Committee for Risk Assessment (RAC)

Committee for Socio-economic Analysis (SEAC)

Annex to the Background document

to the Opinion on the Annex XV dossier proposing restrictions on
FOUR PHTHALATES (DEHP, BBP, DBP, DIBP)

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Annankatu 18, P.O. Box 400, FI-00121 Helsinki, Finland | Tel. +358 9 686180 | Fax +358 9 68618210 | echa.europa.eu

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Abbreviations

ADHD	Attention-Deficit Hyperactivity Disorder
AfA	Application for Authorisation
AGD	Anogenital distance
AOP	Adverse Outcome Pathway
ART	Assisted Reproductive Treatment or Assisted Reproductive Technology
ASE	Alkylsulphonic phenyl ester (Sulfonic acids, C10-21- alkane, Ph esters)
ATBC	Tributyl o-acetylcitrate
BBP	Benzyl butyl phthalate
CHAP	Chronic Hazard Advisory Panel
CMR	Carcinogenic, Mutagenic or toxic for Reproduction
CN	Combined Nomenclature
COMGHA	Glycerides, castor-oil mono-, hydrogenated, acetates
CoRAP	Community Rolling Action Plan
CPSC	Consumer Product Safety Commission
DBP	Dibutyl phthalate
DEGD	Diethylene glycol dibenzoate
DEHA/DOA	Bis(2-ethylhexyl) adipate
DEHP	Bis(2-ethylhexyl) phthalate
DEHS	Bis(2-ethylhexyl) sebacate
DEHT/DOTP	Bis(2-ethylhexyl) terephthalate
DGD	Oxydipropyl dibenzoate
DIBP	Diisobutyl phthalate
DIDP	Di-isodecyl phthalate
DINA	Diiso-nonyl adipate
DINCH	1,2-Cyclohexanedicarboxylic acid, 1,2-diisononyl ester

DINP	Di-“isononyl” phthalate
DNEL	Derived No-Effect Level
DPHP	Bis(2-propylheptyl) phthalate
ECHA	European Chemicals Agency
ECPI	European Council for Plasticisers and Intermediates
EDC	Endocrine Disrupting Chemical
EEE	electrical and Electronic Equipment
EFSA	European Food Safety Authority
EU	European Union
EuPC	European Plastics Convertors
FUE	Urinary Excretion Fraction
GC-MS	Gas Chromatography with Mass Spectrometry
GM	Geometric Mean
GTA	Triacetin
HI	Hazard Index
ICSI	Intracytoplasmic Sperm Injection
INBP	1,2-Benzene dicarboxylic acid, benzyl C7-9-branched and linear alkyl esters
IUPAC	International Union of Pure and Applied Chemistry
IVF	In vitro fertilisation
JRC ED EAG	Joint Research Centre’s Endocrine Disruptor Expert Advisory Group
LOAEL	Lowest Observed Adverse Effect Level
MSC	Member State Committee
N(L)OAEL	No-Observed (Lowest Observed) Adverse Effect Level
NOAEL	No-Observed Adverse Effect Level
OECD	Organisation for Economic Co-operation and Development
PNEC	Predicted No Effect Concentration

PNW	Postnatal week
PODI	Point of Departure Index
PPAR	Peroxisome Proliferator Activated Receptor
PU	Polyurethane
PVA	Polyvinyl acetate
PVC	Polyvinyl chloride
Qm	Quantification Maximum i.e. compositional Limits
RAAF	Read-Across Assessment Framework
RAC	Risk Assessment Committee
RCR	Risk Characterisation Ratio
RDRPPM	R&D, Reformulation, Process and Plant Modification
RMO	Risk Management Option
RoHS	Restriction of Hazardous Substances
RPF	Relative Potency Factors
SAR	Structure-Activity Relationship
SEAC	Socio-Economic Analysis Committee
SER	Specific Emission Rate
SME	Small and Medium sized Enterprise
SML	Specific Migration Limit
SVHC	Substance of Very High Concern
SVOC	Semi-Volatile Organic Compounds, i.e. organic compounds with vapour pressures between 10^{-14} and 10^{-4} atm (10^{-9} to 10 Pa) as defined in Weschler and Nazaroff (2008)
TBT	Technical Barriers to Trade
TDS	Testicular Dysgenesis Syndrome
TEF	Toxic Equivalent Factors
TIE	Toy Industries of Europe

TOTM/TEHTM	Tris(2-ethylhexyl) benzene- 1,2,4-tricarboxylate
WTO	World Trade Organisation
WTP	Willingness to Pay

Annex A: Manufacture and uses

A.1. Introduction

DEHP, DBP, DIBP, and BBP are used in a wide range of applications. The following article groups fall in the scope of the proposed restriction:

- Flooring (and heavy wall covering)
- Film & sheets (& plates, foil, strip and other flat shapes) of plastics
- Bags
- Coated clothing
- Coated paper/wallpaper/tapestry
- Mattresses
- Balls for training and physical exercises
- Bathing equipment (swim-coats/wings/belts and pools - inflatable and others)
- Footwear
- Insulation and mouldings of wires and cables
- Other moulded products (e.g., decorative items, office supplies, etc.)
- Miscellaneous: These are items not falling within the classification groups listed above such as: adult sex toys; handles of bicycles or garden tools; car interiors; other interior construction products; mixtures such as coating and finishes incorporated in the articles above; some hoses & tubes; toys & childcare articles; etc.

The article groups were selected to correspond to the defined scope of the proposed restriction specified in Annex D, i.e., all uses of the four phthalates in articles for indoor use and for outdoor use with potential for dermal contact (excluding the specified derogations). The following two main criteria were used for selecting which article groups fall within the scope of this restriction proposal:

- whether there is a possibility for exposure to these articles leading to human health risks and
- whether there is empirical evidence that the four phthalates are used in EU produced or imported articles placed on the EU28 market.

This Annex discusses the use of the four phthalates in these article groups in EU production and imports. The Annex begins with a general overview of the manufacture, import and export of the four phthalates and continues with a description of their function and specific use by article category.

A.2. Manufacture, import and export of the four phthalates

DEHP, DBP, DIBP, and BBP continue to be produced and used in the EU albeit at decreasing rate. Information from EuroStat presented in Table A1 illustrates that between 2004 and 2013 their consumption has declined by more than 10.5% annually, on average. The regulatory changes which have been the driving force of this decline intensified it in recent years, leading to a faster rate of decline: 13% annually of consumption and more than 24.5% decrease in production between 2010 and 2013.

Table A1 Historical production, export, import and consumption of DEHP, DBP and DIBP in EU28

in tonnes	Production*	Exports**	Imports**	Consumption***
2004	348 671	60 604	4 405	292 472
2005	408 893	60 089	3 932	352 735
2006	341 703	55 419	4 402	290 687
2007	375 530	53 630	4 478	326 378
2008	251 895	37 898	10 565	224 562
2009	211 546	39 248	2 510	174 809
2010	211 232	72 289	4 397	143 341
2011	146 333	31 124	4 753	119 962
2012	120 958	15 639	5 621	110 941
2013	89 615	3 443	8 029	94 201
Average percent change per year				
2011-2013	-24.7%	-61.6%	23.1%	-13.0%
2004-2013	-12.5%	-15.3%	23.0%	-10.6%

Notes:
* ProdCom code 20143410 - Dibutyl and dioctyl orthophthalates, which includes DEHP, DBP and DIBP primarily
** CN code 29173200 - Dioctyl orthophthalates, which includes largely DEHP. As exports and imports of DBP, DIBP and BBP are included together with "other esters of orthophthalic acid", their tonnages are not reflected in consumption, imports and export data.
*** Consumption is defined as Production and Imports, excluding Exports.
Source: EuroStat

The data presented in the table does not reflect the impact of the inclusion of the four phthalates on the Authorisation list (Annex XIV of REACH) with a sunset date of 21 February 2015. Three manufacturers of the substance: Deza a.s., Arkema France, and Grupa Azoty Zakłady Azotowe Kędzierzyn S.A. (Grupa Azoty) submitted applications[1] for the use of DEHP in formulation of DEHP in compounds, dry-blends and plastisol formulations and in polymer processing by calendaring, spread coating, extrusion, injection moulding to produce polyvinyl chloride (PVC) articles. The tonnage applied for is confidential and since the application, the Arkema France plant ceased production of DEHP (2014) and withdrew their application.[2] As there are no applications by downstream users (e.g., converters or article manufacturers), it is uncertain how many tonnes of DEHP continue to be used in the EU28, under the decision-

[1] A decision by the European Commission on the authorisation applications is pending as of the submission date of this dossier: 1 April 2016.

[2] The application was withdrawn on request of the applicant on 2 December 2015, ECHA, http://www.echa.europa.eu/addressing-chemicals-of-concern/authorisation/applications-for-authorisation-previous-consultations

pending authorisations. [redacted]

To date, there are no applications for authorisation for the use of the DBP, DIBP, and BBP in articles in the scope of this restriction proposal. Deza a.s.'s applications for the use of less than 1 200 tonnes of DBP are for industrial uses of the substance.[3] Therefore, it can be assumed that the use of these three phthalates in the manufacturing of articles in the scope of this proposal was fully phased out in EU28 as of 2015.

Table A2 presents a summary of REACH registration data on the four substances. The specific tonnages are confidential, and it is likely that a maximum of three active registrations (two for DEHP and one for DBP) would remain following the decisions on authorisation.

Table A2 Summary of registration information - tonnes/year

	Manufacturers		Importers or Only Representatives	
Substance	**tonnage**	**# registrants**	**tonnage**	**# registrants**
DEHP	> 5 000	6	> 5 000	12
DBP	> 1 000	1	< 2 000	9
DIBP	< 2 000	2	< 2 000	6
BBP	> 1 000	1	-	-

Source: ECHA registration dossiers

A.3. Uses

DEHP, DBP, DIBP and BBP are commonly used plasticisers. They belong to the group of orthophthalates. Phthalates in general – covering orthophthalates and terephthalates - are the most commonly used plasticisers in the world. According to market intelligence they accounted for over 78% of the world consumption of plasticisers in 2012 (IHS 2013). DEHP alone accounted for more than 50% of the phthalates used worldwide (ECPI 2012) and 60% of all plasticisers used in China (BASF 2011). China is the largest plasticiser market in the world, accounting for nearly 38% of world consumption in 2012; it also has the highest forecast growth rate between 2011 and 2018, spurred by increased plasticiser consumption in goods for both domestic and export markets. Other Asian countries taken together, including Japan, constitute the second-largest plasticiser consuming region, with nearly 21% in 2012, followed by Western Europe (16%) and North America (about 13%) (IHS 2013). Information received during a Technical Barriers to Trade consultation on the proposal (see Annex G) did not contradict this.

In Europe, approximately 95% of produced orthophthalates are used in flexible PVC. Examples of materials that may contain phthalates are:

- Polyvinyl chloride (PVC) and related polymers, such as polyvinylidene chloride (PVDC) and polyvinyl acetate (PVA);

[3] A decision by the European Commission on the authorisation applications is pending as of the submission date of this dossier: 1 April 2016.

- Soft or flexible plastics, except polyolefins;
- Soft or flexible rubber, except silicone rubber and natural latex;
- Foam rubber or foam plastic, such as polyurethane (PU);
- Surface coatings, non-slip coatings, finishes, decals, and printed designs;
- Adhesives and sealants. (AFIRM 2014)

They can be found in a variety of articles including electrical cables, hoses, flooring, wall coverings, coated textiles, luggage, sports equipment, toys, roofing membranes, pool liners, footwear as well as medical devices such as tubing and blood bags (ECPI 2015).

The typical concentration of the four phthalates in articles is about 30% of the soft PVC content; however, it varies substantially depending on the article type and, to a less extent, on the manufacturing process.[4] Table A3 below represents an estimate of the phthalate content in articles within the scope of this restriction proposal. It is based on:

- information of domestic use of the four plasticisers in consumer articles, specified in applications for authorisation
- import/export estimates by article group of
    - the average proportion of soft PVC in the article
    - the average plasticiser content of the PVC in the article
    - the (estimated) share of DEHP, DBP, DIBP and BBP of total plasticiser use by geographic region. (For further details regarding the methodology, please see Annex C: Baseline.)

It was estimated that in 2014, approximately 171 135 tonnes of DEHP, BBP, DBP and DIBP contained in articles was placed on the EU28 market – a decrease of 4.5% since 2011 in total (or about 1.4% annually). The tonnages of the four phthalates in EU article production has decreased significantly which however, has been compensated by increase in imports. Despite the decline in EU production, exports continued to grow: by nearly 9% between 2011 and 2014. The tonnes of the four phthalates contained in imported articles represent an increasing share of the total tonnes in articles placed on the EU28 market – from 56.5% in 2011 to close to 73% in 2014. This is due, first, to the declining use of the four phthalates in the EU article manufacturing and in particular, DBP, DIBP and BBP. Second, to the increasing import of articles to the EU, particularly from Asia. Between 2004 and 2014, import volumes of articles in scope increased by more than 44%, or by more than 4% annually, on average. During this period, China's share of article import volumes grew from under 59% to over 67% of total imports to the EU28 of articles within the scope of the proposed restriction.

[4] Refined estimates of PVC and plasticiser content gathered for the purpose of the Danish PVC tax. See Annex C, Table C2 for further details.

Table A3 Estimated total tonnes of DEHP, DBP, DIBP and BBP contained in articles in the scope of this proposal placed on the EU28 market

in tonnes	2011	2012	2013	2014
Total tonnes in articles placed on the market	179 222	169 350	180 525	171 135
Tonnes used in EU28 article manufacturing	92 403	84 259	73 458	62 612
Tonnes contained in Exported articles	14 438	14 924	15 755	15 722
Tonnes contained in Imported articles	101 256	100 015	122 822	124 245
Share of tonnes imported of total placed on EU28 market	56.5%	59.1%	68.0%	72.6%
% change – annual		-5.5%	6.6%	-5.2%

Notes: Estimates derived on the basis of EuroStat import, export, and manufacturing statistics; AFA 2013; Danish phthalate tax database, and market intelligence. See Annex C: Baseline for details.

A.3.1. Use of DEHP in articles

DEHP is a general purpose plasticiser. It has been used for more than 50 years in almost all soft/flexible PVC applications due to its recognised plasticising efficiency, fusion rate and viscosity: it is slower fusing, exhibiting higher viscosity and lower volatility compared to the other three phthalates (DBP, DIBP and BBP), yet quicker fusing, with lower viscosity and higher volatility than other general purpose phthalates which are used as substitutes for DEHP. It is recognised as the international standard for PVC plasticisers due to being in the mid-range of plasticiser properties, at an attractive price; therefore, properties of other phthalates are often conveyed in relation to those of DEHP. Market information on DEHP by application area is provided in section A.4.

Due to regulatory pressures, the use of DEHP in the EU, North America and Northeast Asia has been declining. In the EU, the regulatory action with the strongest effect related to DEHP's classifications as toxic to reproduction in 2001 and its ban in toys and childcare articles (REACH Regulation, Annex XVII, entry 51) while in North America it related to its inclusion in California's list of chemicals known to cause cancer or reproductive toxicity.[5] DEHP has been substituted largely with DINP; however, other plasticisers have also gained share, due to their specific advantages in certain applications (e.g., extreme weather properties, fire retardancy, heating resistance, low volatility, etc.) or due to preference for non-phthalate plasticisers.

Outside of these geographic regions, DEHP continues to be dominant, particularly in China, which accounts for 50% of world DEHP consumption. Use of DEHP in China is forecast to grow by more than 5% annually and its manufacture by 29% between 2011 and 2019. (TOC 2012)

The typical concentration of the DEHP in articles is between 15-30% of the soft PVC content; however, it varies substantially depending on the article type and sampling of various consumer articles has shown that DEHP is present in concentrations up to 461 000 mg/kg. See Table B31 in Annex B for further information.

As shown in Table A4, DEHP use in articles placed on the EU28 market, declined by 2.5% from 2011 to 2014, largely due to its substitution in EU28 articles manufacturing, which outpaced the growth in DEHP tonnages contained in imported articles. The use of DEHP is likely to

[5] Proposition 65, the Safe Drinking Water and Toxic Enforcement Act of 1986, California Office of Environmental Health Hazard Assessment, http://oehha.ca.gov/prop65.html

continue to decline past 2014 due to the entry into force of the authorisation requirements under REACH as of February 21, 2015 – the sunset date of the substance, as well as other regulatory pressures (e.g., amendments of the RoHS directive). Between 2011 and 2014, the tonnages of DEHP in imported articles placed on the EU market grew by close to 22%, as the authorisation requirements do not apply to imported articles and the volumes of articles imported have been increasing.

Table A4 Estimated total tonnes of DEHP contained in articles in the scope of this proposal placed on the EU28 market

in tonnes	**2011**	**2012**	**2013**	**2014**
Tonnes used in EU28 article manufacturing	[redacted]	[redacted]	[redacted]	[redacted]
Tonnes contained in Exported articles	12 743	13 186	13 931	13 909
Tonnes contained in Imported articles	91 957	90 494	111 592	112 088
Total tonnes in articles placed on the market	[redacted]			
Share of tonnes imported of total placed on EU28 market				
% change - annual		-4.7%	8.5%	-5.8%
% change - 2011-2014 average/total	-0.6% / -2.5%			

Notes: Estimates derived on the basis of EuroStat import, export, and manufacturing statistics; AFA 2013; Danish phthalate tax database, and market intelligence. See Annex C: Baseline for details.

Currently, there are several uses of DEHP in the EU with pending authorisation decisions. These relevant for this proposal include formulation of DEHP in compounds, dry-blends and plastisol formulations and industrial use in polymer processing by calendaring, spread coating, extrusion, injection moulding to produce PVC articles. The applicants specified that they did not apply for the following uses (in addition to the uses explicitly restricted under other EU legislation):

- Erasers;
- Adult toys (sex toys and other articles for adults with intensive contact with mucous membranes);
- Small (<10 cm) PVC items available in the home environment (without attachment to larger objects), which can be swallowed by small children;
- Textiles/clothing intended to be worn against the bare skin;
- Formulation and processing of rubber articles;
- Formulation of end product mixtures such as sealants, adhesives, and paints.[6]

Another application for authorisation with a pending decision is for formulation of recycled soft PVC in compounds and dry-blends and industrial use of recycled soft PVC in polymer processing by calendaring, extrusion, compression and injection moulding to produce PVC articles.[7] All other uses of DEHP in the EU are not permitted under REACH as of the sunset date of the substance: 21February 2015.

[6] For professional uses, as for consumer uses the substances are already restricted under REACH Regulation, Annex XVII, entry #30.

[7] The following authorised uses of DEHP in the EU fall outside the scope of this restriction proposal: use of DEHP in stop-off formulation during the diffusion bonding and manufacture of aero engine fan blades; ceramic sheets and printing pastes for production of lambda sensor elements; manufacture of solid propellants and motor charges for rockets and tactical missiles.

Although not supported in the EU any longer (and would not be in the future unless new applications are submitted and authorisations are granted), the end use of DEHP in the articles mentioned above are included in the scope of this restriction proposal. These articles are produced using DEHP internationally and subsequently imported to the EU. As the authorisation requirements do not apply to imported articles and this proposal demonstrates that the risks to human health from those articles are not adequately controlled, a restriction is necessary to appropriately manage them. Between 2011 and 2014, the share of total DEHP tonnages in articles placed on the EU market originating from imports has grown by more than 25% to [redacted] of the total tonnes in articles placed on the EU market. The share of imported articles is anticipated to increase in the future.

A.3.2. Use of DBP in articles

DBP (and DIBP) exhibit low viscosity and good solvating properties but their high volatility has limited their use to that of a speciality plasticiser often used in combination with other plasticisers, including DEHP. DBP is used as a plasticiser for PVC, poly vinyl alcohol (PVA) and rubber as well as a solvent and a fixative in paint. DBP is essentially used for its viscosity reducing properties and compatibility with non-PVC mixtures (lacquers, printing inks, sealants, adhesives) or as processing aid for PVC (plastisols, compounds) in concentrations of 5 to 10 % w/w due to their higher polarity (ECHA 2013). Its soft PVC uses include flooring, packaging material, shoes, home furnishing, and clothing. (UML 2011, ECPI 2015) Recent sampling of articles showed that DBP can be present in concentrations up to 345 000 mg/kg. (See Table B31 in Annex B)

Use of DBP has been declining. Substitutes are available for all its uses within the scope of this restriction proposal and some of the often cited alternatives are benzoates and terephthalates (ECHA 2013).

Table A5 Estimated total tonnes of DBP, DIBP and BBP contained in articles in the scope of this proposal placed on the EU28 market

in tonnes	**2011**	**2012**	**2013**	**2014**
Tonnes used in EU28 article manufacturing	[redacted]	[redacted]	[redacted]	[redacted]
Tonnes contained in Exported articles	1 695	1 739	1 825	1 813
Tonnes contained in Imported articles	9 299	9 522	11 230	12 158
Total tonnes in articles placed on the market	[redacted]			
Share of tonnes imported of total placed on EU28 market				
% change - annual		-8.7%	-1.1%	-2.7%
% change - 2011-2014 average/total	-4.2% / -12.2%			

Notes: Estimates derived on the basis of EuroStat import, export, and manufacturing statistics; Applications for authorisation; Danish phthalate tax database, and market intelligence. See Annex C: Baseline for details.

Since 21 February 2015, DBP cannot be used or placed on the market for use in the EU in articles within the scope of this proposal. Selected applications outside the scope of this restriction proposal have been authorised.[8]

No applications for DBP have been received from recyclers which suggest that DBP content in the recycling waste stream does not exceed 0.3% w/w of the plasticised material.[9]

DBP entering the EU via imported articles is not within the scope of the authorisation process. Thus, from 2015, the imported articles represent the only source of exposure to DBP via articles. By mid-2015, ECHA has received notifications on a wide variety of imported articles containing DBP under art. 7 of REACH. Their reported concentration ranges are presented in brackets (see Table A9):

- Shoes (soles) (1.1% – 8.4%)
- Personal protective equipment (rain gear), rain jackets (3%)
- Inflatable mattresses, boats (0.5% – 20%)
- Pencil cases (0.2%)
- Textile print (no information available)
- Textile accessories (0.2% – 1.6%)
- Packaging material (no information available)
- Pool covers, grain covers, truck covers, general covers, tents, membrane structure covers, warehouse covers, roofing, etc. (0.5% – 3.5%)

Uses in electric cables and cords has also been notified although it has been reported that DBP (and DIBP) can be unsuitable in these applications as they have greater potential for volatisation (as cables and cords undergo heating (typically up to 60°C) and can be in constant use). (RIVM 2013)

A.3.3. Use of DIBP in articles

DIBP has very similar application properties to DBP and may therefore be used to substitute DBP in most, if not all, of its applications. These range from the plasticisation of PVC to the production of paints, printing inks and adhesives. (ECPI 2015) Alternatives to DIBP are available for all its applications (ECHA 2013).

DIBP is not restricted in toys and childcare articles under entry 51 in Annex XVII; however, its concentration is limited to 5%[10] under the Toy Safety Directive, i.e., Directive 2009/48/EC of the European Parliament and of the Council of 18 June 2009 on the safety of toys. For the preparation of this proposal, a literature review was conducted on the use of DIBP in toys (ECHA (2016)). The conclusions of the review are that the available data indicates that 1-3% of toys with flexible PVC contain DIBP. The concentration is highly variable and, due to the differences in the scope of the studies and surveys, it is not possible to aggregate the data. It seems likely that the average concentration of DIBP in the DIBP-containing articles is in the

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8 Authorised uses of DBP include: manufacture of solid propellants and motor charges for rockets and tactical missiles; specialty paint in manufacture of motors for rockets and tactical missiles; ceramic sheets and printingpastes for production of capacitors and lambda sensor elements; propellants – formulation and industrial use; in the manufacture of maleic anhydride as absorption solvent.

9 CLP Regulation general classification limit: Repr. 1A and 1B – 0.3 %.

10 The 9th Amendment to CLP is likely to be adopted by the Commission in the second quarter of 2016 and will become effective 18 months after entry into force; this is estimated to be at the end of 2017. The consentration limit for DIBP will become 0.3%.

range 10-20%, i.e., about half of the average concentration of plasticisers in PVC of approximately 30%. On this basis, it is roughly estimated that DIBP represents 0.5-1.5% of the plasticisers used in toys or 50-300 tonnes. This estimate however is associated with considerable uncertainties.

Some of the studies reviewed by ECHA (2016) explicitly stated that most samples with DIBP also contained the phthalates DEHP, DINP, DBP or DIDP and others, revealed non-compliance with the Toy Safety Directive concentration limit. These conclusions are also supported by entries in the RAPEX database, presented in Table A6 and a recent study by PROSAFE (2016). The latter found DIBP in about 4% of samples, more than half of which showed non-compliance with the Toy Safety Directive.

ECHA (2016) did not find data on the trend in the use of DIBP in toys, as the available number of surveys was too small to indicate any trend. Therefore, it cannot be confirmed that DIBP has replaced the use of DBP in toys and childcare articles but this cannot be excluded, given their structural and pricing similarities. Such substitution is not desirable, as DIBP has very similar hazard profile to DBP (see Annex B).

Table A6 Notifications to the RAPEX database 2011-2016 with the search string "DIBP"

Member state notifying	Year	Product	Phthalate concentration on DIBP-containing part, % by weight	Country of origin
Germany	2015	Air-filled ball	27.2% DIBP	China
Sweden	2014	A wallet made of black and brown artificial leather	19% DEHP and 1.1% DIBP	China
Germany	2014	Remote-controlled toy car	Tyres contain 3.3% DEHP, 4% DINP, 38% DBP and 1.3% DIBP	China
Sweden	2013	Two soft plastic bears	One bear contained 29% DIBP and 5.1% DEHP	China
Sweden	2013	Toy set of a doll and a pony made in soft plastic	Different concentrations in different parts with the highest being: 6.3% DIBP, 11% DBP, 0.23% DINP and 35% DEHP	China
Sweden	2013	A baby doll made of soft plastic	The head contained 40% DEHP, 0.36% DIBP and up to 0.16% DBP. The shower hose contained up to 3.1% DEHP and 45% DBP	China
Sweden	2013	Six plastic fish	29% DIBP and 2.5% DEHP	China
Italy	2013	Packs of strands for braiding, known as "scoubidous"	Packs contained 16% and 15% of DIBP, respectively.	China
Sweden	2013	Six gel pens in various colours with plastic grip in soft plastic	Grip contained 24 % DEHP and 3.4 % DIBP	Unknown
Germany	2012	Plastic toy set consisting of a pair of high-heeled shoes, two hair bobbles and a handbag.	Plastic straps of the toy shoes contained 3.5% DEHP and 14.5% DIBP	France

Source: ECHA (2016)

Recent sampling of articles showed that DIBP can be present in concentrations up to 355 000 mg/kg (Table B31 in Annex B). Some of the articles included school bags and children's risk watches, where DIBP was found in concentration of respectively 830-3 100 mg/kg and 70-50 000 mg/kg. Art. 7 notifications for DIBP are similar to those of DBP. Additional notifications include PVC flooring and ammunition. (See Table A9.) During the public consultation on the restriction dossier, comments were received from the Norwegian and Swedish competent authority which suggest that DIBP's use in toys and childcare articles is being phased out.

No authorisation applications have been received to date and since 21 February 2015, DIBP cannot be used or placed on the market for use in the EU. Therefore, starting from 2015, imported articles – not within the scope of the authorisation process – represent the only source of exposure to DIBP via articles.

A.3.4. Use of BBP in articles

BBP is used mainly as a specialty plasticiser for PVC or other polymers. It is fast fusing plasticiser, exhibiting lower volatility than DBP or DIBP but it is more volatile than DEHP and exhibits poor low temperature properties. Its high solvency results in poor plastisol shelf life, requiring the need to blend it with DEHP or DINP. BBP is used primarily as a fast fusing plasticiser for foamed plastisols and in polysulfides (ECHA 2013).

BBP is used in some soft PVC products such as flooring, packaging, and artificial leather as well as car care products and together with other polymers in sealants, adhesives, paints, coatings

and inks. (ECPI 2015) Analysis of articles within the scope of the restriction showed BBP in concentrations between 2 and 73,000 mg/kg (Table B31 in Annex B).

BBP use in the EU and internationally has been declining. Alternatives exist for all uses, with benzoates and teraphthalates being the main substitutes (ECHA 2013). No authorisation applications have been received to date and since 21 February 2015, BBP cannot be used or placed on the market for use in the EU. Therefore, starting from 2015, imported articles – not subject to authorisation - represent the only source of exposure to BBP via articles.

Art. 7 notifications for BBP include the following (see Table A9):

- Medical supplies (disposable medical devices) – tubes, bags, connectors (0.5%)
- Outdoor sitting furniture with textile cover
- Electrical cables and cords
- Packaging material

Similar to DBP and DIBP, no applications for authorisation have been received from recyclers. This suggests that BBP content in the plasticised materials in the recycling waste stream of non-integrated recyclers does not exceed 0.3% w/w [11]

[11] CLP Regulation general classification limit: Repr. 1A and 1B – 0.3 %

A.4. Uses of DEHP, DBP, DIBP, and BBP in articles in the scope of the restriction proposal

A.4.1. Flooring and wall covering

Definition of the product group

Flooring and heavy style wall covering made of PVC are vinyl materials with and without textile or polyurethane (PUR or PU) backing material. The thickness of the material will generally be in the range of 1-3 mm. The products may be used for covering of ceilings as well.

Possible routes of exposure

To the extent the materials contain phthalates these substances may be released and bound to dust in indoor environment. Phthalates present on the surface of wall covering and flooring may be a source for exposure of small children touching the vinyl with fingers, etc. Phthalates present on the surface of wall covering and flooring may be removed by washing and thereby be disposed of with residues from the washing process, e.g., washing water directed to sewer systems.

Import to and production of articles in the EU

Import of flooring to EU28 is estimated to around 340 000 tonnes, with China accounting for approximately two-thirds of that (2014). EU28 consumption of flooring products was estimated at approximately €1.6 billion in 2014.

Plasticisers in use

This product group is diverse and phthalate concentrations vary extensively depending on flooring type (ECHA 2012a).

Recent sampling of flooring products revealed the following concentrations of the four phthalates: DEHP 49-325 mg/kg, DBP 129 mg/kg, DIBP 56-73 650 mg/kg, and BBP 113 mg/kg (see Table B31 in Annex B). In addition, ECHA notifications in articles showed the presence of the four phthalates in imported articles in the following concentrations: DEHP 2.7% – 23%% and DIBP - 7% (see Table A9). Flooring has been reported to be the main application of BBP (ECHA 2012a).

The tonnages of DEHP in flooring placed on the EU market is estimated at [redacted] tonnes in 2014. The absence of applications for authorisation, including by recyclers, suggests that BBP, DBP, and DIBP are fully phased out in PVC article manufacturing in the EU. Internationally these three phthalates continue to be used. BBP imports in flooring are estimated at 1 640 tonnes and the combined use of DBP/DIBP in imports – about 3 200 tonnes in 2014. The estimates assume that the PVC constitutes 50-100% of the article weight, while the plasticiser is approximately 20% of the PVC weight.

Alternatives available

DINP, DIDP, DPHP, DEHT, DINCH, DEHA and dibenzoates (such as secondary plasticisers DGD and DEGD) are used in PVC flooring products. (ECHA 2012a, AfA 2013a)

Trends and perspectives

The substitution of DEHP in flooring products is expected to continue, likely at a faster pace in domestic article manufacturing due to REACH authorisation pressure. However, their use internationally is not expected to change substantially without further regulatory pressures, as major import partners, e.g., China, have large DEHP manufacturing capacity in particular. By 2020, the tonnages of the four phthalates in flooring on the EU28 market is forecast to decline to [redacted] tonnes of DEHP and to reach about 5 150 tonnes for BBP and DBP/DIBP.[12]

A.4.2. Film & sheet

Definition of the product group

This group includes plates, sheets, film, foil and strips of plastics, non-cellular and not reinforced, laminated, supported or similarly combined with other materials. These products are generally used for packaging goods or in applications such as construction materials, furniture, office supplies, etc. Film and sheet used in bags and brief/suitcases as well as for tablecloth, curtains, shower curtains and similar items (not industrial uses) are reported in the categories of Coated products. However, it is possible that there is a percentage of semi-finished articles that are reported in this category (as an intermediate product) and also reported in other categories of Coated products.

Possible routes of exposure

This diverse group contains articles that could lead to dermal exposure due to contact with skin, as well as inhalation or oral exposure to phthalates due to phthalate release to indoor air and deposition in dust.

Import to and production of articles in the EU

Import of film & sheet to EU28 is estimated to around 290 000 tonnes, with China accounting for approximately 35% of that (2014). The film & sheet placed on the EU market are estimated at approximately €4.3 billion in 2013. (EuroStat)

Plasticisers in use

ECHA notifications of articles showed the presence of the phthalates in imported articles in the following concentrations: DEHP 5.3% – 14.7% (see Table A9).

The total tonnages of DEHP in this article category is estimated at [redacted] tonnes in 2014. BBP imports are estimated at 910 tonnes and the combined use of DBP/DIBP in imports – 1 810 tonnes in 2014. The estimates assume that PVC constitutes 19-100% of the article weight, while the plasticiser constitutes 19-30% of the PVC weight.

[12] See Annex C: Baseline for details.

Alternatives available

Alternatives for all film and sheet applications are available. DINP is the most common substitute for DEHP in this article group.

Trends and perspectives

The substitution of DEHP domestically is expected to continue; however, its use internationally is not expected to change substantially without further regulatory pressure. By 2020, the consumption of the four phthalates in articles on the EU28 market is forecast to decline to [redacted] tonnes of DEHP and to reach about 2 770 tonnes for BBP and DBP/DIBP.[13]

A.4.3. Coated products

Definition of the product group

This section combines information on bags and brief-/suitcases, clothing and textile materials, paper/wallpaper/tapestry, and similar materials containing PVC parts or coating.

Possible routes of exposure

To the extent these materials contain phthalates, these substances could be ingested with dust (as phthalates can be released by evaporation to indoor air and bound to dust) or could be a source of dermal exposure due to contact with skin.

Trends and perspectives for Coated products

Substitution of the four phthalates in coated products has been observed over the past decade. Recent substitution has been driven by regulatory pressures related to the REACH authorisation procedure. In 2014, DEHP tonnages in coated products placed on the EU28 market was about [redacted] tonnes. Pressures related to authorisation requirements are anticipated to continue to drive substitution of DEHP domestically and in 2020, DEHP tonnages in this article group is forecast to decline to [redacted] tonnes, with imports being the largest contributor.

In the absence of further regulatory pressures, the other three phthalates are anticipated to reach about 1 350 tonnes in 2020.[14]

A.4.3.1. Bags, brief-/suitcases and similar items

Definition of the product group

The product group covers plasticised (mainly PVC) parts on bags, brief- and suitcases and similar items. These include coated fabrics such as thin PVC film typically used outside or inside the bags and cases as well as moulded products such as leather-looks, and PVC marks, figures, profiles sewn, welded or otherwise attached to the outer surface or bottom of the bags and cases.

[13] See Annex C: Baseline for details.

[14] See Annex C: Baseline for details.

Import to and production of articles in EU

Import to EU28 of bags within the scope of this restriction proposal is estimated to around 370 000 tonnes, with China accounting for more than 85% of total imports (2014). EU28 consumption is estimated at approximately €7.7 billion in 2013. (EuroStat)

Plasticisers in use

Historical sampling of bags on the EU market has revealed the presence of DEHP in concentrations from 12 to 21%, of DBP and DIBP below 0.1%, and of BBP below 1% w/w (ECHA 2012a).

Recent sampling of bags revealed the following concentrations of the four phthalates: DEHP 12-202 000 mg/kg, DBP 14-60 mg/kg , DIBP 10-509 mg/kg (see Table B31 in Annex B). In addition, ECHA notifications of articles showed the presence of the phthalates in imported articles in the following concentrations: DEHP 0.46% – 20%, DBP 0.2% – 1.6% w/w (see Table A9).

Previous studies have assumed that 20% of all bags contain PVC, and that 70-90% of these bags contain 1-5% PVC, while the remainder is made entirely of PVC (ECHA 2012a). Similarly to other article groups, these assumptions were refined on the basis of information gathered for the purpose of the Danish PVC tax. Thus, for the purpose of this restriction proposal, the majority of bags are assumed to contain about 5% PVC material of the article weight, with 25% of that being the plasticiser, primarily DEHP.

Available alternatives

Producers and suppliers to the Danish market have informed that alternatives are available for all parts of bags. Alternatives are varied, such as DINP, DIDP, DPHP, DEHT, DINCH, DEHA, ASE, etc. (ECHA 2012a, AfA 2013a)

A.4.3.2. Clothing and other textile products

Definition of the product group

The product group covers tablecloth, curtains, shower curtains and similar items made of PVC film or coated fabrics for home and office purposes but not for industrial purposes.[15]

Import to and production of articles in the EU

Import to EU28 of clothing within the scope of this restriction proposal is estimated to around 330 000 tonnes, with China accounting for close to 80% of total imports (2014). EU28 consumption is estimated at approximately €4.4 billion in 2013. (EuroStat)

Plasticisers in use

Recent analysis of PVC-containing textile products revealed the following concentrations of the four phthalates: DEHP 15-296 000 mg/kg, DBP 13-63 mg/kg, DIBP 64-173 mg/kg (see Table

[15] At the time of the writing of this dossier, a European Commission consultation is running on a possible restriction of hazardous substances (CMR 1A and 1B) in textile articles and clothing for consumer use under Article 68(2) of Regulation EC No 1907/2006 (REACH).

B31 in Annex B). In addition, ECHA notifications of articles showed the presence of DEHP in imported articles in the following concentrations: 2%-37% w/w (see Table A9).

No applications for authorisation have been submitted for the use of the four phthalates in clothing materials intended to be worn against the bare skin. Their use in EU article production is therefore not allowed since 21 February 2015.

For the purpose of the current study, the majority of PVC-containing clothing and other textile materials are assumed to contain about 10-20% PVC material of the article weight, with 15% of that being the plasticiser.[16] All four phthalates are assumed to be found in imported articles of this group, while domestically, only DEHP is assumed to be used at a declining rate.

Alternatives available

DINP is used as substitutes for DEHP in table cloths, dinner mats and shower curtains. Other plasticisers than phthalates in use for tablecloth/cover are ATBC, DINCH and DOA in combination with ESBO. Phthalates-free table cloth/covers of PVC film and PVC-coated textile are available on the European market. Plasticisers used include, among others, TBC (tributyl citrate but probably ATBC, often used for PVC for food contact), DINCH, DOA and ESBO. Various alternatives to PVC shower curtains are available at low cost. Many synthetic, woven textiles, for example of polyester, but also plastic film of EVA/PEVA, are marketed. European retailers are also marketing PVC-free plastic coated table cloths (oil cloth style), for example coated with acrylics (ECHA 2012a).

A.4.3.3. Paper, wallpaper, tapestry

Definition of the product group

The product group covers wall paper or wall covering made of or coated with plasticised PVC. PVC wallpaper is also called vinyl wall covering, but should be differentiated from the thicker wall covering products used for bathroom walls etc. (vinyl flooring style coverings), see section A.4.1.

Import to and production of articles in the EU

EU production of wallpaper is estimated to be around 1.9 million tonnes (2013), while import, more than half of which originates in North America, is estimated to be around 200 000 tonnes (2014). EU28 consumption is estimated at approximately €2.3 billion in 2013, 15% of that represented imports. (EuroStat)

Plasticisers in use

Recent analysis of paper/wallpaper/tapestry products revealed the following concentrations of the four phthalates: DEHP 10-24 mg/kg, DBP 9-30 mg/kg, DIBP 5-626 mg/kg (see Table B31 in Annex B).

For the purpose of the current study, the majority of paper/wallpaper/tapestry products are assumed to contain 2-25% PVC material of the article weight, with 25-30% of that being the

[16] See Annex C: Baseline for details.

plasticiser.[17] All four phthalates are assumed to be found in imported articles of this group, while domestically, only DEHP is assumed to be used at a declining rate.

Alternatives available

DINP, DINCH, DOA and DEHT are all used by the wall coverings industry, and that there are no wallpaper qualities which cannot be produced without DEHP, BBP, DBP and DIBP (ECHA 2012a).

A.4.4. Wires and cables

Definition of the article group

The article group consists of isolated electrical wires and cables, as well as optical fibre cables, of types used indoor in homes and offices. PVC in reality serves as insulation as well as coating material. Cable and wire types used in the homes include flexible cables used for connecting electrical devices, construction cables for low voltages, 230 and 400V, low voltage cables used inside electrical and electronic devices, and optical cables. These articles are covered under the RoHS directive but are included in the scope of the proposed restriction to ensure consistence in EU legislation and reduce ambiguity for stakeholders.

Possible routes of exposure

Consumers can be exposed to phthalates via direct dermal contact with the wires and cables or via ingestion of dust containing phthalates released in indoor environment.

Import to and production of articles in the EU

Import of cables within the scope of the proposal to EU28 is estimated to be around 465 000 tonnes, with China accounting for approximately 40% of imports (2014), while production in EU28 is estimated to around 2.4 million tonnes (2013). Wires and cables used within EU28 (imports and production, excluding exports) was estimated at approximately €10 billion in 2013. (EuroStat)

Plasticisers in use

Previous studies have shown that:

- the average PVC content of regular PVC insulated wires and cables is around 30% for single solid copper conductor wire (used for 230-400V installations), whereas it is around 65% for 2-3 conductors flexible connecting cords used in the home or office, and around 70% for 3- and 5-conductors construction cables (230-400V installations);
- DIDP, DINP and DEHP are likely the main plasticisers used for cables in the EU, with DIDP being the main substitute to DEHP;
- BBP, DBP and DIBP are reported to have limited applicability in cable and wire, probably due to their high volatility (cables are heated during use and this increases the volatilisation); however, some use of DBP as a secondary plasticiser was reported in the past. (ECHA 2012a)

[17] See Annex C: Baseline for details.

ECHA notifications of articles showed the presence of the phthalates in imported articles in the following concentrations: DEHP 0.13%-20% and DBP 0%-30% (see Table A9). The estimates of DEHP content in imported cables & wires used indoors assumes that PVC insulation in wires and cables constitutes 20-35% of the article weight, while the plasticiser is approximately 25% of the PVC weight.

Alternatives available

Substitutes for all wires and cables exist, with the most common being DIDP. Other plasticisers used are DINP, DPHP, TOTM and adipates. PVC-free cables are available with insulation made of PE or silicon rubber. (ECHA 2012a)

Trends and perspectives

The use of the four phthalates in wires and cables for the EU28 market is anticipated to be fully phased out by 2019, when the concentration limit of 0.1% w/w on their use under RoHS comes into effect. [18]

A.4.5. Moulded and other products containing soft PVC

Definition of the product group

This article group contains diverse products made of or containing soft PVC material, such as mattresses, balls for training and physical exercises, bathing equipment, footwear, etc.

Many of these products could be manufactured from various materials in addition to plastics. Thus, it was challenging to identify relevant statistical codes that could adequately encompass the products falling within the scope of this restriction and therefore, be used in the estimation of the phthalate content of domestic and imported articles. The statistics below provide only an indication of the phthalate content.

Possible routes of exposure

To the extent these materials contain phthalates, these substances may be released to indoor air or could be a source of dermal exposure due to contact with skin. Some of the smaller items may also lead to exposure of children to phthalates from mouthing.

Trends and perspectives for Moulded products

Substitution of the four phthalates in these articles has been observed over the past decade. Recent substitution in the EU28 has been driven by regulatory pressures related to the REACH authorisation procedure. Articles such as erasing rubber, adult sex toys and similar that may lead to prolonged contact with mucus membranes were not included in the scope of the authorisation applications and the use of all four phthalates was phased out in EU manufacturing as of the sunset date: February 2015.

In 2014, DEHP in moulded products consumed within the EU28 was about [redacted] tonnes. Pressures related to authorisation requirements are anticipated to continue to drive substitution

[18] See Annex C: Baseline for details.

of DEHP domestically and in 2020, DEHP tonnages in this article group is forecast to increase to [redacted], with imports accounting for the largest share.

In the absence of further regulatory pressures, the tonnages of the other three phthalates in imported articles are anticipated to reach about 3 500 tonnes in 2020.[19]

A.4.5.1. Mattresses

Definition of the product group

The product group covers water beds and air mattresses produced of PVC film or coated fabrics.

Import to and production of articles in the EU

EU manufacture of mattresses within the scope of this restriction proposal is estimated at about 1 500 tonnes (2013). EU28 imports are estimated around 7 200 tonnes (2014), with almost all imports originating from China. The value of EU28 mattress production is estimated at approximately €10 billion (2013) and imports, at about €29.3 billion (2014). (EuroStat)

Plasticisers in use

Recent analysis of mattresses revealed the following concentrations of the four phthalates: DEHP 31-304 000 mg/kg and DIBP 11 mg/kg (see Table B31 in Annex B). In addition ECHA notifications of articles showed the presence of the phthalates in imported articles in the following concentrations: DEHP: 8.8% – 20% and DBP: 20% (see Table A9).

For the purpose of the current study, the majority of mattresses are assumed to contain about 20% PVC material of the article weight, with 25% of that being the plasticiser.[20] All four phthalates are assumed to be found in imported articles of this group, while domestically, only DEHP is assumed to be used at a declining rate past 2015.

Alternatives available

ASE and DINP are reported as substitutes to the four phthalates in this article group. Traditional rubber/cotton solution may be regarded as an alternative material to PVC. (ECHA 2012a)

A.4.5.2. Footwear

Definition of the product group

The product group covers sandals and slippers/flip flops made partly or completely of PVC. The group include statistics primarily for footwear which could lead to dermal exposure.

Import to and production of articles in the EU

[19] See Annex C: Baseline for details.
[20] See Annex C: Baseline for details.

Import of the relevant footwear to EU28 is estimated to be around 630 000 tonnes (2014), with China accounting for 85% of that. The value of these imports is about €4.4 billion. EU production of these articles is minimal – about €70 million (2013). (EuroStat)

Plasticisers in use

Recent analysis of footwear revealed the following concentrations of the four phthalates: DEHP 11-461 000 mg/kg, DBP 1-345 000 mg/kg, DIBP 3-212 000 mg/kg, and BBP ND-79 mg/kg (see Table B31 in Annex B). In addition ECHA notifications of articles showed the presence of the phthalates in imported articles in the following concentrations: DEHP 14.6% – 30%, DBP 1.1% – 8.4% and DIBP 11%-15.9% (see Table A9).

For the purpose of the current study, PVC-containing footwear concerned is assumed to contain 50% PVC material of the article weight, with 20% of that being the plasticiser. DEHP and BBP are assumed to be found in imported articles of this group, while domestically, only DEHP is assumed to be used at a declining rate past 2015.

Alternatives available

Alternatives available include other materials (e.g., polyurethane (PU)) as well as other plasticisers, e.g., DINP, DINCH, ATBC and other (ECHA 2012a).

A.4.5.3. Balls and bathing equipment

Definition of the product group

The product group covers a variety of bathing equipment made of plasticised PVC film or coated fabrics inclusive of pools (inflatable and non-inflatable), swim-coats/wings/belts. It also includes balls made entirely of PVC, PVC-film and coated fabrics for playing and physical exercises. This group excludes articles that can be considered toys or childcare articles, as these are already covered by existing restriction entry 51 in Annex XVII of the REACH Regulation.

Import to and production of articles in the EU

Import of the relevant articles in this category to EU28 is estimated to be around 240 000 tonnes (2014), with China accounting for more than 80% of that. The value of EU28 consumption of these articles is about €1.8 billion with imports accounting for about half of that (2013). (EuroStat)

Plasticisers in use

Recent analysis of balls and bathing equipment revealed the following concentrations of the four phthalates: DEHP 9-439 000 mg/kg, DBP 10-21 mg/kg and DIBP 18-355 000 mg/kg (see Table B31 in Annex B). In addition, ECHA notifications of articles showed the presence of the phthalates in imported articles in the following concentrations: DEHP 2% - 25% and DBP 0.5% – 3.5% w/w (see Table A9).

For the purpose of the current study, articles in this category are assumed to contain 5-30% PVC material of the article weight, with 15-35% of that being the plasticiser.[21] DEHP and minor quantities of BBP are assumed to be found in imported articles of this group, while domestically, only DEHP is assumed to be used at a declining rate past 2015.

Alternatives available

The alternatives, other materials as well as other plasticisers, are available for all types of balls, e.g.,:

- soccer balls, e.g., PU
- fitness balls – DINP, ATBC, DIDP, DIOP
- large plastic balls – phthalate-free plasticisers. (ECHA 2012a)

The situation is similar for bathing equipment. DINP is already used as plasticiser in most bathing equipment applications. The use of non-orthophthalate alternatives is also reported: DINCH and DEHT, among others (ECHA 2012a).

A.4.5.4. Other products

Definition of the product group

This group includes other articles made of or coated with plastics such as office supplies (including erasing rubber) and decorative items. The use of erasing rubber has been explicitly excluded from the scope of the authorisation application for DEHP; therefore, the use of the four phthalates in this application has been banned in the EU28 since February 21, 2015. This does not apply to imported erasers.

Import to and production of articles in EU

Import of articles in this category to EU28 is estimated to be around 780 000 tonnes (2014), with China accounting for more than 60% of that. The value of EU28 consumption of these articles is about €18 billion with exports representing less than 25% of that (2013). (EuroStat)

Plasticisers in use

Recent analysis revealed the following concentrations of DEHP in articles in this category between 170 000-440 000 mg/kg (see Table B31 in Annex B).

For the purpose of the current study, articles in this category are assumed to contain 3-50% PVC material of the article weight, with 30% of that being the plasticiser.[22] All four plasticisers are assumed to be found in imported articles of this group, while domestically, only DEHP is assumed to be used at a declining rate past 2015.

[21] See Annex C: Baseline for details.
[22] See Annex C: Baseline for details.

Alternatives available

A number of phthalate and non-phthalate plasticisers are already in use in these applications, with DINP being one of the most common substitutes.

A.4.6. Miscellaneous

Definition of the product group

These are items not falling within the classification groups listed above such as: adult sex toys; handles of bicycles or garden tools; articles used in car interiors such as floor mats, steering wheel covers, car seat cushions; etc. The phthalate content of these articles has not been estimated as this has been significantly impeded by the absence of individual statistical codes and the even greater diversity of articles and the wide variety of materials used in these article categories. For example, the plastic handles of garden tools and bicycles are reported as part of the statistical codes for these article groups, while many of the car interior products, such as steering wheel covers, gear shift knobs or upholstery of synthetic leather, are included in the statistics for the automotive sector.

Furthermore, some garden hoses could lead to prolonged dermal exposure but the majority of the articles within the category of hoses, profiles & tubes do not lead to the exposure within the scope of this restriction proposal. Similarly, large share of construction materials would not lead to relevant exposure as they would be used outdoors or within the building frames. Therefore, the tonnages of the four phthalates contained in hoses, profiles & tubes in the scope of this restriction proposal were not estimated.

In this article category we also include toys and childcare articles that may contain DIBP. DIBP has been found to be used in these articles (See section A.3.3). As there were no applications for authorisation, DIBP is possibly used only in imported articles. The tonnages of DIBP entering the EU via these imported articles is likely small and therefore, was not estimated.

In addition, the four phthalates are also used in surface coatings, non-slip coatings, finishes, decals, printed designs, adhesives and sealants which are incorporated in articles defined within the scope of the proposed restriction. When applied to articles, which may or may not have been produced using the four phthalates, the mixtures for example such as coatings and finishes, form a surface layer that could expose humans, in particularly via dermal and mucous membrane contact. The risk of exposure from these articles would depend on the strength of the chemical bond that the phthalates have formed with the rubber for example or the PVC (if a coating is applied to a PVC article). Therefore, articles containing the four phthalates because of the application of these mixtures are also in the scope of this proposal. The tonnages of the four phthalates placed on the EU market as a result of the application of these mixtures on articles in the scope of this proposal are reported by article category above.

Plasticisers in use

DEHP has traditionally been used in many of these diverse applications. The other three plasticisers have had the role of secondary plasticisers, with exception of some niche applications, e.g., coatings, finishes, etc.

The use of adult sex toys has been explicitly excluded from the scope of the authorisation application for DEHP; therefore, the use of the four phthalates in this application has been banned in the EU28 since 21 February 2015. This doesn't apply for imported articles.

Recent analysis revealed the following concentrations of the four phthalates: DEHP: between 180 - 702 000 mg/kg, DBP: 50-45 000 mg/kg and DIBP: 20-66 000 mg/kg (see Table B31 in Annex B). In addition, ECHA notifications of articles showed the presence of DEHP in imported articles in the excess of 0.5% w/w (see Table A9).

Alternatives available

A number of phthalate and non-phthalate plasticisers are already in use in these applications, with DINP being one of the most common substitutes but also benzoates and terephthalates replacing DBP, DIBP, and BBP.

A.5. Summary

Table A7Table A7 and present summary estimates and projections for the amount of the four phthalates in articles in consumption, production and import to the EU28. While the amount of the four phthalates in articles produced in the EU is expected to continue to decline, the total amount of the four phthalates in articles placed on the EU28 market is expected to decline at much slower rates in the short term. In the long run, this is expected to be reversed due to the tonnages of the four phthalates in imported articles and even to surpass the amount phased out in domestic production if modest historical trends continue in the future. Annex C: Baseline discusses the projections of the amount of the four phthalates in articles placed on the EU market in greater detail.

Table A7Table A7 and A8 below present a summary of the tonnages of the four phthalates contained in articles in the scope of the proposed restriction placed on the EU market.

Table A7 Summary table - DEHP in articles in the scope of the restriction (tonnes)

Articles types	Tonnes contained in articles - 2014			Tonnes contained in articles - 2020		
	Total on EU market	Used in EU production	In Imports	Total on EU market	Used in EU production	In Imports
Film & sheet	[redacted]		17 530	[redacted]		17 946
Flooring & wall covering			31 306			33 146
Coated products			10 522			11 153
Bags, brief/suitcases & similar items			3 835			4 068
Clothing & other textiles			6 278			6 658
Paper, wallpaper, tapestry			409			427
Wires & cables	[redacted]		16 939			
Moulded & other products			35 790	[redacted]		37 950
Mattresses			207			220
Balls & bathing equipment			11 572			12 279
Footwear			9 942			10 553
Other			14 070			14 899
Miscellaneous (not quantified)						
Total	[redacted]		112 088	[redacted]		100 196

Notes: Results derived on the basis of EuroStat import, export, and manufacturing statistics; applications for authorisations; Danish phthalate tax database, and market intelligence. See Annex C: Baseline for details.

Table A8 Summary table – DBP, DIBP and BBP in imported articles in the scope of the restriction (tonnes)

Article types	**2014**	**2020**
Wires & cables	-	-
Film & sheet	2 717	2 771
Flooring & wall covering	4 856	5 140
Coated products	1 275	1 350
Bags, brief-/suitcases & similar items	208	220
Clothing & other coated textile products	995	1 055
Paper, wallpaper, tapestry	72	75
Moulded & other products	3 310	3 507
Mattresses	32	34
Balls & bathing equipment	595	631
Footwear	477	507
Other	2 206	2 335
Miscellaneous (not quantified)		
Total	12 158	12 769

Notes: Results derived on the basis of EuroStat import, export, and manufacturing statistics; applications for authorisations; Danish phthalate tax database, and market intelligence. See Annex C: Baseline for details.

A.6. Uses advised against by the registrants

The main use advised against is the use of DEHP, DBP, and BBP in toys and childcare articles in concentrations greater than 0.1% by weight of the plasticised material. Furthermore, effective from 20 July 2013, The Toy Safety directive bans the use of substances that are classified as carcinogenic, mutagenic or toxic for reproduction (CMR) of category 1A, 1B or 2 under Regulation (EC) No 1272/2008 in toys, in components of toys or microstructurally distinct parts of toys unless certain specified conditions are met. The limit in the Toy Safety Directive is 0.3% (higher than the proposed limit for other articles in this proposal). However, a specific limit of 5% of for DIBP applies.

In addition, the supply to the general public of the four phthalates as substances, constituents of other substances or in mixtures is restricted under restriction entry 30 of Annex XVII due to their classification as toxic to reproduction category 1B.

Other specific legislation limits the use of the four phthalates in specific uses, such as FCM, wires and cables, and medical devices. See Table D1 in Annex D: Impacts for further information.

In addition, the four phthalates are subject to authorisation requirements. The substances cannot be used, including in the production of articles within the scope of this proposal, without specific authorisation granted by the European Commission. Pending and granted authorisations for the four phthalates are discussed in sections A.3.1, A.3.2, A.3.3 and A.3.4 of this Annex.

Table A9: Articles containing one or more of the four phthalates (data from notification of substances in articles, i.e., Art.7 of REACH)

Articles type	**Bis (2-ethylhexyl) phthalate (DEHP)**		**Dibutyl phthalate (DBP)**		**Diisobutyl phthalate (DIBP)**		**Benzyl butyl phthalate (BBP)**		**Articles User Group**
	Tonnage	Concentration in whole article (%)	Tonnage of substance	Concentration in whole article (%)	Tonnage	Concentration in whole article (%)	Tonnage	Concentration in whole article (%)	
Footwear	>3.1	14.6 - 30 %	1.1	1.1 - 8.4 %	4.8	11 - 15.9 %			Consumers
Fabric bags and accessories (e.g. bags, wallets, umbrellas, carry sleeves, suitcases)	>11.6	0.13 - 20 %	>2	0.2 -1.6%					Workers/ consumers
Clothing/ textile (e.g. rainwear, Clothes with print)	>18.7	0,1 % - 30%	1	3%					Workers/ consumers
Cables and wires (e.g. electric and electronic)	>11.04	0.13 - 20 %	1	0-30%					Consumers
Flooring (e.g. PVC tiles, coverings and sheets)	1428.6	14- 23 %				7%			Workers/ consumers
Tablecloth	9	2 - 37 %							Consumers
Inflatable mattresses and articles (inflatable boats, ventilation articles)	>2	8.8 – 25 %	21	0.5 -20 %					Workers/ consumers
Electronics (e.g. sewing machine, air conditioning units, lamps, hairdryers)	>52.1	0.1-20%	23	1%					Workers/ consumers
[redacted]									
Mats (e.g. shower and bath mats)	>2	10.2 – 30 %							Workers/ consumers
Plastic garden furniture	>13.38	*1.24 - 1.4%*							Consumers
Covers (e.g. pool covers, truck covers, tarpaulin, roofings)	*>205.70*	2-25%	>20	0.5- 3.5%					Workers/ consumers

Articles type	**Bis (2-ethylhexyl) phthalate (DEHP)**		**Dibutyl phthalate (DBP)**		**Diisobutyl phthalate (DIBP)**		**Benzyl butyl phthalate (BBP)**		**Articles User Group**
	Tonnage	Concentration in whole article (%)	Tonnage of substance	Concentration in whole article (%)	Tonnage	Concentration in whole article (%)	Tonnage	Concentration in whole article (%)	
Bathroom accessories (e.g. shower caps, shower curtains, hanging organisers, pillow for bathtub, shower tray and rack, faucet)	>93	0.1 - 35%			1	0.1- 35 %			Workers/ consumers
Vehicle and mechanical parts (e.g. motor cycles, seals, valves, bellows)	>101	1-35%			1	2%			Workers/ consumers
Insect frames and screens	1.12	0.31 -11.8 %							Workers/ consumers
PVC sheets and articles (e.g. car windows, boot windows, tent windows)	51	2-20%							Workers/ consumers
Packaging	< 1	9%							Workers/ consumers
Pavilion tent	< 3.38	1.24 %							Workers/ consumers
Tools and tool box	11.41	0.5%							Consumers
Sign materials	< 200	2 - 25%							Workers/ consumers
Conveyor belt (rubber)	1	6%							Workers/ consumers
Personal Protective Equipment - goggles	12.39	14-38 %							Workers/ consumers

Annex B: Information on Hazard and Risk

B.1. Identity of the substance(s) and physical and chemical properties

This proposal concerns four phthalates with similar modes of action. Even though the phthalates have different potency their similar modes of action makes it reasonable to perform dose addition calculations to predict the combination effects of these chemicals.

B.1.1. Name and other identifiers of the substances

Chemical Name: 1,2-Benzenedicarboxylic acid, bis(2-ethylhexyl) ester (**DEHP**)
IUPAC Name: Bis(2-ethylhexyl) phthalate
EC Number: 204-211-0
CAS Number: 117-81-7

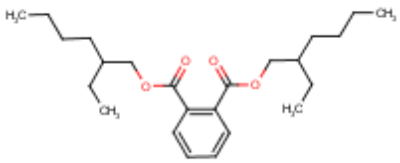
Chemical Name: Benzyl butyl phthalate (**BBP**)
IUPAC Name: Benzyl butyl phthalate
EC Number: 201-622-7
CAS Number: 85-68-7

Chemical Name: Dibutyl phthalate **(DBP)**
IUPAC Name: Dibutyl phthalate
EC Number: 201-557-4
CAS Number: 84-74-2

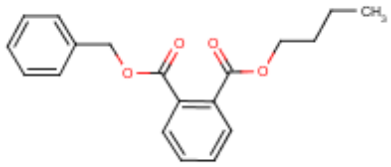
Chemical Name: Diisobutyl phthalate (**DIBP**)
IUPAC Name: Diisobutyl phthalate
EC Number: 201-553-2
CAS Number: 84-69-5

B.1.2. Composition of the substance(s)

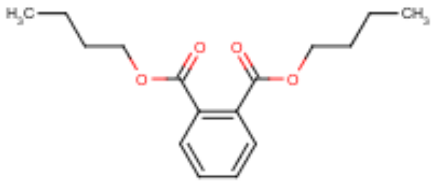
Chemical Name: 1,2-Benzenedicarboxylic acid, bis(2-ethylhexyl) ester (**DEHP**)
Molecular weight: 390.6 g/mol
Molecular formula: C24H38O4
Structural formula:



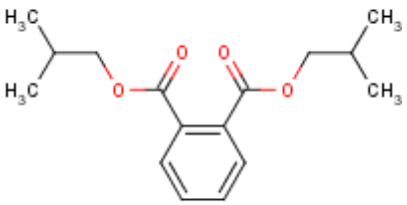
Chemical Name: Benzyl butyl phthalate (**BBP**)
Molecular weight: 312.35 g/mol
Molecular formula: C19H20O4
Structural formula:



Chemical Name: Dibutyl phthalate (**DBP**)
Molecular weight: 278.34 g/mol
Molecular formula: C16H22O4
Structural formula:



Chemical Name: Diisobutyl phthalate (**DIBP**)
Molecular weight: 278.34 g/mol
Molecular formula: C16H22O4
Structural formula:



B.1.3. Physicochemical properties

Table B1. Physicochemical properties of the four phthalates

Property	Substance	Value	Reference
Physical State	DEHP	Colourless oily liquid	EU RAR (2008a)
	BBP	Liquid	EU RAR (2008b)
	DBP	Oily liquid	EU RAR (2004)
	DIBP	Colourless liquid	Annex XV dossier (2009)
Melting point	DEHP	-55°C or -50°C	EU RAR (2008a)
	BBP	<-35°C	EU RAR (2008b)
	DBP	-69°C	EU RAR (2004)
	DIBP	-37°C at 1,013	Annex XV dossier (2009)
Boiling point	DEHP	385° C at 1,013 hPa	EU RAR (2008a)
	BBP	370° C at 10.10 hPa	EU RAR (2008b)
	DBP	340° C at 1,013 hPa	EU RAR (2004)
	DIBP	320° C	Annex XV dossier (2009)
Relative density	DEHP	0.98 g/cm^3 at 20°C	EU RAR (2008a)
	BBP	1.116 g/cm^3 at 20°C	EU RAR (2008b)
	DBP	1.045 g/cm^3 at 20°C	EU RAR (2004)
	DIBP		
Vapour pressure	DEHP	0.000034 Pa at 20° C	EU RAR (2008a)
	BBP	0.00112 Pa at 20° C	EU RAR (2008b)
	DBP	9.7±3.3 x 10^{-3} Pa at 25°C	EU RAR (2004)
	DIBP	0.01 Pa at 20°C	Annex XV dossier (2009)
Water solubility	DEHP	3 µg/l at 20°C	EU RAR (2008a)
	BBP	2.8 mg/L at 25 to 30°C	EU RAR (2008b)
	DBP	10 mg/L at 20°C	EU RAR (2004)
	DIBP	20 mg/L at 20°C	Annex XV dossier (2009)
Partition coefficient n-octanol/water (log value)	DEHP	7.5	EU RAR (2008a)
	BBP	4.84	EU RAR (2008b)
	DBP	4.57	EU RAR (2004)
	DIBP	4.11	Annex XV dossier (2009)

B.1.4. Justification for grouping

Phthalates are a group of substances of which some have been associated with effects on the development of the reproductive system of male laboratory animals and endocrine disrupting effects. The four phthalates DEHP, DBP, DIBP and BBP are classified as reproductive toxicants. There is a general consensus among toxicologists that evaluating combined exposure to all anti-androgenic phthalates is needed (Koch et al. 2011). Several arguments are presented below to justify the conclusion that the four phthalates follow a similar mode of action and can be grouped.

