriteerit dioksiinipitoisuuksille ympäristössä ovat vastaavasti epäselviä. Yleisesti ottaen tulisi sallittavia dioksiini- ja dlPCB-pitoisuuksia arvioida lähemmin, yhteydessä muihin elintarvike-, terveys- ja ympäristökriteereihin.
4. Pääasiallinen vaihtoehto EUn suosituksille elintarvikkeiden dioksiinien ja PCB-yhdisteiden terveysriskien hallitsemiseksi muodostuu käytännössä ravintosuosituksista ja valistuksesta. Myös tällä strategialla on rajoituksia ja ongelmia. Sillä saavutettava riskienvähennys on epävarma; esim. neuvot eivät ulotu tai vaikuta kaikkiin. Toisaalta tieto-ohjaukseen perustuva linja voi hyvin ottaa huomioon alueellisia ja muita spesifisiä tekijöitä sekä erityisiä riskiryhmiä ja –tilanteita. Se voidaan yhdistää joustavasti muihin riskienhallinnan strategioihin ja alueisiin sekä muuttuviin arvioihin. Erityisen tärkeää on, että elintarvikkeiden riskejä ja hyötyjä voidaan helposti tasapainottaa.
5. Sekä EUn suosituksilla että niiden vaihtoehdoilla on hyviä ja huonoja seurauksia ja puolia itämerikalan tapauksessa. On suuri vaara, että vähentyvän dioksiinisaannin aiheuttama terveyshyöty mitätöityy, jos samalla vähennetään rasvaisen kalan syönnin suurempia terveyshyötyjä (ks. A.12, B.3). Vähenevä silakan ja kilohailin kalastus aiheuttaa ongelmia työllisyydelle erityisesti Ruotsissa ja Suomessa. Silakan kalastuksen ja kulutuksen tärkeitä kulttuurisia ja systeemifunktioita on vaikea luoda uudestaan jos ne menetetään. Vähenevä ja muuttuva

kalastus voi lisätä ekotoksikologisia riskejä muuttamalla dioksiinien kiertoa. EUn suosituksilla on hyviä puolia lähinnä kuluttajien altistuksen varman vähentämisen kautta. Moniulotteisten riskien hallinta yhdellä tavoitteella (ihmisen toksikologisten riskien minimointi) ja kriteerillä (kalan pitoisuudet) on silti kapeakatseista, potentiaalisesti tehotonta ja voi aiheuttaa monia haittoja.

6. Keskeinen periaatteellinen kysymys on, missä määrin ihmisten voidaan antaa itse valita mitä he syövät, ja missä määrin tarvitaan kalan tarjonnan säätelyä määräyksillä. Se liittyy säätelyn subsidiariteettiin, joustavuuteen ja normatiivisuuteen. Jotkut ihmiset eivät voi itse valita. Heitä kuitenkin altistetaan legitiimisti dioksiineille ja muille riskeille myös ravinnossa, esim. äidinmaidossa ja markkinoiduissa elintarvikkeissa. Vapaaehtoisen valinnan toteutuminen tehokkaasti voi edellyttää mm. kalan pyyntialueen merkitsemistä.
7. Avainkysymys on: Voidaanko itämerikalan dioksiiniriskejä ihmisen terveydelle välttää samalla kun kalan terveyshyödyt varmistetaan, so. onko hyviä vaihtoehtoja dioksiinipitoiselle kalalle. Itämeren silakan vähentynyt kulutus ei johda aina ja automaattisesti paljon epäterveellisempään dieettiin. Huomionarvoisia vaihtoehtoja ovat dioksiiniköyhä kala ja kalatuotteet Itämerestä ja muualta sekä muut terveelliset rasvat. Vaihtoehtojen toteutumista voidaan edistää mm. tukitoimilla ja ravintoneuvoilla, mutta niissä on rajoituksia ja epävarmuustekijöitä (vrt. B.5). Itämeren kalan terveyshyödyt ovat siten suhteellisia eivätkä muodosta kaikkivoipaa argumenttia tämän kalan kulutukselle.
8. Toivottava strategia Itämeren dioksiinien ja PCB-aineiden riskien hallitsemiseksi painottaisi nykyistä enemmän dioksiinien muodostumisen estämistä. Se sisältää tuotteet ja prosessit perussyihin kuten yhteiskunnan tarpeisiin asti. Ehkäisevä riskienhallinta voi vähentää riskejä kattavasti ja tehokkaasti. Dioksiinien ehkäisyä voidaan tehostaa mm. kemikaalisäädösten ja muiden instrumenttien avulla paremmalla implementoinnilla. Dioksiinien synty on otettava huomioon kaikessa halogenidien tuotannossa ja käsittelyssä. Monia prekursoreita käytetään ja tuotetaan suuria määriä myös EUssa ja rantavaltioissa, mutta ei priorisoida riskienvähennyksessä. Riski-hyötysuhteita on analysoitava perusteellisesti, sillä kaikki vaihtoehdot prekursoreille eivät ole riskittömiä.
9. Päästöjen rajoittamisessa on syytä priorisoida lähteitä, joissa voidaan tehokkaasti vähentää riskejä Itämerelle. Monet helpot mahdollisuudet on jo hyödynnetty, mutta valmistellut toimet tulevat yhä vähentämään riskejä. Keinot dioksiinipäästöjen vähentämiseen sisältävät olemassa olevien tekniikoiden laajemman ja tehokkaamman soveltamisen sekä uusien tekniikoiden kehittämisen mm. metalliteollisuudessa ja poltossa, ml. päästöjen puhdistusjätteiden käsittelyn. Mahdollisuuksia päästöjen vähentämiseen tehokkaasti on erityisesti Puolassa. Toimet voivat olla perusteltuja myös muista syistä kuin dioksiinien takia. Dioksiinien ja PCB-aineiden saastuttamien rajattujen alueiden kunnostuksessa tarvitaan monipuolista harkintaa. Kunnostus on yleensä tehokkaampaa maalla kuin vesistöissä. Kymijoen kunnostus voi poistaa 4 kg WHO-TEq_{DFP} (n. 10 % meren varannosta) 5-100 M€:lla, riippuen tekniikoista ja sisällytettävistä kustannuksista, mutta vaikutukset mm. kalan pitoisuuksiin ovat epävarmoja.
10. Mereen kohdistuvia toimia tulee hyödyntää enemmän, yhteydessä meristrategioiden kehittämiseen. Erityisesti silakan ja kilohailin kalastus muodostaa olennaisen osan dioksiiniongelmien ratkaisua (vrt. B.5). Mahdollisuudet käsittävät puhdistuskalastuksen dioksiinien poistamiseksi merestä, perkuujätteiden dumppauksen rajoittamisen, valikoivan kalastuksen painottaen nuoria ikäluokkia kalan dioksiinitason vähentämiseksi sekä viljellyn itämerikalan laatukontrollin. Ruoppauksen rajoittamista ja ohjaamista korostaa matalien sedimenttien merkitys dioksiinilähteenä. Suojelualueet ja –toimet antavat mahdollisuuksia vähentää riskejä eräille dioksiinien ja PCB-aineiden uhkaamille eläimille.
11. Riskienvähennystoimia kalan merestä poistamisen jälkeen voidaan yhdistää dioksiinien kierron ja käsittelyn muihin vaiheisiin ja osa-alueisiin. Erityisen saastunut kala ohjataan vähemmän vaaralliseen käyttöön ja käsittelyyn. Kalaöljyä voidaan puhdistaa PCDD/F-aineista ja eräistä PCB-aineis-

ta sekä rehukäyttöön että ihmisravinnoksi (vrt. B.7). Puhtaampaa korvaavaa ravintoa voidaan antaa useille lajeille (ks. A.12). Vaikuttamista dioksiinien käyttäytymiseen kudoksissa ja terapeuttisia toimia voidaan käyttää rajoitetusti. Rasvaisen merikalan terveyshyötyjä voidaan lisätä mm. kehittämällä tällaisen kalan käyttöä sydän- ja verisuontitautien ehkäisyssä. Riskienhallinta käsittää myös taloudellisen kompensaation riskeistä ja haitoista. Tämä ei kuitenkaan korvaa kalastusta, ja sisältää ongelmia. Vastuukysymyksiä ratkaistaessa on otettava huomioon mm. rajoitettu tuottamuksellisuus ja retroaktiivisuus sekä muiden taloudellisten ohjauskeinojen käyttö.

12. Vaihtoehtoisten strategioiden etuja, haittapuolia ja seurauksia ei voida määritellä yksiselitteisesti. Suosituksia on siksi kehitettävä jatkuvan keskustelun, kokeilun ja arvioinnin kautta. Monet tekijät puoltavat liberaalia suhtautumista Itämeren rasvaisen luonnonkalan tarjontaan markkinoille, osana kokonaisvaltaisempaa riskienhallintaa. Tiettyä markkinaohjausta voidaan harjoittaa ilman kattavaa dioksiinipitoisuuksien säätelyä, esim. silloin kun riski-hyötysuhteita voidaan helposti optimoida normiohjauksella. Tarkkailu on tärkeä osa riskienhallintaa (vrt. A.16). Tällainen monen rintaman strategia voi parhaiten välttää itämerikalaan yhä sisältyvät riskit samalla kun sen kestävä käyttö varmistetaan. Se voi myös edistää laajemmin integroivaa EUn strategiaa dioksiineille elintarvikkeissa ja rehussa. Monet hallintakeinot voivat keittyä innovaatioiden kautta. Eri keinojen yhdistämistä tarvitaan kaikissa strategioissa. Dioksiinien synnyn ehkäisy vaikuttaa vasta viipeiden jälkeen, ja sitä on myös tästä syystä järkevää täydentää altistuksen vähentämisellä nopeavaikutteisin toimin.

13. Dioksiinien riskienhallinta riippuu olennaisesti varovaisuusperiaatteen tulkinnoista. On vältettävä sekä myöhästyneet toimet että paniikkireaktiot. Dioksiiniriskit muodostavat tärkeän syyn varovaisuusperiaatteelle, mutta sitä ei tule soveltaa epäkriittisesti ja yksipuolisesti. Liioiteltuna se voi johtaa tuhlaukseen ja jopa suurempiin riskeihin, kuten kalan terveyshyötyjen menetykseen. Näin olisi todennäköisesti käynyt jopa pitoisuuksien ollessa korkeampia, jos myrkkykeskeinen varovaisuus olisi vähentänyt silakan syöntiä. Itämerikalan dioksiiniriskit ovat vähentyneet jo pieneen osaan toteutetuilla toimilla. Toisaalta harkintaa ja todisteiden puutetta ei tule käyttää verukkeena lisätoimien viivyttämiselle, myös muiden myrkkyjen vähentämiseksi jotka yhä lisääntyvät. Sopivaa varovaisuutta tulee etsiä toimista jotka ovat helposti toteutettavissa, tehokkaita, monipuolisesti hyödyllisiä, vaarattomia ja laajasti hyväksyttäviä (vrt. B.8, B.12).

14. Tieto on riittämätöntä joillekin päätöksille ja toimille muttei kaikille. EUn suositusten ja vaihtoehtojen seurauksia ei voida arvioida kattavasti ja tarkkaan etukäteen (ks. B.12). Yhtä varmasti optimaalista ratkaisua ei ole olemassa, sillä on monia mahdollisia tavoitteita ja epävarmoja kehityskulkuja. On kuitenkin järkeviä toimien yhdistelmiä ja neuvottelu- ja toimintaprosesseja. Tiedon tarve EUn riskienhallinnan kehittämisedellytyksistä ja vaihtoehdoista tai parannuksista on joka tapauksessa ilmeinen. Tarvittava tieto on nykyistä monitieteisempää ja monimuotoisempaa. Riskien kvantifioinnilla on suuria rajoituksia mutta se myös valaisee asioita olennaisesti. Sen olennainen täydennys on moniulotteinen riski-, hyöty-, vaikutus- ja päätösanalyysi jossa otetaan huomioon myös laadullisia aspekteja. Perusopetus on epäillä varmoja arvioita ja johtopäätöksiä, omiakin.

15. Riskiviestintää tarvitaan tutkijoiden ja päätöksentekijöiden, toimintasektorien, hallinnon tasojen, valtioiden ja intressiryhmien välillä. Monet ihmiset ovat hämmentyneitä riskeistä ja epävarmuudesta, ja epärationaalisia suhtautumisessaan niihin. Viestinnän tilanteet ja tavoitteet myös vaihtelevat suuresti. Siksi tarvitaan monen tasoista ja monimuotoista viestintää. Ihanteena on tuoda esiin avoimesti ja selkeästi argumentit ja myös niiden taustalla olevat arvot. Yhtä tärkeää on viestintä riskeistä yli- ja aliarvioimatta niitä ja niihin liittyvää epävarmuutta, vaikka asioita joudutaan yksinkertaistamaan (vrt. ed.). Informaatio kalan dioksiinipitoisuuksista vaatii huolellisuutta. Yksipuoliset varoitukset itämerikalan käytöstä tai sen varauksettomat suositukset eivät ole perusteltuja.

16. Muut prosessit on otettava huomioon Itämeren dioksiiniriskien hallinnassa. Prosessien synergia ja konfliktit antavat rajoi-

tuksia ja mahdollisuuksia. Monet Itämerta koskevat prosessit ovat keskittyneet muihin ongelmiin kuin dioksiineihin. Toisaalta dioksiinikentässä tapahtuu jo niin paljon, että koordinointi on vaikeaa. Yhteydet rehevöitymisen hillintään ja öljyntorjuntaan ovat tärkeitä. Rehevöitymistä ei voida pitää positiivisena dioksiiniriskienkään kannalta (ks. A.3). Dioksiineja ja monia muita aineita voidaan hallita yhdennetysti. Riskienhallinnan koordinointia ja tiivistämistä tarvitaan monilla alueilla sekä horisontaalisesti eri sektorien ja toimijoiden välillä että vertikaalisesti dioksiineihin kohdistuvien ja muiden, yleisempienkin instrumenttien välillä. Parempi integrointi kalastuksen säätelyn kanssa on erityisen tärkeää. Koordinaatiota tarvitaan myös kansallisen ja alueellisen sekä EU- ja globaalin tason kesken.

Nämä arvioinnit voidaan kiteyttää seuraaviin pääkohtiin:

- Dioksiinit ja niiden kaltaiset aineet itämerikalassa aiheuttavat monia osin huonosti tunnettuja riskejä, jotka ovat jo vähentyneet pieneen osaan 1970-luvun huipputasoista
- Suurimmat riskit liittyvät kehitys- ja lisääntymishäiriöihin, joita aiheutuu eräistä rasvaiseen kalaan kertyvistä dioksiinien kaltaisista furaaneista ja PCB-aineista
- Ei ole selvää näyttöä, että nämä aineet ovat tai eivät ole aiheuttaneet terveyshaittoja väestössä itämerikalan syönnin johdosta
- Dioksiinien kaltaiset PCB-aineet ovat todennäköisesti osaltaan aiheuttaneet vakavia haittoja eräissä eläimissä jotka syövät pääasiassa itämerikalaa; nämä haitat ovat väistyneet jo 1980-luvulla, mutta lievempiä haittoja saattaa yhä aiheutua
- Terveyshyödyt itämerikalasta ylittävät todennäköisesti suuresti terveysriskit
- Kalankulutuksen rajoittaminen vain pitoisuuksien nojalla on huonosti perusteltua
- Riskejä voidaan hallita ja on jo vähennetty monin tavoin; tehokkaimmat toimet perustuvat dioksiinien synnyn ehkäisemiseen ja monipuoliseen ravintoneuvontaan riskiryhmille
- Itämeren dioksiinit korostavat tarvetta täydentää varovaisuusperiaatetta tieteellisillä todisteilla, monipuolisella harkinnalla sekä laajentaa eri alojen yhteistyötä.

References

References to Chapter 1

Ahlborg UG et al. 1989. Nordisk dioxinriskbedömning. København, Nordiska ministerrådet. Miljørapport 1989:7

Ahlborg UG et al. 1992a. Risk assessment of polychlorinated biphenyls (PCBs). Copenhagen, Nordic Council Ministers and Stockholm, Nordic Council 1992b

Ahlborg UG et al. 1992b. Impact of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls on human and environmental health, with special emphasis on application of the toxic equivalency factor concept. Eur J Pharmacol: Environ Toxicol Pharmacol. 228;4:179

Ahlborg UG et al. 1994. Toxic equivalency factors for dioxin-like PCBs. Report on a WHO-ECEH and IPCS consultation, Dec 1993. Chemosphere 28:1049

ATSDR. 1998. Toxicological profile for polychlorinated biphenyls. Agency Toxic Substances Disease Registry, U.S. Dept Health Human Services, Atlanta, Georgia, Dec 1998. www.atsdr.cdc.gov/

Boenke A. 2001. Contribution of European research to risk analysis. Food Addit Contam. 18;12:1135

Chen G, Bunce NJ. 2004. Interaction between halogenated aromatic compounds in the Ah receptor signal transduction pathway. Environ Toxicol. 19;5:480

Christensen FM et al. 2003. Risk terminology—a platform for common understanding and better communication. J Haz Mater. 103;3:181-203

COT. 2001. COT statement on the tolerable daily intake for dioxins and dioxin-like polychlorinated biphenyls. Committee On Toxicity, London. COT/2001/07

CPON. Copenhagen Post Online News 13.8.1999

Darnerud PO et al. 2003. Swedish consumption of fatty Baltic Sea fish in relation to the total dioxin intake and the recommended TDI. Organohalogen Compds. 62:183

Denison MS et al. 2002. Ligand binding and activation of the Ah receptor. Chemico-Biol Interact. 141:3

DN. Dagens Nyheter 28.11.2001

Dudewicz EJ, Mishra SN. 1988. Modern mathematical statistics. John Wiley & Sons, New York et al. 1988

Dybing E et al. 2002. Hazard characterisation of chemicals in food and diet: dose-response, mechanisms, and extrapolation issues. Food Chem Toxicol. 40:237

EC. 2001. Community strategy for dioxins, furans and polychlorinated biphenyls. Communication from the Commission to the Council, the European Parliament and the Economic and Social Committee. COM(2001)593, Final. Off J EC 17.11.2001, C322/2-18. www.europa.eu.int

EC. 2002a. Commission recommendation of 4 March 2002 on the reduction of the presence of dioxins, furans and PCBs in feedingstuffs and foodstuffs. Notified under number C(2002a) 836. Off J EC 9.3.2002 L 67/69 (2002/201/EC)

EC. 2003a. Technical guidance document in support of commission directive 93/67/EEC on risk assessment for new notified substances and commission regulation (EC) No 1488/94 on risk assessment for existing substances. 2nd ed. JRC, Ispra. 4 vol. EUR 20418 EN/1-IV. www.ecb.jrc.it/

EC. 2004a. Synthesis of baseline reports in the framework of the European Environment and Health Strategy (COM(2003)338 final). European Commission, Brussels, Jan 2004.

ECETOC. 1993. Environmental hazard assessment of substances. Eur Centre Ecotoxicol Toxicol Chem, Brussels 1993. ECETOC Tech Report 51.

FAO & WHO. 1997b. Food consumption and exposure assessment of chemicals. Report of FAO/WHO consultation (Report No. WHO/FSF/FOS/97.5). World Health Organization, Geneva 1997

FAO & WHO. 2002b. Plan for the project to update the principles and methods for the assessment of chemicals in food. FAO, WHO, Rome and Geneva 9.4.2002. www.who.int/ipcs/food/principles/en/

Finkel AM. 1993. Confronting uncertainty in risk management. A guide for decision-makers. Resources For the Future, Washington, DC 1993. RFF Report. 68 p.

Guston DH et al. 2000. Report, Workshop on boundary organizations in environmental policy and science, 9-10 Dec 1999, Bloustein Sch Planning Public Pol, Rutgers Univ, New Brunswick, NJ. Belfer Center Sci Int Affairs (BCSIA) Disc Paper 2000-32

Harremoës P. 2003. Ethical aspects of scientific incertitude in environmental analysis and decision making. J Cleaner Product. 11;7:705

HCN. 1996a. Dioxins. Advisory report no. 1996/10E. Rijswijk 6.8.1996, Health Council of The Netherlands

HCN. 1996b. Risk is more than just a number: reflections on the development of the environmental risk management approach. The Hague, Health Council of the Netherlands 1996. Gesondheidsraad E 1996

Higginbotham GR et al. 1968. Chemical and toxicological evaluations of isolated and synthetic chloro derivatives of dibenzo-p-dioxin. Nature 220;168:702

Hildén M. 1997b. Risk, uncertainty, indeterminacy and ignorance in fisheries management – an analysis of management advice. Finn Environ Inst, Helsinki 1997. Monographs Boreal Environ Res. 5

Horlick-Jones T. 1998. Meaning and contextualisation in risk assessment. Rel Eng System Saf. 59;1:79

IARC. 1987. Polychlorinated biphenyls (Group 2A). IARC, Lyon. Summaries & evaluations. Suppl 7:322

IARC. 1997. Polychlorinated dibenzo-para-dioxins. 2,3,7,8-Tetrachlorodibenzo-para-dioxin (Group 1); Polychlorinated dibenzo-para-dioxins (other than 2,3,7,8-tetrachlorodibenzodioxin): 2,7-DCDD, 1,2,3,7,8-PeCDD, 1,2,3,6,7,8-/1,2,3,7,8,9-HxCDD, 1,2,3,4,6,7,8-HpCDD (Group 3); Dibenzo-para-dioxin (Group 3). IARC, Lyon. Summaries & evaluations. Vol 69:33

ICES. 2005a. Report of the Study Group on Management Strategies. ICES Headquarters, 31 Jan–4 Feb 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:09

ICES. 2005b. Report of the Baltic Salmon and Trout Working Group (WGBAST). Helsinki, 5-14 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:18

ICES. 2005c. Report of the Baltic Fisheries Assessment Working Group (WGBFAS). Hamburg, 12-21 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:19
IOM. 2003. Dioxins and dioxin-like compounds in the food supply: Strategies to decrease exposure. Committee on Implications of Dioxin in the Food Supply, Food Nutr Bd, Inst Med of the Natl Academies. Natl Acad Press, Washington, DC
IPCS. 1989. Polychlorinated dibenzo-p-dioxins and dibenzofurans. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 88
IPCS. 1992. Polychlorinated biphenyls and terphenyls. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 140
IPCS. 1994a. Polybrominated biphenyls. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 152
IPCS. 1998. Polybrominated dibenzo-p-dioxins and dibenzofurans. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 205
IPCS. 2001a. Clorinated naphthalenes. Int Programme for Chemical Safety. Concise Int Chem Assess Doc 34. 57 p.
IPCS. 2001d. Integrated risk assessment. Report prepared for the WHO/UNEP/ILO Int Programme on Chemical Safety. WHO/IPCS/IRA/01/12. WHO, Geneva, Dec 2001. www.who.int/ipcs/publications/new_issues/ira/en/
IPCS & OECD. 2003. Descriptions of selected key generic terms used in chemical hazard/risk assessment. Int Progr Chem Saf Joint Project with OECD Harmonis Hazard/Risk Assess Terminol. Geneva, IPCS 2003
Iwata H et al. 1993. Distribution of persistent organochlorines in the oceanic air and surface seawater and the role of ocean on their global transport and fate. Environ Sci Technol. 27:1080
JECFA. 2001. Summary and conclusions, Fifty-seventh meeting Rome, 5-14 Jun 2001. Joint FAO/WHO Expert Committee on Food Additives
Jones EL, Krizek H. 1962. A technic for testing acegenic potency in rabbits, applied to the potent acnegen, 2,3,7,8-tetrachlorodibenzo-p-dioxin. J Invest Dermatol. 39:511
Kimmig J, Schultz KH. 1957. Occupational acne (so called chloracne) caused by chlorinated aromatic cyclic ethers. Dermatologica 115:540-6
Kiviranta H et al. 2003. PCDD/Fs and PCBs in Baltic herring during the 1990s. Chemosphere 50;9:1201
NATO/CCMS. 1988a. International toxicity equivalency factor (I-TEF) method of risk assessment for complex mixtures of dioxins and related compounds. NATO Committee Challenges of Modern Society. Report No. 176
Neubert D. 1997-98. Reflections on the assessment of the toxicity of “dioxins” for humans, using data from experimental and epidemiological studies. Teratogen Carcinogen Mutagen. 17;4-5:157
NFAF. 2004. Uudistetut kalan syöntisuositukset EU-kalat tutkimushankkeen seurauksena. Information from Natl Food Agency Finland 28.04.2004. www.elintarvikevirasto.fi/
Notermans S, Mead GC. 1996. Incorporation of elements of quantitative risk analysis in the HACCP system. Int J Food Microbiol. 30;1-2:157
NRC. 1983. Risk assessment in the federal government: Managing the process. Natl Res Council of the Natl Academies. Natl Acad Press, Washington, DC
SCAN. 2000. Opinion of the Scientific Committee on Animal Nutrition on the dioxin contamination of feedingstuffs and their contribution to the contamination of food of animal origin. EC, Brussels. Adopted 06 Nov 2000
SCF. 2000. Opinion of the Scientific Committee for Food on the risk assessment of dioxins and dioxin-like PCBs an food. EC, Brussels. Adopted Nov 2000. europa.eu.int/comm./food/Fs/sc/scf/out78_en.pdf
SCF. 2001. Opinion of the Scientific Committee for Food on the risk assessment of dioxins and dioxin-like PCBs an food. Update based on new scientific information available. Adopted 30th May 2001. europa.eu.int/comm./food/Fs/sc/scf/out90_en.pdf
SCOOP. 2000. Reports in tasks for scientific cooperation. Report of experts participating in task 3.2.5, Brussels 7.6.2000. DG-SANCO. europa.eu.int/comm/dgs/health_consumer/ library/pub/pub08_en.pdf
Selin H, Eckley N. 2003. Science, politics and persistent organic pollutants. The role of scientific assessments in international environmental co-operation. Int Environ Agreements: Politics Law Econ. 3:17
Skolimowski H. 1966. The structure of thinking in technology. Technol Culture 7;3:371
SNFA. 2002. Fisk och vår hälsa. Swed Natl Food Agency, Uppsala. www.slv.se/ (last updated 2002-05-29)
SNFA. 2005. Mat för två – kostråd för gravida. Swed Natl Food Agency, Uppsala. www.slv.se/ (last updated 2005-05-17)
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
Star SL, Griesemer JR. 1989. Institutional ecology, ‘translations,’ and boundary objects: Amateurs and professionals in Berkeley’s Museum of Vertebrate Zoology, 1907-1939. Soc Studies Sci. 19:387
Sweeney MH, Mocarelli P. Human health effects after exposure to 2,3,7,8-TCDD. Food Addit Contam. 17;4(2000):303
TWGIM. 2004a. Baseline report on “Integrated monitoring of dioxins & PCBs in the Baltic Region” in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 09 Jan 2004
TWGIM. 2004b. Final report on actions and rcommendation for “Integrated monitoring of dioxins & PCBs in the Baltic Region” in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 23 Feb 2004
USEPA. 1994. Health assessment document for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Vol. III: External review draft. OHEA, ORD, USEPA. US Gov Printing Office, Washington, DC. EPA/600/BP-92/001C
USEPA. 1997b. Guidance for cumulative risk assessment: Part 1, Planning and Scoping. USEPA, Washington, DC. www.epa.gov/swerosps/bf/htmldoc/cumrisk2.htm
USEPA. 2000a. Exposure and human health reassessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Draft final. USEPA, Washington, DC. EPA/600/6-88/005Ca. www.epa.gov/ncea/dei.html
USEPA. 2002a. Lessons learned on planning and scoping for environmental risk assessments. Planning Scoping Workgroup of Sci Pol Council Steering Committee. USEPA, Washington, DC
USPCCRARM. 1997. Framework for environmental health risk management. Final report, vol 1. US Presidential/Congressional Commission Risk Assess Risk Manage, Washington, DC. Internet Edition www.riskworld.com
Van den Berg M et al. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ Health Perspect. 106:775
Van Leeuwen FRX, Younes M. 2000. Assessment of the health risk of dioxins: re-evaluation of the tolerable daily intake (TDI). Food Addit Contamin. 17;4
Van Tongelen B. 2002. Community strategy for dioxins, furans and PCBs. Organohalogen Compds. 57:265

Verstraete F. 2002. Development and implementation of an EC strategy on dioxins, furans and dioxin-like PCBs in food and feed. Environ Sci Pollut Res Int. 9;5:297
Verta M et al. 2003. Continued transport of PCDD/F contaminated sediments from River Kymijoki to the Gulf of Finland, the Baltic Sea. Organohalogen Compds. 61:405-8. CD-ROM, Vol. 2, Section 3
Vulykh N, Shatalov V. 2001. Investigation of dioxin/furan composition in emissions and in environmental media. Selection of congeners for modeling. MSC-E Tech Note 6/2001. MSC-E, Moscow. www.msceast.org/publications
Wenborn M et al. 1999. Releases of dioxins and furans to land and water in Europe. Final Report Issue 2. Report for LUW Nordrhein-Westfalen, Germany on behalf of EC DG-Environ, Sep 1999. Report AEAT-4703. http://europa.eu.int/comm/environment/dioxin/download.htm
Wenger E. 2000. Communities of practice and social learning systems. Organization 7;2:225
WHO. Consultation on Tolerable Daily Intake from food of PCDDs and PCDFs. WHO Reg Off Eur 1991. EUR/UCP/PCS 030.
Wynne B. 2002. Science and social responsibility. Ansell J, Wharton F (eds.) Risk analysis and management. John Wiley & Sons, Chichester 2002, p. 137-52

References to Chapter 2

Addink R, Olie K. 1995. Mechanisms of formation and destruction of polychlorinated dibenzo-p-dioxins and dibenzofurans in heterogeneous systems. Environ Sci Technol. 29;6:1425
Agrell C et al. 2001. Atmospheric and river input of PCBs,DDTs and HCHs to the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Ecol Studies 148:149-76. Springer Verlag, Berlin & Heidelberg
Ahlborg UG et al. 1989. Nordisk dioxinriskbedömning. København, Nordiska ministerrådet. Miljørapport 1989:7
Amin S, et al. 2000. Gestational and lactational exposure to TCDD or coplanar PCBs alters adult expression of saccharin preference behavior in female rats. Neurotoxicol Teratol. 22;5:675
Aneer G. 1987. High natural mortality of Baltic herring (Clupea harengus) eggs caused by algal exudates. Mar Biol. 94:163
Assmuth T. 2003. Indicative analysis of relative risks and data gaps of dioxin-like compounds in Baltic Sea fish, based on body burden, biokinetic and bioactivity ratios. Organohalogen Compds. 62:37
Assmuth T, Vartiainen T. 1994. Concentration patterns of 2,3,7,8-chlorinated dibenzo-p-dioxins and dibenzofurans at landfills and disposal sites for chlorophenolic wood preservative wastes. Chemosphere 28;5:971
Axelman J et al. 2001. Dynamics and distribution of hydrophobic organic compounds in the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Springer Verlag, Berlin & Heidelberg. Ecol Studies 148:257
Barabas N et al. 2004. Modified polytopic vector analysis to identify and quantify a dioxin dechlorination signature in sediments. 1. Theory. Environ Sci Technol. 38;6:1813
Barkovskii AL, Adriaens P. 1996. Microbial dechlorination of historically present and freshly spiked chlorinated dioxins and diversity of dioxin-dechlorinating populations. Appl Environ Microbiol. 62;12:4556
Barkovskii AL, Adriaens P. 1998. Impact of humic constituents on microbial dechlorination of polychlorinated dioxins. Environ Toxicol Chem. 17:1013
Behnisch PA et al. 2003. Brominated dioxin-like compounds: in vitro assessment in comparison to classical dioxin-like compounds and other polyaromatic compounds. Environ Int. 29;6:861
Berggren P et al. 1999. Patterns and levels of organochlorines (DDTs, PCBs, non-ortho PCBs and PCDD/Fs) in male harbour porpoises (Phocoena phocoena) from the Baltic Sea, the Kattegat-Skagerrak Seas and the West Coast of Norway. Mar Pollut Bull. 38;12:1070
Bergqvist P-A et al. 2005. Kartläggning av utsläppskällor för oavsiktligt bildade ämnen: PCDD/F, PCB och HCB. Miljökemi, Kem Inst, Umeå Univ, Mar 2005. MK 2005:01
Bergström S et al. 2001. Climate and hydrology of the Baltic basin. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Ecol Studies 148:75
Bignert A et al. 1998. Temporal trends of organochlorines in Northern Europe, 1967 1995. Relation to global fractionation, leakage from sediments and international measures. Environ Pollut. 99;2:177
Birnbaum LS et al. 1987a. Teratogenicity of three polychlorinated dibenzofurans in C57BL/6N mice. Toxicol Appl Pharmacol. 90;2:206
Birnbaum LS et al. 1991. Teratogenic effects of 2,3,7,8-tetrabromodibenzo-p-dioxin and three polybrominated dibenzofurans in C57BL/6N mice. Toxicol Appl Pharmacol. 107;1:141
Birnbaum LS. 1994b. The mechanism of dioxin toxicity: Relationship to risk assessment. Environ Health Perspect. 102;Suppl 9:157
Birnbaum LS et al. 2003. Health effects of polybrominated dibenzo-p-dioxins (PBDDs) and dibenzofurans (PBDFs). Environ Int. 29;6:855
Brewster DW et al. 1988a. Toxicity and disposition of 2,3,4,7,8-pentachlorodibenzofuran (4PeCDF) in the rhesus monkey (Macaca mulatta). Toxicol Appl Pharmacol. 93;2:231
Brouwer A et al. 1995. Functional aspects of developmental toxicity of polyhalogenated aromatic hydrocarbons in experimental animals and human infants. Eur J Pharmacol. 293;1:1
Campfens J, Mackay D. 1997. Fugacity-based model of PCB bioaccumulation in complex aquatic food webs. Environ Sci Technol. 31;2:577
Chen G et al. 2001. Synthesis of polybrominated diphenyl ethers and their capacity to induce CYP1A by the Ah receptor mediated pathway. Environ Sci Technol. 35;18:3749
Chen J et al. 2002. Quantitative predictive models for octanol-air partition coefficients of persistent organic pollutants at different environmental temperatures. Organohalogen Compds. 57:467
Chen J et al. 2003. Quantitative relationships between molecular structures, environmental temperatures and octanol–air partition coefficients of polychlorinated biphenyls. Comput Biol Chem. 27;3:405
Christiansen C et al. 2002. Material transport from the nearshore to the basinal environment in the Southern Baltic Sea: I. Processes and mass estimates. J Mar Syst. 35;3-4:133
Connor K et al. 2004. Estimating the total TEq in human blood from naturally-occurring vs. anthropogenic dioxins: A dietary study. Organohalogen Compds. 66:3408
deBruyn AMH, Gobas FAPC. 2004. Modelling the diagenetic fate of persistent organic pollutants in organically enriched sediments. Ecol Modell. 179;3:405
Denison MS, Nagy SR. 2003. Activation of the aryl hydrocarbon receptor by structurally diverse exogenous and endogenous chemicals. Annu Rev Pharmacol Toxicol. 43:309

Easton MDL et al. 2002. Preliminary examination of contaminant loadings in farmed salmon, wild salmon and commercial salmon feed. Chemosphere 46;7:1053
Eduljee GH, Dyke P. 1996. An updated inventory of potential PCDD and PCDF emission sources in the UK. Sci Total Environ. 177;1-3:303
Emeis K et al. 2002. Material transport from the near shore to the basinal environment in the southern Baltic Sea. II: Synthesis of data on origin and properties. J Mar Syst. 35;3-4:151
Eriksson P, Fredriksson A. 1998. Neurotoxic effects in adult mice neonatally exposed to 3,3',4,4',5-pentachlorinated biphenyl or 2,3,3',4,4'-pentachlorinated biphenyl. Changes in brain nicotinic receptors and behaviour. Environ Toxicol Pharmacol. 5:17
Falandysz J et al. 1994d. Most toxic and highly bioaccumulative PCB congeners in cod-liver oil of Baltic origin processed in Poland during the 1970s and 1980s, their TEq-values and possible intake. Sci Total Environ. 145;3:207
Falandysz J et al. 1994e. Congener-specific data on polychlorinated biphenyls in tissues of common porpoise from Puck Bay, Baltic Sea. Arch Environ Contam Toxicol 26;3:267
Finkel AM. 1993. Confronting uncertainty in risk management. A guide for decision-makers. Resources For the Future, Washington, DC
Fishbein L. 1990. Health-risk estimates for 2,3,7,-tetrachlorodibenzodioxin: An overview. Adv Modern Toxicol. 17:25-68
Fiskeriverket. 2001. Småskaligt kustfiske och insjöfiske – en analys. Sammanfattning. Swed Natl Fisheries Agency. www.fiskeriverket.se/
Floderus S, Pihl L. 1990. Resuspension in the Kattegat: impact of variation of wind climate and fishery. Estuar Coastal Shelf Sci. 31;4:487
Fox GA. 2001. Wildlife as sentinels of human health effects in the Great Lakes--St. Lawrence basin. Environ Health Perspect. 109 Suppl 6:853
Gasiewicz TA. 1997. Dioxins and the Ah receptor: Probes to uncover processes in neuroendocrine development. Neurotoxicol. 18;2:393
Gilek M et al. 1996. Enhanced accumulation of PCB congeners by Baltic Sea blue mussels, Mytilus edulis, with increased algae enrichment. Environ Toxicol Chem. 15;9:1597
Glynn AW et al. 2001. Polychlorinated biphenyl congeners as markers of toxic equivalents of polychlorinated biphenyls, dibenzo-p-dioxins and dibenzofurans in breast milk. Environ Res. 86;3:217.
Gunnarsson JS, Rosenberg R. 1996. Eutrophication increases the association of PCB to dissolved organic matter in marine microcosms. Mar Poll Bull. 33:100
Gutleb AC et al. 2000. Effects of oral exposure to polychlorinated biphenyls (PCBs) on the development and metamophosis of two amphibian species (Xenopus laevis and Rana temporaria). Sci Total Environ. 262;1-2:147
Hagenmaier H et al. 1994. Correlation of environmental occurrence of polychlorinated dibenzo-p-dioxins and dibenzofurans with possible sources. Chemosphere 29:2163
Hamm JT et al. 2003. A mixture of dioxins, furans, and non-ortho PCBs based upon consensus toxic equivalency factors produces dioxin-like reproductive effects. Toxicol Sci. 74;1:182
Hankinson O. 2005. Role of coactivators in transcriptional activation by the aryl hydrocarbon receptor. Arch Biochem Biophys. 433;2:379
Hansen E, Hansen CL. 2003. Substance flow analysis for dioxin 2002. Min Environ, Danish Environ Protect Agency, Copenhagen 2003. Report by Cowi A/S. Environmental project nr. 811. ISBN – electronic 87-7972-675-5. www.mst.dk
Hansson MC et al. 2004. Unprecedented genomic diversity of AhR1 and AhR2 genes in Atlantic salmon (Salmo salar L.). Aquat Toxicol. 68:219-32
Harper N et al. 1993. Immunotoxic potencies of polychlorinated biphenyl (PCB), dibenzofuran (PCDF) and dibenzo-p-dioxin (PCDD) congeners in C57BL/6 and DBA/2 mice. Toxicol. 80;2-3:217
Hassoun EA et al. 2002. Induction of oxidative stress in the tissues of rats after chronic exposure to TCDD, 2,3,4,7,8-pentachlorodibenzofuran, and 3,3',4,4',5-pentachlorobiphenyl. J Toxicol Environ Health A 65;12:825
HELCOM. 2002a. Fourth periodic assessment of the state of the environment of the Baltic Marine Area, 1994-1998. Baltic Sea Environ Proc. 82B:1
Hites RA et al. 2004a. Global assessment of organic contaminants in farmed salmon. Science 303;5655:226
Hoekstra EJ et al. 1999. Natural formation of chlorinated phenols, dibenzo-p-dioxins and dibenzofurans in soil of a Douglas fir forest. Environ Sci Technol. 33:2543
Hoffman DJ et al. 1998. Comparative developmental toxicity of planar PCB congeners in chickens, American kestrels and common terns. Environ Toxicol Chem. 17:747
Holene E et al. 1998. Behavioural hyperactivity in rats following postnatal exposure to sub-toxic doses of polychlorinated biphenyl congeners 153 and 126. Behav Brain Res. 94;1:213
Holsapple MP et al. 1986. Direct suppression of antibody responses by chlorinated dibenzodioxins in cultured spleen cells from (C57BL/6 C3H)F1 and DBA/2 mice. Immunopharmacol. 12;3:175
Håkansson H et al. 1991. In vivo and in vitro toxicity of fractionated fish lipids, with particular regard to their content of chlorinated organic compounds. Pharmacol Toxicol. 69;6:459
Håkansson H et al. 1994. Effect on tissue vitamin A levels in the rat following subchronic exposure to four individual PCB congeners (IUPAC 77, 118, 126, and 153). Chemosphere 29;9-11:2309
ICES. 2003a. Report of the ICES Advisory Committee on Ecocystems, 2003. ICES Cooperat Res Report 262:1
ICES. 2003c. Report of the Baltic Salmon and Trout Working Group. Karlskrona, Sweden 2-11 Mar 2003. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen 2003. ICES CM2003/ACFM:20.
ICES. 2003d. Report of the Study group on multispecies assessment in the Baltic. Charlottenlund, Denmark 2-4 Apr 2003. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen 2003. ICES CM2003/H03 (Ref. by WGBFAS, ICES 2005c)
ICES. 2003e. Report of the Working Group for Fisheries Assessment. ICES Headquarters, Copenhagen, 3-16 Apr 2003. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen 2003. ICES CM2003/ACFM:21
ICES. 2005a. Report of the Study Group on Management Strategies. ICES Headquarters, 31 Jan–4 Feb 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:09
ICES. 2005b. Report of the Baltic Salmon and Trout Working Group (WGBAST). Helsinki, 5-14 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:18
ICES. 2005 c. Report of the Baltic Fisheries Assessment Working Group (WGBFAS). Hamburg, 12-21 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:19. Ref. H. www.ices.dk
IOM. 2003. Dioxins and dioxin-like compounds in the food supply: Strategies to decrease exposure. Committee on Implications of Dioxin in the Food Supply, Food Nutr Bd, Inst Med of the Natl Academies. Natl Acad Press, Washington, DC
IPCS. 1998. Polybrominated dibenzo-p-dioxins and dibenzofurans. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 205

Isosaari P et al. 2002b. Feeding trial on rainbow trout: comparison of dry fish feed and Baltic herring as a source of PCDD/Fs and PCBs. Chemosphere 48;8:795
Jackson JB et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293;5530:629
Jacobs M et al. 2002a. Investigation of polychlorinated dibenzo-p-dioxins, dibenzo-p-furans and selected coplanar biphenyls in Scottish farmed Atlantic salmon (Salmo salar). Chemosphere 47;2:183
Jacobs MN et al. 2002b. Investigation of selected persistent organic pollutants in farmed Atlantic salmon (Salmo salar), salmon aquaculture feed, and fish oil components of the feed. Environ Sci Technol. 36;13:2797
Jansson B et al. 1975. Identification by GC-MS of phenolic metabolites of PCB and p,p'-DDE isolated from Baltic guillemot and seal. Ambio 4;2:93
Johnson KL et al. 1997. Promotion of endometriosis in mice by polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls. Environ Health Perspect. 105;7:750
Johnson MD, Weber WJ Jr. 2001. Rapid prediction of long-term rates of contaminant desorption from soils and sediments. Environ Sci Technol. 35;2:427
Jokinen MP et al. 2003. Increase in cardiovascular pathology in female sprague-dawley rats following chronic treatment with 2,3,7,8-tetrachlorodibenzop- dioxin and 3,3',4,4',5-pentachlorobiphenyl. Cardiovasc Toxicol. 3;4:299
Jonsson A et al. 2003a. Variations in the Baltic Sea wave fields. Ocean Eng. 31;1:107
Jonsson P. 2000. Sediment burial of PCBs in the offshore Baltic Sea. Ambio 29:260
Jung RE, Walker MK. 1997. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on development of anuran amphibians. Environ Toxicol Chem. 16;2:230
Karlsson L et al. 1999. The diet of salmon (Salmo salar) in the Baltic Sea and connections with the M74 syndrome. Ambio 28;1:37
Kerkvliet NI et al. 1990. Influence of the Ah locus on the humoral immunotoxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin: evidence for AhReceptor-dependent and Ah Receptor-independent mechanisms of immunosuppression. Toxicol Appl Pharmacol. 105;1:26
Kiviranta H et al. 1999. Levels and trends of PCDD/Fs and PCBs in human milk in Finland. Chemosphere 38;2:311
Kiviranta H et al. 2002a. Polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in fishermen in Finland. Environ Health Perspect. 10;4:355
Kiviranta H et al. 2003. PCDD/Fs and PCBs in Baltic herring during the 1990s. Chemosphere 50;9:1201
Kogevinas M. 2001. Human health effects of dioxins: cancer, reproductive and endocrine system effects. Hum Reprod Update 7;3:331
Kohler M et al. 2002. Coplanar polychlorinated biphenyls (PCB) in indoor air. Environ Sci Technol. 36;22:4735
Koistinen J. 1990. Residues of planar polychloroaromatic compounds in Baltic fish and seal. Chemosphere 20;7-9:1043
Koistinen J et al. 1995a. PCDEs, PCBs, PCDDs AND PCDFs in black guillemots and white-tailed sea eagles from the Baltic Sea. Chemosphere 30:9:1671
Koistinen J et al. 1997b. 2,3,7,8-Tetrachlorodibenzo-p-dioxin equivalents in extracts of Baltic white-tailed sea eagles. Environ Toxicol Chem. 16;7:1533
Kääriä J et al. 1988. Effects of coastal eutrophication on the spawning grounds of the Baltic herring in the SW archipelago of Finland. Kieler Meeresforsch Sonderh. 6:348
Laine AO. 2003. Distribution of soft-bottom macrofauna in the deep open Baltic Sea in relation to environmental variability. Estuar Coastal Shelf Sci. 57;1-2:87
Lassen C et al. 2002c. Inventory of dioxin and furan releases in Latvia. Final report. DANCEE - Danish Cooperat Environ East Eur, Min Environ Protect Reg Devel Republic Latvia, Aug 2002.
Lassen C et al. 2002b. Inventory of dioxin and furan releases in Lithuania. Final report. DANCEE – Danish Cooperat Environ East Eur, Min Environ Republic Lithuania, Jun 2002
Lassen C et al. 2003. Danish Cooperation for Environment in Eastern Europe (DANCEE). Survey of dioxin sources in the Baltic Region (extended summary). Environ Sci Pollut Res Int. 10;1:49-56
Lassen C et al. 2002a. Inventory of dioxin and furan releases in Estonia. Draft final report. DANCEE - Danish Cooperat Environ East Eur, Est Min Environ, May 2002
Lassen C et al. 2002d. Inventory of dioxin and furan releases in Poland. DANCEE - Danish Cooperat Environ East Eur, Min Environ Poland, Aug 2002
Lee HM et al. 2000. Endocrine disruptive effects of polychlorinated aromatic hydrocarbons on intestinal cholecystokinin in rats. Endocrinol. 141;8:2938
Legare ME et al. 2000. 2,3,7,8-Tetrachlorodibenzo-p-dioxin alters hippocampal astroglia-neuronal gap junctional communication. Neurotoxicol. 21;6:1109
Lind PM et al. 2000b. Change of bone tissue composition and impaired bone strength in rats exposed to 3,3',4,4',5-pentachlorobiphenyl (PCB126). Toxicol. 150;1-3:41
Lind PM et al. 2004. The dioxin-like pollutant PCB 126 (3,3',4,4',5-pentachlorobiphenyl) affects risk factors for cardiovascular disease in female rats. Toxicol Lett. 150;3:293
Lind Y et al. 2002. Exponering för organiska miljökontaminanter via livsmedel – intagsberäkningar av ΣPCB, PCB 153, p,p'-DDE, PCDD/F, dioxinlika PCB, PBDE och HBCD baserade pa konsumtionsdata från Riksmaten 1997-98. Uppsala, Swed Natl Food Authority. Livsmedelsverket Rapport 26
McKay G. 2002. Dioxin characterisation, formation and minimisation during municipal solid waste (MSW) incineration: review. Chem Eng J. 86;3:343
Miltner A, Emeis K-C. 2001. Terrestrial organic matter in surface sediments of the Baltic Sea, Northwest Europe, as determined by CuO oxidation. Geochim Cosmochim Acta 65;8:1285
Mimura J et al. 1999. Identification of a novel mechanism of regulation of Ah (dioxin) receptor function. Genes Dev. 13;1:20
Muto T et al. 2002. Mammary gland differentiation in female rats after prenatal exposure to 3,3',4,4',5-pentachlorobiphenyl. Toxicol. 177;2-3:197
Muto T et al. 2003. Estrous cyclicity and ovarian follicles in female rats after prenatal exposure to 3,3',4,4',5-pentachlorobiphenyl. Toxicol Lett. 143;3:271
Myrberg K, Andrejev O. 2003. Main upwelling regions in the Baltic Sea – a statistical analysis based on three-dimensional modelling. Boreal Environ Res. 8:97

Neubert D. 1997-98. Reflections on the assessment of the toxicity of “dioxins” for humans, using data from experimental and epidemiological studies. Teratogen Carcinogen Mutagen. 17;4-5:157
Norén K, Meironyté D. 2000. Certain organochlorine and organobromine contaminants in Swedish human milk in perspective of past 20-30 years. Chemosphere 40:1111
Norstrom RJ. 2002. Chemical, biological, ecological and environmental properties fundamental to understanding bioaccumulation of POPs in food webs. Organohalogen Compds. 55:5
Ojaveer E, Lehtonen H. 2001. Fish stocks in the Baltic Sea: Finite or infinite resource? Ambio 30;4-5:217
Okey AB et al. 1994. The Ah receptor: mediator of the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Toxicol Lett. 70;1:1
Paasivirta J, Sinkkonen S. 1998. Temperature dependencies of vapour pressures and solubilities of PCBs, PCDDs and PCDFs for modelling their environmental fate. Organohalogen Compds. 36:389
Paasivirta J et al. 1999. Estimation of vapor pressures, solubilities and Henry’s law constants of selected persistent organic pollutants as functions of temperature. Chemosphere 39;5:811
Pflieger-Bruss S et al. 1999. Effects of single polychlorinated biphenyls on the morphology of cultured rat tubuli seminiferi. Andrologia 31;2:77
Pollenz RS. 2002. The mechanism of AH receptor protein down-regulation (degradation) and its impact on AH receptor-mediated gene regulation. Chem Biol Interact. 141;1-2:41
Quass U et al. 2004a. The European dioxin air emission inventory project--final results. Chemosphere 54;9:1319
Rappe C. 1992. Sources of PCDDs and PCDFs. Introduction. Reactions, levels, patterns, profiles and trends. Chemosphere 25;1-2:41
Render JA et al. 2001. Squamous epithelial proliferation in the jaws of mink fed diets containing 3,3’,4,4’,5-pentachlorobiphenyl (PCB 126) or 2,3,7,8-tetra-chlorodibenzo-P-dioxin (TCDD). Vet Hum Toxicol. 43;1:22
Reyes H S et al. 1992. Identification of the Ah receptor nuclear translocator protein (Arnt) as a component of the DNA binding form of the Ah receptor. Science 256;5060:1193
Rice DC. 1999. Behavioral impairment produced by low-level postnatal PCB exposure in monkeys. Environ Res. 80;2:S113
Sakai S et al. 2005. Emission factors of PBDD/DFs and PBDEs from textile processing and BFR production, and the tentative PBDEs emission inventory. Organohalogen Compds. 2159-62. CD-ROM ID 1754
SCAN. 2000. Opinion of the Scientific Committee on Animal Nutrition on the dioxin contamination of feedingstuffs and their contribution to the contamination of food of animal origin. EC, Brussels. Adopted 06 Nov 2000
Shu XO et al. 1999. Breast-feeding and risk of childhood acute leukemia. J Natl Cancer Inst. 91;20:1765
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
Stark KDC et al. 2002. Risk assessment following the hypothetical import of dioxin-contaminated feed for pigs – an example of quantitative decision support under emergency conditions. Food Control 13;1:1
Stigebrandt A. 2001. Physical oceanography of the Baltic Sea. Wulff F, Rahm L et al. (eds.) A systems analysis of the Baltic Sea. Ecol Studies 148:19-74. Springer Verlag, Berlin & Heidelberg
Storr-Hansen E, Spliid H. 1993a. Coplanar polychlorinated biphenyl congener levels and patterns and the identification of separate populations of harbor seals (Phoca vitulina) in Denmark. Arch Environ Contam Toxicol. 24;1:44
Svensson BG et al. 1995a. Fish consumption and exposure to persistent organochlorine compounds, mercury, selenium and methylamines among Swedish fishermen. Scand J Work Environ Health 21;2:96
Söderström G, Marklund S. 2002. PBCDD and PBCDF from incineration of waste-containing brominated flame retardants. Environ Sci Technol. 36;9:1959
Tamade Y et al. 2002. A study of the compound release from appliance - recycling facility. Organohalogen Compds. 56:189
Tame NW et al. 2003b. Assessing influence of experimental parameters on formation of PCDD/F from ash derived from fires of CCA-treated wood. Environ Sci Technol. 37:4148
Tan Y et al. 2004. Ortho-substituted but not coplanar PCBs rapidly kill cerebellar granule cells. Toxicol Sci. 79;1:147
Tuomisto JT et al. 2004b. Risk-benefit analysis of eating farmed salmon. Science 305:478
Turner A, Rawling MC. 2001. The influence of salting out on the sorption of neutral organic compounds in estuaries. Water Res. 35;18:4379
TWGIM. 2004a. Baseline report on “Integrated monitoring of dioxins & PCBs in the Baltic Region” in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 09 Jan 2004
Tysklind M et al. 1989. PCDD and PCDF emissions from scrap metal melting processes at a steel mill. Chemosphere 19;1-6:705
USEPA. 2000a. Exposure and human health reassessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Draft final. USEPA, Washington, DC. EPA/600/6-88/005Ca. www.epa.gov/ncea/dei.html
van Birgelen AP et al. 1995b. Subchronic effects of 2,3,7,8-TCDD or PCBs on thyroid hormone metabolism: use in risk assessment. Eur J Pharmacol. 293;1:77
Van den Berg M et al. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ Health Perspect. 106:775
van der Oost R et al. 2003. Fish bioaccumulation and biomarkers in environmental risk assessment: a review. Environ Toxicol Pharmacol. 13;2:57
Van Leeuwen FX et al. 2000. Dioxins: WHO’s tolerable daily intake (TDI) revisited. Chemosphere 40;9-11:1095
Vartiainen T et al. 1998. Birth weight and sex of children and the correlation to the body burden of PCDDs/PCDFs and PCBs of the mother. Environ Health Perspect. 106;2:61
Vartiainen T et al. 1997c. Ympäristömyrkkyjen kertyminen silakkaaan. Ympäristö ja Terveys 28;7-8:18
Villeneuve DL et al. 2001. In vitro response of fish and mammalian cells to complex mixtures of polychlorinated naphthalenes, polychlorinated biphenyls, and polycyclic aromatic hydrocarbons. Aquat Toxicol. 54;1-2:125
Vuorinen PJ et al. 1997. The M74 syndrome of Baltic salmon (Salmo salar) and organochlorine concentrations in the muscle of female salmon. Chemosphere 34;5-7:11516
Waern F et al. 1991. Relative liver tumour promoting activity and toxicity of some polychlorinated dibenzo-p-dioxin- and dibenzofuran-congeners in female Sprague-Dawley rats. Pharmacol Toxicol. 69;6:450
Walker NJ et al. 2005. Dose-additive carcinogenicity of a defined mixture of “dioxin-like compounds”. Environ Health Perspect. 113;1:43

Wania F et al. 2001. A multicompartmental, multi-basin fugacity model describing the fate of PCBs in the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Ecol Studies 148:417-47. Springer Verlag, Berlin & Heidelberg
Weber R et al. 1999. Formation and destruction of PCDD/PCDF during heat treatment of fly ash samples from fluidized bed incinerators. Chemosphere 38;11:2633
Weber R, Sakurai T. 2001. Formation characteristics of PCDD and PCDF during pyrolysis processes. Chemosphere 45;8:1111
Weidema I (ed.) 2000. Introduced species in Nordic countries. Nord Environ. 2000;13. 242 p.
Wenborn M et al. 1999. Releases of dioxins and furans to land and water in Europe. Final Report Issue 2. Report for LUW Nordrhein-Westfalen, Germany on behalf of EC DG-Environ, Sep 1999. Report AEAT-4703. europa.eu.int/comm/environment/dioxin/download.htm
Öberg LG, Rappe C. 1992. Biochemical formation of PCDD/Fs from chlorophenols. Chemosphere 25:49

References to Chapter 3

Abbors T. 2003. Kalakauppa ja kalatuotteiden valmistus- toimialaraportti. KTM:n ja TE-keskusten julkaisu, Finn Min Trade Ind and Labor Econ Centre Report, Nov 2003. www.toimialaraportit.fi/files/ 30/Kalatuotteet_toimialaraportti_2003.pdf
Abraham K et al. 1989. Absorption and tissue distribution of various polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDDs and PCDFs) in the rat. Arch Toxicol. 63;3:193
Abraham K et al. 1996. Intake, fecal excretion, and body burden of polychlorinated dibenzo-p-dioxins and dibenzofurans in breast-fed and formula-fed infants. Pediatr Res. 40;5:671
Adolf T et al. 1994. Ergebnisse der Nationalen Verzehrsstudie (1985-1988) über die Lebensmittel- und Nahrstoffaufnahme in der Bundesrepublik Deutschland. Wissens Fachverlag Dr. Fleck, Niederkleen. (Ref. by AUH 1995)
Adriaens P et al. 1995. Bioavailability and transformation of highly chlorinated dibenzo-p- dioxins and dibenzofurans in anaerobic soils and sediments. Environ Sci Technol. 29;9:2252
Adriaens P et al. 1996. Dechlorination of PCDD/F by organic and inorganic electron transfer molecules in reduced environments. Chemosphere 32;3:433
Agrell C et al. 1999. Evidence of latitudinal fractionation of polychlorinated biphenyl congeners along the Baltic Sea region. Environ Sci Technol. 33;8:1149
Agrell C et al. 2001. Atmospheric and river input of PCBs, DDTs and HCHs to the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Ecol Studies 148:149. Springer Verlag, Berlin & Heidelberg
Agrell C et al. 2002. PCB congeners in precipitation, wash out ratios and depositional fluxes within the Baltic Sea region, Europe. Atm Environ. 36;2:371
Ahlborg UG et al. 1994. Toxic equivalency factors for dioxin-like PCBs. Report on a WHO-ECEH and IPCS consultation, Dec 1993. Chemosphere 28:1049
Alcock RE et al. 2000. A generic model of human lifetime exposure to persistent organic contaminants: development and application to PCB-101. Environ Pollut. 110;2:253
Ankarberg E et al. 2004. Study of dioxin and dioxin-like PCB levels in fatty fish from Sweden 2000-2002. Organohalogen Compds. XX:314
Anon. 2000a. (Pentachlorophenol and dioxins)
Anon. 2004. Dioxins & PCBs: Environmental levels and human exposure in candidate countries. Final report. ENV.C.2/SER/2002/0085. Consortium: Environmental Levels In Candidate Countries (ELICC) under supervision of Gunther Umlauf (JRC). EC, Brussels 16.6.2004
AP. Yushchenko shows record dioxin level. AP Newsbreak: Viktor Yushchenko's dioxin level is second highest in history, scientist says. The Associated Press, London, Dec 15, 2004.
Armitage J, Cousins IT. 2005. Application of a simplified bioaccumulation model: Estimation of elimination half-lives for PCDD/PCDFs. Organohalogen Compds. 1791-3. CD-ROM ID 1464
Asplund L et al. 1990. Analysis of non-ortho polychlorinated biphenyls and polychlorinated naphthalenes in Swedish dioxin survey samples. Chemosphere 20:1481
Asplund L et al. 1994. Polychlorinated biphenyls, 1,1,1-trichloro-2,2- bis(p-chlorophenyl)ethane (p,p'-DDT) and 1,1-dichloro-2,2-bis(p-chlorophenyl)-ethylene (p,p'-DDE) in human plasma related to fish consumption. Arch Environ Health 49;6:477
Atkinson R. 1991. Atmospheric lifetimes of dibenzo-p-dioxins and dibenzofurans. Sci Total Environ. 104:17
Atuma SS et al. 1996a. Survey of consumption fish from swedish waters for chlorinated pesticides and polychlorinated biphenyls. Chemosphere 33;5:791
Atuma SS et al. 1998a. Organochlorine pesticides, polychlorinated biphenyls and dioxins in human milk from Swedish mothers. Food Addit Contam. 15;2:142
Aune M et al. 2003. Large differences in dioxin and PCB levels in herring and salmon depending on tissue analysed. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 4
Axelman J et al. 1997. Field measurements of PCB partitioning between water and planktonic organisms: influence of growth, particle size and solute–solvent interactions. Environ Sci Technol. 31:665
Axelman J et al. 2001. Dynamics and distribution of hydrophobic organic compounds in the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Springer Verlag, Berlin & Heidelberg. Ecol Studies 148:257
Aylward LL et al. 2005. Concentration-dependent TCDD elimination kinetics in humans: toxicokinetic modeling for moderately to highly exposed adults from Seveso, Italy, and Vienna, Austria, and impact on dose estimates for the NIOSH cohort. J Expo Anal Environ Epidemiol. 15;1:51
Ayotte P et al. 1996. Health risk assessment for inuit newborns exposed to dioxin-like compounds through breast feeding. Chemosphere 32;3:531
Barabas N et al. 2004. Modified polytopic vector analysis to identify and quantify a dioxin dechlorination signature in sediments. 1. Theory. Environ Sci Technol. 38;6:1813
Becher G et al. 1995. PCDDs, PCDFs, and PCBs in human milk from different parts of Norway and Lithuania. J Toxicol Environ Health 46;2:133-48
Becker W, Pearson M. 1999. Riksmaten 1997-98. Kostvanor och näringsintag I Sverige - Metod- och resultatanalys. Livsmedelsverket, Uppsala 1999. www.slv.se
Bergek S et al. 1992. Concentrations of PCDDs and PCDFs in seals from Swedish waters. Ambio 21;8:553
Berggren P et al. 1999. Patterns and levels of organochlorines (DDTs, PCBs, non-ortho PCBs and PCDD/Fs) in male harbour porpoises (Phocoena phocoena) from the Baltic Sea, the Kattegat-Skagerrak Seas and the West Coast of Norway. Mar Pollut Bull. 38;12:1070

Bergman Å et al. 1992. PCB and PCB methyl sulfones in mink treated with PCB and various PCB fractions. Ambio 21;8:570
Bergman Å et al. 1994a. PCB and DDE methyl sulfones in mammals from Canada and Sweden. Environ Toxicol Chem. 13;1:121
Bergqvist P-A et al. 2005. Kartläggning av utsläppskällor för oavsiktligt bildade ämnen: PCDD/F, PCB och HCB. Miljökemi, Kem Inst, Umeå Univ, Mar 2005. MK 2005:01
Bignert A et al. 1989. Polychlorinated dibenzo-p-dioxins (PCDD) and dibenzo-furans (PCDF) in seal blubber. Chemosphere 19;1-6:551
Bignert A et al. 1998. Temporal trends of organochlorines in Northern Europe, 1967 1995. Relation to global fractionation, leakage from sediments and international measures. Environ Pollut. 99;2:177
Bignert A et al. 2005. Spatial and seasonal variation of the dioxin and PCB content in herring from the northern Baltic Sea. Organohalogen Compds. 1403-5. CD-ROM ID 1639
Birnbaum LS et al. 2003. Health effects of polybrominated dibenzo-p-dioxins (PBDDs) and dibenzofurans (PBDFs). Environ Int. 29;6:855
Bjerselius R et al. 2002b. PCDD/PCDF contribute with half of the total TEq found in fatty fish from the Baltic Sea. Organohalogen Compds. 57:209
Bjerselius R et al. 2003. Study of dioxin levels in fatty fish from Sweden 2001-2002. Part II. Organohalogen Compds. 62:193. CD-ROM Vol. 3, Section 1
Björk M. 1998. Ecological and physiological aspects of contaminant accumulation and transport by the filter-feeding mussel Mytilus edulis. Doctoral Thesis, Stockholm Univ
Björk M et al. 2000. In situ determination of PCB biodeposition by Mytilus edulis in a Baltic coastal ecosystem. Mar Ecol Prog Ser. 194:193
BLAD. 2002a. Dioxine – Daten aus Deutschland. Bund/Länder-Arbeitsgruppe Dioxine. Daten zur Dioxinbelastung der Umwelt. 3. Bericht der BLAD, BMU 2002
Blankenship AL et al. 2000. Relative potencies of individual polychlorinated naphthalenes and halowax mixtures to induce Ah receptor-mediated responses. Environ Sci Technol. 34;15:3153
Blanz T et al. 1999. Chlorobiphenyls in suspension and sediment of the southern Baltic Sea: a mass balance calculation since the onset of PCB-production. Continental Shelf Res. 19;7:891
Blomkvist G et al. 1992. Concentrations of sDDT and PCB in seals from Swedish and Scottish waters. Ambio 21;8:539
Blomqvist S, Heiskanen A-S. 2001. The challenge of sedimentation in the Baltic Sea. Wulff F, Rahm L, Larsson P (eds.) A systems analysis of the Baltic Sea. Springer Verlag, Berlin & Heidelberg. Ecol Studies 148:211
Breivik K, Wania F. 2002. Evaluating a model of the historical behavior of two hexachlorocyclohexanes in the Baltic Sea environment. Environ Sci Technol. 36;5:1014
Breivik K et al. 2002a. Towards a global historical emission inventory for selected PCB congeners - a mass balance approach. 1. Global production and consumption. Sci Total Environ. 290;1-3:181
Breivik K et al. 2002b. Towards a global historical emission inventory for selected PCB congeners – a mass balance approach. 2. Emissions. Sci Total Environ. 290:199
Breivik K, et al. 2004. Primary sources of selected POPs: regional and global scale emission inventories. Environ Pollut. 128;1-2:3
Brewster DW, Birnbaum LS. 1987. Disposition and excretion of 2,3,4,7,8-pentachlorodibenzofuran in the rat. Toxicol Appl Pharmacol. 90;2:243
Brewster DW, Birnbaum LS. 1988a. Disposition of 1,2,3,7,8-pentachlorodibenzofuran in the rat. Toxicol Appl Pharmacol. 95;3:490
Brewster DW et al. 1988a. Toxicity and disposition of 2,3,4,7,8-pentachlorodibenzofuran (4PeCDF) in the rhesus monkey (Macaca mulatta). Toxicol Appl Pharmacol. 93;2:231
Broman D et al. 1989. The composition, distribution and flux of PCDDs and PCDFs in settling particulate matter (SPM) - A sediment trap study in the northern Baltic. Chemosphere 19:445
Broman D et al. 1990a. An in situ study on the distribution, biotransformation and flux of polycyclic aromatic hydrocarbons (PAHs) in an aquatic food chain (seston-mytilus edulis l.-Somateria mollissima L.) from the Baltic: An ecotoxicological perspective. Environ Toxicol Chem. 9;4:429
Broman D et al. 1991a. Long-term high and low-volume air sampling of polychlorinated dibenzo-p-dioxins and dibenzofurans and polycyclic aromatic hydrocarbons along a transect from urban to remote areas on the Swedish Baltic coast. Environ Sci Technol. 25;11:1841
Broman D et al. 1991b. Occurrence and dynamics of polychlorinated dibenzo-p-dioxins and polycyclic aromatic hydrocarbons in the mixed surface layer of remote coastal and offshore waters of the Baltic. Environ Sci Technol. 25:1850
Broman D et al. 1992a. Occurrence and dynamics of polychlorinated dibenzo-p-dioxins and dibenzofurans and other combustion related organic pollutants in the aquatic environment of the Baltic. Chemosphere 25:125
Broman D et al. 1992b. Using ratios of stable nitrogen isotopes to estimate bioaccumulation and flux of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in two food chains from the northern Baltic. Environ Toxicol Chem. 11;3:331
Broman D et al. 1994. Time trend analysis of PAHs and PCBs in the Northern Baltic proper. Chemosphere 29:1325
Broman D et al. 1996. Significance of bacteria in marine waters for the distribution of hydrophobic organic contaminants. Environ Sci Technol. 30;4:1238
Brown JF, Wagner RE. 1990. PCB movement, dechlorination, and detoxification in the Acushnet Estuary. Environ Toxicol Chem. 9:1215
Bruhn R, McLachlan MS. 2002. Seasonal variation of polychlorinated biphenyl concentrations in the southern part of the Baltic Sea. Mar Pollut Bull. 44;2:156
Bucheli TD, Gustafsson Ö. 2003. Soot sorption of non-ortho and ortho substituted PCBs. Chemosphere 53;5:515
Bärring H et al. 2002. Soot–water distribution coefficients for polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans and polybrominated diphenylethers determined with the soot cosolvency-column method. Chemosphere 49;6:515
Carrer S et al. 2000. Modelling the fate of dioxins in a trophic network by coupling an ecotoxicological and an Ecopath model. Ecol Modell. 126;2-3:201
Carrier G et al. 1995a. Modeling of the toxickinetics of polychlorinated dibenzo-p-dioxins and dibenzofurans in mammalians, including humans. I. Nonlinear distribution of PCDD/PCDF body burden between liver an adipose tissues. Toxicol Appl Pharmacol. 131;2:253
Carrier G et al. 1995b. Modeling of the toxicokinetics of polychlorinated dibenzo-p-dioxins and dibenzofurans in mammalians, including humans. II. Kinetics of absorption and disposition of PCDDs/PCDFs. Toxicol Appl Pharmacol. 131;2:267
Chen CY et al. 2001. Disposition of polychlorinated dibenzo-p-dioxins, dibenzofurans, and non-ortho polychlorinated biphenyls in pregnant Long Evans rats and the transfer to offspring. Toxicol Appl Pharmacol. 173;2:65
Chen J et al. 2001a. Quantitative structure-property relationships (QSPRs) on direct photolysis quantum yields of PCDDs. Chemosphere 43;2:235
Chen J et al. 2001c. Quantitative structure-property relationships (QSPRs) on direct photolysis of PCDDs. Chemosphere 45;2:151

Choi JW et al. 2003a. Polybrominated dibenzo-p-dioxins, dibenzofurans, and diphenyl ethers in Japanese human adipose tissue. Environ Sci Technol. 37;5:817
Clench-Aas J et al. 1992. PCDD and PCDF in human milk from Scandinavia, with special emphasis on Norway. J Toxicol Environ Health 37;1:73
Cook PM et al. 2003. Effects of aryl hydrocarbon receptor-mediated early life stage toxicity on lake trout populations in Lake Ontario during the 20th century. Environ Sci Technol. 37;17:3864
Czub G, McLachlan MS. 2004. A food chain model to predict the levels of lipophilic organic contaminants in humans. Environ Toxicol Chem. 23;10:2356
Dahl P et al. 1995. Absorption of polychlorinated biphenyls, dibenzo-p-dioxins and dibenzofurans by breast-fed infants. Chemosphere 30;12:2297
Darnerud PO et al. 1996b. Foetal uptake of coplanar polychlorinated biphenyl (PCB) congeners in mice. Pharmacol Toxicol. 78;3:187
Darnerud PO et al. 2003. Swedish consumption of fatty Baltic Sea fish in relation to the total dioxin intake and the recommended TDI. Organohalogen Compds. 62:183
de Boer J. 2000. Polychlorinated terphenyls. Paasivirta J (ed.), New types of persistent halogenated compounds. The handbook of environmental chemistry. Vol 3, Anthropogenic compounds, Part K:43. Springer, Berlin et al.
de Boer J et al. 1993. Non-ortho and mono-ortho substituted chlorobiphenyls and chlorinated dibenzo-p-dioxins and dibenzofurans in marine and freshwater fish and shellfish from The Netherlands. Chemosphere 26;10:1823
deBruyn AMH, Gobas FAPC. 2004. Modelling the diagenetic fate of persistent organic pollutants in organically enriched sediments. Ecol Modell. 179;3:405
De Felip E et al. 1996. Structure-dependent photocatalytic degradation of polychlorobiphenyls in a TiO2 aqueous system. Chemosphere 33;11:2263
DeVito MJ et al. 1995. Comparisons of estimated human body burdens of dioxinlike chemicals and TCDD body burdens in experimentally exposed animals. Environ Health Perspect. 103;9:820
DeVito MJ et al. 1997. Dose-response relationships for polyhalogenated dioxins and dibenzofurans following subchronic treatment in mice. I. CYP1A1 and CYP1A2 enzyme activity in liver, lung, and skin. Toxicol Appl Pharmacol. 147;2:267
DeVito MJ et al. 1998. Dose-response relationships for disposition and hepatic sequestration of polyhalogenated dibenzo-p-dioxins, dibenzofurans, and biphenyls following subchronic treatment in mice. Toxicol Sci. 46;2:223
de Wit C et al. 1990. Results from the first year of the Swedish dioxin survey. Chemosphere 20;10-12:1473
de Wit C et al. 1992. Polychlorinated dibenzo-p-dioxin and polychlorinated dibenzofuran levels and patterns in fish and fish-eating wildlife in the Baltic Sea. Chemosphere 25;1-2:185
de Wit C, Strandell M. (eds.) 1999. Levels, sources and trends of dioxins and dioxin-like substances in the Swedish environment – The Swedish dioxin survey. Vol. 1. Introduction, toxicology, sampling strategies, chemical analyses, biological test methods and results, foodstuffs. Solna, Statens Naturvardsverk. Swed Environ Protect Agency Report 5047
Drouillard KG et al. 2001. Bioaccumulation and toxicokinetics of 42 polychlorinated biphenyl congeners in American kestrels (Falco sparverius). Environ Toxicol Chem. 20;11:2514
Drouillard KG et al. 2003. Development and validation of a herring gull embryo toxicokinetic model for PCBs. Ecotoxicol. 12;1-4:55
Duarte-Davidson R et al. 1997. Exploring the balance between sources, deposition, and the environmental burden of PCDD/Fs in the U.K. terrestrial environment: an aid to identifying uncertainties and research needs. Environ Sci Technol. 31:1
Dudzinska MR, Czerwinski J. 2003. First attempt to estimate the PCDD/Fs loads from sewage sludge to the soil environment in Poland. Organohalogen Compds. 60-65. CD-ROM
Dybing E. 2003. Panel discussion: application of physiological-toxicokinetic modelling. Toxicol Lett. 138;1-2:173
Dyke P et al. 1997. Dioxins in ambient air, bonfire night 1994. Chemosphere 34;5-7:1191
EC. 2002a. Commission recommendation of 4 March 2002 on the reduction of the presence of dioxins, furans and PCBs in feedingstuffs and foodstuffs. Notified under number C(2002a) 836. Off J EC 9.3.2002 L 67/69 (2002/201/EC)
Eduljee GH, Gair AJ. 1996. Validation of a methodology for modelling PCDD and PCDF intake via the foodchain. Sci Total Environ. 187;3:211
Emond C et al. 2003a. Application of a physiologocially based pharmacokinetic (pbpk) model to aid in understanding relative potency factors for dioxinlike chemicals. Organohalogen Compds. 60-65. CD-ROM, Vol. 6, Section 1
Emond C et al. 2004. Physiologically based pharmacokinetic model for developmental exposures to TCDD in the rat.Toxicol Sci. 2004 Mar 31
Engwall M et al. 1997b. Toxic potencies of extracts of sediment and settling particulate matter collected in the recipient of a bleached pulp mill effluent before and after abandoning chlorine bleaching. Environ Toxicol Chem. 16:1187
Engwall M et al. 1999. Levels of dioxin-like compounds in sewage sludge determined with a bioassay based on EROD induction in chicken embryo liver cultures. Chemosphere 38;10:2327
EUSSC. 2000. Harmonisation of risk assessment procedures, 1st Rep. Eur Union Sci Steering Committee. EC, Brussels. http://europa.eu.int/comm/food/fs/sc/index_en.htm
Fagt S et al. 2002. Danskernes kostvaner 2000-2001. Udviklingen I danskernes kost – forbrug, indkøb og vaner. Fødevaredirektoratet, Søborg 2002. FødevareRapport 2002:10.
Falandysz J et al. 1994a. Congener-specific analysis of polychlorinated biphenyls in white-tailed sea eagles Haliaeetus albicilla collected in Poland. Arch Environ Contam Toxicol. 26;1:13
Falandysz J et al. 1994d. Most toxic and highly bioaccumulative PCB congeners in cod-liver oil of Baltic origin processed in Poland during the 1970s and 1980s, their TEq-values and possible intake. Sci Total Environ. 145;3:207
Falandysz J et al. 1994e. Congener-specific data on polychlorinated biphenyls in tissues of common porpoise from Puck Bay, Baltic Sea. Arch Environ Contam Toxicol 26;3:267
Falandysz J et al. 1996a. Congener-specific analysis of chloronaphthalenes in white-tailed sea eagles Haliaeetus albicilla breeding in Poland. Chemosphere 33;1:51
Falandysz J et al. 1996b. Polychlorinated naphthalenes in sediment and biota from the Gdansk basin, Baltic Sea. Environ Sci Technol. 30:3266
Falandysz J et al. 1997a. Concentrations and biomagnification of polychlorinated naphthalenes in black cormorants Phalacrocorax carbo sinensis from the Gulf of Gdansk, Baltic Sea. Sci Total Environ. 204;1:97
Falandysz J et al. 1997b. Spatial distribution and bioaccumulation of polychlorinated naphthalenes (PCNs) in mussel and fish from the Gulf of Gdansk. Sci Total Environ. 203:93

Falandysz J et al. 1998d. Polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCs) in water of the Vistula River et the Kiezmark site, Poland. Organohalogen Compds. 39:215
Falandysz J et al. 1998f. Polychlorinated naphthalenes in three-spined stickleback gasterosteus aculeatus from the Gulf of Gdansk. Chemosphere 37;9-12:2473
Falandysz J et al. 1999b. [Organochlorine pesticides and polychlorinated biphenyls in the Vistula river water, in Polish with English abstract.] Rocz Panstw Zakl Hig. 50;2:123
Falandysz J et al. 2000b. Relative contribution of chlorinated naphthalenes, -biphenyls, -dibenzofurans and –dibenzo-p-dioxins to toxic equivalents in biota from the South coast of the Baltic Sea. Organohalogen Compds. 47:9
Falandysz J et al. 2002a. Polychlorinated biphenyls (PCBs) and their congener-specific accumulation in edible fish from the Gulf of Gdansk, Baltic Sea. Food Addit Contam. 19;8:779
Falandysz J et al. 2002b. Multivariate analysis of the bioaccumulation of polychlorinated biphenyls (PCBs) in the marine pelagic food web from the southern part of the Baltic Sea, Poland. J Environ Monit. 4;6:929
Feeley MM, Jordan SA. 1998. Dietary and tissue residue analysis and contaminant intake estimations in rats consuming diets composed of Great Lakes salmon: a multigeneration study. Regul Toxicol Pharmacol. 27;1 Pt 2:S8
Fiedler H. 1999. Dioxin and furan inventories. National, Regional Emissions of PCDD/PCDF. UNEP Chemicals, Geneva
Fitzsimmons PN et al. 2001. Branchial elimination of superhydrophobic organic compounds by rainbow trout (Oncorhynchus mykiss). Aquat Toxicol. 55;1-2:23
Flesch-Janys D et al. 1996. Elimination of polychlorinated dibenzo-p-dioxins and dibenzofurans in occupationally exposed persons. J Toxicol Environ Health 47;4:363
Friesen KJ et al. 1995. Effect of the input pathway on the distribution of 1,2,3,4,7-pentachlorodibenzo-p-dioxin in an aquatic mesocosm. Environ Toxicol Chem. 14;11:1921
Friesen KJ et al. 1996. Aquatic photodegradation of polychlorinated dibenzofurans: rates and photoproduct analysis. Environ Sci Technol. 30:2504
Frignani M et al. 2001. Accumulation of polychlorinated dibenzo-p-dioxins and dibenzofurans in sediments of the Venice Lagoon and the industrial area of Porto Marghera. Mar Pollut Bull. 42;7:544
Fu QS et al. 1999. Reductive transformation of dioxins: an assessment of the contribution of dissolved organic matter to dechlorination reactions. Environ Sci Technol. 33;21:3837
Fu QS et al. 2005. Microbial dechlorination of dioxins in estuarine enrichment cultures: effects of respiratory conditions and priming compound on community structure and dechlorination patterns. Mar Environ Res. 59:3:177
Fueno H et al. 2002. Theoretical study of the dechlorination reaction pathways of octachlorodibenzo-p-dioxin. Chemosphere 48;8:771
Fødevaredirektoratet. 1999. Indhold af dioxiner og dioxinlignende PCB i fisk og sundhedsmæssig vurdering i forhold til de kostrad
Fødevaredirektoratet giver den danske befolkning. 3.9.1999. www.vfd.dk/diverse/dioxin_1/dioxin_b1.htm
Gaus C et al. 2002. Transformation processes, pathways, and possible sources of distinctive polychlorinated dibenzo-p-dioxin signatures in sink environments. Environ Sci Technol. 36;16:3542
Geyer HJ et al. 2002. Half-lives of tetra-, penta-, hexa-, hepta-, and octachlorodibenzo-p-dioxin in rats, monkeys, and humans--a critical review. Chemosphere 48;6:631
Gilek M et al. 1997. The role of the blue mussel, Mytilus edulis, in the cycling of hydrophobic organic contaminants in the Baltic Proper. Ambio 26:202
Glynn AW et al. 2000a. Serum concentrations of organochlorines in men: a search for markers of exposure. Sci Total Environ. 263;1-3:197
Glynn AW et al. 2001. Polychlorinated biphenyl congeners as markers of toxic equivalents of polychlorinated biphenyls, dibenzo-p-dioxins and dibenzofurans in breast milk. Environ Res. 86;3:217.
Glynn AW et al. 2003. PCBs and dioxins in breast milk - levels and trends in Sweden 1996-2001. Organohalogen Compds. 60-65, CD-Rom Vol. 5, Section 1.
Glynn A et al. 2005. Associations between dietary habits and body burden of PCBs in pregnant women from Sweden. Organohalogen Compds. 1774-6. CD-ROM ID 1157
Grandjean P et al. 2004. Underestimation of risk due to exposure misclassification. Int J Occup Med Environ Health 17;1:131
Green NJL et al. 2001. PCDD/F deposition time trend to Esthwaite Water, U.K., its relevance to sources. Environ Sci Technol. 35:2882
Grimvall E et al. 1997. Monitoring of polychlorinated biphenyls in human blood plasma: methodological developments and influence of age, lactation, and fish consumption. Arch Environ Contam Toxicol. 32;3:329
Gullett BK et al. 2001. Emissions of PCDD/F from uncontrolled domestic waste burning. Chemosphere 43:721
Gunnarsson M et al. 2005. Identification and quantification of sources to PCDD, PCDF, PCB and HCB in Sweden. Organohalogen Compds. 1319-22. CD-ROM ID 1131
Gustafsson Ö et al. 2003a. Halogenated and aromatic contaminants in sediments: Kd matters. Organohalogen Compds. 60-65 (CD-ROM)
Guvenius DM et al. 2002. Metabolites of polychlorinated biphenyls in human liver and adipose tissue. Environ Toxicol Chem. 21;11:2264
Guvenius DM et al. 2003. Human prenatal and postnatal exposure to polybrominated diphenyl ethers, polychlorinated biphenyls, polychlorobiphenylols, and pentachlorophenol. Environ Health Perspect. 111;9:1235
Hagberg J et al. 2005. Occurrence and levels of PCDD/F and PBDD/Fs in two Swedish lake sediments. Organohalogen Compds. 2030-2. CD-ROM ID 1130
Haglund P et al. 2005. High levels of potentially biogenic dibromo and tribromo dibenzo-p-dioxins in Swedish fish. Organohalogen Compds. 1267-70. CD-ROM ID 1927
Hagmar L et al. 1998. Consumption of fatty fish from the Baltic Sea and PCB in whole venous blood, plasma and cord blood from delivering women in the Aland/Turku archipelago. J Toxicol Environ Health A 53;8:581
Hagmar L et al. 2004b. Tidstrender för halter av persistenta klororganiska miljögifter i blod hos vuxna svenska män i relation till konsumtion av fet östersjöfisk. Rapport till Naturvårdsverket – 2004-03-18. www.imm.ki.se/Datavard/PDF/Rapport%20HAEMI%20dioxin.pdf
Hallikainen A et al. 2004. Kotimaisen järvi- ja merikalan dioksiinien, furaanien, dioksiinien kaltaisten PCB-yhdisteiden ja polybromattujen difenyylieettereiden pitoisuudet. EU-KALAT. Edita Express, Helsinki, Finland. Elintarvikeviraston julkaisuja (Publ Natl Food Admin) 1/2004
Hanari N et al. 2005. Concentrations and compositions of polybrominated biphenyls, -dibenzo-p-dioxins and -dibenzofurans in technical polybrominated diphenyl ether preparations. Organohalogen Compds. 426-9. CD-ROM ID 550.

Hansen E. 2000. Substance flow analysis for dioxins in Denmark. Ministry of Environment, Danish Environ Protect Agency, Copenhagen. Report by Cowi A/S. Environmental Project No. 570. www.mst.dk
Hansen E, Hansen CL. 2003. Substance flow analysis for dioxin 2002. Ministry of Environment, Danish Environ Protect Agency, Copenhagen. Report by Cowi A/S. Environmental project nr. 811. ISBN – electronic 87-7972-675-5. www.mst.dk
Haraguchi K et al. 1992. PCB and PCB methyl sulfones in selected groups seals from Swedish waters. Ambio 21;8:546
Hario M et al. 2004. Organochlorine concentrations in diseased vs. healthy gull chicks from the northern Baltic. Environ Pollut. 127;3:411
Harju M et al. 2003. Determination of atropisomeric and planar polychlorinated biphenyls, their enantiomeric fractions and tissue distribution in grey seals using comprehensive 2D gas chromatography. J Chromatogr A 1019;1-2:127
Hays SM, Aylward LL. 2003. Dioxin risks in perspective: past, present, and future. Regul Toxicol Pharmacol. 37;2:202
Heaton SN et al. 1995. Dietary exposure of mink to carp from Saginaw Bay, Michigan. 1. Effects on reproduction and survival, and the potential risks to wild mink populations. Arch Environ Contam Toxicol. 28;3:334
Heiskanen AS, Leppänen JM. 1995. Estimation of export production in the coastal Baltic Sea: effect of resuspension and microbial decomposition on sedimentation measurements. Hydrobiologia 316:211
Heudorf U et al. 2002. Polychlorinated biphenyls in the blood plasma: current exposure of the population in Germany. Rev Environ Health 17;2:123
Hickie BE et al. 1999. Lifetime pharmacokinetic model for hydrophobic contaminants in marine mammals. Environ Toxicol Chem. 18;11:1622
Hochstein MS Jr et al. 2001. Chronic toxicity of dietary 2,3,7,8-tetrachlorodibenzo-p-dioxin to mink. Vet Hum Toxicol. 43;3:134
Hofelt CS et al. 2001. Development of a metabolism factor for polycyclic aromatic hydrocarbons for use in multipathway risk assessments of hazardous waste combustion facilities. Regul Toxicol Pharmacol. 33;1:60
Holoubek I et al. 2000. Persistent, bioaccumulative and toxic chemicals in Central and Eastern European countries – State-of-the-art report. TOCOEN Report No. 150a. Brno, Czech Republic
Holoubek I et al. 2003a. UNEP/GEF project regional based assessment of persistent toxic substances - European regional report. Organohalogen Compds. 60-65, CD-ROM Vol. 3, Section 4.
Hovander L et al. 2002. Identification of hydroxylated PCB metabolites and other phenolic halogenated pollutants in human blood plasma. Arch Environ Contam Toxicol. 42;1:105
Hu K, Bunce NJ. 1999. Metabolism of polychlorinated dibenzo-p-dioxins and related dioxin-like compounds. J Toxicol Environ Health B 2;2:183
Hurst CH et al. 2000b. Tissue disposition of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in maternal and developing Long-Evans rats following subchronic exposure. Toxicol Sci. 57;2:275
Huwe JK et al. 2003. CALUX and high resolution GC/MS analysis of dioxin-like compounds in chlorophenoxy pesticide formulations. Organohalogen Compds. 60-65, CD-ROM Vol 1, Section 3
ICES. 2005c. Report of the Baltic Fisheries Assessment Working Group (WGBFAS). Hamburg, 12-21 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:19
Imai T et al. 2003. Comparison of the recyclability of flame-retarded plastics. Environ Sci Technol. 37;3:652
IPCS. 1998. Polybrominated dibenzo-p-dioxins and dibenzofurans. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 205
IPCS. 2000. Human exposure assessment. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 214
Ishaq R et al. 2000. Tissue distribution of polychlorinated naphthalenes (PCNs) and non-orthochlorinated biphenyls (non-ortho CBs) in harbour porpoises (Phocoena phocoena) from Swedish waters. Chemosphere 41;12:1913
Isosaari P. 2004. Polychlorinated dibenzo-p-dioxin and dibenzofuran contamination of sediments and photochemical decontamination of soils. Kuopio, Finland, NPIH. Publ Natl Public Health Inst. A 11/2004. Doctoral dissertation, Univ Kuopio
Isosaari P et al. 2000. Assessment of levels, distribution, and risks of polychlorinated dibenzo-p-dioxins and dibenzofurans in the vicinity of a vinyl chloride monomer production plant. Environ Sci Technol. 34:2684
Isosaari P et al. 2002b. Feeding trial on rainbow trout: comparison of dry fish feed and Baltic herring as a source of PCDD/Fs and PCBs. Chemosphere 48;8:795
Isosaari P et al. 2002c. Spatial distibution and temporal accumulation of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in the Gulf of Finland. Environ Sci Technol. 36;12:2560
Isosaari P et al. 2003. Dioxin levels in fish caught from the Baltic Sea in 2001-2002. Organohalogen Compds. 60-65, CD-ROM Vol. 3, Section 1
Janák K et al. 1998. Methyl sulfonyl polychlorinated biphenyls and 2,2-bis(4-chlorophenyl)-1,1-dichlorethene in gray seal tissues determined by gas chromatography with electron capture detection and atomic emission detection. Environ Toxicol Chem. 17;6:1046
Jensen AA. 1987. Polychlorobiphenyls (PCBs), polychlorodibenzo-p-dioxins (PCDDs) and polychlorodibenzofurans (PCDFs) in human milk, blood and adipose tissue. Sci Total Environ. 64;3:259
Jensen AA. 2003. Kortlæging av dioxinforurening samt kilder til dioxinforurening i Østersøen. Miljøministeriet, Copenhagen. Report by dk-TEKNIK ENERGI & MILJO. Miljøprojekt nr. 796 (2003). www.mst.dk/udgiv/publikationer/2003/87-7972-570-8/html/bilag01.htm
Jensen S et al. 1969. DDT and PCB in marine animals from Swedish waters. Nature 224:247
Jensen S et al. 1977a. Levels of DDT and PCB in littoral fishes along the Swedish coast. Ambio Spec Report 5:75
Jones PD et al. 2001. Accumulation of 2,3,7,8-tetrachlorodibenzo-p-dioxin by rainbow trout (Onchorhynchus mykiss) at environmentally relevant dietary concentrations. Environ Toxicol Chem. 20;2:344
Jonsson A et al. 2003b. Global accounting of PCBs in the continental shelf sediments. Environ Sci Technol.37;2:245
Jonsson P. 2000. Sediment burial of PCBs in the offshore Baltic Sea. Ambio 29:260
Jonsson P et al. 1993. Pulp mill related polychlorinated organic compounds in baltic sea sediments. Ambio 22:37
Kakareka SV. 2002. Sources of persistent organic pollutants emission on the territory of Belarus. Atm Environ. 36;8:1407
Kankaanpää H et al. 1997. Seasonal sedimentation of organic matter and contaminants in the Gulf of Finland. Boreal Environ Res. 2;3:257
Kannan K et al. 1992. Temporal trends of organochlorine concentrations in cod-liver oil from the southern Baltic proper, 1971-1989. Mar Pollut Bull. 24;7:358
Kannan K et al. 2002a. Polychlorinated biphenyls, dibenzo-p-dioxins, dibenzofurans and DDE in white-tailed sea eagle livers from Eastern Germany, 1979-1998. Organohalogen Compds. 57:455. CO-ROM.
Karl H, Ruoff U. 2004. Dioxins and dioxin-like PCBs in fish in general and in particular from the Baltic Sea. Organohalogen Compds. 66:1910
Karl H et al. 2002. Levels of dioxins in fish and fishery products on the German market. Chemosphere 49;7:765
Karlson K et al. 2000. PCBs, DDTs and methyl sulphone metabolites in various tissues of harbour porpoises from Swedish waters. Environ Pollut. 110;1:29

Kawamoto K, Ishikawa N. 2005. Experimental evidence for de novo synthesis of PBDD/PBDF and PXDD/PXDF as well as dioxins in the thermal processes of ash samples. Organohalogen Compds. 2219-21. CD-ROM ID 1571
Kerger B et al. 2005. An age-dependent half-life model for estimating childhood body burdens of dibenzodioxins and dibenzofurans. Proc Soc Toxicol 44th Ann Meeting, New Orleans, LA, Mar 6-10 2005
Kihlström JE, Berglund E. 1978. An estimation of the amounts of polychlorinated piphenyls in the biomass of the Baltic. Ambio 7;4:175
Kihlström JE et al. 1992. Effects of PCB and different fractions of PCB on the reproduction of the mink (Mustela vison). Ambio 21;8:563
Kim M, O'Keefe PW. 2000. Photodegradation of polychlorinated dibenzo-p-dioxins and dibenzofurans in aqueous solutions and in organic solvents. Chemosphere 41;6:793
Kim Y, Lee D. 2002. Solubility enhancement of PCDD/F in the presence of dissolved humic matter. J Haz Mater. 91;1-3:113
Kirchner JW, Hooper RP et al. 1996. Testing and validating environmental models. Sci Total Environ. 183,1-2:33
Kitamura K et al. 2002. Anatomical variation levels of dioxin congeners in human sebum. Organohalogen Compds. 55:311
Kiviranta H et al. 1999. Levels and trends of PCDD/Fs and PCBs in human milk in Finland. Chemosphere 38;2:311
Kiviranta H et al. 2001. Dietary intakes of polychlorinated dibenzo-p-dioxins, dibenzofurans and polychlorinated biphenyls in Finland. Food Addit Contam. 18;11:945
Kiviranta H et al. 2002a. Polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in fishermen in Finland. Environ Health Perspect. 10;4:355
Kiviranta H et al. 2002b. PCDD/Fs in Baltic herring in the Gulf of Finland during the 1990's. Organohalogen Compds. 57:153
Kiviranta H et al. 2003. PCDD/Fs and PCBs in Baltic herring during the 1990s. Chemosphere 50;9:1201
Kiviranta H et al. 2004. Market basket study on dietary intake of PCDD/Fs, PCBs, and PBDEs in Finland. Environ Int. 30;7:923
Kiviranta H. et al. 2005. Polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in the general population in Finland. Chemosphere 60;7:854
Kjeller L-O, Rappe C. 1995. Time trends in levels, patterns, and profiles for polychlorinated dibenzo-p-dioxins, dibenzofurans and biphenyls in a sediment core from the Baltic Proper. Environ Sci Technol. 29:346
Kleeman JM et al. 1986a. Metabolism and disposition of 2,3,7,8-tetrachlorodibenzo-p-dioxin in rainbow trout. Toxicol Appl Pharmacol. 83;3:391
Kodavanti PR et al. 1998. Congener-specific distribution of polychlorinated biphenyls in brain regions, blood, liver, and fat of adult rats following repeated exposure to Aroclor 1254. Toxicol Appl Pharmacol. 153;2:199
Koistinen J. 1990. Residues of planar polychloroaromatic compounds in Baltic fish and seal. Chemosphere 20;7-9:1043
Koistinen J et al. 1995a. PCDEs, PCBs, PCDDs AND PCDFs in black guillemots and white-tailed sea eagles from the Baltic Sea. Chemosphere 30:9:1671
Koistinen J et al. 1995c. Contamination of pike and sediment from the Kymijoki River by PCDEs, PCDDs and PCDFs: contents and patterns compared to the pike and sediment from the Bothnian Bay and seals from Lake Saimaa. Environ Sci Technol. 29:2541
Koistinen J et al. 1997a. Polychlorinated diphenyl ethers, dibenzo-p-dioxins, dibenzofurans and biphenyls in seals and sediment from the Gulf of Finland. Chemosphere 35;6:1249
Koistinen J et al. 1997b. 2,3,7,8-Tetrachlorodibenzo-p-dioxin equivalents in extracts of Baltic white-tailed sea eagles. Environ Toxicol Chem. 16;7:1533
Koistinen J et al. 2002. PCDD/Fs, PCNs, PBDEs and PCBs in food sources of Baltic seals. Organohalogen Compds. 57:157
Korhonen M et al. 2001. Concentrations of polychlorinated dibenzo-p-dioxins and furans in fish downstream from a Ky-5 manufacturing. Chemosphere 43;4-7:587
Korhonen M et al. 2002. The deposition and sedimentation of PCDD/Fs in the Gulf of Finland. Organohalogen Compds. 57:317-20.
Korhonen M et al. Unpublished. The deposition of PCDD/Fs in Southern Finland. Manuscript, SYKE, Sep 2005
Korner W et al. 2002. Tissue concentrations and induction of a hepatic monooxygenase in male Wistar rats after repeated doses of defined polychlorinated dibenzo-p-dioxin and dibenzofuran (PCDDs and PCDFs) mixtures. Arch Toxicol. 75;11-12:653
Kotz A et al 2005. PBDE, PBDD/F and mixed chlorinated-brominated PXDD/F in pooled human milk samples from different countries. Organohalogen Compds. 1540-4. CD-ROM ID 1099
Kreuzer PE et al. 1997. 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and congeners in infants. A toxicokinetic model of human lifetime body burden by TCDD with special emphasis on its uptake by nutrition. Arch Toxicol. 71;6:383
Krowke R et al. 1990. Transfer of various PCDDs and PCDFs via placenta and mother's milk to marmoset offspring. Chemosphere 20;7-9:1065
LaKind JS et al. 2000. Methodology for characterizing distributions of incremental body burdens of 2,3,7,8-TCDD and DDE from breast milk in North American nursing infants. J Toxicol Environ Health A 59;8:605
Larsson P et al. 1996. Persistent pollutants in a salmon population (Salmo salar) of the southern Baltic Sea. Can J Fish Aquat Sci. 53:62
Lassen C et al. 2002a. Inventory of dioxin and furan releases in Estonia. Draft final report. DANCEE - Danish Cooperat Environ East Eur, Est Min Environ, May 2002
Lassen C et al. 2002b. Inventory of dioxin and furan releases in Lithuania. Final report. DANCEE - Danish Cooperat Environ East Eur, Min Environ Republic Lithuania, Jun 2002
Lassen C et al. 2002c. Inventory of dioxin and furan releases in Latvia. Final report. DANCEE - Danish Cooperat Environ East Eur, Min Environ Protect Reg Devel Republic Latvia, Aug 2002
Lassen C et al. 2002d. Inventory of dioxin and furan releases in Poland. DANCEE - Danish Cooperat Environ East Eur, Min Environ Poland, Aug 2002
Lassen C et al. 2003. Danish Cooperation for Environment in Eastern Europe (DANCEE). Survey of dioxin sources in the Baltic Region (extended summary). Environ Sci Pollut Res Int. 10;1:49
Lemmetyinen R, Rantamäki P. 1980. DDT and PCB residues in the arctic tern (Sterna paradisaea) nesting in the archipelago of southwestern Finland. Ann Zool Fenn. 7:141
Lemmetyinen R et al. 1982. Levels of DDT and PCBs in different stages of life cycle of the arctic tern Sterna paradisaea and the herring gull Larus argentatus. Chemosphere 11:1059
Leung HW et al. 1990. A physiological pharmacokinetic description of the tissue distribution and enzyme-inducing properties of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the rat. Toxicol Appl Pharmacol. 103;3:399
Li X-W et al. 2003. Establishment of free energies of the formation of polybrominated dibenzo-p-dioxins and furans. Organohalogen Compds. 60-65. CD-ROM, Vol. 4, Section 2