A grouping of the four phthalates is justified by their structural and metabolic similarities. They are all ortho-phthalates with alkyl side chains, linear or branched, of length C4-C6, which show similar severe reproductive effects (including effects on reproductive organs, fertility, and development) in experimental animals, especially antiandrogenic effects (Fabjan et al. 2006).

The four phthalates have all been shown to be anti-androgenic as a result of inhibition of foetal testosterone production. The main lines of evidence support this mode of action are: decreased foetal testosterone production (Howdeshell et al. 2008; Hannas et al. 2011, 2012); reduced male anogenital distance (Saillenfait et al. 2008; Lee et al. 2006; Martino-Andrade et al. 2009; Mylchreest et al. 1999; Tyl et al. 2004; Gray et al. 2009); and decreased gene expression related to steroid biosynthesis (Hannas et al. 2012; Lehmann et al. 2004). In addition, increased nipple retention in male offspring, which is an early marker of antiandrongenic effects, is seen consistently in connection with the other effects (Christiansen et al. 2010, Lee et al. 2004, Mylchreest et al. 1999, Tyl et al. 2004).

In addition to the anti-androgenic effects, DBP, DIBP and DEHP induce changes in germ cell differentiation (multinucleated germ cells), which are considered to be independent of foetal testosterone reduction (Borch et al. 2006, Gaido et al. 2007; Lambrot et al. 2009). Multinucleated germs cells are also seen with DINP (Boberg et al. 2011; Clewell et al. 2011)

All four phthalates show effects on reproductive organs and fertility in experimental animals exposed prenatally. The spectrum of adverse effects observed in rats include increased nipple retention, increased male mammary gland changes (vacuolar degeneration and alveolar atrophy), decreased anogenital distance, increased incidence of genital malformations (hypospadias and cryptorchidism), delayed puberty onset (delayed prepubertal separation), reduced semen quality (reduced number of spermatocytes) and testicular changes including decreased testes and epididymides weight, tubular atrophy and Leydig cell hyperplasia (EU RAR 2004, 2008a, 2008b; ECHA 2009d). Although increased nipple retention and decreased anogenital distance may not be adverse to the affected animal per se, these effects are early markers of e.g. hypospadias and undescended testes (cryptorchidism), all of which are consistently observed for all four phthalates (OECD 2008; RAC 2012). Similarly, alveolar atrophy in the male mammary gland may result from a decreased level of serum testosterone (OECD 2009).

Lastly, the four phthalates have a similar use and exposure pattern which is an additional reason for grouping.

B.1.5. Use of dose addition in combined risk assessment of phthalates

The concept

Dose addition is often referred to as concentration addition, simple similar action or Loewe additivity. The concept of dose addition has been introduced by Loewe & Muischnek in 1926, and the model assumes that all the chemicals in the mixture act on the same biological site (receptor or target organ), by the same mechanism of action, and that they differ only in their individual potency (Backhaus et al., 2004). The additive effects are described mathematically by summing up the doses of the individual chemicals in a mixture adjusted for their differences in potencies. Combination effects based on dose addition can result from chemicals at or below their respective no observed adverse effect levels (NOAELs), provided that sufficiently large numbers of chemicals sum up to a suitably high total effect dose (Kortenkamp, 2007).

The equation for dose addition (4 components) is:

$$\mathrm{EDx}_{\mathrm{mixture}} = \left(\frac{p_1}{\mathrm{EDx}_1} + \frac{p_2}{\mathrm{EDx}_2} + \frac{p_3}{\mathrm{EDx}_3} + \frac{p_4}{\mathrm{EDx}_4} \right)^{-1}$$

Here, EDx_1, EDx_2, EDx_3 and EDx_4 are the effect doses of four chemicals that on their own produce the same quantitative effect x as the mixture, and p_1, p_2, p_3 and p_4 are the relative proportions of the corresponding individual doses present in the total mixture dose ("fraction in mixture").

Practical use

Dose addition can be considered as the default approach for combined risk assessment of similar acting chemicals in general and many phthalates in particular.

In June 2009 a report from an Expert workshop on combination effects of chemicals was published. The workshop was organised under the auspices of the Danish Ministry of the Environment and the Danish Environmental Protection Agency and gathered international experts to discuss endocrine disrupters from a regulatory perspective. The application of dose addition as an assessment method was recommended as a default, until evidence as to the suitability of alternative assessment concepts emerges. It should replace the current risk assessment paradigm that is focused on single chemicals, without considering contribution from other substances (Kortenkamp & Hass, 2009).

In addition, the European Union Scientific Committees SCCS, SCHER and SCENIHR addressed the need for methods to evaluate the risk of chemical mixtures, i.e. real-world complex mixtures as well as mixtures of few individual chemicals, in their Opinion on "Toxicity and Assessment of Chemical Mixtures" (SCHER/SCENIHR/SCCS 2011). They list a number of conclusions including the statement that *"chemicals with common modes of action may act jointly to produce combination effects that are larger than the effects of each mixture component applied singly. These effects can be described by dose/concentration addition"*. It is noted that the knowledge gap in information on mode of action of a wide range of chemicals is important, and that dose/concentration addition could also be applied when no information on

knowledge on mode of action is available. Further, a decision tree on evaluating risk of chemical mixtures is proposed.

In relation to phthalates, an important recent development is the establishment of a US National Academy of Sciences committee on combined risk assessment for phthalates and related chemicals at the request of the US EPA. A report of this committee was made publicly available in December 2008 and in the summary of this report is stated: *"Thus, the evidence supports the use of dose-addition as an approximation in estimating cumulative risk posed by phthalates and other anti-androgens. The use of a dose-addition model is also supported by data that show cumulative effects at doses at which individual mixture components did not induce observable effects" and moreover; "Cumulative risk assessment based on common adverse outcomes is a feasible and physiologically relevant approach to the evaluation of the multiplicity of human exposures"* (NCR 2008).

In December 2009, a report commissioned by the European Commission called "state of the art on mixtures" was published. This report details the findings of a project on mixture toxicology and ecotoxicology. It describes the scientific state of the art in the field, and gives an account of the regulatory state of the art for dealing with combined exposures in EU, USA, Japan and in international bodies. Here it was stated that there is a consensus in the field of mixture toxicology that the customary chemical-by-chemical approach to risk assessment might be too simplistic and that therefore there is a danger of underestimating the risk of chemicals to human health and to the environment. Moreover, it was concluded that here is unanimous agreement across all disciplines that, in the case of mixtures of similar compounds, combination effects require special consideration (Kortenkamp et al. 2009).

Recently, both the chronic hazard advisory panel on phthalates and phthalate alternatives (CHAP) and Health Canada applied the concept of dose addition for combined risk assessments of phthalates acting through a common mode of action with biological pathways leading to common effects on reproductive development (CHAP 2014; Health Canada 2015).

Scientific background

Experimental data on combination effects of phthalates from multiple studies provide strong evidence that dose-addition can produce good approximations of mixture effects across the relevant dose range when the effects of all components are known (Hannas et al. 2011; Howdeshell et al. 2008; Howdeshell et al. 2015).

Figure B1 illustrates how dose addition modelling based upon data from five individual phthalates (DEHP, DBP, DIBP, BBP and DPP[23]) provided an accurate prediction of the observed effect of the mixture on foetal testosterone production (Howdeshell et al., 2008). Dose additive, inhibitory effects on foetal steroidogenesis were observed. The dose addition predictions fell within the confidence limits calculated for the observed effects of the mixture at every dosage level. The authors conclude that *"there is a credible rationale for the use of foetal testicular testosterone inhibition rat data in the USEPA risk assessment for DBP (and other*

[23] DPP is the abbreviation of dipentyl phthalate. A sixth phthalate, diethyl phthalate (DEP), was shown not to be anti-androgenic.

phthalates) as well as the inclusion of cumulative risk to multiple antiandrogenic phthalate exposure" (Howdeshell et al. 2008).

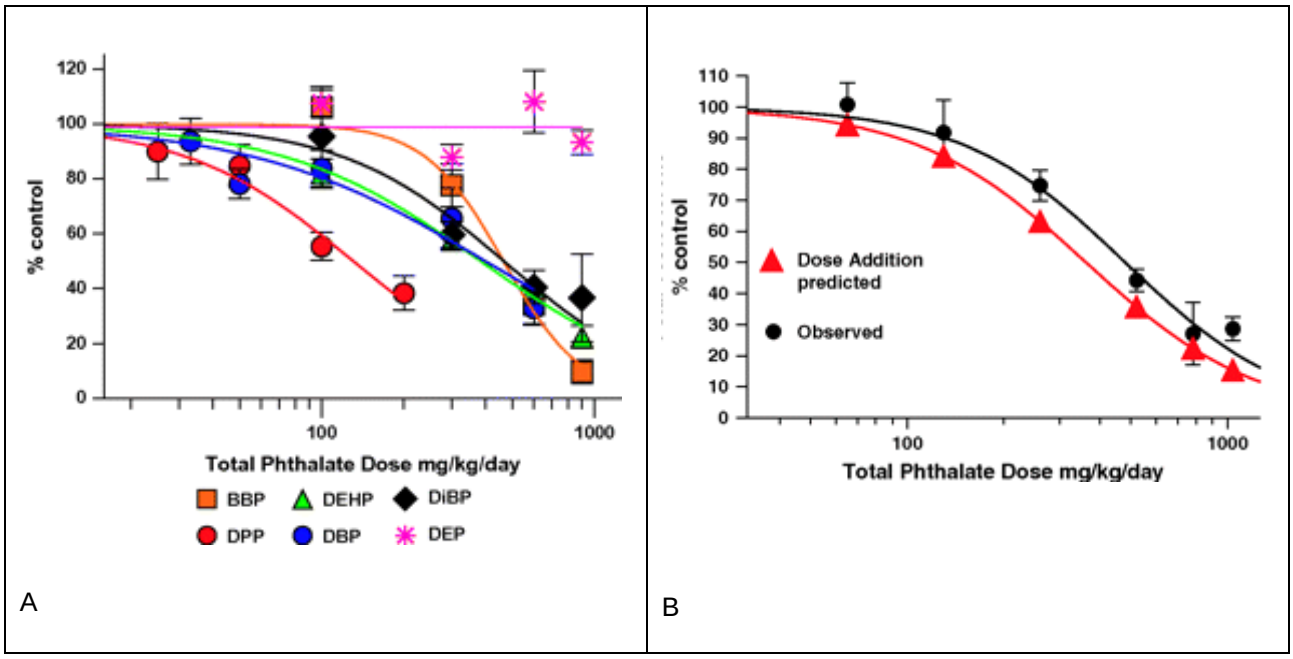


Figure B1 Mixture effects on foetal testosterone production in GD 18 SD rat foetuses exposed to a combination of five phthalates

Note: Mixture effects on foetal testosterone production in GD 18 SD rat foetuses exposed to a combination of five phthalates administered to the dam via gavage on GD 8–18 (from Howdeshell et al. 2008). foetal testicular testosterone production, single phthalates (A) and mixture (B). B: Red triangles indicate predicted dose addition response. The top dose of the mixture contained a total of 1300 mg phthalate/kg/day, including 100 mg DPP/kg/day and 300 mg/kg/day of each of the following: BBP, DBP, DEHP, and DiBP. Data are litter means ± SE.

Other examples in the literature also show that the effects of combinations of DEHP, DBP, DIBP and BBP and other anti-androgens correspond to effects predicted using dose addition (e.g., Hannas et al. 2012; Rider et al. 2009; Hass et al. 2007, Howdeshell et al. 2007; Hotchkiss et al. 2004).

It is also feasible and justified to utilise dose addition also for combinations of dissimilarly acting chemicals (Kortenkamp et al. 2012). Indeed, mixture studies and multi-component mixture studies (including DEHP, DBP, BBP and DIBP and other substances) indicate that even disparate mechanisms may still produce cumulative dose-additive effects as a result of interactions among the signalling pathways in differentiating tissues (Rider et al. 2010).

Christiansen et al. (2009) observed that the induction of malformations of external sex organs by anti-androgens was synergistic, i.e., the observed responses were greater than would be predicted from the toxicities of the individual chemicals. This was shown in an experiment with a mixture of DEHP, vinclozolin, finasteride, and prochloraz.

Recently, the US CHAP concluded that the assumption of dose addition is adequate for mixtures of phthalates to provide the foundation of a combined risk assessment (CHAP 2014).

Overall, based on both scientific literature and detailed reports on this subject (e.g., NCR 2008, CHAP 2014, Health Canada 2015a) it can be concluded that the mixture effects of

phthalates are adequately predicted with dose addition models. No convincing experimental data is currently available suggesting that using another approach than dose addition would result in an overall better estimation of the hazard component of the combined risk assessment (CHAP 2014).

Calculation methods for dose addition

Several approaches exist to calculate combination effects using the dose addition concept in risk assessment. The most common are Hazard Index (HI), Point of Departure Index (PODI), Relative Potency Factors (RPF) and Toxic Equivalent Factors (TEF).

RAC (2012) judged the use of the HI method appropriate in the case of the four phthalates. The HI approach is described as a useful approach by the Scientific Committees in their joint opinion on "Toxicity and assessment of chemical mixtures" (SCHER/SCENIHR/SCCS 2011). Both CHAP (2014) and Health Canada (2015) evaluated the combined risk of similar acting phthalates (including DEHP, DBP, DIBP and BBP) via the concept of dose addition through a HI approach.

The approach chosen in this assessment is the HI as the most appropriate method.

$$HI = \sum C_i/ADI_i \qquad \text{or modified} \qquad HI = \sum Ci/DNEL_i$$

Where
C_i = the concentration in the mixture or the estimated exposure for the included substance;
ADI_i is the ADI for the included substance; and
$DNEL_i$ is the DNEL of the included substance.

The risk is not controlled if HI>1.

This approach is able to clearly define a potential risk; if the summed HI exceeds 1, there is a risk to e.g. human health. This approach allows for individual substances to act on different relevant endocrine endpoints (common adverse outcome pathways) and the HI approach also can be used to look at substances without allocated ADIs, using DNELs instead. The individual DNELs are used to calculate individual Risk Characterisation Ratios (RCRs) (exposure divided by DNEL). Then, these substance-specific RCRs are summed to a total RCR for all four phthalates and for all exposure routes.

B.2. Classification and labelling

Harmonised classification and labelling in Annex VI of Regulation (EC) No 1272/2008 (CLP Regulation) is presented in Table B2.

Table B2. Harmonised classification and labelling of the four phthalates according to Regulation 1272/2008.

Substance	**CAS no.**	**Classification and labelling according to Regulation 1272/2008**	
		Hazard class and category codes	**Hazard statement codes**
DEHP	117-81-7	Repr. 1B	H360-FD
BBP	85-68-7	Repr. 1B Aquatic Acute 1 Aquatic Chronic 1	H360-Df H400 H410
DBP	84-74-2	Repr. 1B Aquatic Acute 1	H360-Df H400
DIBP[24]	84-69-5	Repr. 1B	H360-Df

B.3. Environmental fate properties

This report mainly focusses on Human Health effects of the four phthalates and the environmental properties are only presented qualitatively. See ECHA (2012a), Annexes 2,4,7 and 9 for more details.

B.4. Human health hazard assessment

The four phthalates DEHP, DBP, DIBP and BBP are all classified as toxic to reproduction in category 1B. The ECHA Member State Committee (MSC) has confirmed that these four phthalates are endocrine disruptors (see also section B.4.5.3). This section briefly summarises the human health effects of the four phthalates DEHP, DBP, BBP and DIBP, with a focus on reproductive toxicity, and it is reproductive effects that forms the basis of the N/LOAELs carried forward for the combined risk assessment of these substances. More detailed information on the human health properties of these phthalates is given in Annex 2-5 of ECHA (2012a).

[24] DIBP has a specific concentration limit (SCL) of 25% for mixture classification (Repr. 1B).

B.4.1. Toxicokinetics

To be able to estimate the risk from exposure by different routes, internal dose estimates can be established for exposure estimates that are compared to derived no-effect level estimates (here called “DNELs for internal dose”). Table 3 lists the applied absorption fractions used in the calculation of internal exposure for humans and experimental animals.

For DEHP, an oral absorption fraction of 70% has been determined in rat and monkey studies. The EU RAR for DEHP uses a 100% oral absorption fraction for infants/children as these are considered susceptible and an oral absorption fraction of 50% for adults. Recent studies provide information on the oral absorption of DEHP in humans. Anderson et al. (2011) dosed D_4-DEHP as a single oral dose to 10 male and 10 female subjects. Excreted amounts were then calculated using urine volumes collected over 48h post-dose. The four metabolites of DEHP amounted to 47.1 ± 8.5% fractional excretion on a molar basis. Another study (Koch et al., 2005) based on only one subject but using 3 different doses, separated by a week, resulted in a 65% urinary excretion.). After the opinion of RAC on the four phthalates in 2012 (RAC 2012), the Committee has re-evaluated available data on DEHP studies with human volunteers in later opinions (RAC 2013 a,b). The amount of DEHP recovered is dependent on type and amount of metabolites measured in urine. According to RAC, “measuring all metabolites most likely would result to in near 100% recovery of radioactivity in urine” (ECHA 2013). In this report it is therefore decided to follow the opinion of RAC and use an oral absorption fraction of 100% in adults. For DBP and BBP, the absorption fractions used are equal to those used in the EU RAR. Due to similarities between DBP and DIBP it is assumed that DIBP has the same absorption fraction as DBP.

Table B3. Absorption fractions for calculation of internal doses according to RAC (ECHA 2012a, 2013b,c).

	Absorption fraction, oral	**Absorption fraction, dermal**	**Absorption fraction, inhalation**
DEHP	70% rats, all ages 100% adult humans 100% infants/children	5% human, all ages	75% adults 100% infants/children
DBP and DiBP	100% (exp animals and humans)	10% human, all ages	100% human, all ages
BBP	100% (exp animals and humans)	5% human, all ages	100% human, all ages

B.4.2. Toxicity for reproduction

Previously, when looking at all the available relevant reproductive toxicity studies in the Annex XV restriction report from Denmark, RAC (ECHA 2012a) recognised that multiple mechanisms may have occurred at the same time, leading to several effects that follow from an anti-androgenic mode of action. The effects include early marker effects, morphological and functional effects. Although early marker effects may not be adverse per se, RAC concluded that in the case of the four phthalates all effects attributable to an anti-androgenic mode of action (be it functional or an early marker) are relevant endpoints, since they are so consistently observed in connection with each other in the available studies (ECHA 2012a).

Table B4 lists studies on developmental effects on male reproduction in rodent studies on the four phthalates, i.e. effects in male offspring. No data on direct effects of adult phthalate exposure on male fertility or male reproductive organs are presented, as these effects are only seen at higher doses than effects in offspring and therefore not critical for selection if the starting point for DNEL derivation. However, as described in details in risk assessment reports for DEHP, DBP and BBP and in the SVHC (CMR) dossier for DIBP, these phthalates also affect mating and fertility in adult males, sperm count, male reproductive organ weights and histology (EU RAR 2004, 2008a, 2008b; ECHA 2009d). Also female reproductive function is affected by these phthalates, but as effects are seen at higher dose levels than developmental effects on male reproduction, female reproductive effects are not presented here, as dose levels protecting male offspring also will protect female offspring and adult males/females.

Table B4. Summary of studies on developmental effects on male reproduction in rodent studies on DEHP, DBP, DiBP and BBP.

	NOAEL	**LOAEL**	**Notes on key studies**	**Endpoint and species**	**Commentary**	**Reference**
	mg/kg bw/day					
DEHP	3	10	NOAEL 3 is close to NOAEL 5 from Wolfe and Layton	↓ AGD, ↑ Nipple retention, rat		Christiansen et al. (2010)
	-	3		↑ mild dysgenesis of external genitalia, rat	'alternate' LOAEL used by Christensen et al. 2014	Christiansen et al. (2010)
	4.8	14	NOAEL 5 is the highest NOAEL below the lowest LOAEL. Accepted in EU RAR and EFSA	Reproduction (germ cell depletion, ↓ testis weight), developmental toxicity, rat	EU RAR (2008), EFSA	Wolfe and Layton (2003)
	100	300		↓ Testosterone GD 18, rat		Howdeshell et al. (2008)
	100	300		↓ Testosterone GD 18, rat		Hannas et al. (2011)
	1.2	5		Reproduction (↑ cryptorchidism); ↓ daily sperm production at 15		Andrade et al. (2006)
DBP		**2**	Lowest LOAEL but much lower than from other studies. Accepted in EFSA opinion 2005.	Reduced spermatocyte development PND 21, mammary gland changes in adult males, rat.	EFSA (2005) Dose 20 ppm equals 1.5 to 3 mg/kg bw/day	Lee et al. (2004)
	200	1000	Same study as above	↓ AGD ↑ nipple retention, rat	Dose 2000 ppm equals 148 to	Lee et al. (2004)

	NOAEL	LOAEL	Notes on key studies	Endpoint and species	Commentary	Reference
	mg/kg bw/day					
					291 mg/kg bw/day	
	-	100		↓ AGD GD 21 (↓ testosterone and ↑ nipple retention and histological testis effects from 500)		Martino-Andrade et al (2009)
	-	250		↑ cryptorchidism (↓ AGD and ↑ hypospadias from 500), rat		Jiang et al. (2007)
	50	250	Alternative NOAEL 50	↓ AGD, repro organs, sperm prod., rat		Zhang et al. (2004)
	-	250		Hypospadias, ↓ sperm prod, cryptorchidism, rat		Gray et al. (1999)
	-	100		At 100: delayed preputial separation. At 250: ↓ AGD ↑ nipple retention. At 500: testicular effects, rat.	EFSA (2005), EU RAR (2003)	Mylchreest et al. (1999)
	50	100	Alternative NOAEL 50	↑ nipple retention, rat. At 500: ↓ AGD, repro organs		Mylchreest et al. (2000)
	10	50		↓ Testosterone GD 19, rat		Lehmann et al. (2004)
	100	300		↓ Testosterone GD 18, rat		Howdeshell et al. (2008)
	-	52		Embryotoxicity, rat. ↓ No of live pups, pup wt. Organ wt change from 520, testicular effects from 256.	EU RAR (2003)	Wine et al. (1997)
DiBP	ND	125		Testes atrophy, ↓ prostate weight, ↓ AGD, rat		Sallenfait et al. (2008)
	100	300		↓Testosterone GD 18, rat		Howdeshell et al. (2008)
	100	300		↓ Testosterone GD 18, rat		Hannas et al. (2011)
BBP	**50**	250	50 – Highest NOAEL	↓ AGD, rat (nipple retention and repro organs from 750)	EFSA, EU RAR (2007)	Tyl et al. (2004)

	NOAEL	LOAEL	Notes on key studies	Endpoint and species	Commentary	Reference
	mg/kg bw/day					
	100	500		↓ AGD, repro organs, rat	EU RAR (2007)	Nagao et al. (2000)
	-	100		↓ AGD (sign. testicular effects at 400)		Aso et al. (2005)
	100	300		↓ Testosterone GD 18, rat		Howdeshell et al. (2008)
	185	375		Developmental toxicity, rat	NTP (2003)	Ema et al. (1990)
	20	100		reduced reproductive organ weights and altered sperm counts and motility	Not available in EU RAR (2007) and RAC opinion (ECHA 2012a)	Ahmad et al. (2014)
	182	910		Developmental toxicity, mice	NTP (2003)	Price et al. (1990) in NTP-CEPHR (2003)

[#] Selected NOAEL/LOAEL to be used in DNEL setting (in bold) and studies accepted in EU RAR or by EFSA opinions.

The N(L)OAELs selected for risk assessment are based on developmental effects on male reproduction such as altered testicular development, delayed puberty onset, and increased incidence of hypospadias and cryptorchidism. Additionally, decreases in anogenital distance (AGD) and increases in nipple retention in male offspring are considered robust markers of anti-androgenic effects of chemicals. These are clearly related to adverse reproductive effects in offspring such as altered development of reproductive organs, impaired semen quality, and increased incidence of hypospadias and cryptorchidism (Christiansen et al. 2008, Hotchkiss et al. 2007, McIntyre et al. 2002). Therefore, changes in nipple retention and AGD are also used for NOAEL selection in cases when these changes are present at lower doses than the other reproductive effects[25] observed for these four phthalates.

The selected NOAELs for DEHP and BBP are supported by several studies leading to NOAELs within the same range. For DBP, the studies listed in Table B4 show a large variation in NOAELs/LOAELs. DIBP is structurally similar to DBP, but few reproductive studies have been published for this substance. For the four phthalates, details on the critical studies are described in the subsequent sections (B.4.2.1 to B.4.2.4), and a summary of selected NOAELs is given in section B.4.2.5. A summary of studies on developmental effects on male reproduction for the four phthalates are given in Table B4.

Figure B2 illustrates the cellular targets and the associated changes in gene expression and subsequent hormonal and organ responses after exposure to antiandrogenic phthalates. The spectrum of effects is known as the “phthalate syndrome” (Foster 2006; NRC 2008;

[25] The term reproductive toxicity is here used for effects on reproductive organs and fertility of offspring investigated in one- or multi-generation studies and also direct effects on reproductive organs and fertility of animals exposed as adults in a repeated dose study. Here, the term developmental toxicity is used for effects on offspring of exposed dams.

Kortenkamp et al. 2011; CHAP 2014; Health Canada 2015). It is well understood that the cause for the phthalate syndrome is suppression of foetal androgen action (Kortenkamp et al. 2011). It is hypothesized that these disorders may comprise a "testicular dysgenesis syndrome" (TDS) in humans with a common origin in foetal life. Testicular cancer may also be part of TDS in humans.

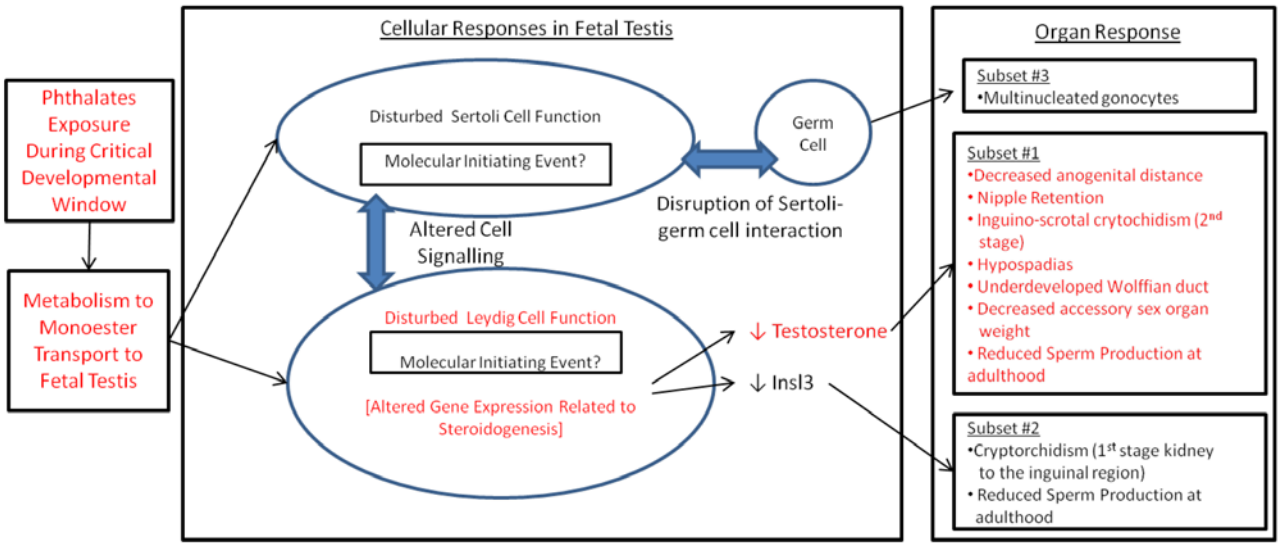


Figure B2. Representation of the cellular targets of the rat "phthalate syndrome". The associated changes in gene expression, and subsequent hormonal and organ responses. Source: Health Canada (2015a).

B.4.2.1. DEHP

Studies by Wolfe and Layton (2003), Christiansen et al. (2010) and Andrade et al. (2006) are critical for the NOAEL selection. Andrade et al. (2006), described increased incidence of cryptorchidism from 5 mg/kg bw/day in male rats exposed to DEHP from GD 6 to PND 21, with a NOAEL of 1.215 mg/kg bw/day. Christiansen et al. (2010), found reduced anogenital distance and increased nipple retention in male rats perinatally (GD7 to PND 16) exposed by gavage to 10 mg DEHP/kg bw/day and above, with a NOAEL of 3 mg/kg bw/day (Christiansen et al., 2010). A NOAEL of 4.8 mg/kg bw/day was found in the study by Wolfe and Layton (2003), who observed testicular toxicity in offspring exposed to 14 mg DEHP/kg bw/day and above in a multigeneration study with dietary exposure (Wolfe and Layton 2003). When combining NOAELs and LOAELs from these three studies, the highest NOAEL is 4.8 mg/kg bw/day based on the study by Wolfe and Layton (2003) and the lowest LOAEL is 5 mg/kg bw/day based on the findings in the study by Andrade et al. (2006). If the study by Andrade et al. (2006) is not taken into consideration (due to the observation of cryptorchidism in only few animals), the lowest LOAEL would be 10 mg/kg bw/day based on the study by Christiansen et al. (2010), and that would not change the NOAEL determination from the EU RAR. The **NOAEL of 4.8 mg/kg bw/day** is selected for combined risk assessment. The effects can be attributed to an anti-androgenic mode of action. This study was used as a starting point in the EU RAR (2008) and by RAC (ECHA 2012,2013c).

Details of the key studies

The study by Wolfe and Layton (2003), is described in detail in EU RAR for DEHP and considered acceptable and used as a key study for selection of overall NOAEL by the EU RAR. The study by Wolfe and Layton (2003) is also used as the critical study in the registration dossier for DEHP. Details on the study by Wolfe and Layton (2003) are presented in the EU RAR (page 413 to 424 of the EU RAR) presented in Annex 2. In brief, this study is a multigeneration study in Sprague-Dawley rats exposed to dietary concentrations of DEHP of 1.5, 10, 30, 100, 300, 1000, 7500, and 10000 ppm of DEHP (n=17 males and females), corresponding to 0.1, 0.47, 1.4, 4.8, 14, 46 and 359 mg/kg bw/day in F2 animals. It should be noted that the control group received 1.5 ppm of DEHP, as this was the amount of DEHP found in control feed. Testicular effects were most prominent in F1 and F2 animals, and a NOAEL of 100 ppm corresponding to 4.8 mg/kg bw/day in F2 animals was determined by the EU RAR as the critical NOAEL for testicular toxicity and developmental (testicular) toxicity (EU RAR page 424). For a comprehensive discussion of this NOAEL selection is referred to the relevant pages of the EU RAR, Annex 2.

The study by Christiansen et al. (2010), is not described in the EU RAR (2008) but is included in the registration dossier for DEHP. Christiansen et al. (2010), describes two non-guideline, non-GLP studies with exposure of time-mated Wistar rats from GD 7 to PND 16 by gavage with DEHP in corn oil. Study 1 included 16 mated dams in the control group and 8 mated dams per group in six exposure groups receiving either 10, 30, 100, 300, 600 or 900 mg/kg bw/day of DEHP. Study 2 included 16 mated dams in the control group, 16 mated dams receiving 3 mg/kg bw/day of DEHP, and 8 mated dams per group receiving either 10, 30, or 100 mg/kg bw/day of DEHP. A number of reproductive endpoints were investigated postnatally and at PND 16. In a combined evaluation of the two studies the anogenital distance was significantly decreased and the number of nipples significantly increased at 10 mg/kg bw/day of DEHP with a NOAEL of 3 mg/kg bw/day. At the same dose (10 mg/kg) and above, decreased weights of ventral prostate and levator ani/bulbocavernosus muscle were observed, though these effects did not show a clear dose-response relationships. Additionally, mild dysgenesis of external genitals[26] was observed at all doses and also in one of the male control rats. When the two studies were combined the incidences of mild dysgenesis were significantly increased at all dose levels except 30 mg/kg ($p = 0.075$ for litter incidences). See Figure B3, below. There is thus not a clear dose-response relationship in the percentage of affected males. Although DEHP appears to affect external genitals at the lowest dose level, the effect on reduced anogenital distance at 10 mg/kg bw/day may be considered a more robust LOAEL for DEHP. Indeed, the authors concluded that the results are consistent with the NOAEL of 5 mg/kg bw/day.

[26] Christiansen et al. (2010) defined mild dysgenesis of the external genitalia as follows: “Score 1, mild dysgenesis of the external genitalia:a small cavity on the caudal surface of the genital tubercle or a minor cleft (+) in the preputial opening is observed. The furless area around anus expands towards the base of the genital tubercle, but thick fur is still present at the base of the genital tubercle.”

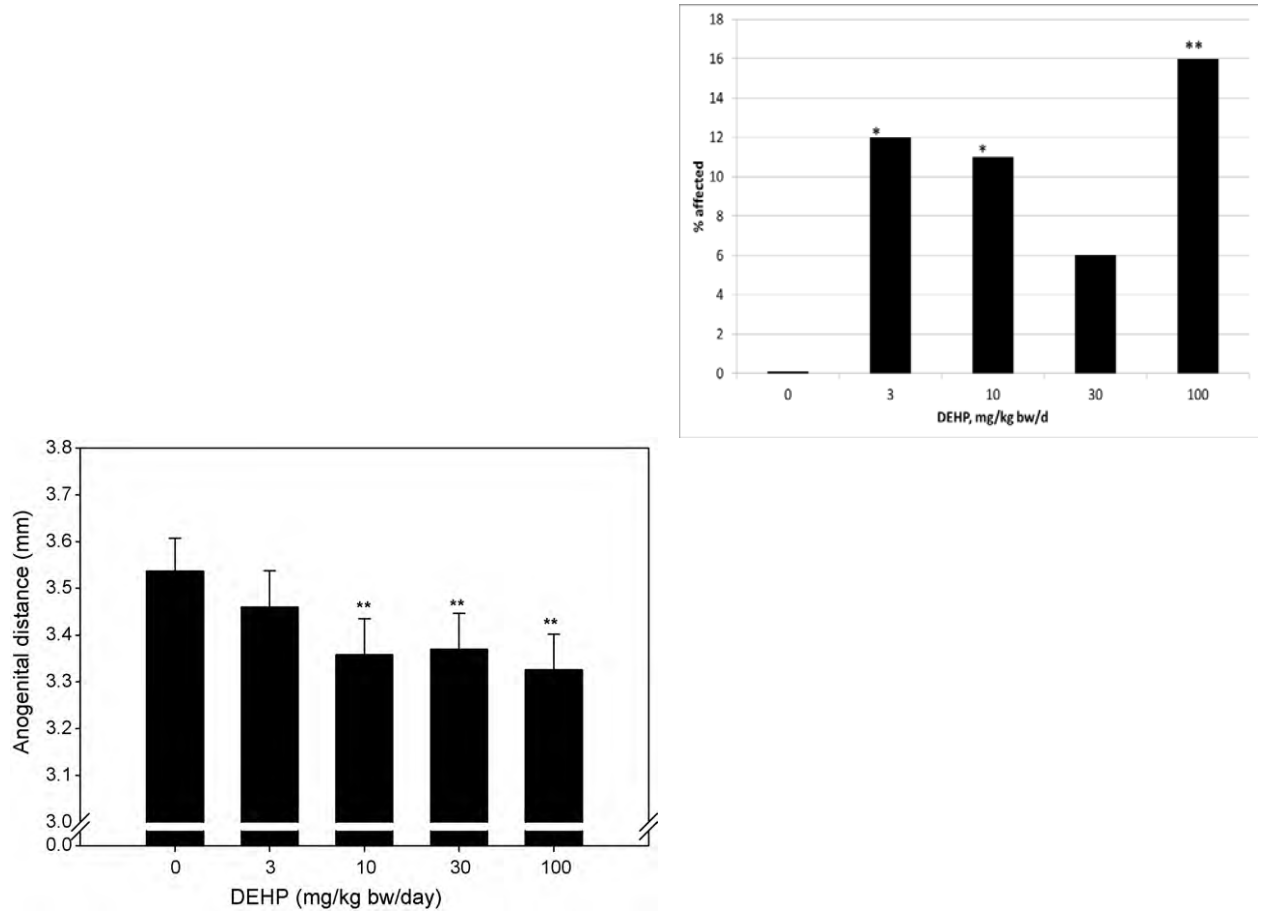


Figure B3 Mean anogenital distance (AGD) on PND 1 (left) and mild external dysgenesis on PND 16 (right) in male rat offspring of dams administered corn oil (control), 3, 10, 30 or 100 mg/kg-d DEHP from GD 7 to PND 16. Least square means + SEM are shown for AGD and the data are corrected for body weight and litter effect. Frequency of affected male offspring is shown for mild external dysgenesis ; *Indicates $p < 0.05$; **Indicates $p < 0.01$. From Christiansen et al. (2010).

Andrade et al. (2006), describes a study on in utero and lactational exposure of Wistar rats to DEHP at low and high doses by gavage, and showed effects on daily sperm production from 15 mg/kg bw/day and a low, but increased incidence of cryptorchidism at 5 mg/kg bw/day. Pregnant Wistar rats were gavaged from GD 6 to PND 21 with 0, 0.015, 0.045, 0.135, 0.405, 1.215, 5, 15, 45, 135 and 405 mg DEHP/kg bw/day (n=11 to 16 litters per dose). Effects on hormone levels were seen at low doses, but did not exhibit dose-response relationships. In males exposed to 1.215 mg/kg bw/day and at doses from 15 mg/kg bw/day and above (i.e. not at 5 mg/kg bw/day), daily sperm production was reduced by 19-25% compared to controls from the same study and compared to historical controls. The authors concluded a LOAEL of 15 mg/kg bw/day for this effect. Three animals exposed to 5, 135, and 405 mg/kg bw/day of DEHP, respectively, had undescended testes (cryptorchidism). The authors concluded a NOAEL of 1.215 mg/kg bw/day based on cryptorchidism despite the low number of affected animals, as cryptorchidism is less common in Wistar rats compared to other rat strains. The weights of testes and epididymides were not affected in any treatment group, whereas the weight of

seminal vesicle plus coagulating glands was significantly reduced at the highest dose group. Ventral prostate weight was also reduced at this dose, although not statistically significantly.

However, as this LOAEL of 5 mg/kg bw/day is above the selected NOAEL of 4.8 mg/kg bw/day in the study by Wolfe and Layton (2003), including this finding does not affect this overall NOAEL selection.

Other studies

Other studies on DEHP were considered for the NOAEL selection but were not considered critical. A large number of reproductive, developmental and mechanistic studies published up to 2005 are described in the EU RAR (2008) (Table 4.58 "Important reproductive studies with DEHP in laboratory animals" in EU RAR), but as the EU RAR did not consider these studies critical with respect to reproductive effects of DEHP, these studies were not evaluated further.

Additionally, the registration dossier for DEHP includes the following studies, which were also not considered critical:

- Studies by Howdeshell et al. (2008) and Hannas et al. (2011) are included in Table B4 and describe effects of DEHP on foetal testosterone production in rats at doses from 300 mg/kg bw/day. These are some of several mechanistic studies describing inhibitory effects of DEHP on foetal testosterone at higher levels than those inducing other male reproductive effects.
- Hannas et al. (2011) compared the exposure of SD and Wistar rats to 0, 100, 300, 500, 625, 750 or 875 mg DEHP /kg/day from GD 14 to 18. Testicular testosterone production ex vivo was assessed by incubation of testes of 18 day old foetuses for 3 hours and testosterone measurement in the media. Despite differences in testosterone production values in the two strains, the same response was seen, i.e. a decrease in testosterone production at 300 mg/kg bw/day and above with a NOAEL of 100 mg/kg bw/day.
- Howdeshell et al. (2008), found that DEHP decreased foetal testosterone production in rats at doses from 300 mg/kg bw/day (NOAEL 100 mg/kg bw/day). In this study, pregnant Sprague-Dawley rats were exposed to 0, 100, 300, 600, or 900 mg/kg bw/day of DEHP from GD 8 to 18 by gavage in corn oil (n=5 to 8 dams per group). Testicular testosterone production ex vivo was assessed by incubation of testes of 18 day old foetuses for 3 hours and testosterone measurement in the media. Dose-related decreases in testosterone production was seen for DEHP and the other tested phthalates (BBP, DBP, and DIBP) from 300 mg/kg bw/day and above, and for DPP (dipentyl phthalate) from 100 mg/kg bw/day.
- The following studies published after 2003 all included doses higher than 10 mg/kg bw/day (=LOAEL) and were therefore not taken into consideration for NOAEL determination: Tanaka et al. (2003), Tanaka et al. (2005), Gray et al. (2009), Wilson et al. (2007), Howdeshell et al. (2007), Shirota et al. (2005), Borch et al. (2005), Tomonari et al. (2006), Cammack et al. (2003), Wilson et al. (2004), Liu et al. (2008), Noriega et al. (2009).

The following recent studies examined doses at or below 10 mg/kg bw, but were not considered critical:

- A study by Jones et al. (2014) found permanently reduced mRNA levels of the gene Hsd3b in adult testes following in utero exposure to 10 mg/kg bw/day of DEHP. Hsd3b codes for the steroidogenic enzyme 3beta hydroxysteroid dehydrogenase (3βHSD), and reduced

levels of this protein were seen in Leydig cells of these DEHP exposed adult animals. This points to permanent effects of foetal exposure to low doses of DEHP.

- A study by Zhang et al. (2014) examined the ovarian effects of a very low dose of DEHP (40 µg/kg bw/day by gavage in DMSO and water from GD 1 to 19). Reduced primordial follicle number and increased secondary follicle number was observed in F1 offspring at PD 21. Also the second generation females (F2) had reduced numbers of primordial follicles and increased numbers of secondary follicles at PD 21, indicating accelerated follicle recruitment in two generations. These findings were associated with changes in gene methylation and expression of several genes in F1 ovaries. These findings may indicate an effect at a lower dose than the NOAEL selected above for DNEL determination, but only one dose group is included and the number of dams/litters per endpoint is not clearly presented. It is not clear whether these possible low-dose effects are related to an anti-androgenic mode of action and this study was not considered relevant for DNEL determination.
- Two studies by Zhang et al. (2013a and 2013b) described effects of low doses of DEHP on mouse testes and ovaries, respectively. As intradermal exposure was applied, these studies were not considered relevant for DNEL determination.

B.4.2.2. DBP

Lee et al. (2004), found reduced spermatocyte development in prepubertal rats and mammary gland changes in adult male rats perinatally (GD 15 to PND 21) exposed to 2 mg DBP/kg bw/day and above via the diet. No NOAEL was determined. In that study, anogenital distance was reduced and nipple retention was increased in males at 1000 mg DBP/kg bw/day with a NOAEL of 200 mg/kg bw/day. The EU RAR (2004) on DBP from 2003 uses an overall LOAEL of 52 mg/kg bw/day for embryotoxicity based on a study by Wine et al., 1997. In 2005, EFSA concluded a Tolerable Daily Intake (TDI) of 0.01 mg/kg bw/day for DBP based on delayed germ cell development and male mammary gland changes in the study by Lee et al. (2004) (see ECHA 2012a, Annex 3).

The reduced spermatocyte development observed in the study by Lee et al. (2004), was statistically significant at PND 21 in the lowest dose group and severity was increasing with dose. Impairment of spermatocyte development persisted to adulthood in the higher dose groups only (see study details below). These changes can be related to an anti-androgenic effect of DBP, and although the low-dose effects appear to be reversible, they are a clear sign of developmental influences on testicular development already at these low doses, and are therefore considered relevant for NOAEL determination.

It is also reasonable to regard the observed mammary gland effects as anti-androgenic. A 28-day study on the androgen receptor antagonist flutamide showed a dose-related induction of lobular atrophy in male mammary glands (Toyoda et al., 2000). The authors suggested that the observed lobular atrophy of the mammary glands may be due to an anti-androgenic action on acinar cells, as also seen in *in vitro* studies (Toyoda et al., 2000; Boccuzzi et al., 1995; Sourla et al., 1998). The same mechanism of action may apply to the lobular atrophy observed with DBP in the study by Lee et al. (2004).

As the observed effects of DBP on mammary gland and testes are considered anti-androgenic, and as EFSA has chosen to change the TDI in favour of the study by Lee et al., the **LOAEL of 2 mg/kg bw/day** is suggested for use in the current combined risk assessment.

Details of the key studies

Details of the studies by Lee et al. (2004) can be found in the EFSA opinion from 2005 and is cited below. It should be noted that doses in mg/kg feed per day can be divided by 10 to reach doses in mg/kg bw/day. According to the authors and the EFSA opinion, 20 mg/kg feed corresponds to 1.5 to 3.0 mg/kg bw/day.

> *"In a recent developmental toxicity study (Lee et al., 2004) with exposure during the period from late gestation (Gestational day 15) to the end of lactation on postnatal day 21 (PND 21), maternal rats were given DBP at dietary concentrations of 0, 20, 200, 2000 and 10000 mg/kg. Major results of this study are summarised below. At PND 2, anogenital distance was significantly reduced in 10000 mg/kg male offspring. At PND 14, the incidence of retained nipples/areolae was increased in all treated male offspring compared with controls but the increase was only significant at 10000 mg/kg. At PND 21, in males, reduction of spermatocyte development as manifested by a decreased number of spermatocytes was observed from 20 mg/kg with dose-dependent increased incidence or/and severity. A significant increase in scattered foci of aggregated Leydig cells was observed at 2000 mg/kg and 10000 mg/kg. In the epididymis, significantly decreased ductular cross sections, indicating reduced coiling, were observed at 2000 and 10000 mg/kg. In the mammary glands, dilatation of alveolar buds and/or ducts was seen in male offspring from 20 mg/kg with low incidence but not achieving statistical significance in any group. In female offspring, hypoplasia of the alveolar buds of the mammary glands was observed in animals from 20 mg/kg with a statistically significant increase at 20, 200, 2000 and 10000 mg/kg ($P<0.05$). At postnatal week 11 (PNW 11), in males, loss of germ cell development was significant at 2000 mg/kg and above. This lesion differed markedly in severity between animals. Significant increases in vacuolar degeneration in the mammary glands of males was present from 20 mg/kg but with similar incidence and qualitative gradation of change across the dose groups"* (EFSA opinion 2005).

Other studies

Other studies on DBP were included in the NOAEL determination by EFSA but were not considered critical. These studies include developmental and reproductive studies described in detail in the EU RAR (page 86 to 98, see Annex 4): Lamb et al. (1987), Morrisey et al. (1989), Gray et al. (1999), Mylchreest et al. (2000), NTP (1995), Wine et al. (1997). The EU RAR also describes the following studies that were not discussed in the EFSA opinion from 2005: Nikoronow et al. (1973), IRDC (1984), Hamano et al. (1997), Shiota et al. (1980), Ema et al. (1993), Mylchreest et al. (1998), Mylchreest et al. (1999) (see EU RAR for precise references).

The registration report for DBP quotes strictly the EU RAR regarding reproductive and developmental toxicity and therefore describes the same studies. The EU RAR determines a LOAEL of 52 mg/kg bw/day based embryotoxicity in the study by NTP (1995)/Wine et al. (1997). As embryotoxicity is not considered an anti-androgenic effect, this LOAEL is not considered for the current combined risk assessment. The lowest LOAEL reported in the EU RAR and related to anti-androgenicity is 100 mg/kg bw/day in a study by Mylchreest et al.

(1999), in which delayed preputial separation was seen at the lowest dose of 100 mg/kg bw/day. The study description from the EU RAR follows below:

> *"In a follow-up study of Mylchreest et al. (1999) DBP was shown to disrupt the androgenregulated male sexual differentiation during prenatal exposure, without interacting directly with the androgen receptor, as does flutamide, a known antiandrogen. At the highest dose-level of 500 mg/kg bw (in corn oil), given orally by gavage to pregnant rats during day 12-21 of gestation, one dam showed weight loss after day 18 of pregnancy and delivered dead and moribund fetuses. At all dose levels (100, 250 and 500 mg/kg bw) delayed preputial separation in F1 males (killed at sexual maturity at the age of 100-105 days) was seen. At the lowest dose level of 100 mg DBP/kg bw this delay (of 2 days) was attributable at least in part, to one markedly affected litter. Furthermore malformations of the (F1) male reproductive tract were observed at 250 and 500 mg/kg bw, i.e. retained thoracic nipples and decreased anogenital distance. In addition, at 500 mg/kg bw hypospadias, cryptorchidism, agenesis of the prostate, epididymis, and vas deferens, degeneration of seminiferous epithelium and interstitial cell hyperplasia (5 animals from 2 litters) of the testis were seen. Interstitial cell adenoma occurred at 500 mg/kg bw in 2 males (in one litter). In F1 females no abnormal uterine or vaginal development or kidney agenesis were seen. In contrast to flutamide, DBP caused a low incidence of prostate agenesis and hypospadias with no vaginal pouch."* (EU RAR, page 95).

A second follow-up study by Mylchreest et al. (2000), is mentioned by EFSA (2005), but not by the EU RAR from 2003. In this study a NOAEL of 50 and a LOAEL of 100 mg/kg bw/day was determined based on nipple retention in male pups at 100 mg/kg bw/day and above. This study examined exposure of pregnant CD rats to DBP by gavage from GD 12 to 21 at the doses of 0, 0.5, 5, 50, 100 or 500 mg/kg bw/day. Nipple retention was the only effect observed at 100 mg/kg bw/day, and at 500 mg/kg bw/day decreased anogenital distance of males, hypospadias and absence or malformations of epididymis, vas deferens, seminal vesicles and ventral prostate was seen together with decreased widths of male reproductive organs and histological changes in testes.

A number of reproductive/developmental studies have been published after the EU RAR from 2003. A study by Zhang et al. (2004), detected a NOAEL of 50 mg/kg bw/day based on decreased anogenital distance of males and effects on male reproductive organs and sperm production of rats exposed to 250 or 500 mg/kg bw/day of DBP in utero and during lactation (GD 1 to PND 21).

A number of reproductive, developmental and/or mechanistic studies applying large doses of DBP are not described here, as these were not considered relevant for NOAEL determination (Ryu et al., 2008, Jiang et al., 2007 and more). Among the mechanistic studies are dose-response studies on the inhibitory effect of DBP on foetal testosterone production: Howdeshell et al. (2008), described that DBP decreased foetal testosterone production in rats at doses from 300 mg/kg bw/day (NOAEL 100 mg/kg bw/day). In this study, pregnant Sprague-Dawley rats were exposed to 33, 50, 100, 300, or 600 mg/kg bw/day of DBP from GD 8 to 18 by gavage in corn oil (n=3 to 4 dams per group). Testicular testosterone production ex vivo was assessed by incubation of testes of 18 day old foetuses for 3 hours and testosterone measurement in the media. Dose-related decreases in testosterone production was seen for BBP and the other tested phthalates (DIBP, BBP, and DEHP) from 300 mg/kg bw/day and above, and for DPP (dipentyl phthalate) from 100 mg/kg bw/day.

Lehmann et al. (2004) exposed pregnant Sprague-Dawley rats to 0, 0.1, 1, 10, 50, 100, or 500 mg/kg bw/day of DBP from GD 12 to 19 by gavage in corn oil (n=1 to 4 litters per group, analysis of testosterone in 3-4 males per litter). Testosterone concentration and testicular expression of steroidogenesis related genes were measured in testes of 19 day old foetuses. found decreased foetal testosterone concentration in rats exposed to DBP at doses from 50 mg/kg bw/day with a NOAEL of 10 mg/kg bw/day. This corresponded to a reduction in the expression of genes involved in steroid synthesis (SR-B1, StAR, P450scc, 3βHSD) from 50 mg/kg bw/day. At lower doses some of these genes were also sporadically reduced, but effects were not statistically significant at 10 mg/kg bw/day and the most consistent effects were seen above 50 mg/kg bw/day. The power of this study was relatively low with 3 to 5 foetuses examined for each endpoint, and a larger study might reveal effects on e.g. gene expression at lower doses also.

B.4.2.3. DIBP

Few reproductive toxicity studies have been published on DIBP compared to the number of studies published for DEHP and DBP. No two-generation studies are available. In addition, DIBP has only been studied at doses >100 mg/kg bw/day (Saillenfait et al. 2006, 2008; Borch et al. 2006; Boberg et al. 2008; Howdeshell et al. 2008; Hannas et al. 2011, 2012).

Description of key studies

Details of the studies by Saillenfait et al. (2008), Howdeshell et al. (2008) and Hannas et al. (2011 and 2012):

- Saillenfait et al. (2008), describes a study on exposure of pregnant Sprague-Dawley rats from GD 12 to 21 by gavage to 0, 125, 250, 500, or 625 mg/kg bw/day of DIBP (n=11-14 dams per group). Reduced male neonatal anogenital distance and an increased number of nipples in males were observed from 250 mg/kg bw/day exhibiting clear dose-response relationships over the tested dose range. A subtle and not statistically significant reduction in anogenital distance was seen in the lowest dose group. Prostate weight at postnatal week 16-17 was significantly reduced at all doses except at 250 mg/kg bw/day, but these data did not show a clear dose-response. At postnatal week 11-12, reductions in prostate weight were statistically significant from 250 mg/kg bw/day with data showing a clear dose-response pattern and a non-significant reduction also at the lowest dose group. Reductions in other reproductive organ weights were seen from 500 mg/kg bw/day. At the highest doses, 500 and 625 mg/kg bw/day, delays in preputial separation and incidence of malformations (hypospadias, cleft prepuce and undescended testes) were observed in young adulthood and histological changes of testes were observed in adulthood. The observation of histological changes of testes was most marked at 500 and 625 mg/kg bw/day, but mild/infrequent effects were also seen at the two lowest doses. Two of 24 control males had tubular degeneration grade 1 (of 5 grades), whereas 2 of 20 males exposed to 125 mg/kg bw/day of DIBP had tubular degeneration at grade 2 and grade 5, respectively, and 7 of 28 males exposed to 250 mg/kg bw/day of DIBP had tubular degeneration at grade 1 to grade 5. No statistical analysis is presented for histological data. The results seen for the positive control DBP at a dose of 500 mg/kg bw/day showed comparable effects to those seen with 500 and 625 mg/kg bw/day of DIBP. Specifically the effects on anogenital distance, nipple retention, reproductive organ weights and reproductive tract malformations (hypospadias, exposed os penis, cleft prepuce and

cryptorchidism) and puberty onset seen with 500 mg/kg bw/day of DIBP were comparable or *less* marked than the effects seen with 500 mg/kg bw/day of DBP, whereas the effects seen with 625 mg/kg bw/day of DIBP were comparable or *more* marked than the effects seen with 500 mg/kg bw/day of DBP.

- Howdeshell et al. (2008), described that DIBP decreased foetal testosterone production in rats at doses from 300 mg/kg bw/day (NOAEL 100 mg/kg bw/day). In this study, pregnant Charles River Sprague-Dawley rats were exposed to 0, 100, 300, 600, or 900 mg/kg bw/day of DIBP from GD 8 to 18 by gavage in corn oil (n=5 to 8 dams per group). Maternal body weight at GD 18 was reduced at 600 and 900 mg/kg bw/day, whereas maternal body weight gain, the number of live foetuses and total resorptions were decreased at 900 mg/kg bw/day. Testicular testosterone production ex vivo was assessed by incubation of testes of 18 day old foetuses for 3 hours and testosterone measurement in the media. Dose-related decreases in testosterone production was seen for DIBP and the other tested phthalates (BBP, DBP, and DEHP) at 300 mg/kg bw/day and above, and for DPP (dipentyl phthalate) from 100 mg/kg bw/day.
- Hannas et al. (2011), describes a study in which pregnant Harlan Sprague-Dawley rats were exposed to 0, 100, 300, 600, or 900 mg/kg bw/day of DIBP from GD 14 to 18 by gavage (n=3 dams per group). Testicular testosterone production ex vivo was assessed by incubation of testes of 18 day old foetuses for 3 hours and testosterone measurement in the media. Dose-related decreases in testosterone production was seen for DIBP and the other tested phthalates (DEHP and DIHP (diisohexyl phthalate)) from 300 mg/kg bw/day and above (NOAEL 100 mg/kg bw/day), and for DINP from 500 mg/kg bw/day.
- Furr et al. (2014) studied foetal testosterone production in several experiments with the aim to develop and validate the 'Fetal Phthalate Screen' assay.
  In the single dose experiment 27 substances were tested. Pregnant rats were dosed daily by oral gavage at 0 and 750 mg/kg bw/day from GD 14 to 18 (n=3-4 dams per group). At GD 18, foetal testes were removed and 3 testes were per litter were used to measure ex vivo testosterone production. The testosterone production was reduced to a level of about 20% of the control in the DIBP dosed animals and at about 12% for DEHP, DBP and BBP.
  In the dose-response experiment 11 substances were tested. Pregnant Harlan Sprague Dawley rats were dosed daily by oral gavage at 0, 100, 200, 300, 500, 600, 750 and 900 mg/kg bw/day using the same protocol as for the single dose experiment. The ED_{50} was about 290 mg/kg bw/day with DIBP in Harlan SD rats. The ED_{50} was about 160 and 320 mg/kg bw/day with DBP (Harlan and Charles River SD rats respectively); 120 and 340 mg/kg bw/day with DEHP (Harlan SD and Charles River SD rats respectively); 170 mg/kg bw/day with BBP in Harlan SD rats; and 750 mg/kg bw/day with DINP in Harlan SD rats. Harlan SD rats appeared to be more sensitive than Charles River SD rats to reduction of foetal testosterone production from phthalate exposure.
- Hannas et al. (2012), describes gene expression studies in testes at GD 18 in the same study as described by Hannas et al., 2011. A number of steroidogenesis-related genes were downregulated at 300 mg/kg of DIBP and above and also by other phthalates examined (dihexyl-, diheptyl-, dipentyl-, and diisononyl phthalate), but not by diisodecyl phthalate. DIBP downregulated the expression of: StAR, Cyp11a1, HSD3b, Cyp17a1, Scarb1, Insl3, Cyp11b1 and Rxrg, and upregulated the expression of Amhr2 and Sox9. These data were applied for potency ranking of these phthalates and it was concluded that several phthalates including DIBP affected the same pathways.

Other studies

Other studies on DIBP were considered for determination of the starting point for DNEL setting but were not considered critical. The described effects on male anogenital distance and foetal testosterone production confirm findings in a study by Borch et al. (2006), showing decreased anogenital distance and decreased testicular testosterone production and –content in foetal male Wistar rats exposed to 600 mg/kg bw/day of DIBP from GD 7 to 21.

The available registration dossier for DIBP also includes the following studies, which all applied oral doses at or above the LOAEL of 250 mg/kg bw/day and were therefore not taken into consideration for NOAEL determination: Saillenfait et al. (2006), Boberg et al. (2008), and Zhu et al. (2010). A study by Ray et al. (2012) applied intraperitoneal exposure to DIBP and was not considered relevant for DNEL determination. In addition, the registration dossier for DIBP presents a study on DBP (NTP 1995) and justifies use of read-across for effects on fertility; however, this study is not used for NOAEL/LOAEL determination.

For direct effects on male adult fertility, these were reviewed by CPSC (CPSC 2011). Short-term oral exposure to DiBP causes significant adverse testicular effects in male adolescent rats including decreased testes weights, increased numbers of apoptotic spermatogenic cells, disorganized or reduced vimentin filaments in Sertoli cells, elevated testicular testosterone levels, decreased testicular zinc levels, and marked inhibition of spermatogenesis and desquamation of spermatocytes. Effects were seen at doses as low as 500 mg/kg-day (Zhu et al., 2010; Oishi and Hiraga, 1980a). Similar findings were reported in rats treated with MIBP (Foster et al., 1981; Oishi and Hiraga, 1980b) and included studies by Zhu et al. (2010) and Oishi and Hiraga (1980a).

Derivation of the point of departure for DIBP in previous assessments

In the Background Document to the Opinion on the Annex XV dossier proposing restrictions on four phthalates (RAC and SEAC 2012), a DNEL of 0.42 mg/kg bw/day was set for anti-androgenic/reproductive effects of DIBP at 125 mg/kg bw/day (Saillenfait et al., 2008). RAC noted that the LOAEL for effects of DIBP on histological effects in adult testes and epididymides can be considered “conservative” given the low incidences found at the LOAEL, but that a steep dose-response curve was seen in this study.

Also the available registration dossier for DIBP used a LOAEL of 125 mg/kg bw/day for DIBP as a point of departure.

A starting point of 9.8 mg/kg bw/day was applied by US consumer product safety commission (CPSC) in their toxicity review of DIBP (CSPC 2011). This was based on a BMDL10 for effects on foetal testosterone in the study by Howdeshell et al. (2008).

Deriving a new point of departure for DIBP

The experimental data leaves a high degree of uncertainty regarding the established LOAEL of DIBP as the substance has not been tested below 100 mg/kg bw/day but has structural similarity to the other phthalates, in particular DBP. Therefore it was considered important to evaluate the new mechanistic evidence regarding potency and to derive a new point of departure using all the available evidence.

The available experimental studies for reproductive adverse effects of DIBP are limited, and the dose-response curve in Saillenfait et al. (2008) is steep with high incidences (up to 100%) of histological changes in testes at PNW 11-12 and nipple retention on PND 12-14 at 500 and 625 mg/kg bw/day (see

Table B5). Subtle effects are also seen at 125 mg/kg bw/day on anogenital distance, tubular degeneration, oligospermia/azoospermia, and prostate weight. It is noted that non-parametric statistical analyses were applied to the data and did not include body weight in the statistical analysis. The finding that no significant change in body weight was apparent at 250 mg/kg bw/day supports the conclusion that the change in anogenital distance at 250 mg/kg bw/day is real and not a cause of decreased body weight. Delay in preputial separation was similar with DIBP and DBP. The effects on absolute organ weights (testis, epididymis, seminal vesicles and prostate) was similar to DBP in PNW 11-12 but seemed less pronounced at PNW 16-17 with DIBP in comparison to DBP.

Table B5 Incidences of histological changes in testes, nipple retention, AGD and age at preputial separation in Saillenfait et al. (2008)

	DIBP					**DBP**
Dose level (mg/kg bw/day)	**0**	**125**	**250**	**500**	**625**	**500**
Incidences of tubular degeneration-atrophy/hypoplasia at PNW 11-12 (adult)	2/24 (8.3%)	2/20 (10.0%) But higher grade than control	7/28 (25.0%)	16/22 (72.7%)	20/20 (100.0%)	Not reported
Oligospermia + azoospermia	0/24 (0.0%)	2/20 (10.0%)	6/28 (21.4%)	12/22 (54.5%)	19/20 (95.0%)	Not reported
Incidences of nipple retention on PND 12-14	0/76 (0.0%)	0/78 (0.0%)	8/96 (8.3%)	47/79 (59.5%)	56/76 (73.7%)	44/59 (74.6%)
Incidences of nipple retention at PNW 11-12 (adult)	0/46 (0.0%)	0/40 (0.0%)	4/55 (7.3%)	24/44 (54.5%)	29/38 (76.3%)	29/39 (74.4%)
Male AGD PND 1 (mm) Difference compared with control (%)	2.55 ±0.17	2.44 ±0.15 (-4%)	2.28 ±0.30* (-11%)	2.02 ±0.13** (-21%)	1.98 ±0.16** (-22%)	1.94 ±0.17** (-24%)
Age at preputial separation	46.9±1.5	45.1±1.6*	46.3±1.8	51.5±3.1**	49.8±3.2*	50.1±3.1**

* significantly different from control group, $p<0.05$

* significantly different from control group, $p<0.01$

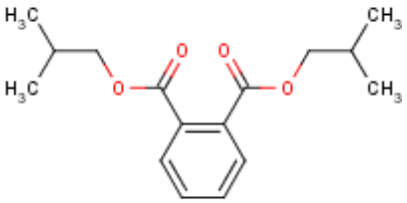
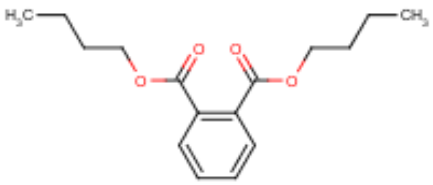
The experimental data leaves a high degree of uncertainty when the selected point of departure is a LOAEL of 125 mg/kg bw/day as DIBP has not been tested below 100 mg/kg bw/day. Therefore it was considered important to evaluate the new mechanistic evidence regarding potency and to explore the potential to derive a new point of departure using all the available evidence.

Health Canada (2015b) recently proposed 3 categories for phthalate esters based on a structure-activity relationship analysis (SAR) covering 3 key events in the developmental effects of antiandrogenic phthalates (see Figure B2) - changes in foetal gene expression, foetal testosterone production and reduced anogenital distance. This approach is similar to the category approach as described in the Read-Across Assessment Framework (RAAF) scenario 4, where the hypothesis is that different compounds have the same type of effects (ECHA 2015). The same hypothesis applies to RAAF scenario 2, where the approach for the read-across is analogue, i.e. effects of the target substance (DIBP) is expected to be quantitatively equal to a single source substance, DBP.

DIBP is structurally very similar to DBP. Indeed, DIBP is a branched isomer of DBP having the same molecular weight and physiochemical properties (see Table B6). The main toxic compound of phthalates is the mono ester form (Foster et al. 2001, 2006). The mono ester metabolite of DBP is the closest structural analogue to the mono ester metabolite of DIBP in the same category with both compounds affecting the 3 key events leading to developmental effects observed for antiandrogenic phthalates. Health Canada (2015b) grouped DIBP and DBP in the same subcategory, medium chain phthalate esters, with the longest carbon backbone length 3-7. Biomonitoring studies often assume that the molar FUE of DIBP is equal to that of DBP (e.g. UBA 2011; Fromme et al. 2013; Kasper-Sonnenberg et al. 2014).

The SAR analysis found DBP to be more potent than DIBP with regard to reducing expression of 5 genes in the steroidogenic pathway (SR-B1, StAR, Cyp11a, 3bHSD, Cyp17a1) (Hannas et al. 2011, 2012, Lehmann et al. 2004). DIBP was found to be slightly more potent than DBP in reducing foetal testosterone levels (Hannas et al. 2011,12, Howdeshell et al. 2008), and DIBP and DBP being equipotent in reducing AGD (Saillenfait et al. 2008, Mylchreest et al. 2009)(all reviewed in Health Canada 2015b). Overall, DIBP and DBP affect similar mechanistic targets leading to similar adverse developmental effects as other phthalates within the medium chain phthalate esters group, and DBP mono ester is the closest structural analogue of DIBP mono ester. This makes DBP the most relevant phthalate for read across for DIBP.

Table B6 Comparison of structure and physicochemical properties of DIBP and DBP

Properties	**Diisobutyl phthalate (DIBP)**	**Dibutyl phthalate (DBP)**
Structure		
MW	278.34 g/mol	278.34 g/mol
Vapour pressure	0.01 Pa at 20°C	0.01 Pa at 25°C
Water solubility	20 mg/L at 20°C	10 mg/L at 20°C
Partition coefficient ($logP_{ow}$)	4.11	4.57

When comparing effects on foetal testosterone production and gene expression, DIBP and DBP appear to be equally potent. When expanding the comparison to also cover other reproductive developmental effects for these phthalates, Saillenfait et al. (2008) investigated effects on anogenital distance, nipple retention, reproductive organ weights and reproductive tract malformations (hypospadias, exposed os penis, cleft prepuce and cryptorchidism) and puberty onset for DIBP (125, 250, 500 and 625 mg/kg bw/d) using DBP (on dose at 500 mg/kg bw/d) as a positive control. The effects of DIBP seen at 500 mg/kg bw/day and 625 mg/kg bw/day were comparable to the effects seen with 500 mg/kg bw/day of DBP (Saillenfait et al., 2008). The potency difference between DIBP and DBP for these reproductive developmental endpoints thus appears to be minor. Effects on neonatal AGD found in Saillenfait et al. 2008 is supported by others. Comparison of developmental effects on male reproduction in rodent studies for DIBP and DBP is provided in Table B7.

Table B7 Comparison of developmental effects on male reproduction in rodent studies on for DIBP with DBP

Phthalate	Protocol (species, duration: doses in mg/kg bw/day)	Effect LOAEL/NOAEL (mg/kg bw/day)	Comment	Reference
AGD				
DIBP	Pregnant rats, gavage GD 12-21: 0, 125, 250, 500, 650	LOAEL 250 NOAEL 125 (some effects on AGD, not statistically significant).	Overall, effects on male AGD appear around 100 mg/kg bw/d of DBP (though only examined in one study) and around 125 mg/kg bw/d of DIBP (only one study with several doses available, others find sign effects at 600 mg/kg with this dose the only one tested) Health Canada calculated BMDL10 values (10% decrease in AGD from controls) of 204 and 208 mg/kg bw/d for DIBP and DBP, respectively (Health Canada 2015b)	Saillenfait et al. 2008 Saillenfait et al. 2008 included DBP as a positive control, see comparisons from this study below
DBP	GD 13-21: 100, 500	LOAEL 100		Martino-andrade et al. 2009
	GD12-21: 100, 250, 500	LOAEL 250 NOAEL 100		Mylchreest et al. 1999; Zhang et al. 2004
	GD12/13-20/21: 100, 500	LOAEL 500 NOAEL 100		Barlow et al. 2004; Mylchreest et al. 2000; Johnson et al. 2011
↓ foetal testosterone				
DIBP	Pregnant rats, gavage GD 8-18: 0, 100, 300, 600, 900	LOAEL 300 NOAEL 100	When comparing effects on foetal testosterone production, DIBP and DBP appear to be equally potent. Howdeshell et al. calculated derived ED50 values for DIBP and DBP of 466 and 399	Hannas et al. 2011, 2012 Similar effects in Howdeshell et al. 2008
	Pregnant Harlan SD rats, gavage GD 14-18: 0, 750	ED50 288 (95% CI 248-335)		Furr et al. 2014

DBP	Pregnant rats, gavage GD 8-18: 0, 100, 300, 600, 900	LOAEL 300 NOAEL 100	mg/kg/day, respectively (for DEHP 383). Comparing with the potency from Hannas et al., the derived ED50 value for DIBP was 305 mg/kg/day, i.e. lower than for DEHP (383 mg/kg/day). Furr et al. 2014 showed differences in species sensitivity and slightly lower ED50s than calculated by Hannas et al. and Howdeshell et al. Based on the ED50 values the substances are estimated to be roughly of equal potency.	Howdeshell et al. 2008
	Pregnant rats, gavage GD 14-18: 0, 750	ED50 (Harlan SD rats) 158 (95% CI 101-248) ED50 (CR SD) 337 (95% CI 250-454)		Furr et al. 2014
Gene expression related to steroid biosynthesis pathway				
DIBP	Pregnant rats, gavage GD 14-18: 0, 100, 300, 600, 900	LOAEL: 300 (cyp11a, 3bhsd, cyp17a1, sr-b1, star) NOAEL: 100	Sporadic reductions in gene expression for 3bhsd and sr-b1 at 0.1 and 1 mg/kg bw/day, but not at 1 mg/kg bw/day. Both studies have low power due to few animals per dose group.	Hannas et al. 2012
DBP	Pregnant rats, gavage GD 12-19: 0, 0.1, 1, 10, 50, 100, 500	LOAEL: 50 (sr-b1cyp11a, star, 3bhsd), 500 (cyp17a1)		Lehmann et al. 2004
Nipple retention				
DIBP			Not sign at any doses (100-625 mg/kg)	Saillenfait et al. 2008
DBP	Pregnant rats, gavage GD 12-21: 0,5, 5, 50, 100, 500 Pregnant rats, gavage GD 12-21: 100,500	LOAEL 100 NOAEL 50 LOAEL 100	Mylchreest et al. 1999 and Martino-Andrade et al. 2009 do not find sign. effects on NR up to 500 mg/kg. Lee et al. 2004: LOAEL 712 mg/kg NOAEL 150 mg/kg	Mylchreest et al. 2000 Barlow et al. 2004
Mammary gland development				
DIBP			No studies available investigating mammary gland development after DIBP exposure	
DBP	Pregnant rats, dietary concentrations GD 15-PND21: 0, 20, 200, 2000, 10000	LOAEL 2	In the mammary glands, dilatation of alveolar buds and/or ducts was seen in male offspring from 20 mg/kg with low incidence but not achieving statistical significance in any group. In female offspring, hypoplasia	Lee et al. 2004; EFSA opinion 2005

			of the alveolar buds of the mammary glands was observed in animals from 20 mg/kg with a statistically significant increase at 20, 200, 2000 and 10000 mg/kg. Significant increases in vacuolar degeneration in the mammary glands of males was present from 20 mg/kg but with similar incidence and qualitative gradation of change across the dose groups"	
Other reproductive effects				
DIBP with DBP as positive control	Pregnant rats, gavage GD 12-21: 0, 125, 250, 500, 650 DBP dose: 500		The effects (reproductive tract malformations, AGD, nipple retention, reproductive organ weights, puberty onset) seen with 500 mg/kg bw/d of DIBP were comparable or slightly less marked than the effects seen with 500 mg/kg bw/d of DBP, whereas the effects seen with 625 mg/kg bw/d were comparable or more marked than the effects seen with 500 mg/kg bw/d of DBP. The potency difference between DIBP and DBP thus appears to be minor. If using current data for extrapolation from DIBP LOAEL at high doses to DBP LOAEL at lower doses, the potency difference between effects for DIBP and DBP thus appear to be around 25%. Prepubertal spermatogenesis is not included in this study, but educed spermatocyte development for DIBP in adult rats was associated with tubular degeneration, occurring in all DIBP treated groups. Their severity increased with the dose. These effects are not reported for DBP.	Saillenfait et al. 2008

Based on a structure-activity relationship analysis (SAR) covering 3 key events in the developmental effects of antiandrogenic phthalates (see Figure B2) - changes in foetal gene expression, foetal testosterone production and reduced anogenital distance - Health Canada (Health Canada 2015b) grouped DIBP and DBP in the same subcategory, medium chain

phthalate esters, with the longest carbon backbone length 3-7. The SAR found DBP to be more potent than DIBP with regard to reducing expression of Cyp11a, a gene in the steroidogenic pathway. DIBP to be more potent than DBP in reducing foetal testosterone levels, and DIBP and DBP being equipotent in reducing AGD (Health Canada 2015b). Overall, this conclusion concur with the proposed relevance of read across between DIBP and DBP.

Conclusion

The current data suggests that DIBP has similar potency to DBP for effects on decreasing foetal testosterone production which is central for the effects observed for antiandrogenic phthalates, and thus that the LOAEL of 125 mg/kg bw/day used previously as the starting point for DNEL derivation for DIBP does not seem to appropriately reflect this potency. In comparison, a markedly lower LOAEL for DBP (factor of 62 lower) was derived from a study showing reduced spermatocyte development in prepubertal rats and mammary gland changes in adult male rats (Lee et al., 2004, described above). No studies on prepubertal spermatogenesis or adult mammary histology have been published for DIBP.

However, a possible potency difference between DIBP and DBP has been observed, based on the available data from Saillenfait et al. A very rough estimate based on the available data indicates that a 25% higher dose of DIBP would be required to see the same reproductive adverse effects as with DBP (anogenital distance, nipple retention, reproductive organ weights and reproductive tract malformations and puberty onset) . This estimate is based on comparable effects seen with 500 mg/kg bw/day of DBP and 625 mg/kg bw/day of DIBP in the study by Saillenfait et al. (2008). Whether the equipotency for decreasing foetal testosterone production and the observed approximately 25% lower potency for the dose-response function of DIBP for developmental effects will still follow the same path as DBP at lower doses is an assumption and introduces a degree of uncertainty to the applied read-across.

If this potency difference of 25% between DBP and DIBP is extrapolated from the high dose area to the lower dose area, a new LOAEL for DIBP would be 25% higher than the current LOAEL of 2 mg/kg bw/day for DBP, leading to a LOAEL for DIBP of 2.5 mg/kg bw/day.

A **LOAEL for DIBP of 2.5 mg/kg bw/day** is used as the starting point for DNEL derivation.

B.4.2.4. BBP

Tyl et al. (2004) found a NOAEL of 50 mg/kg bw/day due to reduced anogenital distance in male rats exposed to the next dose of 250 mg BBP/kg bw/day. In a study by Nagao et al. (2000), a NOAEL of 100 mg/kg bw/day was determined due to reproductive organ effects and reduction of absolute anogenital distance in males at the next dose of 500 mg/kg bw/day. The EU Risk assessment report from 2007 uses both NOAELs, i.e. a NOAEL of 50 for developmental effects in the study by Tyl et al. (2004) and a NOAEL of 100 for effects on fertility and reproductive organs in the study by Nagao et al. A **NOAEL of 50 mg/kg bw/day** is selected here, as this level is used for developmental effects in the EU risk assessment, and this is based on an anti-androgenic endpoint. Both studies were two-generation studies, one with dietary exposure (Tyl et al. 2004), one with gavage administration (Nagao et al. 2000). The registration dossier includes a two-generation study not reported in the EU RAR and performed by Aso et al. (2005). This study revealed decreasing AGD in male offspring in all doses from 100 mg/kg bw/day of BBP and no NOAEL was determined. Ahmad et al. (2014) found reduced

reproductive organ weights and altered sperm counts and motility at 100 mg/kg bw/day in adult male rats exposed in utero. The corresponding NOAEL for these endpoints was 20 mg/kg bw/day. Combining the LOAEL of 100 mg/kg from studies by Aso et al. (2005) and Ahmad et al. (2014) with the results of the study by Tyl et al. (2004) and Nagao et al. (2000), an overall **NOAEL of 50 mg/kg bw/day** can be determined together with a LOAEL of 100 mg/kg bw/day.

Details of the key studies

The study by Aso et al. (2005), is a two-generation study with exposure of male and female Crj:CD Sprague Dawley IGS rats to BBP by gavage at doses of 0, 100, 200, or 400 mg/kg bw/day. BBP was administered starting at 5 weeks of age for the F0 parents and 3 weeks of age for the F1 parents for 10 weeks prior to mating, and continued through weaning. Effects in male offspring are summarised here: F1 males had significantly lower body weights from 100 mg/kg bw/day and lowered epididymal weights and increased liver weight from 200 mg/kg bw/day, and reduced seminal vesicle weights and increased thyroid weights at 400 mg/kg bw/day. Aplasia and/or dysplasia and small epididymes and testes were seen at 400 mg/kg. Softening testes and histological changes, including atrophy of testicular seminiferous tubules, decreased spermatozoa and residual germ cells in epididymal lumina, were seen starting at 100 mg/kg bw/day, but reached statistical significance ($p<0.05$) only at 400 mg/kg bw/day. The F2 male offspring had decreased AGD at all doses when corrected by division with the cube root of body weights. The NOAEL for decreased AGD was therefore <100 mg/kg bw/day, the NOAEL for testicular effects 200 mg/kg bw/day.

In the study by Ahmad et al. (2014), rats were gavaged with 4, 20 or 100 mg/kg bw/day of BBP from GD 14 to parturition. Statistically significant reductions in epididymis weight, prostate weight, kidney weight, serum testosterone level, sperm count, sperm motility (%) and sperm abnormalities (%) in adulthood were seen at 100 mg/kg bw/day of BBP. Measurement of anogenital distance at PND 5 and 25 did not show any statistically significant effect of BBP. As no effects on anogenital distance were seen with DBP (up to 50 mg/kg bw/day) in the same study, the sensitivity of the AGD measurement in this study may be questioned. Pup weight was reduced at PND 1 and 21 at all three doses of BBP. At 20 mg/kg of BBP a reduction in adult body weight was seen. No information on the number of litters or numbers of examined males were reported in the paper, but these results may be used as supporting information for the NOAEL/LOAEL determination based on other studies.

The following is a summary of the two studies by Nagao et al (2000) and Tyl et al. (2004) from the EU RAR (citation in italics):

> *"Regarding toxicity to reproduction, fertility as well as developmental studies are available. When taking the available data base into account a NOAEL at 100 mg/kg bw/day for effects on the reproductive organs/fertility from a 2-generation study in rats is used in the risk assessment (Nagao et al., 2000). The NOAEL is based on atrophy of the testis, epididymis, and seminal vesicle, and reduced reproductive organ weights at 10 or 18 weeks of age in the F1 generation at 500 mg/kg bw/day. In this two-generation study BBP was administered by gavage (0, 20, 100 and 500 mg/kg bw/day) to Sprague-Dawley rats. The results were as following; a significant reduction in fetal body weight was reported at 100 and 500 mg/kg bw/day on pnd 0. Furthermore, in male offspring (preweanling rats) a reduction in AGD (absolute), testis weight, epididymis weight, decreased FSH level and number of*

spermatogonia and spermatocytes in the seminiferous tubules was reported at 500 mg/kg bw/day. In postweanling rats at 500 mg/kg bw/day a decreased body, testis and epididymis weight was reported. Furthermore, at 500 mg/kg bw/day, a delay in preputial separation in males, decreased testosterone and LH levels and increased incidence of testicular atrophy with decreased number of germ cells in the seminiferous tubules and decreased number of sperm in the epididymis was reported. In another recent 2-generation study (Tyl et al., 2004) significantly reduced mating and fertility indices were reported in F1 parents to make F2 offspring at 750 mg/kg bw/day. In the same study a significantly reduced relative and absolute paired ovaries and uterus weight was reported in F0 females. In adult F1 males a significant increase in reproductive tract malformations was reported (53.33% compared to 0% in controls). No increases in reproductive tract malformations were reported in females. Systemic toxicity reported at 750 mg/kg bw/day was limited to organ weight changes (liver, kidney) in males and females and histopathological lesions graded as minimal in females. The NOAEL for fertility was 250 mg/kg bw/day from this study. For development a NOAEL at 50 mg/kg bw/day for offspring is used in the risk assessment (Tyl et al., 2004). This NOAEL value is based on a dose-related significant reduction in absolute and adjusted AGD in both F1 and F2 offspring from 250 mg/kg bw/day. At the next higher dose, 750 mg/kg bw/day a significant increase in F1 and F2 male pups with one or more nipples and/or areolae was reported. At weanling in F1 and F2 offspring a significant reduction in testis weight was reported. At post natal day 21 necropsies the percentage of males with reproductive tract malformations (RTM) were significantly increased in the F1 and F2 offsprings, and at adult necropsies the percentage of males with RTM were significantly increased in the F1 offspring (F2 offspring was not evaluated as adults). In F1 parental male a significant decrease in the testis, epididymis, prostate and seminal vesicle weight was reported (not evaluated in the F2 generation). The NOAEL for maternal toxicity was 750 mg/kg bw/day and was based on organ weight changes (liver and kidney) and histopathological lesions graded as minimal in the liver at 750 mg/kg bw/day. In this 2-generation study BBP was administered in the feed at doses of 0, 750, 3,750 and 11,250 ppm corresponding to approximately 0, 50, 250 and 750 mg/kg bw/day" (EU RAR for BBP page 213-214).

Howdeshell et al. (2008), described that BBP decreased foetal testosterone production in rats at doses from 300 mg/kg bw/day (NOAEL 100 mg/kg bw/day). In this study, pregnant Sprague-Dawley rats were exposed to 0, 100, 300, 600, or 900 mg/kg bw/day of BBP from GD 8 to 18 by gavage in corn oil (n=4 to 9 dams per group). Testicular testosterone production ex vivo was assessed by incubation of testes of 18 day old foetuses for 3 hours and testosterone measurement in the media. Dose-related decreases in testosterone production was seen for BBP and the other tested phthalates (DIBP, DBP, and DEHP) from 300 mg/kg bw/day and above, and for DPP (dipentyl phthalate) from 100 mg/kg bw/day.

Other studies

Other studies on BBP were included in the NOAEL selection but were not considered critical. These studies include the following reproductive and developmental studies described in detail in the EU RAR: Agrawal et al. (1985), Piersma et al. (2000), NTP (1997), Piersma et al. (1995), Monsanto (1993), Lake et al. (1978), Hammond et al. (1987), NTP (1990), Parks et al. (1999), Saillenfait et al. (2003), Gray et al. (2000), Ema et al. (1990, 1991, 1992 (a, b, c)), 1993, 1994, 1998, 2002), Sharpe et al. (1995), Ashby et al. (1997), TNO (1998 (a, b)), Bayer (1998) (see EU RAR for precise references). Additionally, the registration report for BBP includes a study by Field (1989), which was not considered critical as doses were above the

LOAEL of 250 from the study by Tyl et al. (2004): Field (1989). A number of reproductive/developmental studies have been published after the publications listed in the EU RAR (after 2004). The following studies published after 2004 included only doses above the overall LOAEL of 100 mg/kg bw/day and were not considered further for NOAEL determination: Hotchkiss et al. (2004), Liu et al. (2005), Martin et al. (2008), Rider et al. (2009), Kwack et al. (2009), and Moral et al. (2011). In a study by Min et al., 2014 effects on neurotransmitter levels were observed at 50 mg/kg bw/day, but as this effect is not clearly related to an anti-androgenic mode of action this LOAEL is not applied for DNEL determination here. Reduction of body and uterus weights at 20 and 200 mg/kg bw/day were seen in a 20-day female pubertal study by Ahmad et al. (2013). In a uterotrophic study, oral exposure of 21-day old rats to 20 or 200 mg/kg BBP for three days did not increase uterus or ovary weights, but rather reduced uterus weight at 200 mg/kg. These studies did not contribute with information relevant to setting a NOAEL for anti-androgenic effects of BBP.

B.4.2.5. Summary of starting points for the four phthalates

In the EU risk assessment reports for three phthalates (DEHP, DBP and BBP), the reproductive and developmental effects are considered critical, i.e. these effects are seen at the lowest dose levels, and Table B8 compares NOAELs selected here to overall NOAELs concluded in EU Risk assessment reports.

Table B8 Selected starting points for DNEL setting in the current assessment compared with selected starting points from EU risk assessment reports (EU RAR).

mg/kg bw/day	**N(L)OAEL for this assessment**	**Endpoint for NOAEL selection in this assessment**	**NOAEL and endpoint for NOAEL selection in EU RAR**	
DEHP	**4.8**	Small male reproductive organs (testes/epididymes/ seminal vesicles) and minimal testis atrophy	4.8	Testis toxicity, developmental toxicity
DBP	**2 (LOAEL)**	Reduced spermatocyte development PND 21, mammary gland changes (vacuolar degeneration and alveolar atrophy) in adult male offspring	52 (LOAEL)	Embryotoxicity
DIBP	**2.5 (LOAEL for DBP, adding 25% for DNEL derivation)**	-	*	*
BBP	**50**	Reduced anogenital distance	50/100	↓ AGD, reproductive organ weight

* No EU Risk assessment report is available for DIBP

Table B8 shows that the largest difference between NOAEL selection in the EU RAR and in this report is seen for DBP. DBP appears to be a more potent anti-androgen than DEHP, but the DNEL for DBP is based on a study on prepubertal testis development and male mammary gland changes during development and in adulthood in the study by Lee et al. (2004), and it has not been possible to find any literature on the effects of developmental phthalate exposure on male mammary glands for other phthalates. Thus, NOAELs/LOAELs for mammary gland effects have not been determined for any of the other phthalates listed in Table B4. It cannot

be excluded that if this effect was studied on the other phthalates DNELs for the other phthalates could have been lower. Further, it should be noted that the study by Lee et al. (2004), was not included in the EU RAR for DBP from 2003, but the study is considered in an EFSA opinion from 2005. It is also noted that the EFSA opinion used an uncertainty factor of 200, whereas 300 was used for setting the DNEL in the current report. This means that the applied DNEL is lower than the EFSA TDI. If instead the EFSA TDI for DBP had been used, the risk calculations for DBP would have resulted in 1/3 lower values.

As mentioned in Section B.1.4, the use of DNELs in the HI method attempts to account for potency differences of the four phthalates. However, there is some uncertainty in the determination of DNELs, as NOAELs are very dependent on dose selection, endpoint selection and the sensitivity of the critical endpoints. For each compound there are differences in study designs, doses and selected endpoints, and there are also likely to be inter-laboratory differences in the sensitivity of the methods to detect effects on certain endpoints. It is therefore not possible to use dose-response curves for (different) reproductive endpoints to compare potency of these phthalates. One lab has examined all four phthalates and compared dose-response curves for the same endpoint, i.e. inhibition of foetal testosterone production (Howdeshell et al., 2008). As evident from Table B8, this study reveals the same NOAEL for the four phthalates, and closer examination of their data show comparable dose-response curves for all four phthalates with respect to inhibition of foetal testosterone production (Howdeshell et al., 2008). However, as stated in the introduction to the justification for grouping (B.1.4.1), the focus is on reproductive effects and that the data on inhibition of testosterone is primarily to be used as underlying knowledge documenting similar modes of action.

B.4.2.6. Epidemiological studies

The four phthalates DEHP, DBP, DIBP and BBP are all classified as toxic to reproduction in category 1B. These four phthalates are considered to have endocrine disrupting effects with similar mode of action. As in the description of experimental animal studies, the focus is here on developmental effects on male reproduction, although associations have been found for a range of endpoints, including direct effects on male fertility, and only studies on developmental exposures are included. For clarity, studies on associations between adult exposure and effects and studies on effects on female reproduction have not been included.

Epidemiological studies on the relationship between maternal phthalate exposure and reproductive development in offspring may be useful to determine the human exposure levels associated with effects. As most humans in the epidemiological studies are exposed not only to phthalates, but also to other substances with similar mode of action, it is not possible to determine a human “no effect level” and neither is it possible identify effects that are “only” caused by one substance. Most epidemiological studies describe associations between urinary levels of phthalate metabolites and adverse outcomes (e.g. AGD, cryptorchidism), but do not calculate external exposure levels. Data to evaluate human exposure levels associated with effects are not currently available.

The endocrine disrupting effects of the four phthalates suspected to be relevant in humans are congenital malformations of the male reproductive organs, reduced semen quality, reduced

male reproductive hormone levels, and changes in pubertal timing including changes in breast development. Based on the current knowledge of the biology of testicular cancer and breast cancer as well as of the shared risk factors of these cancers and some of the above mentioned effects, it has also been speculated whether prenatal exposure to phthalates may play a role in the increasing incidence levels of these two hormone dependent cancers.

Regarding congenital malformations of the male genitalia

Although prenatal exposure to DBP and DEHP in relatively high doses in rat studies results in cryptorchidism and hypospadias, no direct link between pre- and perinatal phthalate exposure and these malformations of the male genitalia has been proven in humans. There are, however, some circumstantial findings that indicate that phthalate exposure could play a role. Cryptorchidism and hypospadias are both conditions caused by insufficient testosterone action during respectively descent of the testes and the formation of the penis. In a case-control study of phthalate levels in breast milk samples (n=130) from mothers of cryptorchid and healthy boys (62 cryptorchid/68 healthy boys) from a large prospective Danish-Finnish mother-child cohort, no association between phthalate levels and cryptorchidism could be seen. However, an association between phthalate levels in maternal milk and the male reproductive hormone profiles in the male infants was observed, indicating that testicular function of the more exposed boys were affected (Main et al., 2006). The findings are in line with another recent human study showing decreased virilisation in infant boys exposed to phthalates prenatally. The perinatal exposure of the boys was based on maternal urinary phthalate levels during mid-pregnancy and the effects were manifested as a significantly decreased anogenital distance (AGD) in the most exposed boys compared to the least exposed boys (Swan et al., 2005). Also in this study, no significant direct association between prenatal exposure to phthalates and cryptorchidism was found. Still, however, the boys with shorter AGD were more likely to be cryptorchid than the boys with longer AGD. In a more recent and larger study, Swan et al., 2015, examined concentrations of DEHP metabolites in human first trimester maternal urine and compared with anogenital distance in newborn males and females. Increasing levels of certain DEHP metabolites were associated with decreasing AGD in boys, but not in girls.

Jensen et al., 2015, examined the association of DINP and DEHP metabolites with testosterone levels, cryptorchidism and hypospadias in a case-control study. DINP and DEHP metabolites, steroid hormones and the Leydig cell product Insulin-like factor 3 (Insl3) were measured in amniotic fluid samples collected in second trimester of pregnancy in the period 1980-1996. For a DEHP metabolite, a possible interference with human male foetal gonadal function was identified. The authors highlight the advantage of investigating exposures close to the sensitive window important for foetal masculinisation, and the study of steroid hormones and insl-3 in amniotic fluid is also considered a relevant matrix. The measured insl-3 is expected to originate from the male foetus, whereas the measured testosterone may also originate from the adrenals, and the authors speculate that the differential associations between metabolite levels reflect differential actions of phthalate load upon the foetal testis and the foetal adrenal gland. However, the findings are not consistent for DINP and DEHP, and no firm conclusions can be made regarding the associations between exposure to DINP or DEHP and possible effect on hypospadias/cryptorchidism in humans.

Axelsson et al., 2015a, performed a human epidemiological study comparing maternal serum levels of DEHP and DINP metabolites with testicular size, semen quality and reproductive

hormones in 112 adolescent sons. Men were recruited in 2008-2010 at military health board examination or through announcements. Corresponding maternal samples were collected from a Swedish biobank of samples obtained from the 6th to the 35th week of pregnancy (mean 12th week, majority sampled between 8 and 14 weeks). Men in the highest exposure tertile of a DEHP metabolite had lower semen volume than men in the lowest tertile, whereas men in the highest exposure tertile of a DINP metabolite had lower total testicular volume, higher levels of FSH and lower semen volume. It is concluded by the authors that prenatal levels of DEHP and DINP metabolites seemed negatively associated with reproductive function of adolescent men.

Regarding semen quality

Large internationally coordinated studies on semen quality of men from the general population in different European countries including France, Finland, UK (Scotland), Estonia and Denmark found large differences between the countries; especially in Denmark a large proportion of the men had semen quality in the sub-fertile range while Finnish men seem to have significantly better semen quality. Although genetic differences cannot entirely be ruled out as causes, the considerable differences indicate different exposures or lifestyle differences in the four populations (Jørgensen et al., 2001 and Jørgensen et al., 2002). However, in the aforementioned study on phthalate levels in breast milk from a Danish-Finnish mother-child cohort (Main et al., 2006), the levels of phthalate metabolites measured were actually higher in Finnish breast milk than in Danish, while levels of other endocrine disrupting compounds including some dioxins, PCBs, hexachlorobenzene and dieldrin were significantly higher in the Danish samples. This seems to imply that Danes are not more exposed to phthalates than Finns and consequently that phthalate exposure is not likely to play a major role for the poorer semen quality of Danish men. However, although current exposures may have an effect on men's semen quality, a man's general potential for producing sperm is dependent on his testicular capacity, which in general is determined already during the foetal development of the testes. According to the Testicular Dysgenesis Syndrome (TDS) hypothesis, foetal exposure may play an important role for the testicular function in adult life. Recently, this link between foetal development – and more specifically foetal testosterone activity – and sperm quality in adult life was further corroborated by a new study showing a significant positive correlation between a man's AGD and his semen quality (Mendiola et al., 2011). We do not know the phthalate exposure in Denmark nor in Finland at the time when the men who delivered semen for the semen quality studies were born 20-35 years ago as our measurements of phthalates in Danish and Finnish breast milk were done on samples collected only 10-12 years ago. The latest studies in Finland seem to indicate that the semen quality of young Finnish men has continued to decrease and now approach the levels seen in the Danish male population (Jørgensen et al., 2011).

Regarding pubertal timing

A substantial decline in the age of onset of puberty in Danish girls was observed over a 15-year period (from 1991-93 to 2006-08) and manifested as earlier age at breast development (Aksglaede et al., 2009). A study from Puerto Rico found that the phthalate concentration in samples from girls with premature thelarche (breast development) was significantly higher than in samples from age-matched controls (Colon et al., 2000). However, it was debated whether the data could be flawed by methodological problems as the phthalate levels measured was much higher than measured in e.g. the USA. Subsequently a small case-control study on girls with precocious puberty found no difference in urinary phthalate levels of

healthy girls and girls with precocious puberty (Lomenick et al.,2010). The contradicting findings may lay in the differences in the inclusion criteria between the two studies. In the Puerto Rico study the inclusion as case was based on the presence of premature thelarche only, while in the USA study the inclusion criteria for cases was presence of central precocious puberty. Premature thelarche is sometimes seen isolated without activation of the pituitary-gonadal hormone axis. In this respect it is interesting that although a one year decrease in age at breast development was observed in the Danish study the girls did not have significantly different endogenous hormone levels compared to age-match girls studied 15 years earlier. Thus, it seems to be breast development but not the activation of the pituitary-gonadal hormone axis that seems to be advanced.

In rat studies prenatal phthalate exposure is associated with delayed – not advanced – puberty (estimated as the age of vaginal opening) in female offspring, while exposure of prepubertal rats has been shown to advance the age of vaginal opening (Ma et al., 2006) so the effects of phthalate exposure on puberty development may depend on the timing of exposure. Humans are the only mammal that has permanently protuberant breasts, even when not lactating and as such breast development during puberty cannot directly be compared with puberty in rats. However, in utero exposure to phthalates has been shown to induce modifications in the morphology and the gene expression profile of the mammary gland that persist postnatally and potentially may make the breast tissue more susceptible to subsequent exposures (Moral et al., 2011).

Regarding testicular cancer

A significant increase in the incidence of testicular cancer has occurred over the last decades in many countries, indicating that environmental or lifestyle factors play a role in testicular cancer (Skakkebaek et al. 2016). Unfortunately no animal model for testicular cancer exists, as no laboratory animal species seems to develop testicular cancer. However, testicular cancer is linked to other problems with male reproduction: cryptorchidism or poor semen quality are risk factors for developing testicular cancer, which indicates that these conditions share aetiology. Thus, any exposure that is suspected to play a role in cryptorchidism or decreased testicular function may also be suspected to play a role in testicular cancer. Interestingly, Finnish men have not only experienced a recent adverse trend in semen quality as mentioned above; they are also experiencing a concurrent adverse trend in the incidence of testicular cancer (Jørgensen et al., 2011).

As shown above, epidemiological studies are generally associated with considerable uncertainties due to exposures to many substances, limited study populations, uncertainties in back-calculation of urine concentrations to estimated daily exposures, behaviour and societal background, genotype, smoker/non-smoker, diet, weight etc. It is therefore difficult to draw exact conclusions on these studies, but they could be seen as contributing to the overall picture.

B.4.2.7. Possible species differences

Marmoset

In Tomonari et al. (2006), no reproductive effects were seen in male marmosets (n=5-6 per dose group) exposed to DEHP by oral gavage at 100, 500 and 2500 mg/kg bw/day from 3 months of age until sexual maturity (18 months). Similarly, no reproductive effects were seen in Kurata et al. (1998), where male marmosets (n=4 per dose group) were administered 100, 500 and 2500 mg/kg bw/day DEHP during 12-15 months of age. These studies may be of limited human relevance because of the poor capacity of the marmoset to metabolise parent phthalates to their active metabolites (Rhodes et al. 1986).

In contrast, a neonatal study with marmosets of 4 days of age (5 co-twins and 4 non-twins, total n=14) treated 14 days with 500 mg/kg bw/day of MBP (monobutylphthalate, a metabolite of DBP) revealed increased Leydig cell volume which is suggestive of Leydig cell hyperplasia or hypertrophy in compensation of testosterone inhibition (Hallmark et al. 2007). A second study with a single dose of 500 mg/kg bw/day of MBP revealed suppressed (~50%) blood testosterone levels in male marmosets of 2-7 days old (n = 9, measurement 5h after dose) (Hallmark et al. 2007). On the other hand, no effects on germ cell number or differentiation were apparent in a study by McKinnell et al. (2009) where 4 day old co-twin marmosets (5 co-twins, n=10) were exposed to 500 mg/kg/day MBP neonatally during 14 days.

The above studies in the marmoset are neonatal studies and thus the exposure period did not cover gestation week 7-15 which is considered to be the critical programming window for male development in the marmoset (Mitchell et al. 2008; McKinnell et al. 2009). However, the susceptibility period might be challenged considering that treatment of male marmosets shortly after birth (when the testes are actively secreting testosterone) with 500 mg/kg/d MBP caused a reduction in blood testosterone, Leydig cell hyperplasia/hypertrophy, consistent with compensated Leydig cell steroidogenic suppression (Hallmark et al., 2007). Moreover, no effects on testicular morphology, reproductive tract, testosterone levels at birth, germ cell number or germ cell proliferation were observed in male offspring (n=6) of pregnant marmosets exposed to 500 mg/kg bw/day MBP during gestation (GD 49-105) (McKinnell et al. 2009). However, unusual clusters of undifferentiated germ cells were found in two of six males examined at birth. The significance of this observation is unclear. There is no current explanation for this foetus-neonate difference in susceptibility of the marmoset testis to MBP compared with rats (SCENIHR 2016).

Mouse

A study in foetal mice exposed to DBP did reveal changes in several immediate early genes[27] that are also targeted in the rat foetal testes following phthalate exposure, but no or minimal decrease of testosterone production by the foetal testis and expression of genes related to cholesterol homeostasis or steroidogenesis were observed (Gaido et al. 2007). However,

[27] Gaido et al. (2007) state the following regarding 'immediate early genes'*: "Both the fetal mouse and rat quickly respond to DBP with increased testicular expression of a number of immediate early genes including early growth response 1, early growth response 2, Ier3 (immediate early response 3), Nr4a1 (nuclear subfamily 4, group A, member 1; also known as Nur77), as well as many others. Nonreproductively toxic phthalates do not induce immediate early gene expression in the rat (Liu et al., 2005). Interestingly, immediate early gene induction also occurs in the prepubertal rat testis following DBP exposure (Lahousse et al., 2006)."*

exposure to DBP did affect germ cells, similarly to the rat (significantly increased seminiferous cord diameter, the number of multinucleated gonocytes per cord, and the number of nuclei per multinucleated gonocytes). These findings might give some evidence to suggest that germ cell effects may not result from reduced foetal testosterone. Changes in the seminiferous chords may be important to germ cell development and may be related to persistent effects on testes as seen in the testicular dysgenesis syndrome (Toppari et al. 2010).

In vitro

A time- and dose-dependent reduction of testosterone production in rat foetal testis was observed in a study with MEHP, but not with DEHP, in cultured foetal rat testes explant tissue (Chauvigné et al. 2009). MEHP also resulted in decreased gonocyte proliferation and increased apoptosis. Habert et al. (2014) also showed decreased foetal testosterone production in a similar experiment. However, Hallmark et al. (2007) did not observe reduced in vitro rat foetal testosterone production with MBP.

In human foetal testis no foetal testosterone production was observed following in vitro tissue culture with MEHP or MBP (Hallmark et al. 2007; Lambrot et al. 2009; Habert et al. 2014). In these in vitro studies human testis samples were from first or second trimester foetuses, but it is not clear whether these ages correspond to the sensitive window for phthalate exposure in rats (Lambrot et al. 2009; Hallmark et al. 2007). It is possible that the in vitro systems are unsuitable for detecting the reduction of foetal testosterone by exposure to phthalates (Habert et al. 2014), For example, the in vitro systems cannot study long-term effects since the whole testis or sections are only viable for a few days, any possible indirect effects cannot be studied (e.g., via the hypothalamic–pituitary–gonadal axis), and the time between abortion and obtaining the foetal testes is variable (Habert et al. 2014). Heger et al. (2012) considered the inconsistent responses suggests that the in vitro model is a poor phthalate endocrine-disruption model.

Interestingly, an in vitro study on adult human testes has shown that exposure to DEHP and MEHP impaired testosterone production, and that the measured concentrations of phthalate metabolites in the incubated testes were as low as the phthalate metabolite levels measured in humans (Desdoits-Lethimonier et al. 2012).

In vitro studies on phthalate exposure of foetal testis tissue have been able to show comparable changes in germ cells using testes from rats, mice or humans (Lambrot et al. 2009; Lehraiki et al. 2009; Chauvigné et al. 2009; Habert et al. 2009). This clearly supports the possibility that reproductive effects of phthalates are relevant to humans.

Xenograft

Mitchell et al. (2012) inserted pieces of testicular tissue from foetal rats or humans under the skin of a castrated immune-compromised (nude) mouse (5 or 6 grafts per mouse). DBP exhibited a testosterone inhibiting effect when using rat foetal testis explants, but not when using human foetal testis explants (n=12). However, there were several differences in study design between the foetal rat testis graft and the foetal human testis graft study, including duration of grafting before exposure and timing of exposure and age of the testis explant at the time of exposure. The graft was retrieved after 6 weeks with human testis whereas 4 days with rat testes. In the foetal human graft study, mice were supplied with hCG to promote testosterone production, whereas no LH stimulation was necessary for the rat graft to produce

testosterone, and absolute testosterone levels therefore greatly differed in the two experimental setups (Mitchell et al. 2012). Furthermore, the study covered the period after the period that is thought to correspond to the male programming window (Mitchell et al. 2012).

Heger et al. (2012) inserted whole foetal testes from rats and mice or diced human foetal testis (from spontaneous abortions) into the renal subcapsular space of rat hosts. The human xenografts consisted of fragments from different parts of the foetal testis and were of variably size. Importantly, only 2 of the 26 donor testis were from within what is thought to be the masculinisation window (gestation weeks 8-14) and gestational ages were very variable (from 10 to 24 weeks). No significant changes in steroidogenic gene expression were observed in human foetal xenografts but expression was highly variable (testosterone levels were not measured). Rats, mice and human testis xenografts all exhibited induction of multinucleated germ cells (MNGs).

Spade et al. (2014) inserted diced human foetal testis (from spontaneous abortions) into the renal subcapsular space of adult male athymic nude mice (8 xenografts per host). The 7 donor testis dated from gestational weeks 16-22, thus after what is thought to be the masculinisation window. The mice were treated with human chorionic gonadotropin to ensure sufficient testosterone synthesis in the experimental model. A small sample (3 donor testes) received the DBP treatment. Abiraterone acetate, a very potent inhibitor of testosterone synthesis used as a drug to treat castrate-resistant prostate cancer, did significantly reduce serum testosterone (to levels below the castrated sham mice) whereas DBP did not. Seminal vehicle weight was lower in the DBP treatment group, but not significant. A near-significant increase in multinucleated germ cells (MNGs) was observed in the DBP treatment group.

Metabolism

The results in Kurata et al. (2012) suggest that DEHP following oral administration is rapidly metabolised to MEHP and its oxidative metabolites. Most of the metabolites were excreted as glucuronides in humans, whereas in rodents most of these metabolites were excreted as free forms. This observation is relevant as glucuronides are not biologically active and thus may lead to some species-differences in potency of DEHP toxicity.

Conclusion

There may be two different pathways through which phthalates effect the foetal testes, namely a) suppression of steroidogenic gene expression and suppressed testosterone secretion and b) increase in multinucleated gonocyte number (Health Canada 2015a; Johnson et al. 2012). A better understanding of molecular mechanisms responsible for the differences in sensitivity or resistance to developmental phthalate exposure and more insight into the molecular pathways controlling steroidogenesis in the human foetal testis is warranted (Johnson et al. 2012). In relation to risk assessment, Johnson et al. (2012) conclude that “molecular mechanistic understanding will be needed for risk assessment to progress beyond the default protective assumption that humans respond similarly to the most sensitive species“. Similarly, Albert and Jegou (2014) stated that “Caution before concluding that phthalates are innocuous in the human foetal testis should be kept until these issues have been addressed”. The variation among species in the window of susceptibility to the effects of phthalates and variation among species in timing of the development of the testis is an important consideration (Albert and Jegou 2014). According to Jobling et al. (2011) the available evidence suggests that only

exposure in an early sensitive window (e13.5 to e15.5) results in loss of foetal germ cells and only exposure in a late sensitive window (e19.5–e20.5) results in aggregation and multinucleated gonocytes, and that these effects may have separate causes. In this respect, it may be relevant that none of the xenograft studies used donor testes covering the window that is thought to correspond to the sensitive window in humans.

In contrast to the possible differences seen between species regarding sensitivity (or sensitive windows) to phthalate exposure, there appears to be similarities between rats, mice, marmosets and humans regarding influence of phthalate exposure on germ cell proliferation and differentiation. The study in foetal DBP-exposed mice showing no influence on steroidogenesis did reveal comparable changes in germ cells to those seen in foetal rats, i.e. increased seminiferous cord diameter, and increased numbers of multinucleated gonocytes (Gaido et al. 2007). Moreover, SCENIHR (2016) stated that *"exposure to DBP can alter the timing of differentiation of foetal germ cells in the rat (Jobling et al. 2011). Adverse changes in this process underlie the foetal origins of testicular germ cell cancer in humans (Rajpert-de Meyts 2006). In this regard, Yao et al. (2012) showed that MEHP promotes invasion and migration of an embryonal carcinoma cell line (derived from a testis germ cell cancer)."*

The human relevance of the reproductive effects of phthalates is supported by the similarities between the pattern of effects seen in rodent studies of perinatal phthalate exposure and the human "Testicular dysgenesis syndrome", which is a syndrome of reproductive abnormalities suggested to share the same origin, which may be an altered endocrine influence during development (Fisher et al. 2003). Furthermore, some epidemiology supports the theory of TDS (e.g., Bustamante-Montes et al. 2013; Suzuki et al. 2012; Huan et al. 2009; Main et al. 2006; Swan et al. 2005,2008; Skakkebæk et al. 2001,2016; Carlsen et al. 1992). This indicates that the pattern of phthalate effects in rats is conceivable also in humans.

It can be concluded that there are indications of species differences in metabolism and possibly in effects on foetal steroidogenesis, but the evidence is insufficient to deviate from the default assumption that humans are more sensitive than the test species (rat) (ECHA guidance Chapter R.8). The default assumption in DNEL derivation is that there is an interspecies differences of a factor 10 (4 for allometric scaling and 2.5 for remaining differences). There are indications that the neonatal period may be a sensitive window of exposure for humans. This conclusion is in line with the recent opinion by SCENIHR (2016)[28] and with the previous conclusions by RAC (ECHA 2012a).

B.4.3. Toxicity other than toxicity for reproduction

DEHP, DBP, DIBP and BBP are not classified for other human health endpoints than reproductive toxicity (see below).

Available data on the four phthalates DEHP, DBP, DIBP and BBP indicate that all four are of low acute toxicity and induce no skin and eye irritation or skin sensitisation.

[28] SCENIHR (2016) considered that the new studies are in line with the previously derived TDI value or not sufficiently robust to justify the derivation of a new TDI and thus supported the previously derived TDI value for DEHP (which used an interspecies differences of a factor 10).

B.4.3.1. Repeated dose toxicity

In repeated dose toxicity studies with experimental animals, the main organs affected besides reproductive organs (testis, in particular) are the liver (lowest NOAELs for non-peroxisome related effects for DEHP, DBP, and BBP 28.9, 152, and 151 mg/kg bw/day, respectively) and the kidneys (lowest NOAELS for DEHP, DBP and BBP 28.9, 152, and 151 mg/kg bw/day, respectively. For DIBP only few, rather old repeated dose toxicity studies are available.

B.4.3.2. Mutagenicity and carcinogenicity

Available data on mutagenicity result in the conclusion that the four phthalates are not mutagenic in *in vitro* tests.

Although the findings of Leydig cell tumours in one study in rats was not confirmed in four other lifetime studies or in multigeneration studies with DEHP, the EU RAR (2008) considered the induction of Leydig cell tumours in the rat relevant for humans. BBP tested negative for carcinogenicity in mice; in rats findings of mononuclear cell leukaemia, benign pancreas tumours and urinary bladder tumours were of doubtful significance. For DBP and DIBP, no carcinogenicity studies are available.

However, Leydig cell dysgenesis seem to occur in connection with other reproductive developmental effects in rodents (Barlow and Foster 2003, Barlow et al. 2004, Borch et al. 2006; Lee et al. 2004; Parks et al. 2000). Barlow et al. (2003) found Leydig cell aggregates and/or increased numbers of gonocytes and multinucleated gonocytes at post-natal day 3, 7 and 16 after prenatal exposure to 500 mg/kg bw/d DBP in addition to several testicular and epididymal lesions. In another study, Leydig cell lesions were observed later in life at PND 180, 370 and 540. Leydig cell adenomas were only observed in small numbers, and Leydig cell hyperplasias were noted at 500 mg/kg bw/d on PND 180, at 100 and 500 mg/kg bw/d on PND 370 and at 0, 100 and 500 mg/kg bw/d on PND 540. Not significant for any dose groups (Barlow et al. 2004). However, the authors described testicular dysgenesis as aberrant seminiferous tubules associated with proliferative Leydic cells was seen and significant in the high dose group at all 3 time points.

Leydig cell dysgenesis is possibly caused by decreased foetal testosterone. Unfortunately, no rodent model for testicular cancer exist, as no laboratory animal species seem to develop the same kind of testicular cancer as the prevalent form in humans, the testicular germ cell cancer (TGCC). However, as human incidence peaks at 20-45 yrs, suggesting an early onset of malignant process (fetal) and the testes showing comprised development and function of fetal Leydig and Sertoli cells (Skakkebaek et al. 2016),

In humans, testicular cancer, cryptorchidism, hypospadias and reduced semen quality are risk factors for each other at an individual level and at the population level, and occurring at increasing incidence rates. Increasing evidence also link reduced AGD in humans to this group of risk factors. Further, these effects are often seen in association with dysgenesis in parts of the testicular tissue, including clusters of incompletely differentiated Sertoli cells and clustered Leydig cells. In animal studies, unusual clusters of undifferentiated germ cells and multinucleated germ cells have been observed across species (Skaekkebaek et al. 2016).

Although testicular germ cell cancer observed in humans is not seen in animal studies, the other effects on male reproductive development as well as the dysgenesis of Sertoli, Leydig

and germ cells in the testes suggest that the increasing incidences of testicular germ cell cancer in humans is part of the TDS of common fetal origin in both experimental animals and humans. The role of Leydig cell dysgenesis and multinucleated gonocytes observed in rodents in relation to the testicular dysgenesis syndrome (TDS) and testicular cancer may be better understood in the future.

In rodent carcinogenicity studies, DEHP induced liver tumours in both rats and mice.

Regarding the role of peroxisome proliferation in liver carcinogenesis with DEHP, Klaunig et al. (2003) published an extensive in-depth assessment of peroxisome proliferators and the role of PPARα in the tumorigenesis. According to the review by Klaunig et al. (2003), activation of PPARα is causally related to carcinogenesis in rodent liver. One of the arguments is that PPARα-null mice are refractory to the potent agonist WY 14643. This finding lead to the decision by IARC in 2000 to review the classification of DEHP from 'possibly carcinogenic to humans' to 'not classifiable as to its carcinogenicity to humans'. As routine screening showed that even compounds with a very high specificity to PPARα or PPARγ produced significant peroxisome proliferation, it was pointed out that the hypothesis of PPARα mediated carcinogenesis would be strengthened with further bioassays using other known peroxisome proliferators (Klaunig et al. 2003). This suggestion materialised with the publications by Ito et al. (2007) and Ito and Nakajima (2008). The authors studied DEHP in PPARα null mice and unexpectedly found that the incidence of liver tumours was higher (26%) in PPARα-null mice than in wildtype mice (10%) exposed dietary to 0.05% DEHP over two years (corresponding intake in mg/kg bw/day was not reported). The authors suggested the involvement of a non-PPARα pathway in rodent carcinogenesis. They hypothesised that oxidative stress and subsequent induction of inflammation might result in tumorigenesis in PPARα-null mice, and/or expression of protooncogenes. Their hypothesis was supported by dose dependant increased 8-OhdG and NFκB levels in both knock-out and wild type. Furthermore the protooncogene c-jun-mRNA was induced.

Rusyn and Corton (2012) reviewed the evidence that gathered the decade following the publication of Klaunig et al. (2003). The authors considered that activation of PPARα remains an important mode of action for DEHP carcinogenicity, but were of the opinion that the data suggest that multiple pathways in several cell types contribute to cancer in rats and mice. These were summed up as follows: *"(i) rapid metabolism of the parental compound to primary and secondary bioactive metabolites that are readily absorbed and distributed throughout the body; (ii) receptor-independent activation of hepatic macrophages and production of oxidants; (iii) activation of PPARa in hepatocytes and sustained increases in expression of peroxisomal and non-peroxisomal metabolism-related genes; (iv) enlargement of many hepatocellular organelles (peroxisomes, mitochondria, etc.); (v) rapid, but transient increases in cell proliferation and decreases in apoptosis; (vi) sustained hepatomegaly; (vii) chronic low-level oxidative stress and accumulation of DNA damage; (viii) selective clonal expansion of initiated cells; (ix) appearance of pre-neoplastic nodules; (x) development of adenomas and carcinomas.".* The authors furthermore concluded that the overall body of evidence on human cancer hazard of DEHP remains inconclusive.

In the light of the new evidence IARC has in 2011 reviewed the classification of DEHP and changed their conclusion back to 'possibly carcinogenic to humans (Group 2B)' (Grosse et al. 2011; IARC 2012).

B.4.3.3. Immunotoxicity

A significant increase in the prevalence of atopic allergy and asthma has been observed since the Second World War (Kimber and Dearman 2010; Bornehag and Nanberg 2010). It has been proposed that exposure to chemicals, or to specific classes of chemicals such as phthalates, might be a contributing element to the prevalence of these immunological diseases. Phthalates appear not to be sensitisers as such, but instead they are suspected to be adjuvants. In the context of hypersensitisation, an adjuvant means a substance or material which can impact the vigour (and/or quality) of a specific immune response without itself being an antigen[29].

Epidemiological studies

Robinson and Miller (2015); Braun et al. (2013); Kimber and Dearman (2010); Bornehag and Nanberg (2010); and Jaakkola and Knight (2008) have reviewed the human associations between exposure to phthalates and immunological outcomes.

Robinson and Miller (2015) reviewed 11 epidemiological studies and concluded that the studies confirm an association between exposure to DEHP and increased risk of developing a wheeze, asthma and allergy. Such an association for other phthalates was evaluated to be less clear. A critical window of susceptibility has not been established, although it would appear that the prenatal period would be especially important. The latter conclusion would appear to be based on the findings from Whyatt et al. (2014) and Just et al. (2012). Whyatt et al. (2014) found that the maternal urinary MBzP levels were associated with the early onset of eczema in children, and a >70% higher risk of asthma among children with maternal prenatal BBP and DBP urinary metabolite concentrations in the third (31.9 ng/ml MBzP and 80.5 ng/ml MnBP) versus the first tertile (5.8 ng/ml MBzP and 19.3 ng/ml MnBP). Just et al. (2012) found that levels of maternal urinary MBzP were associated with increased an early onset of eczema.

Braun et al. (2013) reviewed epidemiological studies on associations between exposure to DEHP, BBP and DBP and allergic diseases including asthma and eczema (three case-control, one prospective, and one cross-sectional study). The authors highlighted that children from homes with high concentrations of DEHP and BBP in dust had high incidences of allergy, asthma, rhinitis or eczema (Bornehag et al. 2004; Hsu et al. 2012; Kolarik et al. 2008) and that higher maternal BBP exposure in pregnancy was associated with early-onset eczema in children (Just et al. 2012). The authors concluded that there is converging evidence that DEHP and BBzP exposures during childhood may be associated with the development of allergic disease. However, they stressed the need for further longitudinal studies to identify susceptible periods and the need to control for other environmental risk factors for asthma that may co-vary with phthalate exposure.

Kimber and Dearman (2010) reviewed 16 epidemiological studies and reported that in general these studies all reported associations between PVC materials and increased immunological symptoms such as asthma and other respiratory symptoms in workers, adults at home, and children. Also associations between PVC materials and increased development of atopic disease

[29] Adjuvants are used in vaccines to induce broader immune responses and to enhance immune responses to poorly immunogenic antigens. Substances known to have adjuvant properties include aluminium containing substances, lipopolysaccharides, saponins, emulsions and cationic liposomes (Mastelic et al. 2010; Kimber and Dearman 2010).

(rhinitis or eczema) were reported in several studies with children. The authors stressed that causality was not demonstrated in these studies.

Bornehag and Nanberg (2010) reviewed eight papers (corresponding to 5 study populations) reporting on the relationship between phthalates in the environment and/or other sources of exposure, and airway diseases in children. The authors underlined that most of the studies are cross-sectional which reduce the possibility for conclusive results and that only two reports included data on phthalate exposure, whereas other studies reported instead PVC related material in the home as a proxy to phthalate exposure. Despite these shortcomings, the authors concluded that the epidemiological studies provide indications for a relationship between phthalate exposure and asthma in children.

Jaakkola and Knight (2008) reviewed the epidemiological literature on the role of exposure to phthalates from PVC-containing products in the development of asthma and allergies. Several cases of occupational asthma (n=29) have been linked to exposure to fumes emitted from PVC film. Ten epidemiologic studies in workers provided evidence for associations between heated PVC fumes and asthma and respiratory symptoms (one of these studies also examined the role of exposure at home). Five studies in children showed an association between PVC surface materials in the home and the risk of asthma. A meta-analysis of 4 studies provided an odds ratio (OR) for asthma of 1.55 (95% CI 1.18–2.05) and a meta-analysis of 3 studies an odds ratio (OR) for allergic rhinitis of 1.32 (95% CI 1.09–1.60).

Experimental studies

Usually Balb/c mice have been used to study the modulation of the immune response by phthalates, usually with DEHP as test compound and ovalbumin (OVA) as antigen.

Intraperitoneal or subcutaneous route of exposure (mice)

DEHP administered by intraperitoneal or subcutaneous injection (± 5 mg/kg bw) induced adjuvant effects (Larsen et al. 2001b, Lee et al. 2004; Larsen and Nielsen 2007). MEHP, the monoester metabolite of DEHP, had an adjuvant effect at 50 µg/kg bw administered subcutaneously (Larsen et al. 2001a). Also intraperitoneal administration of DINP (Lee et al. 2004) and subcutaneous administration of DINP, DBP and DNOP (Larsen et al. 2002) and the metabolite MINP showed adjuvant effects (Larsen et al. 2001a). At higher doses in the same experiments (at 5 mg/kg bw), MEHP, MINP and DINP appeared to show immunosuppressive effects (Larsen et al 2001a, 2002).

Takano et al. (2006) induced atopic dermatitis in male NC/Nga mice[30] by intradermal injection with mite allergen in the ear. Intraperitoneal exposure to DEHP at a dose of 0.8–20 µg (on days 4, 3, 10, and 17) but not 100 µg caused deterioration of skin lesions.

Yanagisawa et al. (2008) administered DEHP to pregnant female NC/Nga mice at a dose of 0, 0.8, 4, 20, or 100 µg by intraperitoneal injection at gestation days 0, 7, and 14. In a second experiment maternal exposure occurred at these doses during postnatal days 1, 8, and 15

[30] NC/Nga mice are a strain that is prone to develop atopic dermatitis and therefore are used as a model to study atopic dermatitis (Suto et al. 1999).

(during lactation). Male offspring were injected intradermally with mite allergen into the ear. Maternal exposure to 100 µg DEHP during the neonatal period (second experiment), but not during fetal period (first experiment), caused deterioration of skin lesions.

Dermal route of exposure (mice)

Topical application of 25-100% DEHP did not induce adjuvant effect on IgE or OVA-specific IgG1 (Dearman et al. 2008).

Topical application of BBP (100 mg) significantly increased anti-OVA IgG1 production but there was no impact on anti-OVA IgE antibody responses (Dearman et al. 2009).

Butala et al. (2004) reported absence of effects on IgE and IL-4 or IL-13 secretion in lymph node derived cells following topical application of DEHP, BBP, DIHP and DINP on female B6C3F1 mice (applied on the flanks undiluted five times weekly for 2 weeks, 7 days later animals were challenged, and sacrificed 7 days after that).

Li et al. (2014) dermally exposed male mice with 0.4, 4 and 40 mg/kg bw/day DBP from day 1 to 40 and on days 41 and 42 sensitised the mice with 0.5% fluorescein isothiocyanate (FITC) (in 1:1 acetone/DBP). Ear thickness was measured following challenge with 0.5% FITC on the ear. Total IgE, Th cytokines, histopathology and ear swelling suggest that high doses of DBP (4.0 and 40 mg/kg bw/day) applied topically may promote and aggravate atopic dermatitis.

Shigeno et al. (2009) induced contact hypersensitivity in female BALB/c mice by applying topically (on the abdomen or left ear) 0.5% fluorescein isothiocyanate (FITC) in acetone:DBP (1:1), or with 3% picryl chloride in acetone:olive oil. Elicitation was performed on the right ear. DBP increased thymic stromal lymphopoietin in the skin during the sensitisation phase. DBP appeared to be indispensable in both the sensitisation or elicitation phases of FITC-induced contact hypersensitivity. The authors concluded that thymic stromal lymphopoietin induced by DBP during the sensitisation phase plays a role in the cause of FITC-induced contact hypersensitivity and may contribute to the Th2 commitment.

Previously Sato et al. (1998) had already shown that DBP had an adjuvant effect during the sensitisation phase in mice when a solution with FITC and 50% DBP was applied on the skin during sensitisation (80 µl per forelimb) and challenge (20 µl on the right ear).

A similar sensitisation protocol was used in Imai et al. (2006). A strong enhanced ear-swelling response and increased FITC positive dendritic cells in the draining lymph nodes were observed with DBP and DPP but not with DEHP and DINP.

Inhalation route of exposure (mice)

A long-term inhalation study (20min/day, 5 days/week, 14 weeks) with DEHP aerosols of 0.022-13 mg/m^3 induced adjuvant effects at 13 mg/m^3 (Larsen et al. 2007). In a study by the same group, MEHP, the monoester metabolite of DEHP, had an adjuvant effect at 0.03 and 0.4 mg/m^3 in female mice following exposure during 14 weeks (20 min/day, 5 days/week for 2 weeks and thereafter weekly) (Hansen et al. 2007). MEHP caused increased numbers of macrophages, but not other inflammatory cells, in bronchoalveolar lavage fluid (BALF) in Larsen et al. (2004).

Oral route of exposure

A. Enhancement of inflammatory response (adjuvant effect)

In Tonk et al. (2012), juvenile (10 days old) and young adult (50 days old) **rats** were exposed by gavage for 40 days to vehicle or 7 different doses of DEHP (1-1000 mg/kg bw/day). At necropsy, reproductive organs were weighted and serum hormone levels determined. A subgroup of rats (n=4-8) were immunised with subcutaneous Keyhole limpet hemocyanin (KLH) injections (0.2 ml of 5 mg/ml KLH) two times with an interval of 40 days to determine the primary and secondary IgM or IgG response to KLH. At necropsy, the spleen was removed and cytokine production after KLH stimulation was measured on spleen cell cultures. In the non-immunised rats, parameters as lymphocyte subpopulations, natural killer cell (NK) cell activity and lymphocyte proliferation were investigated. The data were analysed using the benchmark dose approach. Effects on immune parameters included a dose-dependent increase in primary and secondary anti-KLH IgG in both juvenile and adult animals (BMDL5 levels of 49 and 66 mg/kg bw/day); a significant dose-dependent decrease in NK cell activity and decrease in white blood cells, neutrophils, lymphocytes and monocyte counts in juveniles only (BMDL5 levels of 12 - 30 mg/kg bw/day); and a significant dose-dependent increase in TNFα production by adherent splenocytes in adult rats only. Effects were seen at lower levels in juveniles for IL-2, IL-4, IL-6, IL-13, IFN-γ, and TNFα in ex-vivo re-stimulation of splenocytes with KLH, with IL-10 being equally responsive in adult animals (BMDL5 levels of 0.09 - 63 mg/kg bw/day). The authors concluded that the results show that DEHP immunotoxicity during maturation of the immune system may influence the prevalence of early-onset immune-related diseases in humans.