Lignell S et al. 2005. Time trend of dioxins in breast milk in Sweden 1996-2004. Organohalogen Compds. 1770-3. CD-ROM ID 1155
Lind Y et al. 2002. Exponering för organiska miljökontaminanter via livsmedel – intagsberäkningar av ΣPCB, PCB 153, p,p'-DDE, PCDD/F, dioxinlika PCB, PBDE och HBCD baserade pa konsumtionsdata från Riksmaten 1997-98. Uppsala, Swed Natl Food Authority. Livsmedelsverket Rapport 26
Litten S et al. 2002. Identification of a novel PCB source through analysis of 209 PCB congeners by US EPA modified method 1668. Chemosphere 46:1457
Lorber M, Phillips L. 2002. Infant exposure to dioxin-like compounds in breast milk. Environ Health Perspect. 110;6:A325
Lundgren K. 2003. Properties and analysis of dioxin-like compounds in marine samples from Sweden. PhD Thesis, Univ Umea www.diva-portal.org/diva/getDocument?urn_ nbn_se_umu_diva-24-1_fulltext.pdf
Lundgren K et al. 2002a. Development of a high-performance liquid chromatography carbon column based method for the fractionation of dioxin-like polychlorinated biphenyls. J Chromatogr A 962;1-2:79
Lundgren K et al. 2003a. Flux estimates and sedimentation of polychlorinated naphthalenes in the northern part of the Baltic Sea. Environ Pollut. 126;1:93
Lundgren K et al. 2004. Biomagnification of mono-ortho and non-ortho PCBs in a benthic food chain in the Baltic Sea. Organohalogen Compds. 66:2352
Lundstedt-Enkel K et al. 2002. Different PCDD/PCDF congener composition in salmon and brown trout from Swedish waters. Organohalogen Compds. 57:185
Luthardt P et al. 2002. Total TEq emissions (PCDD/F and PCB) from industrial sources. Chemosphere 46;9-10:1303
Mackay D, Bentzen E. 1997. The role of the atmosphere in Great Lakes contamination. Atm Environ. 31:4045
Mackay D et al. 1992. Illustrated handbook of physico-chemical properties and environmental fate for organic chemicals. Volume 2. Polynuclear aromatic hydrocarbons, polychlorinated dioxins and dibenzofurans. Lewis Publ, Boca Raton, FL
Mackenzie BR et al. 2004. Fish, fishing, and pollutant reduction in the Baltic Sea. Environ Sci Technol. 38;7:1970
Malisch R, van Leeuwen FXR. 2003. Results of the WHO-coordinated exposure study on the levels of PCBs, PCDDs and PCDFs in human milk. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 1
Malmvärn A et al. 2005. Identification of brominated dibenzo-p-dioxins in blue mussels (Mytilus edulis) from the Baltic Sea. Organohalogen Compds. 1229-32. CD-ROM ID 1129
Marklund S et al. 1991. Environmental deposition of PCDDs and PCDFs as determined by the analysis of snow samples from the Northern Sweden. Chemosphere 23;8-10:1359
Maruyama W et al. 2003. Simulation of dioxin accumulation in human tissues and analysis of reproductive risk. Chemosphere 53;4:301
Maruyama W et al. 2004. Dioxin health risk to infants using simulated tissue concentrations. Environ Toxicol Pharmacol. 18;1:21
Matscheko N et al. 2002. Application of sewage sludge to arable land-soil concentrations of polybrominated diphenyl ethers and polychorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls, and their accumulation in earthworms. Environ Toxicol Chem. 21;12:2515
McLachlan MS et al. 1996. Polychlorinated dibenzo-p-dioxins and dibenzofurans in sewage sludge: Sources and fate following sludge application to land. Sci Total Environ. 185:109
Metcalfe TL, Metcalfe CD. 1997. The trophodynamics of PCBs, including mono- and non-ortho congeners, in the food web of North-Central Lake Ontario. Sci Total Environ. 201;3:245
Miao X-S et al. 1999. Degradation pathways of PCBs upon UV irradiation in hexane. Chemosphere 39;10:1639
Moilanen R et al. 1982. Time trends of chlordane, DDT, and PCB concentrations in pike (Esox lucius) and Baltic herring (Clupea harengus) in the Turku archipelago, Northern Baltic Sea for the period 1971-1982. Bull Environ Contam Toxicol. 29:334
Moser GA, McLachlan MS. 2002. Modeling digestive tract absorption and desorption of lipophilic organic contaminants in humans. Environ Sci Technol. 36;15:3318
Mousa MA et al. 1998. Altered biologic activities of commercial polychlorinated biphenyl mixtures after microbial reductive dechlorination. Environ Health Perspect. 106 Suppl 6:1409
MSC-E. 2005. POPs emissions. Meterol Synthesizing Centre East, Moscow. www.msceast.org/ Updated 06 Jan 2005.
Männistö S et al. (eds.) 2003. The national FINDIET 2002 study. Publ Natl Inst Public Health, Nutr Unit, Helsinki 2003. www.ktl.fi/portal/suomi/osiot/ktl_tutkii/tutkimus/ravitsemus
Nagao T et al. 1995. Tissue distribution after a single subcutaneous administration of 2,3,7,8-tetrabromodibenzo-P-dioxin in comparison with toxicokinetics of 2,3,7,8-tetrachlorodibenzo-p-dioxin in female Wistar rats. Life Sci. 58;4:325
Naito W et al. 2003. Dynamics of PCDDs/DFs and coplanar-PCBs in an aquatic food chain of Tokyo Bay. Chemosphere 53;4:347
Neubert D. 1997-98. Reflections on the assessment of the toxicity of "dioxins" for humans, using data from experimental and epidemiological studies. Teratogen Carcinogen Mutagen. 17;4-5:157
NMR. 2001. Livsmedelskonsumtionen i Norden 1965-1998, Nationell, årlig per capita statistik. Food consumption in the Nordic countries 1965-1998. København, Nordiska Ministerrådet. TemaNord 2001:527. www.norden.org
Nordsieck HO, Mücke W. 2002. Characterization and quantification of dioxin-like compounds in flue gas condensates of municipal solid waste incinerators. Organohalogen Compds. 56:281
Norén K, Lundén A. 1991. Trend studies of polychlorinated biphenyls, dibenzo-p-dioxins and dibenzofurans in human milk. Chemosphere 23;11-12:1895
Norén K, Meironyté D. 2000. Certain organochlorine and organobromine contaminants in Swedish human milk in perspective of past 20-30 years. Chemosphere 40:1111
Nylund K et al. 1992. Analysis of some polyhalogenated organic pollutants in sediment and sewage sludge. Chemosphere 24:1721
Nyman M et al. 2002. Current levels of DDT, PCB and trace elements in the Baltic ringed seals (Phoca hispida baltica) and grey seals (Halichoerus grypus). Environ Pollut. 119;3:399
Nyman M et al. 2003. Contaminant exposure and effects in Baltic ringed and grey seals as assessed by biomarkers. Mar Environ Res. 55;1:73
Näf C et al. 1992. Flux estimates and pattern recognition of particulate polycyclic aromatic hydrocarbons, polychlorinated dibenzo-p-dioxins, and dibenzofurans in the waters outside various emission sources on the Swedish Baltic coast. Environ Sci Technol. 26:1444
Odsjö T et al. 1997. The Swedish environmental specimen bank - application in trend monitoring of mercury and some organohalogenated compounds. Chemosphere 34;9-10:2059

Ogura I et al. 2001. Atmospheric deposition of polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans, and dioxin-like polychlorinated biphenyls in the Kanto Region, Japan. Chemosphere 44;6:1473
Olsson A et al. 2000b. Nestling blood of the white-tailed sea eagle (Haliaeetus albicilla) as an indicator of territorial exposure to organohalogen compounds - An evaluation. Environ Sci Technol. 34;13:2733
Olsson M et al. 1974. DDT and PCB levels in seals from Swedish waters. The occurrence of aborted seal pups. SNV PM 591:43
Olsson M et al. 1992a. Contaminants and diseases of seals from Swedish waters. Ambio 21;8:561
Olsson M et al. 1992b. Seals and seal protection: Summary and comments. Ambio 21;8:606
Olsson M et al. 1994. Diseases and environmental contaminants in seals from the Baltic and the Swedish west coast. Sci Total Environ. 154;2-3:217
Olsson M et al. 2005. Miljögifter i Östersjön – från upptäckt till samhällsreaktion. Östersjö 2005:21
Olsson M et al. 2005. Dioxiner i kustlevande fisk från södra Bottenhavet – en studie av presumtiva föroreningskällor. Unpubl study funded by Swedish EPA. Stockholm 2005 06 08
Paasivirta J. 1990. Orgaanisten klooriyhdisteiden ympäristömerkitys vesiekosysteemissä. Kemia-Kemi 17;1:34
Paasivirta J, Linko R. 1980. Environmental toxins in Finnish Wildlife. A study on time trends of residue contents in fish during 1973-1978. Chemosphere 9;10:643
Pacyna JM (coord.) 2002. Impacts of input changes of selected POPs to the Baltic Sea (BALPOP). Final Project Report, Reporting period 1 Jan 2001 – 31 Apr 2002. 17 p
Pacyna JM et al. 2003. European atmospheric emissions of selected persistent organic pollutants, 1970–1995. Atm Environ. 37;Suppl 1:119
Patandin S et al. 1999. Dietary exposure to polychlorinated biphenyls and dioxins from infancy until adulthood: A comparison between breast-feeding, toddler, and long-term exposure. Environ Health Perspect. 107;1:45
Paustenbach D. 2004. An approach to calculating childhood body burdens of dioxin using age-dependent half lives. Organohalogen Compds. 66
Pekar M et al. 1999. Long-range transport potential of selected persistent organic pollutants. EMEP Report 4/99, Jul 1999. MSC-E, Moscow
Petroff BK et al. 2001. A review of mechanisms controlling ovulation with implications for the anovulatory effects of polychlorinated dibenzo-p-dioxins in rodents. Toxicol. 158;3:91
Pettersen H et al. 1999. The relative contribution of spatial-, sampling- and analytical variation to the PAH and PCB concentrations in Baltic Sea sediments. Chemosphere 38;5:1025
Pluess N et al. 1987. The metabolism of some pentachlorodibenzofurans in the rat. Xenobiotica 17;2:209
Quass U, Fermann M. 1997. Identification of relevant industrial sources of dioxins and furans in Europe (The European dioxin inventory). Final Report. Materialien No. 43, LUA NRW. europa.eu.int/comm/environment/dioxin/download.htm.
Quass U et al. 2000. Steps towards a European dioxin emission inventory. Chemosphere 40;9-11:1125.
Quass U et al. 2004a. The European dioxin air emission inventory project--final results. Chemosphere 54;9:1319
Quass U et al. 2004b. The DG Environment project “Dioxin emissions in candidate countries”: Scope, approach and first results. Organohalogen Compds. 66:878
Raccanelli SD et al. 2002. Monitoring POPs (PCDD/F, PCB, HCB, PAH, DDT) in atmospheric deposition: Sampling and analytical problems. Organohalogen Compds. 58:49
Rappe C. 1993. Sources of exposure, environmental concentrations and exposure assessment of PCDDFs and PCDFs. Chemosphere 27;12-3:211
Rappe C et al. 1989a. Long-range transport of PCDDs and PCDFs on airborne particles. Chemosphere 18;1-6:1283
Rayne S et al. 2002. Photochemical mass balance of 2,3,7,8-TeCDD in aqueous solution under UV light shows formation of chlorinated dihydroxybiphenyls, phenoxyphenols, and dechlorination products. Environ Sci Technol. 36;9:1995
Robson M, Harrad S. 2004. Chiral PCB signatures in air and soil: implications for atmospheric source apportionment. Environ Sci Technol. 38;6:1662
Rolff C et al. 1993. Potential biomagnification of PCDD/Fs - new possibilities for quantitative assessment using stable isotope trophic position. Chemosphere 27;1-3: 461
Roos A et al. 1998. Time trend studies on ΣDDT and PCB in juvenile grey seals (Halichoerus grypus), fish and guillemot eggs from the Baltic Sea. Organohalogen Compds. 39:109
Roots O et al. 2003. Dioxins in the Baltic herring and sprat in Estonian coastal waters. Organohalogen Compds. 60-65 (CD-ROM).
Rose M, Startin J. 2003. Accuracy and comparability of analytical data for PCDD/Fs and PCBs in food. Organohalogen Compds. 60-65, CD-ROM Vol 1, Section 2
Ross PS et al. 1995. Contaminant-related suppression of delayed-type hypersensitivity and antibody responses in harbor seals fed herring from the Baltic Sea. Environ Health Perspect. 103:162
Ross PS et al. 1996c. Contaminant-induced immunotoxicity in harbour seals: Wildlife at risk? Toxicol. 112;2:157
Routti H et al. 2005. Accumulation of dietary organochlorines and vitamins in Baltic seals. Mar Environ Res. 60;3:267
Rozman K et al. 1987. Effect of a sublethal dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin on interscapular brown adipose tissue of rats. Toxicol Pathol. 15;4:425
Rykiel EJ Jr. 1996. Testing ecological models: the meaning of validation. Ecol Modell. 90;3:229
Rylander L et al. 1995. Decreased birthweight among infants born to women with a high dietary intake of fish contaminated with persistent organochlorine compounds. Scand J Work Environ Health 21;5:368
Räisanen S, Salkinoja-Salonen M. 1983. Kloorатuista dioksiineista ja furaaneista. Kemia-Kemi 11:903
Sabljic A. 2001. QSAR models for estimating properties of persistent organic pollutants required in evaluation of their environmental fate and risk. Chemosphere 43;3:363
Safe S. 1989b. Polyhalogenated aromatics: uptake, disposition and metabolism. Kimbrough RD, Jensen AA (eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Elsevier Science Publ BV. 2nd ed.
Safe S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). Crit Rev Toxicol. 21;1:51
Sakai S et al. 2005. Emission factors of PBDD/DFs and PBDEs from textile processing and BFR production, and the tentative PBDEs emission inventory. Organohalogen Compds. 2159-62. CD-ROM ID 1754

Sandberg J. 2004. Test of the suitability of ECOPATH/ECOSIM modelling software as a compliment to estimate flows of carbon, C-14 and radionuclides in the Öregrundsgrepen area. Dept Systems Ecol, Stockholm Univ. SKB Rapport R-04-37
Sapota G. 1996. Chlorinated hydrocarbons in marine biota and sediments from the Gulf of Gdansk. ICES Cooperat Res Report 257:38
Sapota G. 1997. Chlorinated hydrocarbons in sediments from the Vistula Lagoon. Oceanol Stud. 26:61
SCAN. 2000. Opinion of the Scientific Committee on Animal Nutrition on the dioxin contamination of feedingstuffs and their contribution to the contamination of food of animal origin. EC, Brussels. Adopted 06 Nov 2000
Schramm K-W et al. 1995. PCDD/F sources and levels in River Elbe sediments. Water Res. 29;9:2160
Schrey P et al. 1998. Human fecal PCDD/F-excretion exceeds the dietary intake. Chemosphere 37;9-12:1825
Schulz-Bull DE et al. 1995. Distribution of individual chlorobiphenyls (PCB) in solution and suspension in the Baltic Sea. Mar Chem. 48;3-4:245
SCOOP. 2000. Reports in tasks for scientific cooperation. Report of experts participating in task 3.2.5, Brussels 7.6.2000. DG-SANCO. europa.eu.int/comm/dgs/health_consumer/ library/pub/pub08_en.pdf
Shelepchikov A et al. 2005. Contamination of Russian Baltic fish by polychlorinated dibenzo-p-dioxins, dibenzofurans and dioxin-like biphenyls. Organohalogen Compds. 1502-7. CD-ROM ID 2331
Sinkkonen S. 2000. Polychlorinated dibenzothiophenes (PCDTs), thianthrenes (PCTAs) and their alkylated derivatives. Paasivirta J (ed.), New types of persistent halogenated compounds. The handbook of environmental chemistry. Vol 3, Anthropogenic compounds, Part K:289. Springer, Berlin et al.
Sinkkonen S, Paasivirta J. 2000. Degradation half-life times of PCDDs, PCDFs and PCBs for environmental fate modeling. Chemosphere 40;9-11:943
Sjödin A et al. 2001. Flame retardants in indoor air at an electronics recycling plant and at other work environments. Environ Sci Technol. 35;3:448
Sjödin A et al. 1998. Identification of the parent compounds to selectively retained hydroxylated PCB metabolites in rat blood plasma. Organohalogen Compds. 37:365
SNFA. 2003. Delrapport 3 –dioxinanalyser av fet fish från Sverige 2000-2002. Swed Natl Food Admin. www.slv.se/
SNFA. 2004. Interim report 4 – Study of dioxin levels in fatty fish from Sweden 2000-2002. Swed Natl Food Admin. www.slv.se/
SNFA. 2005. Interim report 5- Study of dioxin-like PCB levels in fatty fish from Sweden 2000-2002. Swed Natl Food Admin. www.slv.se/
SNV. 1987. Ref. by Paasivirta 1990 (op. cit.; no reference given in literature list)
Sormo EG et al. 2003. Partitioning of persistent organic pollutants in grey seal (Halichoerus grypus) mother–pup pairs. Sci Total Environ. 302;1-3:145
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
Stephens RD et al. 1995. Biotransfer and bioaccumulation of dioxins and furans from soil: chickens as a model for foraging animals. Sci Total Environ. 175;3:253
Storr-Hansen E, Spliid H. 1993a. Coplanar polychlorinated biphenyl congener levels and patterns and the identification of separate populations of harbor seals (Phoca vitulina) in Denmark. Arch Environ Contam Toxicol. 24;1:44
Strandberg B et al. 1998b. Concentrations, biomagnification and spatial variation of organochlorine compounds in a pelagic food web in the northern part of the Baltic Sea. Sci Total Environ. 217;1-2:143
Strandberg B et al. 1998c. Concentrations and biomagnification of 17 chlordane compounds and other organochlorines in harbour porpoise Phocoena phocoena from southern Baltic Sea. Chemosphere 37:2513
Strandberg B et al. 1998d. Concentrations and spatial variations of cyclodienes and other organochlorines in herring and perch from the Baltic Sea. Sci Total Environ. 215;1-2:69
Strandberg B et al. 1998e. Occurrence, sedimentation, and spatial variations of organochlorine contaminants in settling particulate matter and sediments in the Northern part of the Baltic Sea. Environ Sci Technol. 32;12:1754
Su M-C, Christensen ER. 1997. Apportionment of sources of polychlorinated dibenzo-p-dioxins and dibenzofurans by a chemical mass balance model. Water Res. 31;12:2935
Sundberg H et al. 2005. The distribution and relative toxic potential of organic chemicals in a PCB contaminated bay. Mar Pollut Bull. 50;2:195
Suzuki N et al. 2000. Simulation of long-term environmental dynamics of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans using the dynamic multimedia environmental fate model and its implication to the time trend analysis of dioxins. Chemosphere 40;9-11:969
Svensson BG et al. 1991. Exposure to dioxins and dibenzofurans through the consumption of fish. New Engl J Med. 324;1:8
Svensson BG et al. 1995a. Fish consumption and exposure to persistent organochlorine compounds, mercury, selenium and methylamines among Swedish fishermen. Scand J Work Environ Health 21;2:96
Söderström G, Marklund S. 2002. PBCDD and PBCDF from incineration of waste-containing brominated flame retardants. Environ Sci Technol. 36;9:1959
Takasuga T et al. 2005. Isotope dilution analysis of polychlorinated biphenyls (PCBs) in transformer oil and global commercial PCB formulations by high resolution gas chromatography-high resolution mass spectrometry. Chemosphere 2005 Jun 7, Epub ahead of print
Tarhanen J et al. 1989. Toxic significance of planar aromatic compounds in Baltic ecosystem — New studies on extremely toxic coplanar PCBs. Chemosphere 18;1-6:1067
Thyen S et al. 2000. Organochlorine and mercury contamination of little terns (Sterna albifrons) breeding at the western Baltic Sea, 1978-96. Environ Pollut. 108;2:225
Tiedje JM et al. 1993-94. Microbial reductive dechlorination of PCBs. Biodegrad. 4(4):231
Tuomisto JT et al. 2004a. Soft-tissue sarcoma and dioxin: A case-control study. Int J Cancer 108;6:8930
TWGIM. 2004a. Baseline report on “Integrated monitoring of dioxins & PCBs in the Baltic Region” in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 09 Jan 2004
Tysklind M et al. 1993. Atmospheric transport and transformation of polychlorinated dibenzo-p-dioxins and dibenzofurans. Environ Sci Technol. 27:2190
UNEP. 2005. Standardized toolkit for the identification and quantification of dioxin and furan releases. 2nd ed., UNEP Chemicals, Geneva. www.pops.int
Upton AC. 1994. Science and judgment in dioxin risk assessment: Needs and opportunities. Environ Health Perspect. 102:908

USEPA. 2000a. Exposure and human health reassessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Draft final. USEPA, Washington, DC. EPA/600/6-88/005Ca. www.epa.gov/ncea/dei.html
van Birgelen APJM et al. 1995a. Subchronic dose-response study of 2,3,7,8-tetrachlorodibenzo-p-dioxin in female Sprague-Dawley rats. Toxicol Appl Pharmacol. 132;1:1
Van den Berg M et al. 1987. Presence of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans in fish-eating birds and fish from the Netherlands. Arch Environ Contam Toxicol. 16:149
Van den Berg M et al. 1994. The toxicokinetics and metabolism of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) and their relevance for toxicity. Crit Rev Toxicol. 24;1:1
Van der Molen GW et al. 2000. Estimation of dioxin and furan elimination rates with a pharmacokinetic model. J Expo Anal Environ Epidemiol. 10;6 Pt 1:579
van der Plas SA et al. 1998. Toxicokinetics of an environmentally relevant mixture of dioxin-like PHAHs with or without a non-dioxin-like PCB in a semi-chronic exposure study in female Sprague Dawley rats. Chemosphere 37;9-12:1941
Vartiainen T, Hallikainen A. 1995. Polyklooridibentso-p-dioksiinien ja –dibentsofuraanien (PCDD/F) sekä PCB:n kertyminen kirjoloheen käytettäessä silakkaa tai kuivarehua ravintona. Finnish Natl Food Admin, Helsinki 1995. Elintarvikevirasto tutkimuksia (Res Natl Food Admin) 1/1995.
Vartiainen T et al. 1998. Birth weight and sex of children and the correlation to the body burden of PCDDs/PCDFs and PCBs of the mother. Environ Health Perspect. 106;2:61
Vartiainen T et al. 1997b. PCDDs and PCDFs in human milk from two areas of Finland. Chemosphere 34;12:2571
Vaz R. 1995. Average Swedish dietary intakes of organochlorine contaminants via foods of animal origin and their relation to levels in human milk, 1975-90. Food Addit Contam. 12;4:543
Verstraete F. 2002. Development and implementation of an EC strategy on dioxins, furans and dioxin-like PCBs in food and feed. Environ Sci Pollut Res Int. 9;5:297
Verta M et al. 1999a. Organoklooriyhdisteet ja raskasmetallit Kymijoen sedimentissa: esiintyminen, kulkeutuminen, vaikutukset ja terveysriskit. Suomen Ympäristö (The Finnish Environ) 334(1999a)
Verta M et al. 1999b. High concentrations of PCDDs and PCDFs in river Kymijoki sediments, southeastern Finland. Organohalogen Compds. 43:261
Verta M et al. 2003. Continued transport of PCDD/F contaminated sediments from River Kymijoki to the Gulf of Finland, the Baltic Sea. Organohalogen Compds. 61:405-8. CD-ROM, Vol. 2, Section 3
Verta M et al. 2004. Dioxin concentrations in sediments of the Baltic Sea - A preliminary survey of existing data. Organohalogen Compds. 66:1401
Verta M et al. Accepted. Dioxin concentrations in sediments of the Baltic Sea - a survey of existing data. Chemosphere
Verta M et al. Unpublished. Kymijoen sedimentteihin varastoituneet PCDD/F- ja elohopeayhdisteet sekä niiden kulkeutuminen. Draft report, SYKE 27.1.2005
Vikelsøe J. 2002. Dioxins in Danish soil. Organohalogen Compds. 57:373
Vikelsøe J et al. 2005. Role of PCDD/F in deposition for soil, percolate and sediment. Organohalogen Compds. 1170-3. CD-ROM ID 1891
Vulykh N, Shatalov V. 2001. Investigation of dioxin/furan composition in emissions and in environmental media. Selection of congeners for modeling. MSC-E Tech Note 6/2001. MSC-E, Moscow. www.msceast.org/publications
Vuorinen PJ et al. 1998b. Comparisons and temporal trends of organochlorines and heavy metals in fish from the Gulf of Bothnia. Mar Pollut Bull. 36;3:236
Vuorinen PJ et al. 2002. PCDD, PCDF, PCB and thiamine in Baltic herring (Clupea harengus L.) and sprat [Sprattus sprattus (L.)] as a background to the M74 syndrome of Baltic salmon (Salmo salar L.). ICES J Mar Sci. 59;3:480
Vuorinen PJ et al. 2004. Differences in PCDD/F concentrations and patterns in herring (Clupea harengus) from Southern and Northern Baltic Sea. Organohalogen Compds. 52
Vuorinen P et al. 2005. PCDD/F, PCB, PBDE and PCN in six fish species from the Finnish Baltic Sea coastal and inland waters. Organohalogen Compds. 1263-6. CD-ROM ID 1916
Wallberg P et al. 1997. Potential importance of protozoan grazing on the accumulation of polychlorinated biphenyls (PCBs) in the pelagic food web. Hydrobiol. 357:53
Wallberg P et al. 2001. Trophic transfer and passive uptake of a polychlorinated biphenyl in experimental marine microbial communities. Environ Toxicol Chem. 20;10:2158
Wallin E et al. 2003. Intra-individual variations over time for 2,2',4,4',5,5'-hexachlorobiphenyl (CB 153) in relation to consumption of fatty fish from the Baltic Sea. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 1
Wang X et al. 1997. Determination of parameters responsible for pharmacokinetic behavior of TCDD in female Sprague–Dawley rats. Toxicol Appl Pharmacol. 147;1:151
Wania F et al. 1998. A review of processes involved in the exchange of persistent organic pollutants across the air-sea interface. Environ Pollut. 102;1:3
Wania F et al. 2001. A multicompartmental, multi-basin fugacity model describing the fate of PCBs in the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Ecol Studies 148(2001):417. Springer Verlag, Berlin & Heidelberg
Wania F et al. 2000. The POPCYCLING-Baltic model. A non-steady state multicompartment mass balance model for the fate of persistent organic pollutants in the Baltic Sea environment. NILU OR 10/2000. www.scar.utoronto.ca/~wania
Weber R et al. 2002a. Effects of selected metal oxides on dechlorination and destruction of PCDD and PCDF. Chemosphere 46:1247
Wenborn M et al. 1999. Releases of dioxins and furans to land and water in Europe. Final Report Issue 2. Report for LUW Nordrhein-Westfalen, Germany on behalf of EC DG-Environ. Report AEAT-4703. europa.eu.int/comm/environment/dioxin/download.htm
Wiberg K et al. 1992. Analysis of bromo-, chloro- and mixed bromo/chloro-dibenzo-p-dioxins and dibenzofurans in salmon, osprey and human milk. Chemosphere 24:1431
Wodarg D et al. 2004. A baseline study of polychlorinated biphenyl and hexachlorobenzene concentrations in the western Baltic Sea and Baltic Proper. Mar Chem. 87;1-2:23
Zober MA et al. 1992. Morbidity study of extruder personnel with potential exposure to brominated dioxins and furans. I. Results of blood monitoring and immunological tests. Br J Ind Med. 49;8:532
Öberg M et al. 2002b. Subchronic toxicity of Baltic herring oil and its fractions in the rat I: Fractionation and levels of organohalogen pollutants. Pharmacol Toxicol. 91;5:220

References to Chapter 4

Abnet CC et al. 1999a. Transactivation activity of human, zebrafish, and rainbow trout aryl hydrocarbon receptors expressed in COS-7cells: greater insight into species differences in toxic potency of polychlorinated dibenzo-p-dioxin, dibenzofuran, and biphenyl congeners. Toxicol Appl Pharmacol. 159;1:41

Akhtar FZ et al. 2004. Cancer in US Air Force veterans of the Vietnam War. J Occup Environ Med. 46;2:123

Alaluusua S et al. 1996a. Polychlorinated dibenzo-p-dioxins and dibenzofurans via mother's milk may cause developmental defects in the child's teeth. Environ Toxicol Pharmacol. 1;3:193

Alaluusua S et al. 1996b. Developmental dental defects associated with long breast feeding. Eur J Oral Sci. 104;5-6:493

Alaluusua S et al. 1999. Developing teeth as biomarker of dioxin exposure. Lancet 353;9148:206

Alaluusua S et al. 2002. Natal and neonatal teeth in relation to environmental toxicants. Pediatr Res. 52;5:652

Alaluusua S et al. 2004. Developmental dental aberrations after the dioxin accident in Seveso. Environ Health Perspect. 112;13:1313

Albert CM et al. 1998. Fish consumption and risk of sudden cardiac death. JAMA 279;1:23

Amin S et al. 2000. Gestational and lactational exposure to TCDD or coplanar PCBs alters adult expression of saccharin preference behavior in female rats. Neurotoxicol Teratol. 22;5:675

Anderson HA et al. 1998. Profiles of Great Lakes critical pollutants: a sentinel analysis of human blood and urine. The Great Lakes Consortium. Environ Health Perspect. 106;5:279

Annas A et al. 2000. CYP1A-dependent activation of xenobiotics in endothelial linings of the chorioallantoic membrane (CAM) in birds. Arch Toxicol. 74;6:335

Aoki Y. 2001. Polychlorinated biphenyls, polychlorinated dibenzo-p-dioxins, and polychlorinated dibenzofurans as endocrine disrupters--what we have learned from Yusho disease. Environ Res. 86;1:2

Aro E. 2000. The spatial and temporal distribution patterns of cod (Gadus morhua callaris L.) in the Baltic Sea and their dependence on environmental variability – implications for fishery management. PhD Thesis, Univ Helsinki, Finland. Finn Game Fisheries Res Inst, Helsinki

Arvola L et al. 2004. The effects of climate and landuse on TOC concentrations and loads in Finnish rivers. Boreal Environ Res. 5;9:381

Ascherio A et al. 1995. Dietary intake of marine n-3 fatty acids, fish intake, and risk of coronary heart disease among men. N Engl J Med. 332:977

Asplund L et al. 1994. Polychlorinated biphenyls, 1,1,1-trichloro-2,2-bis(p-chlorophenyl)ethane (p,p'-DDT) and 1,1-dichloro-2,2-bis(p-chlorophenyl)-ethylene (p,p'-DDE) in human plasma related to fish consumption. Arch Environ Health 49;6:477

Asplund L et al. 1999. Organohalogen substances in muscle, egg and blood from healthy Baltic salmon (Salmo salar) and Baltic salmon that produced offspring with the M74 syndrome. Ambio 28:67

ATSDR. 1998. Toxicological profile for polychlorinated biphenyls. Agency Toxic Substances Disease Registry, U.S. Dept Health Human Services, Atlanta, Georgia, Dec. 1998. www.atsdr.cdc.gov/

Aulerich RJ, Ringer R. 1977. Current status of PCB toxicity to mink, and after-effects on their reproduction. Arch Environ Contam Toxicol. 6:279

Aulerich RJ et al. 1987. Toxicity of 3,4,5,3',4',5'-hexachlorobiphenyl to mink. Arch Environ Contam Toxicol. 16;1:53

Aulerich RJ et al. 1988. Biological effects of epidermal growth factor and 2,3,7,8-tetrachlorodibenzo-p-dioxin on developmental parameters of neonatal mink. Arch Environ Contam Toxicol. 17;1:27

Aulerich RJ et al. 2001. Dietary exposure to 3,3',4,4',5-pentachlorobiphenyl (PCB 126) or 2,3,7,8-tetrachlorodibenzo-p-dixon (TCDD) does not induce proliferation of squamous epithelium or osteolysis in the jaws of weanling rats. Vet Hum Toxicol. 43;3:170

Axmon A et al. 2000a. Miscarriages and stillbirths in women with a high intake of fish contaminated with persistent organochlorine compounds. Int Arch Occup Environ Health 73;3:204

Axmon A et al. 2000b. Time to pregnancy and infertility among women with a high intake of fish contaminated with persistent organochlorine compounds. Scand J Work Environ Health 26;3:199

Axmon A et al. 2001. Polychlorinated biphenyls in blood plasma among Swedish female fish consumers in relation to time to pregnancy. J Toxicol Environ Health A 64;6:485

Axmon A et al. 2002. Female fertility in relation to the consumption of fish contaminated with persistent organochlorine compounds. Scand J Work Environ Health 28;2:124-

Axmon A et al. 2004a. Altered menstrual cycles in women with a high dietary intake of persistent organochlorine compounds. Chemosphere 56;8:813

Axmon A et al. 2004b. Polychlorinated biphenyls in serum and time to pregnancy. Environ Res. 96;2:186

Baccarelli A et al. 2002. Immunologic effects of dioxin: new results from Seveso and comparison with other studies. Environ Health Perspect. 110;12:1169

Bang HO et al. 1971. Plasma lipid and lipoprotein pattern in Greenlandic West-coast Eskimos. Lancet 1;7710:1143-5

Barrow LL et al. 2002. Aryl hydrocarbon receptor nuclear translocator 2 (ARNT2): structure, gene mapping, polymorphisms, and candidate evaluation for human orofacial clefts. Teratol. 66;2:85

Bengtsson B-E et al. 1999. Reproductive disturbances in Baltic fish: A synopsis of the FiRe project. Ambio 28;1:2

Bergman A. 1999. Health condition of the Baltic grey seal (Halichoerus grypus) during two decades. Gynaecological health improvement but increased prevalence of colonic ulcers. APMIS 107:270

Bergman A, Olsson M. 1986. Pathology of Baltic grey seal and ringed seal females with special reference to adrenocortical hyperplasia: Is environmental pollution the cause of a widely distributed disease syndrome? Finn Game Res. 44:47

Bergman A et al. 1992a. Skull-bone lesions in the Baltic grey seal (Halichoerus grypus). Ambio 21;8:517

Bergman A et al. 1992b. Influence of commercial polychlorinated biphenyls and fractions thereof on liver histology in female mink (Mustela vison). Ambio 21;8:591

Bergman A et al. 2001. Renal lesions in Baltic grey seals (Halichoerus grypus) and ringed seals (Phoca hispida botnica). Ambio 30;7:397

Bergman Å et al. 1992c. PCB and PCB methyl sulfones in mink treated with PCB and various PCB fractions. Ambio 21;8:570

Berkers JA et al. 1995. Interactive effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin and retinoids on proliferation and differentiation in cultured human keratinocytes: quantification of cross-linked envelope formation. Arch Toxicol. 69;6:368

Bertazzi PA et al. 2001. Health effects of dioxin exposure: a 20-year mortality study. Am J Epidemiol. 153;11:1031

Besselink HT et al. 1998. Low inducibility of CYP1A activity by polychlorinated biphenyls (PCBs) in flounder (Platichthys flesus): Characterization of the Ah receptor and the role of CYP1A inhibition. Toxicol Sci. 43;2:161
Birnbaum LS. 1995a. Developmental effects of dioxins. Environ Health Perspect. 103;Suppl 7:89
Birnbaum LS. 1995b. Developmental effects of dioxins and related endocrine disrupting chemicals. Toxicol Lett. 82-83:743
Birnbaum LS, Tuomisto J. 2000. Non-carcinogenic effects of TCDD in animals. Food Addit Contam. 17;4:275
Birnbaum LS et al. 1987a. Teratogenicity of three polychlorinated dibenzofurans in C57BL/6N mice. Toxicol Appl Pharmacol. 90;2:206
Birnbaum LS et al. 1987b. Teratogenic effects of polychlorinated dibenzofurans in combination in C57BL/6N mice. Toxicol Appl Pharmacol. 91;2:246
Birnbaum LS et al. 1991. Teratogenic effects of 2,3,7,8-tetrabromodibenzo-p-dioxin and three polybrominated dibenzofurans in C57BL/6N mice. Toxicol Appl Pharmacol. 107;1:141
Birnbaum LS et al. 2003. Health effects of polybrominated dibenzo-p-dioxins (PBDDs) and dibenzofurans (PBDFs). Environ Int. 29;6:855
Bjerke DL et al. 1994. Partial demasculinization and feminization of sex behavior in male rats by in utero and lactational exposure to 2,3,7,8-tetra chlorodibenzo-p-dioxin is not associated with alterations in estrogen receptor binding or volumes of sexually differentiated brain nuclei. Toxicol Appl Pharmacol. 127;2:258
Bjerselius R et al. 2002b. PCDD/PCDF contribute with half of the total TEq found in fatty fish from the Baltic Sea. Organohalogen Compds. 57:209
Bjørnstad ON, Grenfell BT. 2001. Noisy clockwork: Time series analysis of population fluctuations and animals. Science 293:638
Boersma ER, Lanting CI. 2000. Environmental exposure to polychlorinated biphenyls (PCBs) and dioxins. Consequences for longterm neurological and cognitive development of the child lactation. Adv Exp Med Biol. 478:271
Bol J et al. 1989. Interactive effects of PCDD's, PCDF's and PCB's as assessed by the E.L.S.-bioassay. Chemosphere 19;1-6:899
Bonvalot Y et al. 1989. Uncertainty in quantitative carcinogenic risk assessment procedures for 2,3,7,8-tetrachlorodibenzo-p-dioxin. Chemosphere 19;1-6:623
Bookstaff RC et al. 1990a. Altered regulation of pituitary gonadotropin-releasing hormone (GnRH) receptor number and pituitary responsiveness to GnRH in 2,3,7,8-tetrachlorodibenzo-p-dioxin-treated male rats. Toxicol Appl Pharmacol. 105;1:78
Bookstaff RC et al. 1990b. 2,3,7,8-tetrachlorodibenzo-p-dioxin increases the potency of androgens and estrogens as feedback inhibitors of luteinizing hormone secretion in male rats. Toxicol Appl Pharmacol. 104;2:212
Boroush M, Gough M. 1994. Can cohort studies detect any human cancer excess that may result from exposure to dioxin? Maybe. Regulat Toxicol Pharmacol. 20;2:198
Bosveld ATC et al. 2000. Biochemical and developmental effects of dietary exposure to polychlorinated biphenyls 126 and 153 in common tern chicks (Sterna hirundo). Environ Toxicol Chem. 19:719
Bouwman CA et al. 1999. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin or 2,2',4,4',5,5'-hexachlorobiphenyl on vitamin K-dependent blood coagulation in male and female WAG/Rij-rats. Chemosphere 38;3:489
Bowman RE et al. 1989a. Behavioral effects in monkeys exposed to 2,3,7,8-TCDD transmitted maternally during gestation and for four months nursing. Chemosphere 18;1-6:235
Brandt I et al. 1992. Comparative studies on adrenocorticolytic DDT-metabolites. Ambio 21;8:602
Breitholtz M et al. 2001. Toxic substances and reproductive disorders in Baltic fish and crustaceans. Ambio 30;4-5:210
Brewster DW, Matsumura F. 1989. Differential effect of 2,3,7,8-tetrachlorodibenzo-p-dioxin on adipose tissue lipoprotein lipase activity in the guinea pig, rat, hamster, rabbit, and mink. Comp Biochem Physiol C 93;1:49
Brewster DW et al. 1987. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin on guinea pig heart muscle. Toxicol Appl Pharmacol. 89;3:408
Brewster DW et al. 1988a. Toxicity and disposition of 2,3,4,7,8-pentachlorodibenzofuran (4PeCDF) in the rhesus monkey (Macaca mulatta). Toxicol Appl Pharmacol. 93;2:231
Brewster DW et al. 1988b. Rabbit serum hypertriglyceridemia after administration of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). J Toxicol Environ Health 25;4:495
Brouwer A et al. 1989. Polychlorinated biphenyl (PCB)-contaminated fish induces vitamin A and thyroid hormone deficiency in the common seal Phoca vitulina. Aquat Toxicol. 15:99
Brouwer A et al. 1995. Functional aspects of developmental toxicity of polyhalogenated aromatic hydrocarbons in experimental animals and human infants. Eur J Pharmacol. 293;1:1
Brouwer A et al. 1998a. Interactions of persistent environmental organohalogens with the thyroid hormone system: mechanisms and possible consequences for animal and human health. Toxicol Ind Health 14;1-2:59
Brouwer A et al. 1998b. Report of the WHO working group on the assessment of health risks for human infants from exposure to PCDDs, PCDFs and PCBs. Chemosphere 37;9-12:1627
Brouwer A et al. 1999. Characterization of potential endocrine-related health effects at low-dose levels of exposure to PCBs. Environ Health Perspect. 107 Suppl 4:639
Brown DJ et al. 2004. Analysis of Ah receptor pathway activation by brominated flame retardants. Chemosphere 55;11:1509
Brown SB et al. 1998. Biochemical and histological responses in rainbow trout (Oncorhynchus mykiss) exposed to 2,3,4,7,8-pentachlorodibenzofuran. Environ Toxicol Chem. 17;5:915
Brown SB et al. 2004. Altered thyroid status in lake trout (Salvelinus namaycush) exposed to co-planar 3,3',4,4',5-pentachlorobiphenyl. Aquat Toxicol. 67;1:75
Brunström B, Halldin K. 2000. Ecotoxicological risk assessment of environmental pollutants in the Arctic. Toxicol Lett. 112-113:111
Brunström B et al. 1990. Embryotoxicity of polycyclic aromatic hydrocarbons (PAHs) in three domestic avian species, and of PAHs and coplanar polychlorinated biphenyls (PCBs) in the common eider. Environ Pollut. 67;2:133
Brunström B et al. 1991b. Effects of a technical PCB preparation and fractions thereof on ethoxyresorufin O-deethylase activity, vitamin A levels and thymic development in the mink (Mustela vison). Pharmacol Toxicol. 69;6:421
Brunström B et al. 1995. EROD induction in cultured chick embryo liver: A sensitive bioassay for dioxin-like environmental pollutants. Environ Toxicol Chem. 14;5:837
Brunström B et al. 2001. Reproductive toxicity in mink (Mustela vison) chronically exposed to environmentally relevant polychlorinated biphenyl concentrations. Environ Toxicol Chem. 20;10:2318
Buck GM et al. 1999. Paternal Lake Ontario fish consumption and risk of conception delay, New York State Angler Cohort. Environ Res. 80;2 Pt 2: S13

Buck GM et al. 2000. Parental consumption of contaminated sport fish from Lake Ontario and predicted fecundability. Epidemiol. 11;4:388
Burdge GC. 1998. The role of docosahexaenoic acid in brain development and fetal alcohol syndrome. Biochem Soc Trans. 26:246
Burleson GR et al. 1996. Effect of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on influenza virus host resistance in mice. Fundam Appl Toxicol. 29;1:40
Byers T. 1999. The role of epidemiology in developing nutritional recommendations: past, present, and future. Am J Clin Nutr. 69;6:1304S
Bylund G et al. 2000. Proliferative and neoplastic conditions of flounder (Platichthys flesus) in the northern Baltic Sea in relation to DNA adducts. Mar Environ Res. 50;1-5:434
Bäcklin B-M, Bergman Å. 1992. Morphological aspects on the reproductive organs in female mink (Mustela vison) exposed to polychlorinated biphenyls and fractions thereof. Ambio 21;8:598
Bäcklin B-M et al. 2003. Proliferative effects of estradiol, progesterone, and two CB congeners and their metabolites on gray seal (Halichoerus grypus) uterine myocytes in vitro. Toxicol Sci. 75;1:154
Calabrese EJ, Baldwin LA. 1998. Can the concept of hormesis be generalized to carcinogenesis? Regul Toxicol Pharmacol. 28;3:230
Calabrese EJ, Baldwin LA. 2002. Hormesis and high-risk groups. Regul Toxicol Pharmacol. 35;3:414
Calabrese EJ, Baldwin LA. 2003. The hormetic dose-response model is more common than the threshold model in toxicology. Toxicol Sci. 71:246
Calabrese EJ et al. 1999. Hormesis: a highly generalizable and reproducible phenomenon with important implications for risk assessment. Risk Anal. 19;2:261
Calvert GM et al. 1999. Evaluation of diabetes mellitus, serum glucose, and thyroid function among United States workers exposed to 2,3,7,8-tetra chlorodibenzo-p-dioxin. Occup Environ Med. 56;4:270
Chang KJ et al. 1982. Immunologic evaluation of patients with polychlorinated biphenyl poisoning: evaluation of delayed-type skin hypersensitive response and its relation to clinical studies. J Toxicol Environ Health 9;2:217
Chao WY et al. 1997. Middle-ear disease in children exposed prenatally to polychlorinated biphenyls and polychlorinated dibenzofurans. Arch Environ Health 52;4:257
Chen YJ, Hsu CC. 1994. Effects of prenatal exposure to PCBs on the neurological function of children: a neuropsychological and neurophysiological study. Dev Med Child Neurol. 36;4:312
Chiba I et al. 2001. Negative correlation between plasma thyroid hormone levels and chlorinated hydrocarbon levels accumulated in seals from the coast of Hokkaido, Japan. Environ Toxicol Chem. 20;5:1092
Chiu CC et al. 2003. Omega-3 fatty acids for depression in pregnancy. Am J Psychiatry 160;2:385
Clark DA et al. 1983. Cellular and genetic basis for suppression of cytotoxic T cell generation by haloaromatic hydrocarbons. Immunopharmacol. 6;2:143
Cleland LG et al. 2003. The role of fish oils in the treatment of rheumatoid arthritis. Drugs 63;9:845
Cohen GM et al. 1979. Anticarcinogenic effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin on benzo(a)pyrene and 7,12-dimethylbenz(a)anthracene tumor initiation and its relationship to DNA binding. Cancer Res. 39;10:4027
Cook PM et al. 2003. Effects of aryl hydrocarbon receptor-mediated early life stage toxicity on lake trout populations in Lake Ontario during the 20th century. Environ Sci Technol. 37;17:3864
Courval JM et al. 1999. Sport-caught fish consumption and conception delay in licensed Michigan anglers. Environ Res. 80;2:S183
Crump KS et al. 2003. Meta-analysis of dioxin cancer dose-response for three occupational cohorts. Environ Health Perspect. 111:681
Dallaire F et al. 2004. Acute infections and environmental exposure to organochlorines in Inuit infants from Nunavik. Environ Health Perspect. 112;14:1359
Dallinga JW et al. 2002. Decreased human semen quality and organochlorine compounds in blood. Hum Reprod. 17;8:1973
Dar E et al. 1992. Fish consumption and reproductive outcomes in Green Bay, Wisconsin. Environ Res. 59;1:189
Darnerud PO. 2003. Toxic effects of brominated flame retardants in man and in wildlife. Environ Int. 29:6:841
Darnerud PO et al. 1986. 3,3',4,4'-tetrachloro[14C]biphenyl in pregnant mice: enrichment of phenol and methyl sulphone metabolites in late gestational fetuses. Xenobiotica 16;4:295
Davis D, Safe S. 1991. Halogenated aryl hydrocarbon-induced suppression of the in vitro plaque-forming cell response to sheep red blood cells is not dependent on the Ah receptor. Immunopharmacol. 21;3(1991):183
de Boer J et al. 1993. Non-ortho and mono-ortho substituted chlorobiphenyls and chlorinated dibenzo-p-dioxins and dibenzofurans in marine and freshwater fish and shellfish from The Netherlands. Chemosphere 26;10:1823
de Deckere EA 1999. Possible beneficial effect of fish and fish n-3 polyunsaturated fatty acids in breast and colorectal cancer. Eur J Cancer Prev. 8;3:213
De Felip E et al. 2004. Dioxin-like compounds and endometriosis: a study on Italian and Belgian women of reproductive age. Toxicol Lett. 150;2:203
de Heer C et al. 1995. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) to the human thymus after implantation in SCID mice. Toxicol Appl Pharmacol. 134;2:296
Demers A et al. 2002. Plasma concentrations of polychlorinated biphenyls and the risk of breast cancer: a congener-specific analysis. Am J Epidemiol. 155;7:629
Den Hond E et al. 2002. Sexual maturation in relation to polychlorinated aromatic hydrocarbons: Sharpe and Skakkebaek's hypothesis revisited. Environ Health Perspect. 110:771
de Roode DF et al. 2002. Embryotoxic potential of persistent organic pollutants extracted from tissues of guillemots (Uria aalge) from the Baltic Sea and the Atlantic Ocean. Environ Toxicol Chem. 21;11:2401
Despres C et al. 2005. Neuromotor functions in Inuit preschool children exposed to Pb, PCBs, and Hg. Neurotoxicol Teratol. 27;2:245
de Swart RL et al. 1994. Impairment of immune function in harbour seals (Phoca vitulina) feeding on fish from polluted waters. Ambio 23:155
de Swart RL et al. 1996a. Impaired immunity in harbour seals (Phoca vitulina) exposed to bioaccumulated environmental contaminants: review of a long-term feeding study. Environ Health Perspect. 104 Suppl 4:823
DeVito MJ et al. 1995. Comparisons of estimated human body burdens of dioxinlike chemicals and TCDD body burdens in experimentally exposed animals. Environ Health Perspect. 103;9:820
de Voogt P et al. 2001. Do polychlorinated biphenyls contribute to reproduction effects in fish-eating birds? Environ Toxicol Chem. 20;6:1149-51
Dewailly E et al. 2000. Susceptibility to infections and immune status in Inuit infants exposed to organochlorines. Environ Health Perspect. 108;3:205

Dickerson R et al. 1990. The structure-dependent effects of heptachlorodibenzofuran isomers in male C57BL/6 mice: immunotoxicity and mono-oxygenase enzyme induction. Fundam Appl Toxicol. 15;2:298
Diliberto JJ et al. 2002. Using tissue dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) as a predictive response for reversible biochemical changes. Organohalogen Compds. 55:171
Downs TJ, Ambrose RF. 2001. Syntropic ecotoxicology: A heuristic model for understanding the vulnerability of ecological systems to stress. Ecosystem Health 7;4:266
Dyerberg J et al. 1978. Eicosopentaenoic acid and prevention of thrombosis and atherosclerosis. Lancet 2:117
Dyerberg J et al. 2004. Effects of trans- and n-3 unsaturated fatty acids on cardiovascular risk markers in healthy males. An 8 weeks dietary intervention study. Eur J Clin Nutr. 58;7:1062
EC. 2003a. Technical guidance document in support of commission directive 93/67/EEC on risk assessment for new notified substances and commission regulation (EC) No 1488/94 on risk assessment for existing substances. 2nd ed. JRC, Ispra. 4 vol. EUR 20418 EN/1-IV. www.ecb.jrc.it/
Edqvist L-E et al. 1992. Biochemical blood parameters in pregnant mink fed PCB and fractions of PCB. Ambio 21;8:577
Elliott JE et al. 1996. Biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans and biphenyls in bald eagle (Haliaeetus leucocephalus) chicks. Environ Toxicol Chem. 15:782
Elliott JE et al. 2001a. Assessment of biological effects of chlorinated hydrocarbons in osprey chicks. Environ Toxicol Chem. 20;4:866
Eskenazi B et al. 2002a. Serum dioxin concentrations and menstrual cycle characteristics. Am J Epidemiol. 156;4:383
Eskenazi B et al. 2002b. Serum dioxin concentrations and endometriosis: a cohort study in Seveso, Italy. Environ Health Perspect. 110;7:629
Eskenazi B et al. 2003. Maternal serum dioxin levels and birth outcomes in women of Seveso, Italy. Environ Health Perspect. 111;7:947
Evans JS et al. 1994. Use of probabilistic expert judgment in uncertainty analysis of carcinogenic potency. Regul Toxicol Pharmacol. 20;1 Pt 1:15
Fair PA, Becker PR. 2000. Review of stress in marine mammals. J Aquat Ecosyst Stress Recovery 7;4:335
Falandysz J et al. 1996c. [Dioxins and furans in edible species of fish from the Gulf of Gdansk, in Polish with English abstract] Rocz Panstw Zakl Hig. 47;2:197
Falandysz J et al. 2002b. Multivariate analysis of the bioaccumulation of polychlorinated biphenyls (PCBs) in the marine pelagic food web from the southern part of the Baltic Sea, Poland. J Environ Monit. 4;6:929
Fan F et al. 1996. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin on humoral and cell-mediated immunity in Sprague-Dawley rats. Toxicol. 106:221
Faqi AS et al. 1998. Reproductive toxicity and tissue concentrations of low doses of 2,3,7,8-tetrachlorodibenzo-p-dioxin in male offspring of rats exposed throughout pregnancy and lactation. Toxicol Appl Pharmacol. 150;2:383
Fattore E et al. 2000. Relative potency values derived from hepatic vitamin A reduction in male and female Sprague-Dawley rats following subchronic dietary exposure to individual polychlorinated dibenzo-p-dioxin and dibenzofuran congeners and a mixture thereof. Toxicol Appl Pharmacol. 165;3:184
Feeley MM. 1995. Workshop on perinatal exposure to dioxin-like compounds. III. Endocrine effects. Environ Health Perspect. 103 Suppl 2:147
Fingerhut MA et al. 1991. Cancer mortality in workers exposed to 2,3,7,8-tetrachlorodibenzo-p-dioxin. N Engl J Med. 324;4:212
Fisk AT et al. 1997. Accumulation, depuration and hepatic mixed-function oxidase enzyme induction in juvenile rainbow trout and lake whitefish exposed to dietary 2,3,7,8-tetrachlorodibenzo-p-dioxin. Aquat Toxicol. 37;2-3:201
Flinkman J et al. 1998. Changes in northern baltic zooplankton and herring nutrition from 1980s to 1990s: Top-down and bottom-up processes at work. Mar Ecol Prog Ser. 165:127
Fox GA. 1991. Practical causal inference for ecoepidemiologists. J Toxicol Environ Health 33;4(1991):359
Fox GA. 2001. Wildlife as sentinels of human health effects in the Great Lakes--St. Lawrence basin. Environ Health Perspect. 109 Suppl 6:853
Fox LL, Grasman KA. 1999. Effects of PCB 126 on primary immune organ development in chicken embryos. J Toxicol Environ Health A 58;4:233
Fristad RF et al. 2004. Does consumption of different categories of seafood affect birthweight? The HUMIS study. Organohalogen Compds. 66:3252
Fujita H et al. 2002. Characterization of the aryl hydrocarbon receptor repressor gene and association of its Pro185Ala polymorphism with micropenis. Teratol. 65;1:10
Fukusato T et al. 2005. Prenatal and lactational exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) impairs renal development in offspring of rhesus monkeys. Organohalogen Compds. 2540-2. CD-ROM ID 772
Ganther HE. 1999. Selenium metabolism, selenoproteins and mechanisms of cancer prevention: complexities with thioredoxin reductase. Carcinogenesis 20;9:1657
Gastel JA. 2001. Early indicators of response in biologically based risk assessment for nongenotoxic carcionogens. Regul Toxicol Pharmacol. 33;3:393
Gaylor DW, Aylward LL. 2004. An evaluation of benchmark dose methodology for non-cancer continuous-data health effects in animals due to exposures to dioxin (TCDD). Regulat Toxicol Pharmacol. 40:1:9
Gehrs BC, Smialowicz RJ. 1999. Persistent suppression of delayed-type hypersensitivity in adult F344 rats after perinatal exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Toxicol. 134;1:79-88
Gehrs BC et al. 1997. Alterations in the developing immune system of the F344 rat after perinatal exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin: II. Effects on the pup and the adult. Toxicol. 122;3:229. Erratum in: Toxicol. 30;1(1998):71.
Gierthy JF et al. 1993. Correlation of in vitro and in vivo growth suppression of MCF-7 human breast cancer by 2,3,7,8-tetrachlorodibenzo-p-dioxin. Cancer Res. 53;13:3149
Giesy JP et al. 1995. Contaminants of fishes from Great Lakes-influenced sections and above dams of three Michigan rivers: III. Implications for health of bald eagles. Arch Environ Contam Toxicol. 29;3:309
Giesy JP et al 2002. Effects of chronic dietary exposure to environmentally relevant concentrations to 2,3,7,8-tetrachlorodibenzo-p-dioxin on survival, growth, reproduction and biochemical responses of female rainbow trout (Oncorhynchus mykiss). Aquat Toxicol. 59;1-2:35
Gilbertson M. 1989. Effects on fish and wildlife populations. Kimbrough RD, Jensen AA (eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Elsevier Science Publ. BV, 2nd ed
Gilbertson M et al. 1991. Great Lakes embryo mortality, edema, and deformities syndrome (GLEMEDS) in colonial fish-eating birds: similarity to chick-edema disease. J Toxicol Environ Health 33;4:455
Gladen BC et al. 1988b. Development after exposure to polychlorinated biphenyls and dichlorodiphenyl dichloroethene transplacentally and through human milk. J Pediatr. 113;6:991

Glynn AW et al. 2000b. Organochlorines and bone mineral density in Swedish men from the general population. Osteoporos Int. 11;12:1036
Goldstein JA, Safe S. 1989. Mechanism of action and structure-activity relationships for the chlorinated dibenzo-p-dioxins and related compounds. Kimbrough RD, Jensen AA (eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Elsevier Science Publ BV, 2nd ed
Goodman DG, Sauer RM. 1992. Hepatotoxicity and carcinogenicity in female Sprague-Dawley rats treated with 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD): a pathology working group reevaluation. Regul Toxicol Pharmacol. 15;3:245
Grandjean P et al. 2001. Neurobehavioral deficits associated with PCB in 7-year-old children prenatally exposed to seafood neurotoxicants. Neurotoxicol Teratol. 23;4:305
Grasman KA, Fox GA. 2001. Associations between altered immune function and organochlorine contamination in young Caspian terns (Sterna caspia) from Lake Huron, 1997-1999. Ecotoxicol. 10;2:101
Grasman KA et al. 1996. Organochlorine-associated immunosuppression in prefledgling Caspian terns and herring gulls from the Great Lakes: An ecoepidemiological study. Environ Health Perspect Suppl. 104:829
Grasman KA et al. 2000a. Geographic variation in hematological variables in adult and prefledgling herring gulls (Larus argentatus) and possible associations with organochlorine exposure. Arch Environ Contam Toxicol. 38;2:244
Grassman JA et al. 2002. Expression of Cytochromes P450 1A1, 1A2 and 1B1 are correlated with dioxin levels in human liver. Organohalogen Compds. 59:183
Gray LE Jr, Ostby JS. 1995. In utero 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) alters reproductive morphology and function in female rat offspring. Toxicol Appl Pharmacol. 133;2:285
Gray LE et al. 1997a. A dose-response analysis of the reproductive effects of a single gestational dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin in male Long Evans Hooded rat offspring. Toxicol Appl Pharmacol. 146;1:11
Gray LE et al. 1997b. In utero exposure to low doses of 2,3,7,8-tetrachlorodibenzo-p-dioxin alters reproductive development of female Long Evans hooded rat offspring. Toxicol Appl Pharmacol. 146;2:237
Gray LE Jr et al. 1998. The value of mechanistic studies in laboratory animals for the prediction of reproductive effects in wildlife: Endocrine effects on mammalian sexual differentiation. Environ Toxicol Chem. 17;1:109
Greene JF et al. 2003. Basis for a proposed reference dose (RfD) for dioxin of 1-10 pg/kg-day: a weight of evidence evaluation of the human and animal studies. J Toxicol Environ Health B 6;2:115
Grinwis GC et al. 2000b. Toxicity of TCDD in European flounder (Platichthys flesus) with emphasis on histopathology and cytochrome P450 1A induction in several organ systems. Aquat Toxicol. 50;4:387
Grinwis GC et al. 2001. Toxicity of PCB 126 in European flounder (Platichthys flesus) with emphasis on histopathology and cytochrome P4501A induction in several organ systems. Arch Toxicol. 75;2:80
Guiney PD et al. 1997. Correlation of 2,3,7,8-tetrachlorodibenzo-p-dioxin induction of cytochrome P4501A in vascular endothelium with toxicity in early life stages of lake trout. Toxicol Appl Pharmacol. 143;2:256
Guo SW. 2004. The link between exposure to dioxin and endometriosis: a critical reappraisal of primate data. Gynecol Obstet Invest. 57;3:157
Gustafsson PA et al. 2004. Breastfeeding, very long polysaturated fatty acids (PUFA) and IQ at 6½ years of age. Acta Paediatr. 93:1280
Guvenius DM et al. 2003. Human prenatal and postnatal exposure to polybrominated diphenyl ethers, polychlorinated biphenyls, polychlorobiphenylols, and pentachlorophenol. Environ Health Perspect. 111;9:1235
Hagmar L et al. 1995. High consumption of fatty fish from the Baltic Sea is associated with changes in human lymphocyte subset levels. Toxicol Lett. 77;1-3:335
Hagmar L et al. 2001a. Plasma levels of persistent organohalogens and hormone levels in adult male humans. Arch Environ Health 56;2:138
Hagmar L et al. 2001b. Plasma concentrations of persistent organochlorines in relation to thyrotropin and thyroid hormone levels in women. Int Arch Occup Environ Health 74;3:184
Hagmar L et al. 2004a. Persistent organochlorine pollutants and risk for skeletal fractures and impaired bone mineral density in humans – results from the “COMPARE” project. Organohalogen Compds. 66:3508
Hagmar L et al. 2004b. Tidstrender för halter av persistenta klororganiska miljögifter i blod hos vuxna svenska män i relation till konsumtion av fet östersjöfisk. Rapport till Naturvårdsverket – 2004-03-18. www.imm.ki.se/Datavard/PDF/Rapport%20HAEMI%20dioxin.pdf
Hall W. 1986. The Agent Orange controversy after the Evatt Royal Commission. Med J Aust. 145;5:219
Hamm JT et al. 2003. A mixture of dioxins, furans, and non-ortho PCBs based upon consensus toxic equivalency factors produces dioxin-like reproductive effects. Toxicol Sci. 74;1:182
Hanlon PR et al. 2005. Identification of novel TCDD-regulated genes by microarray analysis. Toxicol Appl Pharmacol. 202;3:2158
Hansson S. 1985. Effects of eutrophication on fish communities with special reference to the Baltic Sea – a literature review. Rep Inst Freshw Res Drottningholm 62:36
Haraguchi K et al. 1992. PCB and PCB methyl sulfones in selected groups seals from Swedish waters. Ambio 21;8:546
Hardell L et al. 2004. Concentrations of polychlorinated biphenyls in blood and the risk for testicular cancer. Int J Androl. 27;5:282
Hario M et al. 2000. Polychlorinated biphenyls in diseased lesser black-backed gull (Larus fuscus fuscus) chicks from the Gulf of Finland. Environ Pollut. 107;1:53
Hario M et al. 2004. Organochlorine concentrations in diseased vs. healthy gull chicks from the northern Baltic. Environ Pollut.127;3(2004):411
Harper CR, Jacobson TA. 2003. Beyond the Mediterranean diet: the role of omega-3 Fatty acids in the prevention of coronary heart disease. Prev Cardiol. 6;3:136
Harper N et al. 1993. Immunotoxic potencies of polychlorinated biphenyl (PCB), dibenzofuran (PCDF) and dibenzo-p-dioxin (PCDD) congeners in C57BL/6 and DBA/2 mice. Toxicol. 80;2-3:217
Harper N et al. 1995a. Immunosuppressive activity of polychlorinated biphenyl mixtures and congeners: nonadditive (antagonistic) interactions. Fundam Appl Toxicol. 27;1:131
Harris WS, Isley WL. 2001. Clinical trial evidence for the cardioprotective effects of omega-3 fatty acids. Curr Atheroscler Rep. 3;2:174
Harvey CJ et al. 2003. An ecosystem model of food web and fisheries interactions in the Baltic Sea. ICES J Mar Sci 60;5:939
Hawksley O. 1957. Ecology of breeding population on arctic terns. Bird-Banding 28:57-92 (Ref. by Lemmetyinen and Rantamäki 1980)
Hays SM, Aylward LL. 2003. Dioxin risks in perspective: past, present, and future. Regul Toxicol Pharmacol. 37;2:202
He K et al. 2004. Accumulated evidence on fish consumption and coronary heart disease mortality: a meta-analysis of cohort studies. Circulation 109;2705.

Heaton SN et al. 1995. Dietary exposure of mink to carp from Saginaw Bay, Michigan. 1. Effects on reproduction and survival, and the potential risks to wild mink populations. Arch Environ Contam Toxicol. 28;3:334
Heide-Jørgensen MP et al. 1992. Long-term effects of epizootic in harbor seals in the Kattegat-Skagerrak and adjacent areas. Ambio 21;8:511
Heilier JF et al. 2005. Increased dioxin-like compounds in the serum of women with peritoneal endometriosis and deep endometriotic (adenomyotic) nodules. Fertil Steril. 84;2:305
Heilmann C et al. 2003. Decreased childhood vaccine response in children exposed to PCBs from maternal seafood diet. Organohalogen Compds. 60-65. CD-ROM, Vol. 4, Section 4
Helander B. 2003. Havsörnarna – finns nu längs hela Östersjökusten. Miljötillståndet i Egentliga Östersjön Rapport 2003:20. Swedish report on the environmental state of the Baltic Proper, including English summaries
Helander B et al. 2002. The role of DDE, PCB, coplanar PCB and eggshell parameters for reproduction in the white-tailed sea eagle (Haliaeetus albicilla) in Sweden. Ambio 31;5:386
Helder T. 1980. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on early life stages of the pike (Esox lucius L.). Sci Total Environ. 14;3:255
Helland IB et al. 2003. Maternal supplementation with very-long-chain n-3 fatty acids during pregnancy and lactation augments children's IQ at 4 years of age. Pediatrics 111;1:e39
Helle E et al. 1976. DDT and PCB levels and reproduction in ringed seal from the Bothnian Bay. Ambio 5;4:188
Hendriks AJ. 1995. Modelling equilibrium concentrations of microcontaminants in organisms of the Rhine delta: Can average field residues in the aquatic foodchain be predicted from laboratory accumulation? Aquat Toxicol. 31;1:1
Hendriks AJ, Enserink EL. 1996. Modelling response of single-species populations to microcontaminants as a function of species size with examples for waterfleas (Daphnia magna) and cormorants (Phalacrocorax carbo). Ecol Modell. 88;1-3:247
Hennig B et al. 2002. Proinflammatory properties of coplanar PCBs: in vitro and in vivo evidence. Toxicol Appl Pharmacol. 181;3:174
Hennig B et al. 2005. Proinflammatory mechanisms induced by PCBs: Implications in vascular diseases. Organohalogen Compds. 2329-32. CD-ROM ID 443
Hewitt S et al. 1998. Induction of EROD activity in European eel (Anguilla anguilla) by different polychlorobiphenyls (PCBs). Water Sci Technol. 38;7:245
Hill AB. 1965. The environment and the disease: association or causation. Proc R Soc Med 58:295
Hites RA et al. 2004a. Global assessment of organic contaminants in farmed salmon. Science 303;5655:226
Hoffman DJ et al. 1998. Comparative developmental toxicity of planar PCB congeners in chickens, American kestrels and common terns. Environ Toxicol Chem. 17:747
Hojo R et al. 2002. Sexually dimorphic behavioral responses to prenatal dioxin exposure. Environ Health Perspect. 110;3:247
Holcomb M, Safe S. 1994. Inhibition of 7,12-dimethylbenzanthracene-induced rat mammary tumor growth by 2,3,7,8-tetrachlorodibenzo-p-dioxin. Cancer Lett. 82;1:43
Holsapple MP et al. 1991. A review of 2,3,7,8-tetrachlorodibenzo-p-dioxin-induced changes in immunocompetence: 1991 update. Toxicol. 69;3:219
Hooper L et al. 2004. Omega 3 fatty acids for prevention and treatment of cardiovascular disease. The Cochrane Database Systematic Rev. 4. Art. No.: CD00317. pub2. DOI: 10.1002/14651858.CD00317.pub2
House RV et al. 1990. Examination of immune parameters and host resistance mechanisms in B6C3F1 mice following adult exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin. J Toxicol Environ Health 31;3:203
HS. 11.7.2005. Helsingin Sanomat.
Hu K et al. 1995. Screening assay for dioxin-like compounds based on competitive binding to the murine hepatic Ah receptor. 2. Application to environmental samples. Environ Sci Technol. 29;10:2603
Huisman M et al. 1995a. Perinatal exposure to polychlorinated biphenyls and dioxins and its effect on neonatal neurological development. Early Hum Dev. 41;2:111
Hurst CH et al. 2000b. Tissue disposition of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in maternal and developing Long-Evans rats following subchronic exposure. Toxicol Sci. 57;2:275
Håkansson H, Hanberg A. 1989. The distribution of [14C]-2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and its effect on the vitamin A content in parenchymal and stellate cells of rat liver. J Nutr. 119:573
Håkansson H et al. 1991. In vivo and in vitro toxicity of fractionated fish lipids, with particular regard to their content of chlorinated organic compounds. Pharmacol Toxicol. 69;6:459
Håkansson H et al. 1992. Effects of technical PCB preparations and fractions thereof on vitamin A levels in the mink (Mustela vison). Ambio 21;8:588
ICES. 2003a. Report of the ICES Advisory Committee on Ecocystems, 2003. ICES Cooperat Res Report 262:1
Ilsen A et al. 1996. Signs of enhanced neuromotor maturation in children due to perinatal load with background levels of dioxins. Follow-up until age 2 years and 7 months. Chemosphere 33;7:1317
IPCS. 2001c. Principles for evaluating health risks to reproduction associated with exposure to chemicals. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 225
Isosaari P et al. 2002b. Feeding trial on rainbow trout: comparison of dry fish feed and Baltic herring as a source of PCDD/Fs and PCBs. Chemosphere 48;8:795
Jackson JB et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293;5530:629
Jacobs MN et al. 2004. Time trend investigation of PCBs, PBDEs, and organochlorine pesticides in selected n-3 polyunsaturated fatty acid rich dietary fish oil and vegetable oil supplements; nutritional relevance for human essential n-3 fatty acid requirements. J Agric Food Chem. 52;6:1780
Jacobson JL, Jacobson SW. 1996. Intellectual impairment in children exposed to polychlorinated biphenyls in utero. N Engl J Med. 335;11:783
Jacobson JL, Jacobson SW. 1997. Reply to Middaugh and Egeland. N Engl J Med. 336;9:661
Jacobson JL et al. 1990a. Effects of in utero exposure to polychlorinated biphenyls and related contaminants on cognitive functioning in young children. J Pediatr. 116;1:38
Jacobson JL et al. 1990b. Effects of exposure to PCBs and related compounds on growth and activity in children. Neurotoxicol Teratol. 12:319
Jacobson JL et al. 1992. Effects of prenatal PCB exposure on cognitive processing efficiency and sustained attention. Devel Psychol. 28:297
Jacobson SW et al. 1985. The effect of PCB exposure on visual recognition memory. Child Devel. 56:853
Jacobson SW et al. 1999. Breastfeeding effects on intelligence quotient in 4- and 11-year-old children. Pediatrics 103;5:e71

Jensen S et al. 1969. DDT and PCB in marine animals from Swedish waters. Nature 224:247
Johnson KL et al. 1997. Promotion of endometriosis in mice by polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls. Environ Health Perspect. 105;7:750
Jokinen MP et al. 2003. Increase in cardiovascular pathology in female Sprague-Dawley rats following chronic treatment with 2,3,7,8-tetrachlorodibenzop-dioxin and 3,3',4,4',5-pentachlorobiphenyl. Cardiovasc Toxicol. 3;4:299
Jongbloet PH et al. 2002. Where the boys aren't: dioxin and the sex ratio. Environ Health Perspect.110;1:1
Jämsä T et al. 2001. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin on bone in two rat strains with different aryl hydrocarbon receptor structures. J Bone Miner Res. 16;10:1812
Kahila J. 2005. Selkämeri – vielä puhtainta Itämerta. Ympäristö 6/2005:4.
Kakeyama M, Tohyama C. 2003. Developmental neurotoxicity of dioxin and its related compounds. Ind Health 41;3:215
Kannan N et al. 1995. Chlorobiphenyls: Model compounds for metabolism in food chain organisms and their potential use as ecotoxicological stress indicators by application of the metabolic slope concept. Environ Sci Technol. 29:1851
Karmaus W. 2001. Of jugglers, mechanics, communities, and the thyroid gland: how do we achieve good quality data to improve public health? Environ Health Perspect. 109 Suppl 6:863
Karmaus W, Zhu X. 2004. Maternal concentration of polychlorinated biphenyls and dichlorodiphenyl dichlorethylene and birth weight in Michigan fish eaters: a cohort study. Environ Health 3;1:1
Kattainen H et al. 2001. In utero/lactational 2,3,7,8-tetrachlorodibenzo-p-dioxin exposure impairs molar tooth development in rats. Toxicol Appl Pharmacol. 174;3:216
Kayajanian GM. 2002. The J-shaped dioxin dose response curve. Ecotoxicol Environ Saf. 51;1:1
Keenan RE et al. 1991. Pathology reevaluation of the Kociba et al. (1978) bioassay of 2,3,7,8-TCDD: implications for risk assessment. J Toxicol Environ Health 34;3:279
Kennedy SW et al. 1996. Cytochrome P4501A induction in avian hepatocyte cultures: a promising approach for predicting the sensitivity of avian species to toxic effects of halogenated aromatic hydrocarbons. Toxicol Appl Pharmacol. 141;1:214
Kennedy SW et al. 2003a. Hepatic EROD activity is not a useful biomarker of polychlorinated biphenyl exposure in the adult herring gull (Larus argentatus). Ecotoxicol. 12;1-4:153
Kennedy SW et al. 2003b. Sensitivity of bald eagle (Haliaeetus leucocephalus) hepatocyte cultures to induction of cytochrome P4501A by 2,3,7,8-tetrachlorodibenzo-p-dioxin. Ecotoxicol. 12;1-4:163
Kerkvliet NI, Brauner JA. 1990. Flow cytometric analysis of lymphocyte subpopulations in the spleen and thymus of mice exposed to an acute immunosuppressive dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Environ Res. 52;2:146
Kerkvliet NI et al. 1990. Influence of the Ah locus on the humoral immunotoxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin: evidence for Ah-receptor-dependent and Ah-receptor-independent mechanisms of immunosuppression. Toxicol Appl Pharmacol. 105;1:26
Kerkvliet NI. 2002. Recent advances in understanding the mechanisms of TCDD immunotoxicity. Int Immunopharmacol. 2;2-3:277
Kihlström JE et al. 1992. Effects of PCB and different fractions of PCB on the reproduction of the mink (Mustela vison). Ambio 21;8:563
Kim AH et al. 2002. Impact of physiologically based pharmacokinetic modeling on benchmark dose calculations for TCDD-induced biochemical responses. Regul Toxicol Pharmacol. 36;3:287
Kim HA et al. 2003. Immunotoxicological effects of Agent Orange exposure to the Vietnam War Korean veterans. Ind Health 41;3:158
Kimbrough RD et al. 2001. Analysis of research studying the effects of polychlorinated biphenyls and related chemicals on neurobehavioral development in children. Vet Hum Toxicol. 43;4:220
Kimbrough RD, Krouskas C. 2001. Polychlorinated biphenyls, dibenzo-p-dioxins, and dibenzofurans and birth weight and immune and thyroid function in children. Regul Toxicol Pharmacol. 34;1:42
Kimbrough RD, Krouskas C. 2003. Human exposure to polychlorinated biphenyls and health effects : a critical synopsis. Toxicol Rev. 22;4:217
Kleeman JM et al. 1988. Species differences in 2,3,7,8-tetrachlorodibenzo-p-dioxin toxicity and biotransformation in fish. Fundam Appl Toxicol. 10;2:206
Kociba RJ et al. 1978. Results of a two-year chronic toxicity and oncogenicity study of 2,3,7,8-tetrachlorodibenzo-p-dioxin in rats. Toxicol Appl Pharmacol. 46;2:279
Kodavanti PR et al. 1995. Increased [3H]phorbol ester binding in rat cerebellar granule cells by polychlorinated biphenyl mixtures and congeners: structure-activity relationships. Toxicol Appl Pharmacol. 130;1:140
Kogevinas M. 2001. Human health effects of dioxins: cancer, reproductive and endocrine system effects. Hum Reprod Update 7;3:331
Kogevinas M et al. 1997. Cancer mortality in workers exposed to phenoxy herbicides, chlorophenols, and dioxins. An expanded and updated international cohort study. Am J Epidemiol. 145;12:1061
Kohn MC et al. 1994. The importance of biological realism in dioxin risk assessment models. Risk Anal. 14;6:993
Koistinen J et al. 1995a. PCDEs, PCBs, PCDDs AND PCDFs in black guillemots and white-tailed sea eagles from the Baltic Sea. Chemosphere 30:9:1671
Koistinen J et al. 1997b. 2,3,7,8-Tetrachlorodibenzo-p-dioxin equivalents in extracts of Baltic white-tailed sea eagles. Environ Toxicol Chem. 16;7:1533
Koivusaari J et al. 1972. Decrease in eggshell thickness in the white-tailed sea eagle in Finland during 1884-1971. Ornis Fenn. 49:11
Koivusaari J et al. 1980. Relationships between productivity, eggshell thickness and pollutant contents of addled eggs in the population of white-tailed eagles Haliaëtus albicilla L. in Finland during 1969-1978. Environ Pollut. 23;1:41
Koopman-Esseboom C et al. 1994. Effects of dioxins and polychlorinated-biphenyls on thyroid-hormone status of pregnant women and their infants. Pediatr Res. 36:468
Koopman-Esseboom C et al. 1996. Effects of polychlorinated biphenyl dioxin exposure and feeding type on infants mental and psychomotor development. Pediatrics 97:700
Koopman-Esseboom C et al. 1997. Newborn infants diagnosed as neurologically abnormal with relation to PCB and dioxin exposure and their thyroid-hormone status. Dev Med Child Neurol. 39;11:785
Koppe JG. 1995. Nutrition and breast-feeding. Eur J Obstet Gynecol Reprod Biol. 61;1:73
Kris-Etherton PM et al. 2003. Fish consumption, fish oil, omega-3 fatty acids, and cardiovascular disease. Arterioscler Thromb Vasc Biol. 23;2:e20-30. Erratum in: Arterioscler Thromb Vasc Biol. 23;2:e31. Comment in: Arterioscler Thromb Vasc Biol. 23;2:151
Kromhout D. 2001. Diet and cardiovascular diseases. J Nutr Health Aging 5;3:144

Kromhout D et al. 1985. The inverse relation between fish consumption and 20-year mortality from coronary heart disease. N Engl J Med. 312:1205
Kubiak TJ et al. 1989. Microcontaminants and reproductive impairment of the Forster's tern on Green Bay, Lake Michigan--1983. Arch Environ Contam Toxicol. 18;5:706
Kuchiiwa S et al. 2002. In utero and lactational exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin decreases serotonin-immunoreactive neurons in raphe nuclei of male mouse offspring. Neurosci Lett. 317;2:73
Kuriyama S et al. 2003. Effect of low dose mono-ortho 2,3',4,4',5 pentachlorobiphenyl on thyroid hormone status and EROD activity in rat offspring: consequences for risk assessment. Toxicol. 186;1-2:11
Kuroda M et al. 2005. A dioxin sensitive gene, mammalian WAPL, is implicated in spermatogenesis. FEBS Lett. 579;1:167
Käkelä R et al. 2002a. Effects of gender, diet, exogenous melatonin and subchronic PCB exposure on plasma immunoglobulin G in mink. Comp Biochem Physiol C Toxicol Pharmacol. 132;1:67
König A. et al. 2005. A quantitative analysis of fish consumption and coronary heart disease mortality. Am J Prev Med. 29;4: 335-46
Landergren P et al. 1999. Reproductive failure in Baltic Sea trout (Salmo trutta) compared with the M74 syndrome in Baltic salmon (Salmo salar). Ambio 28;1:87
Landi MT et al. 2005. CYP1A1 and CYP1B1 genotypes, haplotypes, and TCDD-induced gene expression in subjects from Seveso, Italy. Toxicol. 207;2:191
Lang T et al. 1999. Spatial distribution of grossly visible diseases and parasites in flounder (Platichthys flesus) from the Baltic Sea: a synoptic survey. ICES J Mar Sci. 56;2:138
Lans MC et al. 1993. Structure-dependent, competitive interaction of hydroxy-polychlorobiphenyls, -dibenzo-p-dioxins and -dibenzofurans with human transthyretin. Chem Biol Interact. 88;1:7
Lans MC et al. 1994. Different competition of thyroxine binding to transthyretin and thyroxine-binding globulin by hydroxy-PCBs, PCDDs and PCDFs. Eur J Pharmacol. 270;2-3:129
Larson JM et al. 1996. Reproductive success, developmental anomalies, and environmental contaminants in double-crested cormorants (Phalacrocorax auritus). Environ Toxicol Chem. 15:553
Larsson P et al. 1996. Persistent pollutants in a salmon population (Salmo salar) of the southern Baltic Sea. Can J Fish Aquat Sci. 53:62
Lawrence GS, Gobas FA. 1997. A pharmacokinetic analysis of interspecies extrapolation in dioxin risk assessment. Chemosphere 35;3:427-52
Le TN, Johansson A. 2001. Impact of chemical warfare with agent orange on women's reproductive lives in Vietnam: a pilot study. Reprod Health Matters 9;18:156
Lemmetyinen R, Rantamäki P. 1980. DDT and PCB residues in the arctic tern (Sterna paradisaea) nesting in the archipelago of southwestern Finland. Ann Zool Fenn. 7:141
Leonards PE et al. 1998. Studies of bioaccumulation and biotransformation of PCBs in mustelids based on concentration and congener patterns in predators and preys. Arch Environ Contamin Toxicol. 35;4:654
Levin M et al. 2005a. Association between lymphocyte proliferation and polychlorinated biphenyls in free-ranging harbor seal (Phoca vitulina) pups from British Columbia, Canada. Environ Toxicol Chem. 24;5:1247
Levin M et al. 2005b. PCBs and TCDD, alone and in mixtures, modulate marine mammal but not B6C3F1 mouse leukocyte phagocytosis. J Toxicol Environ Health A 68;8:635
Li X et al. 1997. 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) increases release of luteinizing hormone and follicle-stimulating hormone from the pituitary of immature female rats in vivo and in vitro. Toxicol Appl Pharmacol. 142;2:264
Li Z et al. 2004. Fish consumption shifts lipoprotein subfractions to a less atherogenic pattern in humans. J Nutr. 134:1724
Lim Y et al. 2004. Assessment of human health risk of dioxin in Korea. Environ Monit Assess. 92;1-3:211
Lind PM et al. 2000a. Bone tissue composition, dimensions and strength in female rats given an increased dietary level of vitamin A or exposed to 3,3',4, 4',5-pentachlorobiphenyl (PCB126) alone or in combination with vitamin C. Toxicol. 151;1-3:11
Lind PM et al. 2000b. Change of bone tissue composition and impaired bone strength in rats exposed to 3,3',4,4',5-pentachlorobiphenyl (PCB126). Toxicol. 150;1-3:41
Lind PM et al. 2003. Bone mineral density in male Baltic grey seal (Halichoerus grypus). Ambio 32;6:385
Longnecker MP et al. 2000. Correlations among human plasma levels of dioxin-like compounds and polychlorinated biphenyls (PCBs) and implications for epidemiologic studies. Arch Environ Health 55;3:195
Longnecker MP et al. 2001. Polychlorinated biphenyl serum levels in pregnant subjects with diabetes. Diabetes Care 24;6:1099
Loonen H & al. 1996. Ecological hazard assessment of dioxins: hazards to organisms at different levels of aquatic food webs (fish-eating birds and mammals, fish and invertebrates). Sci Total Environ. 182;1-3:93
Lorenzen A et al. 1999. Relationships between environmental organochlorine contaminant residues, plasma corticosterone concentrations, and intermediary metabolic enzyme activities in Great Lakes herring gull embryos. Environ Health Perspect. 107;3:179
Lu Y-C, Wu Y-C. 1985. Clinical findings and immunological abnormalities in Yu-Cheng patients. Environ Health Perspect. 5:17
Lucas M et al. 2004. Gestational age and birth weight in relation to n-3 fatty acids among Inuit (Canada). Lipids 39;7:617
Lucier GW et al. 1990. Placental markers of human exposure to polychlorinated dibenzofurans and polychlorinated biphenyls: implications for risk assessment. IARC Sci Publ. 104:55
Luebke RW et al. 2002. Mortality in dioxin-exposed mice infected with influenza: mitochondrial toxicity (reye's-like syndrome) versus enhanced inflammation as the mode of action. Toxicol Sci. 69;1:109
Luster MI et al. 1980. Examination of bone marrow, immunologic parameters and host susceptibility following pre- and postnatal exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Int J Immunopharmacol. 2;4:301
Ma X et al. 1992. Protein tyrosine phosphorylation as an indicator of 2,3,7,8-tetrachloro-p-dioxin exposure in vivo and in vitro. Biochem Biophys Res Commun. 189;1:59
Mably TA et al. 1992a. In utero and lactational exposure of male rats to 2,3,7,8-tetrachlorodibenzo-p-dioxin. 1. Effects on androgenic status. Toxicol Appl Pharmacol. 114;1:97
Mably TA et al. 1992b. In utero and lactational exposure of male rats to 2,3,7,8-tetrachlorodibenzo-p-dioxin. 2. Effects on sexual behavior and the regulation of luteinizing hormone secretion in adulthood. Toxicol Appl Pharmacol. 114;1:108
Mably TA et al. 1992c. In utero and lactational exposure of male rats to 2,3,7,8-tetrachlorodibenzo-p-dioxin. 3. Effects on spermatogenesis and reproductive capability. Toxicol Appl Pharmacol. 114;1:118

Mac MJ, Edsall CC. 1991. Environmental contaminants and the reproductive success of lake trout in the Great Lakes: an epidemiological approach. J Toxicol Environ Health 33;4:375
MacLusky NJ et al. 1998. Hormonal interactions in the effects of halogenated aromatic hydrocarbons on the developing brain. Toxicol Ind Health 14;1-2:185
Mahajan SS, Rifkind AB. 1999. Transcriptional activation of avian CYP1A4 and CYP1A5 by 2,3,7,8-tetrachlorodibenzo-p-dioxin: Differencec in gene expression and regulation compared to mammalian CYP1A1 and CYP1A2. Toxicol Appl Pharmacol. 155:96
Malcolm CA et al. 2003. Scotopic electroretinogram in term infants born of mothers supplemented with docosahexaenoic acid during pregnancy. Invest Ophthalmol Vis Sci. 44;8:3685
Marchioli R et al. 2002. Early protection against sudden death by n-3 polyunsaturated fatty acids after myocardial infarction: time-course analysis of the results of the Gruppo Italiano per lo Studio della Sopravvivenza nell' Infarto Miocardico (GISSI)-Prevenzione. Circulation 105:1897
Marckmann P, Gronbaek M. 1999. Fish consumption and coronary heart disease mortality. A systemtic review of prospective cohort studies. Eur J Clin Nutr. 53:585
Markowski VP et al. 2001. Altered operant responding for motor reinforcement and the determination of benchmark doses following perinatal exposure to low-level 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environ Health Perspect. 109;6:621
Maruyama W et al. 2004. Dioxin health risk to infants using simulated tissue concentrations. Environ Toxicol. Pharmacol. 18;1:21
Mason CF, Madsen AB. 1993. Organochlorine pesticide residues and PCBs in Danish otters (Lutra lutra). Sci Total Environ. 133;1-2:73
Mason G et al. 1987. Polybrominated dibenzo-p-dioxins and related compounds: Quantitative in vivo and in vitro structure-activity relationships. Toxicol. 44;3:245
Masten SA, Shiverick KT. 1996. Characterization of the aryl hydrocarbon receptor complex in human B lymphocytes: evidence for a distinct nuclear DNA-binding form. Arch Biochem Biophys. 336;2:297
Matsumura F. 2003. On the significance of the role of cellular stress response reactions in the toxic actions of dioxin. Biochem Pharmacol. 66,4:527
Mattson M et al. 1998. Elevated levels of cytochrome P4501A (CYP1A) in ringed seals from the Baltic Sea. Aquat Toxicol. 43;1:41
Mayura K et al. 1993. Teratogenicity and immunotoxicity of 3,3',4,4',5-pentachlorobiphenyl in C57BL/6 mice. Toxicol. 77;1-2:123
McCann K et al. 1998. Weak trophic interactions and the balance of nature. Nature 395:794
McConnell EE. 1989. Acute and chronic toxicity and carcinogenesis in animals. Kimbrough RD, Jensen AA (eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Elsevier Science Publ BV, 2nd ed
McGrath LF et al. 1995. Alternative models for low dose-response analysis of biochemical and immunological endpoints for tetrachlorodibenzo-p-dioxin. Regul Toxicol Pharmacol. 21;3:382
McGregor DB et al. 1998. An IARC evaluation of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans as risk factors in human carcinogenesis. Environ Health Perspect. 106 Suppl 2:755
McGuinness BM et al. 2001. Infecundity and consumption of polychlorinated biphenyl-contaminated fish. Arch Environ Health 56;3:250
McMichael A et al. 1999. Globalization and the sustainability of human health: An ecological perspective. Bioscience 49;3:205
McNabb FM, Fox GA. 2003. Avian thyroid development in chemically contaminated environments: is there evidence of alterations in thyroid function and development? Evol Dev. 5;1:76
Meerts IA et al. 2002. Placental transfer of a hydroxylated polychlorinated biphenyl and effects on fetal and maternal thyroid hormone homeostasis in the rat. Toxicol Sci. 68;2:361
Mendola P et al. 1997. Consumption of PCB-contaminated freshwater fish and shortened menstrual cycle length. Am J Epidemiol. 146;11(1997):955
Middaugh JP, Egeland GM. 1997. Intellectual function of children exposed to polychlorinated biphenyls in utero. Reply to Jacobson and Jacobson. N Engl J Med. 336;9:660
Miettinen HM et al. 2003. In utero and lactational TCDD exposure affects rat bone development. Organohalogen Compds.60-65, Vol. 5, Section 3
Miettinen HM et al. 2005. Effects of in utero and lactational TCDD exposure on bone development in differentially sensitive rat lines. Toxicol Sci. 2005 Mar 2; [Epub ahead of print]
Mocarelli P et al. 1986. Clinical laboratory manifestations of exposure to dioxin in children. A six-year study of the effects of an environmental disaster near Seveso, Italy. J Am Med Assoc. 256;19:2687
Mocarelli P et al. 2000. Paternal concent-rations of dioxin and sex ratio of offspring. Lancet 355;9218:1858
Molgaard C, Michaelsen KF. 2003. Vitamin D and bone health in early life. Proc Nutr Soc. 62;4:823
Mollmann C, Koster FW. 1999. Food consumption by clupeids in the Central Baltic: evidence for top-down control? ICES J Mar Sci. 56;1:100
Mora MA et al. 1993. Polychlorinated biphenyls and chliruinated insecticides in plasma of Caspian terns: Relationships with age, productivity, and colony site tenacity in the Great Lakes. Arch Environ Contam Toxicol. 24:320
Mortensen P et al. 1992. Prevalence of skull lesions in harbour seals (Phoca vitulina) in Swedish and Danish Museum collections: 1835-1988. Ambio 21;8:520
Moshammer H, Neuberger M. 2000. Sex ratio in the children of the Austrian chloracne cohort. Lancet 356;9237:1271
Mundt KA, May S. 2001. Epidemiological assessment of hormesis in studies with low-level exposure. Human Ecol Risk Assess. 7:795 (Ref. by Rodricks)
Murata M et al. 2002. Population-level ecological risk assessment of planar polychlorinated aromatic hydrocarbons in great cormorant (Phalacrocorax carbo) around Tokyo Bay, Japan. Environ Toxicol Chem. 22;10:2508
Murk AJ et al. 1994a. Toxic and biochemical effects of 3,3',4,4'-tetrachlorobiphenyl (CB-77) and clophen A50 on eider duckling (Somateria mollissima) in a semi-field experiment. Environ Pollut. 86;1:21
Murk AJ et al. 1994b. In vitro metabolism of 3,3',4,4'-tetrachlorobiphenyl in relation to ethoxyresorufin-O-deethylase activity in liver microsomes of some wild species and rat. Eur J Pharmacol - Environ Toxicol Pharmacol Sect. 270:253
Murk AJ et al. 1996. Effects of polyhalogenated aromatic hydrocarbons and related contaminants on common tern reproduction: integration of biological, biochemical, and chemical data. Arch Environ Contam Toxicol. 31;1:128
Murray FJ et al. 1979. Three-generation reproduction study of rats given 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in the diet. Toxicol Appl Pharmacol. 50;2:241
Muto T et al. 2002. Mammary gland differentiation in female rats after prenatal exposure to 3,3',4,4',5-pentachlorobiphenyl. Toxicol. 177;2-3:197