In Guo et al. (2012), 5-6 week old mice received vehicle, three doses of DEHP only (30, 300, 3000 µg/kg bw/day), OVA only, or three doses of DEHP+OVA by oral gavage for 52 days. The animals were sensitized with OVA + Al(OH)3 on days 25, 39 and 47. The animals in the OVA only and in the OVA+DEHP dose groups were subsequently challenged with an ovalbumin aerosol for 7 days. The animals in the in the saline control and in the DEHP only groups received a saline aerosol for 7 days. Total IgE levels in serum were significantly higher at 30 µg/kg bw/day and above in the DEHP+OVA groups compared to the OVA only group. Compared with OVA only, DEHP+OVA at 3000 µg/kg bw/day significantly increased IL-4 and the ratio of IL-4/IFN-γ in lung tissue samples; the relative number of eosinophil to total cell numbers in BALF; and expiratory resistance and inspiratory resistance. Histological examination confirmed a dose-dependent increase in severity of airway pathology (thickening of the mucosa, epithelial folding and airway obstruction). The authors concluded that the data supports that DEHP may promote and aggravate allergic asthma via an adjuvant mode of action.

In You et al. (2014), methods were comparable to the study by Guo et al. (2012), although only one (higher) oral dose of DEHP was examined and the dosing period was shorter. In You et al. (2014), mice received vehicle, DEHP only at 30 mg/kg bw/day, OVA only, OVA+VitE, DEHP+OVA, DEHP+OVA+VitE by oral gavage for 14 days. The animals were sensitised with OVA + Al(OH)3 on days 1, 11 and 14. The animals in the OVA only, OVA+VitE, DEHP+OVA, and DEHP+OVA+VitE dose groups were subsequently challenged with an ovalbumin aerosol for 4 days. The animals in the saline control and in the DEHP only groups received a saline aerosol. Total IgE levels in serum and IL-4 in lung tissue were significantly higher in the DEHP+OVA group compared to the OVA only group. The total IgE levels in serum and IL-4 in

lung tissue of DEHP+OVA+Vit E were significantly lower than the DEHP+OVA group. The number of eosinophils and ratio of neutrophils in BALF was increased in the DEHP+OVA group compared with the OVA only group, but not significantly. DEHP had a significant negative effect on airway parameters with MCH at doses of 0.05 or 0.1 mg/kg bw. Histological examination showed severe changes (i.e. airway remodelling) in the DEHP+OVA group (e.g., goblet cell hyperplasia, eosinophil and neutrophil infiltration, prominent augmented mucus secretion, small airway wall thickening, fibrosis in surrounding peribronchial areas, epithelial folding and thickened subepithelial cell layers). ROS levels were significantly increased, and GSH levels decreased, in the DEHP+OVA group compared with the OVA group. The authors concluded that the adjuvant effect of DEHP might result from oxidative stress induced by DEHP.

Han et al. (2014) admistered weanling mice with vehicle only or with DEHP at 30, 300 and 3000 µg/kg bw/day by oral gavage from day 1-28. The animals were OVA sensitised at day 7 and 16 subcutaneously. OVA-specific IgE and IgG1 were significantly increased in serum in all DEHP dose groups compared with the control. Spleen tissue cryosections revealed significant increased germinal center formation in the 300 and 3000 µg/kg bw/day dose levels. Furthermore, the data suggested DEHP could stimulate the expansion of CD4+CXCR5+ICOS+/CD4+CXCR5+PD-1+Tfh cells and CD19+CD138+GL7+ plasma cells. Tfh cell intracellular cytokine levels (IL-21 and IL-4) and Bcl-6 and c-Maf gene transcription was significantly increased in all DEHP dose groups. The authors concluded that the study shows that DEHP has adjuvant effects on Tfh cells.

In Zuo et al. (2014), mice were exposed by gavage for 32 days to 0.45 or 45 mg/kg bw/day of **DBP**. Mice were immunised with OVA and adjuvant (Al(OH)3) on day 7, 21 and 28 of treatment, and challenged with a 1% OVA aerosol on days 33-39. An increase in serum IgE and IL-4 was seen at 0.45 mg/kg bw/day but not at 45 mg/kg. The authors suggested this may be due to spleen injury at high DBP doses. The authors concluded that DBP may exert an adjuvant effect.

Sadakane et al. (2014) orally administered NC/Nga mice once a week for 4 weeks with DEHP at 9.3, 166.3, or 3325 µg/animal (≈0.372, 6.652 or 133 mg/kg bw/day assuming a bw of 25g) or with DINP at 6.6, 131.3, or 2625 µg/animal. Animals were treated with mite allergen by subcutaneous injection in the ear for 2–3 days a week (a total of 8 times). Clinical scores of skin lesions did not significantly differ among the given doses. Eosinophil counts in the subcutaneous tissue were increased in the DEHP dose groups (with statistical significance in the 3358 µg group). DEHP exposure tended to increased IL-14 and degranulation of mast cells but not at the highest dose, with local expression of the proinflammatory protein MIP-1a tending to increase except at the highest dose. The authors concluded that oral administration of DEHP (or DINP) may increase the allergic response of atopic dermatitis.

Shin et al. (2014) dietary administered pregnant BALB/c mice with DEHP at 2500 ppm (≈313 mg/kg bw/day) from gestation day 13 to lactation day 21. Pups were sensitised with OVA+ Al(OH)3 by intraperitoneal injection on postnatal days (PNDs) 9 and 15. They received an airway challenge of OVA on PNDs 22, 23 and 24. Offspring from treated mice showed reduced asthmatic responses induced by OVA challenge.

Yang et al. (2008) administered DEHP to Wistar **rats** at 0.7 and 70 mg/kg bw/day by gastric gavage for 30 days. DEHP significantly increased airway hyperresponsiveness, airway

remodelling, and infiltration eosinophils in OVA-immunised rats. The authors concluded that DEHP administered by gavage displays an adjuvant effect on the respiratory system in the ovalbumin-immunised rat model.

In Piepenbrink et al. (2005), pregnant female **rats** were dosed (GD 6 to 21) with four doses of DEHP (37.5 to 300 mg/kg bw/day). KLH immunisation at 3 and 4 weeks of age was followed by immune assessment at 5 weeks of age; KLH immunisation at 11 and 12 weeks of age was followed by immune assessment at 13 weeks of age. Various immune parameters were examined in adult offspring. DEHP did not appear to have measurable effects on immune parameters after *in utero* exposure.

In Badr et al. (2007) **rats** were exposed dietary at a single dose level of 12 000 ppm/day (≈1 200 mg/kg bw/day) of DEHP over 21 days and were then primed with Mycobacterium bovis purified protein derivative (PPD) in a Complete Freund's Adjuvant (induced Th1 response). Fifteen days after they were challenged with Sepharose 4B beads covalently coupled with PPD. Exposure to DEHP altered the balance of Th1/Th2 cytokines in the liver towards a Th2 phenotype.

B. Other studies

Yang et al. (2000) found that DEHP resulted in atrophy of thymus and spleen in male C57BL/6 mice receiving 0.1% DEHP in the diet (≈70 mg/kg bw/day). This study did not include a sensitisation and challenge protocol.

He et al. (2013) administered female mice via the intranasal route with a saline, OVA or OVA+DEHP solution at 0.0004, 0.008, 0.16 or 3.2 µg DEHP/16 µl (8 ml/per anterior naris) for four times at 2-week intervals and then four times at 2-day intervals. DEHP increased the number of neutrophils in nasal cavity lavage fluids (NCLF), but not significantly. Histological findings were slightly worsened in the OVA+DEHP groups than in the OVA group. IL-13 levels in NCLF were significantly higher in the treated mice. The authors concluded that although DEHP did not markedly aggravate inflammatory pathology, DEHP may have the potential to disrupt local immune systems in the nasal cavity.

<u>In vitro studies</u>

Bornehag and Nanberg (2010) reviewed in vitro studies and concluded that the data on isolated primary lymphocytes and the EL4T cell line suggests that several phthalates would have an antigen-dependent direct cellular effect, thus promoting Th2 differentiation. The limited available data suggest direct potentiation effects on mast cells and basophils, with data on monocytes and macrophages being inconclusive. The authors concluded that "overall a picture of modulation of immune responses is emerging".

DEHP increased the production of non-allergic pro-inflammatory chemokines and cytokines in human macrophages (Nishioka at al. 2012) and caused mast cells to release histamine and betahexoaminidase (Kuo et al. 2013), which supports the earlier conclusion by Bornehag and Nanberg (2010).

Discussion

As described in Section B.8, humans are mainly exposed to DEHP via the oral route. For DBP, BBP and DIBP dermal exposure is considered to be significant. Also inhalation exposure contributes to the exposure of humans to the four phthalates.

Case reports and epidemiological data show clear associations between PVC materials or phthalate exposure and increased immunological symptoms (asthma, other respiratory symptoms, rhinitis and eczema). However, the epidemiological studies have not established a causal link between phthalate exposure and increased immunological symptoms.

Experimental studies generally show that DEHP administered via the intraperitoneal or subcutaneous route of exposure induced adjuvant effects in mice. Similar conclusions can be reached for subcutaneous administration of the monoester metabolite of DEHP (MEHP) as well as DBP and other phthalates or phthalate metabolites. In isolation, the biological significance of these observations is unclear as a consequence of the route of exposure in these experiments. However, they have a value in providing evidence of 'proof of concept' and support positive findings in studies using inhalation, oral or dermal routes of exposure.

The evidence from three dermal studies is generally not suggestive for adjuvant effects in mice following topical application of DEHP, BBP and DINP. However, all four studies with DBP confirm an adjuvant effect via the dermal route of application. The applied concentrations in these studies were high compared to exposure situations in humans. However, humans are exposed over longer periods to much larger surfaces. It is unclear whether the adjuvant effects observed with DBP could also be exerted when exposure occurs at lower concentrations but over longer periods and larger surface areas. If this were the case, then the concentrations of DBP used in these studies might be relevant to humans[31]. DEHP administered by gavage displays an adjuvant effect on the respiratory system in both rats and mice which provides some evidence that the adjuvant properties of phthalates might be systemic in nature (Yang et al. 2008; Guo et al. 2012).

Inhalation studies provide evidence that DEHP and its monoester metabolite MEHP induced adjuvant effects in mice at 30 µg/m^3 (LOAEC in Hansen et al. 2007) and above[32]. However, the test substance used in Hansen et al. (2007) was MEHP and not DEHP. The study by Larsen et al. (2007) with similar duration and study protocol showed adjuvant effects of DEHP only at

[31] As a matter of illustration, a NOAEL of 0.4 mg/kg bw/day for DBP from Li et al. (2014) might be used as a starting point, an AS factor for mice of 7, a default AF for other interspecies differences of 2.5, and a default AF for intraspecies differences of 10 (total AF = 175). Assuming dermal absorption in humans and the mouse is the same, the external dermal DNEL for DBP would thus be 2.3 µg/kg bw/day, or the internal DNEL would be **0.23 µg/kg bw/day** assuming 10% dermal absorption (lower than the internal DNEL of 6.7 µg/kg bw/day for reproductive toxicity). In comparison, the reasonable worst case internal dermal exposure was estimated to be around 6 µg/kg bw/day for DBP.

[32] As a matter of illustration, a LOAEC of 30 µg/m^3 for MEHP from Hansen et al. (2007) might be used as a starting point, the corrected LOAEC is 0.3 µg/m^3 when corrected for the exposure duration (0.33/24 x 5/7 = 0.01). No AS factor should be applied, thus the AF for (other) interspecies differences is 2.5 and the default AF for intraspecies differences is 10. To extrapolate from a LOAEC to the NAEC a default factor of 3 can be used. The total AF = 75. The inhalation DNEL for MEHP would then be 0.004 µg/m^3.

higher doses (13 mg/m^3). Nevertheless, Yang et al. (2008) and Guo et al. (2012) found that DEHP administered by gavage displays an adjuvant effect on the respiratory system in both rats and mice. There is also general evidence from both human and animal studies which indicates that effective sensitisation of the respiratory tract can result from dermal contact with a chemical respiratory allergen (ECHA Guidance R.8). Thus, there is evidence that aggregated exposure to phthalates from several routes might be relevant in aggravating respiratory allergies [33].

Oral studies provide the strongest basis to suggest an adjuvant effect of DEHP in hypersensitivity. In **all** studies with direct oral exposure (7), DEHP displayed adjuvant effects on airway hyperresponsivenes, atopic dermatitis or liver response. The only oral study with DBP (Zuo et al. 2014) suggests DBP may exert an adjuvant effect in mice. The explanation why effects are seen at 0.45 mg/kg bw/day but not at 45 mg/kg bw/day was spleen injury. This explanation might be plausible since the spleen was reported to undergo a serious atrophy compared with the control at this dose level.

The study by Tonk et al. (2012) provides some evidence that juveniles are more sensitive to immunotoxicity of DEHP than adults. Exposure to DEHP in utero in mice and rats does not appear to lead to enhanced immune response in rats (Shin et al. 2014; Piepenbrink et al. 2005), which suggests that in utero exposure may not be a sensitive window for immunotoxicity of DEHP. The epidemiological studies from Whyatt et al. (2014) and Just et al. (2012) on the other hand would suggest that prenatal exposure might be a sensitive window of exposure to DBP and BBP.

Robinson and Miller (2015) concluded that most studies appear to support that DEHP and DBP are adjuvants. Similarly, Bornehag and Nanberg (2010) concluded that the majority of studies identified adjuvant effects on Th2 differentiation, production of Th2 cytokines, enhanced levels of Th2 promoted immunoglobulins (IgG1 and IgE) and some data suggests enhanced mast cell degranulation and eosinophilic infiltration. The studies that became available after the review confirm this conclusion. Bornehag and Nanberg (2010) underlined that the relevance of these findings to exposure in the human population is an open question and stressed that the molecular targets are yet to be identified.

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[33] A NOAEC of 13000 µg/m^3 for DEHP from Larsen et al. (2007) might be used as a starting point, the corrected NOAEC is 130 µg/m^3 when corrected for the exposure duration (0.33/24 x 5/7 = 0.01). No AS factor should be applied, thus the AF for (other) interspecies differences is 2.5 and the default AF for intraspecies differences is 10. The total AF = 25. A ***tentative*** **inhalation DNEL** for DEHP would then be **5.2 µg/m^3**. In comparison, the estimated reasonable worst case inhalation exposure of DEHP via indoor air alone is **4.8 µg/m^3**.

Assuming a standard respiratory volume of 2.02 m^3/kg bw in mice and 100% absorption over the oral and inhalation routes, the NOAEL would be 262.6 µg/kg bw/day. The total AF is 175: AS of 7, thus the AF for (other) interspecies differences is 2.5 and the default AF for intraspecies differences is 10. Thus a ***tentative*** **oral or internal DNEL** would be **1.5 µg/kg bw/day**. The DNEL is lower than the DNEL for reproductive toxicity. The 95th percentile of exposure to DEHP in children based on DEMOCOPHES biomonitoring data is **12 µg/kg bw/day (above the DNEL**).

Starting point for DNEL derivation

The available studies using the intraperitoneal and subcutaneous exposure route do not allow setting DNELs for the reasons discussed above. DEHP did not induce adjuvant effects via the dermal route. DBP on the other hand induced adjuvant effects via the dermal route. However, no appropriate point of departure from the dermal studies with DBP by Shinego et al. (2009), Sato et al. (1998) and Imai et al. (2006) is available (single application at high concentration at induction and at challenge) and the study by Li et al. (2014) is not considered sufficiently robust to use as a starting point for DNEL derivation.

A NOAEC of 13000 µg/m^3 for DEHP from Larsen et al. (2007) might be used as a starting point to derive a tentative[34] inhalation DNEL of 5.2 µg/m^3 (duration scaling is applied)[33]. However it is not clear whether the inhaled dose and duration of exposure, or rather the concentration, drive the effect[35]. Without duration scaling, such tentative inhalation DNEL would be 100 times higher.

Regarding the oral studies with DEHP, BMDL5 levels of 0.09 - 63 mg/kg bw/day may be used as a point of departure in the rat (equivalent in the mouse: 0.05 - 36 mg/kg bw/day[36]) (Tonk et al. 2012). Guo et al. (2012) and Han et al. (2014) are indicative of effects in the mouse at 0.03 mg/kg bw/day and 0.3 mg/kg bw/day respectively. A tentative oral DNEL using a NOAEL of 0.3 mg/kg bw/day from these studies as a starting point would be 1.7 µg/kg bw/day using a total AF of 175.

The oral study with DBP is not considered sufficiently robust (questions with regard to the dose-response relationship) to allow deriving a DNEL from this study.

Conclusion

All studies with direct oral exposure to DEHP, the only oral study with DBP, and two inhalation studies with DEHP and its monoester metabolite, MEHP, displayed adjuvant effects in rodents. The two studies with prenatal exposure suggest that in utero exposure may not be a sensitive window of exposure for immunotoxicity. All studies with DBP confirm an adjuvant effect via the dermal route of application, however studies with DEHP, BBP and DINP are generally not suggestive for adjuvant effects following dermal application. Further supportive evidence for adjuvant properties of phthalates is provided by studies using the intraperitoneal or subcutaneous route and from epidemiological studies.

It can be concluded that there are indications that phthalate exposure could lead to immunological disorders in humans (allergy, asthma and eczema), possibly at levels lower

[34] 'Tentative' is used to indicate that the available data and the assessment and interpretation of this data does not allow to derive a firm (robust) DNEL. However, the tentative DNEL provides an indication of the dose level where a DNEL might be situated.

[35] ECHA guidance R.8 states *"Since there are currently no available methods to determine the thresholds and to establish DNEL for respiratory hypersensitivity, only qualitative risk assessment for this endpoint can be performed"*. In particular, there is no guidance with regard to immunotoxicity with an adjuvant mode of action.

[36] The AS factor for the mouse is 7, whereas for the rat the AS factor is 4, thus a factor of 0.57 can be applied to the dose levels in the rat.

than reproductive toxicity. However, in order to take effects on the immune system into consideration for quantitative risk assessment, there is a need for further robust data.

B.4.3.4. Effects on the metabolism

Associations between prenatal phthalate exposure and obesity or diabetes in adulthood have been investigated in epidemiological studies, and in vitro and animal studies have provided mechanistic knowledge indicating obesogenic effects of phthalates, e.g. by promoting differentiation of and accumulation of lipid in lipid cells (reviewed by Kim and Park 2014). The fetal period is considered critical to phthalate exposure, but few studies have been able to clarify the role of prenatal exposure to phthalates in the obesity epidemic.

B.4.3.5. Neurodevelopment

Altered neurodevelopment has been associated with high phthalate exposures in children, as reviewed by Miodovnik et al. (2014). Numerous behavioural disorders including autism spectrum disorders, ADHD, learning disabilities, and altered play behaviour have been associated with higher phthalate exposure in humans (reviewed by Braun et al. 2013). Animal studies examining behavioural effects of phthalate exposure have shown some effects that may be related to altered sex differentiation, whereas other behavioural effects are not clearly linked with disruption of sex hormones. Different modes of action for phthalate effects on neurodevelopment have been proposed, including interference with the thyroid hormone system, altered calcium signalling, relation to activation of peroxisome proliferator activated receptors (PPARs) in brain and altered lipid metabolism (Miodovnik et al. 2014).

B.4.3.6. Conclusion

It can be concluded that there are indications that phthalate exposure could lead to immunological disorders (allergy, asthma and eczema), possibly at levels lower than reproductive toxicity. Effects on other endpoints such as metabolism and neurodevelopment have not been elucidated yet.

B.4.4. Derivation of DNELs

The DNELs are based on N(L)OAELs for anti-androgenic effects seen in developmental studies, i.e. where doses are administered to adult female rats during gestation and lactation. Therefore, risk calculations for pregnant women are particularly relevant. Risk calculation for infants and children are also based on these N(L)OAELs and it is possible the DNELs for this age group would be higher. As sufficient dose-response studies in animal models mimicking direct exposure of children are lacking, DNELs based on NOAELs of dams are used for toddlers and children, but some uncertainty is associated with this DNEL. The prenatal and early postnatal period is considered the most sensitive period for the effects of phthalates (Welsh et al., 2008) and this could point towards higher NOAELs for children than foetuses and

newborns, i.e. the selected N(L)OAELs for DNEL setting may lead to an overestimation of the risk. However, the N(L)OAELs in experimental studies are based on the dose levels given to the dams and are not the dose levels given directly to the foetuses and the newborns. The internal dose levels received by the newborn experimental animals via lactation are most likely lower than the dose levels given to the pregnant dams as only a fraction is likely to be transported across the placenta or excreted in maternal milk. This means that internal N(L)OAELs of pups (neonatal and lactating) may actually be lower than internal N(L)OAELs of the dams, which could lead to an underestimation of the risk for directly exposed children.

In some cases a NOAEL from a repeated dose study may be preferred for risk assessment in a child-specific scenario. However, in this case the anti-androgenic endpoints are selected as relevant for combined risk assessment of phthalates, and repeated dose studies would not be preferred as these studies do not include hormone-sensitive endpoints.

Overall, N(L)OAELs based on anti-androgenic effects in developmental studies are considered the most relevant available data for the current risk assessment. An overview of selected overall N(L)OAELs for DNEL derivation and the applied assessment factors are provided below and in Table B9.

DEHP

The **NOAEL of 4.8 mg/kg bw/day** is selected for DNEL setting and use in combined risk assessment. This NOAEL was also used by the EU RAR and EFSA. This is based on testicular effects that can be attributed to an anti-androgenic mode of action.

DBP

As the observed effects of DBP on the male mammary gland and testes are considered anti-androgenic, and as EFSA has chosen the study by Lee et al. to derive the TDI, the **LOAEL of 2 mg/kg bw/day** is selected for DNEL setting and use in the current combined risk assessment.

DIBP

Few reproductive toxicity studies have been published on this compound compared to DEHP and DBP. No two-generation studies are available and the substance has not been tested at doses <100 mg/kg bw/d. Current data suggest that DIBP could have similar effects to DBP, if studied at lower dose levels. If the potency difference between DIBP and DBP, as a very rough estimate of the observed effects in Saillenfait et al. (2008) (type of effects seen at 500 and 625 mg/kg bw /day, corresponding to a difference of 25%), is extrapolated from the high dose area to the lower dose area, an estimated LOAEL for DIBP would be 25% higher than the current LOAEL for DBP (2 mg/kg bw/day). Available information is shown in Table B7. A **LOAEL for DIBP of 2.5 mg/kg bw/day** is selected for use in the current combined risk assessment.

BBP

A **NOAEL of 50 mg/kg bw/day** is selected for DNEL setting and use in combined risk assessment, as this level is used for developmental effects in the EU risk assessment, and this is based on an anti-androgenic endpoint (reduced anogenital distance in male rats).

DNEL derivation is given in Table B9, below.

Table B9 Overview of DNEL derivation.

	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint and study reference	AFs	Correction for absorption§	DNEL internal dose (mg/kg bw/day)
DEHP	4.8	14	Small male reproductive organs (testes/epididymes/seminal vesicles) and minimal testis atrophy in Wolfe and Layton (2003)	4*2.5*10 = 100	0.7	0.034
DBP	–	2	Reduced spermatocyte development at postnatal day 21, and mammary gland changes (vacuolar degeneration and alveolar atrophy) in adult male offspring in Lee et al. (2004)	4*2.5*10*3 = 300	1	0.0067
DIBP	-	2.5	Read-across from DBP	4*2.5*10*3 = 300	1	0.0083
BBP	50	100	Reduced anogenital distance in Aso et al. (2005), Tyl et al. (2004) and Nagao et al. (2000). Reduced reproductive organ weights and altered sperm counts and motility in Ahmad et al. (2014)	4*2.5*10 = 100	1	0.50

§oral absorption fraction=0.7 in rats for DEHP and 1 for other compounds see Table B3

For the available studies, Table B9 lists the DNELs for consumers and the general public, including pregnant women and children. In accordance with ECHA guidance Chapter R.8, DNEL calculation uses an uncertainty factor of 2.5 for interspecies differences; an allometric scaling factor of 4 for rats and 7 for mice; a factor of 10 for intraspecies differences; and a factor of 3 as extrapolation from LOAEL to NAEL if no NOAEL is available. From the studies described in Table B4, one NOAEL is selected for each phthalate and used for the combined risk assessment. When selecting NOAELs for combined risk assessment particular attention was given to the assessments in the EU risk assessment reports (EU RARs) or by EFSA opinions. It should be noted that NOAELs are based on different types of specific effects assumed to have the same mode of action.

No other assessment factors for e.g. different duration/exposure time (e.g. developmental studies and 2-generation studies versus subchronic or chronic studies) were considered relevant. The DNEL for internal dose and the selected NOAELs for the four phthalates are given in Table B9.

B.4.5. Uncertainties

Several uncertainties in the hazard characterisation of the four phthalates are suggesting that the DNELs carried forward for combined risk assessment may be lower, making it possible that the current DNELs may contribute to underestimating the risk from these 4 substances. The elements are described below.

B.4.5.1. DNEL setting

DEHP

The proposed NOAEL for DEHP may be associated with some uncertainty. A more cautious NOAEL could be based on the findings of cryptorchidism in a few animals at 5 mg/kg bw/day in the study by Andrade et al. (2006) and the presence of mild dysgenesis of external genitalia at 3 mg/kg bw/day in the study by Christiansen et al. (2010). The studies by Christiansen et al. (2010) determining a LOAEL of 3 mg/kg bw/day and the NOAEL of 1.2 mg/kg bw/day in the study by Andrade et al. (2006) are presented below.

Researchers from the United States Environmental Protection Agency recently published a paper presenting hazard indexes for combined risk assessment of the five anti-androgenic phthalates DEHP, DBP, DIBP, BBP and DINP (Christensen et al. 2014). In this paper, the study by Christiansen et al. (2010) was applied for setting an “alternate value” as reference dose for DEHP. The LOAEL of 3 mg/kg bw/day was applied as point of departure, and a total uncertainty factor of 1000 was applied, which included an AF of 10 for LOAEL to NOAEL extrapolation (Christensen et al. 2014). The resulting reference value was thus 0.003 mg/kg bw/day. The value is similar to the DNEL of DBP.

A DNEL derived on the basis of these studies (using an AF of 3 for LOAEL to NOAEL extrapolation in the Christiansen study) would result in alternate DNELs of 0.007 and 0.008 mg/kg bw/day, respectively; close to the DNEL for DBP. This is in good agreement with the knowledge that DEHP and DBP have relatively similar potencies for effects on e.g. fetal testosterone production (Howdeshell et al. 2008).

	NOAEL	LOAEL	Uncertainty factor	DNEL, external dose	DNEL, internal dose [a]	Endpoint and species	Reference
DEHP	-	3	2,5*4*10*3=300	0.01	0.007	↑ mild dysgenesis of external genitalia, rat	Christiansen et al. (2010)
DEHP	1.2	5	2.5*4*10 = 100	0.012	0.008	Reproduction (↑ cryptorchidism); ↓ daily sperm production at 15	Andrade et al. (2006)

DBP

No NOAEL has been established for DBP. A LOAEL is available. An uncertainty regarding establishing a no-effect level therefore exist for DBP.

As the observed effects of DBP on the male mammary gland and testes (delayed germ cell development) are considered anti-androgenic, EFSA (EFSA 2005a) has chosen the LOAEL of 2 mg/kg bw/day in the study by Lee et al. (2004) to derive the TDI. This LOAEL has been supported by RAC (RAC 2012). However, as effects on the mammary gland and delayed germ cell development have only been investigated for DBP, it is not possible to compare DBP, DIBP, DEHP and BBP based on potency differences for these effects. The DNELs for DEHP and BBP do not therefore account for these effects, and may not be sufficiently protective for these endpoints.

DIBP

In the absence of conclusive experimental data, read-across from DBP has been performed, as DBP is the linear isomer of DIBP. The potency difference between DIBP and DBP was estimated to be 25%.This assumption is based on structural similarities, comparable observations at equal or similar doses of effects on fetal testosterone reduction, AGD, steroid gene expression related to the steroid biosynthesis pathway and adverse effects on reproductive tract malformations and reduced reproductive organ weights. The experimental evidence for concluding that DIBP is of similar anti-androgenic potency is considered robust, but the assumption of potency difference (25%) is uncertain.

BBP

For DBP and DEHP the lowest LOAELs were seen on endpoints including testicular histology, mammary histology of male adults and presence of mild dysgenesis of external genitalia, and these endpoints have not been examined for BBP. The potency of BBP to reduce foetal testosterone production appears to be comparable to DEHP and DBP (Howdeshell et al. 2008), and it may be speculated that further studies on BBP including endocrine sensitive endpoints (such as reduced spermatocyte development and male mammary gland changes) would reveal effects at lower doses than 50 mg/kg bw/day.

B.4.5.2. Toxicity other than toxicity for reproduction

In recent years, a number of experimental and epidemiological studies have examined the possible influence of phthalate exposure on the immune system, the metabolic system and neurological development. Some of these studies indicate that reproductive toxicity may not be the most sensitive endpoint for the effects of DEHP and that the DNELs selected for the current combined risk assessment may not be sufficiently protective against other effects of phthalates. See also section B.4.4 and Annex D.

B.4.5.3. Threshold for phthalates

In December 2014 the Member State Committee (MSC) unanimously acknowledged that for DEHP, DBP, DIBP and BBP there is "*scientific evidence on the endocrine activity and on the link between this activity and the adverse effects to human health. However, the MSC did not reach unanimous agreement on whether this constitutes an equivalent level of concern to CMRs (majority view), as a minority of members were of the view that the concern related to endocrine disruption is already covered by the existing identification as SVHC due to toxicity to reproduction*" (ECHA 2014). As no unanimous agreement could be reached in the MSC, the Commission will take the final decision.

According to current policy, substances identified as having endocrine disruptive properties according to Article 57 (f) do not have a threshold, except where it can be demonstrated that a threshold exists (European Commission 2014). Even though RAC has previously established DNELs for reproductive toxicity for the four phthalates (ECHA 2012a, 2013c,d,e), these did not take into account the need to specifically assess and document the existence of a threshold if the phthalates are identified as having endocrine disruptive properties.

The Joint Research Centre's Endocrine Disruptor Expert Advisory Group (JRC ED EAG) investigated the issue of determining thresholds for endocrine disruptors and *"considered that thresholds of adversity are likely to exist for EDs but may be very low for individual EDs, depending on the mode of action, potency and toxicokinetics and that these thresholds may be particularly low during foetal development (i.e. critical windows of sensitivity) due to the immarturity of homeostatic mechanisms, the immature metabolism as well as the absence of some endocrine axes during sensitive periods of foetal life comparted to adult life stages. For these reasons some experts considered it uncertain whether there is a threshold during development. Several experts also expressed the view that, although thresholds may exist, it might be difficult to estimate with any confidence the biological thresholds of adversity based on currently available standard tests. In addition, small changes in hormone levels during development could have permanent serious consequences for the organism. Other experts expressed the view that a threshold of adversity for EDs might be lower in the developing organism than in the adult and the nature of the effect might be different (severe, permanent change in the foetus versus a less severe effect in the adult), but a threshold of adversity must exist and can be estimated with appropriate testing (involving developmental exposure)."* (Munn and Goumenou 2013).

Participants of an expert meeting at the Chief Scientific Advisor to the Commissions President in October 2013 concluded that it is possible thresholds do not exist; the reason of the uncertainty is the limitation of the experimental constraints and the understanding of the biology (European Commission 2013). Further, that it is not possible to define thresholds only by experiments in whole organisms due to lack of sensitivity, and that the existence of thresholds must be defined by understanding better the mechanisms of action in a quantitative systems approach (European Commission 2013).

The uncertainty regarding determining thresholds is a key uncertainty to the assessment of endocrine disruptors. Other uncertainties pertaining to the determination of thresholds for endocrine disruptors were also highlighted by both the JRC ED EAG and at the meeting of experts in October 2013 (Munn and Goumenou 2014, European Commission 2013) and considered relevant for determining the relevant route for authorisation of EDs under REACH

(European Commission 2014). These include current test methods limitations on sensitivity as well a lack of a study involving exposure through the whole life cycle of a mammal, from contraception to old age and lack of adverse effects which may be induced during fetal or pubertal developments but emerge later in life like certain cancers and effects on reproductive senescence. Further, the immature metabolism, the absence or immaturity of the homeostatic mechanisms of as well as the absence of fully developed endocrine axes during fetal life could mean that a small change in hormone levels during development could have permanent serious consequences for the organism. In addition, puberty, pregnancy and menopause also constitute vulnerable life stages.

The issue of non-monotonic dose response relationships and low dose effects are also relevant, however not necessarily specific to endocrine disruptors, as well as uncertainties related to inter and intra-species extrapolation (European Commission 2014).

Thus, as the existence of a threshold has not yet been assessed and documented for DEHP, DBP, DIBP and BBP this leads to uncertainties regarding the appropriateness of the derived DNELs.

As an illustrative example, the dose-response curves for the effects of DEHP on anogenital distance (AGD) and the frequency of mild external genital malformations in Christiansen et al. (2010) shows significant effects at all doses studied (Figure B4).

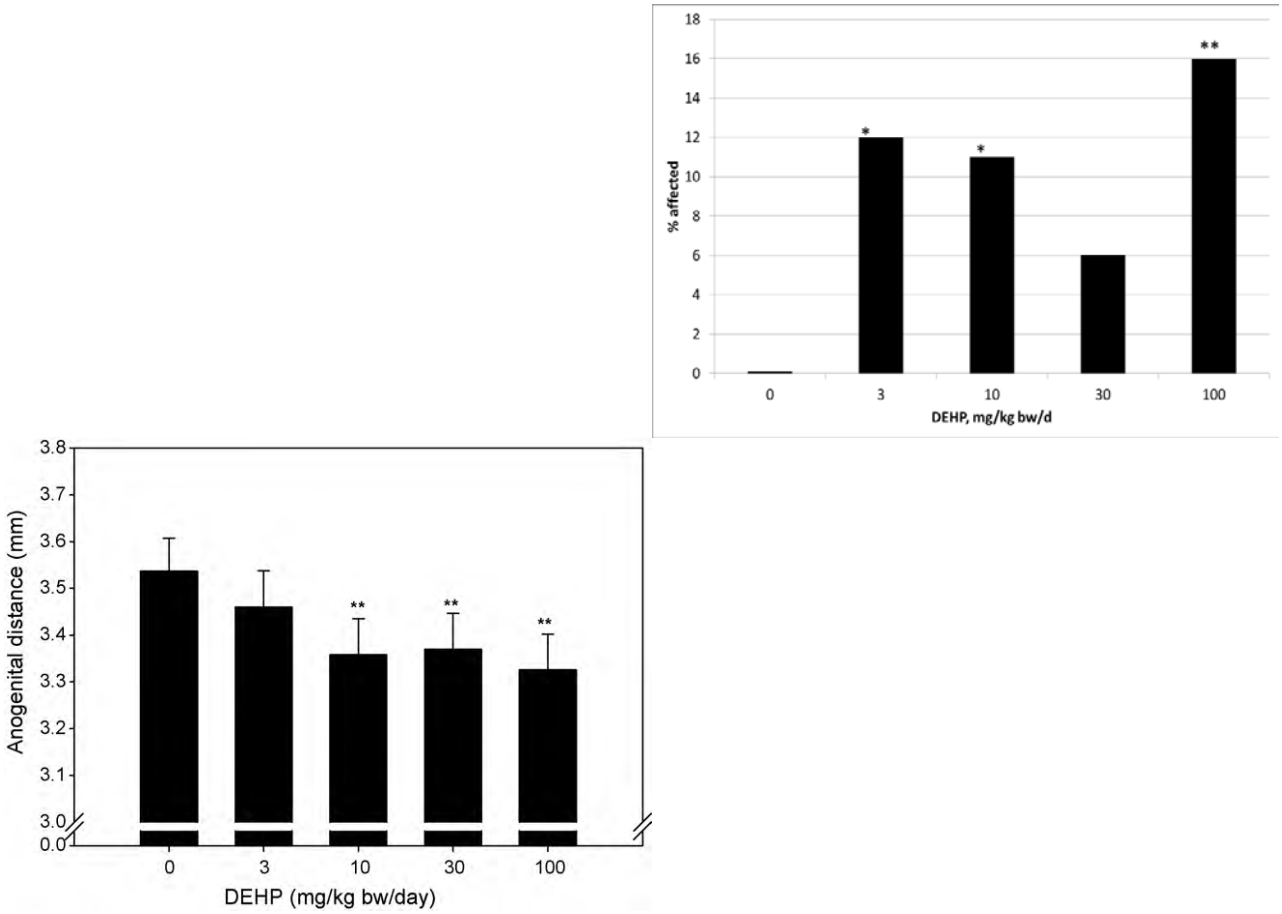

Figure B4 Mean anogenital distance (AGD) on PND 1 (left) and mild external dysgenesis on PND 16 (right) in male rat offspring of dams administered corn oil (control), 3, 10, 30 or 100 mg/kg-d DEHP from GD 7 to PND 16. Least square means + SEM are shown for AGD and the data are corrected for body weight and litter effect. Frequency of affected male offspring is shown for mild external dysgenesis ; *Indicates $p < 0.05$; **Indicates $p < 0.01$.

B.5. Human health hazard assessment of physico-chemical properties

Not relevant for this dossier.

B.6. Environmental hazard assessment

This report is targeted to effects on human health. A summary of the environmental hazards and risks described on the EU Risk Assessment Reports for DEHP, DBP and BBP is presented in Appendix B2. For DIBP no summary is presented as no EU RAR is available.

B.7. PBT and vPvB assessment

Not relevant for this dossier.

B.8. Exposure assessment

B.8.1. General discussion on releases and exposure

B.8.1.1. Summary of the existing legal requirements

DEHP, DBP and BBP are restricted in toys and childcare articles in concentrations above 0.1% according to REACH Regulation Annex XVII, entry 51.

CMR substances are restricted in toys from July 2013 in concentrations above the generic or specific classification limit (Directive 2009/48/EC on the safety of toys). The relevant concentration is currently the specific concentration limit of 5% in the case of DIBP, since mixtures containing 5% DIBP or more are classified as toxic for reproduction category 2. For DEHP, BBP and DBP the relevant concentration would be the generic limit for reproductive toxicity category 1B, namely 0.3%[37]. However, with the 9th amendment to CLP (adopted at the REACH committee on 4th February 2016) the specific concentration limit for DIBP for reproductive toxicity is repealed. Therefore, in the future (latest 18 months after publication), the concentration limit for DIBP will be the generic concentration limits (0.3% for Repr. 1B and 3% for Repr. 2).

All four phthalates are included in the candidate list, resulting in the obligation for suppliers to provide safety information to the recipient of the article (including, on request, consumers) if the content of one or more of the four phthalates is above 0.1%. All four phthalates are furthermore included in Annex XIV of the REACH Regulation and are thus subject to the authorisation process. However, the authorisation process does not cover placing on the market of articles containing the phthalates and therefore does not cover imported articles. Imported articles could therefore still contain the four phthalates.

In the RoHS Directive Bis(2-ethylhexyl) phthalate (DEHP), Butyl benzyl phthalate (BBP), Dibutyl phthalate (DBP) and Diisobutyl phthalate (DIBP) will be restricted in electrical and electronic products from 22 July 2019 . The limit value will be 0.1 %. However, the restriction of DEHP, BBP, DBP and DIBP in electrical and electronic products shall apply to medical devices, including in vitro medical devices, and monitoring and control instruments, including industrial monitoring and control instruments, from 22 July 2021.

The restriction of DEHP, BBP, DBP and DIBP will not apply to cables or spare parts for the repair, the reuse, the updating of functionalities or upgrading of capacity of EEE placed on the market before 22 July 2019, and of medical devices, including in vitro medical devices, and monitoring and control instruments, including industrial monitoring and control instruments, placed on the market before 22 July 2021.

The restriction of DEHP, BBP and DBP under the RoHS Directive will not apply to toys which are already subject to the restriction of DEHP, BBP and DBP through entry 51 of Annex XVII to Regulation (EC) No 1907/2006.

[37] The restriction in toys and childcare articles according to restriction entry 51 in REACH Annex XVII limits concentrations above 0.1% and is thus the applicable limit.

Within the RoHS Directive it is possible to apply for an exemptions if no alternatives are available.

As far as food packaging is concerned, the use of DEHP in food contact materials is already restricted under Commission Regulation 10/2011 on plastic materials and articles intended to come into contact with food. DEHP can be used in non-fatty food contact materials for repeated use which includes such items as tubes and conveyer belts, provided the migration of the plasticiser does not exceed the Specific Migration Limit (SML) of 1.5 mg/kg food.

B.8.1.2. Summary of the effectiveness of the implemented operational conditions and risk management measures

Phthalates are found primarily in PVC as softeners but can also be found in other plastics in low concentrations. Phthalates can also be found in e.g. dispersions, paints and varnishes, as emulsifiers, repellents and carrier fluids in biocides, etc.

Several reports indicate that single articles (containing very high concentrations of phthalates) e.g., single types of plastic sandals (Danish EPA 2010c) and single erasers (Danish EPA 2007) will contribute to a very high exposure to one or more of the four phthalates. Individuals are exposed to these phthalates through inhalation (phthalates emitted from wall paper, floor covering and other sources), ingestion (via e.g., food, toddlers suckling on plastic materials), and dermal exposure for their whole lifetime since the intrauterine life.

As shown in the rest of this section the exposure to these four phthalates raises concern with risk characterisation ratios (RCRs) well above 1. This implies that the existing and already implemented operational conditions and risk management measures are insufficient and the exposure from indoor environment, food and contact with articles poses a risk.

B.8.2. Human exposure

The general population is exposed to phthalates via different routes and different sources. Oral exposure occurs from ingestion of food and dust, and from mouthing of articles. Exposure also occurs from inhalation of air and dust and from dermal contact with articles and dust. The main sources of exposure are considered to be food, indoor environment and direct contact with articles. Additionally, medical devices may contribute to exposure of the general population.

Occupational exposure has also been considered.

Urinary biomonitoring data has been used to estimate the exposure to phthalates. Biomonitoring data indicate the level of phthalates that a population in general has been exposed to. Such data are therefore used to show the exposure level of phthalates, but biomonitoring studies have limited capability in identifying the sources of exposure and thus modelling was also performed to better characterise the contributing sources of exposure.

Although the foetus is thought to be more sensitive to the effects of the four phthalates, neonates, infants and children are still considered to be among the sensitive population

because their reproductive system is still developing (David 2006; Foster et al. 2001; den Hond and Schoeters 2005; Jacobson-Dickman and Lee 2009). Thus both exposure to women and children is assessed.

B.8.3. Exposure based on biomonitoring data

B.8.3.1. Background to biomonitoring

Exposure estimates from urinary biomonitoring data reflect all sources of exposure. Provided that the number of samples is sufficiently large, the sample taken is representative to the population studies and the quality of the study is good, then exposure estimates based on biomonitoring data can be considered to provide more exact exposure estimates compared to modelled estimates. In isolation, biomonitoring does not give information on the source or route of exposure. However, in combination with duplicate diet studies, questionnaires on exposure-relevant behaviour, professional occupation and socio-demographic data, and exposure modelling, it is possible to interpret the overall exposure as estimated from biomonitoring data in terms of the contribution of different sources to the overall burden of exposure.

For the phthalates DEHP, DBP, BBP and DIBP a lot of urinary biomonitoring data is available. The current assessment in particular relies on the urinary biomonitoring data generated in the EU-wide DEMOCOPHES project.

B.8.3.1.1. Metabolites

Phthalates are rapidly metabolised and most of the metabolites are nearly completely excreted within 24h after exposure (Wittassek et al. 2001).

The first step in the metabolism of DEHP, DINP, DIDP and DPHP is the formation of short-lived monoesters. The monoester of DEHP, MEHP, is not the best indicator for DEHP exposure as a result of its short half-life of 5 hour (Koch et al. 2005). After a day, only about 3% of MEHP remains to be eliminated (Lorber et al. 2011). The major share of the simple monoester is further metabolised to produce a number of oxidative metabolites (Wittassek et al. 2011). The secondary, oxidised metabolites have half-lives of 10-15 hour (Lorber et al. 2011) and are the main metabolites excreted in human urine (Wittassek et al. 2011). Oxidative metabolism of DEHP is enzyme-mediated and oxidative metabolites cannot result from accidental contamination of samples with DEHP during sampling, storage or analysis (Wittassek et al. 2011). After a day, about 25% of secondary metabolites remain to be excreted (about 2 half-lives have passed).

The fraction of the phthalates excreted in urine (FUE) value from for the sum of MEHP, 5-oxo-MEHP, 5OH-MEHP and 5cx-MEHP is 45.3% within 24h and 47.1% within 48h (Anderson et al. 2011). This suggests that 96% of DEHP is excreted within 24h. Indeed, Anderson et al. (2011) reported that over 90% is excreted in 24h. Anderson et al. (2001) were not able to detect any labelled phthalate metabolites in urine after 24h post dosing.

For DBP, DIBP and BBP the major urinary metabolites are the simple monoesters (about 70% of the oral dose) and the monoesters are normally measured for these phthalates (Wittassek et al. 2011). The half-life of MnBP appears to be in the order of 6 hours (Seckin et al. 2009). No data on half-lives appear to be available in the literature for the metabolites of DIBP or BBP.

B.8.3.1.2. Sampling time and variability in exposure

There are essentially two protocols for collecting urine samples: studies using spot samples and studies using 24 hour samples. In the latter all urine excreted in one day is collected, thus giving the absolute metabolite amounts excreted in one day. The method is however logistically difficult (Wittassek et al. 2011).

Since phthalates are metabolised relatively rapidly there is both a diurnal and a day to day variation in the quantities of metabolites excreted in urine in response to the variation in intakes of these compounds over a 24 hour period. This variability is relevant to the sampling approach taken and to the interpretation of data.

Preau et al. (2010) studied variability of urinary metabolite concentrations of DEHP in eight participants from the metropolitan Atlanta area (USA). The variability of DEHP metabolites in samples from individual volunteers showed a greater within day than between day variability. Significantly higher[38] concentrations were seen in the evening compared to the morning (Preau et al. 2010). Similarly the within person variability was greater (69-83% of total variability explained) than between persons (Preau et al. 2010). However, there is also an important interpersonal variability with DEHP exposure: some individuals showed on average comparatively higher chronic exposures than other individuals. Indeed there is a factor of 127 difference between the geometric mean (GM) values of the lowest exposed participant and the highest exposed participant (n=8) in the study by Preau et al. (2010).

Fromme et al. (2007) measured phthalate metabolite levels over 8 consecutive days in 49 participants (27 female, 23 male) from Munich and the surrounding area in 2005. A significant correlation of creatinine-adjusted urine levels was observed for BBP, DBP and DIBP (but not for DEHP) metabolites measured on several or all days 2-8 compared with day 1. The lowest within person variability was observed for BBP and the highest for DEHP metabolites. No systematic trend in urinary levels was observed across the observation time.

As a result of the variabilities in metabolite concentrations, a single spot urine sample may not be representative for the mean daily concentration (Wittassek et al. 2011). A single low concentration measured in a spot sample may represent a recent low exposure or might reflect a very high exposure the previous day or two (Lorber et al. 2011). Zeman et al. (2013) warned that a low concentration could mistakenly be thought to imply a low daily intake when in fact there may have been a high, short intake. It also cannot be excluded that individuals with high exposure to a phthalate will show high exposure over longer periods. It is not certain that a high exposure value was taken at the highest peak of urinary excretion (thus exposure

[38] Preau et al. (2010) reported also that for MEHHP (5OH-MEHP), the urinary concentration were significantly higher for samples collected in the evening (geometric mean = 33.2 µg/l) compared to morning (geometric mean= 18.7 µg/l) or afternoon samples (geometric mean=18.1 µg/l).

might be higher). Of course the reverse is true as well: a peak exposure may not be representative for the central tendency of exposure for the individual. The variabilities might be balanced out in larger spot sample studies where urine samples were collected from different individuals at different times of day rather than at a specific time (Wittassek et al. 2011; Preau et al. 2010).

Following from the observation by Preau et al. (2010) that concentrations of MEHHP (5OH-MEHP) were significantly higher in the evening (33.2 µg/L) compared to the morning (18.7 µg/L), spot samples taken in the morning may systematically underestimate exposure (CHAP 2014). The samples collected in DEMOCOPHES are mostly spot morning urine samples and thus might underestimate exposure to DEHP and possibly also the other four phthalates with a factor of 1.5 (CHAP 2014). However, Frederiksen et al. (2011) report that in their study of children 40-48% of the metabolites were excreted in the first morning void and that this percentage decreased with increasing age. This is in contrast with Preau et al. (2010), although the age difference may be an explaining factor for this difference.

B.8.3.1.3. Extrapolating spot sample urinary excretion data to full day excretion

Spot sample studies normalise urinary metabolite concentrations against creatinine or daily urinary volume reference values to estimate the amount excreted over a full day. The assumption in this extrapolation is that the phthalate metabolite excretion is steady over time (Lorber et al. 2011). However, as discussed above, this assumption is not correct and leads to a first possible uncertainty. Furthermore, a second uncertainty relates to the normalisation method itself as discussed below.

Creatinine is a degradation product from creatinine phosphate which is present in the skeletal muscles. Creatine is fairly constantly converted to creatinine at a rate of about 2% of total body creatine per day (Barr et al. 2005), and this proportionate to the muscle mass (Samra and Abcar 2012). Since individuals vary in the rate that they excrete urine, urinary biomonitoring data are often adjusted to the more constant creatinine excretion rate.

Normalisation of urinary metabolite levels against creatinine introduces some uncertainties related to the potential variability of creatinine excretion rates. Indeed, creatinine excretion is dependent on muscle mass (e.g., typically higher in young men than in women) and activity. There are age, gender, race, BMI, fat-free mass and health related (kidney function, hyperthyroidism, hypertension, and diabetes) variations in rates of creatinine excretion, as well as the time of the day of taking samples (Barr et al. 2005). Also meat consumption and strenuous exercise results in elevated urinary creatinine (Samra and Abcar 2012).

Lorber et al. (2011) warned that “an underprediction of intakes might be frequent and a strong deterrent to the regular use of the creatinine correction approach for DEHP”. The authors however acknowledge that they cannot suggest a viable alternative method. Also Koch et al. (2007) estimated that creatinine adjusted results might underestimate daily DBP and BBP intake. Langer et al. (2014) did not adjust for creatinine, reasoning that creatinine excretion in children depends more on physical development than on urine dilution, and that humans are exposed intermittently (rather than continuously) to DEHP which is also metabolised quickly. Wittassek et al. (2007a) and Koch et al. (2007) found that values for children were on average about two times higher with the volume based-model in comparison with the creatinine-based

model. Data from children and adults in Hartmann et al. (2015) does not suggest a significant difference between the two methods. A comparison of approaches based on DEMOCOPHES data confirms that the volume correction approach results in typically higher exposure estimates (see Table B13): a difference of about a factor 2 was observed for adults, however the difference for children appeared to be small. Wittassek et al. (2007a) and Koch et al. (2007) consider that both approaches must be regarded equivalent.

Of the studies published since January 2011, 6 used the volume based method, 3 the creatinine correction method and one study reported results for both methods (see Table B19-Table B22). Thus the most recent publications showed a tendency to report volume based intake estimates rather than creatinine corrected estimates.

Studies using 24 h collection do not have the correction problem encountered with spot sample collection since the absolute metabolite amounts are available. However, as mentioned before, such studies are logistically difficult, which explains why DEMOCOPHES data and nearly all of the studies in Table B19-Table B22 are based on spot samples.

B.8.3.2. DEMOCOPHES data

The exposure assessment is mainly based on the DEMOCOPHES urinary biomonitoring data from samples taken in the period from September 2011 until February 2012 (FPS 2013). Spot urine samples were collected from mother-child pairs in 16 Member States and Switzerland. The majority of the samples collected were morning samples (99.2% in children and 98.8% in mothers), other samples were spot urine samples collected throughout the day (FPS 2013). The morning samples are not necessarily first morning voids. Children were 6-11 years old. The median age of the mothers was 39 years with a 25th percentile of 35 years and a 75th percentile of 42 years (FPS 2013).

Table B10 Overview of the available urinary biomonitoring data from the DEMOCOPHES project

Country	**Country code**	**Source**
Overall		Den Hond et al. (2015)
Belgium	BE	National report
Switzerland	CH	National report
Cyprus	CY	National report
Czech Republic	CZ	National report; Černá et al. (2015)
Germany	DE	National report
Denmark	DK	National report; Frederiksen et al. (2013)
Spain	ES	National report; Cutanda et al. (2015)
Hungary	HU	National report; Černá et al. (2015); Középesy (2016)
Ireland	IE	National report
Luxembourg	LU	National report; Gutleb (2015)
Poland	PL	National report
Portugal	PT	National report
Romania	RO	National report
Sweden	SE	National report; Larsson et al. (2014)
Slovenia	SI	National report; Horvat (2015)
Slovak Republic	SK	National report; Černá et al. (2015); Jajcaj (2015)
United Kingdom	UK	National report

B.8.3.2.1. Methods

The formula from Frederiksen et al. (2013) using creatinine corrected urinary concentration of metabolites was used to estimate the daily intake (µg/kg bw/day) from the spot samples gathered in the DEMOCOPHES project:

$$DI\left(\frac{\mu g}{kg \times day}\right) = \frac{\left[\left(\frac{UE_{m1crea}\left(\frac{\mu g}{g_{crea}}\right)}{MW_{m1}\left(\frac{\mu g}{\mu mol}\right)}\right) + \left(\frac{UE_{m2crea}\left(\frac{\mu g}{g_{crea}}\right)}{MW_{m2}\left(\frac{\mu g}{\mu mol}\right)}\right) + \cdots\right] \times MW_p\left(\frac{\mu g}{\mu mol}\right) \times CE_{smoothed}\left(\frac{g}{day}\right)}{FUE \times BW(kg)}$$

Where
DI = estimation of daily intake of phthalate diesters;
$CE_{smoothed}$ = 24-h urinary creatinine excretion;
BW = body weight ;
$UE_{m1\ crea}$, $UE_{m2\ crea}$,... = creatinine adjusted urinary concentration of phthalates metabolites;
MW_{m1}, MW_{m2} ,... = molecular masses of each of the respective metabolites;
MW_p = molecular mass of the specific phthalate diester; and
FUE = fraction of the phthalate diester excreted in urine.

Molar urinary excretion fractions (FUE values) for the different phthalate metabolites have been derived in kinetic studies with adult[39] human volunteers. These fractions allow to estimate external oral exposure from urine concentrations.

For **DEHP**, the mean 24h FUE values (molar) from Anderson et al. (2011) were used in the calculations, i.e., **6.2%** for **MEHP**, **10.9%** for **5-oxo-MEHP**, and **14.9%** for **5OH-MEHP**.

Anderson et al. (2001) studied elimination of 255 µg and 510 µg of ^{13}C-DBP in human volunteers. The low dose gave yields of 64% (n=6) and the high dose 73% (n=7). A molar FUE value for DBP of 69% was derived as an average. The higher bound is consistent with a study by Seckin et al. (2009) using a high dose (3.6 mg DBP) resulting in an FUE of 78% (n=17). In a study by Koch et al. (2012) an FUE of 84% was derived with a high dose of 5.38 mg (n=1). In the current report a FUE value for **DBP** of **74%** was assumed as an average between the value from Anderson et al. (2001) and Seckin et al. (2009). The estimate by Koch et al. (2012) was considered less robust since it is based on one volunteer only and a high dose level (exposure levels in humans are several orders of magnitude lower). When deriving a weighted average according to number of volunteers the same FUE value is obtained (74%)[40].

[39] Seckin et al. (2009) also included 4 children in a total population of 17. No specific FUE was reported for the children however.

[40] The weighted average of the values 64% (n=6); 73% (n=7); 78% (n=17); and 84% (n=1) is 74.

For **BBP** an FUE value of **73%** from Anderson et al. (2001) was used.

For DIBP an FUE of 70% derived in a human volunteer study by Koch et al. (2012) with a high dose of 5.001 mg (n=1). Because previously no experimental FUE was available for DIBP and because DBP and DIBP are isomers and thus have the same MW and similar structure, it has often been assumed (e.g. UBA 2011; Fromme et al. 2013; Kasper-Sonnenberg et al. 2014) that the molar FUE is 69% for DIBP, which is equal to the FUE value for DBP from Anderson et al. (2001). In the current report an FUE value of **70%** for **DIBP** is used from Koch et al. (2012).

The FUE values show some intra- and inter-individual variation resulting from variations in metabolism and thus their use leads to some (minor) uncertainty to the estimates from the biomonitoring data. No specific excretion fractions have been established for children. Children aged 6-11 years appeared to excrete a smaller fraction (about 50% less) of the simple monoester of DEHP (MEHP) compared to adults (Koch et al. 2007; Koch and Angerer 2012). Thus, using the FUE of adults for children may for DBP, DIBP and BBP result in an underestimation of exposure since only the monoesters are measured for these phthalates (Koch et al. 2007).

Ideally, the above formula is applied using data from the individual participants. No data on body weight, age, height, creatinine levels, and urinary metabolite levels for individual participants was made available to ECHA by the project members. This leads to some loss of precision or accuracy of the exposure estimates from the biomonitoring data:

- In the current assessment a fixed value was used for "$CE_{smoothed}$" (not the smoothed value), as the data needed for the calculation of $CE_{smoothed}$ (age, body weight and height of each individual participant) was not available to the Dossier Submitter. Values of 1.2 g/d for women and 0.5 g/d for children aged 6-11 were used from Aylward et al. (2009).
- The bodyweight values used are the country-specific median values from the DEMOCOPHES National Reports.
- Since individuals exposed to high levels of DEHP will have high metabolite values in their urine, the current assessment assumed that using the high percentile values of the individual metabolites in the above formula would not result in significant bias in the assessment[41]. It is possible some bias is introduced caused by measurement variation and errors on the individual metabolites and possibly also by the variation in metabolic activity between individuals.

[41] This assumption was to some extent verified. Data from Slovenia was available for the secondary metabolites 5-oxo-MEHP and 5OH-MEHP from individual participants (Horvat 2015). The approach taken in the current assessment resulted in a difference in the intake estimates for mothers of -3.68% for the median intake, 1.79% for the 90th percentile, -1.62% for the 95th percentile and 0.00% for the maximum value when compared to the more accurate estimate using data from individual participants. For children the difference was -3.59% for the median intake, 1.21% for the 90th percentile, 6.03% for the 95th percentile and 3.77% for the maximum value. This confirms that indeed the bias is not significant. For MEHP the verification was not possible and it is not unlikely that differences would be larger since MEHP and oxidative metabolites have a weaker correlation in comparison with the correlation between the oxidative metabolites, probably explained by the different half-lives and possible external contamination of MEHP (Wittassek et al. 2011). This assumption has been evaluated by estimating the intake of DEHP on the basis of only the oxidative metabolites 5-oxo-MEHP and 5OH-MEHP. The resulting in take estimates of DEHP were consistently higher (on average 10.4%) compared to those based on the three metabolites MEHP, 5-oxo-MEHP and 5OH-MEHP. These results confirm that adding the high percentile levels is clearly not leading to overestimates of the intake of DEHP.

Table B11 phthalate metabolites used as biomarkers and their FUE

Substance	Chemical name of metabolite	Abbreviation used in this report	Synonyms	FUE
DEHP	mono-(2-ethylhexyl) phthalate	MEHP		6.2% (Anderson et al. 2011)
	mono-(2-ethyl-5-oxohexyl)phthalate	5-oxo-MEHP	MEOHP	10.9% (Anderson et al. 2011)
	mono-(2-ethyl-5-hydroxyhexyl)phthalate	5OH-MEHP	MEHHP	14.9% (Anderson et al. 2011)
	mono(2-ethyl-5-carboxypentyl) phthalate	5cx-MEPP	MECPP	Generally not measured in DEMOCOPHES, therefore not used in the intake estimates
	mono(2-carboxy-methylhexyl) phthalate	2cx-MMHP		Not measured in DEMOCOPHES
DBP	mono-n-butyl phthalate	MnBP		74% (see above)
DiBP	mono-isobutyl phthalate	MiBP		70% (Koch et al. 2012)
BBP	mono-benzyl phthalate	MBzP		73% (Anderson et al. 2001)

B.8.3.2.2. Estimation of the reasonable worst case

The reasonable worst case exposure estimate should reflect a reasonable foreseeable way of behaviour or reasonable foreseeable circumstances. If it is demonstrated that individuals with a reasonable foreseeable way of behaviour or reasonable foreseeable circumstances are not sufficiently protected (i.e., their exposure is above the DNEL), the individuals would be at risk. It should be considered that certain sub-populations may be exposed differently from others.

When measurement data is used to estimate the reasonable worst case, a specific percentile of the exposure distribution needs to be selected. When deriving occupational exposure estimates it is customary to use the 90th percentile of an exposure distribution for chronic effects as a measure for the reasonable worst case (for acute effects 95th percentiles can be used, see ECHA Guidance R14). For consumers, the guidance does not specify a percentile but the 95th percentile of measured data may be used to estimate the reasonable worst case of exposure. In some specific cases (depending on the substance, the exposure pattern, accuracy of the measurements, etc.), it may be argued that the reasonable worst case of exposure corresponds to a higher percentile than the 95th percentile (e.g., the 99th percentile, see RIVM 2014).

In this case, the measurements are urinary metabolite levels of phthalates from the DEMOCOPHES project. The following elements were considered in selecting the appropriate percentile to estimate the reasonable worst case of exposure:

- As there are rather few data per country, the 99th percentile would effectively correspond to the one or two highest values. Maxima from measurements are not normally used in risk assessment and might be outliers resulting from analytical and methodological errors or might result from non-representative exposure situations.

- Peak exposures may be indicative of certain reasonable foreseeable ways of behaviour or reasonable foreseeable circumstances that may be typical for some individuals or sub-populations.
- No biomonitoring is available for infants.
- Even a short elevated exposure level within the 'critical windows of exposure'[42] may be sufficient to cause adverse effects on the developing foetus which makes peak exposures particularly relevant in the case of the four phthalates (as opposed to substances where the critical effects are caused following chronic exposure).
- It can be questioned whether the sample sizes in the countries participating in the DEMOCOPHES are sufficiently large to even out some of the variability caused by taking spot samples (see discussion above). As a result of this, there are relatively high uncertainties to whether the actual 95th percentile exposure in the entire population is lower or higher.
- Further uncertainties result from the methods used in the current assessment[43].

Based on the above considerations, it is considered appropriate to use the 95th percentile urinary exposure levels from DEMOCOPHES as an estimate of the reasonable worst case of exposure. There are however indications that the selection of a 95th percentile may lead to underestimation of the reasonable worst case exposure level.

B.8.3.2.3. Results

The creatinine corrected urinary concentrations from DEMOCOPHES are reported in Table B12. The intake estimates (µg/kg bw/day) from DEMOCOPHES based on creatinine corrected urinary metabolite concentrations are reported in Table B13. Published intake estimates (µg/kg bw/day) from DEMOCOPHES for DK are reported in Table B15-Table B18.

[42] The critical time period for inducing malformations is thought to be gestation days 15 to 19 in rats (ECHA 2013d; Welsh et al. 2008). The corresponding programming window in humans might be during the 8th and 14th weeks of gestation (Welsh et al. 2008). However, in humans testosterone peaks also at other times during pregnancy, and postnatally around 3 months of age and at puberty. This indicates there may be more than one critical time period for exposure. Exposure within these periods may not need to be chronic to result in effects, even a short elevated level may be sufficient to cause adverse effects on the developing foetus (Wittassek et al. 2007a).

[43] No body weights from the individual participants were available to the Dossier Submitter. Instead, the bodyweight values used are the country-specific median values from the DEMOCOPHES National Reports. Since the resulting exposure estimates (mg/kg bw/day) are not derived using the actual bodyweight of the participants, the 95th percentiles might be over or underestimated. Similarly, the creatinine excretion correction was done using a average excretion value. It was not possible to refine the value further and calculate a $CE_{smoothened}$ since no body weights, age and height from the individual participants were available to the Dossier Submitter.

Table B12 Creatinine corrected urinary concentrations from DEMOCOPHES

				creatinine corrected urinary concentrations					
				DEHP			DBP	BBP	DiBP
Country	N	Population		MEHP	5OH	5oxo	MnBP	MBzP	MiBP
BE	125	Mother	P50	1.94	9.48	6.70	25.34	5.52	29.50
			P95	6.56	31.54	21.59	79.31	19.82	142.68
	125	Child	P50	1.94	16.06	11.24	33.68	8.01	46.42
			P95	10.13	92.59	64.81	99.35	31.94	278.89
CH	117	Mother	P50	2.30	7.70	4.60	14.40	3.30	15.00
			P95	11.70	56.20	6.20	57.40	13.60	48.10
	119	Child	P50	2.00	13.30	11.90	20.30	3.90	19.40
			P95	6.40	52.40	37.50	60.90	26.10	62.70
CY	59	Mother	P50	2.89	5.35	4.60	14.37	1.97	44.26
			P95	27.41	106.10	53.93	41.28	9.38	106.15
	60	Child	P50	2.35	10.67	7.93	20.85	3.36	53.39
			P95	10.62	69.22	35.35	55.33	15.16	124.70
CZ	117	Mother	P50	3.20	17.77	11.20	57.70	4.19	NA
			P95	12.59	55.82	34.03	157.00	41.60	NA
	120	Child	P50	3.27	36.27	23.82	109.80	6.95	NA
			P95	11.39	116.06	74.19	315.47	53.48	NA
DE	116	Mother	P50	1.98	10.00	6.56	28.33	3.96	21.24
			P95	8.14	24.20	18.35	75.38	17.91	58.97
	120	Child	P50	2.04	19.07	12.85	40.86	5.22	35.20
			P95	8.67	53.48	38.44	125.30	34.99	99.22
DK	143	Mother	P50	1.70	13.00	6.20	21.00	4.10	37.00
			P95	7.50	43.00	19.00	41.00	17.00	100.00
	142	Child	P50	2.20	22.00	11.00	33.00	7.40	58.00
			P95	8.30	70.00	33.00	72.00	36.00	165.00
ES	118	Mother	P50	6.91	19.43	12.57	30.44	7.34	36.06
			P95	21.22	50.69	34.81	68.82	29.76	77.03
	119	Child	P50	7.28	37.05	23.54	46.10	13.13	54.34
			P95	16.35	98.08	58.33	213.75	50.10	237.21
HU	115	Mother	P50	3.45	14.98	9.90	32.60	3.39	NA
			P95	13.81	57.70	37.33	101.92	17.17	NA
	117	Child	P50	3.24	27.03	19.58	52.70	6.14	NA
			P95	15.10	99.35	70.10	162.11	28.12	NA
IE	120	Mother	P50	2.79	16.17	8.42	18.40	2.79	22.20
			P95	11.11	50.53	25.93	51.98	18.04	93.71
	120	Child	P50	3.03	30.85	16.96	25.69	4.73	39.15
			P95	12.37	94.23	50.61	66.06	21.70	139.97
LU	58	Mother	P50	1.46	7.69	4.85	19.21	3.22	19.54
			P95	8.78	32.25	23.22	45.36	13.26	69.39
	60	Child	P50	1.59	12.32	8.76	26.34	4.15	35.41
			P95	4.03	31.16	18.12	57.85	19.98	193.76
PL	119	Mother	P50	4.33	20.38	12.62	44.00	3.69	45.78
			P95	24.19	85.66	50.00	178.95	23.08	180.00
	115	Child	P50	3.79	39.09	25.00	78.29	8.85	101.47
			P95	12.77	143.66	100.69	277.78	60.66	349.12
PT	117	Mother	P50	3.52	16.91	9.08	19.08	4.64	24.10
			P95	17.40	73.29	47.55	44.47	14.20	70.42
	116	Child	P50	3.02	24.29	14.53	29.75	7.45	36.32
			P95	11.96	81.83	38.32	82.52	39.15	118.33
RO	117	Mother	P50	4.41	20.21	12.67	21.31	2.15	28.27
			P95	39.76	205.16	168.34	50.35	9.60	78.02
	119	Child	P50	3.52	35.34	21.98	39.37	3.71	47.33
			P95	26.34	269.58	132.97	140.71	19.38	170.89
SE	96	Mother	P50	2.42	12.96	7.34	58.15	11.06	NA
			P95	9.11	39.00	28.54	161.19	74.16	NA
	97	Child	P50	3.19	26.69	17.74	83.24	22.37	NA
			P95	11.47	93.50	60.54	236.58	96.58	NA
SI	120	Mother	P50	2.98	12.88	6.73	17.90	3.79	NA
			P95	11.20	42.32	22.99	86.70	16.10	NA
	120	Child	P50	2.04	22.62	13.40	31.80	6.32	NA
			P95	6.44	61.91	36.21	102.00	28.60	NA
SK	125	Mother	P50	3.33	17.79	11.61	59.75	3.73	NA
			P95	11.70	48.18	31.98	170.28	14.14	NA
	127	Child	P50	3.15	40.73	26.58	95.64	6.46	NA
			P95	11.54	110.37	80.66	264.37	32.18	NA
UK	21	Mother	P50	1.23	7.89	4.61	14.33	2.01	14.96
			P95	4.08	20.47	12.42	32.44	4.82	70.71
	21	Child	P50	1.31	21.46	12.36	25.04	3.67	25.05
			P95	9.96	39.29	25.54	66.41	21.61	75.67

NA = not available

Table B13 Intake estimates (µg/kg bw/day) from DEMOCOPHES based on creatinine corrected urinary metabolite concentrations

				intake			
				DEHP	DBP	BBP	DiBP
Country	N	Population		µg/kg/d	µg/kg/d	µg/kg/d	µg/kg/d
BE	125	Mother	P50	1.49	0.84	0.18	1.04
			P95	4.92	2.64	0.65	5.02
	125	Child	P50	2.11	0.98	0.23	1.43
			P95	12.06	2.90	0.92	8.60
CH	117	Mother	P50	1.15	0.46	0.10	0.50
			P95	5.83	1.82	0.43	1.61
	119	Child	P50	2.11	0.64	0.12	0.64
			P95	7.45	1.91	0.81	2.08
CY	59	Mother	P50	1.03	0.46	0.06	1.51
			P95	14.99	1.33	0.30	3.62
	60	Child	P50	1.42	0.57	0.09	1.54
			P95	7.77	1.51	0.41	3.60
CZ	117	Mother	P50	2.53	1.83	0.13	NA
			P95	8.05	4.98	1.30	NA
	120	Child	P50	4.41	3.10	0.19	NA
			P95	14.03	8.90	1.49	NA
DE	116	Mother	P50	1.39	0.86	0.12	0.68
			P95	3.82	2.28	0.54	1.89
	120	Child	P50	2.45	1.19	0.15	1.09
			P95	7.26	3.66	1.01	3.06
DK	143	Mother	P50	1.61	0.66	0.13	1.22
			P95	5.37	1.28	0.52	3.30
	142	Child	P50	2.84	0.93	0.21	1.73
			P95	7.75	2.03	1.00	4.92
ES	118	Mother	P50	3.17	1.00	0.24	1.25
			P95	8.70	2.25	0.96	2.67
	119	Child	P50	4.74	1.30	0.37	1.62
			P95	12.05	6.03	1.39	7.07
HU	115	Mother	P50	2.21	1.03	0.11	0.00
			P95	8.49	3.21	0.53	0.00
	117	Child	P50	3.47	1.49	0.17	0.00
			P95	12.86	4.57	0.78	0.00
IE	120	Mother	P50	2.05	0.56	0.08	0.71
			P95	6.58	1.58	0.54	3.00
	120	Child	P50	3.32	0.68	0.12	1.09
			P95	10.27	1.75	0.57	3.91
LU	58	Mother	P50	1.08	0.60	0.10	0.65
			P95	4.98	1.42	0.41	2.29
	60	Child	P50	1.63	0.77	0.12	1.09
			P95	3.84	1.69	0.58	5.98
PL	119	Mother	P50	2.89	1.37	0.11	1.51
			P95	12.39	5.59	0.71	5.94
	115	Child	P50	4.57	2.14	0.24	2.93
			P95	17.31	7.58	1.63	10.07
PT	117	Mother	P50	2.47	0.65	0.15	0.86
			P95	11.59	1.51	0.47	2.52
	116	Child	P50	2.82	0.81	0.20	1.05
			P95	8.91	2.25	1.05	3.41
RO	117	Mother	P50	3.13	0.72	0.07	1.01
			P95	34.60	1.70	0.32	2.79
	119	Child	P50	4.23	1.11	0.10	1.41
			P95	29.85	3.97	0.54	5.10
SE	96	Mother	P50	1.73	1.79	0.34	NA
			P95	5.84	4.96	2.25	NA
	97	Child	P50	3.21	2.27	0.60	NA
			P95	11.16	6.46	2.60	NA
SI	120	Mother	P50	NA	0.56	0.12	NA
			P95	NA	2.71	0.50	NA
	120	Child	P50	NA	0.84	0.16	NA
			P95	NA	2.70	0.75	NA
SK	125	Mother	P50	2.53	1.87	0.11	NA
			P95	7.11	5.32	0.44	NA
	127	Child	P50	4.90	2.70	0.18	NA
			P95	14.10	7.46	0.90	NA
UK	21	Mother	P50	1.00	0.42	0.06	0.47
			P95	2.69	0.95	0.14	2.20
	21	Child	P50	2.53	0.73	0.11	0.77
			P95	5.41	1.94	0.62	2.33

NA = not available

Overall unweighted[44] creatinine corrected urinary concentrations from DEMOCOPHES are presented in Table B14 and are taken from Table S3 in Den Hond et al. (2015). The table also presents intake estimates based on these values. It is worth mentioning that these values include data from all 17 participating countries to the study (thus including non-EU countries and the UK).

[44] *Additionally, Den Hond et al. (2015) presented weighted GM and 90th percentile values (µ/l) in Table 2 of the publication. The values were derived by weighting all countries equally (assuming n = 120), with the exception of LU and CY who received half the weight (n = 60). It appears however that the results from the UK were also weighted with 120 which seems at par with the weighting approach (n = 21 is inflated to 120). The small sample from the UK can be considered non-representative to the UK population.

Table B14 Overall unweighted creatinine corrected urinary concentrations from DEMOCOPHES and corresponding intake estimates.

		DEHP		DBP		BBP		DIBP	
		Urinary concentration* (µg/gC)	Intake (µg/kg bw/day)	Urinary concentration MnBP (µg/gC)	Intake (µg/kg bw/day)	Urinary concentration MBzP (µg/gC)	Intake (µg/kg bw/day)	Urinary concentration MiBP (µg/gC)	Intake (µg/kg bw/day)
Mothers	min	1.8	0.1	0.9	0.0	0.1	0.0	1.0	0.0
	GM	26.8	2.1	22.3	0.7	4.2	0.1	28.3	0.9
	P50	25.5	2.0	22.0	0.7	4.0	0.1	27.7	0.9
	P90	70.2	5.6	51.6	1.6	13.7	0.4	69.4	2.3
	P95	104.3	8.3	67.6	2.1	20.8	0.7	94.3	3.2
	max	1 540.0	122.5	2 059.0	65.3	433.0	13.6	347.0	11.6
Children	min	2.0	0.1	0.8	0.0	0.2	0.0	1.7	0.1
	GM	46.8	3.3	34.6	1.0	7.1	0.2	45.9	1.4
	P50	45.5	3.2	33.9	1.0	6.6	0.2	44.8	1.3
	P90	120.3	8.5	88.1	2.5	26.0	0.7	125.9	3.8
	P95	170.1	12.0	131.3	3.7	42.2	1.2	167.6	5.0
	max	3 616.0	255.7	883.0	24.9	603.0	16.8	1 634.0	48.7

*Sum metabolites MEHP, 5OH-MEHP and 5oxo-MEHP

B.8.3.2.4. Comparison with published exposure estimates from DEMOCOPHES

Published intake estimates based on DEMOCOPHES urinary biomonitoring data are available for Denmark. The intake levels for the Danish population of children and mothers estimated with our methodology differ slightly from those estimated on the basis of the same DEMOCOPHES samples as published by Frederiksen et al. (2013), see Table B15-Table B18 and Table B19-Table B22. As discussed in section “Methods” above, the current assessment did not use the smoothed value for daily creatinine excretion and body weight was the median value instead of the weight of the individual participants. For DEHP, Frederiksen et al. (2013) used 4 metabolites to estimate the daily intake whereas our estimates used 3 metabolites and we added high percentile values of the DEHP metabolites. We used an FUE of 74% instead of 84% for DBP and 70% instead of 71% for DIBP (which should lead to slightly higher intake estimates compared with Frederiksen et al. 2013). Moreover, we did not have separate data for the rural and urban population.

Overall, with the exception of an underestimation of the 95th percentile exposure estimates for DEHP and DIBP for children in our calculations, the exposure estimates are very similar to those reported by Frederiksen et al. (2013)[45].

[45] The median exposure levels for DEHP and DBP are slightly higher than the levels reported by Frederiksen et al. (2013), whereas the median exposure levels for DIBP are lower and for BBP are within the range of the urban and rural values reported by Frederiksen et al. (2013).
The 95th percentile exposure estimate for DEHP for children is lower than the levels reported by Frederiksen et al. (2013) (7.75 µg/kg bw/day compared to 8.13 µg/kg bw/day from urban and 12.54 µg/kg bw/day from rural environments). The same is observed for DIBP (4.92 µg/kg bw/day compared to 7.55 from urban and 7.44 µg/kg bw/day from rural environments). The 95th percentile exposure estimate for DEHP and BBP for mothers is only just slightly higher. The 95th percentile exposure estimates for DBP (mother and children), as well as DIBP (mothers) and BBP (children) are within the range of the urban and rural values reported by Frederiksen et al. (2013).

Table B15 Published intake estimates (µg/kg bw/day) of DEHP based on urinary biomonitoring from the DEMOCOPHES project

Sample year	Country	Age, population size	Metabolite	Median	95th percentile	Max	Basis for estimated intake	Reference
Sep–Dec 2011	DK	**Urban** 6-11 years (n= 74) Mothers (n=75) **Rural** 6-11 years (n= 67) Mothers (n=69)	MEHP 5-oxo-MEHP 5-OH-MEHP 5-cx-MEPP	 2.69 1.56 2.43 1.53	 8.13 5.12 12.54 4.37	 85.6 90.1 21.8 30.7	First morning urine Creatinine method FUE of 45.3% for DEHP metabolites (MEHP; 5-oxo-MEHP 5-OH-MEHP; 5-cx-MEPP)	Frederiksen et al. (2013)

Table B16 Published intake estimates (µg/kg bw/day) of DBP based on urinary biomonitoring from the DEMOCOPHES project

Sample year	Country	Age, population size	Metabolite	Median	95th percentile	Max	Basis for estimated intake	Reference
Sep–Dec 2011	DK	**Urban** 6-11 years (n= 74) Mothers (n=75) **Rural** 6-11 years (n= 67) Mothers (n=69)	MnBP	 0.704 0.490 0.856 0.543	 2.23 0.996 2.03 1.34	 2.49 1.91 7.37 1.81	First morning urine Creatinine method FUE of 84%	Frederiksen et al. (2013)

Table B17 Published intake estimates (µg/kg bw/day) of DIBP based on urinary biomonitoring from the DEMOCOPHES project

Sample year	Country	Age, population size	Metabolite	Median	95th percentile	Max	Basis for estimated intake	Reference
Sep–Dec 2011	DK	**Urban** 6-11 years (n= 74) Mothers (n=75) **Rural** 6-11 years (n= 67) Mothers (n=69)	MiBP	 2.35 1.66 2.75 1.64	 7.55 3.04 7.44 5.21	 9.08 6.28 31.7 6.03	First morning urine Creatinine method FUE of 71%	Frederiksen et al. (2013)

Table B18 Published intake estimates (µg/kg bw/day) of BBP based on urinary biomonitoring from the DEMOCOPHES project

Sample year	Country	Age, population size	Metabolite	Median	95th percentile	Max	Basis for estimated intake	Reference
Sep–Dec 2011	DK	**Urban** 6-11 years (n= 74) Mothers (n=75) **Rural** 6-11 years (n= 67) Mothers (n=69)	MBzP	0.173 0.094 0.227 0.131	1.10 0.432 1.09 0.470	5.49 0.797 2.61 0.900	First morning urine Creatinine method FUE of 73%	Frederiksen et al. (2013)

In Den Hond et al. (2015), data for the for MnBP for the countries CZ, HU, SE and SK in Figure 1 is marked as “no biomarker data available”. Schoeters (2016) clarified that Den Hond et al. (2015) had chosen not to include the data because in some labs no good chromatographic separation was obtained for MnBP and MiBP. It is possible that the levels measured for MnBP are overestimated as they may account also for MiBP. Larsson et al. (2014) however did publish the Swedish data for MnBP and did not report any problems regarding chromatographic separation.

It should be noted that MiBP was not measured in CZ, HU, SE and SK and thus there is no risk of double counting the metabolites in the combined exposure to all four phthalates. Moreover, DBP and DIBP are roughly equipotent and the FUEs are practically equal. Thus if indeed some of the MiBP would’ve been measured under MnBP it may still not sufficiently account for MiBP and thus lead to overall underestimation of risks from combined exposure to all four phthalates. In conclusion, there are no good reasons to omit the data from the assessment, but they need to be interpreted carefully.

Černá et al. (2015) reported geometric mean values for urinary concentrations for DEHP and BBP for the Czech, Hungarian and Slovak populations in the DEMOCOPHES project. Cutanda et al. (2015) reported urinary concentrations for DEHP, DBP, DIBP and BBP for the Spanish populations, and Larsson et al. (2014) for the Swedish population in the DEMOCOPHES project. Since no intake values were presented in these publications, the numerical results are not reported in the tables above.

B.8.3.2.5. Other studies

Table B19-Table B22 below give an overview of the published intake estimates (µg/kg bw/day) of phthalates based on urinary biomonitoring data from Europe. In addition to DEMOCOPHES, there are many other studies reporting urinary metabolite levels of the four phthalates in Germany and Denmark, and some information is available from other countries (Austria, France and Spain). Data from samples taken after 2008, the year of entry into force of the food contact material legislation, are available for infants (DK) and children (AT, DE and DK) and for adults (AT, DK). Older data is presented as well as it might give useful information for trend analysis.

Additional to the studies reported in the tables below, there are further studies (e.g., Becker et al. 2009; Casas et al. 2011; Frederiksen et al. 2014; Geens et al. 2014; Hildenbrand et al. 2009; Tranfo et al. 2012; Sabaredzovic et al. 2015; Strømmen et al. 2016; Koch et al. 2016; Dereumeaux et al. 2016) that reported urinary metabolite concentrations but did not report intake estimates. These studies are therefore not reported here. There are also data for the USA and some data from elsewhere in the world (not reported here).

Table B19 Intake estimates (µg/kg bw/day) of DEHP based on urinary biomonitoring data from Europe (gray-shaded values are used by RAC 2010)

Sample year	Country	Age, population size	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
2011-2013	GR	4 years n=500 Mothers: see below	5-oxo-MEHP 5-OH-MEHP	Median 4.02	17.30	Max 206.92	Spot samples Volume method UV children 0.0224 L/kg bw FUE as in Koch et al. (2004, 2005). MEHP measured but not used in the calculation of intake.	Myridakis et al. (2016)
Nov 2011-May 2012	DE	1.7-6.7 years n=663	5-OH-MEHP	Median 3.26	11.86	Not reported	Spot samples: 1-3 h after day care and for some also first morning samples on Monday before going to the day care Volume method FUEs: 0.149 for 5-OH-MEHP from Anderson et al. (2011)	Fromme et al. (2013b)

Sample year	Country	Age, population size	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
							5-oxo-MEHP was measured but not used to estimate the intake. The authors confirmed that estimates were similar using 5-oxo-MEHP	
2010-2011	AT	6-8 years n=30/31 7-15 years n=214/219 18-64 years n=267/269 65-81 years n=69/71	MEHP 5-oxo-MEHP 5-OH-MEHP 5-cx-MEPP	Median 1: 3.3 2: 3.3 1: 1.5 2: 1.3 1: 0.75 2: 0.53 1: 0.91 2: 0.86	 1: 7.7 2: 15 1: 6.9 2: 7.2 1: 3.2 2: 2.2 1: 8.7 2: 8.5	 1: 0.26- 9.1 2: 0.54-16 1: 0.19-16 2: 0.13-21 1: 0.0-15 2: 0.0-10 1: 0.06-17 2: 0.14-14	Spot samples (before midday) 1. volume model 2. creatinine model FUEs: MEHP= 0.062; 5-oxo-MEHP= 0.109: 5-OH-MEHP = 0.149 (5-cx-MEPP appears not to have been used for deriving intake estimates)	Hartmann et al. (2015)
March 2009 – June 2011	GR	2 years n=239 Mothers: see below	5-oxo-MEHP 5-OH-MEHP	Median 4.0	 21.6	Max 69.6	Spot samples Volume method UV children 0.0224 L/kg bw FUE as in Koch et al. (2004, 2005). MEHP measured but not used in the calculation of intake.	Myridakis et al. (2015)
Nov 2009-Oct 2010	DE	8-10 years n= 465	MEHP 5-oxo-MEHP 5-OH-MEHP 5-cx-MEPP	Median 1.31	 4.31	Max 79.3	First morning urine Volume method FUE for DEHP of 0.627 for 4 metabolites (Koch et al. 2005)	Kasper-Sonnenberg et al. (2014)
Oct 2009 – Jan 2010	DE	15-21 months n= 25	MEHP 5-oxo-MEHP 5-OH-MEHP 2-cx-MMHP	Median 2.6*/4.9**	 6.3*/20.6**	Max 11.4*/26.9**	Morning urine on 7 consecutive days Volume method	Fromme et al. (2013a)

Sample year	Country	Age, population size	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
			5-cx-MEPP				FUE: 5-OH-MEHP = 0.149 (Anderson et al. 2011). Unclear if also other metabolites were used in the estimate of intake. *based on median from 7 sampling days for each subject **based on 95th p from 7 sampling days for each subject	
April-Sep 2008	DK	Men 18-22 years n=33	MEHP 5-oxo-MEHP 5-OH-MEHP 5-cx-MEPP	Median 2.9; 3.6; 2.9	35.2;8.0; 8.3	0.65-97.7; 1.1-10.0; 0.73-12.1	24h sampling: 3 days with intervals of 40-46 days FUE of 45.3% according to Anderson et al. (2011)	Kranich et al. (2014)
Aug 2008 – April 2009	DK	3-6 years n= 441	MEHP 5-oxo-MEHP 5-OH-MEHP 5-cx-MEPP	Median 4.42	16.9	0.38-533.3	Morning urine Volume method FUE: MEHP = 0.059; 5-oxo-MEHP = 0.15; 5-OH-MEHP = 0.233; 5-cx-MEPP = 0.185	Bekö et al. (2013)
2008	DK	Mothers n=52 Infants (1-5 months) n=47	MEHP 5-oxo-MEHP 5-OH-MEHP 2-cx-MMHP 5-cx-MEPP	Median 5.7 Infants: lower than for mothers	23.3 Infants: lower than for mothers		Spot samples Mothers: at 7 time points (34th-37th week of pregnancy, before delivery and 1, 2, 3, 4 and 5 months after delivery) Infants: At 1, 2, 3, 4 and 5 months after delivery Volume method UV infants 0.044 l/kg UV mothers 0.02 l/kg FUE as in Fromme et al. (2013).	Völkel et al. (2014)

Sample year	Country	Age, population size	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
May 2007 – May 2008	GR	Mothers n=239 Children: see above	5-oxo-MEHP 5-OH-MEHP	Median 4.4	 25.6	Max 1015.0	Spot samples At 10th – 13th week of gestation Volume method UV mothers 2L FUE as in Koch et al. (2004, 2005). MEHP measured but not used in the calculation of intake.	Myridakis et al. (2015)
Nov 2007	DK	 6-10 years (♂/♀) n=24/25 17- 21 years (♂/♀) n=14/11 Total n=103	MEHP 5-oxo-MEHP 5OH-MEHP 2cx-MMHP	Median 5.67/5.37 (♂/♀) 2.96/1.74 (♂/♀) 4.04	 44.2/10.8 (♂/♀) Not available 10.7	Max 53/11 10.3/ 3.79 52.9	24h sampling 2 consecutive morning urine + pool of all urine in between FUE according to Koch et al. (2007)/ Wittassek et al. (2007)	Frederiksen et al. (2011)
Oct 2007	FR	 Pregnant women n=279	MEHP 5-oxo-MEHP 5-OH-MEHP 5-cx-MEPP 2cx-MMHP	Median 5.8	 65.1		Spot samples Creatinine corrected FUEs: MEHP = 0.059; 5-oxo-MEHP = 0.15; 5-OH-MEHP = 0.233; 5-cx-MEPP = 0.185; 2cx-MMHP = 0.042	Zeman et al. (2013)
Febr/March 2007	DE	 children age 5-6 n = 108	MEHP 5-oxo-MEHP 5OH-MEHP 5cx-MEPP 2cx-MMHP	Median 4.5	 18.0	Max 44.5	Spot Creatinine corrected FUE: 0.669 for 5 metabolites (Koch et al. 2005)	Koch et al. (2011)
Autumn 2005	DE	 boys 5–8 years	MEHP MEHHP	Median 2.27		0.78-12.73	volume based	UBA (2011)

Sample year	Country	Age, population size	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
		n= 10	MEOHP MECPP 2cx-MMHP		7.02 (90th percentile)			
January 2003	DE	<7 years n=36	MEHP 5-oxo-MEHP 5OH-MEHP	GM 1.0 using MEHP 3.5 using 5OH-MEHP 3.8 using 5-oxo-MEHP	3.3 using MEHP 7.1 using 5OH-MEHP 7.4 using 5-oxo-MEHP		First morning urine Creatinine corrected: creatinine clearance rate (CE) 20 mg/kg adults; 11 mg/kg/day for children and 9.8 mg/kg/day for infants FUEs: 0.13 DEHP (as MEHP); 0.23 as 5OH-MEHP; 0.15 as 5-oxo-MEHP	Koch et al. (2004) reported in Calafat and McKee (2006)
 2001 2003	DE	20-29 years (♂/♀) n=30/30 n=30/29 n=119	MEHP 5-oxo-MEHP 5OH-MEHP 5cx-MEPP 2cx-MMHP	Median 3.3/2.7 (♂/♀) 2.2/2.5 (♂/♀) Overall: 2.7	 6.8/13.6 (♂/♀) 6.4/5.7 (♂/♀) Overall: 6.4	0.82-20.1	24 hr sampling FUE: According to Koch et al. (2005) Not all data shown here: ESB, data reported for 1988-2003	Wittassek et al. (2007b)
April 2002	DE	7-64 years n= 85	5-oxo-MEHP 5OH-MEHP	Median 4.6 * 13.8 **	 17.0 * 52.1 **	58.2-166	First morning urine Creatinine corrected FUE: * according to Koch et al. (2005) ** according to Schmid and Schlatter (1985)	Koch et al. (2003) reported in Wittassek et al. (2007b)
			MEHP 5-oxo-MEHP 5OH-MEHP	GM 2.7 using MEHP 6.5 using 5OH-MEHP 7.4	 7.5 using MEHP 16.3 using 5OH-MEHP 18.9		First morning urine Creatinine clearance rate (CE) 20 mg/kg adults; 11 mg/kg/day for children and 9.8 mg/kg/day for infants	Koch et al. (2003) recalculated in Calafat and McKee (2006)

Sample year	Country	Age, population size	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
				using 5-oxo-MEHP	using 5-oxo-MEHP		FUE:0.13 as MEHP; 0.23 as 5OH-MEHP; 0.15 as 5-oxo-MEHP	
			MEHP 5-oxo-MEHP 5OH-MEHP	GM 10.3 using MEHP 13.5 using 5OH-MEHP 14.2 using 5-oxo-MEHP 13.8 using 5OH-MEHP + 5-oxo-MEHP	 38.3 using MEHP 51.4 using 5OH-MEHP 52.8 using 5-oxo-MEHP 52.1 using 5OH-MEHP + 5-oxo-MEHP		First morning urine FUE from Schmid and Schlatter (1985)	Koch et al. (2003) recalculated in Matsumoto et al. (2008)
March 2001-March 2002 (GerES IV)	DE	children 3-14 years n= 254	MEHP 5-oxo-MEHP 5OH-MEHP	GM 0.7 using MEHP 2.6 using 5OH-MEHP 3.1 using 5-oxo-MEHP	 2.8 using MEHP 10.7 using 5OH-MEHP 11.7 using 5-oxo-MEHP	Not given	Morning urine Creatinine clearance rate (CE) 20 mg/kg adults; 11 mg/kg/day for children and 9.8 mg/kg/day for infants FUE:0.13 as MEHP; 0.23 as 5OH-MEHP; 0.15 as 5-oxo-MEHP	Becker et al. (2004) reported in Calafat and McKee (2006)
March 2001-March 2002	DE	 2-4 years (n=31) 5-6 years (n=46) 7-8 years (n=53) 9-11 years (n=56) 12-14 years (n=53)	MEHP 5-oxo-MEHP 5OH-MEHP	Median 1: 10.7 2: 5.7 1: 10.0 2: 6.1 1: 7.7 2: 4.9 1: 8.1	 1: 45.0 2: 23.4 1: 19.4 2: 14.7 1: 18.3 2: 12.1 1: 25.4	 1: 0.4-409 2: 1.8-140 1: 2.9-43.7 2: 1.3-28.8 1: 2.0-22.3 2: 2.0-19.7 1: 1.5-139	Morning urine 1. volume model 2. creatinine model FUE according to Koch et al. (2005)	Wittassek et al. (2007a)

Sample year	Country	Age, population size	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
		Total (2-14 years) n= 239		2: 3.3 1: 4.8 2: 2.7 1: 7.8 2: 4.3	2: 13.9 1: 16.8 2: 8.2 1: 25.2 2: 15.2	2: 0.6-73.5 1: 1.2-34.0 2: 0.8-33.1 1: 0.4-409 2: 0.6-140		

Table B20 Intake estimates (µg/kg bw/day) of DBP based on urinary biomonitoring data from Europe (gray-shaded values are used by RAC 2010)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
2011-2013	GR	4 years n=500 Mothers: see below	MnBP	Median 0.70	4.25	Max 22.94	Spot samples Volume method UV children 0.0224 L/kg bw FUE from Anderson et al. (2001)	Myridakis et al. (2016)
Nov 2011-May 2012	DE	1.7-6.7 years n=663	MnBP	Median 1.31	4.92	Not reported	Spot samples: 1-3 h after day care and for some also first morning samples on Monday before going to the day care Volume method FUE: 0.78 from Seckin et al. (2009)	Fromme et al. (2013b)
2010-2011	AT	6-8 years n=30/31 7-15 years n=214/219	MnBP	Median 1: 0.99 2: 0.84 1: 0.40 2: 0.34	1: 2.4 2: 10 1: 1.6 2: 1.8	1: 0.07-2.6 2: 0.15-15 1: 0.0-2.5 2: 0.0-19	Spot samples (before midday) 1. volume model 2. creatinine model FUE of 0.84	Hartmann et al. (2015)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
		18-64 years n=267/269 65-81 years n=69/71		1: 0.28 2: 0.24 1: 0.35 2: 0.35	1: 1.4 2: 1.1 1: 1.9 2: 1.9	1: 0.0-2.4 2: 0.0-2.0 1: 0.0-2.9 2: 0.0-2.6		
March 2009 – June 2011	GR	2 years n=239 Mothers: see below	MnBP	Median 1.0	6.6	Max 50.8	Spot samples Volume method UV children 0.0224 L/kg bw FUE from Anderson et al. (2001)	Myridakis et al. (2015)
Nov 2009- Oct 2010	DE	8-10 years n= 465	MnBP MCPP OH-MnBP	Median 1.82	5.86	Max 21.5	First morning urine Volume method FUE for DBP of 0.69 for 2 metabolites	Kasper-Sonnenberg et al. (2014)
Oct 2009 – Jan 2010	DE	15-21 months n= 25	MnBP	Median 1.6*/2.2**	3.6*/6.2**	Max 5.9*/9.2**	Morning urine on 7 consecutive days Volume method FUE of 0.69 taken from Seckin et al. (2009) *based on median from 7 sampling days for each subject **based on 95th p from 7 sampling days for each subject	Fromme et al. (2013a)
April-Sep 2008	DK	Men 18-22 years n=33	MnBP MiBP	Median DBP: 0.90;0.83;0.76 DBP + DiBP: 2.6;2.4;2.2	DBP:2.86; 4.22; 5.11	DBP: 0.33-3.40; 0.31-7.40; 0.29-12.00	24h sampling: 3 days with intervals of 40-46 days urinary levels of MnBP and MiBP were analysed together as one (the	Kranich et al. (2014)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
					DBP + DiBP: 8.2; 12.1; 14.6	DBP + DiBP:0.95-9.7; 0.87-21.1; 0.84-34.3	isoforms could not be separated by the chromatographic method used) Intake of DBP was estimated from the sum of DBP + DiBP using a ratio of 35:65 FUE of 69% from Anderson et al. (2001)	
Aug 2008 – April 2009	DK	3-6 years n= 441	MnBP	Median 3.26	10.03	0.25-162.9	Morning urine Volume method FUE of 0.69	Bekö et al. (2013)
2008	DK	Mothers n=52 Infants (1-5 months) n=47	MnBP	Median 1.8 lower than for mothers	5.6 lower than for mothers		Spot samples Mothers: at 7 time points (34^{th}-37^{th} week of pregnancy, before delivery and 1, 2, 3, 4 and 5 months after delivery) Infants: At 1, 2, 3, 4 and 5 months after delivery Volume method UV infants 0.044 l/kg UV mothers 0.02 l/kg FUE of 0.78 from Seckin et al. (2009)	Völkel et al. (2014)
May 2007 – May 2008	GR	Mothers n=239 Children: see above	MnBP	Median 1.9	11.4	Max 4839.8	Spot samples At 10^{th} – 13^{th} week of gestation Volume method UV mothers 2L FUE from Anderson et al. (2001)	Myridakis et al. (2015)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
Nov 2007	DK	6-10 years (♂/♀) n=24/25 17-21 years (♂/♀) n=14/11 Total n = 103	MnBP + **MiBP** DBP + DiBP median 5.27/5.28 (♂/♀) 3.8/2.53 (♂/♀) 4.29	DBP + DiBP 14.6/21.4 (♂/♀) not available 11.3	DBP + DiBP Max 15.9/24.1 (♂/♀) 6.45/4.17 (♂/♀) 35.5	24h sampling (total urine collected) urinary levels of MnBP and MiBP were analyzed together as one (the isoforms could not be separated by the chromatographic method used) According to Koch et al. (2007)/ Wittassek (2007)	Frederiksen et al. (2011)
Oct 2007	FR	Pregnant women n=279	MnBP Median 1.5	6.6		Spot samples Creatinine corrected FUE of 0.69 from Anderson et al. (2001)	Zeman et al. (2013)
Feb-March 2007	DE	children 5-6 year n=108	MnBP median 1.9	6.4	max 11.2	Spot Creatinine corrected FUE: as published in Koch and Calafat (2009) and Wittassek et al. (2011)	Koch et al. (2011)
2001 and 2003	DE	20-29 years n= 119	MnBP median 2.2	7.3	0.49-116	24 hour sampling FUE from Anderson et al. (2001) Not all data shown here: ESB, data reported for 1988-2003	Wittassek et al. (2007b)
April 2002	DE	7-64 years n= 85	MnBP median 5.2	16.2		First morning urine volume based	Koch et al. (2003) reported in Wittassek et al. (2007b);

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
							FUE from Anderson et al. (2001)	Koch and Calafat (2009)
March 2001-March 2002 (GER-ES IV)	DE	 2-4 years (n=31) 5-6 years (n=46) 7-8 years (n=53) 9-11 years (n=56) 12-14 years (n=53) Total (2-14 years) n= 239	MnBP	median 1: 10.5 2: 6.46 1:7.47 2: 5.03 1: 7.17 2: 4.85 1: 8.47 2: 4.02 1: 5.29 2: 3.09 1: 7.61 2: 4.07	 1: 37.2 2: 18.1 1: 19.5 2: 12.3 1: 33.0 2: 23.3 1: 27.2 2: 9.10 1:24.5 2: 11.2 1: 30.5 2: 14.9	 1: 0.09-54.8 2: 1.36-25.9 1: 3.16-31.5 2: 1.93-25.3 1: 1.60-88.9 2: 1.69-76.4 1: 1.55-40.5 2: 0.81-11.8 1: 0.91-110 2: 0.66-73.3 1: 0.91-110 2: 0.66-76.4	First morning urine 1. volume method 2. creatinine excretion of children FUE from Anderson et al. (2001)	Koch et al. (2007)

Table B21 Intake estimates (µg/kg bw/day) of DIBP based on urinary biomonitoring data from Europe (gray-shaded values are used by RAC 2010)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
2011-2013	GR	4 years n=500 Mothers: see below	MiBP	Median 1.20	 5.31	Max 23.84	Spot samples Volume method UV children 0.0224 L/kg bw FUE from DBP	Myridakis et al. (2016)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
Nov 2011- May 2012	DE	1.7-6.7 years n=663	MiBP	Median 1.80	6.24	Not reported	Spot samples: 1-3 h after daycare and for some also first morning samples on Monday before going to the daycare Volume method FUE: assumed to be the same as for DBP	Fromme et al. (2013b)
2010- 2011	AT	6-8 years n=30/31 7-15 years n=214/219 18-64 years n=267/269 65-81 years n=69/71	MiBP	Median 1: 2.4 2: 2.3 1: 1.3 2: 1.1 1: 0.99 2: 0.78 1: 1.0 2: 0.96	 1: 6.9 2: 14 1: 4.7 2: 7.0 1: 4.6 2: 3.4 1: 4.4 2: 4.9	 1: 0.25-7.9 2: 0.52-17 1: 0.0-7.1 2: 0.0-34 1: 0.0-12 2: 0.0-16 1: 0.08-5.3 2: 0.15-8.7	Spot samples (before midday) Two methods: 1. volume model 2. creatinine model FUE of 0.7 from Koch et al. (2012)	Hartmann et al. (2015)
March 2009 – June 2011	GR	2 years n=239 Mothers: see below	MiBP	Median 1.4	8.2	Max 36.0	Spot samples Volume method UV children 0.0224 L/kg bw FUE from DBP	Myridakis et al. (2015)
Nov 2009- Oct 2010	DE	8-10 years n= 465	MiBP OH-MiBP	Median 2.18	9.65	Max 44.4	First morning urine Volume method FUE for DIBP of 0.69 for 2 metabolites	Kasper-Sonnenberg et al. (2014)
Oct 2009 – Jan 2010	DE	15-21 months n= 25	MiBP	Median 2.2*/3.9**	1.13*/ 9.02**	Max 6.1*/13.9**	Morning urine on 7 consecutive days Volume method	Fromme et al. (2013a)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
							FUE as for DBP *based on median from 7 sampling days for each subject **based on 95th p from 7 sampling days for each subject	
April-Sep 2008	DK	Men 18-22 years n=33	MiBP MnBP	Median DiBP: 1.66; 1.54; 1.41 DBP + DiBP: 2.6; 2.4; 2.2	DiBP: 5.30; 7.84; 9.49 DBP + DiBP:8.2; 12.1; 14.6	DiBP: 0.62-6.31; 0.57-13.75; 0.55-22.28 DBP + DiBP: 0.95-9.7; 0.87-21.1; 0.84-34.3	24h sampling: 3 days with intervals of 40-46 days urinary levels of MnBP and MiBP were analysed together as one (the isoforms could not be separated by the chromatographic method used) Intake of DBP was estimated from the sum of DBP + DiBP using a ratio of 35:65 FUE not given	Kranich et al. (2014)
Aug 2008 – April 2009	DK	3-6 years n= 441	MiBP	Median 2.93	10.02	0.26-152.4	Morning urine Volume method FUE as for DBP	Bekö et al. (2013)
2008	DK	Mothers n=52 Infants (1-5 months) n=47	MiBP	Median 3.1 lower than for mothers	10.1 lower than for mothers		Spot samples Mothers: at 7 time points (34th-37th week of pregnancy, before delivery and 1, 2, 3, 4 and 5 months after delivery) Infants: At 1, 2, 3, 4 and 5 months after delivery Volume method: UV infants 0.044 l/kg ; UV mothers 0.02 l/kg	Völkel et al. (2014)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
							FUE of 0.78 (Seckin et al. 2009)	
May 2007 – May 2008	GR	Mothers n=239 Children: see above	MiBP	Median 2.1	11	Max 30.6	Spot samples At 10th – 13th week of gestation Volume method UV mothers 2L FUE from DBP	Myridakis et al. (2015)
Nov 2007	DK	 6-10 years (male/female) n=24/25 17-21 years (male/female) n=14/11 Total n = 103	MiBP + **MnBP**	DBP + DiBP median 5.27/5.28 (♂/♀) 3.8/2.53 (♂/♀) 4.29	DBP + DiBP 14.6/21.4 (♂/♀) not available 11.3	DBP + DiBP Max 15.9/24.1 (♂/♀) 6.45/4.17 (♂/♀) 35.5	24 hour sampling urinary levels of MnBP and MiBP were analysed together as one (the isoforms could not be separated by the chromatographic method used) FUE according to Koch et al. (2007)/ Wittassek et al. (2007)	Frederiksen et al. (2011)
Oct 2007	FR	 Pregnant women n=279	MiBP	Median 2.2	11.1		Spot samples Creatinine corrected FUE of 0.69 (as DBP)	Zeman et al. (2013)
Feb-March 2007	DE	 5-6 years n= 108	MiBP	median 2.1	11.0	max 59.4	Spot Creatinine corrected FUE: as published in Koch and Calafat (2009) and Wittassek et al. (2011)	Koch et al. (2011)
2001 and 2003	DE	 20-29 years n= 119	MiBP	median 1.5	4.2	0.29-12.6	24 hour sampling FUE from Anderson et al. (2001)	Wittassek et al. (2007b)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
							Not all data shown here: ESB, data reported for 1988-2003	

Table B22 Intake estimates (µg/kg bw/day) of BBP based on urinary biomonitoring data from Europe (gray-shaded values are used by RAC 2010)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
2011-2013	GR	4 years n=500 Mothers: see below	MBzP	Median 0.17	1.32	Max 15.95	Spot samples Volume method UV children 0.224 L/kg bw FUE from Anderson et al. (2001)	Myridakis et al. (2016)
Nov 2011-May 2012	DE	1.7-6.7 years n=663	MBzP	Median 0.43	2.97	Not reported	Spot samples: 1-3 h after day care and for some also first morning samples on Monday before going to the day care Volume method FUE: 0.73 from Anderson et al. (2001)	Fromme et al. (2013b)
2010-2011	AT	6-8 years n=30/31 7-15 years n=214/219 18-64 years	MBzP	Median 1: 0.22 2: 0.21 1: 0.09 2: 0.08 1: 0.99	 1: 1.9 2: 1.9 1: 0.78 2: 0.77 1: 4.6	 1: 0.0-2.2 2: 0.0-2.1 1: 0.0-1.8 2: 0.0-1.3 1: 0.0-12	Spot samples (before midday) 1. volume model 2. creatinine model FUE of 0.73 from Anderson et al. (2001)	Hartmann et al. (2015)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
		n=267/269 65-81 years n=69/71		2: 0.04 1: 1.0 2: 0.05	2: 0.23 1: 4.4 2: 0.40	2: 0.0-1.0 1: 0.08-5.3 2: 0.0-0.46		
March 2009 – June 2011	GR	2 years n=239 Mothers: see below	MBzP	Median 0.2	1.3	Max 9.0	Spot samples Volume method UV children 0.224 L/kg bw FUE from Anderson et al. (2001)	Myridakis et al. (2015)
Nov 2009-Oct 2010	DE	8-10 years n= 465	MBzP	Median 0.23	1.30	Max 14.9	First morning urine Volume method FUE of 0.73	Kasper-Sonnenberg et al. (2014)
Oct 2009 – Jan 2010	DE	15-21 months n= 25	MBzP	Median 0.3*/0.7**	1.3*/2.5**	Max 2.1*/2.7**	Morning urine on 7 consecutive days Volume method FUE of 0.73 from Anderson et al. (2001) *based on median from 7 sampling days for each subject **based on 95th p from 7 sampling days for each subject	Fromme et al. (2013a)
April-Sep 2008	DK	Men 18-22 years n=33	MBzP	Median 0.7; 0.5; 0.6	2.9; 8.8; 1.8	0.05-3.7; 0.00-9.5; 0.06-2.4	24h sampling: 3 days with intervals of 40-46 days FUE of 73% from Anderson et al. (2001)	Kranich et al. (2014)
Aug 2008 – April 2009	DK	3-6 years n= 441	MBzP	Median 0.49	2.79	0.023-22.3	Morning urine Volume method	Bekö et al. (2013)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
							FUE of 0.73	
2008	DK	Mothers n=52 Infants (1-5 months) n=47	MBzP	Median 0.6 lower than for mothers	3.3 lower than for mothers		Spot samples Mothers: at 7 time points (34th-37th week of pregnancy, before delivery and 1, 2, 3, 4 and 5 months after delivery) Infants: At 1, 2, 3, 4 and 5 months after delivery Volume method UV infants 0.044 l/kg UV mothers 0.02 l/kg FUE as in Fromme et al. (2013)	Völkel et al. (2014)
May 2007 – May 2008	GR	Mothers n=239 Children: see above	MBzP	Median 0.3	1.8	Max 9.9	Spot samples At 10th – 13th week of gestation Volume method UV mothers 2L FUE from Anderson et al. (2001)	Myridakis et al. (2015)
Nov 2007	DK	6-10 years (♂/♀) n=24/25 17-21 years (♂/♀) n=14/11 total n=103 (6-21 years)	MBzP	median 0.96/0.97 (♂/♀) 0.4/0.32 (♂/♀) 0.62	6.28/3.57 (♂/♀) not available 3.78	9.96	24 hour sampling 2 consecutive morning urine + pool of all urine in between FUE according to Koch et al. 2007/ Wittassek et al. (2007)	Frederiksen et al. (2011)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
Oct 2007	FR	Pregnant women n=279	MBzP	Median 0.4	2.4		Spot samples Creatinine corrected FUE of 0.73 from Koch et al. (2003)	Zeman et al. (2013)
Feb-March 2007	DE	5-6 years n= 108	MBzP	median 0.3	2.6	Max 10.4	Spot Creatinine corrected FUE: as published in Koch and Calafat (2009) and Wittassek et al. (2011)	Koch et al. (2011)
2001 and 2003	DE	20-29 years n= 119	MBzP	median 0.22	0.75	0.02-1.74	24 hour sampling FUE from Anderson et al. (2001) Not all data shown here: ESB, data reported for 1988-2003	Wittassek et al. (2007b)
April 2002	DE	7-64 years n= 85	MBzP	median 0.6	2.5	Not reported	First morning urine volume based FUE not reported	Koch et al. (2003) reported in Wittassek et al. (2007b)
March 2001-March 2002 (GER-ES IV)	DE	2-4 years (n=31) 5-6 years (n=46) 7-8 years (n=53) 9-11 years (n=56) 12-14 years (n=53)	MBzP	median 1: 1.25 2: 0.61 1: 0.80 2: 0.49 1: 0.94 2: 0.54 1: 0.74 2: 0.29	 1: 3.92 2: 2.38 1: 3.57 2: 1.56 1: 3.69 2: 2.46 1: 7.79 2: 2.97	 1: 0.05-13.2 2: 0.18-3.88 1: 0.20-5.77 2: 0.15-3.35 1: 0.18-25.1 2: 0.16-13.9 1: 0.14-31.3 2: 0.06-11.7	First morning urine 1. volume method 2. creatine excretion of children FUE by Anderson et al. (2001)	Koch et al. (2007)

Sample year	Country	Age, year	Metabolite	Median/GM	95th percentile	Range	Basis for estimated intake	Reference
		Total (2-14 years) n= 239		1: 0.45 2: 0.30 1: 0.77 2: 0.42	1: 4.12 2: 1.98 1: 4.48 2: 2.57	1: 0.11-5.59 2: 0.09-3.28 1: 0.05-31.3 2: 0.06-13.9		

B.8.3.2.6. Discussion

Comparisons between studies are difficult as a result of differences in sample period, age groups, geographical area of residence of the study population, size of the study population, and methodology used to estimate the intake. Therefore the attempts made below to draw conclusions need to be interpreted with caution and are not necessarily valid for other countries.

Age

The DEMOCOPHES biomonitoring exposure estimates clearly show that exposure of children is higher than that of mothers. This is generally the case also in other biomonitoring studies (Hartmann et al. 2015; Frederiksen et al. 2011; Becker et al. 2009; Geens et al. 2014). As an exception, in Myridakis et al. (2015) mothers had a higher intake than children which may in part be explained by the market changes over the 2.5 year between taking of samples of mothers and children. The higher food and dust intake or exposure through inhalation relative to body weight of children compared to adults might help to explain this difference (Den Hond et al. 2015; Frederiksen et al. 2011). In addition, differences in exposure patterns in children and metabolism can be factors explaining the differences (Den Hond et al. 2015).

Data from the same sample period as DEMOCOPHES (2011-2012) from Fromme et al. (2013b) suggests that German children of 1.7-6.7 years old may be exposed at somewhat higher levels of DEHP, DIBP and BBP compared with children of 6-11 years old. The trend is especially pronounced with BBP and DIBP but is only weakly seen with DBP. The fact that Fromme et al. (2013b) used the volume method for calculating intake may partly explain the difference, but comparison of Fromme et al. (2013b) with Kasper-Sonnenberg et al. (2014) and Fromme et al. (2013a) confirms this observation (except for infants in Fromme et al. (2013a) potentially indicating that infants may have different exposure patterns to BBP compared to the other phthalates). Myridakis et al. (2016) show a higher intake of DBP, DIBP and BBP (but not of DEHP) in 2 years old compared with 4 years old children.

Exposure trend over time

The DEMOCOPHES data from Germany for children aged 6-11 years was compared with the weighted average of creatinine corrected intake estimates for the age groups 5-6, 7-8 and 9-11 years from the years 2001-2002 (Wittassek et al. 2007a; Koch et al. 2007). It can be concluded that over this 10 year period the exposure of German children of 6-11 years has declined by about 50% for DEHP[46], 75% for DBP[47] and 60% for BBP[48]. No data was available to allow for a similar comparison to be made for DIBP.

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[46] The weighted average (weighted by n) of creatinine corrected intake estimates for the age groups 5-6, 7-8 and 9-11 years from Wittassek et al. (2007a) from the years 2001-2002 gives a median intake estimate of 4.7 µg/kg bw/day and 95th p 13.5 µg/kg bw/day for DEHP (compared with a median of 2.45 and 95th p of 7.26 µg/kg bw/day from DEMOCOPHES).

[47] The weighted average (weighted by n) of creatinine corrected intake estimates for the age groups 5-6, 7-8 and 9-11 years from Koch et al. (2007) from the years 2001-2002 gives a median intake estimate of 4.6 µg/kg bw/day and 95th p 14.9 µg/kg bw/day for DBP (compared with a median of 1.19 and 95th p of 3.66 µg/kg bw/day from DEMOCOPHES).

[48] The weighted average (weighted by n) of creatinine corrected intake estimates for the age groups 5-6, 7-8 and 9-11 years from Koch et al. (2007) from the years 2001-2002 gives a median intake estimate of 0.4 µg/kg bw/day

Similarly, comparing DEMOCOPHES intake estimates from Germany with data from 2001/2003 from Wittassek et al. (2007b), it can be assumed that over a period of 8-10 years, exposure in adults declined with 40-50% for DEHP, 60-70% for DBP, 55% for DIBP, and 30-45% for BBP.

A comparison of the DEMOCOPHES data intake estimates from Denmark for children with data from 2007 in Danish children aged 6-10 years (Frederiksen et al. 2011) would suggest exposure declined with 50-70% for DEHP[49] and 80% for BBP[50] in the period 2007-2011.

Similarly, for adults a decline in exposure of 30% for DEHP and 60% for BBP can be assumed.

The data furthermore suggests a decline in exposure of 80% in children and 75% in adults in this period also for the sum of DIBP and DBP metabolites[51].

The exposure to phthalates has declined over time when older biomonitoring studies are compare to the DEMOCOPHES data. The data is summarised in Table B23 and indicates that a significant decline in exposure has taken place in Germany and Denmark over 2001-2011.

Table B23 Comparison of data from DEMOCOPHES and the literature to assess a trend in exposure

Study	Period	Member State	Population	Percentage decline			
				DEHP	BBP	DBP	DIBP
Wittassek et al. 2007a; Koch et al. 2007	2001-2002 to 2011	DE	Children aged 6-11 years	50%	75%	60%	-
Wittassek et al. (2007b)	2001/2003 to 2011	DE	Adults	40-50%	30-45%	60-70%	55%
Frederiksen et al. 2011	2007-2011	DK	Children aged 6-10 years	50-70%	80%		
		DK	Adults	30%	60%	75%	
		DK	Children			80%	

Regarding projected exposure levels beyond 2011, see section D.3.5.3.

and 95th p 2.4 µg/kg bw/day for BBP (compared with a median of 1.19 and 95th p of 3.66 µg/kg bw/day from DEMOCOPHES).

49 A mean of median DEHP exposure values for boys and girls gave 5.5 µg/kg bw/day and a mean of 95th percentiles of 27.5 µg/kg bw/day, whereas the DEMOCOPHES intake estimate for mean and 95th percentile was 2.84 and 7.75 µg/kg bw/day respectively. This allows a rough comparison that suggests a 50-70% decline in exposure.

50 A mean of median BBP exposure values for boys and girls gave 0.965 µg/kg bw/day and a mean of 95th percentiles of 4.9 µg/kg bw/day, whereas the DEMOCOPHES intake estimate for mean and 95th percentile was 0.21 and 1.00 µg/kg bw/day respectively. This allows a rough comparison that suggests a 80% decline in exposure.

51 Frederiksen et al. (2011) analysed MnBP and MiBP metabolites together since the isoforms could not be separated by the chromatographic method used. The sample size is small and reported for boys and girls separately (24 boys and 25 girls). A mean of median value for children of 6-10 years gave 5.3 µg/kg bw/day and a mean of 95th percentiles of 18.0 µg/kg bw/day, whereas the DEMOCOPHES sum of mean intake values for DBP and DIBP and sum of 95th percentiles were 1.14 and 3.03 µg/kg bw/day respectively. This allows a rough comparison that suggests a 80% decline in exposure. A mean of median value for adults gave 3.2 µg/kg bw/day (95th percentiles not available), whereas the DEMOCOPHES sum of mean intake values for DBP and DIBP was 0.78 µg/kg bw/day suggesting a 75% exposure decline.

Comparison by Member State

Data covering almost all Austrian federal states (Hartmann et al. 2015), suggests that exposure levels of children to DEHP and BBP are rather similar to those in Germany from the North-Rhine-Westphalia region (Bochum as urban area and Higher Sauerland District as rural area), whereas Austrian exposure levels to DIBP are significantly higher than those in Germany.

Data from Zeman et al. (2013) would suggest that exposure in adults to DEHP in France might have been about double compared with Denmark in 2007 (Frederiksen et al. 2011).

The median and the 95th percentile intakes of pregnant mothers (2007-2008) in Myridakis et al. (2015) are respectively more than double and 2.6-5.4 times higher in comparison to the overall EU intake estimates from DEMOCOPHES in Table B14.

Exposure to DIBP appears to be a factor of 3-4 higher in Europe compared to the US in both mothers and children from the DEMOCOPHES study (Den Hond et al. 2015).

Co-exposure to multiple phthalates

Frederiksen et al. (2011) observed a significant positive correlation within samples between DBP, BBP, DEHP and DINP metabolites. Becker et al. (2009) observed strong correlations between metabolites of DBP and DIBP, DBP and BBP, DBP and DEHP and DBP and DINP as well as strong correlations between metabolites of DEHP and DINP. Also Frederiksen et al. (2013) found that DBP, DIBP+DBP, DEHP and DINP metabolites were all significantly correlated to each other within samples. This was the case for spot samples, morning urine samples as well as 24 hour samples.

A study by ExxonMobil (Qian et al. 2015), assessed chemical coexposure based on biomonitoring data from the US. The data shows that 172 out of 2604 participants were exposed above the 95th percentile for two or more phthalates simultaneously, or about 7%, and 49 out of 2604 participants were exposed above the 95th percentile for three or more phthalates simultaneously, or 2% (three phthalates is the relevant number since BBP does not contribute much to the RCRs). From a different perspective, 636 participants (24%) were exposed to at least one phthalate above the 95th percentile and of this highly exposed population 27% were also exposed above the 95th percentile for two or more phthalates simultaneously, and 8% were also exposed above the 95th percentile for three or more phthalates simultaneously.

From Figure 2b of Qian et al. (2015) it also appears that about 28% of individuals is exposed to two or more phthalates simultaneously above the 85th percentile, and 14% of individuals is exposed to three or more phthalates simultaneously above the 85th percentile. From a different perspective, about 53% of participants were exposed to at least one phthalate above the 85th percentile and of this highly exposed population about 53% were also exposed above the 85th percentile for two or more phthalates simultaneously, and about 26% were also exposed above the 85th percentile for three or more phthalates simultaneously.

A longitudinal study of urinary phthalates in infants concluded that "*Children with a relatively high exposure to one phthalate tended also to be highly exposed to the other five phthalates, and exposures tended to be present at a relatively high level throughout infancy*" (Frederiksen

et al. 2014). The authors pointed out that their conclusions were supported by the literature (referring to Frederiksen et al. 2012; Koch et al. 2011; Mieritz et al. 2012; Mouritsen et al. 2013).

In conclusion, these consistent results indicate that individuals exposed to high levels of one phthalate often are also highly exposed to other phthalates. The data in Qian et al. (2015) demonstrates that the addition of RCRs based on the 95th percentile is in fact reasonable in the current risk assessment since 2% of the participants were exposed above the 95th percentile for three or more phthalates simultaneously.

Phthalate exposure via food intake

Several studies show that food is an important source for the exposure of phthalates, in particular for exposure to DEHP (Fromme et al. 2013a; Koch et al. 2013; Wittassek et al. 2011; Rudel et al. 2011; UBA 2011). The urinary levels of phthalates were measured in these studies, and either the diet was changed (fasting or low-phthalate diet) or the content of phthalates in the diet was measured.

Fromme et al. (2013a) calculated the exposure of 25 German infants in the age of 15-21 months, based on analyses of duplicate diet samples over 7 consecutive days and took morning spot urine samples of the infants in the same period (2009-2010). The comparison of the total exposure based on biomonitoring data showed that the total exposure of DEHP (based on biomonitoring) mainly derived from uptake via food (approximately 90% or more), whereas for DBP and DIBP the exposure from the diet comprise approximately 24-32% of the total exposure. The relevance of exposure from food for BBP seem to be very limited and only 4% of the exposure seems to originate from food. These results indicate that for infants (15-21 months) there are other sources to the exposure of especially DBP, DIBP and BBP.

Another comparison of the exposure via food and the intake estimated from biomonitoring is made by Koch et al. (2013). Full volumes of urine samples were collected over 48 hours of fasting as well as before and after the fasting period. The samples were taken from five German adults in the age of 27-47 in November 2009. Declining concentration of metabolites of DEHP was seen during fasting. This indicates that the exposure of DEHP is driven by food consumption. For DBP, DIBP and BBP only a weak influence of fasting is seen, indicating that the exposure is driven by other sources.

A poster from Koch et al. in 2006 as reported in Wittassek et al. (2011) described the influence of fasting on urinary metabolite concentrations of DEHP, DINP, DBP, DIBP, BBP. Three adult volunteers fasted for 48 hours (they drank mineral water only) and urine samples were collected before during and after the fasting period. Urinary metabolite levels for DEHP and DINP dropped sharply to very low levels and these low levels persisted throughout the second day of fasting (median levels were about 7.5 times lower than initial median levels). For DBP, DIBP and BBP this pattern was not observed and exposure peaks were evident within the fasting period. Median levels were lower in the fasting period though than initial median levels (about a factor of 2-3, not statistically significant). Interestingly, the exposure sources to these three phthalates seemed to have been the same as the urinary levels followed the same pattern.

Rudel et al. (2011) collected urinary samples during 8 days from 20 participants (10 adults and 10 children) in five families in California in January 2010. The first 2 days the participants ate their normal food, followed by a 3 days diet with almost exclusively fresh and organic food, where preparation techniques avoided contact with plastic utensils, and then followed by 3 days their normal food again. The 3-day meal intervention showed a reduction in geometric mean concentration in the urine of 12% for MBzP (a metabolite of BBP), 25% MBUP (a metabolite of DBP and BBP) and a reduction of 53-56% of DEHP metabolites. Maxima declined by 93-96% for DEHP metabolites (for the sub-group of children, the maxima dropped 30-64%[52]), 12.5%[52] for MBUP and 56%[52] for MBzP. The authors did not observe significant variation between families, which suggests that individual behaviour is more important than the home environment.

An unpublished study from Germany (UBA 2011) has measured the content of 5 DEHP metabolites in urinary samples from 10 boys of 5 to 8 years old in 2005. The content of DEHP in the air and dust from their homes was measured and a duplicate sample of their food and drinks was taken and analysed for DEHP as well. Based on comparison of the mean contributions, the study concludes that 58% of the daily DEHP intake originates from foodstuff, 7% from drinks, 5% from air, and 18% from dust.

Since duplicate diet/drink samples were taken, the intake estimates from this study for food and drinks should be rather accurate. The estimate for dust intake is more uncertain since it relies on the assumption of 100mg dust intake per day. Although the samples were taken before the legislation on food contact materials entered into force, the median and 90th percentile intake as estimated from biomonitoring was 2.27 and 7.02 µg/kg bw/day respectively, which is roughly equal to the data from Germany in DEMOCOPHES from 2011-2012 (median of 2.45 and 90th percentile of 5.18 µg/kg bw/day).

From Fromme et al. (2013a), Koch et al. (2013), Wittassek et al. (2011)[53], Rudel et al. (2011)[54] and UBA (2011) it can be concluded that the exposure of DEHP in adults and infants is driven by food consumption while there seem to be additional important sources of exposure of DBP, DIBP and BBP. For the purposes of the current analysis it has been assumed that 75% of the intake of DEHP is attributable to food (incl. drinks), whereas for DBP, DIBP and BBP it is assumed that 25% is attributable to food. The contribution of food to the intake of DEHP might also be higher than 75% based on Fromme et al. (2011). On the other hand, UBA (2011) data

[52] Calculated from the supplemental material to Rudel et al. (2011).

[53] The information from Koch et al. reported in 2006 in Wittassek et al. (2011) adds to the evidence that food is an important source of exposure to phthalates, in particular to DEHP. It should be noted that the experiment is not recent and that the number of participants was very low (n=3) and thus the exposure sources the participants were exposed to is not representative to the (current) EU population.

[54] Care should be taken when interpreting the specific proportions (percentages) of exposure attributable to FCMs in Rudel et al. (2011). Indeed, the proportions might not necessarily be representative for California (n=20) or the EU, and the study was designed to avoid, but not to exclude, dietary intake of phthalates from food contaminated through food contact materials or other contamination sources. Non-FCM exposure might be different in California compared with the EU. Phthalates contamination of food from packaging or food processing in California might differ from the EU, e.g., there appear to be no migration limits for phthalates in food contact materials in the US nor California. In the EU, migration limits for DEHP, DBP and BBP might result in a lower contribution from FCMs, but regulatory pressure has been higher on the other markets for the four phthalates as well and future decline is expected for the four phthalates especially in articles other than FCMs as a consequence of the authorisation process. Thus, on balance, the proportion of exposure attributable to FCMs might give a good indication for the EU situation.

is from before the FCM legislation thus one would expect that the relative proportion of food to the intake of DEHP would have decreased since and would thus be lower than 65%.