Männistö S et al. (eds.) 2003. The national FINDIET 2002 study. Publ Natl Inst Public Health, Nutr Unit, Helsinki 2003. www.ktl.fi/portal/suomi/osiot/ktl_tutkii/tutkimus/ravitsemus
Nafstad P et al. 2003. Asthma and allergic rhinitis at 4 years of age in relation to fish consumption in infancy. J Asthma 40;4:343
Nagai H et al. 2005. Search for the target genes involved in the suppression of antibody production by TCDD in C57BL/6 mice. Int Immunopharmacol. 5;2:331
Nagayama J et al. 1998b. Postnatal exposure to chlorinated dioxins and related chemicals on thyroid hormone status in Japanese breast-fed infants. Chemosphere 37;9-12:1789
Nagayama J et al. 2005a. Higher level of multiple contaminations in breast milk of mothers who gave birth to neonates with congenital hypothyroidism. Organohalogen Compds. 2413-6. CD-ROM ID 1700
Naito W et al. 2004. Evaluation of population-level ecological risks of fish-eating birds to dioxinlike PCBs exposure. Organohalogen Compds. 66:3350
Narasimhan TR et al. 1994. Relative sensitivities of 2,3,7,8-tetrachlorodibenzo-p-dioxin-induced Cyp1a-1 and Cyp1a-2 gene expression and immunotoxicity in female B6C3F1 mice. Fundam Appl Toxicol. 23;4:598
Natunen A et al. 2005. Effects of TCDD on differentiation of osteoblasts derived from rat mesenchymal stem cells. Organohalogen Compds. 2355-7. CD-ROM ID 1632
Ness DK et al. 1993. Effects of perinatal exposure to specific PCB congeners on thyroid hormone concentrations and thyroid histology in the rat. Toxicol Lett. 68;3:311
Neubert D. 1997-98. Reflections on the assessment of the toxicity of “dioxins” for humans, using data from experimental and epidemiological studies. Teratogen Carcinogen Mutagen. 17;4-5:157
Neubert D et al. 1992a. TCDD-toxicity equivalencies for PCDD/PCDF congeners: Prerequisites and limitations. Chemosphere 25;1-2:65
Neubert R et al. 1992b. Polyhalogenated dibenzo-p-dioxins and dibenzofurans and the immune system. 4. Effects of multiple-dose treatment with 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on peripheral lymphocyte subpopulations of a non-human primate (Callithrix jacchus). Arch Toxicol. 66;4:250
Neubert R et al. 1993a. Effects of small doses of dioxins on the immune system of marmosets and rats. Ann N Y Acad Sci. 685:662
Neubert R et al. 1993b. Chlorinated dibenzo-p-dioxins and dibenzofurans and the human immune system. 1. Blood cell receptors in volunteers with moderately increased body burdens. Life Sci. 53;26:1995
Neubert R et al. 1994b. Risk assessment for possible effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related substances on components and functions of the immune system. Exp Clin Immunogenet. 11;2-3:163
Neubert R et al. 2000. Chlorinated dibenzo-p-dioxins and dibenzofurans and the human immune system: 3. Plasma immunoglobulins and cytokines of workers with quantified moderately-increased body burdens. Life Sci. 66;22:2123
Nikolaidis E et al. 1988a. Effects of the TCDD congeners 3,3’,4,4’-tetrachlorobiphenyl and 3,3’,4,4’-tetrachloroazoxybenzene on lymphoid development in the bursa of Fabricius of the chick embryo. Toxicol Appl Pharmacol 92;2:315
Nikolaidis E et al. 1988b. Effects of TCDD and its congeners 3,3’,4,4’-tetrachloroazoxybenzene and 3,3’,4,4’-tetrachlorobiphenyl on lymphoid development in the thymus of avian embryos. Pharmacol Toxicol. 63;5:333
Nisbet RM et al. 1997. Models relating individual and population response to contaminants. Environ Modeling Assess. 2:7
Nishimura N et al. 2002. Immunohistochemical localization of thyroid stimulating hormone induced by a low oral dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin in female Sprague-Dawley rats. Toxicol.171;2-3:73
Nishimura N et al. 2003. Rat thyroid hyperplasia induced by gestational and lactational exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Endocrinol. 144;5:2075
Nishimura N et al. 2005a. Altered thyroxin and retinoid metabolic response to 2,3,7,8-tetrachlorodibenzo-p-dioxin in aryl hydrocarbon receptor-null mice. Arch Toxicol. 79;5:260
Nishimura N et al. 2005b. Disruption of thyroid hormone homeostasis at weaning of Holtzman rats by lactational but not in utero exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Toxicol Sci. 85;1:607
Nohara K et al. 2002. Effect of low-dose 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on influenza A virus-induced mortality in mice. Toxicol. 170;1-2:131
NTP. 1982. Carcinogenesis bioassay of 2,3,7,8-tetrachlorodibenzo-p-dioxin (CAS no. 1746-01-6) in Osborne-Mendel rats and B6C3F1 mice (gavage study). Carcinogenesis testing program, Natl Cancer Inst, NIH, Bethesda, MD and Natl Toxicol Progr, Res Triangle Park, NC. DHHS publ. no. 82
Nyman M et al. 2002. Current levels of DDT, PCB and trace elements in the Baltic ringed seals (Phoca hispida baltica) and grey seals (Halichoerus grypus). Environ Pollut. 119;3:399
Nyman M et al. 2003. Contaminant exposure and effects in Baltic ringed and grey seals as assessed by biomarkers. Mar Environ Res. 55;1:73-99
Oenga GN et al. 2004. TCDD and PCBs inhibit breast cancer cell proliferation in vitro. Toxicol In Vitro 18;6:811
Ohsako S et al. 2001. Maternal exposure to a low dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) suppressed the development of reproductive organs of male rats: dose-dependent increase of mRNA levels of 5alpha-reductase type 2 in contrast to decrease of androgen receptor in the pubertal ventral prostate. Toxicol Sci. 60;1:132
Ohsako S et al. 2002. Developmental stage-specific effects of perinatal 2,3,7,8-tetrachlorodibenzo-p-dioxin exposure on reproductive organs of male rat offspring. Toxicol Sci. 66;2:283
Ohtake F et al. 2003. Modulation of oestrogen receptor signalling by association with the activated dioxin receptor. Nature 423;6939:545
Oken E et al. 2005. Maternal fish consumption, hair mercury, and infant cognition in a US cohort. Environ Health Perspect. DOI: 10.1289/ehp.8041
Olsen SF, Secher HJ. 2002. Low consumption of seafood in early pregnancy as a risk factor for preterm delivery: prospective cohort study. Br Med J. 324:447
Olsen SF et al. 1993. Frequency of seafood intake in pregnancy as a determinant of birth weight: evidence for a dose dependent relationship. J Epidemiol Community Health 47;6:436
Olsen SF et al. 2000. Randomised clinical trials of fish oil supplementation in high risk pregnancies. Fish Oil Trials In Pregnancy (FOTIP) Team. Br J Obstet Gynecol. 107;3:382
Olsman H et al. 2005. Ah-receptor activity induced by brominated and mixed brominated/chlorinated dibenzodioxins in DR-CALUX and RTL-W1 cell lines. Organohalogen Compds. 2005. CD ROM ID 1722

Olsson A et al. 2000b. Nestling blood of the white-tailed sea eagle (Haliaeetus albicilla) as an indicator of territorial exposure to organohalogen compounds - An evaluation. Environ Sci Technol. 34;13:2733
Olsson M et al. 1974. DDT and PCB levels in seals from Swedish waters. The occurrence of aborted seal pups. SNV PM 591:43
Olsson M et al. 1975. DDT and PCB levels in seals from Swedish waters. The occurrence of aborted seal pups. SNV Report PM 900:1 (Ref. in SNV, undated)
Olsson M. 1986. PCBs in the Baltic environment. Wald JS (ed.), PCBs in the environment, Vol. 3. CRC, Boca Raton, FL (Ref. by Brunström et al. 2001)
Olsson M et al. 1992b. Seals and seal protection: Summary and comments. Ambio 21;8:606
Olsson M et al. 1994. Diseases and environmental contaminants in seals from the Baltic and the Swedish west coast. Sci Total Environ. 154;2-3:217
O´Shea TJ. 2000a. PCB´s not to blame. Science 288;5473:1965
O´Shea TJ. 2000b. Cause of seal die-off in 1988 is still under debate. Science 290:1097
Patandin S et al. 1998. Effects of environmental exposure to polychlorinated biphenyls and dioxins on birth size and growth in Dutch children. Pediatr Res. 44;4:538
Paustenbach DJ et al. 1991. Risk assessment of 2,3,7,8-TCDD using a biologically based cancer model: a reevaluation of the Kociba et al. bioassay using 1978 and 1990 histopathology criteria. J Toxicol Environ Health 34;1:11
Pauwels A et al. 2001. The risk of endometriosis and exposure to dioxins and polychlorinated biphenyls: a case-control study of infertile women. Hum Reprod. 16;10:2050
Pavuk M et al. 2003. Serum 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) levels and thyroid function in Air Force veterans of the Vietnam War. Ann Epidemiol. 13;5:335
Pavuk M et al. 2005. Cancer, dioxin, and calendar period of herbicide spraying in veterans of Operation Ranch Hand. Organohalogen Compds. 2410-2. CD-ROM ID 883
Pazdernik TL, Rozman KK. 1985. Effect of thyroidectomy and thyroxine on 2,3,7,8-tetrachlorodibenzo-p-dioxin-induced immunotoxicity. Life Sci. 36;7:695
Peakall DB, Lincer JL. 1996. Do PCBs cause eggshell thinning? Environ Pollut. 91;1:127
Persky V et al. 2001. The effects of PCB exposure and fish consumption on endogenous hormones. Environ Health Perspect. 109;12:1275
Persson C, Stenberg P. 2004. Growth rate of juvenile cormorants Phalacrocorax carbo sinensis in the nestling stage. www.home.swipnet.se/~w-48087/faglar/materialmapp/skarvmapp/cormdev.html
Pesatori AC et al. 1998. Dioxin exposure and non-malignant health effects: a mortality study. Occup Environ Med. 55;2:126
Pesatori AC et al. 2003. Short- and long-term morbidity and mortality in the population exposed to dioxin after the “Seveso accident”. Ind Health 41;3:127
Peterson RE et al. 1993. Developmental and reproductive toxicity of dioxins and related compounds: cross-species comparisons. Crit Rev Toxicol. 23;3:283
Petroff BK et al. 2001. A review of mechanisms controlling ovulation with implications for the anovulatory effects of polychlorinated dibenzo-p-dioxins in rodents. Toxicol. 158;3:91
Pflieger-Bruss S et al. 1995. Effect of PCBs on spermatogenesis. Lancet 346;8981:1040
Pliskova M et al. 2004. AhR- and ER-mediated activities in human blood samples collected from PCB-contaminated and background region in Slovakia. Organohalogen Compds. 66:3580
Pluim HJ et al. 1992. Effects of dioxins on thyroid function in newborn babies. Lancet 339;8804:1303
Pluim HJ et al. 1993. Effects of pre- and postnatal exposure to chlorinated dioxins and furans on human neonatal thyroid hormone concentrations. Environ Health Perspect. 101;6:504
Pohjanvirta R et al. 1999. Physicochemical differences in the AH receptors of the most TCDD-susceptible and the most TCDD-resistant rat strains. Toxicol Appl Pharmacol. 155;1:82
Potischman N, Weed DL. 1999. Causal criteria in nutritional epidemiology. Am J Clin Nutr. 69;6:1309S
Powell DC et al. 1997a. Organochlorine contaminants in double-crested cormorants from Green Bay,Wisconsin: II. Effects of an extract derived from cormorant eggs on the chicken embryo. Arch Environ Contam Toxicol. 32;3:316
Puga A et al. 2000. The transcriptional signature of dioxin in human hepatoma HepG2 cells. Biochem Pharmacol. 60;8:1129
Puga A et al. 2002. Role of the aryl hydrocarbon receptor in cell cycle regulation. Toxicol. 181-182:171
Purkey HE et al. 2004. Hydroxylated polychlorinated biphenyls selectively bind transthyretin in blood and inhibit amyloidogenesis: rationalizing rodent PCB toxicity. Chem Biol. 11;12:1719
Ramamoorthy K et al. 1999. 3,3’4,4’-Tetrachlorobiphenyl exhibits antiestrogenic and antitumorigenic activity in the rodent uterus and mammary cells and in human breast cancer cells. Carcinogenesis 20;1:115
Rattner BA et al. 2004. Contaminant exposure and reproductive success of ospreys (Pandion haliaetus) nesting in Chesapeake Bay regions of concern. Arch Environ Contam Toxicol. 47;1:126
Remillard RB, Bunce NJ. 2002. Linking dioxins to diabetes: epidemiology and biologic plausibility. Environ Health Perspect. 110;9:853
Rhile MJ et al. 1996. Role of Fas apoptosis and MHC genes in 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD)-induced immunotoxicity of T cells. Toxicol. 110;1-3:153
Rice DC. 1999. Behavioral impairment produced by low-level postnatal PCB exposure in monkeys. Environ Res. 80;2:S113
Rice DC, Hayward S. 1999. Effects of exposure to 3,3’,4,4’,5-pentachlorobiphenyl (PCB 126) throughout gestation and lactation on behavior (concurrent random interval-random interval and progressive ratio performance) in rats. Neurotoxicol Teratol. 21;6:679
Rice J. 2001. Implications of variability on many time scales for scientific advice on sustainable management of living marine resources. Prog Oceanogr. 49;1-4:189
Richter CA, vom Saal FS. 2003. Dioxin interacts with estrogen and androgen response systems to disrupt prostate development. Organohalogen Compds. 60-65
Richter CA et al. 1997. An in vitro rainbow trout cell bioassay for aryl hydrocarbon receptor-mediated toxins. Environ Toxicol Chem. 16;3:543
Richthoff J et al. 2003. Serum levels of 2,2’,4,4’,5,5’-hexachlorobiphenyl (CB-153) in relation to markers of reproductive function in young males from the general Swedish population. Environ Health Perspect. 111;4:409

Riecke K et al. 2002. 2,3,7,8-tetrachloro-dibenzo-p-dioxin induces myocardial fibrosis in marmosets (Callithrix jacchus). Organohalogen Compds. 55:315
Riecke K et al. 2003. Effects of 2,3,7,8-TCDD and PCB 126 on human thymic epithelial cells in vitro. Arch Toxicol. 77;6:358
Rier SE. 2002. The potential role of exposure to environmental toxicants in the pathophysiology of endometriosis. Ann N Y Acad Sci. 955:201; discussion 230, 396
Rier SE, Foster WG. 2003. Environmental dioxins and endometriosis. Semin Reprod Med. 21;2:145
Rier SE et al. 1993. Endometriosis in rhesus monkeys (Macaca mulatta) following chronic exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Fundam Appl Toxicol. 21;4:433-41. Comment in: Fundam Appl Toxicol. 23;1(1994):141
Rier SE et al. 2001a. Serum levels of TCDD and dioxin-like chemicals in Rhesus monkeys chronically exposed to dioxin: correlation of increased serum PCB levels with endometriosis. Toxicol Sci. 59;1:147
Rier SE et al. 2001b. Increased tumor necrosis factor-alpha production by peripheral blood leukocytes from TCDD-exposed rhesus monkeys. Toxicol Sci. 60;2:327
Rignell-Hydbom A et al. 2003. Serum levels of 2,2',4,4',5,5'-hexachlorobiphenyl in relation to semen quality and quantity among Swedish fishermen. Organohalogen Compds. 60-65
Rignell-Hydbom A et al. 2004. Exposure to CB-153 and p,p'-DDE and male reproductive function. Hum Reprod. 19;9:2066
Rignell-Hydbom A et al. 2005a. Exposure to PCBs and p,p'-DDE and human sperm chromatin integrity. Environ Health Perspect. 113;2:175
Rignell-Hydbom A et al. 2005b. Exposure to persistent organochlorine pollutants and seminal levels of markers of epididymal and accessory sex gland functions in Swedish men. Hum Reprod. 20;7:1910
Rodricks JV. 2003. Hormesis and toxicological risk assessment. Toxicol Sci. 71:134
Rogan WJ, Gladen BC. 1992. Neurotoxicology of PCBs and related compounds. Neurotoxicol. 13;1:27
Rogan WJ et al. 1988. Congenital poisoning by polychlorinated biphenyls and their contaminants in Taiwan. Science 241;4863:334
Rolland RM. 2000. A review of chemically-induced alterations in thyroid and vitamin A status from field studies of wildlife and fish. J Wildl Dis. 36;4:615
Roos A et al. 2001. The otter (Lutra lutra) in Sweden--population trends in relation to ΣDDT and total PCB concentrations during 1968-99. Environ Pollut. 111;3:457
Rose DP. 1997. Dietary fatty acids and cancer. Am J Clin Nutr. 66;Suppl:998s
Ross PS et al. 1995. Contaminant-related suppression of delayed-type hypersensitivity and antibody responses in harbor seals fed herring from the Baltic Sea. Environ Health Perspect. 103:162
Ross PS et al. 1996a. Suppression of natural killer cell activity in harbour seals (Phoca vitulina) fed Baltic Sea herring. Aquat Toxicol. 34;1:71
Ross PS et al. 1996b. The immunotoxicity of environmental contaminants to marine wildlife: a review. Annu Rev Fish Dis. 6:151
Ross PS et al. 1996c. Contaminant-induced immunotoxicity in harbour seals: Wildlife at risk? Toxicol. 112;2:157
Ross PS et al. 1997. Impaired cellular immune response in rats exposed perinatally to Baltic Sea herring oil or 2,3,7,8-TCDD. Arch Toxicol. 71;9:563
Ross PS et al. 2000. PCBs are a health risk for humans and wildlife. Science 289:1878
Rozman KK, Doull J. 2003. Scientific foundations of hormesis. Part 2. Maturation, strengths, limitations, and possible applications in toxicology, pharmacology, and epidemiology. Crit Rev Toxicol. 33;3-4:451
Ryan JJ et al. 2002. Sex ratios of children of Russian pesticide producers exposed to dioxin. Environ Health Perspect. 110;11:A699
Rychetnik L, Frommer M. 2000. A proposed schema for evaluating evidence for public health interventions. A discussion paper prepared for the National Public Health Partnership. May 2000
Rychetnik L et al. 2002. Criteria for evaluating evidence on public health interventions. J Epidemiol Community Health 56:119
Rylander L, Hagmar L. 2000. Medical and psychometric examinations of conscripts born to mothers with a high intake of fish contaminated with persistent organochlorines. Scand J Work Environ Health 26;3:207
Rylander L et al. 1995. Decreased birthweight among infants born to women with a high dietary intake of fish contaminated with persistent organochlorine compounds. Scand J Work Environ Health 21;5:368
Rylander L et al. 1996. Dietary intake of fish contaminated with persistent organochlorine compounds in relation to low birthweight. Scand J Work Environ Health 22;4:260
Rylander L et al. 1998. Polychlorinated biphenyls in blood plasma among Swedish female fish consumers in relation to low birth weight. Am J Epidemiol. 147;5:493
Rylander L et al. 2000. Lowered birth weight among infants born to women with a high intake of fish contaminated with persistent organochlorine compounds. Chemosphere 40;9-11:1255
Rylander L et al. 2003. Incidence of hospitalized osteoporotic fractures and persistent organochlorine compounds. Organohalogen Compds. 60-65, CD-ROM Vol. 6, Section 2.
Rylander L et al. 2005. Exposure to persistent organochlorine pollutants in relation to weight and height at 4 and 7 years of age. Organohalogen Compds. 2404-6. CD-ROM ID 488
SAB. 1995. A second look at dioxin. USEPA's Sci Advisory Bd. EPA-SAB-EC-95-02 1. Sep 1995.
SACN and COT. 2004. Advice on fish consumtion: benefits & risks. Sci Advisory Committee Nutr and Committee Toxicol. The Stationery Office, London. www.food.gov.uk/multimedia/pdfs/fishreport2004full.pdf
Safe S, Krishnan V. 1995. Chlorinated hydrocarbons: estrogens and antiestrogens. Toxicol Lett. 82-83:731
Sakamoto KQ et al. 2003. Cytochrome P450 induction and gonadal status alteration in common carp (Cyprinus carpio) associated with the discharge of dioxin contaminated effluent to the Hikiji River, Kanagawa Prefecture, Japan. Chemosphere 51;6:491
Sargent JR. 1997. Fish oils and human diet. Br J Nutr. 78 Suppl 1:S5
SCAN. 2000. Opinion of the Scientific Committee on Animal Nutrition on the dioxin contamination of feedingstuffs and their contribution to the contamination of food of animal origin. EC, Brussels. Adopted 06 Nov 2000
SCF. 2001. Opinion of the Scientific Committee for Food on the risk assessment of dioxins and dioxin-like PCBs an food. Update based on new scientific information available. Adopted 30th May 2001. europa.eu.int/comm./food/Fs/sc/scf/out90_en.pdf
Schantz SL, Bowman RE. 1989. Learning in monkeys exposed perinatally to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Neurotoxicol Teratol. 11;1:13
Schantz SL et al. 1989. Effects of perinatal PCB exposure on discrimination-reversal learning in monkeys. Neurotoxicol Teratol. 11;3:243

Schantz SL et al. 1992. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin on behavior of monkeys in peer groups. Neurotoxicol Teratol. 14;6:433
Schantz SL et al. 1996a. Effects of gestational and lactational exposure to TCDD or coplanar PCBs on spatial learning. Neurotoxicol Teratol. 18;3:305
Schantz SL et al. 2001. Impairments of memory and learning in older adults exposed to polychlorinated biphenyls via consumption of Great Lakes fish. Environ Health Perspect. 109;6:605
Schell JD Jr et al. 2001. PCBs and neurodevelopmental effects in Michigan children: An evaluation of exposure and dose characterization. Regul Toxicol Pharmacol. 33;3:300
Schlezinger JJ, Stegeman JJ. 2000. Induction of cytochrome P450 1A in the American eel by model halogenated and non-halogenated aryl hydrocarbon receptor agonists. Aquat Toxicol. 50;4:375
Selmin O et al. 2005. Transcriptional activation of the membrane-bound progesterone receptor (mPR) by dioxin, in endocrine-responsive tissues. Mol Reprod Dev. 70;2:166
Semba RD et al. 1992. Depressed immune response to tetanus in children and vitamin A deficiency. J Nutr. 122:101
Semba RD et al. 1993. Abnormal T-cell subset proportions in vitamin-A-deficient children. Lancet 341;8836:5
Senthil Kumar K et al. 2002b. Distribution and elimination of polychlorinated dibenzo-p-dioxins, dibenzofurans, biphenyls, and p,p'-DDE in tissues of bald eagles from the Upper Peninsula of Michigan. Environ Sci Technol. 36;13:2789
Sharara FI et al. 1998. Environmental toxicants and female reproduction. Fertil Steril. 70;4:613
Sidhu KS. 2003. Health benefits and potential risks related to consumption of fish or fish oil. Regulat Toxicol Pharmacol. 38;3:336
Simms W, Ross PS. 2000. Vitamin A physiology and its application as a biomarker of contaminant-related toxicity in marine mammals: a review. Toxicol Ind Health 16;6:291
Sjöåsen T et al. 1997. The otter (Lutra lutra) situation in Latvia and Sweden related to PCB and DDT levels. Ambio 26:196
Sleiderink HM et al. 1995. Influence of temperature and polyaromatic contaminants on CYP1A levels in North Sea dab (Limanda limanda). Aquat Toxicol. 32;2-3:189
Slovic P. 1998. If hormesis exists...: implications for risk perception and communication. Hum Exp Toxicol. 17;8:439
Smeets JM et al. 1999b. The anti-estrogenicity of Ah receptor agonists in carp (Cyprinus carpio) hepatocytes. Toxicol Sci. 52;2:178
Smuts CM et al. 2003. A randomized trial of docosahexaenoic acid supplementation during the third trimester of pregnancy. Obstet Gynecol. 101;3(2003):469-79.
Sommer RJ et al. 2005. Early developmental 2,3,7,8-tetrachlorodibenzo-p-dioxin exposure decreases chick embryo heart chronotropic response to isoproterenol but not to agents affecting signals downstream of the beta-adrenergic receptor. Toxicol Sci. 83;2:363
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
Starr TB. 2003. Significant issues raised by meta-analyses of cancer mortality and dioxin exposure. Environ Health Perspect. 111:1443
Staskal D et al. 2005. Inhibition of human and rat CYP1A2 by TCDD and dioxin-like chemicals. Toxicol Sci. 2005 Jan 19; [Epub ahead of print]
Steenland K et al. 2004. Dioxin revisited: developments since the 1997 IARC classification of dioxin as a human carcinogen. Environ Health Perspect. 112;13:1265
Stern N et al. 2002. Subchronic toxicity of Baltic herring oil and its fractions in the rat II: Clinical observations and toxicological parameters. Pharmacol Toxicol. 91;5:232
Stern N et al. 2003. TCDD induces trabecular bone loss and bone fragility in a TCDD-sensitive but not in a TCDD-resistant rat strain. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 3
Stern N et al. 2004. Perinatal exposure to a mixture of persistent pollutants based on blood profiles of Arctic populations affects bone parameters in 35 days old rats. Organohalogen Compds. 66:3048
Stewart PW et al. 2003. Cognitive development in preschool children prenatally exposed to PCBs and MeHg. Neurotoxicol Teratol. 25;1:11
Storr-Hansen E, Spliid H. 1993a. Coplanar polychlorinated biphenyl congener levels and patterns and the identification of separate populations of harbor seals (Phoca vitulina) in Denmark. Arch Environ Contam Toxicol. 24;1:44
Studer M et al. 2005. Effect of different antilipidemic agents and diets on mortality: a systematic review. Arch Intern Med. 165;7:725
Su K-P et al. 2003. Omega-3 fatty acids in major depressive disorder. A preliminary double-blind, placebo-controlled trial. Eur Neuropsychopharmacol. 13;4:267
Sumida H et al. 2005. Testes of rhesus monkeys exposed in utero and lactational period to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Organohalogen Compds. 2537-9. CD-ROM ID 738
Susser M. 1977. Judgement and causal inference: criteria in epidemiologic studies. Am J Epidemiol. 105;1:1-15. Reprint: Am J Epidemiol. 141;8(1995):701
Susser M. 1986. Rules of inference in epidemiology. Regul Toxicol Pharmacol. 6;2:116
Susser M. 1991. What is a cause and how do we know one? A grammar for pragmatic epidemiology. Am J Epidemiol. 133;7:635
Suter GW II (ed.). 1993. Ecological risk assessment. Lewis Publ, Boca Raton et al. 1993. 538 p
Suter GW II et al. 2002. A methodology for inferring the causes of observed imnpairments in aquatic ecosystems. Environ Toxicol Chem. 21;6:1101
Svensson BG et al. 1994. Parameters of immunological competence in subjects with high consumption of fish contaminated with persistent organochlorine compounds. Int Arch Occup Environ Health 65;6:351
Svensson BG et al. 1995b. Mortality and cancer incidence among Swedish fishermen with a high dietary intake of persistent organochlorine compounds. Scand J Work Environ Health 21;2:106
ten Tusscher GW, Koppe JG. 2004. Perinatal dioxin exposure and later effects-- a review. Chemosphere 54;9:1329
ten Tusscher GW et al. 2003. Persistent hematologic and immunologic disturbances in 8-year-old Dutch children associated with perinatal dioxin exposure. Environ Health Perspect. 111;12:1519
Terry P et al. 2002. Fatty fish consumption lowers the risk of endometrial cancer: a nationwide case-control study in Sweden. Cancer Epidemiol Biomarkers Prev. 11;1:143
Terry PD et al. 2003. Intakes of fish and marine fatty acids and the risks of cancers of the breast and prostate and of other hormone-related cancers: a review of the epidemiologic evidence. Am J Clin Nutr. 77;3:532
Terry PD et al. 2004. Long-chain (n-3) fatty acid intake and risk of cancers of the breast and the prostate: recent epidemiological studies, biological mechanisms, and directions for future research. J Nutr. 134;12 Suppl:3412S
Thurow F. 1997. Estimation of the total fish biomass in the Baltic Sea during the 20th century. ICES J Mar Sci. 54;3:444

Thyen S et al. 2000. Organochlorine and mercury contamination of little terns (Sterna albifrons) breeding at the western Baltic Sea, 1978-96. Environ Pollut. 108;2: 225
Tiido T et al. 2005. Exposure to persistent organochlorine pollutants associates with human sperm Y:X chromosome ratio. Hum Reprod. 20;7:1903
Tillitt DE et al. 1989. Planar chlorinated hydrocarbons (PCHs) in colonial fish-eating waterbird eggs from the Great Lakes. Mar Environ Res. 28;1-4:505
Tillitt DE et al. 1991. Characterization of the H4IIE rat hepatoma cell bioassay as a tool for assessing toxic potency of planar halogenated hydrocarbons in environmental samples. Environ Sci Technol. 25:87
Tillitt DE et al. 1996. Dietary exposure of mink to carp from Saginaw Bay. 3. Characterization of dietary exposue to planar halogenated hydrocarbons, dioxin equivalents, and biomagnification. Environ Sci Technol. 30:283
Tilson HA, Kodavanti PRS. 1997. Neurochemical effects of polychlorinated biphenyls: An overview and identification of research needs. Neurotoxicol. 18;3:727
Tonn T et al. 1996. Persistence of decreased T-helper cell function in industrial workers 20 years after exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environ Health Perspect. 104;4:422
Tryphonas H et al. 1991a. Effect of chronic exposure of PCB (Aroclor 1254) on specific and nonspecific immune parameters in the rhesus (Macaca mulatta) monkey. Fundam Appl Toxicol. 16:773
Tsukino H et al. 2005. Associations between serum levels of selected organochlorine compounds and endometriosis in infertile Japanese women. Environ Res. 2005 May 28; [Epub ahead of print]
Tuomisto JT et al. 2004a. Soft-tissue sarcoma and dioxin: A case-control study. Int J Cancer 108;6:8930
Uauy-Dagach R, Mena P. 1995. Nutritional role of omega-3 fatty acids during the perinatal period. Clin Perinatol. 22;1:157
Umbreit TH, Gallo MA. 1988. Physiological implications of estrogen receptor modulation by 2,3,7,8-tetrachlorodibenzo-p-dioxin. Toxicol Lett. 42;1:5
USEPA. 2000a. Exposure and human health reassessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Draft final. USEPA, Washington, DC. EPA/600/6-88/005Ca. www.epa.gov/ncea/dei.html
USPSTF. 1996. Guide to clinical preventive services. US Preventive Services Task Force.
Vallin L et al. 1999. Potential factors influencing reproductive success of Baltic cod, Gadus morhua: A review. Ambio 28;1:92
van Birgelen APJM et al. 1995a. Subchronic dose-response study of 2,3,7,8-tetrachlorodibenzo-p-dioxin in female Sprague-Dawley rats. Toxicol Appl Pharmacol. 132;1:1
van Birgelen AP et al. 1995b. Subchronic effects of 2,3,7,8-TCDD or PCBs on thyroid hormone metabolism: use in risk assessment. Eur J Pharmacol. 293;1:77
van Birgelen APJM et al. 1996a. Relative potencies of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls derived from hepatic porphyrin accumulation in mice. Toxicol Appl Pharmacol. 138;1:98
Van den Berg KJ et al. 1988. Effects of 3,4,3',4'-tetrachlorobiphenyl on thyroid function and histology in marmoset monkeys. Toxicol Lett. 41;1:77
Van den Berg M et al. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ Health Perspect. 106:775
Van Den Heuvel RL et al. 2002. Immunologic biomarkers in relation to exposure markers of PCBs and dioxins in Flemish adolescents (Belgium). Environ Health Perspect. 110;6:595
van der Plas SA et al. 2001. Effects of subchronic exposure to complex mixtures of dioxin-like and non-dioxin-like polyhalogenated aromatic compounds on thyroid hormone and vitamin A levels in female Sprague-Dawley rats. Toxicol Sci. 59;1:92
van der Weiden MEJ et al. 1993. Induction of Cytochrome P450 1A in fish treated with 2,3,7,8-tetrachlorodibenzo-p-dioxin or chemically contaminated sediment. Environ Toxicol Chem. 12:989
van der Weiden MEJ et al. 1994a. Concurrence of P450 1A induction and toxic effects in the mirror carp (Cyprinus carpio), after administration of a low dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Aquat Toxicol. 29;3-4:147
van der Weiden MEJ et al. 1994b. Relative potencies of polychlorinated dibenzo-p-dioxins (PCDDS), dibenzofurans (PCDFS) and biphenyls (PCBS), for cytochrome P450 1A induction in the mirror carp (Cyprinus carpio). Aquat Toxicol. 29;3-4:163
Van Leeuwen FX et al. 2000. Dioxins: WHO's tolerable daily intake (TDI) revisited. Chemosphere 40;9-11:1095
Van Loveren H et al. 2000. Contaminant-induced immunosuppression and mass mortalities among harbor seals. Toxicol Lett. 112-113:319
Van Miller JP et al. 1977. Increased incidence of neoplasms in rats exposed to low levels of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Chemosphere 6;10:625
Vartiainen T et al. 1998. Birth weight and sex of children and the correlation to the body burden of PCDDs/PCDFs and PCBs of the mother. Environ Health Perspect.106;2:61
Vorderstrasse BA et al. 2003. Examining the relationship between impaired host resistance and altered immune function in mice treated with TCDD. Toxicol. 188;1:15
Vorderstrasse BA et al. 2004. A novel effect of dioxin: Exposure during pregnancy severely impairs mammary gland differentiation. Toxicol Sci. 78;2:248
Vos JG et al. 1997-98. Immunotoxic effects of TCDD and toxic equivalency factors. Teratogen Carcinogen Mutagen. 17;4-5:275
Vos JG, Luster MI. 1989. Immune alterations. Kimbrough RD, Jensen AA (eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Elsevier Science Publ BV, 2nd ed
Vos JG, Van Loveren H. 1998. Experimental studies on immunosuppression: how do they predict for man? Toxicol. 129;1:13
Vreugdenhil HJ et al. 2002a. Effects of prenatal PCB and dioxin background exposure on cognitive and motor abilities in Dutch children at school age. J Pediatr. 140;1:48
Vreugdenhil HJ et al. 2002b. Effects of perinatal exposure to PCBs and dioxins on play behavior in Dutch children at school age. Environ Health Perspect. 110;10:A593
Vreugdenhil HJ et al. 2004. Effects of perinatal exposure to PCBs on neuropsychological functions in the Rotterdam cohort at 9 years of age. Neuropsychol. 18;1:185
Vuorinen PJ et al. 1997. The M74 syndrome of Baltic salmon (Salmo salar) and organochlorine concentrations in the muscle of female salmon. Chemosphere 34;5-7:11516
Vuorinen PJ et al. 1998a. Dioxin-like organochlorines in the M74 syndrome of Baltic salmon (Salmo salar L.) Mar Environ Res. 46;1-5:177
Walker MK, Catron TF. 2000. Characterization of cardiotoxicity induced by 2,3,7,8-tetrachlorodibenzo-p-dioxin and related chemicals during early chick embryo development. Toxicol Appl Pharmacol. 167;3:210

Walker MK et al. 1997. Expression of the aryl hydrocarbon receptor (AhR) and AhR nuclear translocator during chick cardiogenesis is consistent with 2,3,7,8-tetrachlorodibenzo-p-dioxin-induced heart defects. Toxicol Appl Pharmacol. 143;2:407
Walter GL et al. 2000. Pathologic alterations in adult rainbow trout, Oncorhynchus mykiss, exposed to dietary 2,3,7,8-tetrachlorodibenzo-p-dioxin. Aquat Toxicol. 50;4:287
Wang C et al. 2004. Effects of omega-3 fatty acids on cardiovascular disease. Agency Healthcare Res Qual., Rockville, MD AHRQ Publ No. 04-E009-2
Warmerdam JM, Greene JF. 2002. Analysis of sex ratios from Seveso and Ranch Hand cohorts using binomial probability. Organohalogen Compds. 59:391
Warner M et al. 2002. Serum dioxin concentrations and breast cancer risk in the Seveso Women's Health Study. Environ Health Perspect. 110;7:625
Warner M, Eskenazi B. 2005. Correspondence: TCDD and puberty: Warner and Eskenazi respond. Environ Health Perspect. 113;1(2005):a18
Warner M et al. 2004. Serum dioxin concentrations and age at menarche. Environ Health Perspect. 112;13:1289
Warngard L et al. 1996. Mechanistical studies of the inhibition of intercellular communication by organochlorine compounds. Arch Toxicol Suppl. 18:149
Weed DL. 2002. Environmental epidemiology: basics and proof of cause-effect. Toxicol. 181-182:399
Weisglas-Kuperus N et al. 2000. Immunologic effects of background exposure to polychlorinated biphenyls and dioxins in Dutch preschool children. Environ Health Perspect. 108;12:1203
Weisskopf MG et al. 2005. Maternal exposure to Great Lakes sport-caught fish and dichlorodiphenyl dichloroethylene, but not polychlorinated biphenyls, is associated with reduced birth weight. Environ Res. 97;2:149
WHO. 1998. Assessment of the health risk of dioxins: re-evaluation of the Tolerable Daily Intake (TDI), executive summary. WHO Consultation, May 25-29 1998, Geneva, Switzerland. WHO, Geneva. http://www.who.int/pcs/dioxin-exec-sum/exe-sum-final.doc
Wiberg K et al. 2002. Concentrations and enantiomer fractions of organochlorine compounds in Baltic species hit by reproductive impairment. Environ Toxicol Chem. 21;12:2542
Williams C et al. 2001. Stereoacuity at age 3.5 y in children born full-term is associated with prenatal and postnatal dietary factors: a report from a population-based cohort study. Am J Clin Nutr. 73:316
Williams DE et al. 1998. Xenobiotics and xenoestrogens in fish: Modulation of cytochrome P450 and carcinogenesis. Mutat Res. 399;2:179
Wolf CJ et al. 1999. Gestational exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) severely alters reproductive function of female hamster offspring. Toxicol Sci. 51;2:259
Wren CD. 1991. Cause-effect linkages between chemicals and populations of mink (Mustela vison) and otter (Lutra canadensis) in the Great Lakes basin. J Toxicol Environ Health 33;4:549
Wright PJ, Tillitt DE. 1999. Embryotoxicity of Great Lakes lake trout extracts to developing rainbow trout. Aquat Toxicol. 47;2:77
Yamaguchi N. 1999. Uncertainty in risk characterization of weak carcinogens. Ann N Y Acad Sci. 895:338
Yamashita N et al. 1991. Effect of eicosapentaenoic and docosahexaenoic acid on natural killer cell activity in human peripheral blood lymphocytes. Clin Immunol Immunopathol. 59,3:335
Yamashita N et al. 1993. Embryonic abnormalities and organochlorine contamination in double-crested cormorants (Phalacrocorax auritus) and Caspian terns (Hydroprogne caspia) from the upper Great Lakes in 1988. Environ Pollut. 79;2:163
Yao Y et al. 1995. Dioxin activates HIV-1 gene expression by an oxidative stress pathway requiring a functional cytochrome P450 CYP1A1 enzyme. Environ Health Perspect. 103:366
Yasuda M et al. 2005 .Defects of the third molar teeth in rhesus monkeys prenatally and lactationally exposed to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Organohalogen Compds. 2543. CD-ROM ID 806.
Yu ML et al. 1998. The immunologic evaluation of the Yucheng children. Chemosphere 37;9-12:1855
Zober MA et al. 1992. Morbidity study of extruder personnel with potential exposure to brominated dioxins and furans. I. Results of blood monitoring and immunological tests. Br J Ind Med. 49;8:532
Åkerman G et al. 1998. Studies with oxythiamine to mimic reproduction disorders among fish early life stages. Mar Environ Res. 46:493 (ref. by Bengtsson BE et al. 1999)

References to Chapter 5

Abraham K et al. 1989. Absorption and tissue distribution of various polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDDs and PCDFs) in the rat. Arch Toxicol. 63;3:193
Ahlborg UG et al. 1994. Toxic equivalency factors for dioxin-like PCBs. Report on a WHO-ECEH and IPCS consultation, Dec 1993. Chemosphere 28:1049
Alcock RE et al. 1998. Dioxin-like PCBs in the environment - human exposure and the significance of sources. Chemosphere 37;8:1457
Alcock RE et al. 2000. A generic model of human lifetime exposure to persistent organic contaminants: development and application to PCB-101. Environ Pollut. 110;2:253
Alder L et al. 1997. Levels of toxaphene indicator compounds in fish. Chemosphere 34;5-7:1389
AMAP. 2002. AMAP assessment 2002: Persistent organic pollutants in the Arctic. Arctic Monitoring and Assessment Progr, Oslo. xvi + 310 p
Anderson HA et al. 1998. Profiles of Great Lakes critical pollutants: a sentinel analysis of human blood and urine. The Great Lakes Consortium. Environ Health Perspect. 106;5:279
Andersson Ö, Wartanian A. 1992. Levels of polychlorinated camphenes (Toxaphene), chlordane compounds and polybrominated diphenyl ethers in seals from Swedish waters. Ambio 21;8:550
Ashizawa AE et al. 2005. Human health research and policy development: experience in the Great Lakes region. Int J Hyg Environ Health 208;1-2:7
Asplund L et al. 1990. Analysis of non-ortho polychlorinated biphenyls and polychlorinated naphthalenes in Swedish dioxin survey samples. Chemosphere 20:1481
Asplund L et al. 1994. Polychlorinated biphenyls, 1,1,1-trichloro-2,2- bis(p-chlorophenyl)ethane (p,p'-DDT) and 1,1-dichloro-2,2-bis (p-chlorophenyl)-ethylene (p,p'-DDE) in human plasma related to fish consumption. Arch Environ Health 49;6:477

Assmuth TW. 2000. Natural risks in ecosystem sustainability policy: General analysis, and lessons from Near-Earth Objects and toxic substances. Global Environmental Issues in the 21st Century: Problems, Causes and Solutions. Pensacola, SETAC. Proc 3rd SETAC World Congress and 10th Ann Meet SETAC Eur, Brighton, UK, 21-25 May 2000. Abstracts 1C/002;11

Assmuth TW. 2003. Indicative analysis of relative risks and data gaps of dioxin-like compounds in Baltic Sea fish, based on body burden, biokinetic and bioactivity ratios. Organohalogen Compds. 62:37

Assmuth T, Louekari K. 2001. Research for management of environmental risks from endocrine disrupters - contexts, knowledge base, methodologies and strategies. The Finnish Environ 448. www.vyh.fi/eng/publica/electro/fd448/fe448.htm

ATSDR. 1998. Toxicological profile for polychlorinated biphenyls. Agency Toxic Substances Disease Registry, U.S. Dept Health Human Services, Atlanta, Georgia, Dec 1998. www.atsdr.cdc.gov/

Atuma SS et al. 1998a. Organochlorine pesticides, polychlorinated biphenyls and dioxins in human milk from Swedish mothers. Food Addit Contam. 15;2:142

Aulerich RJ et al. 1988. Biological effects of epidermal growth factor and 2,3,7,8-tetrachlorodibenzo-p-dioxin on developmental parameters of neonatal mink. Arch Environ Contam Toxicol. 17;1:27

Aw TY et al. 1990. Tributyltin stimulates apoptosis in rat thymocytes. Arch Biochem Biophys. 283:46

Aylward LL, Hays SM. 2002. Temporal trends in human TCDD body burden: decreases over three decades and implications for exposure levels. J Expo Anal Environ Epidemiol. 12;5:319

Aylward LL et al. 2005. Concentration-dependent TCDD elimination kinetics in humans: toxicokinetic modeling for moderately to highly exposed adults from Seveso, Italy, and Vienna, Austria, and impact on dose estimates for the NIOSH cohort. J Expo Anal Environ Epidemiol. 15;1:51

Aylward LL et al. 2003a. TCDD kinetics in Austrian patients: Implementation of the Carrier et al. (1995) model. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 1

Aylward L et al. 2003b. Impact of a concentration-dependent elimination rate for TCDD on dose estimates for the NIOSH cohort. Organohalogen Compds. 60-65, CD-ROM Vol. 5, section 1

Axelman J et al. 2001. Dynamics and distribution of hydrophobic organic compounds in the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Springer Verlag, Berlin & Heidelberg. Ecol Studies 148:257

Axmon A et al. 2002. Female fertility in relation to the consumption of fish contaminated with persistent organochlorine compounds. Scand J Work Environ Health 28;2:124

Axmon A et al. 2004a. Altered menstrual cycles in women with a high dietary intake of persistent organochlorine compounds. Chemosphere 56;8:813

Axmon A et al. 2004b. Polychlorinated biphenyls in serum and time to pregnancy. Environ Res. 96;2:186

Ball DJ. 2002. Environmental risk assessment and the intrusion of bias. Environ Int. 28;6:529

Barnard RC. 1996. A new approach to risk assessment integrating scientific evaluation and economic assessment of costs and benefits. Regul Toxicol Pharmacol. 24;2 Pt 1:121

Baeyens W et al. 2002. PCBs in Northern sea fish: Is there a health risk? Organohalogen Compds. 55:315

Becher G et al. 1995. PCDDs, PCDFs, and PCBs in human milk from different parts of Norway and Lithuania. J Toxicol Environ Health 46;2:133

Bekker DL, Nolet BA. 1990. The diet of otters Lutra lutra in the Netherlands in winter and early spring. Lutra 33:133-44. (Ref. Leonards et al. 1998)

Bergek S et al. 1992. Concentrations of PCDDs and PCDFs in seals from Swedish waters. Ambio 21;8:553

Berggren P et al. 1999. Patterns and levels of organochlorines (DDTs, PCBs, non-ortho PCBs and PCDD/Fs) in male harbour porpoises (Phocoena phocoena) from the Baltic Sea, the Kattegat-Skagerrak Seas and the West Coast of Norway. Mar Pollut Bull. 38;12:1070

Bergqvist P-A et al. 2005. Kartläggning av utsläppskällor för oavsiktligt bildade ämnen: PCDD/F, PCB och HCB. Miljökemi, Kem Inst, Umeå Univ, Mar 2005. MK 2005:01

Bignert A et al. 1989. Polychlorinated dibenzo-p-dioxins (PCDD) and dibenzo-furans (PCDF) in seal blubber. Chemosphere 19;1-6:551

Bignert A et al. 1998. Temporal trends of organochlorines in Northern Europe, 1967 1995. Relation to global fractionation, leakage from sediments and international measures. Environ Pollut. 99;2:177

Bignert A et al. 2005. Spatial and seasonal variation of the dioxin and PCB content in herring from the northern Baltic Sea. Organohalogen Compds. 1403-5. CD-ROM ID 1639

Birnbaum LS. 1994b. The mechanism of dioxin toxicity: Relationship to risk assessment. Environ Health Perspect. 102;Suppl 9:157

Birnbaum LS. 1995a. Developmental effects of dioxins. Environ Health Perspect. 103;Suppl 7:89

Birnbaum LS, Fenton SE. 2002. Role of developmental exposure to endocrine disruptors in cancer. Organohalogen Compds. 56:65

Birnbaum LS et al. 1991. Teratogenic effects of 2,3,7,8-tetrabromodibenzo-p-dioxin and three polybrominated dibenzofurans in C57BL/6N mice. Toxicol Appl Pharmacol. 107;1:141

Birnbaum LS et al. 2003. Health effects of polybrominated dibenzo-p-dioxins (PBDDs) and dibenzofurans (PBDFs). Environ Int. 29;6:855

Bjerselius R et al. 2003. Study of dioxin levels in fatty fish from Sweden 2001-2002. Part II. Organohalogen Compds. 62:193-6. CD-ROM Vol. 3, Section 1

Björnfoth H et al. 2005. Possible risk of prostate cancer associated with adipose tissue concentrations of persistent organic pollutants. Organohalogen Compds. 2005. 1638-42. CD-ROM ID 1551

Blomkvist G et al. 1992. Concentrations of SDDT and PCB in seals from Swedish and Scottish waters. Ambio 21:539

Boersma ER, Lanting CI. 2000. Environmental exposure to polychlorinated biphenyls (PCBs) and dioxins. Consequences for longterm neurological and cognitive development of the child lactation. Adv Exp Med Biol. 478:271

Bogaards JJP et al. 2000. Prediction of interindividual variation in drug plasma levels in vivo from individual enzyme kinetic data and physiologically based pharmacokinetic modelling. Eur J Pharmaceut Sci. 12:117

Bol J et al. 1989. Interactive effects of PCDD's, PCDF's and PCB's as assessed by the E.L.S.-bioassay. Chemosphere 19;1-6:899

Bonvalot Y et al. 1989. Uncertainty in quantitative carcinogenic risk assessment procedures for 2,3,7,8-tetrachlorodibenzo-p-dioxin. Chemosphere 19;1-6:623

Boon JP et al. 1987. The kinetics of individual polychlorinated biphenyl congeners in female harbour seals (Phoca vitulina), with evidence for structure-related metabolism. Aquat Toxicol. 10:307

Boon JP et al. 1997. Concentration-dependent changes of PCB patterns in fish-eating mammals: Structural evidence for induction of cytochrome P450. Arch Environ Contam Toxicol. 33:298

Bosveld ATC, van den Berg M. 2002. Reproductive failure and endocrine disruption by organohalogens in fish-eating birds. Toxicol. 181-182:155
Bosveld ATC et al. 1992. Assessment of the EROD inducing potency of eleven 2,3,7,8-substituted PCDD/Fs and three coplanar PCBs in the chick embryo. Chemosphere 25;7-10:911
Bosveld ATC et al. 2000. Biochemical and developmental effects of dietary exposure to polychlorinated biphenyls 126 and 153 in common tern chicks (Sterna hirundo). Environ Toxicol Chem. 19(2000):719
Breivik K et al. 2002b. Towards a global historical emission inventory for selected PCB congeners – a mass balance approach. 2. Emissions. Sci Total Environ. 290:199
Brewster DW, Birnbaum LS. 1987. Disposition and excretion of 2,3,4,7,8-pentachlorodibenzofuran in the rat. Toxicol Appl Pharmacol. 90;2:243
Brouwer A et al. 1995. Functional aspects of developmental toxicity of polyhalogenated aromatic hydrocarbons in experimental animals and human infants. Eur J Pharmacol. 293;1:1
Brouwer A et al. 1998b. Report of the WHO working group on the assessment of health risks for human infants from exposure to PCDDs, PCDFs and PCBs. Chemosphere 37;9-12:1627
Brown SB et al. 2004. Altered thyroid status in lake trout (Salvelinus namaycush) exposed to co-planar 3,3',4,4',5-pentachlorobiphenyl. Aquat Toxicol. 67;1:75
Bruhn R et al. 1999. Persistent chlorinated organic contaminants in harbour porpoises from the North Sea, the Baltic Sea and Arctic waters. Sci Total Environ. 237-238:351
Brunström B, Halldin K. 1998. EROD induction by environmental contaminants in avian embryo livers. Comp Biochem Physiol C 121;1-3:213
Brunström B et al. 1990. Embryotoxicity of polycyclic aromatic hydrocarbons (PAHs) in three domestic avian species, and of PAHs and coplanar polychlorinated biphenyls (PCBs) in the common eider. Environ Pollut. 67;2:133
Brunström B et al. 1991a. Toxicity and EROD-inducing potency of 24 polycyclic aromatic hydrocarbons (PAHs) in chick embryos. Arch Toxicol. 65;6:485
Brunström B et al. 1992. Extracts from settling particulate matter collected in the Stockholm archipelago waters: Embryolethality, immunotoxicity and EROD-inducing potency of fractions containing aliphatics/monoaromatics, diaromatics or polyaromatics. Environ Toxicol Chem. 11;10:1441
Brunström B et al. 2001. Reproductive toxicity in mink (Mustela vison) chronically exposed to environmentally relevant polychlorinated biphenyl concentrations. Environ Toxicol Chem. 20;10:2318
Canton RC et al. 2003. Expression of Cyp1a1 and 1b1 mRNA in blood lymphocytes from two district populations in Slovakia compared to total TEqs in blood as measured by the DRE-CALUX® assay. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 1
Carrier G et al. 1995a. Modeling of the toxickinetics of polychlorinated dibenzo-p-dioxins and dibenzofurans in mammalians, including humans. I. Nonlinear distribution of PCDD/PCDF body burden between liver an adipose tissues. Toxicol Appl Pharmacol. 131;2:253
Carrier G et al. 1995b. Modeling of the toxicokinetics of polychlorinated dibenzo-p-dioxins and dibenzofurans in mammalians, including humans. II. Kinetics of absorption and disposition of PCDDs/PCDFs. Toxicol Appl Pharmacol. 131;2:267
Carvalho PS et al. 2004. Intra-strain dioxin sensitivity and morphometric effects in swim-up rainbow trout (Oncorhynchus mykiss). Comp Biochem Physiol C Toxicol Pharmacol. 137;2:133
Caudill SP et al. 1992. Effects of measurement error on estimating biological half-life. J Expo Anal Environ Epidemiol. 2;4:463
CCME. 1999. Protocol for the derivation of Canadian tissue residue guidelines for the protection of wildlife that consume aquatic biota. Can Council Ministers Environ, Winnipeg Jan 1998, in corporating Mar 1998 errata. www.ec.gc.ca
Champ MA. 2000. A review of organotin regulatory strategies, pending actions, related costs and benefits. Sci Total Environ. 258;1-2:21
Chen G, Bunce NJ. 2004. Interaction between halogenated aromatic compounds in the Ah receptor signal transduction pathway. Environ Toxicol. 19;5:480
Chen JJ et al. 2001. Using dose addition to estimate cumulative risks from exposures to multiple chemicals. Regul Toxicol Pharmacol. 34;1:35
Cheung MO et al. 1981. Cardiovascular teratogenicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the chick embryo. Toxicol Appl Pharmacol. 61(2):197
Clench-Aas J et al. 1992. PCDD and PCDF in human milk from Scandinavia, with special emphasis on Norway. J Toxicol Environ Health 37(1):73
Cohen J et al. 2005. A quantitative risk-benefit analysis of changes in population fish consumption. Am J Prev Med. 29;4: 325-34
Comber SDW et al. 2003. Development of aquatic quality standards (QSs) for dioxins. Final Report to Dept Environ Food Rural Affairs. Report No: DEFRA 6297, Jul 2003.
Connor K et al. 2004. Estimating the total TEq in human blood from naturally-occurring vs. anthropogenic dioxins: A dietary study. Organohalogen Compds. 66:3408
Cook PM et al. 2003. Effects of aryl hydrocarbon receptor-mediated early life stage toxicity on lake trout populations in Lake Ontario during the 20th century. Environ Sci Technol. 37;17:3864
COT. 2001. COT statement on the tolerable daily intake for dioxins and dioxin-like polychlorinated biphenyls. Committee On Toxicity, London. COT/2001/07
Czub G, McLachlan MS. 2004. A food chain model to predict the levels of lipophilic organic contaminants in humans. Environ Toxicol Chem. 23;10:2356
Dallaire F et al. 2004. Acute infections and environmental exposure to organochlorines in Inuit infants from Nunavik. Environ Health Perspect. 112;14:1359
Daniel V et al. 2001. Associations of blood levels of PCB, HCHS, and HCB with numbers of lymphocyte subpopulations, in vitro lymphocyte response, plasma cytokine levels, and immunoglobulin autoantibodies. Environ Health Perspect. 109;2:173
Darnerud PO. 2003.Toxic effects of brominated flame retardants in man and in wildlife. Environ Int. 29:6:841
Daviglus ML et al. 1997. Fish consumption and the 30-year risk of fatal myocardial infarction. N Engl J Med. 336;15:1046
Davis D, Safe SH. 1989. Dose-response immunotoxicities of commercial polychlorinated biphenyls (PCBs) and their interaction with 2,3,7,8-tetrachlorodibenzo-p-dioxin. Toxicol Lett. 48:35
de Boer J et al. 1993. Non-ortho and mono-ortho substituted chlorobiphenyls and chlorinated dibenzo-p-dioxins and dibenzofurans in marine and freshwater fish and shellfish from The Netherlands. Chemosphere 26;10:1823
De Jongh J et al. 1993. Toxicokinetic mixture interactions between chlorinated aromatic hydrocarbons in the liver of the C57BL/6J mouse: 2. Polychlorinated dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs) and biphenyls (PCBs). Arch Toxicol. 67;9:598

Dekkers S et al. 2001. Critical effect sizes in toxicological risk assessment: a comprehensive and critical evaluation. Environ Toxicol Pharmacol.. 10(1-2):33
de March BGE et al. 1998. Persistent organic pollutants. de March et al. (eds.) AMAP assessment report: Arctic pollution issues. AMAP, Oslo, Norway. (Rwf. by Berggren P et al. 1999)
Denison MS, Nagy SR. 2003. Activation of the aryl hydrocarbon receptor by structurally diverse exogenous and endogenous chemicals. Annu Rev Pharmacol Toxicol. 43:309
De Rosa CT et al. 1999a. Dioxin and dioxin-like compounds in soil, Part I: ATSDR policy guideline. Toxicol Ind Health 15;6:552
Desaulniers D et al. 2003. Effects of postnatal exposure to mixtures of non-ortho-PCBs, PCDDs, and PCDFs in prepubertal female rats. Toxicol Sci. 75;2:468
Despres C et al. 2005. Neuromotor functions in Inuit preschool children exposed to Pb, PCBs, and Hg. Neurotoxicol Teratol. 27;2:245
de Swart RL et al. 1994. Impairment of immune function in harbour seals (Phoca vitulina) feeding on fish from polluted waters. Ambio 23:155
DeVito MJ et al. 1993. Comparative ability of various PCBs, PCDFs, and TCDD to induce cytochrome P450 1A1 and 1A2 activity following 4 weeks of treatment. Fundam Appl Toxicol. 20;1:125
DeVito MJ et al. 1997. Dose-response relationships for polyhalogenated dioxins and dibenzofurans following subchronic treatment in mice. I. CYP1A1 and CYP1A2 enzyme activity in liver, lung, and skin. Toxicol Appl Pharmacol. 147;2:267
DeVito MJ et al. 1998. Dose-response relationships for disposition and hepatic sequestration of polyhalogenated dibenzo-p-dioxins, dibenzofurans, and biphenyls following subchronic treatment in mice. Toxicol Sci. 46;2:223
Dewailly E et al. 2000. Susceptibility to infections and immune status in Inuit infants exposed to organochlorines. Environ Health Perspect. 108;3:205
Dieter HH, Konietzka R. 1995. Which multiple of a safe body dose derived on the basis of default factors would probably be unsafe? Regul Toxicol Pharmacol. 22;3:262
Din JN et al. 2004. Omega 3 fatty acids and cardiovascular disease--fishing for a natural treatment. Br Med J. 328;7430:30-5. Discussion: Br Med J 328;7436:406-7
Dosman DM et al. 2001. Socioeconomic determinants of health- and food safety-related risk perceptions. Risk Anal. 21(2):307
Douglas M. 1996 (1992). Risk and blame – Essays in cultural theory. Routledge, London
Dunn JR et al. 1994. Psychosocial effects of PCB contamination and remediation: the case of Smithville, Ontario. Soc Sci Med. 39(8):1093
EC. 2002a. Commission recommendation of 4 March 2002 on the reduction of the presence of dioxins, furans and PCBs in feedingstuffs and foodstuffs. Notified under number C(2002a) 836. Off J EC 9.3.2002 L 67/69 (2002/201/EC)
EC. 2003a. Technical guidance document in support of commission directive 93/67/EEC on risk assessment for new notified substances and commission regulation (EC) No 1488/94 on risk assessment for existing substances. 2nd ed. JRC, Ispra. 4 vol. EUR 20418 EN/1-IV. www.ecb.jrc.it/
ECETOC. 1995. Assessment factors in human health risk assessment. Eur Centre Ecotoxicol Toxicol Chem, Brussels. ECETOC Publ. 68. 57 p
Elliott JE et al. 1996. Biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans and biphenyls in bald eagle (Haliaeetus leucocephalus) chicks. Environ Toxicol Chem. 15:782
Elliott JE et al. 2001a. Assessment of biological effects of chlorinated hydrocarbons in osprey chicks. Environ Toxicol Chem. 20;4:866
Elliott JE et al. 2001b. Monitoring temporal and spatial trends in polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in eggs of great blue heron (Ardea herodias) on the coast of British Columbia, Canada, 1983-1998. Ambio 30;7:416
Emond C et al. 2003a. Application of a physiologocially based pharmacokinetic (pbpk) model to aid in understanding relative potency factors for dioxinlike chemicals. Organohalogen Compds. 60-65. CD-ROM, Vol. 6, Section 1
Engwall M et al. 1997a. Dioxin-like compounds in HPLC-fractionated extracts of marine samples from the East and West coast of Sweden: Bioassay- and instrumentally-derived TCDD equivalents. Mar Poll Bull. 34;12:1032
Eskenazi B et al. 2003. Maternal serum dioxin levels and birth outcomes in women of Seveso, Italy. Environ Health Perspect. 111;7:947
Fair PA, Becker PR. 2000. Review of stress in marine mammals. J Aquat Ecosyst Stress Recovery 7;4:335
Falandysz J et al. 1994d. Most toxic and highly bioaccumulative PCB congeners in cod-liver oil of Baltic origin processed in Poland during the 1970s and 1980s, their TEq-values and possible intake. Sci Total Environ. 145;3:207
Falandysz J et al. 1994e. Congener-specific data on polychlorinated biphenyls in tissues of common porpoise from Puck Bay, Baltic Sea. Arch Environ Contam Toxicol 26;3:267
Falandysz J et al. 1994f. Concentrations, clearance rates and toxic potential of non-ortho coplanar PCBs in cod liver oil from the southern Baltic Sea from 1971 to 1989. Mar Pollut Bull. 28:259
Falandysz J et al. 1996b. Polychlorinated naphthalenes in sediment and biota from the Gdansk basin, Baltic Sea. Environ Sci Technol. 30:3266
Falandysz J et al. 2000b. Relative contribution of chlorinated naphthalenes, -biphenyls, -dibenzofurans and –dibenzo-p-dioxins to toxic equivalents in biota from the South coast of the Baltic Sea. Organohalogen Compds. 47:9
Falandysz J et al. 2002b. Multivariate analysis of the bioaccumulation of polychlorinated biphenyls (PCBs) in the marine pelagic food web from the southern part of the Baltic Sea, Poland. J Environ Monit. 4;6:929
Falk C et al. 1999. Body burden levels of dioxin, furans, and PCBs among frequent consumers of Great Lakes sport fish. The Great Lakes Consortium. Environ Res. 80;2 Pt 2:S19
FAO & WHO. 2002a. Evaluation of certain food additives and contaminants. Fifty-seventh report of the Joint FAO/WHO Expert Committee on Food Additives. WHO Tech Report Series 909:1-181. World Health Organization, Geneva
Faqi AS et al. 1998. Reproductive toxicity and tissue concentrations of low doses of 2,3,7,8-tetrachlorodibenzo-p-dioxin in male offspring of rats exposed throughout pregnancy and lactation. Toxicol Appl Pharmacol. 150;2:383
Fernandes AR et al. 2003. PCDD/Fs and PCBs in fish oil dietary supplements. Organohalogen Compds. 60-65(CD-ROM)
Feyk LA et al. 2000. Changes in Cytochrome P4501a activity during development in common tern chicks fed polychlorinated biphenyls, as measured by the caffeine breath test. Environ Toxicol Chem. 19;3:712
Fierens S et al. 2005. Gender dependent accumulation of dioxins in smokers. Occup Environ Med. 62;1:61
Finkel AM. 1995. Toward less misleading comparisons of uncertain risks: the example of aflatoxin and alar. Environ Health Perspect. 103;4:376
Finkel AM. 1996. Comparing risks thoughtfully. Risk - Health Saf Environ. 7:3256. www.piercelaw.edu/RISK/rskarts.htm
Fox GA et al. 1998. Monitoring the elimination of persistent toxic substances from the Great Lakes; chemical and physiological evidence from adult herring gulls. Environ Monitor Assess. 53:147

Frank A et al. 1992. Metal concentrations in seals from Swedish waters. Ambio 21;8:529
Fromberg A et al. 2000. Levels of Toxaphene congeners in fish from Danish waters. Chemosphere 40:1227
Fromberg A et al. 2005. Levels of PCB in cod liver from Danish waters 1988 – 2004. Organohalogen Compds. 1247-9. CD-ROM ID 1553
FSAI. 2002. Summary of investigation of dioxins, furans and PCBs in farmed salmon, wild salmon, farmed trout and fish oil capsules. Food Saf Authority Ireland, Mar 2002. http://www.fsai.ie/industry/Dioxins3.htm
Fujita H et al. 2002. Characterization of the aryl hydrocarbon receptor repressor gene and association of its Pro185Ala polymorphism with micropenis. Teratol. 65;1:10
Gastel JA. 2001. Early indicators of response in biologically based risk assessment for nongenotoxic carcionogens. Regul Toxicol Pharmacol. 33;3:393
Gastel JA, Sutter TR. 1995. Biologically bounded risk assessment for receptor-mediated nongenotoxic carcinogens. Regul Toxicol Pharmacol. 22;3:273
Gehrs BC et al. 1997. Alterations in the developing immune system of the F344 rat after perinatal exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin: II. Effects on the pup and the adult. Toxicol. 122;3:229
Giesy JP et al. 1994a. Dioxins, dibenzofurans, PCBs and colonial, fish-eating water birds. Schecter A (ed.) Dioxins and health, Plenum Press, New York
Giesy JP et al. 1995. Contaminants of fishes from Great Lakes-influenced sections and above dams of three Michigan rivers: III. Implications for health of bald eagles. Arch Environ Contam Toxicol. 29;3(1995):309
Giesy JP et al. 2002. Effects of chronic dietary exposure to environmentally relevant concentrations to 2,3,7,8-tetrachlorodibenzo-p-dioxin on survival, growth, reproduction and biochemical responses of female rainbow trout (Oncorhynchus mykiss). Aquat Toxicol. 59;1-2:35
Gill CE, Elliott JE. 2003. Influence of food supply and chlorinated hydrocarbon contaminants on breeding success of bald eagles. Ecotoxicol. 12;1-4:95
Giri BS et al. 2001. Modeling and Monte Carlo simulation of TCDD transport in a river. Water Res. 35;5:1263
Glynn AW et al. 2000a. Serum concentrations of organochlorines in men: a search for markers of exposure. Sci Total Environ. 263;1-3:197
Glynn AW et al. 2000b. Organochlorines and bone mineral density in Swedish men from the general population. Osteoporos Int. 11;12:1036
Goldstein JA, Safe S. 1989. Mechanism of action and structure-activity relationships for the chlorinated dibenzo-p-dioxins and related compounds.
Kimbrough RD, Jensen AA (eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Elsevier Science Publ BV, 2nd ed
Golor G et al. 2001. Kinetics and inductive potency of 1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin (HCDD) in rats. Life Sci. 69;5:493
Gough M. 1991. Agent Orange: exposure and policy. Am J Public Health 81;3:289
Granby K, Spliid NH. 1995. Hydrocarbons and organochlorines in common mussels from the Kattegat and the Belts and their relation to condition indices. Mar Pollut Bull. 30;1:74
Grandjean P et al. 2001. Neurobehavioral deficits associated with PCB in 7-year-old children prenatally exposed to seafood neurotoxicants. Neurotoxicol Teratol. 23;4:305
Grasman KA et al. 2000a. Geographic variation in hematological variables in adult and prefledgling herring gulls (Larus argentatus) and possible associations with organochlorine exposure. Arch Environ Contam Toxicol. 38;2:244
Grassman JA et al. 1998. Animal models of human response to dioxins. Environ Health Perspect. 106 Suppl 2:761
Gray LE et al. 1997a. A dose-response analysis of the reproductive effects of a single gestational dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin in male Long Evans Hooded rat offspring. Toxicol Appl Pharmacol. 146;1:11
Greene JF et al. 2003. Basis for a proposed reference dose (RfD) for dioxin of 1-10 pg/kg-day: a weight of evidence evaluation of the human and animal studies. J Toxicol Environ Health B 6;2:115
Griffin RJ et al. 1995. The effects of community pluralism on press coverage of health risks from local environmental contamination. Risk Anal. 15(4):449
Griffin RJ et al. 1999. Proposed model of the relationship of risk information seeking and processing to the development of preventive behaviors. Environ Res. 80;2 Pt 2:S230
Grimvall E et al. 1997. Monitoring of polychlorinated biphenyls in human blood plasma: methodological developments and influence of age, lactation, and fish consumption. Arch Environ Contam Toxicol. 32;3:329
Grinwis GC et al. 2000a. Toxicology of environmental chemicals in the flounder (Platichthys flesus) with emphasis on the immune system: field, emi-field (mesocosm) and laboratory studies. Toxicol Lett. 112-113:289
Guiney PD et al. 1997. Correlation of 2,3,7,8-tetrachlorodibenzo-p-dioxin induction of cytochrome P4501A in vascular endothelium with toxicity in early life stages of lake trout. Toxicol Appl Pharmacol. 143;2:256
Guiney PD et al. 2000. Hemodynamic dysfunction and cytochrome P4501A mRNA expression induced by 2,3,7,8-tetrachlorodibenzo- p-dioxin during embryonic stages of lake trout development. Toxicol Appl Pharmacol. 168;1:1
Gustafsson Ö et al. 2003b. Accounting for PCBs in the continental shelf sediments as part of a global budget for PCBs: toward predictions of global environmental longevities. Organohalogen Compds. 60-65. CD-ROM, Vol. 2, Section 4
Guttes S et al. 1998. Chlororganic pesticides and polychlorinated biphenyls in breast tissue of women with benign and malignant breast disease. Arch Environ Contam Toxicol. 35;1:140
Guvenius DM et al. 2002. Metabolites of polychlorinated biphenyls in human liver and adipose tissue. Environ Toxicol Chem. 21;11:2264
Haag-Grönlund M et al. 1998. Interactive effects of three structurally different polychlorinated biphenyls in a rat liver tumor promotion bioassay. Toxicol Appl Pharmacol. 152;1:153
Hagen ME et al. 1997. Environmental response to decreased dioxin and furan loadings from British Columbia coastal pulp mills. Chemosphere 34;5-7:1221
Hagmar L et al. 1995. High consumption of fatty fish from the Baltic Sea is associated with changes in human lymphocyte subset levels. Toxicol Lett. 77;1-3:335
Hagmar L et al. 1998. Consumption of fatty fish from the Baltic Sea and PCB in whole venous blood, plasma and cord blood from delivering women in the Aland/Turku archipelago. J Toxicol Environ Health A 53;8:581
Hagmar L et al. 2001b. Plasma concentrations of persistent organochlorines in relation to thyrotropin and thyroid hormone levels in women. Int Arch Occup Environ Health 74;3:184

Hagmar L et al. 2004b. Tidstrender för halter av persistenta klororganiska miljögifter i blod hos vuxna svenska män i relation till konsumtion av fet östersjöfisk. Rapport till Naturvårdsverket – 2004-03-18. www.imm.ki.se/Datavard/PDF/Rapport%20HAEMI%20dioxin.pdf
Hallgren S, Darnerud P-O. 2002. Polybrominated diphenyl ethers (PBDEs), polychlorinated biphenyls (PCBs) and chlorinated paraffins (CPs) in rats - testing interactions and mechanisms for thyroid hormone effects. Toxicol. 177;2-3:227
Hallikainen A et al. 2004. Kotimaisen järvi- ja merikalan dioksiinien, furaanien, dioksiinien kaltaisten PCB-yhdisteiden ja polybromattujen difenyylieettereiden pitoisuudet. EU-KALAT. Edita Express, Helsinki, Finland. Elintarvikeviraston julkaisuja (Publ Natl Food Admin) 1/2004
Hamm JT et al. 2003. A mixture of dioxins, furans, and non-ortho PCBs based upon consensus toxic equivalency factors produces dioxin-like reproductive effects. Toxicol Sci. 74;1:182
Hankinson O. 2005. Role of coactivators in transcriptional activation by the aryl hydrocarbon receptor. Arch Biochem Biophys. 433;2:379
Hansen JC. 2003. Human exposure to persistent organic pollutants in the Arctic. Organohalogen Compds. 60-65. CD-ROM, Vol. 2, Section 2
Hansson MC et al. 2004. Unprecedented genomic diversity of AhR1 and AhR2 genes in Atlantic salmon (Salmo salar L.). Aq Toxicol. 68:219
Hardell L et al. 2002. Is DDT exposure during fetal period and breast-feeding associated with neurological impairment? Environ Res. 88;3:141
Hardell L et al. 2004. Concentrations of polychlorinated biphenyls in blood and the risk for testicular cancer. Int J Androl. 27;5:282
Hario M et al. 2004. Organochlorine concentrations in diseased vs. healthy gull chicks from the northern Baltic. Environ Pollut.127;3(2004):411
Harper CR, Jacobson TA. 2001. The fats of life: the role of omega-3 fatty acids in the prevention of coronary heart disease. Arch Intern Med. 161;18:2185-92
Harper N et al. 1995a. Immunosuppressive activity of polychlorinated biphenyl mixtures and congeners: nonadditive (antagonistic) interactions. Fundam Appl Toxicol. 27;1:131
Harper PA et al. 2002. Polymorphisms in the human AH receptor. Chem Biol Interact. 141;1-2:161
Hart A et al. 2003. Application of uncertainty analysis in assessing dietary exposure. Toxicol Lett. 140-1:437
Hatcher SL. 1982. The psychological experience of nursing mothers upon learning of a toxic substance in their breast milk. Psychiatry 45(2):172
Hauser R et al. 2002. Environmental organochlorines and semen quality: results of a pilot study. Environ Health Perspect. 110;3:229
Hauser R et al. 2003a. The relationship between human semen parameters and environmental exposure to polychlorinated biphenyls and p,p'-DDE. Environ Health Perspect. 111;12:1505
Hauser R et al. 2003b. Lack of an association between environmental exposure to polychlorinated biphenyls and p,p'-DDE and DNA damage in human sperm measured using the neutral comet assay. Hum Reprod. 18;12:2525
Hays SM, Aylward LL. 2003. Dioxin risks in perspective: past, present, and future. Regul Toxicol Pharmacol. 37;2:202
Hays SM et al. 1997. The relative susceptibility of animals and humans to the carcinogenic hazard posed by exposure to 2,3,7,8-TCDD: an analysis using standard and internal measures of dose. Chemosphere 34;5-7:1507
HCN. 1996a. Dioxins. Advisory report no. 1996/10E. Rijswijk 6.8.1996, Health Council of The Netherlands
Head JA et al. 2003. Variation in Cytochrome P4501A mRNA indicibility among individual chickens and herring gulls. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 4
Heaton SN et al. 1995. Dietary exposure of mink to carp from Saginaw Bay, Michigan. 1. Effects on reproduction and survival, and the potential risks to wild mink populations. Arch Environ Contam Toxicol. 28;3:334
Helander B, Bignert A. 1992. Harbor seal (Phoca vitulina): Population trends and reproduction. Ambio 21;8:504
Helander B et al. 2002. The role of DDE, PCB, coplanar PCB and eggshell parameters for reproduction in the white-tailed sea eagle (Haliaeetus albicilla) in Sweden. Ambio 31;5:386
Helle E. 1976. PCB Levels correlated with pathological changes in seal uteri. Ambio 5;5-6:261
Hendriks AJ, Enserink EL. 1996. Modelling response of single-species populations to microcontaminants as a function of species size with examples for waterfleas (Daphnia magna) and cormorants (Phalacrocorax carbo). Ecol Modell. 88;1-3:247
Hertzberg RC, MacDonell MM. 2002. Synergy and other ineffective mixture risk definitions. Sci Total Environ. 288;1-2:31
Hites RA et al. 2004a. Global assessment of organic contaminants in farmed salmon. Science 303;5655:226
Hites RA et al. 2004b. Responses to Rembold CN, Tuomisto JN et al., Lund E et al. and Weaver DE. Science 305:475-98.
Hoffman DJ et al. 1998. Comparative developmental toxicity of planar PCB congeners in chickens, American kestrels and common terns. Environ Toxicol Chem. 17:747
Hoffmann S, Krupnick AJ. 2004. Valuing risk to health – children are not little adults. Resources (Resources For the Future Publication) 154:12
Holmes M et al. 2003. Dietary exposure to dioxins and PCBs including measurement uncertainty and limits of detection. Organohalogen Compds. 60-65, CD-ROM Vol 1, Section 2
Holmström KE et al. 2005. Temporal trends of PFOS and PFOA in guillemot eggs from the Baltic Sea, 1968--2003. Environ Sci Technol. 39;1:80
Hoover SM. 1999. Exposure to persistent organochlorines in Canadian breast milk: a probabilistic assessment. Risk Anal. 19;4:527
Hornung MW et al. 1996a. Toxic equivalency factors of polybrominated dibenzo-p-dioxin, dibenzofuran, biphenyl, and polyhalogenated diphenyl ether congeners based on rainbow trout early life stage mortality. Toxicol Appl Pharmacol. 140;2:227
Houweling DA et al. 2002. Levels, trends and determinants of organohalogen compounds in breast milk in The Netherlands. Organohalogen Compds. 56:337
Hovander L et al. 2002. Identification of hydroxylated PCB metabolites and other phenolic halogenated pollutants in human blood plasma. Arch Environ Contam Toxicol. 42;1:105
Huestis SY et al. 1997. Evaluation of temporal and age-related trends of chemically and biologically generated 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents in Lake Ontario trout, 1997 to 1993. Environ Toxicol Chem. 16;2:154
Hurst CH et al. 2000a. Acute administration of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in pregnant Long Evans rats: association of measured tissue concentrations with developmental effects. Toxicol Sci. 53;2:411
Hurst CH et al. 2000b. Tissue disposition of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in maternal and developing Long-Evans rats following subchronic exposure. Toxicol Sci. 57;2:2750
Inoue K et al. 2000. Ethnic-related differences in the frequency distribution of genetic polymorphisms in the CYP1A1 and CYP1B1 genes in Japanese and Caucasian populations. Xenobiotica 30;3:285
IPCS. 1994a. Polybrominated biphenyls. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 152
IPCS. 1998. Polybrominated dibenzo-p-dioxins and dibenzofurans. WHO and Int Progr Chem Saf, Geneva. Environ Health Criteria 205

Ishaq R et al. 2000. Tissue distribution of polychlorinated naphthalenes (PCNs) and non-orthochlorinated biphenyls (non-ortho CBs) in harbour porpoises (Phocoena phocoena) from Swedish waters. Chemosphere 41;12:1913
Isosaari P et al. 2002b. Feeding trial on rainbow trout: comparison of dry fish feed and Baltic herring as a source of PCDD/Fs and PCBs. Chemosphere 48;8:795
Isosaari P et al. 2002c. Spatial distibution and temporal accumulation of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in the Gulf of Finland. Environ Sci Technol. 36;12:2560
Isosaari P. 2004. Polychlorinated dibenzo-p-dioxin and dibenzofuran contamination of sediments and photochemical decontamination of soils. Kuopio, Finland, NPIH. Publ Natl Public Health Inst. A 11/2004. Doctoral dissertation, Univ Kuopio
Jacobson SW et al. 1999. Breastfeeding effects on intelligence quotient in 4- and 11-year-old children. Pediatrics 103;5:e71
Jakobsson K et al. 2005. Polybrominated Diphenyl Ethers (PBDEs) in serum from Swedish men 1988-2002. A longitudinal study. Organohalogen Compds. 533-6. CD-ROM ID 2140
Jansson B et al. 1993. Chlorinated and brominated persistent organic compounds in biological samples from the environment. Environ Toxicol Chem. 12;7:1163
JECFA. 2001. Summary and conclusions, Fifty-seventh meeting Rome, 5-14 Jun 2001. Joint FAO/WHO Expert Committee on Food Additives
Jensen KG et al. 2000. Induction of aberrant mitosis with PCBs: particular efficiency of 2,3,3',4,4'-pentachlorobiphenyl and synergism with riphenyltin. Mutagenesis 15;1:9
Jensen RK, Sleight SD. 1986. Sequential study on the synergistic effects of 2,2',4,4',5,5'-hexabromobiphenyl and 3,3',4,4',5,5'-hexabromobiphenyl on hepatic tumor promotion. Carcinogenesis 7;10:1771
Jensen S et al. 1977a. Levels of DDT and PCB in littoral fishes along the Swedish coast. Ambio Spec Report 5:75
Johnson BL, DeRosa CT. 1999. Conclusion – Public health implications. Environ Res. 80;2 Pt 2:S183
Jonsson P et al. 1993. Pulp mill related polychlorinated organic compounds in Baltic Sea sediments. Ambio 22:37
Jones PD et al. 2001. Accumulation of 2,3,7,8-tetrachlorodibenzo-p-dioxin by rainbow trout (Onchorhynchus mykiss) at environmentally relevant dietary concentrations. Environ Toxicol Chem. 20;2:344
Judd NL et al. 2003. Assessment of PCB congener analytical methods: do they meet risk assessment needs? Arch Environ Contam Toxicol. 44;1:132
Judd NL et al. 2004. Contribution of PCB exposure from fish consumption to total dioxin-like dietary exposure. Regul Toxicol Pharmacol. 40;2:125
Järnberg U et al. 1993. Polychlorinated biphenyls and polychlorinated naphthalenes in Swedish sediment and biota: Levels, patterns, and time trends. Environ Sci Technol. 27:1364
Järnberg U et al. 1997. Distribution of polychlorinated naphthalene congeners in environmental and source-related samples. Arch Environ Contam Toxicol. 32:232
Kannan N et al. 1989. Critical evaluation of polychlorinated biphenyl toxicity in terrestrial and marine mammals: increasing impact of non-ortho and mono-ortho coplanar polychlorinated biphenyls from land to ocean. Arch Environ Contam Toxicol. 18;6:850
Kannan N et al. 1995. Chlorobiphenyls: Model compounds for metabolism in food chain organisms and their potential use as ecotoxicological stress indicators by application of the metabolic slope concept. Environ Sci Technol. 29:1851
Kannan K et al. 2000. Human Ecol Risk Assess. 6;1:181-201 (Ref. by Shaw et al. 2003)
Kannan K et al. 2001. Polychlorinated naphthalenes, -biphenyls, -dibenzo-p-dioxins, and -dibenzofurans in double-crested cormorants and herring gulls from Michigan waters of the Great Lakes. Environ Sci Technol. 35;3:441
Kannan K et al. 2002b. Perfluorooctanesulfonate and related fluorinated hydrocarbons in marine mammals, fishes, and birds from coasts of the Baltic and the Mediterranean Seas. Environ Sci Technol. 36;15:3210
Karl H, Ruoff U. 2004. Dioxins and dioxin-like PCBs in fish in general and in particular from the Baltic Sea. Organohalogen Compds. 66:1910
Karl H et al. 2002. Levels of dioxins in fish and fishery products on the German market. Chemosphere 49;7:765
Karlson K et al. 2000. PCBs, DDTs and methyl sulphone metabolites in various tissues of harbour porpoises from Swedish waters. Environ Pollut. 110;1:29
Karlsson L et al. 1999. The diet of salmon (Salmo salar) in the Baltic Sea and connections with the M74 syndrome. Ambio 28;1:37
Karmaus W. 2001. Of jugglers, mechanics, communities, and the thyroid gland: how do we achieve good quality data to improve public health? Environ Health Perspect. 109 Suppl 6:863
Karmaus W, Zhu X. 2004. Maternal concentration of polychlorinated biphenyls and dichlorodiphenyl dichlorethylene and birth weight in Michigan fish eaters: a cohort study. Environ Health 3;1:1
Kennedy SW et al. 1996. Cytochrome P4501A induction in avian hepatocyte cultures: a promising approach for predicting the sensitivity of avian species to toxic effects of halogenated aromatic hydrocarbons. Toxicol Appl Pharmacol. 141;1:214
Kikuchi H et al. 2001. Method for evaluation of immunotoxicity of dioxin compounds using human T-lymphoblastic cell line, L-MAT. Chemosphere 43;4-7:815
Kimbrough RD, Krouskas C. 2001. Polychlorinated biphenyls, dibenzo-p-dioxins, and dibenzofurans and birth weight and immune and thyroid function in children. Regul Toxicol Pharmacol. 34;1:42
Kimbrough RD, Krouskas C. 2003. Human exposure to polychlorinated biphenyls and health effects : a critical synopsis. Toxicol Rev. 22;4:217
Kimbrough RD et al. 2001. Analysis of research studying the effects of polychlorinated biphenyls and related chemicals on neurobehavioral development in children. Vet Hum Toxicol. 43;4:220
Kiviranta H et al. 2002a. Polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in fishermen in Finland. Environ Health Perspect. 10;4:355
Kiviranta H et al. 2003. PCDD/Fs and PCBs in Baltic herring during the 1990s. Chemosphere 50;9:1201
Kiviranta H. et al. 2005. Polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in the general population in Finland. Chemosphere 60;7:854
Knutzen J et al. 2003. Polychlorinated dibenzofurans/dibenzo-p-dioxins (PCDF/PCDDs) and other dioxin-like substances in marine organisms from the Grenland fjords, S. Norway, 1975-2001: present contamination levels, trends and species specific accumulation of PCDF/PCDD congeners. Chemosphere 52;4:745
Kodavanti PR et al. 2001. Differential effects of two lots of aroclor 1254: congener-specific analysis and neurochemical end points. Environ Health Perspect. 109;11:1153
Kohn MC et al. 2001. Physiological modeling of a proposed mechanism of enzyme induction by TCDD. Toxicol. 162;3:193

Koistinen J et al. 1997b. 2,3,7,8-Tetrachlorodibenzo-p-dioxin equivalents in extracts of Baltic white-tailed sea eagles. Environ Toxicol Chem. 16;7:1533
Koistinen J et al. 2004. Levels of persistent organic pollutants in Baltic herring. Organohalogen Compds. 66:1783
Koldkjaer OG et al. 2004. Parkinson's disease among Inuit in Greenland: organochlorines as risk factors. Int J Circumpolar Health 63 Suppl 2:366
Konat J, Kowalewska G. 2001. Polychlorinated biphenyls (PCBs) in sediments of the southern Baltic Sea - trends and fate. Sci Total Environ. 280;1-3:1
Koopman-Esseboom C et al. 1994. Effects of dioxins and polychlorinated-biphenyls on thyroid-hormone status of pregnant-women and their infants. Pediatr Res. 36:468
Koopman-Esseboom C et al. 1996. Effects of polychlorinated biphenyl dioxin exposure and feeding type on infants mental and psychomotor development. Pediatrics 97:700
Korkalainen M et al. 2001. The AH receptor of the most dioxin-sensitive species, guinea pig, is highly homologous to the human AH receptor. Biochem Biophys Res Commun. 285;5:1121
Korner W et al. 2002. Tissue concentrations and induction of a hepatic monooxygenase in male Wistar rats after repeated doses of defined polychlorinated dibenzo-p-dioxin and dibenzofuran (PCDDs and PCDFs) mixtures. Arch Toxicol. 75;11-12:653
Kowalewska G et al. 2003. Transfer of organic contaminants to the Baltic in the Odra Estuary. Mar Pollut Bull. 46;6:703
Kreuzer PE et al. 1997. 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and congeners in infants. A toxicokinetic model of human lifetime body burden by TCDD with special emphasis on its uptake by nutrition. Arch Toxicol. 71;6:383
Laden F et al. 2001. Plasma organochlorine levels and the risk of breast cancer: an extended follow-up in the Nurses' Health Study. Int J Cancer 91;4:568
LaKind JS et al. 2000. Methodology for characterizing distributions of incremental body burdens of 2,3,7,8-TCDD and DDE from breast milk in North American nursing infants. J Toxicol Environ Health A 59;8:605
Lawrence GS, Gobas FA. 1997. A pharmacokinetic analysis of interspecies extrapolation in dioxin risk assessment. Chemosphere 35;3:427
Leino O et al. 2005. Risk-benefit analysis of fish in Finland: Dioxins and omega-3 fatty acids. Organohalogen Compds. CD-ROM ID 1097
Leonards PE et al. 1995. Assessment of experimental data on PCB-induced reproduction inhibition in mink, based on an isomer- and congener-specific approach using 2,3,7,8-tetrachlorodibenzo-p-dioxin toxic equivalency. Environ Toxicol Chem. 14;4:639
Leonards PE et al. 1998. Studies of bioaccumulation and biotransformation of PCBs in mustelids based on concentration and congener patterns in predators and preys. Arch Environ Contamin Toxicol. 35;4:654
Liljegren G et al. 1998. Case-control study on breast cancer and adipose tissue concentrations of congener specific polychlorinated biphenyls, DDE and hexachlorobenzene. Eur J Cancer Prev. 7;2:135
Lind Y et al. 2002. Exponering för organiska miljökontaminanter via livsmedel – intagsberäkningar av ΣPCB, PCB 153, p,p'-DDE, PCDD/F, dioxinlika PCB, PBDE och HBCD baserade pa konsumtionsdata från Riksmaten 1997-98. Uppsala, Swed Natl Food Authority. Livsmedelsverket Rapport 26. 103 p
Loonen H & al. 1996. Ecological hazard assessment of dioxins: hazards to organisms at different levels of aquatic food webs (fish-eating birds and mammals, fish and invertebrates). Sci Total Environ. 182;1-3:93
Lorenzen A et al. 1997a. Sensitivity of common tern (Sterna hirundo) embryo hepatocyte cultures to CYP1A induction and porphyrin accumulation by halogenated aromatic hydrocarbons and common tern egg extracts. Arch Environ Contam Toxicol. 32;2:126
Lorenzen A et al. 1999. Relationships between environmental organochlorine contaminant residues, plasma corticosterone concentrations, and intermediary metabolic enzyme activities in Great Lakes herring gull embryos. Environ Health Perspect. 107;3:179
Lucas M et al. 2004. Gestational age and birth weight in relation to n-3 fatty acids among Inuit (Canada). Lipids 39;7:617
Lucier GW. 1991. Humans are a sensitive species to some of the biochemical effects of structural analogs of dioxin. Environ Toxicol Chem. 10;6:727
Lucier GW et al. 1990. Placental markers of human exposure to polychlorinated dibenzofurans and polychlorinated biphenyls: implications for risk assessment. IARC Sci Publ. 104:55
Lund E et al. 2004. Cancer risk and salmon intake. Science 305(2004):477
Lundén A, Norén K. 1998. Polychlorinated naphthalenes and other organochlorine contaminants in Swedish human milk, 1972-1992. Arch Environ Contam Toxicol. 34;4:414
Lundgren K. 2003. Properties and analysis of dioxin-like compounds in marine samples from Sweden. PhD Thesis, Univ Umea. www.diva-portal.org/diva/getDocument?urn_ nbn_se_umu_diva-24-1__fulltext.pdf
Lundgren K et al. 2002b. Polychlorinated naphthalene levels, distribution, and biomagnification in a benthic food chain in the Baltic Sea. Environ Sci Technol. 36;23:5005
Lundstedt-Enkel K et al. 2002. Different PCDD/PCDF congener composition in salmon and brown trout from Swedish waters. Organohalogen Compds. 57:185
Luthardt P et al. 2003. PCDD/F measurements in emissions of a Belgian sintering plant - an exemplary look at the uncertainty. Organohalogen Compds. 60-65, CD-ROM Vol. 1, Section 2
Mably TA et al. 1992c. In utero and lactational exposure of male rats to 2,3,7,8-tetrachlorodibenzo-p-dioxin. 3. Effects on spermatogenesis and reproductive capability. Toxicol Appl Pharmacol. 114;1:118
Marckmann P, Gronbaek M. 1999. Fish consumption and coronary heart disease mortailty. A systemtic review of prospective cohort studies. Eur J Clin Nutr. 53:585
Markowski VP et al. 2001. Altered operant responding for motor reinforcement and the determination of benchmark doses following perinatal exposure to low-level 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environ Health Perspect. 109;6:621
Maruyama W et al. 2002b. Possible range of dioxin concentration in human tissues: simulation with a physiologically based model. J Toxicol Environ Health A 65;24:2053
Maruyama W et al. 2003. Simulation of dioxin accumulation in human tissues and analysis of reproductive risk. Chemosphere 53;4:301
Maruyama W et al. 2004. Dioxin health risk to infants using simulated tissue concentrations. Environ Toxicol Pharmacol. 18;1:21
Masten SA et al. 1998. Population-based studies of dioxin responsiveness: Individual variation in CYP1A1 levels and relationship to dioxin body burden. Organohalogen Compds. 38:13
McConnell EE. 1989. Acute and chronic toxicity and carcinogenesis in animals. Kimbrough RD, Jensen AA (eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Elsevier Science Publ BV, 2nd ed

McGrath LF et al. 1996. Application of a biologically-based RFD estimation method to tetrachlorodibenzo-p-dioxin (TCDD) mediated immune suppression and enzyme induction. Risk Anal. 16;4:539
McLachlan MS. 1993. Exposure toxicity equivalents (ETEs): A plea for more environmental chemistry in dioxin risk assessment. Chemosphere 27;1-3:483
Mehrle PM et al. 1988. Toxicity and bioconcentration of 2,3,7,8-tetrachlorodibenzo-p-dioxin and 2,3,7,8-tetrachlorodibenzofuran in rainbow trout. Environ Toxicol Chem. 7:47
Melanson SF et al. 2005. Measurement of organochlorines in commercial over-the-counter fish oil preparations: implications for dietary and therapeutic recommendations for omega-3 fatty acids and a review of the literature. Arch Pathol Lab Med. 129;1:74
Mertz CK et al. 1998. Judgments of chemical risks: comparisons among senior managers, toxicologists, and the public. Risk Anal. 18;4:391
Mocarelli P et al. 1986. Clinical laboratory manifestations of exposure to dioxin in children. A six-year study of the effects of an environmental disaster near Seveso, Italy. J Am Med Assoc. 256;19:2687
Mocarelli P et al. 2000. Paternal concent-rations of dioxin and sex ratio of offspring. Lancet 355;9218:1858
Moilanen R et al. 1982. Time trends of chlordane, DDT, and PCB concentrations in pike (Esox lucius) and Baltic herring (Clupea harengus) in the Turku archipelago, Northern Baltic Sea for the period 1971-1982. Bull Environ Contam Toxicol. 29(1982):334
Molgaard C, Michaelsen KF. 2003. Vitamin D and bone health in early life. Proc Nutr Soc. 62;4:823
Murk AJ et al. 1994a. Toxic and biochemical effects of 3,3',4,4'-tetrachlorobiphenyl (CB-77) and clophen A50 on eider duckling (Somateria mollissima) in a semi-field experiment. Environ Pollut. 86;1:21
Murk AJ et al. 1996. Effects of polyhalogenated aromatic hydrocarbons and related contaminants on common tern reproduction: integration of biological, biochemical, and chemical data. Arch Environ Contam Toxicol. 31;1:128
Murk AJ et al. 1998. Application of biomarkers for exposure and effect of polyhalogenated aromatic hydrocarbons in naturally exposed European otters (Lutra lutra). Environ Toxicol Pharmacol. 6;2:91
Muto H, Takizawa Y. 1989. Dioxins in cigarette smoke. Arch Environ Health 44;3:171
Nagao T et al. 1993. Teratogenic potency of 2,3,4,7,8-pentachlorodibenzofuran and of three mixtures of polychlorinated dibenzo-p-dioxins and dibenzofurans in mice. Problems with risk assessment using TCDD toxic-equivalency factors. Arch Toxicol. 67;9(1993):591
NATO/CCMS. 1988a. International toxicity equivalency factor (I-TEF) method of risk assessment for complex mixtures of dioxins and related compounds. NATO Committee Challenges of Modern Society. Report No. 176
Nebert DW et al. 1991. Human AH locus polymorphism and cancer: inducibility of CYP1A1 and other genes by combustion products and dioxin. Pharmacogenetics 1;2:68
Neubert D. 1997-98. Reflections on the assessment of the toxicity of “dioxins” for humans, using data from experimental and epidemiological studies. Teratogen Carcinogen Mutagen. 17;4-5:157
Neubert D et al. 1992a. TCDD-toxicity equivalencies for PCDD/PCDF congeners: Prerequisites and limitations. Chemosphere 25;1-2:65
Neumann J, Greenwood H. 2002. Existing literature and recommended strategies for valuation of children’s health effects. NCEE, 09/23/2002. Working paper #02-07
Norén K, Meironyté D. 2000. Certain organochlorine and organobromine contaminants in Swedish human milk in perspective of past 20-30 years. Chemosphere 40:1111
North DW. 1997. Risk characterization: A bridge to informed decision making. Fundam Appl Toxicol. 39;2:81
Nothridge ME. 1995. Annnotation: Public health methods – attributable risk as a link between causality and public health action. Am J Public Health 85;9:1202
Nylund K et al. 1992. Analysis of some polyhalogenated organic pollutants in sediment and sewage sludge. Chemosphere 24:1721
Nyman M et al. 2002. Current levels of DDT, PCB and trace elements in the Baltic ringed seals (Phoca hispida baltica) and grey seals (Halichoerus grypus). Environ Pollut. 119;3:399
Nyman M et al. 2003. Contaminant exposure and effects in Baltic ringed and grey seals as assessed by biomarkers. Mar Environ Res. 55;1:73
Näf C et al. 1992. Flux estimates and pattern recognition of particulate polycyclic aromatic hydrocarbons, polychlorinated dibenzo-p-dioxins, and dibenzofurans in the waters outside various emission sources on the Swedish Baltic coast. Environ Sci Technol. 26:1444
Oetjen K, Karl H. 1998. Levels of toxaphene indicator compounds in fish meal, fish oil and fish feed. Chemosphere 37(1):1
Ohsako S et al. 2001. Maternal exposure to a low dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) suppressed the development of reproductive organs of male rats: dose-dependent increase of mRNA levels of 5alpha-reductase type 2 in contrast to decrease of androgen receptor in the pubertal ventral prostate. Toxicol Sci. 60;1:132
Ojaveer E, Lehtonen H. 2001. Fish stocks in the Baltic Sea: Finite or infinite resource? Ambio 30;4-5:217
Okey AB et al. 1994. The Ah receptor: mediator of the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Toxicol Lett. 70(1):1
Olsson M, Reutergårdh L. 1986. DDT and PCB pollution trends in the Swedish aquatic environment. Ambio 15;2:103
Olsson M et al. 1994. Diseases and environmental contaminants in seals from the Baltic and the Swedish west coast. Sci Total Environ. 154;2-3:217
Olsson M et al. 2005. Miljögifter i Östersjön – från upptäckt till samhällsreaktion. Östersjö 2005:21
Omara FO et al. 1997. Immunotoxicity of environmentally relevant mixtures of polychlorinated aromatic hydrocarbons with methyl mercury on rat lymphocytes in vitro. Environ Toxicol Chem. 16;3:576
Paasivirta J et al. 1993. Studies on toxaphene in the environment. II. PCCs in Baltic and Arctic Sea and lake fish. Chemosphere 27;10:2011
Pan X et al. 2004. Evaluation of relative potebies of PCB 126 and PCB169 for the immunotoxicities in ovalbumin (OVA)-immunized mice. Toxicol. 204:51
Parmanne R et al. 1997. Fishery and biology of herring (Clupea harengus L.) in the Gulf of Finland: A review. Boreal Environ Res. 2;2:209
Patandin S et al. 1998. Effects of environmental exposure to polychlorinated biphenyls and dioxins on birth size and growth in Dutch children. Pediatr Res. 44;4:538
Paustenbach DJ et al. 1991. Risk assessment of 2,3,7,8-TCDD using a biologically based cancer model: a reevaluation of the Kociba et al. bioassay using 1978 and 1990 histopathology criteria. J Toxicol Environ Health 34;1:11
Pereg D et al. 2002. Environmental exposure to polychlorinated biphenyls and placental CYP1A1 activity in Inuit women from northern Quebec. Environ Health Perspect. 110;6:607
Perttilä M et al. 1982. Heavy metals in Baltic herring and cod. Mar Pollut Bull. 13;11:391