In comparison, the modelling performed (see section B.9.2.4) suggests that the contribution to exposure to DEHP from food is only 38%, 51% and 36% in infants, children and adults respectively. For DBP, the modelling suggests a contribution of 32%, 19% and 10% in infants, children and adults respectively; for DIBP 44%, 35% and 18% in infants, children and adults respectively; and for BBP 0% (no recent data available), 34% and 22% in infants, children and adults respectively.

For DEHP, the modelling seems to underestimate the contribution via food relative to other exposure sources. For the other phthalates, considering all uncertainties, the modelling estimates of the proportion of food to overall exposure sources is reasonably similar to the estimates above that were based on 'fasting-urinary biomonitoring' or 'duplicated diet-urinary biomonitoring' studies.

Den Hond et al. (2015) found associations between DEHP levels and chewing gum and ice cream consumption in the DEMOCOPHES study. The authors interpret this association as tendency for eating flavoured, packaged or processed food and thus they saw chewing gum and ice cream not necessarily as exposure sources themselves but rather as a proxy for convenience food.

Geens et al. (2014) reported that consumption of locally grown vegetables was significantly associated with higher levels of DBP and BBP urinary metabolites, possibly explained by use of pesticides.

Medication and medical devices

DBP is used in enteric coatings[55] in medications in concentrations up to 9000 µg per capsule (Seckin et al. 2009). Fromme et al. (2007) observed that creatinine-adjusted DBP metabolite levels increased significantly in case medication was taken, but not with unadjusted metabolite levels and not when mixed linear models were applied to account for repeated measurements on the subjects. The 95th percentiles exposure levels for DBP in DEMOCOPHES biomonitoring do not seem to reflect such high exposure peaks which would be expected to be one or two orders of magnitude higher. There is likely to be an unknown proportion of the EU population where exposure to DBP via medicines results in a significantly higher exposure than estimated in the current restriction report. However, from 1 June 2018, medicines should only exceptionally contain DBP (EMA 2014)[56].

Medical devices may contribute to exposure to DEHP, for example in preterm neonates (SCENIHR 2016). Since the population in biomonitoring studies such as DEMOCOPHES does

[55] The phthalates provide increased resistance to degradation of the coating with the aim to deliver medications to specific areas of the gastrointestinal tract (Gallinger and Nguyen 2013; EMA 2014). Other functions are: maintenance of flexibility of tablet or capsules (e.g. to prevent cracking) and increased ease of swallowing; to control characteristics such as thinness of the coat whilst maintaining adequate barrier to moisture; control of drug-release; increase of the palatability by containing the drug formulation (EMA 2014). Examples of commonly prescribed medicines in the US that may contain phthalates are 5-Aminosalicylates, proton pump inhibitors, and pancreatic enzymes (Gallinger and Nguyen 2013).

[56] Veterenary medicines seem not to be covered and potentially may contribute to human exposure via the food chain.

not include neonates, there may be additional risks from phthalates not accounted for in the current risk assessment.

Exposure of women or children to DEHP from medical devices (e.g., used in blood transfusion) is of acute or short term nature. Patients with haemodialysis were not admissible to the DEMOCOPHES study (participants with metabolic disturbances or abnormal urine excretion were excluded, FPS 2013) but may be chronically exposed to DEHP from medical devices. It is thus highly unlikely that any patients with recent (within a day) exposure from medical devices would have been included in the study population of DEMOCOPHES. Medical procedures using PVC medical devices may lead to exposure that exceeds the daily intake in the general population by several orders of magnitude (Koch and Angerer 2012). Thus, for those children (boys) and women that regularly undergo medical treatment with DEHP containing medical devices, the risk as estimated in the current risk assessment is likely to be underestimated.

Indoor environment

Participants of DEMOCOPHES who reported to have PVC flooring or walls showed significantly higher BBP and DIBP metabolites in children as well as mothers and significantly higher DBP metabolite concentrations in children (Den Hond et al. 2015). However, no association was found between DEHP exposure and the presence of PVC at home in the DEMOCOPHES study (Den Hond et al. 2015). This suggests that although DEHP is certainly present in PVC flooring in Europe (see Annex A), it may not be very common.

Fromme et al. (2013b) reported that the floor covering in 63 daycare centres from Bavaria, Berlin and North Rhine-Westfalia did not significantly correlate with excretion of phthalate metabolites (15 rooms had PVC flooring). The authors however observed a significant correlation between phthalate concentrations in dust samples and urinary levels of DBP, BBP and to a lesser extent also DEHP metabolites. Significant correlations were observed between concentrations in air and urine metabolites for DBP, DEHP and DIBP (only a limited number of samples had concentrations above the LOQ in the air for BBP). In a multivariate analysis, increased indoor air concentrations were associated with higher urinary concentrations and were said to be independent of dust concentrations. However, the phthalate concentrations in air and dust correlated significantly for DEHP, DBP and DIBP.

Geens et al. (2014) observed significantly higher levels of DBP and BBP urinary metabolite levels in Flemish adolescents associated with the presence of wall paper in house (these phthalates are often present in adhesives and printing inks).

Socioeconomic position in society

Phthalate metabolites inversely correlated with educational level of the family in the DEMOCOPHES study (Den Hond et al. 2015). This might reflect associated lifestyle factors (Den Hond et al. 2015). One could also reason that lower educational level might be associated with lower incomes and less awareness about chemical risks, which might result in these families buying more plastic goods (that are typically cheaper than e.g., wood or metal) and eating more processed food. This theory might be supported by the observation that in the diet of European low-income groups there is a higher the contribution of fat to total energy intake and higher frequency of consumption of processed meat than for high-income groups (University of Leeds 2011). Similarly, Fernández-Alvira et al. (2015) observed that children with higher-educated parents and highest incomes were shown to have a tendency of a

healthier diet (consumption of fruits, vegetables and wholemeal products) and less sweet foods and sweetened drinks. Migrants showed a tendency to eat more processed food (snacks and fast food).

Similarly, Geens et al. (2014) observed a trend of increasing phthalate metabolites in Flemish adolescents with decreasing educational level of adolescents, but not with educational level of their parents. A lower household income was also associated with significantly higher urinary DBP metabolite levels. Also for other phthalates, highest incomes showed a (non-significant) tendency for low exposure levels.

Population density

La Rocca et al. (2014) reported that MEHP levels in serum were significantly lower in rural Italian areas compared with urban and metropolitan area. The metropolitan area showed higher MEHP levels than the other areas approximately by an order of magnitude in both fertile and infertile women.

B.8.4. Exposure modelling

To identify the most efficient risk reduction measures, the exposure from the different sources of the four phthalates DEHP, DBP, BBP and DIBP has to be identified. Exposure modelling a useful tool to better characterise the relative contribution of the different exposure sources.

Phthalates are found in high concentrations as softeners in PVC. Phthalates are also – to a lower extent - used in other plastics, in dispersions, paints and varnishes, as emulsifiers, and for other common applications.

The main sources of exposure to the four phthalates are food, contact with articles and the indoor environment. The exposure routes are via ingestion, inhalation and via dermal or mucous contact.

The major source of DEHP exposure is from the diet, whereas for DBP, DIBP and BBP other sources are probably of higher importance (see Section B.8.3.2).

Exposure from contact with articles may be due to direct contact between the article and the skin or mucous membrane, or oral, due to infants mouthing articles. Articles such as vinyl flooring contribute to exposure from the indoor environment, and other articles that are used in food processing or packaging can contribute to exposure via food. Finally, articles release phthalates to the environment and thus contribute to exposure of man via environment to phthalates (mainly via food).

The indoor environment acts as a source for the exposure to phthalates. Phthalates are found in the air (gas phase and particles in air) and the dust in the indoor environment which are contributing to the human intake of phthalates. The exposure levels of phthalates from the indoor environment are found in the literature for the levels in dust, and by calculations and modelling for the levels in indoor air (gas phase and particles in air). The phthalates found in articles (within a building/ indoor environment) will continue to emit phthalates over time, and thereby contribute to the phthalate concentration in the indoor air and dust.

B.8.4.1. Estimation of exposure of the general population via the indoor environment, food and contact with articles

Exposure estimates are calculated from the contribution from the three main sources of exposure to the four phthalates DEHP, DBP, BBP and DIBP, namely the indoor environment, contact with articles and food.

Two scenarios are made, a typical scenario and a reasonable worst case scenario. These two scenarios aim to give an indication of the exposure for the average consumer and for the highly exposed consumer.

When calculating the reasonable worst case estimate, it is a challenge to decide how variation in the input parameters shall be taken into account. It is obvious that the reasonable worst case scenario for all parameters very seldom will appear at the same time for all parameters involved. For that reason, a probabilistic approach using Monte Carlo simulations was used in addition to deterministic calculations. These simulations make it possible to combine distribution functions for different parameters into a new distribution function, where it is assumed that the parameters are not correlated (worst case for one parameter is not correlated with a worst case for another parameter). Monte Carlo simulations were only used to estimate exposure from contact with articles since in this case modelling uses combinations of reasonable worst case assumptions for different parameters. Probabilistic modelling was also used for assessing combined exposure to the four phthalates. The combined exposure assessment using Monte Carlo simulations assumes there is no correlation between high exposure from one phthalate with high exposure from another phthalate is assumed. Since high exposure to one phthalate is in fact correlated with high exposure to other phthalates, the ‘truth’ may lay somewhere in between the combined exposure assessment based on probabilistic and based on deterministic modelling.

The major difficulty of this approach is defining the distributions of probabilities of the entry variables of the models. This information is generally not given in the literature. In this report the varying parameters for which Monte Carlo estimations are carried out are assumed to follow normal distributions.

As described further in the section B.9.3.6.5 on uncertainties, the estimations of the reasonable worst case is considered to fit relatively well.

Exposure from contact with specific articles is also calculated to show that single articles in some cases can cause a high exposure. To illustrate this, calculations of exposure based on migration rates analysed for the concrete articles are carried out in B.9.3.6.6.

Main assumptions

Age groups

As the exposure differs depending on age, the population is divided into three age groups, (male) infants in the age of 6-12 months, (male) children in the age of 6-11 years and women. Infants at the age of 6-12 months are special as they are expected to mouth many articles not only toys. Furthermore, at this age they will stop being breast fed and will start to eat solid food. Children at 6-11 years are chosen as a group to represent school children, this group is comparable to the group 6-11 years old in the biomonitoring studies.

The exposure will differ within these groups, due to different behaviour or stage of development and due to difference in the articles and building materials used in the different homes.

Body weight

A body weight for infants at the age of 6-12 months of 9.2 kg (Höglund et al. 2012) is used in the exposure estimations. For children at the age of 6-11 years a body weight of 31.8 kg is used in the exposure estimates (Höglund et al. 2011) and for women, the default body weight of 60 kg is used, as women are the subpopulation of adults that may need protection from reproductive toxicants (ECHA 2010). The use of a body weight of 60 kg is in line with ECHA guidance R15 (2010)[57].

Table B24 Recommended Values for Body Weight for infants and children – as presented in the Exposure Factors Handbook (Höglund et al. 2012)

Age group	Mean (kg)	5th Percentiles	95th Percentiles
Infants	9.2	7.1	11.3
Children	31.8	19.7	52.5
Women	60	45	85

Internal dose estimates

The exposure estimates are converted to internal dose estimates (µg/kg bw/day). This is done by using the absorption rate of the four phthalates for oral and dermal absorption as given in the section on toxicokinetics (section B.4.1).

B.8.4.2. Exposure from indoor environment

B.8.4.2.1. Introduction

When semi-volatile organic compounds (SVOCs) such as the four phthalates are introduced into an indoor environment via e.g. vinyl floorings, wall paper, furniture and other articles

[57] In the later updated ECHA guidance R15, 2016 does not contain default values on body weight. The Guidance for biocides, Biocides Human Health Exposure Methodology (2015) recommends the following values for risk assessment: 8 kg for infants, 23.9 for children and 60 kg for adult women. These estimates are based on 25th percentiles and not the mean or median.

consisting totally or partly of PVC, they tend to redistribute from their initial location to all indoor surfaces, like e.g. furniture, dust, particles in the air and even hair and skin, to which surfaces they are sorbed (Weschler et al., 2008). Surfaces will then act as sinks (sorptive reservoirs) and thereby as sources for subsequent emission.

Normal house dust, originating from wear and tear of e.g. textiles, furniture, vinyl flooring and outdoor air, acts as a major sink in the indoor environment. This is due to the fact that dust consists of a huge number of small particles with very big surfaces to which SVOCs, such as the four phthalates, can easily sorb. It has been found that airborne particles increase the rate at which DEHP is transported between rooms by a factor 5 relative to gas-phase transport (Xu et.al, 2006), i.e. contains 5 times higher concentration than the air itself.

Because phthalate plasticisers are not chemically bound to the polymer matrix of e.g. vinyl flooring, and emission from the articles to air or other media usually occurs during the entire service life of the articles (Clausen et al., 2010; Xu et al., 2010), phthalates are among the most abundant contaminants in indoor air environment. Phthalates can persist indoors for years after they are introduced, even after the primary source is removed (Weschler et al., 2008).

Clausen et al. (2010) found that when the air flow rate is increased about 7 times, the specific emission rate (SER) is increased almost 6 times, and the system maintains an almost constant bulk air concentration despite different air exchange rates. Therefore, if the surface materials contain phthalates and the only mechanism of removal is normal ventilation, it is impossible to avoid the phthalates in indoor air, and the substances can persist indoors for months to many years (Weschler et al., 2008). When opening a window or vacuum cleaning the surfaces in a room, the concentration of SVOCs tend to fall immediately after, but will reach the earlier “steady state” concentration again after some time as there is always a tendency of gasses/vapours to strive to an equilibrium concentration in articles, in air, particles in air and in dust Only high-frequent cleaning of all indoor surfaces (incl. walls etc.) will possibly reduce the concentration of phthalates in the air. In our simulation cleaning of surfaces is not included.

Exposure of phthalates from indoor environment can happen through indoor air (gas phase and particles in air) or dust. The exposures from these two sources are estimated as the exposure from the indoor environment. The exposure to phthalates in dust is estimated in section B.8.3.4.1, the estimation of phthalates in air (gas-phase and particles) in section B.8.3.4.2 and finally the combined exposure to phthalates in air and dust is estimated in B.8.3.4.3.

B.8.4.3. Exposure to phthalates via dust

This section describes exposure to the four phthalates in a typical room environment. Phthalates are emitted as vapours from vinyl floor coverings, wall coverings and other PVC materials containing phthalates. The phthalate vapours then adsorb to suspended particles in indoor air (see EU Risk Assessments) but they also absorb to other surfaces like walls, carpets etc., from which they can be released as dust or vapours. In addition as PVC materials degrade during use, they will eventually start to release particles of PVC containing phthalate. Exposure to dust in indoor environments occurs through the inhalation of airborne dust, accidental

ingestion of settled dust and dermal contact with settled dust. Small quantities of dust are present on most indoor surfaces, that is readily transferred to hands on contact with surfaces leading to a low level of dermal exposure and also accidental ingestion of settled dust via hand-mouth contact (both subconscious hand-face contact and also while eating, drinking or smoking; EA 2009).

A key study was published by Langer et al (2010), who measured dust levels in 151 daycare centres and 500 children's bedrooms in Denmark in 2009. This study included the largest number of samples of the studies listed in Table B26. This study focused on levels in children's rooms (daycare centres and children's bedrooms) and may therefore be considered particularly relevant for estimating phthalate exposure of both children and women spending time in children's rooms. Langer et al. (2010) detected DEHP in dust samples from all sites (151 daycare centres and 500 children's bedrooms), while DBP, DIBP and BBP were detected in more than 75% of the bedrooms and more than 90% of the daycare centres. The dust levels of several phthalates (BBP, DBP and DEHP) were substantially lower than those measured in a comparable study conducted 6-7 years earlier in Sweden. Although usage patterns in Denmark differ from those in Sweden the current results may also reflect a change in the plasticizers that are used in common articles including toys.

The estimations of internal exposure from ingestion of dust is based on measured phthalate levels in dust (in µg/g dust) and an estimated intake of dust per day. Exposure estimates are calculated as internal exposures using the oral absorption fractions (100 %). Internal exposure from ingestion of dust is thus calculated using the formula below.

$$Exposure_{\text{int}ernal} = \frac{C_{phthalate} \times D_{oral} \times F_{oral}}{BW}$$

Where $Exposure_{internal}$ = internal exposure;
C_{phth} = phthalate concentration in dust;
D_{oral} = daily intake of dust;
F_{oral} = fraction of phthalate absorbed; and
BW = bodyweight.

Generally, median estimates of exposure are comparable between studies, and variations between studies observed may reflect differences in phthalate sources in the indoor environment studied.

For the dust intake the values recommended in ECHA guidance R15 referring to Oomen et al. (2008) are used. For infants R15 recommends a dust intake of 100 mg per day for risk assessment while it recommends a dust intake of 50 mg/day for children and adults.

Infants and small children are likely to have higher exposures to indoor dust than adults because they play on the floor leading to greater dermal contact with dust and are also more likely to put non-food items into their mouth.

The estimation of the phthalate concentration is based on measured dust concentrations found in the most recent studies presented in Table B26.

Hence, daily intake is based on an estimated intake of 0.1 g dust per day for infants (9.2 kg bw) and 0.05 g dust per day for children (31.5 kg bw) and women (60 kg bw). For the risk assessment a weighted median was calculated. The weighted medians are used for the risk characterisation. The ranges of the medians are also shown in brackets in Table B26. This is done to show that there are differences in exposure levels within EU Member States, as well as in other countries. This may be due to amongst other reasons cultural differences. As an example PVC floorings are used more widely in Sweden than in e.g. Denmark, and this can be seen in the Swedish study by Bergh et al. (2010) that reports the highest measurements of DEHP and DBP in dust. The exposure from dust seems to depend on the place of the study and will to a high degree reflect the use of articles in the room.

Only a few of the newer surveys has calculated the 95th percentile of the daily intake. Based on distributions found in studies before 2008 (ECHA 2012a) and the one new study (Fromme et al., 2013) where the 95th percentile has been calculated, the 95 percentile is estimated as 4 times the median (related to differences in dust levels).

Based on the individual phthalates the factor is close to 4 for DEHP and DBP while for DIBP and BBP is somewhat higher (approximately 7 and 8). However, especially for DIBP, only few studies are available. As a conservative estimate a factor 4 was used for all phthalates.

Table B25 Compilation of the factor resulting from the 95th percentile divided by the median intake estimates of phthalates from house dust

	Source	N	DEHP	DBP	DIBP	BBP
Pohner et al. 1997 (Germany)	Background Document 2012, Table 22 (ECHA 2012a)	272	4.4			
Butte et al. (Germany)		286	3.5	3.8		6.5
Becker et al. 2002 (Germany)		199	2.9	3.8		13.8
Clausen et al. 2003 (Denmark)		23	3.0			
Kersten et al. 2003 (Germany)		65	2.7	3.8		12.1
Fromme et al. 2004 (Germany)		30	2.2	2.3		7.3
Becker et al. 2004 (Germany)		252	3.6			
Bornehag et al. 2005 (Sweden)		346	5.3	3.8	6.9	4.4
UBA 2011 (Germany)		10	5.4			
Fromme et al. 2013 (Germany)	Appendix, Table B24	63	8.6	4.5	8.7	15.5
Simple average			4.2	3.9	7.8	9.9
Weighted average			4.1	4.1	7.2	8.2

Note: the weighting is based on the numbers of observations

Furthermore the 95th percentiles were adjusted for lower bodyweight for 5 percentile of the population assuming that the intake and inhalation are not depending on weight. The 5th percentile represent those for which the intake results in a higher exposure per kg weight.

Table B26. Intake estimates of phthalates from house dust in Europe (N = homes, daycare centres or working places). Gray-shaded values are selected for the risk assessment.

			Phthalate concentration (µg/g dust)		Daily intake (µg/kg bw/day)					
					Infants		Children		Women	
Study	**Country**	**N**	**Median**	**95th p**	**Median ***	**95th p**	**Median**	**95th p**	**Median**	**95th p**
DEHP										
Abb et al. (2009)	GER	30	604		6.57		0.95		0.50	
Langer et al. (2010) in homes	DEN	500	210		2.28		0.33		0.18	
Langer et al. (2010) day care centres	DEN	151	500		5.43		0.79		0.42	
Bergh et al. (2011) in home	SWE	10	680		7.39		1.07		0.57	
Bergh et al. (2011) day care	SWE	10	1600		17.39		2.52		1.33	
Bergh et al. (2011). Work	SWE	10	1100						0.92	
Fromme et al. (2013)	GER	63	888	7616	9.65	107.3	1.40	19.33	0.74	16.48
Personal communication UBA, 2011***	GER	10	310	1680 (90-p)	3.37	23.66	0.49	4.26	0.26	3.64
Blanchard et al. (2014)	FRA	30	289		3.14		0.45		0.24	
Weighted arithmetic average[58],					3.94	20.42	0.57	3.68	0.31	1.65

[58] The arithmetic mean is considered to be the appropriate mean, as the values from the different studies are comparable. Even if the distributions of the underlying data are log normal distributed, it is considered to be statistically correct to use the arithmetic average of the medians of the different studies as an estimated of

DBP										
Abb et al. (2009)	GER	30	87		0.95		0.14		0.07	
Langer et al. (2010) in homes	DEN	500	15		0.16		0.02		0.01	
Langer et al. (2010) day care centres	DEN	151	38		0.41		0.06		0.03	
Bergh et al. (2011). Home	SWE	10	130		1.41		0.20		0.11	
Bergh et al. (2011). Day care	SWE	10	150		1.63		0.24		0.13	
Bergh et al. (2011). Work	SWE	10	100						0.08	
Fromme et al. (2013)	GER	63	21	95	0.23	1.34	0.03	0.24	0.02	0.21
Blanchard et al. (2014)	FRA	30	11,9		357	0,13	0,02	0,00	0,01	0,00
Weighted arithmetic average					0.28	1.47	0.04	0.27	0.02	0.12
DIBP										
Study	**Country**	**N**	**Median**	**95-p**	**Median**	**95-p**	**Median**	**95-p daily intake**	**Median**	**95-p**
Langer et al. (2010) in homes	DEN	500	27		0.29		0.04		0.02	
Langer et al. (2010) day care centres	DEN	151	23		0.25		0.04		0.02	
Bergh et al. (2011). Home	SWE	10	4		0.04		0.01		0.003	

the average value on European level. Blanchard et al. (2014) was not included in the Annex XV report and therefore not included in the calculation of weighted averages, used further in the risk characterisation. Had it been included the average median daily intake would have been 3.91 (DEHP), unchanged for children and 0.30 for women

Bergh et al. (2011). Day care	SWE	10	3		0.03		0.005		0.003	
Bergh et al. (2011). Work	SWE	10	37						0.03	
Fromme et al. (2013)	GER	63	20	174	0.22	2.45	0.03	0.44	0.02	0.38
Blanchard et al. (2014)	FRA	30 30	11,9 19	357	0,20 0,13	0,00	0,02 0,03	0,00	0,01 0.02	0,00
Weighted arithmetic average					0.27	1.41	0.04	0.25	0.02	0.11
BBP										
Abb et al. (2009)	GER	30	15		0.16		0.02		0.005	
Langer et al. (2010) in homes	DEN	500	3.7		0.04		0.006		0.003	
Langer et al. (2010) day care centres	DEN	151	17		0.18		0.03		0.01	
Bergh et al. (2011). Home	SWE	10	17		0.18		0.03		0,01	
Bergh et al. (2011). Day care	SWE	10	31		0.34		0.05		0.03	
Bergh et al. (2011). Work	SWE	10	8.8						0,01	
Fromme et al. (2013)	GER	60	6	93	0.07	1.31	0.01	0.24	0.005	0.2
Blanchard et al. (2014)	FRA	30 30	11,9 8.5	357	0,09 0,13	0,00	0,02 0.01	0,00	0,01 0.01	0,00
Weighted arithmetic average					0.08	0.42	0.01	0.08	0.014	0.03

B.8.4.4. Exposure to phthalates from indoor air

As earlier described, semi-volatile organic compounds (SVOCs) tend to redistribute from their initial location (source) to all indoor surfaces, dust, and particles in the air. For DEHP, Xu et al. (2009) describes that after introduction of a phthalate containing source into a room, the air concentration reaches a steady level after about one and a half years. As long as the source of the phthalate is still available, release and steady-state will re-establish also after ventilation and vacuum cleaning.

Afshari et al. (2001) shows that the concentration of phthalates in indoor air is independent of ventilation rates and the area of surface materials containing plasticizers, i.e. a small area of plasticizer containing products emits almost as much as a large area. Therefore, if the surface materials contain plasticizers, it is impossible to avoid the phthalates in indoor air.

To estimate the exposure to phthalates in indoor air, this estimation has to include both the concentration of phthalates in the gas-phase as well as the phthalates in airborne sorptive reservoirs like airborne particles.

B.8.4.4.1. Exposure estimation

To get an impression of the contribution of single sources e.g. furniture, toys and flooring materials to the indoor air environment, indoor air concentrations are estimated for two room scenarios and compared to results from calculations based on data from the EU Risk Assessment Report on DEHP (EU RAR 2008) and literature data on concentrations in air. Table B27 summarises the concentrations of DEHP in indoor air found by simulations, calculations and literature findings.

EU Risk Assessment Report

A calculation has been made by applying the method developed in the EU Risk Assessment Report on DEHP (EU RAR 2008). In the calculation described in EU RAR the only phthalate source in the room is DEHP emitted from the vinyl flooring and the wall paper covered with plasticiser. The result of the calculation (9.4 µg/m^3) is 10 fold higher than the concentrations found in the simulations and in the other references, and it exceeds 5.3 µg/m^3, which is the saturated vapour pressure of DEHP at 20° C. All these levels are brought further to Table B27.

Literature

Larsen et al. (2007) made a review of measured indoor air and particle in air concentrations of DEHP in studies carried out in USA, Denmark, Poland, Japan and Germany. The average mean concentration in these studies is 0.23 µg/m^3, and the average maximum concentration is 1 µg/m^3. These levels are brought further to Table B27.

Indoor air – simulation of gas phase

Two scenarios were simulated: one children's play room and one bathroom. The children's room assumes presence of vinyl flooring, wall paper, an air mattress, a chair covered with artificial leather, and a balance ball containing DEHP[59]. The bathroom assumes presence of vinyl flooring, wall paper and a shower curtain with DEHP. The concentration of DEHP in the articles used as input in the modelling comes from samples taken from the Danish market in 2001 and 2010 (Danish EPA, 2001 and Danish EPA 2010a). Appendix B1 explains the calculations made.

Only results from DEHP are used in the risk assessment and are reported in Table B27. The concentration of the other phthalates in the gas-phase were negligible. However, the limited number of samples may not have been representative for the Danish market, and the Danish market is not representative for the EU. Thus, the exposure to DBP, DIBP and BBP is not taken into consideration in the exposure estimates from indoor air.

Table B27 Summary – Concentrations in indoor air for DEHP (µg/m^3)

Simulations, based on Danish Data, gas phase only		**Calculation according to EU RAR (2008a), gas phase only**		**Literature levels of DEHP (sum of gas phase and particles in air) in indoor air (including offices, kindergartens and workplaces)**	
Children´s play room, typical case	0.16	Children´s play room	9.4	Larsen et al. (2007) max.	1
Children´s play room, reasonable worst case	0.81	DEHP saturated vapour pressure, 20° C	5.3	Larsen et al. (2007) mean	0.23
Bathroom, typical case	0.26				
Bathroom, reasonable worst case	0.8				

It should be stressed that the higher concentrations of phthalates in the air in the reasonable worst case scenarios are solely caused by the higher concentrations of phthalates in vinyl flooring and wall paper samples taken in 2001. The other items assumed to be present in the room are the same in both the reasonable worst case and the typical case scenarios (i.e., an air mattress, a chair covered by artificial leather, a balance ball, and a shower curtain).

For the typical case, the simulations are based on a limited number of samples taken in Denmark in 2010 of vinyl flooring (n=16) and wall paper (n=15). In these samples, the four phthalates were not present or only present at very low concentrations. Thus, in the typical case it is assumed that flooring and wallpaper practically do not significantly contribute to the indoor air gas phase and the concentration mostly comes from the other articles present in the room. Therefore, the result for the typical case might be an underestimate. Simulations

[59] Even if DEHP is not the only phthalate in the indoor air, as also reported for France (2011 data) in Blanchard et al. (2014), the simulations below are based on the data from analysed articles containing very low concentration of DBP, DIBP and BBP, and where only the exposure to DEHP was of any significance.

regarding DBP, DIBP and BBP assuming higher levels of these phthalates in the articles would lead to more significant exposure estimates from indoor air also for these phthalates.

As can be seen from the table above, the result of the EU RAR calculation, applied to the simulated Children's playroom scenario (9.4 µg/m^3) is 10 fold higher than the concentrations found in the simulations and in the literature (which also comprise particles in air), and twice the level of the saturated vapour pressure of DEHP at 20° C. This is probably due to the fixed emission rate of 3 x 10^{-4} µg/m^2/s used in the EU RAR. The actual emission rate is not constant and depends on the concentration in the room air, the time, the sinks (sorptive reservoirs), the material concentration etc. In our calculations, the initial emission rate is approximately 0.0025 µg/m^2/s and drops in the course of time, when the room air becomes saturated. This "steady-state" saturation is not accounted for in the method described in the EU RAR.

In conclusion, as more time is spent in a children's room than in a bathroom, a **gas phase** level of DEHP of **0.81 µg/m^3** from the children's play room can be selected for the "reasonable worst case", and **0.16 µg/m^3** as the "typical case". Only DEHP is taken into consideration as it by far dominated the air concentration of phthalates. Thereby the contribution from other phthalates might be understated.

Indoor air – particles in air

To compare the measured DEHP levels found in the literature with the air levels from the simulations, it is necessary to estimate the concentration of DEHP in particles in air, as the measured level is the sum of gas phase and particles in air.

Xu et al. (2006) estimated that about 80% of airborne DEHP in, for example, indoor settings are associated with particles (total suspended particles) in the air. Airborne particles increase the rate at which DEHP is transported from the emitting surfaces and between rooms by a factor 5 relative to gas-phase transport. DEHP also desorbs very rapidly from the particles. Both DBP and DEHP can be found among the SVOCs that typically have the highest airborne concentrations (sum of gas and particle phases). DEHP, DBP and BBP belong to the group of SVOCs typically having the highest concentrations in particles in air due to their relatively large abundance and low vapour pressure.

Based on a predicted indoor air DEHP concentration at steady state of 0.15 µg/m^3, Xu et al. (2009) predicted the particle DEHP concentration in the air at steady state to be 0.75 µg/m^3 (5 x 0.15 μg/m^3).

A reasonable worst case scenario resp. typical scenario concentration of DEHP in indoor air, at 20° C can be calculated using the steady state gas phase concentrations of 0.81 resp. 0.16 µg/m^3. We assume 0.81 µg/m^3 to be a reasonable worst case gas phase level and 0.16 µg/m^3 to be a typical level in a children´s play room in a relatively new building. Since the Xu et al. (2006, 2009) study found that airborne particles increase the rate at which DEHP is transported between rooms by a factor 5 relative to gas-phase transport (i.e. 5 times more DEHP is bound to airborne particles than is to be found in the gas-phase of the indoor air), it can be inferred that the reasonable worst case of air concentration of DEHP adsorbed to airborne particles is **4 µg/m^3** (5 x 0.81) and the typical case **0.8 µg/m^3** (5 x 0.16=). This is further supported by Fromme et al. (2013), confirming that the percentage of DEHP in air

compared to other phthalates (DiDP, SiNP, DBP, DIBP, DEP and DMP) is 15%, while the percentage of DEHP in dust is 70 %.

These levels are 5-fold higher than the mean and maximum average levels found in the literature. Part of the difference may be due to the difference in furniture and coverings (PVC flooring and wallpaper).

Overall indoor air exposure – gas phase and particles in air

Inhalation exposure through gas/vapour or particles in air were estimated by the following equation:

$$\text{Gas/vapour/particles (µg/kg bw/d)} = \frac{Y * IR * ED * CFi * Rf}{Bw}$$

Where
Y (µg/m^3) = phthalate concentration in inhaled air (air or particles in air);
IR (m^3/d) = inhalation rate;
ED (hr/d) = exposure duration;
CFi (d/hr) = unit conversion factor of 1/24;
BW (kg) = bodyweight and
Rf = respirable fraction.

The calculation is furthermore based on the following assumptions:

	Infants	**Children**	**Women**
Body weight (kg)	9.2	31.8	60
Respiration rate (m^3/d)	7	14	18
Exposure duration (h/d)	21.93	20.73	19.32
Respirable fraction of inhaled substance	1	1	0.75

The respiration volumes are taken from REACH Guideline R15 (2011)[60], Annex R15-5, table R15-1. The exposure durations are taken from US EPA- EFH, 2009, table 16-21 and 16-22.

Internal exposure estimates (µg/kg bw/day)	**Age**	**Median, average "typical case"**	**95th percentile, average "reasonable worst case"**
DEHP, air	Infants Children Women	0.11 0.06 0.03	0.56 0.31 0.15
DEHP, particles in air	Infants Children Women	0.56 0.30 0.14	2.78 1.52 0.72
DEHP, indoor air total	Infants Children Women	0.67 0.37 0.17	3.34 1.83 0.87

For the purposes of risk assessment, RIVM (Oomen et al. 2008) has indicated that concentrations of airborne dust in indoor air are 60 µg/m^3 in homes and moderately crowded places and 100 µg/m^3 in crowded places. In current restriction report 60 µg/m^3 is used for the typical case, while 100 µg/m^3 is used for the reasonable worst case.

The total concentration of phthalates in air (the sum of gas phase concentration of DEHP and DEHP associated to airborne particles) may then be approximately **1 µg/m^3** in the typical case (0.8 + 0.16 µg/m^3) and **4.8 µg/m^3** in the reasonable worst case (4 + 0.81 µg/m^3).

B.8.4.4.2. Overall weighted average of internal exposure to phthalates (in µg/kg bw/day) from indoor environment

Table B28 brings together all calculated weighted average of internal exposure to phthalates from indoor environment (air, particles in air and dust) to be used in B.9.2 as a basis for the estimation of the risk characterisation ratios.

[60] The later updated ECHA guidance R15, 2016 does not contain default values.

Table B28 Internal exposure estimates (µg/kg bw/day) from dust ingestion, and for DEHP also inhalation of phthalates via air and particles in air

	Infants		Children		Women	
	Typical case	**Reasonable worst case**	**Typical case**	**Reasonable worst case**	**Typical case**	**Reasonable worst case**
DEHP (dust)	3.94	20.42	0.57	3.68	0.31	1.65
DEHP (dust + air)[61]	4.61 (4.22)	23.76 (21.85)	0.94 (0.93)	5.51	0.48	2.52
DBP	0.28	1.47	0.04	0.27	0.02	0.12
DIBP	0.27	1.41	0.04	0.25	0.02	0.11
BBP	0.08	0.42	0.01	0.08	0.01	0.03

The public could also be exposed dermally through dust, but this will not be handled as part of the dermal exposure, as the dermal exposure via dust is expected to be relatively low.

B.8.4.4.3. Uncertainties on the measurements and estimations of exposure from indoor environment

The modelling for exposure via indoor air used the concentration measurements of DEHP in articles from samples taken from the Danish market. Due to limitations in the amount of available data, these are supplemented with default values and data from the literature. The simulations are based on emissions from articles containing very low concentrations of DBP, DIBP and BBP and where only DEHP contributed significantly. However, e.g. Blanchard et al. (2014) measures the content of DBP in airborne particles to be 40% of the DEHP level and DIBP to be 70% of the DEHP level, while the content of BBP was about 5% of the DEHP level. Therefore, also inhalation of DBP, DIBP and BBP via air and particles will contribute to the risk.

The estimates from ingestion of phthalates in dust depends on the concentrations of phthalates in dust. This concentration is expected to depend on the articles to be found in the indoor environment containing phthalates. If a study of the content of phthalates in dust was made in an indoor environment with a high number of articles containing phthalates, this could overestimate the exposure. Similarly, the exposure could be underestimated if only a small number of articles containing phthalates were in the room. The scenarios are based on weighted averages of the median values, meaning that some measurements of phthalates in dust have shown higher concentrations and therefore some populations will have a higher exposure of phthalates from dust than calculated here.

Table B26 shows large differences of levels of the phthalates in house dust. Some of this might reflect differences in dust sampling methods, but others reflect real differences. In some Member States the level might also be higher than others.

[61] The values between brackets are used in the further calculations of the exposure of DEHP via air (air and particles in air). The values between brackets are based on WHO default values for body weight and not the default values from US EPA. The US EPA default values are based on more recent data than the data from WHO. However, this inconsistency leads to minimal differences and the risk estimate would only be marginally higher if the US EPA default body weight values had been used.

As mentioned above, both for the typical case as well as for the reasonable worst case, the estimations of dust intake are based on ECHA Guidance R15 (ECHA 2010) which recommends the use of 100 mg/day when calculating infant exposure from dust and 50 mg/day for children and adults.

The recommendations are based on RIVM (Oomen et al. 2008) which based on expert judgement considers these as conservative but realistic estimates of dust ingestion. This expert judgement is based on two arguments.

First, it is assumed that children ingest on average 100 mg soil per day via hand-to-mouth behaviour (Lijzen et al. 2001; Otte et al. 2001). When playing outside, a child's hand is much more loaded with soil than a loading with house dust during indoor playing. The ingestion of soil per time unit will thus be much greater outdoors than indoors. On the other hand, children spend more time indoors than outdoors. Yet, it is very unlikely that average daily dust ingestion will be greater than average daily soil ingestion.

Second, a number of surveys show that the dust ingestion of children is between 20 and 200 mg/day, whereas in most cases 100 mg/day is used as upper level. For adults, in most cases about 50 mg/day was derived as an upper estimate.

Data from recent publications indicates that the typical case might be lower. Wilson et al. (2013) indicates that levels of dust intake of 100 mg/day resp. 50 mg/day are reasonable worst case levels and assumes for 60 mg/day for infants and 30 mg/day as typical levels of dust intake per day. The estimation of are based on the following factors:

- particle loading to indoor surfaces;
- fraction transferred to the hands;
- hand surface area;
- fraction of hand surface area that may be mouthed or contacted to food;
- frequency of hand-to- mouth events, amount dissolved by saliva; and
- exposure time.

The calculations were adapted specifically for Canadian context, estimated mean indoor dust ingestion rates range from 2.2 mg/day for teenagers to 41 mg/day for toddlers.

Furthermore, analysis made by Bierkens et.al (2011) indicates an average dust ingestion rate for children is below 100 mg/day, most probably around 50 mg/day.

EA (2009) has estimated a dust/soil ingestion intake of 25 mg/day for adults and 100 mg/day for small children, while the US EPA 2011 Exposure Factors Handbook (US EPA 2011) indicates that the typical intake of settled dust for babies of 6 weeks to 1 year, children of 1 to <21 years and adults are 30, 60 and 30 mg/day respectively with an upper percentile (reasonable worst case estimate) of 100 mg/day for children aged between 3 and <6 years.

Lower estimations of dust intake will result in nearly a proportional lower estimation of the exposure from the indoor environment. E.g. if the dust intake is as found by Wilson et.al the intake and inhalation will be as listed in Table B29.

Table B29 Dust ingestion and for DEHP also inhalation of phthalates via air and dust – assuming typical daily dust intake of 60 mg for infants and 30 mg for children and Women

	Infants		Children		Women	
	Typical case	**Reasonable worst case**	**Typical case**	**Reasonable worst case**	**Typical case**	**Reasonable worst case**
DEHP	2.65	21.85	0.71	5.51	0.36	2.52
DBP	0.17	1.47	0.02	0.27	0.01	0.12
DIBP	0.16	1.41	0.02	0.25	0.01	0.11
BBP	0.05	0.42	0.01	0.08	0.00	0.03

Some studies have shown that humans could be exposed to phthalates in the air through dermal exposure. Weschler et al. (2015) investigated the dermal uptake of DEP and DBP from air. The concentration of DEP and DBP in the air was relatively high and therefore not comparable with normal air concentrations of phthalates. However, at these relatively high concentrations of DEP and DBP in the air the conclusion from the study was that dermal exposure from air could be an important exposure route. The exposure was measured by urine samples from six adult males staying in the room with six aluminium plates painted with latex paint spiked with DEP and DBP. The subjects only wore shorts, had a restricted diet and had a restricted use of personal care products before and after entering the chamber. The study was made at elevated air concentrations but did show that the dermal uptake of phthalates from air could be an important exposure pathway.

B.8.4.5. Exposure from food

B.8.4.5.1. Contamination and legislation

An important source of exposure to the four phthalates is via intake of food. Food may be contaminated via:

- Food contact materials (FCMs)[62], such as food packaging and articles that are used during the processing of food;
- Non-FCM articles that may come into contact with food, e.g., table mats and oilcloth for tables;
- Non-compliant FCMs; and
- The environment: environmental release of phthalates occurs from phthalate manufacturing plants (DEHP and DBP only), from downstream use of phthalates (DEHP

[62] Food contact materials are either intended to be brought into contact with food, are already in contact with food, or can reasonably be brought into contact with food or transfer their constituents to the food under normal or foreseeable use. This includes direct or indirect contact. Examples include:
- containers for transporting food
- machinery to process food
- packaging materials
- kitchenware and tableware

and DBP only) and from the article service life[63] (including the waste stage). This may lead to contamination of plant and animal based food sources.

Since 2008, DEHP[64], DBP[65] and BBP[66] are authorised to be used in food contact materials with Specific Migration Limit (SML) of resp. 1.5, 0.3 and 30 mg/kg food and Quantifications maximum (Qm) of resp. 0.1, 0.05 and 0.1% in the material.[67] The total SML is 60 mg/kg for DEHP, DBP, BBP, DINP, DIDP, and 15 other substances. DIBP is not authorised for use in FCMs. The SMLs for DEHP and DBP only allocate 50% of the TDI to exposure from FCM to make room for other sources of these phthalates in the total exposure.

Table B30 of "classical" phthalates in the fourth amendment to the plastics directive: Survey of the critical parameter to control in enforcement work.

		SML	**Qm**	**Parameter to control in single use Food Contact Material *)**			**Parameter to control in repeated use Food Contact Material**			**Limit in fatty food simulant **)**
PM-no	Substance	(mg/kg food simulant)	(% in the plastic)	Fatty food	Infant food	Non-fatty food	Fatty food	Non-fatty food	Infant food (non-fatty)	(mg/kg simulant D)
74560	Phthalic acid, benzyl butyl ester (BBP)	30	0.1	Qm		SML	SML			30-150
74640	Phthalic acid, bis(2-ethylhexyl)ester (DEHP)	1.5	0.1	Qm			Qm	SML		Not of relevance
74880	Phthalic acid, dibutyl ester (DBP)	0.3	0.05	Qm			Qm	SML		Not of relevance
75100	Phthalic acid, diester with C8-C10 (DiNP)	9 (SML(T) incl. DiDP)	0.1	Qm		SML	SML			9-45
75105	Phthalic acid, diester with C9-C11 (DiDP)	9 (SML(T) incl. DiNP)	0.1	Qm		SML	SML			9-45

*) Usually packaging made from glasses with lid containing a plasticized gasket is considered a single use material.
**) Taking D-reduction factor in consideration (info for planning of method validation). When simulant D is 50% ethanol no reduction factor is of relevance.
Confirmed by the network of reference laboratories and the Commission working group on food contact materials.

[63] For example runoff water from roofing or from farm equipment that is not covered by FCM legislation may contribute to environmental contamination of food.

[64] Only to be used as: (a) plasticiser in repeated use materials and articles contacting non-fatty foods; (b) technical support agent in concentrations up to 0.1 % in the final product.

[65] Only to be used as: (a) plasticiser in repeated use materials and articles contacting non-fatty foods; (b) technical support agent in polyolefins in concentrations up to 0.05 % in the final product.

[66] Only to be used as: (a) plasticiser in repeated use materials and articles; (b) plasticiser in single-use materials and articles contacting non-fatty foods except for infant formulae and follow-on formulae as defined by Directive 2006/141/EC or processed cereal-based foods and baby foods for infants and young children as defined by Directive 2006/125/EC; (c) technical support agent in concentrations up to 0,1 % in the final product.

[67] Commission Regulation (EU) 10/2011 repealed Directives 80/766/EEC, 81/432/EEC, and 2002/72/EC from 1 May 2011. Commission Directive 2007/19/EC amended Directive 2002/72/EC and required Member States to adopt provisions to prohibit the manufacture and importation into the Community of plastic materials and articles intended to come into contact with food which do not comply with restrictions and specifications for phthalates from 1 June 2008.

No specific legal concentration limits exist for the four phthalates as environmental contaminants in food. Member States may measure phthalates in food and may act on the basis of Article 14(8) in the general food law (Regulation (EC) No 178/2002)[68]. However, measurement of phthalates in food is technically complicated, no standardised methods exist, and there are no legal concentration limits to comply with. This perhaps explains the scarcity of measurement data for phthalates in food.

Thus, summarising the legal requirements, the legislation is not targeted at limiting phthalate content in food per se but allows DEHP, DBP and BBP to be present in FCMs as long as the SMLs are met. At least in principle, repeated contact with compliant FCMs and sources of contamination may result in significant phthalate levels in food. From the limited available measurements of phthalates in food, the four phthalates are found in most food samples.

B.8.4.5.2. Measurements of phthalates in food

Studies show that phthalates are found in food samples and that the concentration vary in different samples.

A research project in Belgium reported in Fierens et al. (2012), Van Holderbeke et al. (2014) and Sioen et al. (2012) studied the concentration of phthalates in 400 food samples bought in Belgium in the period of May 2009 until June 2010. Levels of phthalates were investigated in different food groups.

DEHP was the most detected phthalate, identified in 81 % of all samples, followed by DIBP (75 %), DBP (69 %) and BBP (58%). The highest measured concentrations were for DEHP with a maximum in fish and fish products of 5932 µg/kg fresh weight with a median of 36.0 µg/kg. High levels of DEHP were also found in:

- Fat and oils (maximum of 1827.0 µg/kg fresh weight; median of 93 µg/kg),
- Condiments and sauces (maximum of 2154.0 µg/kg fresh weight; median of 44.0 µg/kg),
- Cereals and cereal products (maximum of 2264.0 µg/kg fresh weight; median of 49.0 µg/kg), and
- Fruits and vegetables (maximum of 1413.0 µg/kg fresh weight; median of 16.0 µg/kg fresh weight).

Several of the maxima are above the SML of 1.5 mg/kg food for DEHP. DEHP was detected in 35 of 37 tested meat and meat products (with concentrations from 10.0-850.0 µg/kg fresh weight) and in all tested packaging materials (30 samples with concentration from 1.1-482.0 µg/kg).

High levels of DIBP were found in Cereals and cereal products (maximum of 1383.0 µg/kg fresh weight; median of 6.0 µg/kg).

[68] *"8. Conformity of a food with specific provisions applicable to that food shall not bar the competent authorities from taking appropriate measures to impose restrictions on it being placed on the market or to require its withdrawal from the market where there are reasons to suspect that, despite such conformity, the food is unsafe."*

Concentrations for other phthalates and food groups were generally quite low. The lowest concentrations of phthalates were detected in baby food and aqueous-based packaging.

In the tested packaging materials, especially in cardboard, the phthalate concentration was primarily due to the presence of DIBP perhaps indicating that DIBP was present in the printing inks and lacquers of food contact materials.

Fromme et al. (2013), analysed 171 diet samples from children for phthalates. All samples contained DEHP and DIBP in concentrations from 4.3-131 µg/kg fresh weight and 1.2-163 µg/kg fresh weight, respectively. Detailed information of the concentration of phthalates in the diet samples are not given.

Sakhi et al. (2014) measured the content of phthalates in foods and beverages in Norway. 37 different food and beverages were analysed for the content of different phthalates. DIBP was found in 67 % of the food samples, DEHP in 65 %, DBP in 62 % and BBP in 30 %. The highest measured concentration of DEHP was found in Fats with a maximum of 323 µg/kg fresh weight and a median of 221 µg/kg. High levels of DEHP were also found in Ready to eat food (maximum of 235 µg/kg fresh weight; median of 136 µg/kg), Milk and dairy products (maximum of 173 µg/kg fresh weight; median of 126 µg/kg) and in single Meat and meat products (maximum of 117 µg/kg fresh weight; median of ND).

In 2014 The Danish Food Authority (Danish Food Authority, 2014) analysed a total of 58 meat samples from the slaughter house and packed meat products to investigate environmental contamination of phthalates and contamination from food contact material. 58 meat samples were analysed for metabolites of DEHP and DBP. Only one sample (vacuum packed beef) showed low concentration of mBP of 63 µg/kg. DEHP and DBP were not detected in any of the samples above the detection limit of 48 µg/kg. It is therefore concluded that the concentration of the metabolite of DBP must originate from the metabolism before the slaughter of the animal. The conclusion of the Danish study is that neither contamination of meat from food contact material nor from the environment seems to be a big problem in Denmark. These results are in contradiction with the data from Fierens et al. (2012) where DEHP were detected in all tested meat and meat products.

A study by Gärtner et al. (2009) analysed the migration of phthalates in infant food packed in recycled paperboard, and this study shows that phthalates and especially DIBP can still be found in infant food collected in the beginning of 2009 after the entry into force of the FCM legislation restricting phthalates in food contact materials. The concentrations are though considered as low and the median concentration in the food is 20.3 ng/g. Serrano et al. (2014) reviewed 35 studies reporting phthalate concentration in food. This review showed that meats, oils and fats and dairy (creams) have high concentrations of phthalates. For poultry this means that more than half of the mean DEHP measurements were higher than 300 µg/kg, while concentrations of other phthalates in general were low. For oils and fats more than half of mean DEHP concentrations were ranging from 404 to 5,591.7 µg/kg. Foods with low concentrations of phthalates were yogurt, milk and eggs; pasta, noodles and rice; fruits and vegetables and beverages and water. Phthalates were found in these food products, but in low concentrations. Seafood; bread and cereal products and spices showed varied concentrations of phthalates in the different studies reviewed.

Van Holderbeke et al. (2014) investigated the contamination pathways of phthalates in food sold in Belgium. Bread was one of the food samples investigated and the result showed that there are several sources of contamination. The contamination mainly takes place during processing, either by the use of contaminated ingredients or by the use of packing equipment containing phthalates. The distribution of phthalates within food was investigated as well, showing that phthalates were uniformly distributed in bread, soft goat's cheese, salami and semi-soft cheese samples but not in apples. The food products except from the apple are all processed in the same manner, indicating that food processing is introducing phthalates to Belgian food.

In conclusion, investigations of phthalates in food show that phthalates are found in many different food samples and the concentrations vary strongly in the different samples. In some samples phthalates cannot be determined. There may also be a variation between countries, as DBP and DEHP do not seem to be a problem in Danish meat, DEHP were detected in all 35 of 37 samples of meat and meat products from Belgium. In Norway DEHP were found in 3 of 8 meat samples (Sakhi et al., 2014). Investigations indicate that there are differences in the concentration of phthalates within countries.

B.8.4.5.3. Market surveillance of FCM legislation

Market surveillance data from Denmark in 2013 (Danish Food Authority 2013) on food contact materials and oils showed non-compliant food contact materials on the market. The market surveillance covered plastic gloves, sealants in metal screw caps, cooking oil and manufacturing equipment with soft plastics, such as hoses for food and conveyor belt. The market surveillance showed 27% of the samples contained one of the four phthalates and more than 1/3 were non-compliant due to the content of phthalates or the migration of phthalates. Phthalates detected in the food contact materials were DEHP, DBP and in one case BBP. DINP and DIDP were detected as well. In the cooking oils high concentration of DEHP were found especially in one (55.1 mg/kg) of the cooking oils and it was concluded that there could be a risk for sensitive groups. However, DEHP was not detected in the plastic cap of the lid of the oil, which were also analysed. This indicates that DEHP comes from the production of the oil or it could be an environmental contamination (Danish Food Authority 2013).

Table B31 Market surveillance data from Denmark in 2013 (Danish Food Authority 2013) on food contact materials and oils

Sample	Number of samples	Samples with phthalates	Samples with concentration of phthalates above Qm	Samples with phthalates above SML	Concentration in %	Migration in mg/kg
Conveyor belt	12	10	4	4	High concentration of DINP in four belts and BBP in one belt – not quantified 0.26-0.78 (DBP in two belts) 0.25-0.52 (DEHP in two belts)	9.1-30 (DINP in oil in four belts)
Hoses	7	4	1	1	25.7 (DIDP)	High in oil (not quantified) 95 (DIDP in 50 % ethanol)
Sealants in metal screw caps	6	0				
Gloves	5	3	2		13.9 DINP 2.97 DIDP	
Oils	2	2			Not detected in the lid	55.5 (DEHP in oil) 0.78 (DEHP in oil)

In 2012 Cyprus analysed 8 duplicate samples of plastic gaskets of glass jar lids, to determine phthalate esters. The glass jars could be used for all kinds of foods including fatty foods. In one of the samples DEHP were detected in 19.20% and DBP in 3.55% in the plasticised material. In the other jar DEHP was detected in 29.05% of the plasticized material (Personal communication, 2015).

A not yet published Nordic enforcement project on food contact materials shows that 4 of 19 food contact materials were non-compliant due to content of DEHP or DBP. The phthalates were found in conveyor belt, gloves for repeated use and a hose. The FCM were taken from The Faroe Islands, Iceland, Finland, Norway, Sweden and Denmark in the period from Autumn 2013 to Winter 2014.

A coordinated European enforcement on lids from 2011, showed a migration of DEHP in 4% of the samples (a total of 415 samples from 21 Member States). It is not clear from the reference whether the laboratory also analysed for DBP, DIBP and BBP. This study shows that migration limits for food contact materials is not always met and due to non-compliant food contact materials (McCombie et al., 2011) there may be an exposure.

Petersen and Jensen, 2010, have reported on two enforcement campaigns in Denmark just after the entry into force of the legislation of phthalates in FCM. The campaign were made in the Winter 2008/2009 and again in autumn 2009. Samples were taken from FCM producers, FCM importer and importers of packed foodstuffs from third-party countries. The results of the enforcement showed that more than 20% of the analysed samples contained DBP or DEHP in concentration above 0.05% and 0.1%, respectively. The highest concentrations of DEHP were found in the gasket from a range of different mango chutneys packed in India, where the concentration was between 29-41%. The trend from the first to the second campaign is not reported except from 25% non-compliant gaskets from glass-packed fatty foodstuff, because of high concentrations of DEHP and DBP in the first campaign and no non-compliant products in the category in the second campaign.

Exposure of phthalates can also happen from non FCM articles but which may still come into contact with food. As an illustrative example cheese left on a dinner mat with DEHP prior to consumption by a child could lead to an exposure of 0.033 mg/kg bw/day[69]. This example illustrates that occasionally high exposure levels may be reached from food contamination from articles not necessarily intended to come into contact with food[70].

In conclusion market surveillance activities show the SML for especially DEHP and DBP is often exceeded. This will lead to an exposure of the population from food even though regulation is in place. Other articles can in some cases also lead to an exposure via food if the food comes into contact with such articles as is the case with the examples with the dinner mat explained above.

B.8.4.5.4. Exposure from food, estimates

Data from the background document from 2012 and new literature data (shaded green) from after the entry into force of the limits in food contact materials are presented in Table B32.

Sioen et al. (2012) based the exposure calculations on analyses of phthalates in food products sold on the Belgian market between 2009 and 2011. 388 food products were analysed. The food products analysed were selected based on consumption data from a Belgian food consumption survey and based on the likelihood on finding phthalates in the food products. The exposure of phthalates was calculated for preschool children (2.5 to 6.5 years old) and for adults (≥ 15 years old). 12 different exposure scenarios are presented in the paper, and the medium bound probabilistic scenario, taking the preparation of food into account, is assumed to be the most realistic scenario and therefore the exposure calculations for mean and 95th percentiles are presented in Table B32.

Fromme et al. (2013) calculated the exposure of food for children in the age of 15-21 months. Exposure calculations were based on measurements of duplicate diet samples collected over 7 consecutive days. The food samples were collected from 25 German infants in the period from

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69 A Danish market surveillance on the information requirements for SVHC substances in articles revealed 10.5 % DEHP in a dinner mat. Using the software “Migratest Lite” rev 2004, an estimation of the concentration of DEHP in a slice of cheese (7.5 cm x 7.5 cm) after one hour on the dinner mat, was made. Based on this result and the assumption that a child of 10 kg would eat the cheese after one hour on the table cloth, the exposure was estimated to be 0.033 mg/kg bw/day. The scenario was developed by the Danish food Agency on a request from the Danish EPA in relation to case where a dinner mat containing a high amount of DEHP was found.

70 Such exposure will not be revealed from studies on duplicate diet samples, as it cannot be taken into account how the food is handled while eating.

October 2009 until January 2010. Data are reported as median, 95th percentile and maximum for average intake and high intake, respectively. In Table B32, data from high intake are presented, where the differences in intake for each individual is considered. Consequently, an exposure based on high intake will show relatively higher exposure values, comprising also the highest exposures.

The risk assessment is based on the exposure calculated in Sioen et al. (2012) and Fromme et al. (2013) as these studies include analyses of food conducted after the entry into force of the legislation of phthalates in food contact materials[71]. From Fromme et al. (2013) and Sioen et al. (2012) no data on the exposure of BBP for infants is available. The estimates from food comprise solid foods, liquid foods (e.g., soup and sauces) and beverages (e.g., milk, coffee, drinking water). The contribution from drinking water to exposure via food is expected to be negligible due to the low water solubility of the four phthalates[72]. Milk was the 3rd highest source of BBP in food for preschool children in Sioen et al. (2012). Alcoholic drinks were the 3rd highest source of BBP in food for adults in Sioen et al. (2012). No other beverages were among the main contributing sources to the exposure via food in Sioen et al. (2012).

Fromme et al. (2013) did not calculate the exposure for infants and BBP was only measured in two food samples by Sioen et al. (2012). To have an estimate of the exposure of BBP for infants, the exposure estimate of BBP from Fromme et al. (2007) is used. Sioen et al. (2012) calculated the exposure of BBP for adults to be approximately 30 % of the exposure calculated by Fromme et al. (2007). The exposure of BBP for infants used in the calculation is therefore 30% of the exposure calculated by Fromme et al. (2007) being 0.15 µg/kg bw/day and 0.24 µg/kg bw/day for median and 95 percentile, respectively.

The risk assessment in the background document from 2012 (ECHA 2012a) was based on a study from 2007 from UK (COT statement 2011). The argumentation for choosing this study and not the other studies are given below, together with a short description of the studies referenced in Table B32.

In the UK study total diet samples were taken and analysed for phthalates. This study is also from before the restriction of phthalates in food contact materials and the study only reports 97.5th percentiles. It was therefore also expected that the migration of phthalates would be lower than reported in this study.

In 2005, EFSA published opinions on DEHP, DBP and BBP for use in food contact materials. In their risk assessments EFSA considered data on phthalate intake from food from two Danish

[71] In the Public Consultation on the restriction proposal, the French Agency ANSES has submitted updated information based on French data for children and adults (ANSES 2014) and for infants (ANSES 2016). For infants the data were based on food specimen collected from July 2011 and July 2012. The results from these studies seem to generally confirm the levels in Table B33 for children and adults: a higher level for DEHP but lower levels for DBP and BBP were reported by ANSES (2014) in comparison to the data in Table B33 (no data for DIBP was reported by ANSES 2014. The exposures of DEHP, DBP and DIBP from food to infants are significantly lower in ANSES (2016) compared to the results in Table B33..

[72] In possible cases of contamination, the contribution of drinking water may be significant. The EU RAR for DEHP reported a case of contaminated groundwater in the Netherlands (20 to 45 µg /l of DEHP) from a publication in 1987. No recent information is available on groundwater or drinking water contaminated with such high levels of phthalates.

studies by Petersen et al. (2000) and Müller et al. (2003) and a British study by MAFF (1996). In Table B32 these studies are compared to other studies on phthalate intake from food published after the EFSA opinions.

Most of the data on exposure from phthalates in Table B32 are within the same range even though different methods are used to calculate the exposure, either measurements in diet or modelling data. It was in the background document from 2012 (ECHA 2012a) decided to use the exposure from the newest study from UK and use this study for the risk assessment. Other studies are presented to show that there are differences in exposure levels between the countries in EU. This could be due to cultural differences, for example the use of phthalates in the processing of the food, and how much food is eaten as finished and wrapped food. In the sections below the interpretation of the data is given for some individual studies.

In the study by Fromme et al., frozen samples of food for children between 15 and 21 months were analysed for the content of phthalates. The results are in the low end of the reported data in the table, but it is not stated when the food samples were taken. The mean values are given and not the median. These measurements are not used further in the risk assessment as it is not clear when the samples were taken.

The UK data (COT statement 2011) are from diet samples from 2007 from UK. The data are given as 97.5th percentile ranges from the whole diet. These measurements are in most cases lower than other measurements referred in the table. The measurements from this study are as the other referenced studies from before the new legislation on food contact materials. The lower measurements in this study could possibly be because of food contact materials with lower content of the four phthalates, but this cannot be confirmed from the study. The UK data from COT statement, 2011, has given the 97.5th percentiles as a range.

It should be noted, that Petersen and Breindahl presented mean values instead of medians as presented for other studies. In addition to mean values, Petersen and Breindahl (2000) presented intake estimates based on the highest measured phthalate concentrations in a total diet sample. These highest intake estimates are selected as 95th percentile values (“reasonable worst case”) for infants and adults (Petersen and Breindahl 2000). The study by Wormuth et al. (2006) is the only study reporting 95th percentile values for children around 6-11 year old (age 4-10), and these values are relatively close to median values in the Müller et al. (2003) study.

In 2003, the Danish Veterinary and Food Administration published a report on ”Human exposure to selected phthalates in Denmark” by Müller et al. (2003) who compared exposure to five phthalates (DEHP, DBP, BBP, DINP, DIDP) from foods, environment and consumer products (Müller et al., 2003). The study showed that food is the dominant pathway of exposure for all age groups, and that toys also contributed significantly to DEHP exposure in young children. The computer program EUSES was used to make a simple and a refined estimate of phthalate exposure, with the refined method including measured levels of phthalates in environment and food samples.

The predicted intake of phthalates in food based on the simple method was lower than measured concentrations of DEHP and DBP in foods, and as measurements of DEHP were available, these were used in a refined estimate of DEHP exposure. A specific estimate for food was made only for DEHP (Table B32). For the other phthalates, Table B32 includes refined

"total daily intake" estimates, which are thus interpreted as total daily intake via food. The refined estimate for DEHP including measured concentrations in foods and environment is reported to be similar to intake estimates based on foods and to the daily intake estimate reported in the EU RAR for DEHP. Also for DBP and BBP the refined estimates (based on environmental concentrations) are considered to be similar to exposure estimates from other studies (Müller et al., 2003).

Petersen and Breindahl (2000) and Fromme et al. (2007) calculated daily intake levels based on phthalate measurements in duplicate diet samples. These intake levels based on dietary levels have been divided by body weight of the relevant age group. To describe intake of 2-year olds, intake values of adults (per kg bw) have been multiplied by two, as 2-year olds have twice the energy intake of adults per kg bodyweight (Danish EPA, 2009). Petersen and Breindahl (2000) calculated mean daily intakes of adults as well as the highest daily intake value (highest daily intake of 29 samples). In Table B32 the mean value is listed as "median" and the highest intake value is listed as "95th percentile". In some cases, the use of mean values instead of median values may lead to an overestimation of typical exposures.

Wormuth et al. (2006) based their estimates of phthalate exposure from foods on concentration measurements published in 1995 to 2002. The authors concluded that the calculated exposure estimates were comparable to exposure estimates based on urinary levels. Food was considered the main source of DIBP, DBP and DEHP accounting for 55-60% of total exposure to toddlers (1-3 years, 13 kg bw) and 90-100% for adults. For BBP, only 18% of intake was from food in toddlers and around 65% was from food in adults. In the Wormuth paper, exposure from food is reported as a percentage of total exposure together with total exposure estimates in µg/kg bw/day. Thus, the listed estimates on exposure from food alone are calculated by multiplying total exposure with the food-fraction. An example for DIBP in children: median "daily internal exposure" is 0.3 µg/kg bw/day of which 65% is from food which is estimated to => 0.2 µg/kg bw/day from food.

Analysis of DEHP from 164 meals and snacks from 10 German boys in the age 5 to 8 years were collected in 2005. These meals were exact duplicates of the boys' intake of food. The concentrations of DEHP in the meals were between 10.00 and 1510.00 µg/kg with a median concentration of 35.20 µg/kg (personal communication UBA, 2011).

The EFSA opinions on phthalates for use in food contact materials do not single out one specific estimate for phthalate exposure via food but refer to a number of studies (The EFSA Journal, DBP, 2005; The EFSA Journal, DEHP, 2005; The EFSA Journal, BBP, 2005).

The EU Risk assessment reports reported specific estimates for exposure via food:

- The EU RAR for DEHP used an exposure estimate of 19 µg/kg bw/day for children and 2 µg/kg bw/day for adults based on an EUSES estimate (these estimates are for food, water and air with the main contribution from food, and these estimates are different from the EUSES estimates reported in Müller et al. (2003)). These levels are comparable to the values selected in Table B32.
- The EU RAR for DBP used an exposure estimate of 27 µg/kg bw/day for adults based on MAFF (1987). This estimate is higher than the values in Table B32 including the more recent data from MAFF (1996).

- The EU RAR for BBP used an exposure estimate of 0.8 µg/kg bw/day for children and 0.3 µg/kg bw/day for adults based on the data from MAFF (1996) listed in Table B32. These estimates are lower than the exposure estimates selected here from Müller et al. (2003) and Petersen and Breindahl (2000). The estimates from Müller et al. (2003) are selected as these are the values used by EFSA for the median daily intake, and the estimates from Petersen and Breindahl (2000) were selected as these report the 95th percentile.

Table B32 Intake estimates for phthalates from food. Gray-shaded values are used in the risk assessment in ECHA (2012a).

Intake from food in µg/kg bw/day		**Infants#**		**Children##**		**Women**	
Study	**Country**	**Median daily intake**	**95-p daily intake**	**Median daily intake**	**95-p daily intake**	**Median daily intake**	**95-p daily intake**
DEHP							
Sioen et al. (2012), based on measurements in food samples###, food samples from 2009-2010	BEL			3.50	5.38	1.49	2.86
Fromme et al. (2013), based on measurements of total diet samples, diet samples from 2009-2010	GER	4.66	7.09				
COT statement (2011), measurement on food samples	GBR		9.9 (97.5-p)		6.7 (97.5-p)		4 (97.5-p)
Fromme et al. (2007); based on measurements in total diet samples	GER	4.8	8			2.4	4.0
MAFF (1996) in The EFSA Journal, DEHP, based on measurements in diet samples	GBR	5	10			2.5	5
Müller et al. (2003); based on EUSES	EUR	26		11		4.5	
Petersen and Breindahl (2000); based on measurements in total diet samples	DEN	10 (mean)	36.7			5 (mean)	18.33
Tsumura et al. (2003), based on measurements in diet samples	JPN	5				2.5	
Fromme et al. based on measurements in food samples	GER	2.6 (mean)	4.1				
Personal communication UBA (2011); based on measurements in diet samples	GER			1.27	3.17		
DBP							
Sioen et al. (2012), based on measurements in food samples###	BEL			0.202	0.301	0.081	0.155
Fromme et al. (2013), based on measurements of total diet samples	GER	0.70	1.24				
COT statement (2011), measurement on food samples	GBR		1.0 (97.5-p)		0.7 (97.5-p)		0.3 (97.5-p)

Intake from food in µg/kg bw/day		**Infants#**		**Children##**		**Women**	
Study	**Country**	**Median daily intake**	**95-p daily intake**	**Median daily intake**	**95-p daily intake**	**Median daily intake**	**95-p daily intake**
Fromme et al. (2007); based on measurements in total diet samples	GER	0.5	3.2			0.3	1.4
MAFF (1996) in the EFSA Journal, DBP (2005), based on measurements in diet samples	GBR	0.4	1.0			0.2	0.5
Müller et al. (2003); based on EUSES	EUR	8.0		3.5		1.6	
Petersen and Breindahl (2000); based on measurements in total diet samples	DEN	9.7 (mean)	24			4.8 (mean)	12
Tsumura et al. (2003), based on measurements in diet samples	JPN	0.4				0.2	
Fromme et al. based on measurements in food samples	GER	0.4 (mean)	0.7				
Wormuth et al. (2006); based on measurements in food	EU, US, Asia	2.2	22	0.7	10	3.1	
DIBP							
Sioen et al. (2012), based on measurements in food samples###	BEL			0.418	0.644	0.143	0.280
Fromme et al. (2013), based on measurements of total diet samples	GER	1.03	9.02				
COT statement (2011), measurement on food samples	GBR		2.7 (97.5-p)		1.8 (97.5-p)		0.9 (97.5-p)
Fromme et al. (2007); based on measurements in total diet samples	GER	1.1	4.2			0.6	2.1
Fromme et al., based on measurements in food samples	GER	0.5 (mean)	1.0				
Wormuth et al. (2006); based on measurements in food	EU, US, Asia	0.48	2.4	0.2	1.0	0.5	1.5
BBP							

Intake from food in µg/kg bw/day		**Infants#**		**Children##**		**Women**	
Study	**Country**	**Median daily intake**	**95-p daily intake**	**Median daily intake**	**95-p daily intake**	**Median daily intake**	**95-p daily intake**
Sioen et al. (2012), based on measurements in food samples###	BEL			0.116	0.205	0.054	0.117
Fromme et al. (2013), based on measurements of total diet samples	GER	0.15	0.24				
COT statement (2011), measurement on food samples	GBR		1.3 (97.5-p)		0.9 (97.5-p)		0.5 (97.5-p)
Fromme et al. (2007); based on measurements in total diet samples	GER	0.5	0.8			0.2	0.4
MAFF (1996) in the EFSA Journal, BBP (2005), based on measurements in diet samples	GBR	0.3	0.6			0.1	0.3
Müller et al. (2003); based on EUSES	EUR	5.9		2.4		1.0	
Petersen and Breindahl (2000); based on measurements in total diet samples	DEN	1.0 (mean)	10.7			0.5 (mean)	5.33
Tsumura et al. (2003), based on measurements in diet samples	JPN	0.12				0.06	
Wormuth et al. (2006); based on measurements in food	EU, US, Asia	0.07	1.1	0.03	0.8	0.2	

The green-shaded values are new estimates based on measurements in food from 2009 and 2011 and used in the risk assessment.
#Study Wormuth et al. (2006) and Müller et al. (2003) include specific data for toddlers (age 1-3 and 1-6, respectively), but in all other studies phthalate intake of toddlers is calculated as twice the estimate for adults, as small children have twice the energy intake of adults per kg bodyweight (Danish EPA, 2009).
##Phthalate intake of children around 6-11 years of age is only listed for study (Wormuth et al., 2006), which includes specific data for 4-10 year olds, and for study (Müller et al., 2003), which provides estimates for 7-14 year-olds, which used as approximate estimates for 6-11 years olds.
###Study from Sion et al., 2012 include data on preschool children in the age of 2.5 to 6.5 years, these intake data are used for 6-11 years old
####BBP was only detected in two samples and the median and 95th percentile was not calculated. The values are based on 30 % of the exposure from Fromme et al. (2007).

B.8.4.5.5. Uncertainties on the measurements of exposure from food

In the opinion from RAC on the four phthalates it was concluded that the exposure from food could be overestimated due to the lack of recent data from after the entry into force of the legislation on phthalates in food contact materials in 2008. However, limited new data on dietary exposure to phthalates from Belgium and from infants in Germany has been reported. It is unclear whether the exposure to the four phthalates is now lower after the entry into force of the FCM legislation limits of phthalates. The new data can only be seen as an indication of the exposure from food as there could be differences among member states in EU. There are differences in the method for calculating the exposure from food in the two additional studies. The Belgium study bases the calculation on analysis of 388 food samples and a food consumption database and calculates the exposure based on the concentration of phthalates in the different food samples and the consumption of food. In the German study the exposure is based on analysis of duplicate diet samples and will therefore show the actual intake. Differences in the methods can also result in differences in the calculated exposure.

As discussed in section B.8.3.2, several studies measured urinary levels of phthalates. The diet in these studies was either changed by letting volunteers fast or the content of phthalates in the diet was measured. From studies it can be concluded that 75% of the intake of DEHP is attributable to food (incl. drinks), whereas for DBP, DIBP and BBP it is assumed that 25% is attributable to food.

The exposure modelling suggests that the contribution to exposure to DEHP from food is only 38%, 51% and 36% in infants, children and adults respectively. For DBP, the modelling suggests a contribution of 32%, 19% and 10% in infants, children and adults respectively; for DIBP 44%, 35% and 18% in infants, children and adults respectively; and for BBP 0% (no recent data available), 34% and 22% in infants, children and adults respectively.

For DEHP, the modelling seems to underestimate the contribution via food relative to other exposure sources. For the other phthalates, considering all uncertainties, the modelling estimates of the proportion of food to overall exposure sources is reasonably similar to the estimates above that were based on ‘fasting-urinary biomonitoring’ or ‘duplicated diet-urinary biomonitoring’ studies.

B.8.4.5.6. Conclusion – exposure from food

As only limited new data on dietary exposure to phthalates can be found in the literature, it is unclear whether the exposure to the four phthalates is lower after the entry into force of the limits of phthalates in food contact materials. As can be seen from the table, values appear lower particularly for adults, but the study from Fromme et al. (2013) does for some phthalates show similar or even higher exposure of phthalates compared to earlier studies, but this might be because to the use of high intake data from the study. The data from Fromme et al. (2013) also shows higher exposure than the study from Sioen et al. (2012). The explanation could be caused by regional differences between countries in EU, and the exposure based on data from Belgium and Germany is therefore only an indication of the exposure on EU-level. The exposure of phthalates from food is also expected to be higher for children due to their higher intake of food in relation to their size.

The newest data on exposure from phthalates from food are used for the risk assessment. These are data from after the entry into force of the limits in food contact materials in 2008. Intake estimates for food used in the risk assessment are shown in the Table B33.

Table B33. Intake estimates for food used in the risk assessment (µg/kg bw/day)

	Infants**		Children*		Women*	
	Median daily intake	95-p daily intake	Median daily intake	95-p daily intake	Median daily intake	95-p daily intake
DEHP	4.66	7.09	3.50	5.38	1.49	2.86
DBP	0.70	1.24	0.20	0.30	0.08	0.16
DIBP	1.03	9.02	0.42	0.64	0.14	0.28
BBP	0.15	0.24	0.12	0.21	0.05	0.12

*Sioen et al. (2012)
**Fromme et al. (2013), except BBP where 30% of the estimate in Fromme et al. (2007) is used

Sioen et al. (2012) include data on preschool children in the age of 2.5-6.5 years and these are in the risk assessment used for children (6-11 years old).
Fromme et al. (2013) include data on children in the age of 15-21 months and these are in the risk assessment used for infants (6-12 months).

B.8.4.6. Exposure from contact with articles

The four phthalates are used in a wide range of different articles, primarily as plasticiser in PVC. When such articles are in direct contact with the body, people might be exposed to the phthalates contained in the articles. However, to be exposed the phthalates have to migrate from the article to a media to which the consumers are exposed, e.g. sweat, saliva, air or dust particles. Furthermore, the phthalates also have to be absorbed into the body (Figure B5).

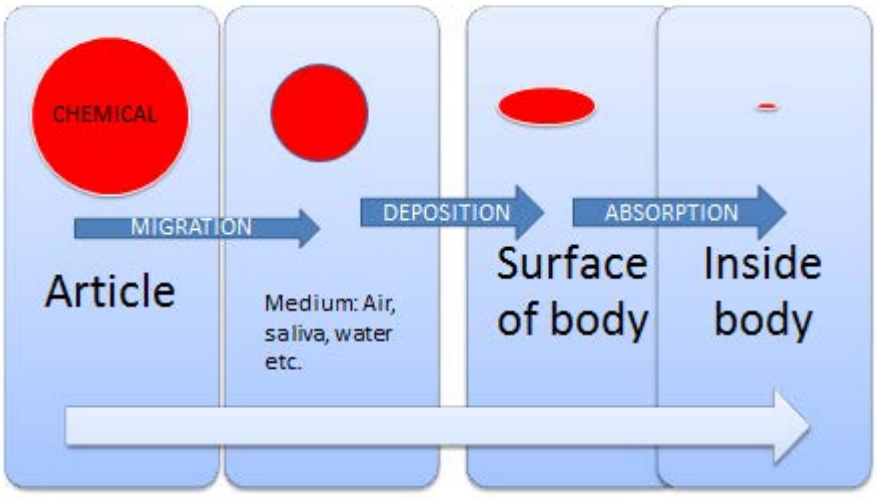


Figure B5 Exposure of chemicals from articles.

B.8.4.6.1. Concentration of phthalates in articles

Analyses of articles containing soft PVC have shown that such articles in many cases contain phthalates. Table B34 below shows articles where the four phthalates have been found.

Table B34 Examples of articles containing one or more of the four phthalates.

Articles type	**Phthalate in mg/kg**				**Year of publication**
	DEHP	**DBP**	**DIBP**	**BBP**	
Bicycle handles	8000	50	20-50		2015
Covers for cell phones and tablets	50000-80000		70		2012, 2015
Children's wrist watches	5000-150000	10-200	50-70000		2015
Gloves for cleaning	260000				2012
School bags	7100-13000	4000	830-3100		2015
PVC mat for a sink	270000	5150			2015
Garden hoses	180-39000				2015
PVC tape	180000				2015
Rubber boots	165000				2015
Rain coats	82-110000				2015
[redacted]					
Plastic sandals	11-461000	1-345000	3-212000	ND-79	2009, 2010
Bag (backpack, toilet bags, handbag)	12-202000	14-60	10-509		2010, 2012
Shower curtain	15-296000	13			2010
Oilcloth and dinner mats	31-254000		9-56		2010, 2015
Tools	542-150500	51-87	44-87000		2013
Synthetic leather furniture	24-109000	125	12-5200		2013
Water wing	333000				2010
Swimming pool	66-258000	10	18		2010
Balance ball	462-439000	21	115-693		2010
Training ball	9		355000		2010
T-shirts	24-220000	1-310		2-73000	2008
Reflectors on children's clothes	124000-147000				2009
Zipper strap	74	43			2009
Soap packaging	133-80130				2009
Shower mats	128625				2009
Floor coverings	49-325	129	56-73650	113	2010
Wall paper	10-24	9-30	5-626		2010
Furniture	24-391500	2-340	8-16250	124-652	2010
Lamp shades	13-365	14-719	9-337		2010
Shower curtains	173-281500	63	64-173		2010
Air mattresses	31-304000		11		2010
Sex toy	730-702000				2006
Eraser	170000-440000				2007

ND: Not detected

Other articles of soft PVC could also contain the four phthalates. This could be the steering wheel in a car, handles on tools and garden tools as well as articles made of artificial leather, as handbags, bags/covers for smartphones and tablets, office materials and furniture.

B.8.4.6.2. Migration rates

"Phthalates are not covalently bound to the PVC matrix. The plasticiser molecules are intercalated between the polymer chains, where electrostatic plasticiser-plasticiser, plasticiser polymer, and polymer-polymer interactions occur between the dipoles (Van der Waals forces). Plasticisers can be released by volatisation, extraction to a liquid, or by migration to a solid or semi-solid" (ECHA, 2013a).

Migration of phthalates depends on type of contact, contact duration, temperature, plasticiser concentration difference, plasticiser concentration level, molecular weight and molecular structure (ECPI 2011b). Another element that seems important in determining the migration rate is the process conditions for PVC manufacturing (Simoneau 200955; RIVM 1998; ExxonMobil 2011b) as well as the analytical methods used for determination of the migration rates (Danish EPA, (2016)). It is important to consider that the actual driver for migration is determined by thermodynamics, i.e. a reduction of free energy (INEOS ChlorVinyls 2012). Phthalates are highly lipophilic, and therefore fatty simulants, such as olive oil, can produce significant migration in contrast with non-lipophilic media (INEOS ChlorVinyls 2012). For articles requiring a long service-life (e.g. flooring, cable), loss of plasticiser results in loss of mechanical performance and leads to product shrinkage and brittleness (ECPI 2011b).

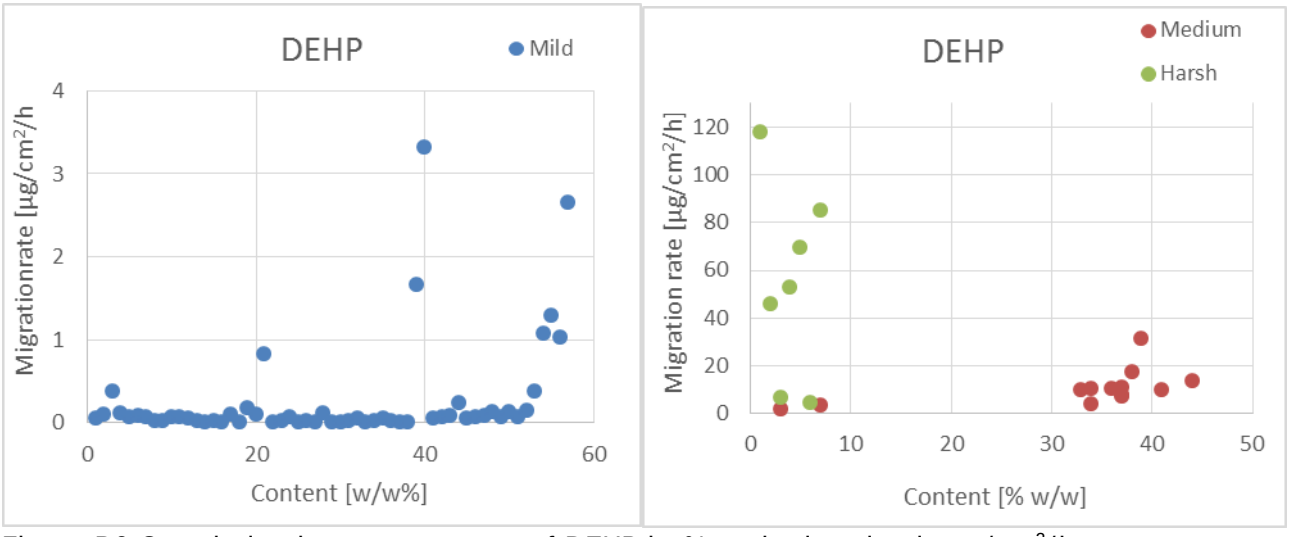


Figure B6 Correlation between content of DEHP in % and migration in µg/cm²/h.
Note: Mild: Static or dynamic conditions at 37±3°C, Medium: Head over heels method at 60 rpm, Harsh: Horizontal shaking at 300 rpm (Danish EPA, 2016).

A relationship between the plasticiser content of PVC and the migration of plasticiser from PVC cannot be established based on experimental data (as also noted by Babich et al. 2004; Simoneau et al. 2009; Health Canada 1998). The likely reason for this is the multitude of factors influencing the migration from PVC in combination with differences in experimental settings amongst the studies and the measurement methods used. Niino et al. (2002a)

reported high effects of especially rotation speed during extraction (a migration rate of ca. 20 µg/cm^2/h at 200 rpm versus approx. 150 µg/cm2/h at 400 rpm) and temperature (a migration rate of app. 80 µg/cm^2/h at 20 °C versus ca.170 µg/cm^2/h at 40 °C).

Figure B6 shows the migration rate as function of the content of DEHP for different analytical methods for determining the migration rate. It shows that the methods for determining the migration rate seems to have a high impact on the migration rate.

Most available migration rates are for DEHP and only few for DBP and DIBP and no migration rates are found for BBP. It is assumed that migration to artificial sweat and saliva are comparable. Migration rates are primarily based on reports and non-published reports from the Danish EPA. Table B35, Table B36, and Table B37 below show positive findings of migration from DEHP, DBP and DIBP, respectively, where the result are given per area. Health Canada (2015c) reports that an evaluation of migration rate data show that a majority of phthalates migrate during the first 1 to 3 hours. The migration rates in Table B35, Table B36, and Table B37 are therefore derived, assuming that the migration will happen in the first hour. Dividing the migration rate with the experimental duration would lead to an underestimation of the migration rates if most of the phthalate will migrate within the first hour. In the Danish EPA (2016) migration rates for DEHP, DBP, DIBP, BBP and DINP are suggested based on determination by head over heels method at 60 rpm. These migration rates can be used for low tier risk assessment of specific articles to indicate whether articles pose a risk to the consumer. Overall migration rates can be determined in different ways as there are many factors influencing the migration.

It is decided to take an average of all migration rates referenced in the literature and shown in the tables below.

Table B35 Migration rates for DEHP

Article type	Content of DEHP w/w%	Migration of DEHP mg/g/h	Migration µg/cm²/h	Method for determination of migration*	Reference
Oilcloth	13	0.004	0.07	Migration to artificial saliva and sweat under static condition for 1 hour at 37±3°C	Danish EPA (2010b), Survey no 109
Oilcloth	13	0.005	0.09		
Toilet bag	17.6	0.004	0.08		
Toilet bag	17.6	0.003	0,06		
Shower curtain	25.1	0.005	0.04		
Shower curtain	25.1	0.007	0.06		
Oilcloth	25.3	0.005	0.05		
Oilcloth	25.3	0.006	0.05		
Swimming pool	25.8	0.003	0.08		
Swimming pool	25.8	0.004	0.11		
Shower curtain	29.6	0.004	0.06		
Shower curtain	29.6	0.005	0.08		
Balance ball	44.2	0.003	0.24		
Balance ball	44.2	0.006	0.38		
Rucksack	20.2	0.01	0.21	Migration to artificial sweat under static condition for 1 hour at 37±3°C	Danish EPA (2010b), Survey no 109
Water wings	33.3	0.004	0.08		
Mobile cover of plastic	8		0.04	Migration to artificial sweat under stirring for 4 hour at 37±3°C	Danish EPA (2015), Survey no 139
wrist watch	15		0.24	Migration to artificial sweat under stirring for 24 hour at 37±3°C	
Sex toys	0.07		0.01	Migration to artificial sweat adjusted to pH 4.5 for one hour at 40°C	Danish EPA (2006a), Survey no 77
Sex toys	17.6		0.06		
Sex toys	20		0.05		
Sex toys	70.2		0.06	Migration to artificial sweat under stirring	Danish EPA not published
Doll - head	21		0.09		

Dinner mat	10	0.001	0.11	for 3 hour at 37°C	
PVC mat for sink	27	0.001	0.23		not published
Garden hose	0.31	0	0.02		
PVC tape	18	0.001	0.04		
Rubber boot	16.5	0.001	0.19		
Rain coat	11	0.002	0.10		
Doll	3		1.86	Migration to artificial saliva using Head over Heels method	Bouma and Schakel 2002
Doll	38		17.64		
Doll	39		31.32		
Doll	44		13.56		
Inflatable furniture	37		7.38		
Inflatable furniture	41		9.84		
Swimming tool	33		9.84		
Swimming tool	36		10.50		
Swimming tool	37		7.86		
Swimming tool	37		10.86		
Apron	7		3.48		
Ball	34		10.62		
Can	34		3.78		
Plate B	47.7		29.50	Migration to saliva simulant with horizontal shaking for 15 min at 35°C	Niino et al., 2003
Plate D	14.7		11.40		
Plate F	13.2		1.60		
Soft doll C	31.1		13.20		
Ball A	18.5		17.40		
Ball B	37.0		21.30		
Plastic sandal	2.2	0.0007	0.17	Migration to sweat simulant, static condition for 16 hours at 37±3°C	Danish EPA (2010c), Survey no 107
Plastic sandal	30	0.0008	0.18		
Plastic sandal	15	0.0003	0.07		
Plastic sandal	15	0.0064	1.57	Migration to sweat simulant, static condition for 16 hours at 37±3°C, with	

				new simulant after 8 hours	
Plastic sandal	15	0.0083	2.85	Migration to sweat simulant, dynamic condition for 16 hours at 37±3°C	
Plastic sandal	15	0.0046	1.52	Migration to sweat simulant with sunscreen on the sample and static condition for 16 hours at 37±3°C	
Plastic sandal	15	0.049	13.14		
Plastic sandal	17	0.0004	0.08	Migration to sweat simulant, static condition for 16 hours at 37±3°C	
Plastic sandal	46	0.0013	0.31		
Plastic sandal	0.03	0.0042	0.96		
Plastic sandal	24	0,0003	0.10		
Plastic sandal	11	0.0083	0.15		
Plastic sandal	21	0.00029	0.09		
Plastic sandal	1	0.0072	1.73		
Plastic sandal	34	0.00042	0.09		
Plastic sandal	<LOD	0.0003	0.04		
Plastic sandal	21	0.0023	0.35		
Plastic sandal	26	0.0011	0.08		
Plastic sandal	33	0.00043	0.08		
Plastic sandal	46	0.00102	0.20		
Plastic sandal	10	0.0207	0.66		
Plastic sandal	15	0.0018	0.22		
Plastic sandal	0.013	0.0049	0.10		

*Further details on the methods can be found in the references.

Table B36 Migration rate for DBP

Article type	**Content of DBP w/w%**	**Migration of DBP mg/g/h**	**Migration µg/cm²/h**	**Method for determination of migration***	**Reference**
PVC mat for sink	0,515	0.00022	0.04	Migration to artificial sweat under stirring for 3 hour at 37°C	Danish EPA Not published
Plate C	47,1		36.20	Migration to saliva simulant with horizontal shaking for 15 min at 35°C	Niino et al., 2003
Plate D	13,5		8.70		
Plate G	12,9		4.30		
Ball A	10		14.50		
Ball B	22		19.80		
Plastic sandal	23	0.028	6.77	Migration to sweat simulant, static con-dition for 16 hours at 37±3°C	Danish EPA, 2010c, Survey no 107
Plastic sandal	27	0.053	10.56		
Plastic sandal	1	0.0006	0.15	Migration to sweat simulant, static con-dition for 16 hours at 37±3°C, but with new simulant after 8 hours	
Plastic sandal	1	0.001	0.34	Migration to sweat simulant, dynamic condition for 16 hours at 37±3°C	
Plastic sandal	1	0.0007	0.21		
Plastic sandal	1	0.98	0.26		
Plastic sandal	8	0.017	3.56		
Plastic sandal	34	0.044	10.08		
Plastic sandal	0.9	0,0008	0.27		
Plastic sandal	0.2	0.004	0.07		
Plastic sandal	0.08	0.00005	0.02		
Plastic sandal	18	0.029	6.98		
Plastic sandal	1.2	0.003	0.47		
Plastic sandal	0.3	0,0008	0.03		
Plastic sandal	0.3	0,0003	0.03		
Plastic sandal	28	0.051	10.08		

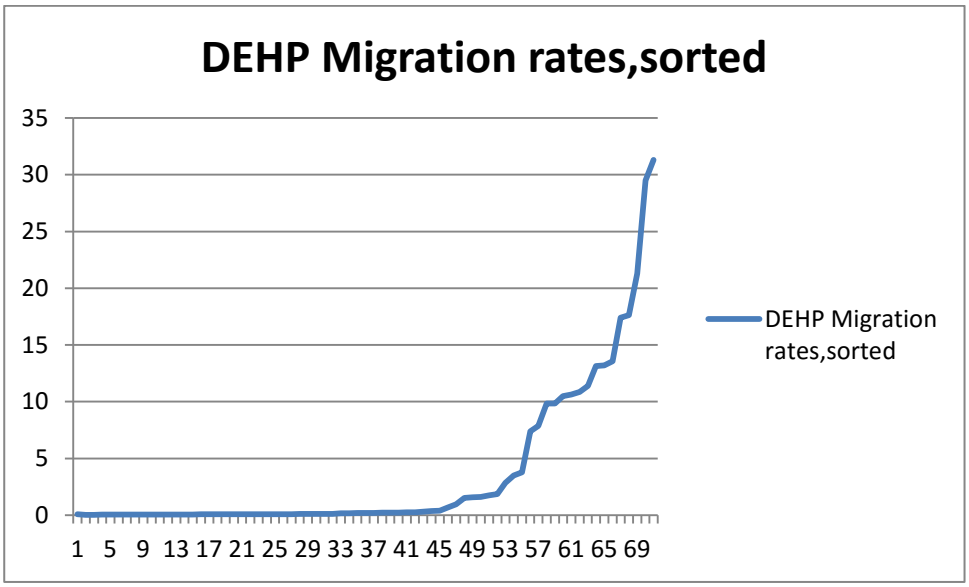
*Further details on the methods can be found in the references.

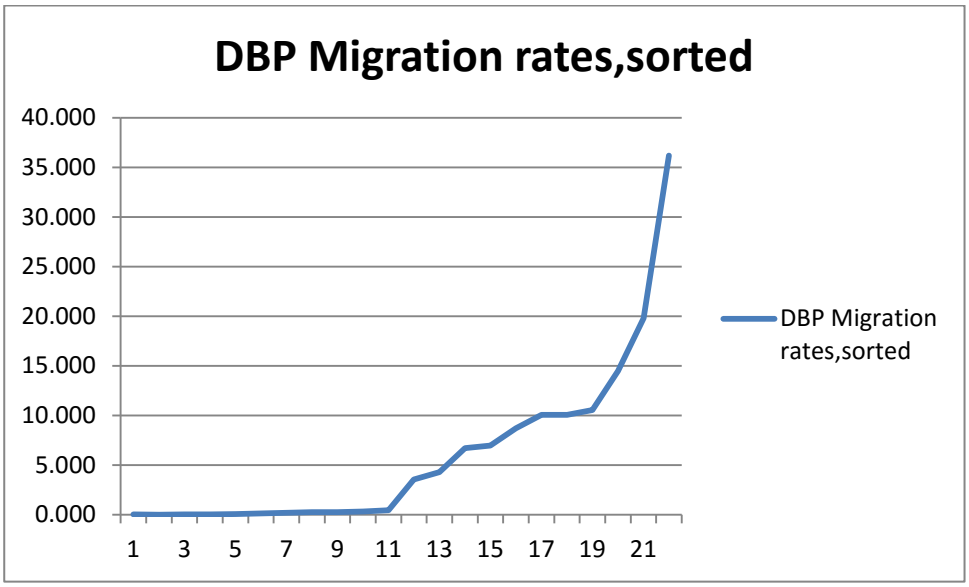
Table B37 Migration rates for DIBP

Article type	Content of DIBP in w/w%	Migration of DIBP in mg/g/hour	Migration in µg/cm²/h	Method for determination of migration*	Reference
Wrist watch	7		0.89	Migration to artificial sweat under stirring for 24 hour at 37±3°C	Danish EPA, 2015 Survey no 139
Training balls	35.5	0.111	3.70	Migration to artificial saliva under static condition for 1 hour at 37±3°C	Danish EPA (2010b) Survey no 109,
	36.5	0.182	5.80		
Plastic sandals	6.3	0.013	3.10	Migration to sweat simulant, static condition for 16 hours at 37±3°C	Danish EPA (2010b) Survey no 109
	21	0.02	4.66		
	21	0.021	5.15	Migration to sweat simulant, static condition for 16 hours at 37±3°C, but with new simulant after 8 hours	Danish EPA (2010c), survey no 107
	21	0.022	7.56	Migration to sweat simulant, dynamic condition for 16 hours at 37±3°C	
	21	0.047	15.58	Migration to sweat simulant with sunscreen on the sample and static condition for 16 hours at 37±3°C	
	21	0.067	17.97		
	7.4	0.023	4.82	Migration to sweat simulant, static condition for 16 hours at 37±3°C	
	6.6	0.0076	2.55		
	2.2	0.12	2.21		
	11.7	0.013	3.85		
	5.3	0.015	3.37		
	1.6	0.017	2.03		
	12.1	0.049	7.55		
	12.1	0.037	6.66		
	2.2	0.038	1.19		
	3.9	0.013	1.56		
	33	0.39	7.86		

*Further details on the methods can be found in the references.

As can be seen from Table B35, Table B36, Table B37 and also from Figure B7 below, there are large differences in the measured migration rates. The migration rates analysed by the Danish EPA seem in general to be lower than migration rates measured by others. These migration rates are determined under static or dynamic (magnetic steering) conditions.


DEHP Migration rates,sorted
35
30
25
20
15
10
5
0
1 5 9 13 17 21 25 29 33 37 41 45 49 53 57 61 65 69
DEHP Migration rates,sorted


DBP Migration rates,sorted
40.000
35.000
30.000
25.000
20.000
15.000
10.000
5.000
0.000
1 3 5 7 9 11 13 15 17 19 21
DBP Migration rates,sorted

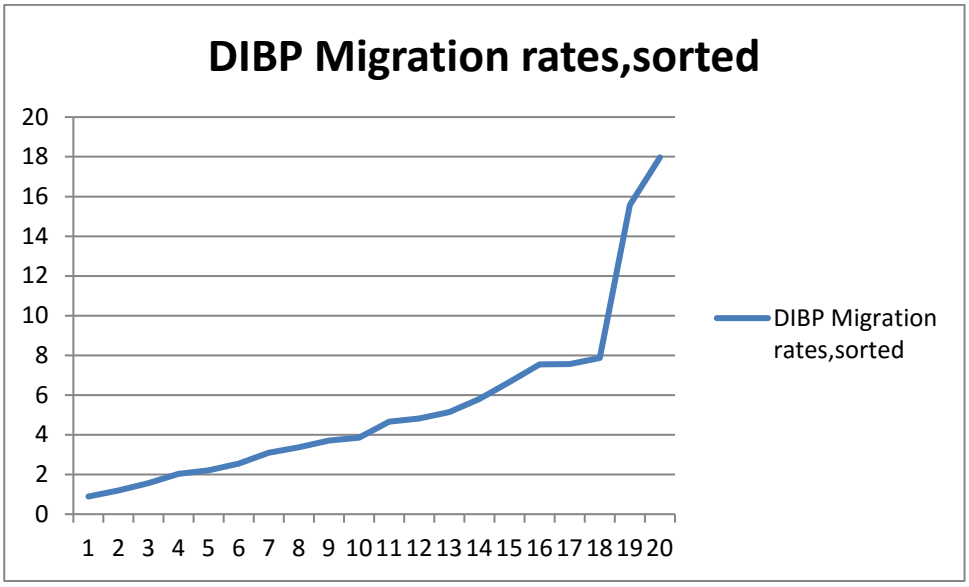


Figure B7 Sorted migration rates µg/cm2/h for test results listed in table B33-B35

Average migration rates for DEHP, DBP and DIBP (from the tables above) are used in the scenarios to estimate the exposure from contact with articles. The average migration rates are based on positive findings of migration. Furthermore, only migration rates given in the unit per area are used. The mean migration rate for BBP reported in Wormuth (2006) (see Table B39) is used in the scenarios to estimate the exposure from BBP.

Table B38 Average of migration rates for DEHP, DBP, DIBP and BBP used in the deterministic exposure modelling

Phthalate	Min migration rate in µg/cm²/h	Average migration rate in µg/cm²/h	Max migration rate in µg/cm²/h
DEHP	0.02 (0.1)	3.7[73]	31.3[73] (15.0)
DBP	0.02 (0.2)	6.1	36.2 (17.2)
DIBP	0.9 (0.2)	5.4	17.9 (17.2)
BBP	0.3 (0.3)	2.5	6.5 (6.5)

Note: Also data on migration from articles with content below 1% is taken into account. If these are excluded, the average migration rates would be DEHP: 3.92, DBP: 8.29 but no change for DIBP. Figures in brackets are estimates of low and high migration rates to be used in Monte Carlo analysis in B.8.3 For DEHP, DBP and DIBP the high exposure levels are calculated as 95th percentile (assuming normal distribution) of the migration rates in Table B35, Table B36 and Table B37, while the low estimate is the 5th percentile (estimates for DPB and DIBP are assessed together). For BBP the figures from Wormuth et al (2006) are used.

[73] In the Annex XV report the average migration rate for DEHP was calculated to 3.8 and the 95th percentile calculated to 15.2, which are the figure used later also in this report in the risk characterisation.

For comparison the mean migration rates for DEHP and DBP reported by Wormuth (2006) (see Table B39) are higher for DEHP and lower for DBP, but in the same level as the average migration rates used to estimate the exposure here.

Table B39 Migration rates from Wormuth (2006)

Phthalate	Min migration in µg/cm²/h	Mean migration in µg/cm²/h	Max migration in µg/cm²/h
DEHP	1.3	4.4	8.4
DBP	0.1	1.2	3.2
BBP	0.3	2.5	6.5

B.8.4.6.3. Distribution between different phthalates

To be able to calculate the total exposure from the four phthalates, assumption on the proportion of articles containing DEHP, DBP, DIBP and BBP has to be made. From the information on the use of the four phthalates it is assumed that the proportion of articles containing DEHP, DBP, DIBP and BBP is 74%, 8%, 8% and 10%, respectively. Compared to the information in Annex C baseline, the contribution from DBP, DIBP and BBP might be lower and the contribution from DEHP somewhat higher.

B.8.4.6.4. Exposure modelling

A general exposure model is established. This covers exposure estimates for exposure to male infants (6-12 months old), both oral and dermal exposure and for male children (6-11 years old), and for women (only dermal).

For each age group a typical value is derived, as well as two reasonable worst case estimates. One is based on the traditional approach of combining worst case assumptions for the two parameters – contact time and contact area[74], while the other is based on Monte Carlo simulations.

In the Monte Carlo simulations the two (three) parameters mentioned above as well as the migration rates for the four phthalates have been combined in a probabilistic approach, as described in section B.8.4.1. By this approach the parameters where the worst case assumptions diverge from the typical case assumptions are combined to address a situation where the parameters are not correlated with each other.

Exposure of infants from mouthing of articles

The oral exposure is estimated based on the above assumptions on migration and weight for infants. Oral exposure of phthalates from articles can occur by mouthing and/or ingestion. The oral exposure is calculated based on a modified equation from the ECHA guidance Chapter R15 (ECHA 2010).

[74] For oral exposure also weight as mouthing behaviour is considered not to be correlated with weight of the infant

$$D_{oral} = \frac{Fc_{migr} \cdot T_{contact} \cdot Q_{prod} \cdot F_{Abs}}{BW} \cdot N$$

where

D_{oral}	Oral exposure daily dose (µg/kg BW/day)
Q_{prod}	Surface area of article (cm^2)
Fc_{migr}	Fraction of compound that migrates per unit time µg/cm^2 per hour
F_{abs}	Fraction of the applied compound that is absorbed through the skin (decimal fraction between 0 and 1)
$T_{contact}$	Duration of exposure per event (hours)
N	Number of events per day
BW	Body weight (kg)

For children, the oral exposure to DEHP, DBP, DIBP and BBP is limited since DEHP, DBP and BBP are banned in toys and childcare articles and DIBP is restricted in toys. However, children might mouth other articles, as mouthing behaviour is used by small children to explore their environment.

One or more of the four phthalates have been identified in some articles that infants could mouth and which infants could be in contacts with. For example: covers for mobile phones and tablets, faux leather hand-bags, furniture with faux leather (e.g., sofa's), oil cloth and dinner mats, shower curtains, balance balls; training balls as well as reflectors on jackets and straps on zippers on jackets.

The draft report on mouthing behaviour from CEN/TC 52 (2015) refers to previous literature showing that children mouth many items other than dummies, teethers and toys. Particularly children under 1 year do mouth other articles due to teething and use mouthing as a method of exploring their environment. The CEN/TC 52 (2015) study also show that especially for children under 1 year, toys not intended to be mouthed are mouthed as much as toys intended to be mouthed. This indicates that especially children under 1 year will mouth other articles than toys, which could contain one or more of the four phthalates.

Mouthing time and contact area

Available reports on mouthing behaviour of children have been explored by ECHA in the report on the existing restrictions of DINP and DIDP under REACH Annex XVII (ECHA 2013). ECHA assumed, based on existing studies on mouthing behaviour, that the mean mouthing time for children in the age of 0-18 months can be estimated to 30 min/day. This covers all articles made of various materials. Smith and Norris (2002) estimated that half of all mouthed articles are made of plastics, whereby mouthing of plastic articles is 15 min/day.

It is expected that children will primarily mouth toys and childcare articles and therefore it is assumed that only 25 % of the mouthing time (3.75 min) is used to mouth articles not being toys and childcare articles.

It is then assumed that 25 % of these plastic articles contain one or more of the four phthalates. The total mouthing time is therefore assumed to be approximately **1 min/day for**

infants in a **typical scenario**. For illustration, this could cover mouthing of bags made of PVC/artificial leather, a training ball used for playing and the shower curtain while bathing. The typical scenario should be seen as an average scenario, covering an average infant.

As a **reasonable worst case scenario a mouthing time of 30 min/day** is used. The mouthing time is based on an assumption that infants in the reasonable worst case scenario mouth plastic articles (including toys and childcare articles) 2 hours every day (ECHA 2013). If it is assumed that 25 % of the mouthing time is used to mouth articles not being toys and childcare articles, and that of these articles all contain one of the four phthalates, the daily mouthing time to the four phthalates would be 30 minutes. A child might on one day for example mouth several articles containing one of the four phthalates such as a zipper strap of a jacket, an oil cloth, a cell phone cover, and synthetic leather furniture totalling 30 min per day. This duration may also reflect a situation where a child's favourite mouthing object (not being a toy) is plasticised with one or more of the four phthalates.

Exposure from non-compliant toys may not be covered by the mouthing times assumed here but naturally contributes to the body burden. There are regularly RAPEX notification on non-compliant toys containing phthalates.

The **surface area of the mouthed articles is assumed to be 10 cm²**, as this is normally used in exposure scenarios for toys; it is a reasonable assumption that the surface area of other mouthed articles will be the same as for toys. Children (6-11 years old) and women are not expected to mouth articles containing one or more of the four phthalates as mouthing behaviour is a behaviour that primarily is used by children under 3 years. The only group of articles expected to be mouthed by children is erasers.

Table B40 Assumptions for mouthing time and oral contact area for infants

	Mouthing time in min/d			**Oral contact area in cm²**
	Minimum Scenario*	**Typical scenario**	**Reasonable worst case scenario**	
Infants	0.12	1	30	10

*For use in Monte Carlo Simulations

Exposure estimates of typical and reasonable worst case scenarios are given in Table B41.

Table B41 Exposure of infants from mouthing of articles (µg/kg bw/day)

	DEHP	DBP	DIBP	BBP
Typical*	0.05	0.01	0.01	0.00
Reasonable worst case*	1.53	0.27	0.23	0.14
Reasonable worst case**	2.76	0.37	0.36	0.18

*Monte Carlo simulations related to variation of migration rate and mouthing time (Table B38)

Direct dermal exposure

The general public, consumers, employees (e.g. in offices and warehouses), and children may be dermally exposed to many kinds of PVC articles which might contain one or more of the four phthalates. It could be vinyl flooring, gloves for dishwashing or cleaning, bags and wallets, shower curtains, sandals, oilcloth and dinner mats and other articles. The dermal exposure is calculated based on a modified equation from the ECHA guidance on safety assessment, chapter R15, from 2010 (ECHA, 2010).

$$D_{der} = \frac{Fc_{migr} \cdot T_{contact} \cdot Q_{prod} \cdot F_{abs}}{BW} \cdot N$$

Where

D_{der}	Dermal exposure daily dose	µg/kg BW/day
Q_{prod}	Surface area of skin coming into contact with the articles	cm^2
Fc_{migr}	Fraction of compound that migrates out of the article per unit time µg/cm^2 per hour	
F_{abs}	Fraction of the applied compound that is absorbed through the skin (decimal fraction between 0 and 1)	
$T_{contact}$	Duration of exposure per event	hours
N	Number of exposures (events)	per day
BW	Body weight (BW)	kg

Contact time and contact area – infants and children

Infants and children can and will most probably have dermal contact with a wide variety of different articles containing phthalates during a normal day, e.g. the articles mentioned in Table B34. It could for example either be shower mats, bicycle handles and covers/bags for smartphones or tablet as well as furniture made of artificial leather. Other examples are use of swimming pools, inflatable mattresses, sandals during a summer period, or a changing mat plasticised with DIBP. 6—11 years old might also use gloves for dishwashing or cleaning when assisting in the house-keeping at home as well as air mattresses and other inflatable PVC items containing phthalates.

In the typical scenario for infants and children it is assumed that the daily dermal contact time with articles containing one or more of the four phthalates is **30 min**. In the reasonable worst case scenario, both infants and children are assumed to have dermal contact for **1½ hour** every day. This reflects that some articles are used very shortly, but frequently during a day and some are used for a longer period like furniture.

The total body surfaces for 6-12 months old and 6-11 years old are 4500 cm^2 and 10800 cm^2, respectively (Höglund et al., 2012). As the purpose of the whole exercise is to calculate exposure per kg body weight and a close correlation between total surface area and body weight exists, the distributions of surface area among the two populations groups – in order to estimate the reasonable worst case for this parameter is not calculated.

The surface area in contact with articles containing one or more of the four phthalates is assumed to be **10 % of the total body surface area**. For comparison, arms (excluding hands) equal to approximately 13 % of the total body surface, and hands equal to approximately 5 % of the total body surface. A contact area of 10% of the body surface could reflect that both shower mats (contact with sole of foot, app. 3% of surface), bicycle handles and covers/bags for smartphones or tablet (contact with the palm of the hand, app. 2% of surface) as well as sitting in furniture's made of artificial leather (contact with underarms, app. 6% of surface) are used during a normal day. Furthermore, DIBP in childcare articles could contribute to the exposure, as DIBP is not covered by the present Annex XVII entry (entry 52). Hence, **450 cm^2 and 1080 cm^2** are used for infants and children in the typical case scenario.

In the reasonable worst case scenario, both infants and children are assumed to have dermal contact with **25 % of the body surface**. The body surface used in the calculations for infants is **1125 cm²** and **2700 cm²** for 6/7-years old.

Central for the calculation of dermal exposure is the combination of exposure time and the exposed area. Some articles might be used for many hours but the exposure area could be rather small, while other articles might be used for a shorter period but expose a larger part of the body.

The exposed surface area could be expected to be large from articles like inflatable swimming pools, air mattresses and balance balls. However, there are several articles where the exposed area is expected to be much smaller, for example sandals, wrist watches and mats for sinks etc. However, for some of these articles a long exposure time could be expected.

For comparison Health Canada (2015c) used a contact time of 1 hour/day and 25% of body surface area in their typical scenario for infants and 4 hours/day and 50% of body surface area in their worst case scenario for infants. This is 5 times higher than used in our scenario. Health Canada's reasoning for the assumptions is the use of changing mats and playing on plastic mats.

Table B42 Assumptions for dermal contact time and area for infants and children

	infants		**Children**	
	Dermal contact time in min/d	**Dermal contact area in cm²**	**Dermal contact time in min/d**	**Dermal contact area in cm²**
Typical scenario	30	450	30	1 080
Reasonable worst case scenario	90	1 125	90	2 700

Contact time and contact area – women

Women are also expected to come into dermal contact with articles containing one or more of the four phthalates. It could be the same articles as for children but the exposure to for instance gloves for dish washing or cleaning, gardening gloves, garden hoses, handles on tools, furniture and PVC tape is expected to be higher. This exposure could take place both at work and at home.

Furthermore, women may be exposed to articles at work, e.g. at stores or production facilities. This exposure is not part of the assessment for occupational exposure as the articles are very heterogeneous and not directly linked to the production and use of the phthalates themselves.

In the typical scenario, it is assumed that women are in contact with articles containing one or more of the four phthalates for **½ hour** every day. In the reasonable worst case scenario, it is assumed that women are in daily contact with articles containing one or more of the four phthalates for **1½ hour**. This scenario takes into account that more articles are plasticized with one or more of the four phthalates and the time of contact is therefore increased compared to the typical scenario. These additional articles might be steering wheels, gloves for

dish washing or cleaning, garden gloves, garden hoses, furniture and PVC tape plasticized with phthalates. Some of these articles might be used for most of the day.

The total body surface area of adult women is 18100 cm^2 (Höglund et al., 2012). In the typical scenario **10%** of the total body surface area (**1810 cm^2**) is considered to be exposed and in the reasonable worst case scenario **12%** of the total body surface area (**2170 cm^2**) is considered to be exposed. The exposure might come from the use of several articles during a day, as for example wrist watches, shoes (sandals and rubber boots), swimming pool, shower mats and more. A contact area of 10-12% of the body surface could reflect the use of shower mats (contact with sole of foot, approximately 3% of surface), bicycle handles and covers/bags for smartphones or tablets (contact with the palm of the hand, approximately 2% of surface) and sitting in furniture made of artificial leather (contact with under arms, approximately 7% of surface).

For comparison Health Canada (2015c) used a contact time of 3 hour/day and 16 % of body surface area in their typical scenario and 3 hours/day and 50% of body surface area in their worst case scenario. This is 10 times higher than used in our scenario. Health Canada's reasoning for their assumptions is the contact with a couch, wearing plastic gloves, holding a plastic steering wheel and/or wearing plastic clothing – use patterns which should not differ from European uses.

Table B43 Assumptions for dermal contact time and area for women

	Women	
	Dermal contact time min/day	**Dermal contact area cm^2**
Typical scenario	30	1 810
Reasonable worst case scenario	90	2 170

Dermal exposure estimation

Table B44 Parameters used in the estimations of dermal exposure from articles

		Unit	**Typical case**	**Reasonable worst case**
Migration rate	DEHP	$\mu g/cm^2$	3.80	15.2
	DBP	$\mu g/cm^2$	6.10	17.2
	DIBP	$\mu g/cm^2$	5.40	17.2
	BBP	$\mu g/cm^2$	2.50	6.50
Exposure time	dermal	hours/day	0.5	1.5
Contact area	Infants(6–12 months)	cm^2	450	1125
	Children (6–11 years)	cm^2	1080	2700
	Women	cm^2	1810	2170
Bodyweight	Infants	kg	9.2	
	Children	kg	31.8	
	Women	kg	60.0	

Table B45 Further parameters used in the estimations of dermal exposure from articles

	DEHP	**DBP**	**DIBP**	**BBP**	**Total**
absorption rate, dermal	0.05	0.1	0.1	0.05	
Phthalate ratio in articles	0.74	0.08	0.08	0.1	1.00

On this basis the dermal exposure is estimated for the three age groups. Exposure estimates of typical and reasonable worst case scenarios are given in Table B46.

Table B46 Internal exposure estimates from dermal exposure (µg/kg bw/d)

	DEHP			**DBP**			**DIBP**			**BBP**		
	TC	RWC	RWC MC	TC	RWC	RWC MC	TC	RWC	RWC MC	TC	RWC	RWC MC
Infants	3.44	25.79	24.91	1.19	8.95	6.10	1.06	7.92	6.39	0.31	2.29	1.57
Children	2.39	17.91	17.26	0.83	6.22	4.39	0.73	5.50	4.49	0.21	1.59	1.13
Women	2.13	7.63	12.06	0.74	2.65	3.17	0.65	2.34	3.09	0.19	0.68	0.77

TC = Typical case scenario
RWC = Reasonable worst case scenario. In this estimate, the reasonable worst case estimates for exposure time and contact area are used, while the typical case migration rate is used.
RWC MC = Monte Carlo simulation of the reasonable worst case scenario (variation of exposure time, contact area and migration rate). Input parameters are given in table above. The minimum estimate of exposure time and contact area is assumed to be 10% of the typical case.

The reasonable worst case is a combination of reasonable worst case assumptions for both exposure time and dermal contact area using the typical estimate for migration of the phthalates. This means that the reasonable worst case estimate is reflecting a scenario where the worst case situation arise for both contact area and exposure time. In the Monte Carlo estimations all three parameters is considered not to be correlated and also the variations in the migration rates are taken into consideration.

In the table above some of the estimates for the dermal exposure in reasonable worst case are lower than the estimates derived in a Monte Carlo simulation. This reflects that in the latter variations in the migration rates are taken into consideration, while the first ones are based on mean values for migration. As shown in Table B38 the migrations rates vary quite much.

B.8.4.6.5. Oral exposure from erasers containing phthalates

The exposure from contact with some specific articles that might lead to high oral exposures was considered. In particular erasers were identified to be a possible source of high oral exposure.

For children (6-11 years), the oral exposures have been calculated for erasers, where the mouthing time is assumed to be 60 minutes for both the typical scenario and the reasonable worst case scenario.

Children might also ingest small parts of erasers. In the reasonable worst case scenario a child is assumed to ingest 8 mg eraser per day. This corresponds to approximately one sesame seed. It is only assumed that children will be exposed to erasers, but it cannot be excluded that women also will be exposed through this route (i.e. sucking the end of a pencil with an eraser attached).

The exposure time estimates for the oral exposure are considered to be realistic for the average child but it will vary between individuals.

The oral exposures are estimated based on analyses of the migration from articles that can be expected to be put into the mouth, based on migration to artificial saliva. The migration rates and assumptions can be seen in ECHA (2012a), table 16. No migration of DBP, DIBP and BBP from the articles was measured.

The oral exposure from articles is calculated and given in Table B47. The values are used to calculate the risk characterisation ratio in B.9.

Table B47 Calculated direct oral exposure of DEHP from erasers.

	Measured migration of DEHP in mg/g	**Exposure of children to DEHP (ug/kg bw/day)**
Eraser mouthing* (typical scenario)	0.0833**	15.8
Eraser mouthing and eating 8 mg* (reasonable worst case scenario)	440 000***	176.0

*calculations for erasers are based on a child weighing 20 kg

** The original measured migration rate is divided with a factor of 6 to take into account that the migration was measured from small pieces of eraser, giving a larger surface, and divided by 2 to take account the uncertainty of the analysis which is 50%. It is assumed that the weight of eraser mouthed is 3.79 g corresponding to 1 cm^2 of the eraser is mouthed.

***content in mg/kg

B.8.4.6.6. Dermal exposure from specific articles containing phthalates

The exposure from some specific articles that might lead to high dermal exposures was considered. In particular plastic sandals and sex toys were identified to be a possible source of high dermal exposure.

As described in detail in ECHA (2012a) exposure durations are given as a typical estimate for the three age groups. Body surface area to be exposed is calculated based on ECHA guidance R15 (2010) and US EPA Child-Specific Exposure Factors Handbook 2002 and the age specific assumptions are given in BD12, Table B12, 13, 14, 16, 16a and 18.

Table B48 contains the result in form of the estimation of dermal exposure of DEHP, DIBP and DBP in specific consumer articles contributing with a high exposure. The age groups from Background Document (ECHA 2012a) are shown.

Table B48. Calculated dermal exposure of DEHP, DIBP and DBP in specific consumer articles (µg/kg bw/day)

Article	DEHP			DIBP			DBP		
	Infants	Children	Women	Infants	Children	Women	Infants	Children	Women
Plastic sandals median exposure	0.90	1.87	0.71	0	0	3.76	0	0	0
Plastic sandals worst case exposure	3.62	0	1.44	3.56	0	2.61	0	3.91	5.450
Sex toys*	0	0	0.001/0.92	0	0	0	0	0	0

*The first value is based on the migration to artificial sweat and the second value is based on the migration to artificial sweat + oil based lubricant (worst case scenario).

It should be noted that a special situation exists for DIBP in relation to childcare articles such as changing mats and car seats for children. DEHP, DBP and BBP are banned in childcare articles, but as DIBP is not banned, it can be present in imported childcare articles. Children could therefore be exposed dermally to DIBP from childcare articles as changing mats, bibs or car seats.

According to market surveillance data, DIBP is used in 1-3% of toys and childcare articles with flexible PVC (often together with other phthalates), with an average concentration of 10-20% of PVC content. Although historical information is not available to confirm that DIBP is replacing DBP in toys, the substitution is feasible, given their structural and pricing similarities. RAPEX entries and a recent PROSAFE survey have shown non-compliance with the Toy Safety Directive limit for DIBP on a number of occasions.

Using for DIBP the same assumptions as done by ECHA in their evaluation of DINP and DIDP in toys and childcare articles (ECHA 2013a,b), reasonable worst case internal exposure estimates for 6-12 months old infants from DIBP in toys and childcare articles would be 118 µg/kg

bw/day (from mouthing) and 5 µg/kg bw/day (from dermal contact). Toys not in compliance with the toys Directive and childcare articles could therefore contribute to a relatively high exposure to DIBP.

Children will also have dermal contact with other articles containing one or more of the four phthalates as wrist watches and covers and bags for mobile phones and tablets (Danish EPA, 2015) and other articles as those listed in Table B34. The exposed dermal surface area will depend on the articles used. If DIBP is used in a changing mat, the exposed dermal area could be as large as half of the body surface area.

B.8.4.6.7. Uncertainties to the exposure estimates from contact with articles

Modelling exposure includes making assumptions on migration of phthalates and use of articles. As can be seen from Table B38 migration rates vary and depend on many factors. The used migration rates for the four phthalates could therefore be both under- or overestimated.

In the calculation of migration rates it is assumed that the migration of phthalates will happen within the first hour of migration even if the migration is measured over several hours. There are studies showing that the migration of DINP is linear at least på to 4 hours (Oomen et al. 2004). As a number of the tests presented in table B33-B36 resulting in quite high migration levels have been carried out for a shorter period than 1 hour, an estimation based on migration rates using the actual migration time would result in higher exposure estimates.

Furthermore, a newly published report from the Danish EPA has suggested migration rates for DEHP, DBP, DIBP, BBP and DINP. The conclusion is that many factors are affecting the migration and in particular the migration method and that the most realistic migration results are from the head-over-heals (HOH) method. Deriving migration rates only from this method results in higher migration rates than the migration rates used and proposed here and would lead to higher exposure and RCR-values.

Humans are different and behave differently and consume differently. This will lead to variations uncertainties in the use of articles. As an example, observations on mouthing time show very large variations in mouthing times for infants and toddlers. In the exposure modelling it has also been assumed that children in the age of 6-11 will not mouth any articles, but experiences from toys show that children in this age group also mouth toys, but it is considered to be more seldom.

In relation to calculation the exposure in the reasonable worst case scenario it is important whether the individual parameters are correlated or not.

Very seldom the reasonable worst case would apply for all parameters at the same time. To estimate the overall reasonable worst Monte Carlo simulations have been carried out using 10 000 iterations of random combinations. For articles, Monte Carlo simulations have been carried out with regard to migration rate from PVC articles, exposure time and contact area.

The calculations are estimations based on assumptions on normal distribution of the individual parameters, e.g. exposure duration. Considering the overall uncertainties, the precise shape of the distributions is considered to be of minor importance. Furthermore some correlation

could be present. E.g. that people living in a very “plasticised” environment would be more exposed to two or more of the four phthalates. Therefore, the result should be interpreted cautiously.

The best case assumptions (representing the situation with the lowest risk) necessary for carrying out Monte Carlo Simulations are not known for most parameters. In these cases the best case is estimated to be 10% of the typical case (exposure duration and contact area). Sensitivity analysis have been carried out to analyse the effect if the best case estimate for the parameter would be zero, which had little effect on the outcome of the estimation of the 95th percentile, which is the parameter considered to be relevant in relation to the reasonable worst case scenario.

B.8.4.6.8. Conclusion – exposure from contact with articles

The modelled exposure from contact with articles depends on the assumptions made on the use of articles, the migration rates and the share between the four phthalates. Normally the highest exposure would be expected from oral exposure as the absorption of phthalates is assumed to be 100%. From the calculations made here only a relatively low exposure is anticipated from oral exposure. This is due to the assumption that the mouthing times are low and it is not expected that there will be an intake of articles. The highest exposure is from DEHP and there is no exposure of BBP from oral exposure. This is due to the low migration rate of BBP together with the low share of BBP used in articles compared to the other three phthalates.

The dermal exposure is highest for infants and lowest for women. As for the oral exposure the highest exposure is calculated for DEHP and the lowest exposure from BBP.

Migration studies from specific articles like erasers, sandals and sex toys, show that individual articles can contribute to a relatively high exposure. The direct exposure from articles will therefore depend on the articles used and the content and migration of phthalates from these articles together with the use of the articles.

B.8.4.7. Aggregated exposure from indoor environment, food and contact with articles

The results from B.8.4.2 – B.8.4.6 the exposure from the individual phthalates is summarised in Table B49:

Table B49 Aggregated exposure from indoor environment, food and contact with articles for each phthalate (µg/kg bw/day)

	Infants			Children			Women		
	Typical	RWC	MC RWC	Typical	RWC	MC RWC	Typical	RWC	MC RWC
DEHP									
Indoor	4.22	21.85	21.85	0.93	5.51	5.51	0.48	2.52	2.52
Food	4.66	7.09	7.09	3.50	5.38	5.38	1.49	2.86	2.86
Articles	3.49	27.32	27.67	2.39	17.91	17.26	2.12	7.63	12.06
Total	12.37	56.26	56.61	6.82	28.80	28.15	4.09	13.01	17.45
Monte Carlo			42.98			22.38			14.17
DBP									
Indoor	0.28	1.47	1.47	0.04	0.27	0.27	0.02	0.12	0.12
Food	0.70	1.24	1.24	0.20	0.30	0.30	0.08	0.16	0.16
Articles	1.20	9.22	6.48	0.83	6.22	4.39	0.74	2.65	3.17
Total	2.18	11.93	9.19	1.07	6.79	4.96	0.84	2.92	3.45
Monte Carlo			6.63			4.63			3.27
DIBP									
Indoor	0.27	1.41	1.41	0.04	0.25	0.25	0.02	0.11	0.11
Food	1.03	9.02	9.02	0.42	0.64	0.64	0.14	0.28	0.28
Articles	1.06	8.16	6.74	0.73	5.50	4.49	0.65	2.34	3.09
Total	2.37	18.59	17.18	1.19	6.40	5.39	0.82	2.74	3.48
Monte Carlo			12.19			4.94			3.28
BBP									
Indoor	0.08	0.42	0.42	0.01	0.08	0.08	0.01	0.03	0.03
Food	0.15	0.24	0.24	0.12	0.21	0.21	0.05	0.12	0.12
Articles	0.31	2.43	1.75	0.21	1.59	1.13	0.19	0.68	0.77
Total	0.54	3.09	2.41	0.34	1.87	1.41	0.25	0.83	0.92
Monte Carlo			2.03[75]			1.25			0.83

Typical = Typical case scenario

RWC = Reasonable worst case scenario

RWC MC = Monte Carlo simulation of the reasonable worst case scenario

[75] Contribution from food to infants' exposure of BBP was not included in the Annex XV report and therefore not included in the calculation of weighted averages, used further in the risk characterisation. The original value of 1.90 from the Annex XV report is brought forward into the risk characterisation estimation. This is considered to be justifiable taking into account that the difference of 0.13 µg/kg bw/day corresponds to an RCR of 0.00026.

For DEHP, food is the dominant source for infants' and children's exposure in the typical case, while contact with articles dominates for women.

For the reasonable worst case, contact with articles seems to be the main source for DEHP sources for all age groups. Indoor environment contributes by 14 and 12 percent for children and women, while for infants indoor environment count for 34%.

Results from biomonitoring studies indicate that food is a dominant source of DEHP exposure. This is not supported by the modelled data. These variations illustrate that exposure estimations in general are uncertain due to large number of varying parameters.

For DBP the main source seems to be contact with articles for all three age groups for both the typical and the reasonable worst case.

The same applies for DIBP, even if the contribution from food seems to be higher, especially for infants and children.

The exposure from dust is though expected to be overestimated, as the assumed intake of dust of 100 mg/d and 50 mg/d for infants and women respectively seem to be very high.