Pesatori AC et al. 2003. Short- and long-term morbidity and mortality in the population exposed to dioxin after the “Seveso accident”. Ind Health 41;3:127
Peters AK et al. 2005. Polybrominated diphenyl ethers (PBDEs) antagonize or inhibit TCDD induced CYP1A1 activity in various in vitro systems. Organohalogen Compds. 2290-3. CD-ROM ID 1437
Pohl H, Holler J. 1995. Halogenated aromatic hydrocarbons and toxicity equivalency factors (TEFs) from the public health assessment perspective. Chemosphere 31;1:2547
Pohl HR, Hibbs BF. 1996. Breast-feeding exposure of infants to environmental contaminants--a public health risk assessment viewpoint: chlorinated dibenzodioxins and chlorinated dibenzofurans. Toxicol Ind Health 12;5:593
Pollenz RS et al. 1998. Female Sprague–Dawley rats exposed to a single oral dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin exhibit sustained depletion of aryl hydrocarbon receptor protein in liver, spleen, thymus, and lung. Toxicol Sci. 42;2:117
Pollitt F. 1999. Polychlorinated dibenzodioxins and polychlorinated dibenzofurans. Regul Toxicol Pharmacol. 30;2 Pt 2:S63
Ponce RA et al. 2000. Use of quality-adjusted life year weights with dose response models for public health decisions: a case study of the risks and benefits of fish consumption. Risk Anal. 20;4:529
Portier CJ et al. 1990. Biologically based models in risk assessment. Vainio H, Sorsa M, McMichael AL. (eds.) Complex mixtures and cancer risk. IARC, Lyon
Powell DC et al. 1996a. Effects of 3,3’,4,4’,5-pentachlorobiphenyl (PCB 126) and 2,3,7,8-tetrachloro-dibenzo-p-dioxin (TCDD) injected into the yolks of chicken (Gallus domesticus) eggs prior to incubation. Arch Environ Contam Toxicol. 31;3:404
Powell DC et al. 1997a. Organochlorine contaminants in double-crested cormorants from Green Bay,Wisconsin: II. Effects of an extract derived from cormorant eggs on the chicken embryo. Arch Environ Contam Toxicol. 32;3:316
Powell DC et al. 1997b. Effects of 3,3’,4,4’,5-pentachlorobiphenyl (PCB 126), 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), or an extract derived from field-collected cormorant eggs injected into double-crested cormorant (Phalacocorax auritus) eggs. Environ Toxicol Chem. 16;7:1450
Rao VR, Unger A. 1995. A novel application of a competitive binding model in dioxin risk assessment. Regul Toxicol Pharmacol. 21;2:108
Rembold CM. The health benefits of eating salmon. Science 305(2004):475
Renwick AG et al. 2003. Risk characterisation of chemicals in food and diet. Food Chem Toxicol. 41;9:1211
Ribas-Fito N et al. 2003. Breastfeeding, exposure to organochlorine compounds, and neurodevelopment in infants. Pediatrics 111;5 Pt 1:e580
Richter-Reichhelm HB et al. 2002. Workshop report. Children as a special subpopulation: focus on immunotoxicity. Fed Inst Health Protect Consumers Vet Med (BgVV), 15-16 Nov 2001, Berlin, Germany. Arch Toxicol. 76;7:377
Rier SE et al. 1993. Endometriosis in rhesus monkeys (Macaca mulatta) following chronic exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Fundam Appl Toxicol. 21;4:433-41. Comment in: Fundam Appl Toxicol. 23;1(1994):141
Riget F et al. 2005. Levels and temporal trends of PCDD/PCDFs and non-ortho PCBs in ringed seals from East Greenland. Mar Pollut Bull. 2005 Jul 21; [Epub ahead of print]
Rignell-Hydbom A et al. 2005a. Exposure to PCBs and p,p’-DDE and human sperm chromatin integrity. Environ Health Perspect. 113;2:175
Rogan WJ, Ragan NB. 1994. Chemical contaminants, pharmacokinetics, and the lactating mother. Environ Health Perspect. 102 Suppl 11:89
Rogers MD. 2003. Risk analysis under uncertainty, the precautionary principle, and the new EU chemicals strategy. Regul Toxicol Pharmacol. 37;3:370
Roos A et al. 1998. Time trend studies on ΣDDT and PCB in juvenile grey seals (Halichoerus grypus), fish and guillemot eggs from the Baltic Sea. Organohalogen Compds. 39:109
Rose M, Startin J. 2003. Accuracy and comparability of analytical data for PCDD/Fs and PCBs in food. Organohalogen Compds. 60-65, CD-ROM Vol 1, Section 2
Rosenberg NA et al. 2002. Genetic structure of human populations. Science 298;5602:2381
Ross PS et al. 1995. Contaminant-related suppression of delayed-type hypersensitivity and antibody responses in harbor seals fed herring from the Baltic Sea. Environ Health Perspect. 103:162
Ross PS et al. 1996a. Suppression of natural killer cell activity in harbour seals (Phoca vitulina) fed Baltic Sea herring. Aquat Toxicol. 34;1:71
Rozman KK, Doull J. 2003. Scientific foundations of hormesis. Part 2. Maturation, strengths, limitations, and possible applications in toxicology, pharmacology, and epidemiology. Crit Rev Toxicol. 33;3-4:451
Rozman K et al. 1987. Effect of a sublethal dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin on interscapular brown adipose tissue of rats. Toxicol Pathol. 15;4:425
Rylander L et al. 1997. The impact of age, lactation and dietary habits on PCB in plasma in Swedish women. Sci Total Environ. 207;1:55
Rylander L et al. 1998. Polychlorinated biphenyls in blood plasma among Swedish female fish consumers in relation to low birth weight. Am J Epidemiol. 147;5:493
SACN and COT. 2004. Advice on fish consumtion: benefits & risks. Sci Advisory Committee Nutr and Committee Toxicol. The Stationery Office, London.www.food.gov.uk/multimedia/pdfs/fishreport2004full.pdf
Safe S. 1993. Development of bioassays and approaches for the risk assessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin and related compounds. Environ Health Perspect. 101 Suppl 3:317
Safe S. 1997-98. Limitations of the toxic equivalence factor approach for risk assessment of TCDD and related compounds. Teratogen Carcinogen Mutage. 17:285
Safe SH. 1998. Development, validation and problems with the toxic equivalency factor approach for risk assessment of dioxins and related compounds. J Anim Sci. 76;1:134
Safe S. 2003. Implications for dietary interactions of TEqs and endogeneous/phytochemical Ah receptor ligands. Organohalogen Compds. 60-65. (CD-ROM)
Salvan A et al. 2001. Use of a toxicokinetic model in the analysis of cancer mortality in relation to the estimated absorbed dose of dioxin (2,3,7,8-te trachlorodibenzo-p-dioxin, TCDD). Sci Total Environ. 274;1-3:21
Sand S et al. 2002. Evaluation of the benchmark dose method for dichotomous data: Model dependence and model selection. Regul Toxicol Pharmacol. 36;2:184
Sanderson JT, Bellward GD. 1995. Hepatic microsomal ethoxyresorufin O-deethylase-inducing potency in ovo and cytosolic Ah receptor binding affinity of 2,3,7,8- tetrachlorodibenzo-p-dioxin: comparison of four avian species. Toxicol Appl Pharmacol. 132;1:131

Sanderson JT et al. 1997. Effects of embryonic and adult exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin on hepatic microsomal testosterone hydroxylase activities in great blue herons (Ardea herodias). Environ Toxicol Chem. 16;6:1304
Sanderson JT et al. 1998. In vitro induction of ethoxyresorufin-o-deethylase and porphyrins by halogenated aromatic hydrocarbons in avian primary hepatocytes. Environ Toxicol Chem. 17;10:2006
Sargent L et al. 1991. Study of the separate and combined effects of the non-planar 2,5,2',5'- and the planar 3,4,3',4'-tetrachlorobiphenyl in liver and lymphocytes in vivo. Carcinogenesis 12;5:7930
SCAN. 2000. Opinion of the Scientific Committee on Animal Nutrition on the dioxin contamination of feedingstuffs and their contribution to the contamination of food of animal origin. EC, Brussels. Adopted 06 Nov 2000
SCF. 2000. Opinion of the Scientific Committee for Food on the risk assessment of dioxins and dioxin-like PCBs an food. EC, Brussels. Adopted Nov 2000. europa.eu.int/comm./food/Fs/sc/scf/out78_en.pdf
SCF. 2001. Opinion of the Scientific Committee for Food on the risk assessment of dioxins and dioxin-like PCBs an food. Update based on new scientific information available. Adopted 30th May 2001. europa.eu.int/comm./food/Fs/sc/scf/out90_en.pdf
Schantz SL, Bowman RE. 1989. Learning in monkeys exposed perinatally to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Neurotoxicol Teratol. 11;1:13
Schantz SL et al. 2001. Impairments of memory and learning in older adults exposed to polychlorinated biphenyls via consumption of Great Lakes fish. Environ Health Perspect. 109;6:605
Schantz SL et al. 2003. Effects of PCB exposure on neuropsychological function in children. Environ Health Perspect. 111;3:357
Scheuplein RJ, Bowers JC. 1995. Dioxin--an analysis of the major human studies: comparison with animal-based cancer risks. Risk Anal. 15;3:319
Schmidt K et al. 2005. Impact of PCB mixture (Aroclor 1254) and TBT and a mixture of both on swimming behavior, body growth and enzymatic biotransformation activities (GST) of young carp (Cyprinus carpio). Aquat Toxicol. 71;1:49
Schramm KW et al. 2000. Liver tumor-promoting activity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in TCDD-sensitive and TCDD-resistant rat strains. Cancer Res. 60;24:6911
SCOOP. 2000. Reports in tasks for scientific cooperation. Report of experts participating in task 3.2.5, Brussels 7.6.2000. DG-SANCO. europa.eu.int/comm/dgs/health_consumer/ library/pub/pub08_en.pdf
Seacat AM et al. 2003. Sub-chronic dietary toxicity of potassium perfluorooctanesulfonate in rats. Toxicol. 183;1-3:117
Sekizawa J, Matsuda T. 2002. Endogeneous ligands of arylhydrocarbon receptor and interactions with dioxins. Organohalogen Compds. 59:433
Shaw SD et al. 2003. Persistent organic pollutants (POPs) and immune function in US Atlantic coast harbor seals (Phoca vitulina concolor). Organohalogen Compds. 60-65. CD-ROM, Vol. 3, Section 1
Shirai JH, Kissel JC. 1996. Uncertainty in estimated half-lives of PCBS in humans: impact on exposure assessment. Sci Total Environ. 187;3:199
Shrader-Frechette KS. 1991. Risk and rationality : philosophical foundations for populist reforms. Univ California Press, Berkeley
Shu XO et al. 1999. Breast-feeding and risk of childhood acute leukemia. J Natl Cancer Inst. 91;20:1765
Simanainen U et al. 2004. Pattern of male reproductive system effects after in utero and lactational 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) exposure in three differentially TCDD-sensitive rat lines. Toxicol Sci. 2004 Apr 14
Simms W et al. 2000. Contaminant-related disruption of vitamin A dynamics in free-ranging harbor seal (Phoca vitulina) pups from British Columbia, Canada and Washington State, USA. Environ Toxicol Chem. 19;11:2844
Sjödin A et al. 2000. Influence of the consumption of fatty Baltic Sea fish on plasma levels of halogenated environmental contaminants in Latvian and Swedish men. Environ Health Perspect. 108:1035
Sjöåsen T et al. 1997. The otter (Lutra lutra) situation in Latvia and Sweden related to PCB and DDT levels. Ambio 26:196
Slovic P. 1987. Perception of risk. Science 236:280
Smialowicz RJ et al. 1990. Immune alterations in rats following subacute exposure to tributyltin oxide. Toxicol. 64;2:169
Smith AG, Gangolli SD. 2002. Organochlorine chemicals in seafood: occurrence and health concerns. Food Chem Toxicol. 40;6:767
SNFA. 2003. Delrapport 3 –dioxinanalyser av fet fish från Sverige 2000-2002. Swed Natl Food Admin. www.slv.se/
SNFA. 2004. Interim report 4 – Study of dioxin levels in fatty fish from Sweden 2000-2002. Swed Natl Food Admin. www.slv.se/
SNFA. 2005. Interim report 5- Study of dioxin-like PCB levels in fatty fish from Sweden 2000-2002. Swed Natl Food Admin. www.slv.se/
Sonnemann GW et al. 2002. Framework for the uncertainty assessment in the Impact Pathway Analysis with an application on a local scale in Spain. Environ Int. 28;1-2:9
Sormo EG et al. 2003. Polychlorinated biphenyls and organochlorine pesticides in Baltic and Atlantic gray seal (Halichoerus grypus) pups. Environ Toxicol Chem. 22;11:2789
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
Stewart PW et al. 2003. Cognitive development in preschool children prenatally exposed to PCBs and MeHg. Neurotoxicol Teratol. 25;1:11
Strandberg B et al. 1998b. Concentrations, biomagnification and spatial variation of organochlorine compounds in a pelagic food web in the northern part of the Baltic Sea. Sci Total Environ. 217;1-2:143
Strandberg B et al. 1998c. Concentrations and biomagnification of 17 chlordane compounds and other organochlorines in harbour porpoise Phocoena phocoena from southern Baltic Sea. Chemosphere 37:2513
Strandberg B et al. 1998d. Concentrations and spatial variations of cyclodienes and other organochlorines in herring and perch from the Baltic Sea. Sci Total Environ. 215;1-2:69
Stronkhorst J et al. 2002. Using the dioxin receptor-CALUX in vitro bioassay to screen marine harbor sediments for compounds with a dioxin-like mode of action. Environ Toxicol Chem. 21;12:2552
Stuart JS, Binzel RP. 2004. Bias-corrected population, size distribution, and impact hazard for the near-Earth objects. Icarus 170;2:295
Su M-C, Christensen ER. 1997. Apportionment of sources of polychlorinated dibenzo-p-dioxins and dibenzofurans by a chemical mass balance model. Water Res. 31;12:2935
Sumida H et al. 2005. Testes of rhesus monkeys exposed in utero and lactational period to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Organohalogen Compds. 2537-9. CD-ROM ID 738
Suter GW II (ed.). 1993. Ecological risk assessment. Lewis Publ, Boca Raton et al. 1993. 538 p
Svensson BG et al. 1991. Exposure to dioxins and dibenzofurans through the consumption of fish. New Engl J Med. 324;1:8
Svensson BG et al. 1994. Parameters of immunological competence in subjects with high consumption of fish contaminated with persistent organochlorine compounds. Int Arch Occup Environ Health 65;6:351

Svensson BG et al. 1995a. Fish consumption and exposure to persistent organochlorine compounds, mercury, selenium and methylamines among Swedish fishermen. Scand J Work Environ Health 21;2:96
Svensson BG et al. 1995b. Mortality and cancer incidence among Swedish fishermen with a high dietary intake of persistent organochlorine compounds. Scand J Work Environ Health 21;2:106
Sweetman AJ et al. 2000b. Declining PCB concentrations in the UK atmosphere: evidence and possible causes. Environ Sci Technol. 34:863
Tarhanen J et al. 1989. Toxic significance of planar aromatic compounds in Baltic ecosystem — New studies on extremely toxic coplanar PCBs. Chemosphere 18;1-6:1067
ten Tusscher GW et al. 2003. Persistent hematologic and immunologic disturbances in 8-year-old Dutch children associated with perinatal dioxin exposure. Environ Health Perspect. 111;12:1519
Thyen S et al. 2000. Organochlorine and mercury contamination of little terns (Sterna albifrons) breeding at the western Baltic Sea, 1978-96. Environ Pollut. 108;2:225
Tillitt DE et al. 1996. Dietary exposure of mink to carp from Saginaw Bay. 3. Characterization of dietary exposue to planar halogenated hydrocarbons, dioxin equivalents, and biomagnification. Environ Sci Technol. 30:283
Toide K et al. 2003. Aryl hydrocarbon hydroxylase represents CYP1B1, and not CYP1A1, in human freshly isolated white cells: Trimodal distribution of Japanese population according to induction of CYP1B1 mRNA by environmental dioxins. Cancer Epidemiol Biomarkers Prev. 12;3:219
Toyoshiba H et al. 2004. Evaluation of toxic equivalency factors for induction of cytochromes P450 CYP1A1 and CYP1A2 enzyme activity by dioxin-like compounds. Toxicol Appl Pharmacol. 194;2:156
Traas TP et al. 2001. Congener-specific model for polychlorinated biphenyl effects on otter (Lutra lutra) and associated sediment quality criteria. Environ Toxicol Chem. 20;1:205
Tryphonas H et al. 2001. Effects of toxaphene on the immune system of cynomolgus (Macaca fascicularis) monkeys. Food Chem Toxicol. 39;9:947
Tuomisto JT et al. 2004b. Risk-benefit analysis of eating farmed salmon. Science 305:478
TWGIM. 2004a. Baseline report on “Integrated monitoring of dioxins & PCBs in the Baltic Region” in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 09 Jan 2004
UBA. 2002. Unpublished commentary on SCF (2001) risk assessment and proposed TDIs. Umweltbundesamt, Berlin
Upton AC. 1994. Science and judgment in dioxin risk assessment: Needs and opportunities. Environ Health Perspect. 102:908
USEPA. 1998b. (Tissue residue based benchmark levels for dioxins to protect wildlife) www.epa.gov/waterscience/cs/vol1/appdx_d.pdf. (Ref. by Wenning et al. 2004)
USEPA. 2000a. Exposure and human health reassessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Draft final. USEPA, Washington, DC. EPA/600/6-88/005Ca. www.epa.gov/ncea/dei.html
van Birgelen AP. 1998. Hexachlorobenzene as a possible major contributor to the dioxin activity of human milk. Environ Health Perspect. 106;11:683
van Birgelen APJM et al. 1994. Toxic potency of 3,3',4,4',5-pentachlorobiphenyl relative to and in combination with 2,3,7,8-tetrachlorodibenzo-p-dioxin in a subchronic feeding study in the rat. Toxicol Appl Pharmacol. 127;2(1994):209
van Birgelen APJM et al. 1996a. Relative potencies of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls derived from hepatic porphyrin accumulation in mice. Toxicol Appl Pharmacol. 138;1:98
Van den Berg M et al. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ Health Perspect. 106:775
Van Den Heuvel RL et al. 2002. Immunologic biomarkers in relation to exposure markers of PCBs and dioxins in Flemish adolescents (Belgium). Environ Health Perspect. 110;6:595
Van der Burght AS et al. 1999. Structure-dependent induction of CYP1A by polychlorinated biphenyls in hepatocytes of cynomolgus monkeys (Macaca fascicularis). Toxicol Appl Pharmacol. 155;1:13
Van der Molen GW et al. 2000. Estimation of dioxin and furan elimination rates with a pharmacokinetic model. J Expo Anal Environ Epidemiol. 10;6 Pt 1:579
Van Leeuwen FX et al. 2000. Dioxins: WHO’s tolerable daily intake (TDI) revisited. Chemosphere 40;9-11:1095
Vartiainen T et al. 1997b. PCDDs and PCDFs in human milk from two areas of Finland. Chemosphere 34;12:2571
Vena J et al. 1998. Exposure to dioxin and nonneoplastic mortality in the expanded IARC international cohort study of phenoxy herbicide and chlorophenol production workers and sprayers. Environ Health Perspect. 106 Suppl 2:645
Verta M et al. accepted. Dioxin concentrations in sediments of the Baltic Sea - a survey of existing data. Chemosphere
Vetter W et al. 2001. Distribution and levels of eight toxaphene congeners in different tissues of marine mammals, birds and cod livers. Chemosphere 43;4-7:611
Viluksela M et al. 1998. Subchronic/chronic toxicity of a mixture of four chlorinated dibenzo-p-dioxins in rats. II. Biochemical effects. Toxicol Appl Pharmacol. 151;1:70
Virtanen JK et al. 2005. Mercury, fish oils, and risk of acute coronary events and cardiovascular disease, coronary heart disease, and all-cause mortality in men in eastern Finland. Arterioscler Thromb Vasc Biol. 25;1:228
Vreugdenhil HJ et al. 2004. Effects of perinatal exposure to PCBs on neuropsychological functions in the Rotterdam cohort at 9 years of age. Neuropsychol. 18;1:185
Vrijens B et al. 2002. Probabilistic intake assessment and body burden estimation of dioxin-like substances in background conditions and during a short food contamination episode. Food Addit Contam. 19;7:687
Vulykh N, Shatalov V. 2001. Investigation of dioxin/furan composition in emissions and in environmental media. Selection of congeners for modeling. MSC-E Tech Note 6/2001. MSC-E, Moscow. www.msceast.org/publications
Vuorinen PJ et al. 1998b. Comparisons and temporal trends of organochlorines and heavy metals in fish from the Gulf of Bothnia. Mar Pollut Bull. 36;3:236
Wagemann R, Muir DCG. 1984. Concentrations of heavy metals and organochlorines in marine mammals in northern waters: Overview and evaluation. Can Tech Report Fisheries Aquat Sci. 1279, pp. v+97. (Ref. by Berggren P et al. 1999).

Walker MK, Peterson RE. 1991. Potencies of polychlorinated dibenzo-p-dioxin, dibenzofuran, and biphenyl congeners, relative to 2,3,7,8-tetrachlorodibenzo-p-dioxin, for producing early life stage mortality in rainbow trout (Oncorhynchus mykiss). Aquat Toxicol. 21;3-4:219
Walker MK, Peterson RE. 1994. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to brook trout (Salvelinus fontinalis) during ealy development. Environ Toxicol Chem. 13;5817
Walker NJ et al. 2005. Dose-additive carcinogenicity of a defined mixture of "dioxin-like compounds". Environ Health Perspect. 113;1:43
Walter GL et al. 2000. Pathologic alterations in adult rainbow trout, Oncorhynchus mykiss, exposed to dietary 2,3,7,8-tetrachlorodibenzo-p-dioxin. Aquat Toxicol. 50;4:287
Wang C et al. 2004. Effects of omega-3 fatty acids on cardiovascular disease. Agency Healthcare Res Qual., Rockville, MD, USA, Mar 2004. AHRQ Publ. No 04-E009-2. www.ncbi.nlm.nih.gov/books/bv.fcgi?rid_hstat1a.chapter.38290
Wania F et al. 2001. A multicompartmental, multi-basin fugacity model describing the fate of PCBs in the Baltic Sea. Wulff F et al. (eds.) A systems analysis of the Baltic Sea. Ecol Studies 148:417-47. Springer Verlag, Berlin & Heidelberg
Warngard L et al. 1996. Mechanistical studies of the inhibition of intercellular communication by organochlorine compounds. Arch Toxicol Suppl. 18:149
Watanabe M et al. 2004. Association of male infertility with Pro185Ala polymorphism in the aryl hydrocarbon receptor repressor gene: implication for the susceptibility to dioxins. Fertil Steril. 82 Suppl 3:1067
Weihe P et al. 2003. Sustained high concentrations of PCBs in Faroese pregnant women despite dietary intervention. Organohalogen Compds. 60-65, CD-ROM Vol. 4, Section 4
Weisglas-Kuperus N et al. 2000. Immunologic effects of background exposure to polychlorinated biphenyls and dioxins in Dutch preschool children. Environ Health Perspect. 108;12:1203
Weisskopf MG et al. 2005. Maternal exposure to Great Lakes sport-caught fish and dichlorodiphenyl dichloroethylene, but not polychlorinated biphenyls, is associated with reduced birth weight. Environ Res. 97;2:149
Wenborn M et al. 1999. Releases of dioxins and furans to land and water in Europe. Final Report Issue 2. Report for LUW Nordrhein-Westfalen, Germany on behalf of EC DG-Environ, Sep 1999. Report AEAT-4703. europa.eu.int/comm/environment/dioxin/download.htm
Wenning RJ et al. 2004. Review of approaches used to establish sediment benchmarks for PCDD/Fs. Organohalogen Compds. 66:3497
Wheatley B, Paradis S. 1996. Balancing human exposure, risk and reality: Questions raised by the Canadian aboriginal methylmercury program. Neurotoxicol. 17;1:241
Wheatley B, Wheatley MA. 2000. Methylmercury and the health of indigenous peoples: a risk management challenge for physical and social sciences and for public health policy. Sci Total Environ. 259;1-3:23
WHO. 1993a. Guidelines for drinking-water quality. Vol. 1, Recommendations. 2nd ed., WHO, Geneva
WHO. 1998. Assessment of the health risk of dioxins: re-evaluation of the Tolerable Daily Intake (TDI), executive summary. WHO Consultation, May 25-29 1998, Geneva, Switzerland. WHO, Geneva. www.who.int/pcs/dioxin-exec-sum/exe-sum-final.doc
Wilson R, Crouch EAC. 1987. Risk assessment and comparisons: An introduction. Science 236:267
Windal I et al. 2003b. Non additive interactions in CALUX. Organohalogen Compds. 60-65. CD-ROM, Vol. 1, Section 3
Witt G et al. 2001. Using fluffy layer material to study the fate of particle-bound organic pollutants in the southern Baltic Sea. Environ Sci Technol. 35;8:1567
Wodarg D et al. 2004. A baseline study of polychlorinated biphenyl and hexachlorobenzene concentrations in the western Baltic Sea and Baltic Proper. Mar Chem. 87;1-2:23
Wolff MS et al. 1997. Proposed PCB congener groupings for epidemiological studies. Environ Health Perspect. 105;1:13
Wong EY et al. 2003. Comparative risk and policy analysis in environmental health. Risk Anal. 23;6:1337
Working PK. 1988. Male reproductive toxicology: comparison of the human to animal models. Environ Health Perspect. 77:37
Wulff F et al. 1993. A mass-balance model of chlorinated organic matter for the Baltic Sea - a challenge for ecotoxicology. Ambio 22:27
Yoshida K, Nakanishi J. 2003. Estimation of dioxin risk to Japanese from the past to the future. Chemosphere 53;4:427
Yoshida K et al. 2000. Assessment of human health risk of dioxins in Japan. Chemosphere 40;2:177
Zabel EW et al. 1995a. Interactions of polychlorinated dibenzo-p-dioxin, dibenzofuran, and biphenyl congeners for producing rainbow trout early life stage mortality. Toxicol Appl Pharmacol. 134;2(1995a):204
Zabel EW et al. 1996. Relative potencies of individual polychlorinated dibenzo-p-dioxin, dibenzofuran, and biphenyl congeners and congener mixtures based on induction of cytochrome P4501A mRNA in a rainbow trout gonadal cell line (RTG-2). Environ Toxicol Chem. 15;12:2310

References to Chapter 6

Allsopp M et al. 1994. Achieving zero dioxin, an emergency strategy for dioxin elimination. Greenpeace International 1994. 45 p
Andersson H. 2002. Critical review of legislation and recommendations protecting the Baltic Sea. Unpubl presentation, WWF Board of Trustees meeting, May 7th 2002
Anon. 1996-97. Environmental policy for a sustainable development. Norwegian Government's report to the Storting, White paper No. 58
Anon. 1999. Risiko, politik og miljø i det moderne samfund : en antologi om en aktuel kontrovers. Forlaget Sociologi, Fredriksberg
Anon. 2002. Finland's programme for the protection of the Baltic Sea – The Finnish Government's decision-in-principle 26.4.2002. Suomen Ympäristö 569. www.ymparisto.fi/SY569 Suomen Itämeren suojeluohjelma.pdf
Anon. 2002-2003. Norwegian Government's environmental policy and the environmental status of Norway. White paper No. 25
Ashford NA. 2002. Implementing a precautionary approach in decisions affecting health, safety, and the environment: risk, technology alternatives, and tradeoff analysis. Freytag et al. (eds.) The role of precaution in chemicals policy. Diplomatic Acad, Vienna. Favorita Papers 01/2002:128-40
Bar S et al. 2000. Closer co-operation, a new instrument for European environmental policy? Eur Integration Online Papers (EIoP) 4;13. www.eiop.or.at/eiop/texte/2000-013a.htm
Copenhagen Post Online News 13.8.1999. Double game in dioxin contamination issue. www.cphpost.dk/get/58104.html
DN (Dagens Nyheter) 28.11.2001. Frågan om dioxingränser delar V och MP. Lena Alfredsson.
Degnbol P et al. 2003. Integrating fisheries and environmental policies - Nordic experiences. TemaNord 2003:521. Nordic Council of Ministers, Copenhagen. 148 p

EC. 2000a. European Commission White paper on food safety. COM (1999) 719 final. EC, Brussels 12 Jan 2000. www.europa.eu.int/comm/dgs/health_consumer/library/pub/pub06_en.pdf
EC. 2001. Community strategy for dioxins, furans and polychlorinated biphenyls. Communication from the Commission to the Council, the European Parliament and the Economic and Social Committee. COM(2001)593, Final. Off J EC 17.11.2001, C322/2-18. www.europa.eu.int
EC. 2002b. Communication from the Commission on the reform of the common fisheries policy. Roadmap. European Commission, Brussels. 32 p. europa.eu.int/comm/fisheries/doc_et_publ/cfp_en.htm
EC. 2003b1. Proposal for a regulation of the European Parliament and of the Council on the persistent organic pollutants and amending Directives 79/117/EEC and 96/59/EC. EC, Brussels 12.6..2003 COM (2003) 333 final
EC. 2003b2. Proposal for a Council Decision concerning the conclusion, on behalf of the European Community, of the Stockholm Convention on Persistent organic pollutants. Brussels, 12.6.2003b2. COM(2003) 331 final, 2003/0118 (CNS)
EC. 2003c. A European environment and health strategy. Community strategy for environment and health (COM(2003)338 final). EC, Brussels. europa.eu.int/comm/environment/health/index_en.htm
EC. 2004. Synthesis of baseline reports in the framework of the European Environment and Health Strategy (COM(2003)338 final). EC, Brussels.
Eckley N. 2002. Dependable dynamism: lessons for designing scientific assessment processes in consensus negotiations. Global Environ Change 12:13
Eckley N, Selin H. 2003. Science, Politics, and Persistent Organic Pollutants. The Role of Scientific Assessments in International Environmental Co-operation. Int Env Agreements: politics, Law and Economics 3:17
Eckley N, Selin H. 2004. All talk, little action: Precaution and its effects on European Chemicals Regulation. J Eur Public Pol. 11;1:78
ENDS Daily 12.1.2000. EU Commission launches food safety initiative.
ENDS Daily 25.10.2001. EU sets out plan to reduce dioxins and PCBs.
ENDS Daily 26.11.2001. Sweden, Finland win dioxin in fish derogation.
Fairman R et al. 1998. Environmental risk assessment : approaches, experiences and information sources. Copenhagen, EEA. Environ Issues Ser. 4
Geiser K, Tickner J. 2003. New directions in European chemicals policies: Drivers, scope, and status. Lowell Center Sustainable Production. www.sustainableproduction.org/downloads/New%20Directions%20in%20European%20Chemicals%20Policy.pdf
Godduhn A, Duffy LK. 2003. Multi-generation health risks of persistent organic pollution in the far north: use of the precautionary approach in the Stockholm Convention. Environ Sci Pol. 6;4:341
Grandjean P. 2004. Implications of the precautionary principle for public health practice and research. Int J Occup Med Environ Health 17;1:5
Grandjean P. 2003. Unpublished Expression of Interest to EU's 5th Framework Programme for RTD.
Gray JS, Bewers JM. 1996. Towards a scientific definition of the precautionary principle. Mar Pollut Bull. 32:11:768
Haag D, Kaupenjohann M. 2001. Parameters, prediction, post-normal science and the precautionary principle - a roadmap for modelling for decision-making. Ecol Modell. 144;1:45
Harremoës P et al. 2002. (eds.). Late lessons from early warnings: the precautionary principle 1896–2000. Copenhagen, EEA. Environ Issue Report No 22
HELCOM. 1998b. Final report on the implementation of the 1988 Ministerial Declaration. Baltic Sea Environ Proc. 71
HELCOM. 2001a. Harmonization of HELCOM Recommendations with EU Directives and OSPAR Decisions and Recommendations. Final Report, Mar 2001a. Helsinki, Helsinki Commission.
HELCOM. 2001b. Activities 2001: Overview. Baltic Sea Environ Proc. 84
HELCOM. 2002a. Fourth periodic assessment of the state of the environment of the Baltic Marine Area, 1994-1998. Baltic Sea Environ Proc. 82B
HELCOM. 2002b. Implementing the HELCOM objective with regard to hazardous substances. Guidance document on dioxins. Helsinki, Helsinki Commission. www. helcom.fi/stc/files/environment/haz_subs/dioxins.pdf
Hildén M. 1997b. Risk, uncertainty, indeterminacy and ignorance in fisheries management – an analysis of management advice. Finn Environ Inst, Helsinki. 61 p. Monographs Boreal Environ Res. 5
Joas R et al. 2001. Effects on the fisheries industry of the Commission proposals (SANCO) on dioxin content of fish, fish oil and fish meal as part of animal feed regulation. Manuscript, Eur Parliament D-G Res Working Paper STOA 101 EN, 10-2001. Sci Technol Options Assess Series. Eur Parliament, Luxemburg
Joas R et al. 2002. Material flow based strategy for dioxin-like PCB levels in feed and food across the European Union and dietary intake estimation. Organohalogen Compds. 55(2002):291
Joerges C. 1996. Scientific expertise in social regulation and the European Court of Justice: Legal frameworks for denationalised governance structures. Joerges C, Ladeur K-H, Vos E. (eds.) Integrating scientific expertise into regulatory decision-making. RSC Working Papers 1996/10. hdl.handle.net/1814/1425
Johnston DM, VanderZwaag DL. 2000. The ocean and international environmental law: swimming, sinking, and treading water at the millennium. Ocean Coastal Manage. 43:141
Jordan A, Jeppesen T. 2000. EU Environmental policy: adapting to the principle of subsidiarity? Eur Environ. 10:64
Keita-Ouane F. 2003. UNEP Chemicals' work: Breaking the barriers to information access. Toxicol. 190;1-2:135
Kovalev V. 2004. Greenpeace takes Vodokanal to court. St. Petersburg Times 1.4.2004. www.sptimes.ru/archive/times/973/news/n_12605.htm
Krueger J, Selin H. 2002. Governance for sound chemical management: The need for a more comprehensive global strategy. Global Governance 8:323
Ledoux L, Turner RK. 2002. Valuing ocean and coastal resources: a review of practical examples and issues for further action. Ocean Coastal Manage. 45:583
Lok C, Powell D. 2000. The Belgian dioxin crisis of the summer of 1999: a case study in crisis communications and management. Technical Report #13, Dept Food Sci, Univ Guelph. www.foodsafetynetwork.ca/crisis/ belgian-dioxin-crisis-feb01-00.htm (last visited Apr 12 2005)
LUA-NRW. 2001. The European dioxin emission inventory stage II. North Rhine-Westphalia State Environ Agency. Prof. Bröker, Dr. Quass, Dr. Fermann. europa.int/comm./environment/dioxin/download.htm#stage2
Lyons G. 1999. Chemical trespass: a toxic legacy. A WWF-UK Report. WWF-UK, Jun 1999. Executive summary. www.wwf-uk.org
MoE. Norway's action plan for hazardous substances. Min of the Environ, Oslo 1999. 26 p

Neyer J. 2000. The regulation of risks and the power of the people: Lessons from the BSE crisis. Eur Integrat online Papers (EioP) 4;6. eiop.or.at/texte/2000-006a.htm
Quass U et al. 2004a. The European dioxin air emission inventory project--final results. Chemosphere 54;9:1319
Raffensperger C, Tickner J. (eds.) 1999. Protecting public health and the environment: Implementing the precautionary principle. Island Press, Washington, DC
Rikskansliet. 2000. The Swedish environmental objectives – Interim targets and action strategies. Summary of Gov. Bill 2000 / 01:130. http:www.regeringen.se/sb/d/108/a/1197
SCF. 2001. Opinion of the Scientific Committee for Food on the risk assessment of dioxins and dioxin-like PCBs an food. Update based on new scientific information available. Adopted 30th May 2001. http://europa.eu.int/comm./food/Fs/sc/scf/out90_en.pdf
Selin H, VanDeveer SD. 2002. Hazardous substances and the Helsinki and Barcelona Conventions: Origins, results and future challenges. Paper, Policy Forum Management of Toxic Substances in Marine Environment: Analysis of the Mediterranean and the Baltic, Javea, Spain 6-8 Oct 2002. www.helcom.fi/land/Hazardous/javeapolicypaper.pdf
Sijtsma R, Doring A. 2002. Feed industry efforts to reduce the risk of dioxin contamination of the food chain. Organohalogen Compds. 57:251
Stolzenberg H-C. 2000. Risk reduction in Germany for chlorinated paraffins used in metal working fluids: Regulator's view on triggers, driving forces, perspectives. Organohalogen Compds. 47:131
Tuomisto J. 2004. Is the precautionary principle used to cover up ignorance? Basic Clin Pharmacol Toxicol. 95;2:49
UNEP. 2003c. Master list of actions on the reduction and/or elimination of the releases of persistent organic pollutants. 5th ed., UNEP Chemicals, Geneva. www.pops.int/documents/meetings/inc7/mastlist5/ml5.pdf
UNEP. 2005. Standardized toolkit for the identification and quantification of dioxin and furan releases. 2nd ed., UNEP Chemicals, Geneva. www.pops.int
Weed DL. 2002. Environmental epidemiology: basics and proof of cause-effect. Toxicol. 181-182:399
Weed DL. 2004a. Methodologic implications of the precautionary principle: causal criteria. Int J Occup Med Environ Health 17;1:77
Weed DL. 2004b. Precaution, prevention, and public health ethics. J Med Philos. 29;3:313
Willis J. 2002. Precaution and the Stockholm Convention. Freytag et al. (eds.) The role of precaution in chemicals policy. Favorita papers 01/2002. Diplomatic Academy, Vienna. P. 174
WWF. 2005. Clean Baltic within Reach? How can a new chemical policy contribute to the protection of the Baltic Sea? Toxic chemicals- a threat to wildlife and humans. WWF DeTox Campaign, Brussels 2005.
WWF-Sweden. 2004. Press release 2.4.2004. www.wwf.se
WWF-Sweden. 2005. Miljögifter fortsatt hot mot östersjöfisk. Press release 25.1.2005. www.wwf.se
WWF-Sweden. Undated. Fisk till middag? WWFs konsumentguide för miljövänligare köp av fisk- och skaldjurprodukter. (Fish for dinner? WWF's consumer guide for more environmetal-friendly purcase of fish and shellfish, in Swedish). 3. ed. (undated). www.wwf.se
WWF-UK. 2002. WWF warning suppressed in toxic chemical debate. 7.8.2002. wwf-uk.org/news
WWF-UK. 2005. Baltic fish may be too toxic for Britain. 24.1.2005. wwf-uk.org/news

References to Chapter 7

Abad E et al. 2002a. Dioxin abatement strategies and mass balance at a municipal waste management plant. Environ Sci Technol. 36;1:92
Ashizawa AE et al. 2005. Human health research and policy development: experience in the Great Lakes region. Int J Hyg Environ Health 208;1-2:7
Assmuth TW. 1995. Toxicant distributions and impacts models in environmental risk analysis of waste sites. Publ Water Environ Res Inst. 20:1. Synthesis part of Doctoral thesis, Univ Helsinki Dept Environ Sci Limnol.
Barkovskii AL, Adriaens P. 1996. Microbial dechlorination of historically present and freshly spiked chlorinated dioxins and diversity of dioxin-dechlorinating populations. Appl Environ Microbiol. 62;12:4556
Baeyens W, Goyens L. 2000. Focus on new dioxin regulations in Europe. Environ Sci Pol. 3:65
Behnisch PA et al. 2002b. Low-temperature thermal decomposition of dioxin-like compounds in fly ash: combination of chemical analysis with in vitro bioassays (EROD and DR-CALUX). Environ Sci Technol. 36;23:5211
Bell JG et al. 2004. Dioxin and dioxin-like PCB content of farmed salmon flesh, the effect of feeding diets contanining marine fish oil or vegetable oils. Proc. 11th ISNFF, Phuket, Thailand:52
Bergqvist P-A et al. 2005. Kartläggning av utsläppskällor för oavsiktligt bildade ämnen: PCDD/F, PCB och HCB. Miljökemi, Kem Inst, Umeå Univ, Mar 2005. MK 2005:01
Berglund O et al. 2001. Influence of trophic status on PCB distribution in lake sediments and biota. Environ Pollut. 113;2:199
Berntssen MHG et al. 2005. Reducing the levels of dioxins and dioxin-like PCBs in Atlantic salmon by substitution of fish oil with vegetable oil in the feed. Aquacult Nutr. 11:219
Beurskens JEM et al. 1995. Dehalogenation of chlorinated dioxins by an anaerobic microbial consortium from sediment. Environ Toxicol Chem. 14:939
Birnbaum LS. 1994a. Endocrine effects of prenatal exposure to PCBs, dioxins, and other xenobiotics: implications for policy and future research. Environ Health Perspect. 102;8:676
Bonte JL et al. 2002. Catalytic destruction of PCDD/F in a fabric filter: experience at a municipal waste. Waste Manage. 22;4:421
Breivik H, Thorstad O. 2004. Removal of organic environmental pollutants from fish oil by short path distillation. The effect of a working fluid. Proc EuroFed Lipid Conf, 5-8 Sep 2004. (Ref. SPCFC 2005)
Bunge M et al. 2003. Reductive dehalogenation of chlorinated dioxins by an anaerobic bacterium. Nature 421;6921:357
Cahu C et al. 2004. Farmed and wild fish in the prevention of cardiovascular diseases: assessing possible differences in lipid nutritional values. Nutr Metab Cardiovasc Dis. 14:34
Carrier G et al. 1995a. Modeling of the toxickinetics of polychlorinated dibenzo-p-dioxins and dibenzofurans in mammalians, including humans. I. Nonlinear distribution of PCDD/PCDF body burden between liver an adipose tissues. Toxicol Appl Pharmacol. 131;2:253
Carrier G et al. 1995b. Modeling of the toxicokinetics of polychlorinated dibenzo-p-dioxins and dibenzofurans in mammalians, including humans. II. Kinetics of absorption and disposition of PCDDs/PCDFs. Toxicol Appl Pharmacol. 131;2:267

Casper RF et al. 1999. Resveratrol has antagonist activity on the aryl hydrocarbon receptor: implications for prevention of dioxin toxicity. Mol Pharmacol. 56;4:784
Chen ASC et al. 1997. Treating contaminated sediment with a two-stage base-catalyzed decomposition (BCD) process: bench-scale evaluation. J Haz Mater. 56;3:287
Connelly NA, Knuth BA. 1998. Evaluating risk communication: examining target audience perceptions about four presentation formats for fish consumption health advisory information. Risk Anal. 18;5:649
CSTEE. 1998. Opinion on the reports by Environmental Resources Management "Assessment of the risks posed by Pentachlorophenol (PCP) through the exposure of man and the environment to dioxins" and "Analysis of the advantage and drawbacks of further restrictions on the marketing and use of PCP", opinion at the 6th CSTEE plenary meeting, Brussels, 27 November 1998. Sci Committee Toxicity Ecotoxicity Environ. europa.eu.int/comm/health/ph_risk/committees/sct/docshtml/sct_out25_en.htm
Darnerud PO et al. 2003. Swedish consumption of fatty Baltic Sea fish in relation to the total dioxin intake and the recommended TDI. Organohalogen Compds.62:183
DEFRA. 2002. Regulatory impact assessment - Priority list of substances under article 16 of the Water Framework Directive. Dept Environ Food Rural Affairs, Jan 2002. www.defra.gov.uk/environment/water/wfd/art16-ria/
Degnbol P et al. 2003. Integrating fisheries and environmental policies - Nordic experiences. TemaNord 2003:521. Nordic Council of Ministers, Copenhagen. 148 p
De Kock J et al. 2004. Removal of dioxins and PCBs from marine oils: Current status and future developments. Proc 11th ISNFF, Phuket, Thailand:53. (Ref. SPCFC 2005)
De Meulenaer B et al. 2003. Selective adsorption of dioxins and PCB's from marine oils onm activated carbon. Organohalogen Compds. 60-65 (CD-ROM), Vol. 1, Section 1
de Percin PR. 1995. Application of thermal desorption technologies to hazardous waste sites. J Haz Mater. 40;2:203
desRosiers PE. 1989. Chemical detoxification of dioxin-contaminated wastes using potassium polyethylene glycolate. Chemosphere 18;1-6:343
Detzel A et al. 1998. Investigation of emissions and abatement measures for persistent organic pollutants in the Federal Republic of Germany. Umweltbundesamt, Berlin. Texte 75/98. Res Report 295 44 365, UBA-FB 98-115/e. IFEU GmbH. 291 p
Deutch B et al. 2003. Smoking as a determinant of high organochlorine levels in Greenland. Arch Environ Health 58;1:30
Deweerd KA, Bedard DL. 1999. Use of halogenated benzoates and other halogenated aromatic compounds to stimulate the microbial dechlorination of PCBs. Environ Sci Technol. 33;12:2057
Du X et al. 2001. [Microbial degradation of polychlorinated dibenzo-p-dioxins, in Chinese with English abstract] Huan Jing Ke Xue 22;3:97
EC. 2001. Community strategy for dioxins, furans and polychlorinated biphenyls. Communication from the Commission to the Council, the European Parliament and the Economic and Social Committee. COM(2001)593, Final. Off J EC 17.11.2001, C322/2-18. www.europa.eu.int
EC. 2002a. Commission recommendation of 4 March 2002 on the reduction of the presence of dioxins, furans and PCBs in feedingstuffs and foodstuffs. Notified under number C(2002) 836. Off J EC 9.3.2002a. L 67/69 (2002/201/EC)
EC & EU. 2002. Submission by the European Community and the member states of the European Union to the sixth session of the Intergovernmental NegotiatingCommittee for an international legally binding instrument for international action on certain persistent organic pollutants (previously UNEP/POPS/INC.6/CRP.6): Best available techniques (BAT) and best environmental practices (BEP) for reducing and/or eliminating emission of POPs by-products. Appendix to Report by the Secretariat on preparatory work for the meeting of UNEP Expert Group on Best Available Techniques and Best Environmental Practices, Res Triangle Park, 10-14 Mar 2003. UNEP/POPS/EGB.1/INF/5, 19 Nov 2002. 17 p
Eduljee GH et al. 1997. The effect of changing waste management practices on PCDD/PCDF releases from household waste recycling and disposal processes. Chemosphere 34;5-7:1615
Engwall M et al. 1999. Levels of dioxin-like compounds in sewage sludge determined with a bioassay based on EROD induction in chicken embryo liver cultures. Chemosphere 38;10:2327
Entec UK Ltd. 2003. Development of UK cost curves for abatement of dioxin emissions to air. Report prepared for Dept Env Food Rural Affairs (DEFRA). Final Rep, Draft for consultation, Nov 2003. www.defra.gov.uk/corporate/consult/dioxins-two/report2.pdf
Environment Canada. 2005. Proposed risk management strategy for pentachlorobenzene and tetrachlorobenzenes. Natl Office Pollut Prevent, Chemicals Control Branch, Environ Canada. Comment Draft Jan 2005. www.ec.gc.ca/nopp/DOCS/consult/CBz/en/p5.cfm
Eppe G et al. 2005. Removal of PCDD/Fs and DL-PCBs from fish oil by activated carbon: Compliance with European legislation. Organohalogen Compds. 1412-6. CD-ROM ID 1931
ERM. 1997. Assessment of the risks posed by pentachlorophenol (PCP) through the exposure of man and the environment to dioxins. Environmental Resources Management Ltd. Final Report to EC/GG III, Dec. 1997.
ERM. 1998. Analysis of the advantage and drawbacks of further restrictions on the marketing and use of PCP. Environmental Resources Management Ltd. Draft final report to EC/DG III, Mar 1998.
Fiedler H. 1996. Sources of PCDD/PCDF and impact on the environment. Chemosphere 32;1:55
Fischer R et al. 2003. Effects of sinter mix composition upon the formation of PCDD/Fs in iron ore sintering. Organohalogen Compds. 60-65 (CD-ROM)
Flesch-Janys D et al. 1996. Elimination of polychlorinated dibenzo-p-dioxins and dibenzofurans in occupationally exposed persons. J Toxicol Environ Health 47;4:363
Foran JA et al. 2005b. Quantitative analysis of the benefits and risks of consuming farmed and wild salmon. J Nutr. 135;11:2639
François F et al. 2000. Reduction of the PCDD/PCDF emission in the Flemish Region (Belgium). Organohalogen Compds. 45:352
Friesen EN et al. 2005. Use of dietary flaxseed oil to reduce PCB loadings in farmed sablefish. Organohalogen Compds. CD-ROM ID 2075
FRTR. 2004. Cost & performance remediation case studies and related information. Fed Remedial Technologies Roundtable, 5th ed. Jun 2004. EPA 542-C-04-004. www.frtr.gov
Geusau A et al. 1999. Olestra increases faecal excretion of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Lancet 354;9186:1266
Gray KA, Hilarides RJ. 1995. Radiolytic treatment of dioxin contaminated soils. Rad Phys Chem. 46;46:1081
Ha DTC et al. 2004. Biodegradation of 2,3,7,8 TCDD by anaeorobic and aerobic microcosms from bioremediation treatments for cleaning up dioxin contaminated soils. Organohalogen Compds. 66:3695

Habe H et al. 2002. Degradation characteristics of a dibenzofuran-degrader Terrabacter sp. strain DBF63 toward chlorinated dioxins in soil. Chemosphere 48;2:201
Hansson S. Unpublished. Fisheries and the cycling of dioxin-like compounds in the Baltic. Paper, workshop on Dioxins in Baltic Sea fish - Scientific basis and information needs of assessment and management, SYKE, Helsinki, 12.-13.6.2003
Haraldsdottir J. 1999. Dietary guidelines and patterns of intake in Denmark. Br J Nutr. 81 Suppl 2:S43
Hashimoto S et al. 2004. Remediation of soil contaminated with dioxins by subcritical water extraction. Chemosphere 54;1:89
Hedman B et al. 2005. Emissions from small-scale energy production using co-combustion of biofuel and the dry fraction of household waste. Waste Manage. 25;3:311
HELCOM. 2002a. Fourth periodic assessment of the state of the environment of the Baltic Marine Area, 1994-1998. Baltic Sea Environ Proc. 82B:1
Hildén M. Assmuth T. Miljöfrågorna är en utmaning för medierna. Debattartikel, Hufvustadsbladet 12.3.2002.
Hildén M. Assmuth T. Förenkla lagom om Östersjön. Debattartikel, Hufvustadsbladet 20.3.2002.
Hites RA et al. 2004a. Global assessment of organic contaminants in farmed salmon. Science 303;5655:226
Huisman M et al. 1995b. Perinatal exposure to polychlorinated biphenyls and dioxins through dietary intake. Chemosphere 31;10:4273
Hwang SY et al. 2004. Panax ginseng improves survival and sperm quality in guinea pigs exposed to 2,3,7,8-tetrachlorodibenzo-p-dioxin. Br J Urol Int. 94;4:663
ICES. 2003a. Report of the ICES Advisory Committee on Ecocystems, 2003. ICES Cooperat Res Report 262:1
ICES. 2003b. Report of the ICES Advisory Committee on the Marine Environment, 2003. ICES Cooperat Res Report 263:1
ICES. 2005c. Report of the Baltic Fisheries Assessment Working Group (WGBFAS). Hamburg, 12-21 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:19
IOM. 2003. Dioxins and dioxin-like compounds in the food supply: Strategies to decrease exposure. Committee on Implications of Dioxin in the Food Supply, Food Nutr Bd, Inst Med of the Natl Academies. Natl Acad Press, Washington, DC
Isosaari P et al. 2001. Use of olive oil for soil extraction and ultraviolet degradation of polychlorinated dibenzo-p-dioxins and dibenzofurans. Environ Sci Technol. 35;6:1259
Isosaari P et al. 2002b. Feeding trial on rainbow trout: comparison of dry fish feed and Baltic herring as a source of PCDD/Fs and PCBs. Chemosphere 48;8:795
Isosaari P et al. 2004. Accumulation and distribution of polychlorinated dibenzo-p-dioxin, dibenzofuran, and polychlorinated biphenyl congeners in Atlantic salmon (Salmo salar). Environ Toxicol Chem. 23;7:1672
Jacobs M et al. 2002a. Investigation of polychlorinated dibenzo-p-dioxins, dibenzo-p-furans and selected coplanar biphenyls in Scottish farmed Atlantic salmon (Salmo salar). Chemosphere 47;2:183
Jacobs MN et al. 2002b. Investigation of selected persistent organic pollutants in farmed Atlantic salmon (Salmo salar), salmon aquaculture feed, and fish oil components of the feed. Environ Sci Technol. 36;13:2797
Joas R et al. 2001. Effects on the fisheries industry of the Commission proposals (SANCO) on dioxin content of fish, fish oil and fish meal as part of animal feed regulation. Manuscript, Eur Parliament D-G Res Working Paper STOA 101 EN, 10-2001. Sci Technol Options Assess Series. Eur Parliament, Luxemburg
Kamimura H et al. 2005. Enhanced faecal excretion of 2,3,4,7,8-pentachlorodibenzofuran in rats by a long-term treatment with activated charcoal beads. Xenobiotica 18;5:585
Kao CM et al. 2001. Evaluation of TCDD biodegradability under different redox conditions. Chemosphere 44;6:1447
Kasai E et al. 2000. Thermal remediation of PCDD/Fs contaminated soil by zone combustion process. Chemosphere 41;6:857
KemI. 2003. Bromerade flamskyddsmedel – förutsättningar för ett nationellt förbud. Rapport från ett regeringsuppdrag. KemI Rapport 4/03. www.kemi.se
KemI. 2004. Dekabromdifenyleter (dekaBDE) - underlag för ett nationellt förbud. Rapport från ett regeringsuppdrag. KemI Rapport 5/04.
KemI. 2005. Försålda kvantiteter av bekämpningsmedel 2004. Sold quantities of pesticides. www.kemi.se/
Kim SC et al. 2001. Removal efficiencies of PCDDs/PCDFs by air pollution control devices in municipal solid waste incinerators. Chemosphere 43;4-7:773
Kim S-C et al. 2003a. PCDDs/PCDFs emission from ferrous metal industry. Organohalogen Compds. 60-65. CD-ROM, Vol. 4, Section 1
Kim S-C et al. 2003b. PCDDs/PCDFs emission from nonferrous metal industry. Organohalogen Compds. 60-65. CD-ROM, Vol. 4, Section 1
Kitamura K et al. 2005. Effect of chlorophyllin-chitosan on excretion of dioxins in a healthy man. Environ Sci Technol. 39;4:1084
Koppe JG. 1995. Nutrition and breast-feeding. Eur J Obstet Gynecol Reprod Biol. 61;1:73
Klimm C et al. 1998. Formation of octa- and heptachlorodibenzo-p-dioxins during semi anaerobic digestion of sewage sludge. Chemosphere 37;9-11:2003
Krauss T et al. 1994. Formation of PCDD/PCDF during composting? Chemosphere 28;1:155
Kris-Etherton PM et al. 2003. Fish consumption, fish oil, omega-3 fatty acids, and cardiovascular disease. Arterioscler Thromb Vasc Biol. 23;2:e20-30. Erratum in: Arterioscler Thromb Vasc Biol. 23;2:e31. Comment in: Arterioscler Thromb Vasc Biol. 23;2:151
Kruse S, Meng W. 2005. Risk management of dioxins in feed by the German Federal Ministry of Consumer Protection. Organohalogen Compds. 2470-2. CD-ROM ID 362
Kuikka S et al. 2004. Kuluttajaan kohdistuvan dioksiiniriskin pienentäminen: markkina- ja ekosysteemitiedon yhdistäminen päätöksenteon näkökulmasta (DIOMAR). Unpublished project proposal to Finn Min Agr Forestry 25.3.2004. 15 p
Lind Y et al. 2002. Exponering för organiska miljökontaminanter via livsmedel – intagsberäkningar av ΣPCB, PCB 153, p,p'-DDE, PCDD/F, dioxinlika PCB, PBDE och HBCD baserade pa konsumtionsdata från Riksmaten 1997-98. Uppsala, Swed Natl Food Authority. Livsmedelsverket Rapport 26
Lozano RB, Pratt JR. 1994. Interaction of toxicants and communities: The role of nutrients. Environ Toxicol Chem. 13;3:361
Mackenzie BR et al. 2004. Fish, fishing, and pollutant reduction in the Baltic Sea. Environ Sci Technol. 38;7:1970
Marklund S et al. 1992. Formation and degradation of chlorinated aromatic compounds in an air pollution control device for MSW combustor. Chemosphere 25;1-2:139
Mills WJ et al. 2002. Quantitative estimates of polychlorinated biphenyls emissions to ambient air from a storage site and cleanup project. Organohalogen Compds. 56:253
Moon J-Y et al. 2005. Panax ginseng extracts accelerate TCDD excretion in rats. Organohalogen Compds. 2564. CD-ROM ID 2050.

Mori C et al. 2005. Reduction of dioxins and polychlorinated biphenyls (pcbs) in human body using colestimide. Organohalogen Compds. 1610-3. CD-ROM ID 2266
Mousa MA et al. 1998. Altered biologic activities of commercial polychlorinated biphenyl mixtures after microbial reductive dechlorination. Environ Health Perspect. 106 Suppl 6:1409
Muir T, Alaee M. 2002. Costs and benefits of brominated flame retardants (BFRs) and alternatives. Organohalogen Compds. 58:237
Nakamiya K et al. 2000. Biodegradation of dioxins by activated sludge. Organohalogen Compds. 45:468B
NCM. 1996. Nordiska näringsämnesrekommendationer. Nordic Council of Ministers, Copenhagen. Nord 28.
Neurath C. 2003. Open burning of domestic wastes: the single largest source of dioxin. Organohalogen Compds. 60-65, CD-ROM Vol. 4, Section 1
NRC. 2001. A risk-management strategy for PCB-contaminated sediments. Committee on Remediation of PCB-Contaminated Sediments, Natl Res Council of the Natl Academies, Natl Acad Press, Washington, DC
Oku A et al. 1995. Destruction of PCDDs and PCDFs. A convenient method using alkali-metal hydroxide in 1,3-dimethyl-2-imidazolidinone (DMI). Chemosphere 39;8:3873
Ojaveer E, Lehtonen H. 2001. Fish stocks in the Baltic Sea: Finite or infinite resource? Ambio 30;4-5:217
OTA. 1991. Dioxin treatment technologies. Off Technol Assess, US Congress, Nov 1991. OTA-BP-O-93. NTIS order #PB92-152511. 71 p
Pagano JJ et al. 1995. Reductive dechlorination of PCB-contaminated sediments in an anaerobic bioreactor system. Environ Sci Technol. 29:2584
Pandelova N et al. 2003. Primary measures for reduction of PCDD/F n co-combustion of lignite coal and waste: Effect of various inhibitors. Organohalogen Compds. 60-65
Pelletier C et al. 2002. Associations between weight loss-induced changes in plasma organochlorine concentrations, serum T(3) concentration, and resting metabolic rate. Toxicol Sci. 67;1:46
Peterson R, Milicic E. 1992. Chemical treatment of dioxin residues from wastewater processing. Chemosphere 25;7-1:1565
Phillipson BE et al. 1985. Reduction of plasma lipids, lipoproteins, and apoproteins by dietary fish oils in patients with hypertriglyceridemia. N Engl J Med. 312;19:1210
Pignatello JJ, Huang LQ. 1993. Degradation of polychlorinated dibenzo-p-dioxin and dibenzofuran contaminants in 2,4,5-T by photoassisted iron-catalyzed hydrogen peroxide. Water Res. 27;12:1731
Pluim HJ et al. 1994b. Influence of short-term dietary measures on dioxin concentrations in human milk. Environ Health Perspect. 102;11:968
Ponce RA et al. 2000. Use of quality-adjusted life year weights with dose response models for public health decisions: a case study of the risks and benefits of fish consumption. Risk Anal. 20;4:529
Poster DL et al. 2003. Degradation of PCBs in a marine sediment treated with ionizing and UV radiation. Environ Sci Technol. 37;17:3808
Ragazzi M, Sibisi N. 2003. PCDD/F emissions and mass balance from municipal solid waste and RDF combustion facilities: a comparison. Organohalogen Compds. 60-65. CD-ROM Vol. 4, Section 1
Raghunathan K, Gullett BK. 1996. Role of sulfur in reducing PCDD and PCDF formation. Environ Sci Technol. 30:1827
Rappe C et al. 1989c. Formation of PCDDs and PCDFs by the chlorination of water. Chemosphere 19;12:1875
Rappe C et al. 1990. Levels of polychlorinated dioxins and dibenzofurans in commercial detergents and related products. Chemosphere 21;1-2:43
Renwick AG et al. 2003. Risk characterisation of chemicals in food and diet. Food Chem Toxicol. 41;9:1211
RPA. 2002. Scope for the use of economic instruments for selected persistent pollutants. Risk & Pol Anal. Ltd. Report to DEFRA. Jul 2002
SACN and COT. 2004. Advice on fish consumtion: benefits & risks. Sci Advisory Committee Nutr and Committee Toxicol. The Stationery Office, London. www.food.gov.uk/multimedia/pdfs/fishreport2004full.pdf
Sakurai K, Todaka E, Saito Y, Mori C. Pilot study to reduce dioxins in the human body. Intern Med. 43;9(2004):792
Schlummer M et al. 2002. Recycling of technical polymers from electronic waste while eliminating brominated flame retardadants and PBDD/F. Organohalogen Compds. 56:417
Schnare DW, Robinson PC. 1986. Reduction of the human body burdens of hexachlorobenzene and polychlorinated biphenyls. IARC Sci Publ. 77:597
Schnare DW et al. 1982. Evaluation of a detoxification regimen for fat stored xenobiotics. Med Hypotheses 9;3:265
Schnare DW et al. 1984. Body burden reductions of PCBs, PBBs and chlorinated pesticides in human subjects. Ambio 13:378
Seierstad SL et al. 2005. Dietary intake of differently fed salmon; the influence on markers of human atherosclerosis. Eur J Clin Invest. 35;1:52-9
Shibata E et al. 2003. Formation behavior of PCDD/Fs in PVC pyrolysis with copper oxide. Chemosphere 50;9:1235
Sidhu KS. 2003. Health benefits and potential risks related to consumption of fish or fish oil. Regulat Toxicol Pharmacol. 38;3:336
SNV. 1997. Statens naturvårdsverks föreskrifter om spridning av kemiska bekämpningsmedel; beslutade den 2 december 1996. SNFS 1997:2
SNV & Boverket. 2002. Omhändertagande av PCB i byggnader. Redovisning av regeringsuppdrag M2002/1114/Kn. SNV:s diarienr. 643-2429-02
Souta I et al. 2004. Bio-remediation in a field with dioxin contaminated soil using a lysimeter. Organohalogen Compds. 66:1269
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
SPIN. 2005. SPIN on the Internet. Substances in Preparations in Nordic countries. www.spin2000.net/spin.html
Stolzenberg H-C. 2000. Risk reduction in Germany for chlorinated paraffins used in metal working fluids: Regulator's view on triggers, driving forces, perspectives. Organohalogen Compds. 47:131
Takada S et al. 1996. Degradation of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans by the white rot fungus Phanerochaete sordida YK-624. Appl Environ Microbiol. 62;12:4323
Takasuga T et al. 2003. Quantitative analysis of toxic compounds formed from combustion of some plastic materials and newspaper. Organohalogen Compds. 60-65. CD-ROM Vol. 4, Section 1
Taoda H et al. 2000. Destruction of PCDD/Fs and coplanar PCBs in flue gas from waste incineration by photocatalyst. Organohalogen Compds. 45:400
Thomae TL et al. 2005. Transforming growth factor-beta 3 restores fusion in palatal shelves exposed to 2,3,7,8-tetrachlorodibenzo-p-dioxin. J Biol Chem. 2005 Jan 24; Epub ahead of print
Tiedje JM et al. 1993-94. Microbial reductive dechlorination of PCBs. Biodegrad. 4;4:231
Tiernan TO et al. 1989. Dechlorination of PCDD and PCDF sorbed on activated carbon using the KPEG reagent. Chemosphere 19;1-6:573
Tilden J et al. 1997. Health advisories for consumers of Great Lakes sport fish: is the message being received? Environ Health Perspect. 105:1360
Toji A, Kusuda T. 2002. Characteristics of dioxins removal by coagulation and sedimentation process. Organohalogen Compds. 58:145
UNEP. 1998. Inventory of world-wide PCB destruction capacity. UNEP Chemicals, Geneva. www.chem.unep.ch

UNEP. 2000. Survey of currently available non-incineration PCB destruction technologies. UNEP Chemicals, Geneva. www.chem.unep.ch
UNEP. 2001. Destruction and decontamination technologies for PCB and other POPs wastes. Part III. UNEP Chemicals, Geneva. www.basel.int
UNEP. 2004c. Draft guidelines on best available techniques (BAT) and guidance on best environmental practices (BEP) relevant to the provisions of Article 5 and Annex C of the Stockholm Convention. UNEP/POPS/EGB.3/2. Draft 29 Jul 2004. UNEP Chemicals, Geneva. www.pops.int/documents/meetings/bat_bep/3rd_session/Default.htm
UNEP. 2004d. Draft guidelines on best available techniques and provisional guidance on best environmental practices. UNEP Chemicals, Geneva. www.pops.int
UNEP. 2004e. Review of the emerging, innovative technologies for the destruction and decontamination of POPs and the identification of promising technologies for use in developing countries. UNEP Chemicals, Geneva. www.unep.org/stapgef
UNEP. 2004f. Technical guidelines for environmentally sound management of wastes consisting of, containing or contaminated with polychlorinated biphenyls, polychlorinated terphenyls or polybrominated biphenyls. Conf Parties to the Basel Convention Control Transboundary Movements of Haz Wastes and Their Disposal 7th meeting, Geneva, 25–29 Oct 2004
USEPA. 2003a. Superfund Innovative Technology Evaluation (SITE) Program. Technology profiles. 11th ed. Vols. 1+2. NRMRL, ORD, USEPA, Cincinnati, OH, Sept 2003. EPA/540/R-03/501.
USEPA. 2003b. On-site incineration: Overview of Superfund operating experience. USEPA, Washington, DC. www.epa.gov
USEPA. 2004 (2002). Great Lakes Strategy 2002 - A Plan for the New Millennium. A Strategic Plan for the Great Lakes Ecosystem. www.epa.gov/glnpo/gls/gls02.html
Van Leeuwen FRX, Younes M. 1998. WHO revises the Tolebale Daily Intake (TDI) for dioxins. Organohalogen Compds. 38:295
Vartiainen T et al. 1997c. Ympäristömyrkkyjen kertyminen silakkaaan. Ympäristö ja Terveys 28;7-8:18
Vesterinen R, Flyktman M. 1996. Organic emissions from co-combustion of RDF with wood chips and milled peat in a bubbling fluidized bed boiler. Chemosphere 32;4:681
Weber R. 2004. Relevance of PCDD/PCDF formation in the evaluation of POPs destruction technologies – Necessary and current status. Organohalogen Compds. 66:1273
Weihe P et al. 2003. Sustained high concentrations of PCBs in Faroese pregnant women despite dietary intervention. Organohalogen Compds. 60-65, CD-ROM Vol. 4, Section 4
WHO. 2004. Chlorobenzenes other than hexachlorobenzene: Environmental aspects. Concise Int Chem Assess Doc 60
Wittich RM. 1998. Degradation of dioxin-like compounds by microorganisms. Appl Microbiol Biotechnol. 49;5:489
Yadav JS et al. 1995. Degradation of polychlorinated biphenyl mixtures (Aroclors 1242, 1254, and 1260) by the white rot fungus Phanerochaete chrysosporium as evidenced by congener-specific analysis. Appl Environ Microbiol. 61;7:2560
Yak HK et al. 2000. Relative resistance of positional isomers of polychlorinated biphenyls toward reductive dechlorination by zerovalent iron in subcritical water. Environ Sci Technol. 34:2792
Yasuhara A et al. 2005. The role of metals in dioxin formation from combustion of newspapers and polyvinyl chloride in an incinerator. Chemosphere 58;7:891
Ye D et al. 1992. Anaerobic dechlorination of polychlorobiphenyls (Aroclor 1242) by pasteurized and ethanol-treated microorganisms from sediments. Appl Environ Microbiol. 58;4: 1110
Zabik ME, Zabik MJ. 1999. Polychlorinated biphenyls, polybrominated biphenyls, and dioxin reduction during processing/cooking food. Adv Exp Med Biol. 459:213
Zabik ME, Zabik MJ. 1995. Tetrachlorodibenzo-p-dioxin residue reduction by cooking/processing of fish fillets harvested from the Great Lakes. Bull Environ Contam Toxicol. 55;2:264
Zabik ME et al. 1996. Pesticide residues, PCBs and PAHs in baked, charbroiled, salt boiled and smoked Great Lakes lake trout. Food Chem. 55;3:231

References to Chapter 8

Ahmed FE et al. 1993. Risk assessment and management of chemical contaminants in fishery products consumed in the USA. J Appl Toxicol. 13;6:395
Andresen S. 1996. Implementation of international environmental commitments: the case of the Northern Seas. Sci Total Environ. 186;1-2:149
Anon. 2000a. (Pentachlorophenol and dioxins)
Anon. 2003. Dioxins & PCBs: Environmental levels and human exposure in candidate countries. Progress report III, Sept 2003. Ref: ENV.C.2/SER/2002/0085. Consortium: Environ Levels In Candidate Countries (ELICC). EC, Brussels. 17 p
Anon. 2004. Dioxins & PCBs: Environmental levels and human exposure in candidate countries. Final report. Reference: ENV.C.2/SER/2002/0085. Consortium: Environmental Levels In Candidate Countries (ELICC) under Supervision of Gunther Umlauf (JRC). EC, Brussels, 16.6.2004
Ball DJ. 2002. Environmental risk assessment and the intrusion of bias. Environ Int. 28;6:529
Beaton GH. 2003. Dietary guidelines: some issues to consider before initiating revisions. J Am Diet Assoc. 103;12 Suppl 2:S56
Belliveau M. 2003. Dioxin pollution prevention and PVC plastic in municipal solid waste: Precautionary state policy. Organohalogen Compds. 65. CD-ROM, Vol. 6, Section 4
Bethune C et al. 2005. Circulating contaminant and fatty acid levels in patients after a controlled diet of farmed salmon. Organohalogen Compds. 1756-9. CD-ROM ID 2163
Booth DE. 1994. Ethics and the limits of environmental economics. Ecol Econ. 9;3:241
Buekens A et al. 2002a. Dioxin-laden residual streams from thermal and metallurgical processes: Inventory and management. Organohalogen Compds. 56:217
Buekens A et al. 2002b. Minization od dioxins in thermal and industrial processes: Mechanisms, monitoring and abatement (MINDIP): Main achievements of this EU-project. Organohalogen Compds. 56:341
Chen G al. 2001. Synthesis of polybrominated diphenyl ethers and their capacity to induce CYP1A by the Ah receptor mediated pathway. Environ Sci Technol. 35;18:3749
Cohen J et al. 2005. A quantitative risk-benefit analysis of changes in population fish consumption. Am J Prev Med. 29;4: 325-34
Collins A et al. 1998. Fishery-pollution Interactions: A modelling approach to explore the nature and incidence of economic damages. Mar Pollut Bull. 36;3:211

Crettaz P et al. 2002. Assessing human health response in life cycle assessment using ED10s and DALYs: part 1--Cancer effects. Risk Anal. 22;5:931
Degnbol P et al. 2003. Integrating fisheries and environmental policies - Nordic experiences. TemaNord 2003:521. Nordic Council of Ministers, Copenhagen. 148 p
Dellinger JA. 2004. Exposure assessment and initial intervention regarding fish consumption of tribal members of the Upper Great Lakes Region in the United States. Environ Res. 95;3:325
Detzel A et al. 1998. Investigation of emissions and abatement measures for persistent organic pollutants in the Federal Republic of Germany. Umweltbundesamt, Berlin. Texte 75/98. Res Report 295 44 365, UBA-FB 98-115/e. IFEU GmbH. 291 p
Downs TJ, Ambrose RF. 2001. Syntropic ecotoxicology: A heuristic model for understanding the vulnerability of ecological systems to stress. Ecosystem Health 7;4:266
EC. 2001. Community strategy for dioxins, furans and polychlorinated biphenyls. Communication from the Commission to the Council, the European Parliament and the Economic and Social Committee. COM(2001)593, Final. Off J EC 17.11.2001, C322/2-18. www.europa.eu.int
EC. 2002a. Commission recommendation of 4 March 2002 on the reduction of the presence of dioxins, furans and PCBs in feedingstuffs and foodstuffs. Notified under number C(2002a) 836. Off J EC 9.3.2002 L 67/69 (2002/201/EC)
Efroymson RA et al. 2004. A framework for net environmental benefit analysis for remediation or restoration of contaminated sites. Environ Manage. 34;3:315
Egeland GM, Middaugh JP. 1997. Balancing fish consumption benefits with mercury exposure. Science 278;5345:1904
Fagt S et al. 2002. Danskernes kostvaner 2000-2001. Udviklingen I danskernes kost – forbrug, indkøb og vaner. Fødevaredirektoratet, Søborg 2002. FødevareRapport 2002:10.
FAO & WHO 1997a. Risk management and food safety : report of a joint FAO/WHO consultation, Rome, Italy, 27 to 31 January 1997. FAO, Rome. FAO Food Nutr Paper 65
Fiedler H. 2003b. (Textile industry dioxin abatement options)
Foran JA et al. 2005a. Risk-based consumption advice for farmed Atlantic and wild Pacific salmon contaminated with dioxins and dioxin-like compounds. Environ Health Perspect. 113;5:552
Foran JA et al. 2005b. Quantitative analysis of the benefits and risks of consuming farmed and wild salmon. J Nutr. 135;11:2639
François F et al. 2000. Reduction of the PCDD/PCDF emission in the Flemish Region (Belgium). Organohalogen Compds. 45:352
François F et al. 2002. Dioxin emission reduction at non-ferrous metal plants in the Flemish Region (Belgium) – Enforcement approach of the Environment Inspection Section. Organohalogen Compds. 56:421
FRTR. 2004. Cost & performance remediation case studies and related information. Fed Remedial Technologies Roundtable, 5th ed. Jun 2004. EPA 542-C-04-004. www.frtr.gov
FSAI. 2002. Summary of investigation of dioxins, furans and PCBs in farmed salmon, wild salmon, farmed trout and fish oil capsules. Food Safety Authority of Ireland, March 2002. http://www.fsai.ie/industry/Dioxins3.htm
Grandjean P. 2004. Implications of the precautionary principle for public health practice and research. Int J Occup Med Environ Health 17;1:5
Green E et al. 2000. Protecting environmental quality and human health: strategies for harmonisation. Sci Total Environ. 256;2-3:205
Grochowalski A. 1998. PCDDs and PCDFs concentration in combustion gases and bottom ash from incineration of hospital wastes in Poland. Chemosphere 37;9-12:2279
Hammitt JK. 2004. Economic implications of hormesis. Hum Exp Toxicol. 23;6 and BELLE Newsletter 12;1
Hammitt JK, Liu J-T. 2004. Is there a “cancer premium” ? Risk in Perspective 12;2. Harvard center for Risk Anal. www.hcra.harvard.edu
Hansson S et al. 1990. Selective predation by herring and mysids, and zooplankton community structure in a Baltic Sea coastal area. J Plankton Res. 12;5:1099
Harremoës P et al. (eds.) 2002. Late lessons from early warnings: the precautionary principle 1896–2000. Copenhagen, EEA. Environmental Issue Report No 22
Heinzerling L, Ackerman F. 2002. Pricing the priceless: Cost-benefit analysis of environmental protection. Georgetown Univ Law Center, Washington, DC
HELCOM and IBSFC. 2002. (Ecosystem approach based fisheries management)
Hildén M. 1997b. Risk, uncertainty, indeterminacy and ignorance in fisheries management – an analysis of management advice. Finn Environ Inst, Helsinki 1997. Monographs Boreal Environ Res. 5
Hildén M. 2005. Distilling research from practice and practice from research. BIREME – Baltic Sea Research Programme Newsletter 1/2005:14
Hildén M et al. 2002. Evaluation of environmental policy instruments – a case study of the Finnish pulp & paper and chemical industries. Finn Environ Inst, Helsinki. Monographs Boreal Environ Res. 21
Hoekstra EJ et al. 1999. Natural formation of chlorinated phenols, dibenzo-p-dioxins and dibenzofurans in soil of a Douglas fir forest. Environ Sci Technol. 33:2543
Houck O. 2003. Tales from a troubled marriage: Science and law in environmental policy. Science 302:1926
ICES. 2003b. Report of the ICES Advisory Committee on the Marine Environment, 2003. ICES Cooperat Res Report 263:1
ICES. 2004a. Report of the Baltic Fisheries Assessment Working Group. ICES, Copenhagen, 13-22 Apr 2004. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2004/ACFM:22
ICES. 2005a. Report of the Study Group on Management Strategies. ICES Headquarters, 31 Jan–4 Feb 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:09
ICES. 2005b. Report of the Baltic Salmon and Trout Working Group (WGBAST). Helsinki, 5-14 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:18
ICES. 2005c. Report of the Baltic Fisheries Assessment Working Group (WGBFAS). Hamburg, 12-21 Apr 2005. Int Council Explorat Seas Advisory Committee Fishery Manage, Copenhagen. ICES CM2005/ACFM:19
IOM. 2003. Dioxins and dioxin-like compounds in the food supply: Strategies to decrease exposure. Committee on Implications of Dioxin in the Food Supply, Food Nutr Bd, Inst Med of the Natl Academies. Natl Acad Press, Washington, DC
Jacobs MN et al. 2002b. Investigation of selected persistent organic pollutants in farmed Atlantic salmon (Salmo salar), salmon aquaculture feed, and fish oil components of the feed. Environ Sci Technol. 36;13:2797
Jentoft S. 1989. Fisheries co-management. Delegating government responsibility to fishermen’s organizations. Mar Pol. 13;2:137

Joas R et al. 2001. Effects on the fisheries industry of the Commission proposals (SANCO) on dioxin content of fish, fish oil and fish meal as part of animal feed regulation. Manuscript, Eur Parliament D-G Res Working Paper STOA 101 EN, 10-2001. Sci Technol Options Assess Series. Eur Parliament, Luxemburg
Joas R et al. 2002. Material flow based strategy for dioxin-like PCB levels in feed and food across the European Union and dietary intake estimation. Organohalogen Compds. 55:291
Joas R et al. 2003. Dioxins and other pops in by-products, recyclates and wastes and their potential to enter the food chain. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 1
Jones KC et al. 2000. Difffuse and secondary sources of atmospheric PCDD/Fs: Are they significant? Organohalogen Compds. 46:39
Judd NL et al. 2003. Assessment of PCB congener analytical methods: do they meet risk assessment needs? Arch Environ Contam Toxicol. 44;1:132
Kishimoto A et al. 2001. Cost effectiveness of reducing dioxin emissions from municipal solid waste incinerators in Japan. Environ Sci Technol. 35;14:2861
Koopman-Esseboom C et al. 1996. Effects of polychlorinated biphenyl dioxin exposure and feeding type on infants mental and psychomotor development. Pediatrics 97:700
Kris-Etherton PM et al. 2003. Fish consumption, fish oil, omega-3 fatty acids, and cardiovascular disease. Arterioscler Thromb Vasc Biol. 23;2:e20-30. Erratum in: Arterioscler Thromb Vasc Biol. 23;2:e31. Comment in: Arterioscler Thromb Vasc Biol. 23;2:151
Kruse S, Meng W. 2005. Risk management of dioxins in feed by the German Federal Ministry of Consumer Protection. Organohalogen Compds. CD-ROM ID 362
Kuikka S et al. 2004. Kuluttajaan kohdistuvan dioksiiniriskin pienentäminen: markkina- ja ekosysteemitiedon yhdistäminen päätöksenteon näkökulmasta (DIOMAR). Unpublished project proposal to Finn Min Agr Forestry 25.3.2004. 15 p
Leino O et al. 2005. Risk-benefit analysis of fish in Finland: Dioxins and omega-3 fatty acids. Organohalogen Compds. 2476-8. CD-ROM ID 1097
Lind Y et al. 2002. Exponering för organiska miljökontaminanter via livsmedel – intagsberäkningar av ΣPCB, PCB 153, p,p'-DDE, PCDD/F, dioxinlika PCB, PBDE och HBCD baserade pa konsumtionsdata från Riksmaten 1997-98. Uppsala, Swed Natl Food Authority. Livsmedelsverket Rapport 26. 103 p
Malve O et al. 2003. Modeling the transport of PCDD/F compounds in a contaminated river and the possible influence of restoration dredging on calculated fluxes. Environ Sci Technol. 37;15:3413
Markowska A, Żylicz T. 1999. Costing an international public good: the case of the Baltic Sea. Ecol Econ. 30;2:301
Melanson SF et al. 2005. Measurement of organochlorines in commercial over-the-counter fish oil preparations: implications for dietary and therapeutic recommendations for omega-3 fatty acids and a review of the literature. Arch Pathol Lab Med. 129;1:74
Neumann P. 2004. Using cost-effectiveness analysis to improve health care: Opportunities and barriers. Risk in Perspective 12;5:1
Norstrom RJ. 2002. Chemical, biological, ecological and environmental properties fundamental to understanding bioaccumulation of POPs in food webs. Organohalogen Compds. 55:5
North DW. 1997. Risk characterization: A bridge to informed decision making. Fundam Appl Toxicol. 39;2:81
NRC. 2001. A risk-management strategy for PCB-contaminated sediments. Committee on Remediation of PCB-Contaminated Sediments, Natl Res Council of the Natl Academies, Natl Acad Press, Washington, DC
Ponce RA et al. 2000. Use of quality-adjusted life year weights with dose response models for public health decisions: a case study of the risks and benefits of fish consumption. Risk Anal. 20;4:529
Prinz B et al. 1993. Standards and guidelines for PCDD/PCDF – An integrated approach with special respect to the control of ambient airpollution. Chemosphere 27;1-3:491
Renn O et al. 1998. How to apply the concept of sustainability to a region. Technol Forecasting Soc Change 58:63
Rogers MD. 2003. Risk analysis under uncertainty, the precautionary principle, and the new EU chemicals strategy. Regul Toxicol Pharmacol. 37;3:370
Ross G. 2004. The public health implications of polychlorinated biphenyls (PCBs) in the environment. Ecotoxicol Environ Saf. 59;3:275
SACN and COT. 2004. Advice on fish consumtion: benefits & risks. Sci Advisory Committee Nutr and Committee Toxicol. The Stationery Office, London. www.food.gov.uk/multimedia/pdfs/fishreport2004full.pdf
SCAN. 2000. Opinion of the Scientific Committee on Animal Nutrition on the dioxin contamination of feedingstuffs and their contribution to the contamination of food of animal origin. EC, Brussels. Adopted 06 Nov 2000
Seierstad SL et al. 2005. Dietary intake of differently fed salmon; the influence on markers of human atherosclerosis. Eur J Clin Invest. 35;1:52-9
Sijtsma R, Doring A. 2002. Feed industry efforts to reduce the risk of dioxin contamination of the food chain. Organohalogen Compds. 57:251
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
Slovic P. 1987. Perception of risk. Science 236:280
Smith GC et al. 2002. Intake estimation of polychlorinated dibenzo-p-dioxins, dibenzofurans (PCDD/Fs) and polychlorinated biphenyls (PCBs) in salmon: the inclusion of uncertainty. Food Addit Contam. 19;8:770
Tuomisto JT et al. 2004b. Risk-benefit analysis of eating farmed salmon. Science 305:478
TWGIM. 2004b. Final report on actions and rcommendation for "Integrated monitoring of dioxins & PCBs in the Baltic Region" in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 23 Feb 2004
USEPA. 2000b. Guidance for assessing chemical contaminant data for use in fish advisories. Vol. 2, Risk assessment and consumption limits. USEPA, Washington, DC. www.epa.gov/ost/fishadvice/es.html
USEPA. 2002c. Guidance for assessing chemical contaminant data for use in fish advisories. Vol. 3, Risk management. USEPA, Washington, DC. www.epa.gov/ost/fishadvice/es.html
USEPA. 2004 (2002). Great Lakes Strategy 2002 - A Plan for the New Millennium. A Strategic Plan for the Great Lakes Ecosystem. www.epa.gov/glnpo/gls/gls02.html
USPCCRARM. 1997. Framework for environmental health risk management. Final report, vol 1. US Presidential/Congressional Commission Risk Assess Risk Manage, Washington, DC. Internet Edition www.riskworld.com
Van Tongelen B. 2002. Community strategy for dioxins, furans and PCBs. Organohalogen Compds. 57:265

Verstraete F. 2002. Development and implementation of an EC strategy on dioxins, furans and dioxin-like PCBs in food and feed. Environ Sci Pollut Res Int. 9;5:297
Verta M et al. 2003. Continued transport of PCDD/F contaminated sediments from River Kymijoki to the Gulf of Finland, the Baltic Sea. Organohalogen Compds. 61:405-8. CD-ROM, Vol. 2, Section 3
Verta M et al. 2004. Dioxin concentrations in sediments of the Baltic Sea - A preliminary survey of existing data. Organohalogen Compds. 66:1401
Verta M et al. Accepted. Dioxin concentrations in sediments of the Baltic Sea - a survey of existing data. Chemosphere
Verta M et al. Unpublished. Kymijoen sedimentteihin varastoituneet PCDD/F- ja elohopeayhdisteet sekä niiden kulkeutuminen. Draft report, SYKE 27.1.2005
Weed DL. 2004a. Methodologic implications of the precautionary principle: causal criteria. Int J Occup Med Environ Health 17;1:77
Wernstedt K et al. 1999. Grounding hazardous waste cleanups: a promising remedy? Land Use Pol. 16;1:45
Wheatley B, Paradis S. 1996. Balancing human exposure, risk and reality: Questions raised by the Canadian aboriginal methylmercury program. Neurotoxicol. 17;1:241
Wheatley B, Wheatley MA. 2000. Methylmercury and the health of indigenous peoples: a risk management challenge for physical and social sciences and for public health policy. Sci Total Environ. 259;1-3:23

References to Chapter 9

Anon. 2003. Dioxins & PCBs: Environmental levels and human exposure in candidate countries. Progress report III, Sept 2003. Ref: ENV.C.2/SER/2002/0085. Consortium: Environ Levels In Candidate Countries (ELICC). EC, Brussels. 17 p
Anon. 2004. Dioxins & PCBs: Environmental levels and human exposure in candidate countries. Final report. Reference: ENV.C.2/SER/2002/0085. Consortium: Environmental Levels In Candidate Countries (ELICC) under Supervision of Gunther Umlauf (JRC). EC, Brussels, 16.6.2004
Beck U. 1995. Ecological politics in an age of risk. Cambridge, Polity Press
Bellett AJ. 1990. Agent Orange controversy. Nature 343;6259:586
Bjørnstad ON, Grenfell BT. 2001. Noisy clockwork: Time series analysis of population fluctuations and animals. Science 293:638
Diamond J. 2005. Collapse: How societies choose to fail or survive. The Penguin Group, London et al
Dietz T et al. 2003. The struggle to govern the Commons. Science 302:1907
Douglas M. 1996. Risk and blame – Essays in cultural theory. Routledge, London. Reprint, first published 1992
Durodié B. 2003a. The true cost of precautionary chemicals regulation. Risk Anal. 23;2:389-98
Durodié B. 2003c. Limitations of public dialogue in science and the rise of new 'experts'. Crit Rev Int Soc Pol Phil. 6;4:82-92
Durodié B. 2005. Inclusion versus experimentation: A reply to Roland Jackson et al. Crit Rev Int Soc Pol Phil. forthcoming
EC. 2001. Community strategy for dioxins, furans and polychlorinated biphenyls. Communication from the Commission to the Council, the European Parliament and the Economic and Social Committee. COM(2001)593, Final. Off J EC 17.11.2001, C322/2-18. www.europa.eu.int
EC. 2002a. Commission recommendation of 4 March 2002 on the reduction of the presence of dioxins, furans and PCBs in feedingstuffs and foodstuffs. Notified under number C(2002a) 836. Off J EC 9.3.2002 L 67/69 (2002/201/EC)
Finkel AM. 1989. Is risk assessment really too conservative? Revising the revisionists. Columbia J Environ Law 14:427
Foran JA et al. 2005b. Quantitative analysis of the benefits and risks of consuming farmed and wild salmon. J Nutr. 135;11:2639
Glassner B. 1999. The culture of fear – Why Americans are afraid of the wrong things. Basic Books.
IOM. 2003. Dioxins and dioxin-like compounds in the food supply: Strategies to decrease exposure. Committee on Implications of Dioxin in the Food Supply, Food Nutr Bd, Inst Med of the Natl Academies. Natl Acad Press, Washington, DC
Jackson JB et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293;5530:629
Joas R et al. 2001. Effects on the fisheries industry of the Commission proposals (SANCO) on dioxin content of fish, fish oil and fish meal as part of animal feed regulation. Manuscript, Eur Parliament D-G Res Working Paper STOA 101 EN, 10-2001. Sci Technol Options Assess Series. Eur Parliament, Luxemburg
Joas R et al. 2002. Material flow based strategy for dioxin-like PCB levels in feed and food across the European Union and dietary intake estimation. Organohalogen Compds. 55:291
Joas R et al. 2003. Dioxins and other pops in by-products, recyclates and wastes and their potential to enter the food chain. Organohalogen Compds. 60-65, CD-ROM Vol. 5, Section 1
Kohn MC et al. 1994. The importance of biological realism in dioxin risk assessment models. Risk Anal. 14;6:993
Kuhn TS. 1977. A function for thought experiments. Kuhn TS. The essential tension – Selected studies in scientific tradition and change. The Univ Chicago Press, Chicago. P. 240-65. Reprinted from L'aventure de la science, Mélanges Alexandre Koyre, Hermann, Paris 1964.
Leino O et al. 2005. Risk-benefit analysis of fish in Finland: Dioxins and omega-3 fatty acids. Organohalogen Compds. CD-ROM ID 1097
Mackenzie BR et al. 2004. Fish, fishing, and pollutant reduction in the Baltic Sea. Environ Sci Technol. 38;7:1970
Mazur A. 2004. True warnings and false alarms. Evaluating fears about technology, 1948-1971. Resources For the Future, Washington, DC
Oreskes N et al. 1994. Verification, validation, and confirmation of numerical models in the earth sciences. Science 263:641
Pacala SW. 2003. False alarm over environmental false alarms. Science 301:1187
Putnam H. 2002. The collapse of the fact/value dichotomy and other essays. Harvard Univ Press, Cambridge and London
Siegel M. 2005. False alarm: The truth about the epidemic of fear. Wiley, John and Sons, Inc., New York.
SPCFC. 2005. Opinion of the Scientific Committee on Contaminants in the Food Chain on a request from the European Parliament related to the safety assessment of wild and farmed fish. Question N EFSA-Q-2004-23. Adopted on Jun 2005. EFSA J. 236:1
Tuomisto JT et al. 2004b. Risk-benefit analysis of eating farmed salmon. Science 305:478.
TWGIM. 2004a. Baseline report on "Integrated monitoring of dioxins & PCBs in the Baltic Region" in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 09 Jan 2004.
TWGIM. 2004b. Final report on actions and rcommendation for "Integrated monitoring of dioxins & PCBs in the Baltic Region" in the framework of the European Environment and Health Strategy (COM(2003)338 final). Tech Working Group Integrated Monitoring, subgroup Monitoring dioxins & PCBs Baltic Region. Version 23 Feb 2004
Van Tongelen B. 2002. Community strategy for dioxins, furans and PCBs. Organohalogen Compds. 57:265
Weed DL. 2002. Environmental epidemiology: basics and proof of cause-effect. Toxicol. 181-182:399

Indexes

(cf. List of abbreviations; excluding the most common terms)

Geographical names	
Africa	47, 119-21
Åland	39, 241
Archipelago Sea	73, 116
(the) Arctic	93, 97, 101, 121, 155, 160-1, 169, 178, 200
Arctic Ocean	169
Arkona Sea	27
(the) Atlantic	46, 77, 160-2, 169, 173, 199, 236, 256, 266
(the) Baltic	most pages
Baltic Marine Area	26-7, 39-41, 170, 172
Baltic Proper	27, 43, 64, 66, 68, 73, 158-9, 171
Baltic states	173, 241
Barcelona	17
Belarus	27
Belt Sea	27, 39, 170
Black Sea	27, 162, 201, 277
Bornholm	27, 39, 68, 268
Bornholm Basin	68
Boston	17
Bothnian Bay	27, 40, 64, 66, 74, 76, 120, 124, 155, 159-61
Bothnian Sea	27, 66, 70, 116, 156, 159-60
British Columbia	156
Canadian Arctic	178
Central Europe	60, 157, 230
Central Finland	84
Czech Republic	27, 60
Dagö	39
Denmark	23, 27-8, 41-3, 49, 57, 61-3, 77-8, 80, 173, 191
East coast (of Sweden)	79, 80, 84, 96, 99, 152, 177
Eastern Europe	60, 207, 221, 230
Eastern (general)	226, 244
Estonia	27-8, 42, 209
EU	17, 22-4, 28, 30-1, 34, 38, 41, 55, 59-60, 63,
Europe	17, 22, 59-60, 148, 152, …, 300, 302
European Economic Area	176
Finland	23, 27-8, 39-43, 49, 61, 65-8, 70, 73, 76-81, 84-6, 118-9, 130, 133, 150, 152, 167, 171, 173, 175-6, 207-11, 220-2, 237-8, 240-1, 249-50, 260, 264-8, 270-2, 283, 297, 299, 302
Fyn	39
Germany	23, 27-8, 42, 57, 60-1, 78, 80, 84, 167, 171, 186, 209-10, 221, 224
Gotland	39, 76
Great Lakes	63, 76, 89, 96-9, 117, 119-22, 124, 137, 139, 149, 152-3, 169, 172-3, 179, 190, 194, 214, 232, 237-9, 254, 270-2
Gulf of Bothnia	27, 40, 64, 66, 70, 72-3, 76, 158-9, 161
Gulf of Finland	27, 40, 65, 68, 70, 73, 76, 120, 139, 158-9
Gulf of Riga	27
Helsinki	20, 27
Ireland	180
Japan	69, 77, 86, 95, 153, 186
Kattegat	27, 43, 53, 74, 116, 124, 126, 159, 194
Lake Michigan	119
Lake Ontario	139, 192
Lapland	76, 124
Latvia	27-8, 42, 85, 118, 209, 212-3
Lithuania	27-8, 42, 86, 140, 209, 212-3
Lolland	39
Mediterranean	17, 162
North Atlantic	162
North-Eastern Atlantic	199, 208
North Sea	27, 39, 43, 49, 57, 63, 68, 80, 199, 208, 256, 264, 266-8, 299
Northern Baltic	128, 133, 222
Northern Europe	161
Northern (general)	198, 208-9, 260
Northern Sweden	118
Norway	178, 207-8, 223
Poland	27-8, 42, 60, 63, 73, 80, 84, 158, 171, 173,
River Kymijoki	38, 42, 59, 62, 64, 70, 158, 160, 283-4, 303
River Oder (Odra)	68. 157
River Wisla	62
Rügen	39
Russian Federation	17, 27, 42, 49, 60, 73, 197-8, 200, 207-9, 212-3, 215, 234, 241, 280
Seveso, Italy	21, 86, 96-7, 101, 108, 150, 152, 157, 191
Sjælland	39
Skagerrak	43, 63, 126, 160-1
Slovakia	27-8
Southern Baltic	66, 73, 76, 119, 121, 156, 159-60
Southern Norway	156
Southern Sweden	118
Soviet (Union)	207
Stockholm	118
Sweden	23, 27-8, 39, 41-2, 49, 57-9, 61, 77, 79, 80, 84, 118, 128, 130, 152, 167, 173, 191, 194, 209, 220-3, 228, 237-8, 240-1, 249, 260, 264-6, 268, 270-1, 297, 302