Table B50 Assumptions used for modelling of exposure

		Unit	Average	Best *	Worst
Migration rate	DEHP	μg/cm²/h	3.8	0.1	15.2
	DBP	μg/cm²/h	6.1	0.2	17.2
	DIBP	μg/cm²/h	5.4	0.2	17.2
	BBP	μg/cm²/h	2.5	0.3	6.5
Exposure time	Dermal	hours/day	0.5	0.05	1.5
	Oral, infants	hours/day	0.017	0.002	0.5
Contact area	Oral, infants	cm²	10	10	10
	Dermal, infants	cm²	450	45	1125
	Dermal, Children	cm²	1080	108	2700
	Dermal, Women	cm²	1810	181	2170
Body weight	Infants	kg	9.2	7.1	11.3
	Children	kg	31.8	19.7	52.5
	Women	kg	60	45	85

* The best estimate is used in the Monte Carlo simulations for values that would result in the lowest risk ratio (in principle the 5th percentile)

Table B51 Further parameters used to assess exposure form contact with articles

	DEHP	DBP	DIBP	BBP
Absorption rate, dermal	0.05	0.1	0.1	0.05
Absorption rate, oral	1	1	1	1
Phthalate ratio	0.74	0.08	0.08	0.1
DNEL, µg/kg bw/day	35	6.7	8.3	500

B.8.5. Mixture effects

The combined risk assessment of DEHP, DBP, DIBP and BBP considers only these phthalates, but other substances may contribute to mixture effects on male reproductive development.

The MoA of these phthalates, i.e. decreased testosterone, is only one of several MoAs that can affect the Adverse Outcome Pathway (AOP) for anti-androgenic effects and thereby lead to adverse effects on male reproductive development. Other important MoAs include decreased dihydro-testosterone level and androgen receptor antagonism (NRC 2008). Several substances are evaluated to be able to affect this AOP, i.e. to cause anti-androgenic effects, based on in vivo studies showing adverse effects on male reproductive development and is some cases also mechanistic data showing anti-androgenic MoA. Exposure to other substances affecting male reproductive development can contribute significantly to the total risk. Therefore, the combined risk assessment of DEHP, DBP, DIBP and BBP alone is likely to be an underestimation of the risk for mixture effects on male reproductive development.

Further, during a meeting of scientific experts organized by the Chief Scientific Advisor to the Commissions President in October 2013 it was noted that *"for many chemicals zero exposure does not exist – in fact, all organisms have a background exposure to a range of chemicals, so it is essential to understand how an ED contributes to and interacts with this background"* (European Commission 2013). This would apply to both persistent and non-persistent chemicals.

B.8.6. Occupational exposure

All four substances are listed in Annex XIV of REACH, implying that the substances may not be used in the EU unless an authorisation is granted. Applications have been submitted for DEHP and DBP, only.

DEHP

Workers are exposed to DEHP during manufacturing of DEHP, the formulation of DEHP (compounds, dry-blends and plastisol formulations) and the production of articles (polymer processing by calendering, spread coating, extrusion, injection moulding).

Workers are furthermore exposed to the four substances during formulation of recycled soft PVC containing DEHP[76] in compounds and dry-blends.

During the service life stage of articles worker exposure may also occur. The applicants for DEHP described the following relevant article service life exposure scenarios for professionals and industrial workers:

[76] There may be additional recyclers whose activities do not fall under the authorisation requirement but where occupational exposure to DEHP, DBP, DIBP and BBP occurs.

- Professional handling of PVC articles: Installation of building materials and similar activities (dermal and inhalation exposure)[77]; and
- Professional and industrial workers wearing PVC work clothes and footwear (waterproof overall/rainwear, waterproof boots, clogs) (dermal exposure)[78].

RAC confirmed that the risk assessment based on the limited exposure data in the application does not demonstrate adequate control of risks for workers from the use applied for. RAC's assessment based on these limited exposure data in the application showed a risk for the use applied for.

DBP

Applications have been submitted for a number of uses.

1- Use of DBP as an absorption solvent in a closed system in the manufacture of maleic anhydride.
2- Industrial use of DBP as a burning rate surface moderant, plasticiser and/or coolant in the formulation of nitrocellulose-based propellant grains.
3- Industrial use of DBP-containing propellant grains in manufacture of ammunition for military and civilian uses, and pyrocartridges for aircraft ejection seat safety systems
4- Industrial use of DBP in ceramic sheets and printing pastes for production of capacitors and lambda sensor elements
5- Industrial use of DBP in manufacture of solid propellants and motor charges and within a specialty paint in manufacture of motors for rockets and tactical missiles.

For all applications for DBP, RAC confirmed that the exposure assessments in the applications demonstrated adequate control of risks from the use applied for provided that the risk management measures and operational conditions as described in the applications are adhered to.

The details are available in the Applications for authorisation and the associated opinions on the applications for authorisation by RAC and SEAC (ECHA 2014d,e,f).

[77] This contributing scenario covers the handling of PVC articles primarily by craftsmen, such as: roofing and flooring products, HVAC products and coated steel wire mesh fencing. It also covers PVC articles handled by workers in other professions, e.g. office clerks, landscape gardeners, shop assistants, such as: office supplies (such as files, slip cases and ring binders), outdoor products (such as garden hoses and tubes, wood-PVC composite profiles (fences, patio profiles)) and many other items (e.g. packaging material, tape and self-adhesive foils, haberdashery, luggage, briefcases, bags, rainwear, footwear as well as curtains and blinds), as exposure (in terms of skin surface exposed and/or exposure duration/frequency) is obviously lower compared to daily dermal contact for several hours to building materials by craftsmen.

[78] This contributing scenario covers waterproof clothes (trousers and jacket or overall) and footwear (boots), as worn e.g. by professional fishermen, but also e.g. in the food industry. It also covers non-waterproof footwear, such as clogs and safety shoes worn in the professional environment, e.g. by hospital staff, pool attendants etc.

B.9. Risk characterisation

Risk characterisation is only performed for the health of the general public. Risks only related to manufacturing, formulation and use of the substances have been assessed by RAC under the authorisation scheme.

In the DEMOCOPHES project, a survey of the phthalate burden in humans in 16 EU Member States and Switzerland has been carried out showing that the phthalates are present in all humans in Europe, and in some countries above the safe level (Section B.9.1).

To evaluate how possible sources might contribute to this risk level, modelling has been carried out (Section B.9.2).

Furthermore Table B47 and Table B48 show how specific articles might contribute to exposure

The dose addition principle is applied to summarise the risk of combined phthalate exposure by adding risk characterisation ratios (RCRs). The RCR for a chemical is defined as the ratio between exposure level and DNEL (ECHA part E 2008). The RCR is calculated as the ratio between the internal exposure estimates and the internal DNEL for the individual phthalates.

$$RCR = \frac{Exposure}{DNEL}$$

If the RCR for a substance exceeds 1, i.e. when exposure exceeds the DNEL, it may be concluded that the risk is not adequately controlled (ECHA guidance part E 2008). In a situation with exposure to several similarly acting chemicals, the dose addition principle will imply that a combined RCR can be calculated by adding the RCR for each chemical, see section B.1.5. When this combined RCR exceeds 1 the risk is considered not to be controlled for the chemicals comprised by the combined RCR.

The exposure and the risk are calculated for the three different population groups:

- Infants (boys of 6-12 months)
- Children (boys of 6-11 years)
- Women (developing male foetus in pregnant women)

However, for biomonitoring DEMOCOPHES provides data only for women and children (6-11 years).

The risk characterisation in the following sections has only been carried out for PVC articles containing the four phthalates. However, there is evidence that some adhesives or sealants used in articles also contain DBP, DIBP and BBP (see Annex A). There is little available evidence on the migration of the phthalates from this material but we assume that the risk from these materials is similar to their plasticiser in PVC uses. This assumption can be further tested in the Public Consultation.

B.9.1. Risk characterisation based on biomonitoring data

The intake of DEHP, DBP, DIBP and BBP from section B.8.3.2 are compared with the oral DNELs from Table B9.

As can be seen from Table B52 and Figure B8 below, RCRs for 95th percentile exposure of children to DBP are above 1 in Poland, the Czech Republic, the Slovak Republic, equal to 1 in Sweden, and in Spain close to 1. The RCRs for 95th percentile exposure of children to DIBP are above 1 in Poland and equal to 1 in Belgium. The RCR for 95th percentile exposure of children to DEHP is close to 1 in Romania and in mothers equal to 1. The geometric mean exposure values in Romania are also high compared to other European countries (see Table B13).

The very low RCRs from BBP exposure is due to the high DNELs in comparison to the DNELs for the other phthalates and BBP also has the lowest estimated intakes amongst the four phthalates (median values are about an order of magnitude lower than for DEHP).

In 13 out of 15 Member States (87%)[79] RCRs for combined 95th percentile exposure to DEHP, DBP, BBP and DIBP are at or above 1 for children. For 5 out of these Member States RCRs are equal to or above 1 also for mothers, with a 6th Member State having an RCR above 1 in mothers but not in children (Cyprus).

In Polish children the geometric mean exposure values approach an RCR of 1 (RCR of 0.86). The RCR corresponding to the median and mean exposure values are 0.81 and 1.13 respectively in Polish children. The 3 Member States with the highest 95th percentile combined exposure values (Poland, Spain and Romania), are also amongst the Member States with highest geometric mean combined exposure values.

Uncertainties to the exposure estimates from DEMOCOPHES data have been discussed in section B.8.3 and are summarised here:

- There are uncertainties to the estimates as a result of data availability issues. The effect appears to be minimal based on a comparison of our estimates and published estimates for DK.
- When using volume based method of intake calculation from urinary biomonitoring data higher RCRs may be obtained (possibly by a factor of 2).
- The children in the study population of DEMOCOPHES were 6-11 years old. Younger children are likely to have higher exposure to the four phthalates (see section B.8.3 - Discussion) and thus the RCR values are not representative for children younger than 6 years.
- The FUEs used for children are for adults and may result in underestimation of exposure to DBP, DIBP and BBP.
- Adding RCRs based on 95th percentiles of several phthalates may lead to some overestimation of the RCRs but consistent evidence indicates that individuals exposed to high levels of one of the four phthalates tend to be exposed at high levels to other phthalates as well.
- The RCRs for combined exposure are underestimated for Slovenia since no measurement of DIBP metabolites was available. For the same reasons, the RCRs for

[79] Excluding the UK (small sample size, n= 21) and Switzerland as it is not part of the EU.

the Slovak Republic, Sweden, Czech Republic and Hungary may also be underestimated, although potential issues with chromatic separation may have compensated for the lack of a measurement value for DIBP (see section B.8.3.2).

- The sample size (n=21) of the UK data is not considered representative for the exposure in the UK.

Additional contribution from exposure to other anti-androgenic phthalates such as DINP[80], DnHP, DIHepP, DnHepP adds further to the body burden (Health Canada 2015; ECHA 2013a). Other substances might also add further to the body burden, e.g., Vinclozolin, Prochloraz, Procymidone and p,p'-DDE[81] (Kortenkamp and Faust 2010).

There are furthermore uncertainties to the DNELs as discussed in section B.4.2 Toxicity for reproduction.

Table B52 RCRs for exposure to the four phthalates as estimated from 95th percentile urinary biomonitoring exposure levels from DEMOCOPHES data from 2011-2012

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.4	0.0	*NA*	0.6	120	0.2	0.4	0.0	*NA*	0.6
UK	21	0.1	0.1	0.0	0.3	0.5	21	0.2	0.3	0.0	0.3	0.7
CH	117	0.2	0.3	0.0	0.2	0.6	119	0.2	0.3	0.0	0.3	0.8
CY	59	0.4	0.2	0.0	0.4	1.1	60	0.2	0.2	0.0	0.4	0.9
PT	117	0.3	0.2	0.0	0.3	0.9	116	0.3	0.3	0.0	0.4	1.0
IE	120	0.2	0.2	0.0	0.4	0.8	120	0.3	0.3	0.0	0.5	1.0
HU	115	0.2	0.5	0.0	*NA*	0.7	117	0.4	0.7	0.0	*NA*	1.1
LU	60	0.1	0.2	0.0	0.3	0.6	60	0.1	0.3	0.0	0.7	1.1
DK	143	0.2	0.2	0.0	0.4	0.7	142	0.2	0.3	0.0	0.6	1.1
DE	116	0.1	0.3	0.0	0.2	0.7	120	0.2	0.5	0.0	0.4	1.1
SE	96	0.2	0.7	0.0	*NA*	0.9	97	0.3	1.0	0.0	*NA*	1.3
SK	125	0.2	0.8	0.0	*NA*	1.0	127	0.4	1.1	0.0	*NA*	1.5
CZ	117	0.2	0.7	0.0	*NA*	1.0	120	0.4	1.3	0.0	*NA*	1.7
BE	125	0.1	0.4	0.0	0.6	1.1	125	0.4	0.4	0.0	1.0	1.8
RO	117	1.0	0.3	0.0	0.3	1.6	119	0.9	0.6	0.0	0.6	2.1
ES	118	0.3	0.3	0.0	0.3	0.9	119	0.4	0.9	0.0	0.9	2.1
PL	119	0.4	0.8	0.0	0.7	1.9	115	0.5	1.1	0.0	1.2	2.9

NA = not available

[80] An RCR of about 0.04 - 0.06 for reproductive toxicity may result from reasonable worst case DINP exposure based on urinary biomonitoring studies in Germany, Denmark and the Netherlands (ECHA 2013a).

[81] Median intake RCRs for Vinclozolin, Prochloraz, Procymidone and p,p'-DDE were calculated to be 0.12, 0.02, 0.03 and 0.003 respectively (Kortenkamp and Faust 2010). Thus, and additional background body burden might be an RCR of around 0.17.

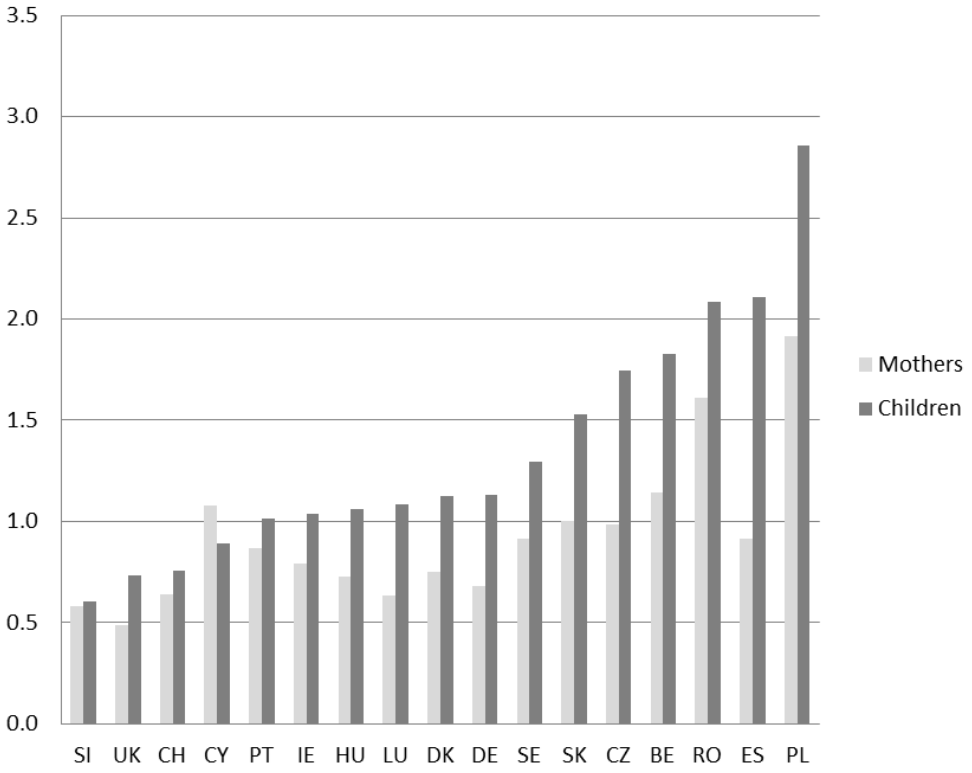


Figure B8 RCRs for combined exposure to the four phthalates as estimated from 95th percentile urinary biomonitoring values

Table B53 RCRs for exposure to the four phthalates as estimated from geometric mean (GM) urinary biomonitoring values

		Mother						Child				
Country	N	DEHP	DBP	BBP	DIBP	SUM	N	DEHP	DBP	BBP	DIBP	SUM
SI	120	0.1	0.1	0.0	*NA*	0.2	120	0.1	0.1	0.0	*NA*	0.2
CH	117	0.0	0.1	0.0	0.1	0.2	119	0.1	0.1	0.0	0.1	0.2
CY	59	0.0	0.1	0.0	0.2	0.3	60	0.0	0.1	0.0	0.2	0.3
LU	58	0.0	0.1	0.0	0.1	0.2	60	0.0	0.1	0.0	0.1	0.3
UK	21	0.0	0.1	0.0	0.1	0.1	21	0.1	0.1	0.0	0.1	0.3
HU	115	0.1	0.2	0.0	*NA*	0.2	117	0.1	0.2	0.0	*NA*	0.3
IE	120	0.1	0.1	0.0	0.1	0.2	120	0.1	0.1	0.0	0.1	0.3
PT	117	0.1	0.1	0.0	0.1	0.3	116	0.1	0.1	0.0	0.1	0.4
DE	116	0.0	0.1	0.0	0.1	0.3	120	0.1	0.2	0.0	0.1	0.4
BE	125	0.0	0.1	0.0	0.1	0.3	125	0.1	0.2	0.0	0.2	0.4
DK	143	0.0	0.1	0.0	0.2	0.3	142	0.1	0.1	0.0	0.2	0.4
SE	96	0.1	0.3	0.0	*NA*	0.3	97	0.1	0.4	0.0	*NA*	0.5
RO	117	0.1	0.1	0.0	0.1	0.3	119	0.1	0.2	0.0	0.2	0.5
SK	125	0.1	0.3	0.0	*NA*	0.4	127	0.1	0.4	0.0	*NA*	0.6
ES	118	0.1	0.1	0.0	0.1	0.4	119	0.1	0.2	0.0	0.2	0.6
CZ	117	0.1	0.3	0.0	0.0	0.3	120	0.1	0.4	0.0	0.0	0.6
PL	119	0.1	0.2	0.0	0.2	0.5	115	0.1	0.4	0.0	0.4	0.9

NA = not available

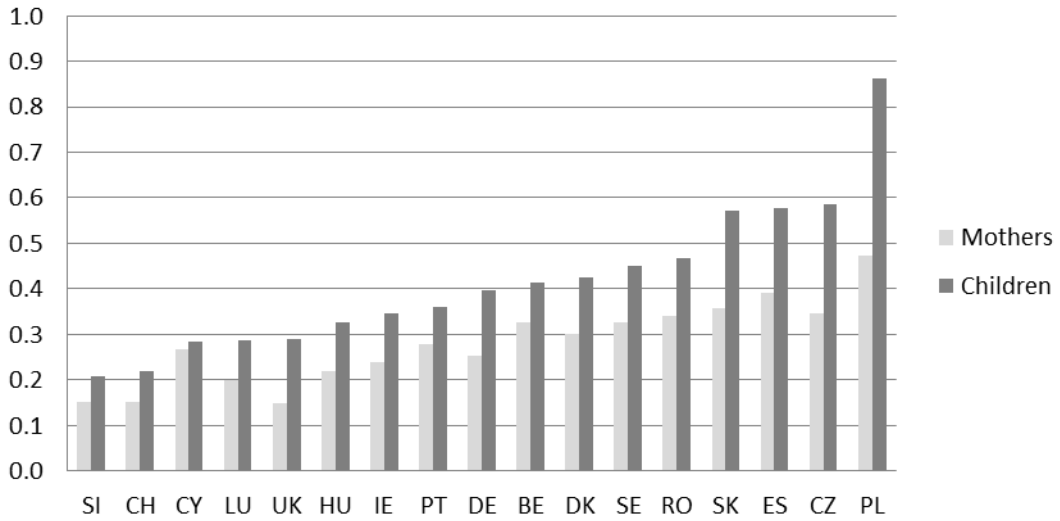


Figure B9 RCRs for combined exposure to the four phthalates as estimated from geometric mean urinary biomonitoring values

Table B54 presents the overall RCRs for exposure to the four phthalates as estimated from unweighted creatinine corrected urinary concentrations from DEMOCOPHES data from 2011-2012. In comparison, the 95th percentile values weighted according to the number of live births in the EU Member States[82] results in an RCR value of 1.1 in mothers and 1.7 in children.

Table B54 Overall RCRs for exposure to the four phthalates as estimated from unweighted creatinine corrected urinary concentrations from DEMOCOPHES data from 2011-2012.

Population		DEHP	DBP	BBP	DIBP	SUM
Mothers	Min	0.0	0.0	0.0	0.0	0.0
	GM	0.1	0.1	0.0	0.1	0.3
	P50	0.1	0.1	0.0	0.1	0.3
	P90	0.2	0.2	0.0	0.3	0.7
	P95	0.2	0.3	0.0	0.4	0.9
	Max	3.6	9.8	0.0	1.4	14.8
Children	Min	0.0	0.0	0.0	0.0	0.0
	GM	0.1	0.1	0.0	0.2	0.4
	P50	0.1	0.1	0.0	0.2	0.4
	P90	0.3	0.4	0.0	0.5	1.1
	P95	0.4	0.6	0.0	0.6	1.5
	Max	7.5	3.7	0.0	5.9	17.1

A comparison of the unweighted 90th percentiles for DEHP from Table B54 and the weighted 90th percentiles in the opinions on the Applications for Authorisation for DEHP (AfA 2013a-c) is made in Table B55. The estimates in the opinions (AfA 2013a-c) were based on preliminary results from a presentation but the data (µg/l) was confirmed in Den Hond et al. (2015), thus any further processing of the data seems not to have had an influence on these figures. The DNEL for DEHP used in the opinions (AfA 2013a-c) are the same as in the current restriction report. Lower exposure estimates and RCRs were derived in the restriction report based on the DEMOCOPHES data when compared to the RAC opinions on the DEHP Applications for Authorisation. This is due to difference in methodology: in AfA (2013a-c) the volume correction

[82] Excluding the UK and Switzerland, thus 15 of the 17 participating countries in DEMOCOPHES.

approach was used whereas the restriction proposal used the creatinine method. Furthermore, Table B54 used unweighted values, whereas AfA (2013a-c) used weighted values. This comparison confirms that the volume correction approach results in typically higher exposure estimates.

Importantly, the current restriction report considered combined exposure from multiple phthalates whereas AfA (2013a-c) was limited to assessing the risks from DEHP.

Table B55 Comparison of the consumer risk assessments for DEHP in the opinions on the Applications for Authorisation for DEHP (AfA 2013a-c) and in this report

	Women		**Children**	
	AfA (2013a-c)	**This report**	**AfA (2013a-c)**	**This report**
90th p urinary DEHP metabolite concentration*	91 µg/l **	70.2 µg/gC	137 µg/l **	120.3 µg/gC
90th p exposure (µg/kg bw/day)	9	5.6	10	8.5
DNEL (µg/kg bw/day)	34	34	34	34
RCR	**0.3**	**0.16**	**0.3**	**0.25**

*sum of the three metabolites MEHP, 5OH-MEHP and 5oxo-MEHP from the 17 countries participating in DEMOCOPHES.
**These values correspond to the weighted values in Table 2 in Den Hond et al. (2015). The non-weighted values in Table S3 were just slightly higher.

A projection was made to estimate the risk in 2014[83]. Comparing data from Table B56 and Table B57 with data from Table B52 and Table B54, it can be seen that no strong decline in the risk levels in 2014 can be expected to have occurred between 2011 and 2014 (e.g., the RCR is 2.6 in Poland in 2014 and 2.9 in 2011).

[83] Projections for 2016 were not available as a result of limitation in the available data. See also Annex D for projections further into the future. Those projections are however more uncertain and not relevant to this section (identified risk).

Table B56 RCRs for exposure to the four phthalates as estimated from 95th percentile urinary biomonitoring exposure levels from DEMOCOPHES data extrapolated from 2011/2012 to 2014

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.4	0.0	*NA*	0.5	120	0.2	0.4	0.0	*NA*	0.6
UK	21	0.1	0.1	0.0	0.2	0.4	21	0.2	0.3	0.0	0.3	0.7
CH	117	0.2	0.2	0.0	0.2	0.6	119	0.2	0.3	0.0	0.2	0.7
CY	59	0.4	0.2	0.0	0.4	1.0	60	0.2	0.2	0.0	0.4	0.8
PT	117	0.3	0.2	0.0	0.3	0.8	116	0.3	0.3	0.0	0.4	0.9
IE	120	0.2	0.2	0.0	0.3	0.7	120	0.3	0.2	0.0	0.4	1.0
LU	60	0.1	0.2	0.0	0.3	0.6	60	0.1	0.2	0.0	0.7	1.0
HU	115	0.2	0.4	0.0	NA	0.7	117	0.4	0.6	0.0	NA	1.0
DK	143	0.2	0.2	0.0	0.4	0.7	142	0.2	0.3	0.0	0.5	1.0
DE	116	0.1	0.3	0.0	0.2	0.6	120	0.2	0.5	0.0	0.3	1.0
SE	96	0.2	0.7	0.0	NA	0.8	97	0.3	0.9	0.0	NA	1.2
SK	125	0.2	0.7	0.0	NA	0.9	127	0.4	1.0	0.0	NA	1.4
CZ	117	0.2	0.7	0.0	NA	0.9	120	0.4	1.2	0.0	NA	1.6
BE	125	0.1	0.4	0.0	0.5	1.1	125	0.4	0.4	0.0	0.9	1.7
ES	118	0.3	0.3	0.0	0.3	0.9	119	0.4	0.8	0.0	0.8	1.9
RO	117	1.0	0.2	0.0	0.3	1.5	119	0.9	0.5	0.0	0.6	2.0
PL	119	0.4	0.8	0.0	0.7	1.8	115	0.5	1.0	0.0	1.1	2.6

NA = not available

Table B57 presents the overall RCRs for exposure to the four phthalates as estimated from unweighted creatinine corrected urinary concentrations from DEMOCOPHES data extrapolated from 2011/2012 to 2014. In comparison, the 95th percentile values weighted according to the number of live births in the EU Member States[84] results in an RCR value of 1.0 in mothers and 1.6 in children.

Table B57 Overall RCRs for exposure to the four phthalates as estimated from unweighted creatinine corrected urinary concentrations from DEMOCOPHES data extrapolated from 2011/2012 to 2014.

Population		**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
Mothers	Min	0.0	0.0	0.0	0.0	0.0
	GM	0.1	0.1	0.0	0.1	0.3
	P50	0.1	0.1	0.0	0.1	0.3
	P90	0.2	0.2	0.0	0.3	0.6
	P95	0.2	0.3	0.0	0.3	0.9
	Max	3.6	8.9	0.0	1.3	13.7
Children	Min	0.0	0.0	0.0	0.0	0.0
	GM	0.1	0.1	0.0	0.1	0.4
	P50	0.1	0.1	0.0	0.1	0.4
	P90	0.2	0.3	0.0	0.4	1.0
	P95	0.4	0.5	0.0	0.5	1.4
	Max	7.5	3.4	0.0	5.3	16.2

Based on the combined RCRs from DEMOCOPHES biomonitoring data, the population at risk in 2014 has been estimated as the percentage of mothers (boys exposed in utero) and children exceeding an RCR value of 1 for the individual 15 EU Member States (except UK). The overall percentage of the population at risk from these 15 Member States was used to extrapolate to the remaining 13 Member States. The estimations assume a lognormal distribution. The standard deviation of the lognormal distributions was derived per country from the natural logarithmic values of the measure 95th percentile and the geometric mean (2011 values

[84] Excluding the UK and Switzerland, thus 15 of the 17 participating countries in DEMOCOPHES.

projected to 2014)[85]. Table B58 shows that in the EU28 about 5.1% of new born boys (130 000) were at risk through in utero exposure and about 15.5% boys (400 000) were at risk from direct exposure in 2014. In 2011, the percentages were 6% and 18%, respectively. Importantly, the Table B58 shows there is a risk in all Member States.

Table B58 Estimate of the population at risk in 2014 in the EU28

Member State	live births/ year	boys born/year	Mothers (boys exposed in utero)			Boys		
			combined 95th percentile RCRs	% at risk	Number of boys at risk	combined 95th percentile RCRs	% at risk	Number of boys at risk
SI	21 165	10 583	0.54	0.8%	88	0.57	0.6%	61
CY	9 258	4 629	1.01	5.2%	240	0.83	2.8%	128
PT	82 367	41 184	0.82	2.7%	1 103	0.94	4.1%	1 683
IE	66 520	33 260	0.74	1.9%	643	0.97	4.5%	1 493
HU	93 300	46 650	0.68	1.5%	712	1.00	5.0%	2 314
LU	6 070	3 035	0.59	0.8%	26	1.00	5.0%	150
DE	700 000	350 000	0.63	0.8%	2 752	1.04	5.7%	20 102
DK	56 870	28 435	0.69	1.1%	305	1.04	5.8%	1 638
SE	114 907	57 454	0.85	2.8%	1 618	1.21	8.8%	5 050
SK	55 033	27 517	0.93	3.9%	1 078	1.42	14.6%	4 015
CZ	109 860	54 930	0.91	3.7%	2 025	1.62	17.9%	9 831
BE	125 014	62 507	1.05	5.7%	3 582	1.69	14.3%	8 965
ES	426 042	213 021	0.85	2.6%	5 450	1.95	21.4%	45 524
RO	183 313	91 657	1.55	11.7%	10 729	1.97	18.3%	16 733
PL	375 160	187 580	1.77	16.6%	31 045	2.64	37.6%	70 563
Sum	2 424 879	1 212 440			61 396			188 251
Other Member States	2 683 487	1 341 744						
Total	5 108 366	2 554 183		**5.1%**	**129 340**		**15.5%**	**396 579**

Conclusion

Based on the 95th percentile of combined exposure to the four phthalates in 2011, RCRs are at or above 1 for children in 13 out of 15 Member States (87%) and in 6 out of 15 Member States in women (40%). Overall, RCRs were at or above 1 at the 95th percentile exposure level in 14 out of 15 Member States (93%) in 2011, and 13 out of 15 Member States (86%) in 2014. In Member States with an RCR = 1 or an RCR < 1 at the 95th percentile exposure level, there was

[85] The SD was calculated as ln(95th p RCR)-ln(GM RCR)/1.64485362695147. Using the NORM.DIST function in Excel the percentile X is calculated where the ln(RCR) = 0 (i.e., where the RCR = 1). The fraction of the population exposed above an RCR of 1 is then 1-X.

still a risk for individuals with the highest exposure levels in the study population in 2014. It can be concluded that there was a risk in all Member States in 2014.

Furthermore, based on the 95th percentile of the DEMOCOPHES biomonitoring estimates, a risk from exposure to individual phthalates was identified. A risk from exposure of children to DBP was identified in Poland, the Czech Republic, the Slovak Republic, and bearing uncertainties in mind, risks from DBP are not likely to be adequately controlled in Sweden and Spain. A risk from exposure of children to DIBP was identified in Poland and risks are not likely to be adequately controlled in Belgium. Furthermore, risks from exposure of mothers to DEHP might not be adequately controlled in Romania. Exposure to BBP does not appear to significantly contribute to the risks.

In several Member States (PL, CZ, ES and SK) the central (median, median and geometric mean) combined exposure tendency indicates that exposure to DEHP, DBP, DIBP and BBP leads to a significant body burden (RCR>0.5). The central tendency of combined exposure in Polish children indicates that there is a risk in a very large proportion of the population.

Approximately 5% of new born boys (130 000) were at risk through in utero exposure in 2014 and about 15.5% boys (400 000) were at risk from direct exposure in 2014.

Evaluation of the uncertainties to the RCRs generally point to possible underestimation of the RCRs.

It can be concluded that a risk has been identified that is not adequately controlled and needs to be addressed.

B.9.2. Risk characterisation based on exposure modelling

B.9.2.1. Indoor environment

The estimated exposure values for the four phthalates in indoor air and dust is repeated in the table below.

Table B59 Dust ingestion and - for DEHP - inhalation of phthalates via air and particles

µg/kg bw/day	Infants		Children		Women	
	Typical case	**Reasonable worst case**	**Typical case**	**Reasonable worst case**	**Typical case**	**Reasonable worst case**
DEHP	4.22	21.85	0.93	5.51	0.48	2.52
DBP	0.28	1.47	0.04	0.27	0.02	0,12
DIBP	0.27	1.41	0.04	0.25	0.02	0.11
BBP	0.08	0.42	0.01	0.08	0.01	0.03

On this basis the RCR values for dust and indoor air are estimated in Table B60.

Table B60 RCR contribution from dust and - for DEHP - inhalation of phthalates via air and particles

	Infants		Children		Women	
	Typical case	**Reasonable worst case**	**Typical case**	**Reasonable worst case**	**Typical case**	**Reasonable worst case**
DEHP	0.12	0.64	0.03	0.16	0.01	0.07
DBP	0.04	0.22	0.006	0.04	0.003	0.01
DIBP	0.03	0.17	0.005	0.03	0.003	0.02
BBP	0.00	0.001	0.00	0.00	0.00	0.00
Combined	0.20	1.03	0.04	0.23	0.02	0.11
Combined MC	0.20	0.76	0.04	0.18	0.02	0.08

From Table B60 it can be seen that DEHP is responsible for the major contribution to the total RCR for indoor environment in both typical and reasonable worst case scenario. One of the reasons for the relatively low RCR values for BBP compared to DEHP, DBP and DIBP is the relative higher DNEL value for BBP. Dust contains the largest amounts of phthalates compared to the concentrations in the air (gas phase and particles in air). The total RCRs show that RCR values are highest for infants and lowest for women. This is due to the assumption that infants have a higher intake of dust compared to children and women. The cumulated RCR values are 0.20, 0.04 and 0.02 for infants, children and women, respectively in the typical case and 0.76, 0.18 and 0.08 for the reasonable worst case scenario.

In a Monte Carlo simulation in relation to the individual phthalates, where it is not assumed that the 95th percentile will be reached for all four substances, the RCR values for the three age groups are 0.76, 0.18 and 0.08, respectively.

Langer et al. (2010) found no correlation among the different phthalates in dust in either homes or day care centres. However, Langer et al. (2014) found significant associations among the individual phthalate metabolites measured in the urine samples. This might indicate that the “real” 95% would be between the simple combined RCR and the Monte Carlo combined RCR.

B.9.2.2. Food

The estimated exposure values for the four phthalates from food are given in Table B61 below together with the RCR values for each phthalate.

RCR values are based on two studies on exposure from food; one from Germany based on analysis of duplicate diet samples, measuring the exposure for infants and another from Belgium based on analysis of 388 food samples and data from a food consumption data base, calculating the exposure for children and adults. It is expected that the exposure based on analysis of duplicate diet samples, would give a more precise measure of the exposure compared to the calculation based on analyses for food samples combined with data from food

consumption data. Exposure from all age groups leads to RCR values below 1 (Table B61), except from the combined reasonable worst case for infants where the RCR from food alone is close to 1.3 assuming non-correlation between contribution for the different phthalates. The RCR values are, depending on the scenario, but data show that an exposure from food is still expected even with the new limits for phthalates in food contact materials. The highest contribution to the RCR value is for children and infants from DEHP, while for infants DIBP contributes most. The contribution to the RCR from BBP is non-existing. This is well in line with other findings referenced below.

Several studies have shown that food and the way food is stored has an impact on the exposure for phthalates. Rudel et al. (2011) took urine samples from 5 families in US in 2010 over a period where the families ate normal food. The families were then given food that were not canned, packed in plastic and where preparation techniques avoided contact with plastic and after this period the families returned to their normal food again. This showed a significant decrease in DEHP metabolites during the period with food that was not canned or packed in plastic. This shows that food and food contact materials are an important source to the phthalate exposure. Other studies comparing the intake of food with biomonitoring showed that the main contributor to DEHP exposure seems to be food, whereas there must be other sources to DBP, DIBP and BBP. Furthermore several market surveillance activities show that phthalates are still found in FCM and food simulants.

Gärtner et al. (2009) have analysed the migration of phthalates in infant food packed in recycled paperboard, and this study shows that phthalates and especially DIBP can still be found in infant food collected in the beginning of 2009 and several other market surveillance activities show non-compliant food contact materials.

Table B61 Exposure estimates from food

µg/kg bw/day	Infants		Children		Women	
	Typical	RWC	Typical	RWC	Typical	RWC
DEHP	4.66	7.09	3.5	5.38	1.49	2.86
DBP	0.7	1.24	0.20	0.30	0.08	0.16
DIBP	1.03	9.02	0.42	0.64	0.14	0.28
BBP	0.15	0.24	0.12	0.21	0.054	0.12

Table B62 RCR from exposure via food

	Infants		Children		Women	
	Typical	RWC	Typical	RWC	Typical	RWC
DEHP	0.14	0.21	0.10	0.16	0.04	0.08
DBP	0.10	0.19	0.03	0.04	0.01	0.02
DIBP	0.12	1.09	0.05	0.08	0.02	0.03
BBP	0.00	0.00	0.00	0.00	0.00	0.00
combined	0.37	1.48	0.18	0.28	0.07	0.14
Combined MC*		1.34		0.25		0.12

*Assuming no correlation between contribution from phthalates. However, both Frederiksen et al. (2011) and Becker et al. (2009) observed a significant positive correlation between samples of DBP, BBP, DEHP and DINP metabolites. This might indicate that the “real” 95% would be between the simple combined RCR and the Monte Carlo cumulative RCR.

B.9.2.3. Contact with articles

The RCR-values related to direct exposure to articles are estimated for both a typical scenario representing the average and a reasonable worst case scenario. In addition to the general scenarios, RCR-values for specific articles are estimated based on migration rates measured from the specific articles.

Table B41 and Table B46 present the exposure estimates.

Table B63 Exposure from oral and dermal contact with articles

µg/kg bw/day	Infants			Children			Women		
	Typical case	RWC	RWC-MC	Typical case	RWC	RWC-MC	Typical case	RWC	RWC-MC
DEHP	3.49	27.32	27.67	2.39	17.91	17.26	2.12	7.63	12.06
DBP	1.20	9.22	6.48	0.83	6.22	4.39	0.74	2.65	3.17
DIBP	1.06	8.16	6.74	0.73	5.50	4.49	0.65	2.34	3.09
BBP	0.31	2.43	1.75	0.21	1.59	1.13	0.19	0.68	0.77

The total RCR values for the three different age groups for both dermal and oral exposure from articles is shown in Table B64 below.

Table B64 RCR related to exposure from contact with articles

	Infants			Children			Women		
	Typical case	RWC	RWC-MC	Typical case	RWC	RWC-MC	Typical case	RWC	RWC-MC
DEHP	0.10	0.80	0.81	0.07	0.53	0.51	0.06	0.22	0.35
DBP	0.18	1.38	0.97	0.12	0.93	0.65	0.11	0.40	0.47
DIBP	0.13	0.98	0.81	0.09	0.66	0.54	0.08	0.28	0.37
BBP	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cumulative	0.41	3.17	2.60	0.28	2.12	1.37	0.25	0.90	0.96
Combined MC			1.69			1.11			0.81

It can be noted that the combined RCRs arising from exposure from contact with articles in the typical case are 0.4 for infants, 0.3 for children and 0.3 for women. In the reasonable worst case scenario assuming non-correlation between high exposures between the four phthalates the RCRs are 1.7 for infants, 1.1 for children and 0.8 for women.

DEHP, DBP and DIBP each contribute to the combined risk from the four phthalates, while BBP does not contribute to the RCR.

The RCR will depend on the use of articles and the migration of the phthalates. A reason for the relatively high RCR values for DBP and DIBP is due to the high migration rates for these phthalates together with low DNELs.

For infants the risk is both related to oral and dermal exposure, However, the dermal exposure is dominating the RCR value, as in the reasonable worst case scenario more than 93% of the cumulative RCR relates thereto, while the oral exposure only accounts for about 7%. For the typical case scenario the oral exposure accounts for less than 1%. This is due to the long dermal exposure time compared to the short oral exposure time.

For children and women only dermal contact with articles are envisaged. Also for children and women DBP contributes most to the RCR, while the contribution from BBP is almost non existing. This is due to the high DNEL values for BBP compared to the DNEL values for DEHP, DIBP and DBP.

B.9.2.4. Risk characterisation of combined exposure to the four phthalates from all sources

Typical case scenario

Table B65 presents the RCRs for the typical case modelling exposure estimates for food, the indoor environment and contact with articles and the range of GM of biomonitoring exposure estimates from different countries. These RCRs are combined to obtain a total RCRs for aggregated exposure sources and combined exposure to the four phthalates for each of the age groups (deterministic modelling). These RCR values are in line with the RCR values based on biomonitoring data (Table B52 and Table B53). The RCRs for combined exposure to the four phthalates in the typical scenario are 1 for infants but below one for children and women.

Table B65 Overview of RCRs for the modelling exposure estimates for the typical scenario (deterministic modelling) and the range of GM of biomonitoring exposure estimates from different countries

	Infants				**Children**					**Mothers**				
	Indoor	Food	Articles	Total	Indoor	Food	Articles	Total	GM BM	Indoor	Food	Articles	Total	GM BM
DEHP	0.12	0.14	0.10	0.36	0.03	0.10	0.07	0.20	0.04-0.14	0.01	0.04	0.06	0.12	0.03-0.10
DBP	0.04	0.10	0.18	0.33	0.01	0.03	0.12	0.16	0.08-0.46	0.00	0.01	0.11	0.13	0.07-0.30
DIBP	0.03	0.12	0.13	0.29	0.00	0.05	0.09	0.14	0.08-0.36	0.00	0.02	0.08	0.10	0.05-0.19
BBP	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00-0.00	0.00	0.00	0.00	0.00	0.00-0.00
Total	**0.20**	**0.37**	**0.41**	**0.98**	**0.04**	**0.18**	**0.28**	**0.50**	**0.23-0.89**	**0.02**	**0.07**	**0.25**	**0.34**	**0.16-0.49**

To show to which extent the different sources contributes to the exposure, Table B66 below summarises RCR values for indoor environment, food and contact with articles in the typical scenario.

Table B66 demonstrates that all the three main sources contribute significantly to the exposure.

The RCR values are based on weighted average values from studies of medians and 95th percentiles for indoor environment and food. This means that some data show higher exposure of phthalates, and some groups of the population are expected to have a higher exposure than estimated here, resulting in higher RCR values. As demonstrated below, the use of specific articles could result in relatively high exposure of phthalates and in higher total RCR values possibly leading to a risk.

Table B66 Relative contribution from sources and phthalates

	Typical case scenario, share of phthalates					**Typical case scenario, share of sources**			
	Infants								
	indoor	food	articles	Total		indoor	food	articles	Total
DEHP	62%	37%	25%	37%		34%	38%	28%	100%
DBP	21%	29%	44%	33%		13%	32%	55%	100%
DIBP	17%	34%	31%	29%		12%	44%	45%	100%
BBP	0%	0%	0%	0%		21%	0%	79%	100%
Total	100%	100%	100%	100%		20%	37%	42%	100%
	Children								
	indoor	food	articles	Total		indoor	food	articles	Total
DEHP	71%	56%	25%	40%		14%	51%	35%	100%
DBP	16%	16%	44%	32%		4%	19%	77%	100%
DIBP	12%	27%	31%	28%		3%	35%	62%	100%
BBP	0%	0%	0%	0%		3%	34%	62%	100%
Total	100%	100%	100%	100%		8%	36%	56%	100%
	Women								
	indoor	food	articles	Total		indoor	food	articles	Total
DEHP	71%	60%	25%	35%		12%	36%	52%	100%
DBP	17%	17%	44%	36%		3%	10%	88%	100%
DIBP	13%	24%	31%	29%		3%	18%	80%	100%
BBP	0%	0%	0%	0%		3%	22%	76%	100%
Total	100%	100%	100%	100%		6%	21%	73%	100%

Reasonable worst case scenario

Table B67, presents the RCRs for the reasonable worst case modelling exposure estimates for food, the indoor environment and contact with articles and the range of 95th percentile of biomonitoring exposure estimates from different countries. These RCRs are combined to obtain a total RCRs for aggregated exposure sources and combined exposure to the four phthalates for each of the age groups by using Monte Carlo simulations. The RCRs for combined exposure to the four phthalates in the reasonable worst case scenario are 2.7 for infants, 1.3 for children and 0.9 for mothers.

Table B67 RCRs for the reasonable worst case modelling exposure estimates and the range of the 95th percentile of biomonitoring exposure estimates from different countries

	Indoor	**Food**	**Articles (MC)**	**Total**	**Aggregated RCR (MC)**	**95th percentile biomonitoring**	**Combined RCR (MC)**
Infants							
DEHP	0.64	0.21	0.81	1.67	1.26	NA	**2.63**
DBP	0.22	0.19	0.97	1.37	1.14	NA	
DIBP	0.17	1.09	0.81	2.07	1.47	NA	
BBP	0.00	0.00	0.00	0.00	0.00	NA	
Total	1.03	1.48	2.60	5.11		NA	
Combined RCR per source (MC)	0.76	1.34	1.69				
Combined RCR (MC)	2.69						
Children							
DEHP	0.16	0.16	0.51	0.83	0.66	0.16-0.88	**1.34**
DBP	0.04	0.04	0.65	0.74	0.69	0.28-1.21	
DIBP	0.03	0.08	0.54	0.65	0.60	0.25-1.21	
BBP	0.00	0.00	0.00	0.00	0.00	0.00-0.01	
Total	0.23	0.28	1.71	2.22		0.75-2.94	
Combined RCR per source (MC)	0.18	0.25	1.11				
Combined RCR (MC)	1.34						
Mothers							
DEHP	0.07	0.08	0.35	0.51	0.42	0.08-1.02	**0.90**
DBP	0.02	0.02	0.47	0.51	0.49	0.15-0.89	
DIBP	0.01	0.03	0.37	0.42	0.40	0.27-0.72	
BBP	0.00	0.00	0.00	0.00	0.00	0.00-0.00	
Total	0.11	0.14	1.20	1.45		0.50-1.98	
Combined RCR per source (MC)	0.08	0.12	0.81				
Combined RCR (MC)	0.91						

Total = simple sum of RCRs, i.e., not using Monte Carlo estimations

NA = Not available

MC = Monte Carlo

Infants have the highest RCR and both DBP, BBP and DEHP contributes all significantly to the risk. The exposure to DEHP is dominated by the contribution from contact with articles and indoor environment, while for DIBP food impose a significant risk – above 1. BBP does only contribute very limited to the overall risk.

Children have a RCR value half of the one calculated for infants, but still significantly above 1. Contact with articles is the main contributor – in average about 77% of the total RCR. The combined RCR values for the different phthalates as well as the combined RCR are in the range found in biomonitoring (Table B52).

For women, the reasonable worst case RCR is just below 1 (0.9). Also here contact with articles seems to be a major contributor. It is noted that this is not in line with the common understanding that food is the main source.

The modelled exposure estimates show that in almost all scenarios and for all of the phthalates contact with articles is the main contributor to the exposure. This is not supported by biomonitoring data from Fromme et al (2013a), Koch et al. (2013), Wittassek et al. (2011), Rudel et al. (2011) and UBA (2011).

As discussed in section B.8.3.2, from studies it can be concluded that 75% of the intake of DEHP is attributable to food (incl. drinks), whereas for DBP, DIBP and BBP it is assumed that 25% is attributable to food. The exposure modelling suggests that the contribution to exposure to DEHP from food is only 38%, 51% and 36% in infants, children and adults respectively. For DBP, the modelling suggests a contribution of 32%, 19% and 10% in infants, children and adults respectively; for DIBP 44%, 35% and 18% in infants, children and adults respectively; and for BBP 0% (no recent data available), 34% and 22% in infants, children and adults respectively.

For DEHP, the modelling seems to underestimate the contribution via food relative to other exposure sources. For the other phthalates, considering all uncertainties, the modelling estimates of the proportion of food to overall exposure sources is reasonably similar to the estimates above that were based on ‘fasting-urinary biomonitoring’ or ‘duplicated diet-urinary biomonitoring’ studies.

The uncertainties to estimates of the risk from contact with articles relate to the assumptions made on migration rates, exposure time, use and the dermal and oral contact area. The RCRs could either be over- or underestimated.

Phthalates in dust and indoor air originate from phthalate containing articles in the homes. Single articles or article applications may be responsible for high exposure to phthalates.

In spite of the limit of phthalates in food contact materials, new data from after the entry into force of the limits, show exposure from food. The exposure from food could come from non-compliant food contact materials, food contact materials migrating phthalates up to the migration limits or as a result of phthalates in the environment. Several studies show that the exposure still can take place from food contact material.

Table B68 Modelled RCR-values

Source		Typical	Best (low)	Worst case	Ref in report
		RCR value			
Indoor environment, infants	DEHP	0.12	0.062	0.64	Table B60
	DBP	0.04	0.021	0.22	
	DIBP	0.03	0.016	0.17	
	BBP	0.00	0.000	0.001	
Indoor environment, children	DEHP	0.03	0.014	0.16	
	DBP	0.006	0.003	0.04	
	DIBP	0.005	0.002	0.03	
	BBP	0.00	0.000	0.00	
Indoor environment, women	DEHP	0.01	0.007	0.07	
	DBP	0.003	0.002	0.02	
	DIBP	0.003	0.001	0.02	
	BBP	0.00	0.000	0.00	
Food, infants	DEHP	0.14	0.07	0.21	Table B61
	DBP	0.10	0.05	0.19	
	DIBP	0.12	0.06	1.09	
	BBP	0.00	0.00	0.00	
Food children	DEHP	0.10	0.05	0.16	
	DBP	0.03	0.02	0.04	
	DIBP	0.05	0.03	0.08	
	BBP	0.00	0.00	0.00	
Food, women	DEHP	0.04	0.02	0.08	
	DBP	0.01	0.01	0.02	
	DIBP	0.02	0.01	0.03	
	BBP	0.00	0.00	0.00	
Articles, infants	DEHP	0.10	0.01	0.81	Table B64
	DBP	0.18	0.02	0.97	
	DIBP	0.13	0.01	0.81	
	BBP	0.00	0.00	0.00	
Articles, children	DEHP	0.07	0.04	0.51	
	DBP	0.12	0.06	0.65	
	DIBP	0.09	0.05	0.54	
	BBP	0.00	0.00	0.00	

Articles, women	DEHP	0.06	0.03	0.35	
	DBP	0.11	0.06	0.47	
	DIBP	0.08	004	0.37	
	BBP	0.00	0.00	0.00	

B.9.2.5. RCR values for exposure from contact with specific articles

The estimated RCR values for exposure from contact with specific articles can be found in Table B69 and Table B70.

Table B69 RCR values for oral exposure of children to DEHP in erasers.

	RCR
Eraser mouthing	0.45
Eraser intake 8 mg	5.03

Table B70 RCR values for dermal exposure to specific articles

Article	**DEHP**			**DIBP**			**DBP**		
	infants	**children**	**women**	**infants**	**children**	**women**	**infants**	**children**	**women**
Plastic sandals median exposure	0.026	0.053	0.020			0.009/0.448			
Plastic sandals worst case exposure	0.103		0.041	0.429		0.314		0.58	0.82
Sex toys*			0/0.026						

* The first value is based on the migration to artificial sweat and the second value is based on the migration to artificial sweat + oil based lubricant (worst case scenario).

From Table B69 and Table B70 it can be seen that exposure from contact with specific articles might result in high contributions to the total RCR. The contribution to the RCR will depend on the article, the composition of the PVC used in the material and the use of the article. Mouthing and intake of eraser will result in high RCR-values. RCR from mouthing erasers is 0.45 (mouthing an eraser 1 hour per day). If it is assumed that the eraser is only mouthed 10 minutes per day the RCR contribution from erasers would be 0.08. Apart from mouthing, regular oral intake of 8 mg of eraser, corresponding to one sesame seed, will lead to a risk. The use of plastic sandals combined with sun screen applied to the skin will also give a high contribution to the total exposure.

The article specific scenarios show that the use of some single articles migrating high amounts of phthalates will contribute highly to the RCR, as for example plastic sandals and erasers. These are examples of the use of articles that for some individuals would lead to relatively high RCR values.

RCR values can also change if for example articles are used in a realistic but not intended way. This could be by eating food directly from a dinner mat, where a risk assessment showed that

eating a piece of cheese from a dinner mat after one hour could lead to a RCR of 0.94 for a child of 10 kg[86].

B.9.3. Uncertainties in the RCR calculations

Modelling

An overview of the uncertainties is presented in Table B71. Overall, the uncertainties suggest that the exposure modelling may underestimate RCRs.

The exposure estimates from modelling and biomonitoring are in the same order of magnitude, indicating that the they are trustworthy. The biomonitoring data suggest that certain groups in the population are exposed at higher levels than estimated by the modelling.

The modelled estimates suggests contact with articles is the main source of exposure, while biomonitoring studies indicate that food is the main source of exposure to DEHP.

[86] The scenario was developed by the Danish food Agency on a request from the Danish EPA in relation to case where a dinner mat containing a high amount of DEHP was found.

Table B71 Overview of sources of uncertainty in the exposure modelling and influence on RCRs for the reasonable worst case.

Source	Description	Effect on RCR
Probabilistic assessment	The Monte Carlo simulations assumed that the contributions from food, indoor environment and contact with articles are not correlated, nor are the contributions from the four phthalates assumed to be correlated with each other. However, articles leading to dermal contact also contribute to exposure via the indoor environment. Similarly biomonitoring shows consistently that individuals which are highly exposed to one phthalate are also exposed to high levels of other phthalates.	↑
Deterministic assessment	Summation of 95th percentiles of several phthalates may overestimate RCRs.	↓
Body weight	The bodyweight used in the risk assessment follows the ECHA guidance R15, 2010. The updated Guidance R15, 2016 does not contain default values on body weight. In the Guidance for biocides (Biocides Human Health Exposure Methodology 2015) recommended values for risk assessment is 8 kg for infants, 23.9 for children and 60 kg for adult women. However, these values are based on 25th percentiles and not on the mean or median. If those values are used, the resulting risk arising from articles and dust (see Section 9, Table B51) would be 15% (infants) and 33% (children) higher. For adult women (and thereby newborns) the 60 kg might be a too low estimate if used as a median. The mean/median might be about 10 % higher (in a Dutch study the median weight is 70.6 kg, while the 25th-weight is 64.1 kg). Hence, the resulting risk for women might be about 10 % lower than indicated in Table B51, while the values for RCR contribution from articles and dust given in Table B51 for women would represent the traditional more cautious risk assessment approach.	↑ (infants and children)
Migration rate	Average migration rates for the four phthalates are used to calculate the reasonable worst case exposure and RCR from direct contact with articles. Migration rates referenced in the literature are varying from very low to high. Studies have shown that migration rates of phthalates are highly dependent on the analytical methods, and the migration rates determined in reports from the Danish EPA in general seem to be very low.	↑
Mouthing time (infants only)	As a reasonable worst case scenario a mouthing time of 30 min/day is chosen. It was assumed that 25 % of the overall mouthing time is used to mouth articles not being toys and childcare articles, and that of these articles all contain one of the four phthalates. It is also considered that a child's favourite mouthing object might be plasticised with one or more of the four phthalates. Mouthing of non-compliant toys may not be covered by the mouthing times assumed here but naturally contributes to the body burden. There are regularly RAPEX notification on non-compliant toys containing phthalates	↑↓
Assumption of intake of dust	The intake of dust is assumed to be 100 mg/day for infants and 50 mg/day for children and adults. Data from recent publications indicates that the typical case intake of dust might be lower, but for the reasonable worst case these assumptions seem appropriate.	-
Exposure from inhalation (indoor air and particles in indoor air)	The exposure to phthalates from indoor air are based on simulations. The simulations are based on data from analysed articles with relatively low concentrations of DBP,	↑

	DIBP and BBP. Articles with higher concentrations of these phthalates have been found and are therefore also expected in many homes. The use of analytical data with low concentrations of DBP, DIBP and BBP will underestimate the exposure of these phthalates from air.	
Exposure from inhalation (indoor air and particles in indoor air)	For infants, children and women respiration rates of 7, 14 and 18 m3/day are used. Reference is made to REACH Guideline R15, Annex R15-5, Table R15-1 (2010), based on AUH, 1995. The updated guidance R15 from 2016 does not contain an Appendix including defaults values. If other updated defaults, such as those given in the US-EPA EFH (2011) and the Biocides Exposure guidance (2015) (for long term inhalation mean respiration rates 5.4 m3/day for infants, 12 m3/day for children (6-12 years), and 16 m3/day for adults) are used, the contribution from inhalation of dust and air would be between 30% (children) and 13% (adults) lower. However, the contribution to the RCR from this exposure route is very limited (table 51) and a correction would only change the RCR by approximately 1% (6.6% of DEHP in indoor air).	↓
Exposure via food	Only two recent studies were available from two Member States (Belgium and Germany). There are therefore uncertainties regarding the representativity of these studies for these countries. Furthermore, the studies may not sufficiently capture the regional differences in exposure via food in the EU28. Especially for DEHP, the modelling seems to underestimate the contribution via food relative to other exposure sources.	↑

An arrow pointing upwards (↑) indicates that uncertainties suggest RCRs may be higher and thus may be underestimated. An arrow pointing downwards (↓) suggest RCRs may be lower and thus may be overestimated. An uncertainty with minimal impact on the RCRs is indicated with a dash (-). Where arrows are pointing in both direction, this indicates that uncertainties may have a significant impact on the RCRs, but it is not possible to evaluate whether the parameter leads to under- or overestimation of the RCRs.

Biomonitoring

A number of uncertainties and their implications for the RCR are listed in Table B72. Overall, uncertainties point in the direction of a possible underestimation of the risks.

Table B72 Overview of sources of uncertainty in the phthalate risk assessment based on biomonitoring data and influence on RCRs.

Source	**Description**	**Effect on RCR**
Hazard		
DNEL DEHP	Alternate (lower) DNELs of 0.007 and 0.008 mg/kg bw/day may be derived from Christiansen et al. (2010) and Andrade et al. (2006) (4.5 times lower).	↑
DNEL DEHP	Endpoints that appeared to be the most sensitive for DBP have not been investigated for DEHP. In view of equipotency for effects on testosterone production as compared to DBP, the PoD could be about 5 times lower.	↑
DNEL BBP	BBP appears to have comparable potency to DEHP and DBP on foetal testosterone production. It may be speculated that further studies on effects of BBP on endocrine sensitive endpoints would reveal effects at	↑

	lower doses than 50 mg/kg bw/day, potentially leading to a lower DNEL (if similar to DEHP the DNEL for BBP would be a factor 10 lower)	
DNEL DIBP	In the absence of conclusive experimental data, read-across from DBP has been performed to DIBP. The experimental evidence for concluding that DIBP is of similar anti-androgenic potency is considered robust, but the assumption of potency difference (25%) is uncertain.	-
DNELs for children	The DNELs are relevant for both pregnant women and for children, albeit it is possible that the DNELs for children would be higher.	↓
Species differences	There are indications of species differences in metabolism and possibly in effects on foetal steroidogenesis, but the evidence is insufficient to deviate from the default assumption that humans are more sensitive than the test species.	↓
Effects on the immune system	A number of experimental and epidemiological studies provide moderate to strong indications for effects on the immune system. Some of these studies indicate that reproductive toxicity may not be the most sensitive endpoint and that the selected DNELs may not be sufficiently protective against these immune effects.	↑
Effects on the metabolic system and neurological development	A number of experimental and epidemiological studies suggested possible effects on the metabolic system and neurological development. It is not clear from the data whether the selected DNELs based on reproductive toxicity are sufficiently protective against these other effects.	?
Effects on the immune system	A number of experimental and epidemiological studies have suggested possible effects on the immune system, the metabolic system and neurological development. Some of these studies indicate that reproductive toxicity may not be the most sensitive endpoint for the effects and that the selected DNELs may not be sufficiently protective against these other effects	↑
Threshold	If it is decided that the four phthalates give rise to equivalent level of concern due to their endocrine disrupting properties for human health, it has to be determined whether a threshold for effects can be demonstrated if any applications for authorisation would be submitted in the future (European Commission 2014). The existence of a threshold has not yet been assessed and documented for DEHP, DBP, DIBP and BBP.	↑
Exposure		
Data availability	There are uncertainties to the estimates as a result of data availability issues. The effect appears to be minimal based on a comparison of our estimates and published estimates for DK.	-
Sampling approach	There is both a diurnal and a day to day variation in the quantities of metabolites excreted in urine in response to the variation in intakes of phthalates over a 24 hour period. As a result of this variability, a single spot urine	↑↓

	sample may not be representative for the mean daily concentration.	
Creatinine based method	When using volume based method of intake calculation from urinary biomonitoring data higher exposure estimates may be obtained (possibly by a factor of 2).	↑
Use of 95th percentile exposure and summation of 95th percentiles of several phthalates	The exposure estimates are derived from a fairly limited number of samples per country (around 120). This results in relatively high uncertainties to whether the actual 95th percentile exposure in the entire population is lower or higher: the sample might not be representative for highly exposed sub-populations. Even a short elevated exposure level may be sufficient to cause adverse effects from exposure within the critical windows of exposure. On the other hand, maxima may arise from analytical and methodological errors or might result from non-representative exposure situations. Furthermore, adding RCRs based on 95th percentiles of several phthalates may lead to some overestimation of the RCRs, although consistent evidence indicates that it is not uncommon that individuals are exposed to high levels of more than one phthalate simultaneously.	↑↓
Selection of population	Patients with haemodialysis were not admissible to the DEMOCOPHES study (FPS 2013) and thus it is highly unlikely that any patients with recent (within a day) exposure from medical devices would have been included in the study population. These specific situations may lead to exposure that exceeds the daily intake in the general population by several orders of magnitude (Koch and Angerer 2012). Thus, for those children and women that regularly undergo medical treatment with DEHP containing medical devices, the risk as estimated in the current risk assessment is likely to be underestimated.	↑
Infants	The children in the study population of DEMOCOPHES were 6-11 years old. Younger children appear to be exposed at higher levels to the four phthalates and thus the estimates may underestimate exposure of younger children. In addition, medical devices may contribute to exposure to DEHP, for example in preterm neonates (SCENIHR 2016). Since the population in biomonitoring studies such as DEMOCOPHES does not include neonates, there may be additional risks from phthalates to infants not accounted for in the current risk assessment. Modelling suggests that the RCR for infants from combined exposure to the four phthalates may be twice that for children.	↑
FUEs used for children	The FUEs used for children are for adults and may result in underestimation of exposure to DBP, DIBP and BBP.	↑
Estimates for specific Member States	The RCRs for combined exposure are underestimated for Slovenia since no measurement of DIBP metabolites was available. For the same reasons, the RCRs for the Slovak Republic, Sweden, Czech Republic and Hungary may also	↑

	be underestimated, although potential issues with chromatic separation may have compensated for the lack of a measurement value for DIBP. Due to the small sample size (n=21), the data from the UK is not considered representative for the exposure in the UK and might be underestimated.	
Other considerations		
Other anti-androgenic substances may contribute significantly to the total risk	The combined risk assessment considers only DEHP, DBP, DIBP and BBP, but other substances may contribute to mixture effects on male reproductive development. Several substances are evaluated to be able to cause anti-androgenic effects. Exposure to other substances affecting male reproductive development can contribute significantly to the total risk. Therefore, the combined risk assessment of DEHP, DBP, DIBP and BBP alone is likely to be an underestimation of the risk for mixture effects on male reproductive development. Examples are other anti-androgenic phthalates such as DINP, DnHP, DIHepP, DnHepP (Health Canada 2015; ECHA 2013a) and other substances e.g., Vinclozolin, Prochloraz, Procymidone and p,p'-DDE (Kortenkamp and Faust 2010).	↑

An arrow pointing upwards (↑) indicates that uncertainties suggest RCRs may be higher and thus may be underestimated. An arrow pointing downwards (↓) suggest RCRs may be lower and thus may be overestimated. An uncertainty with minimal impact on the RCRs is indicated with a dash (-). Where arrows are pointing in both direction, this indicates that uncertainties may have a significant impact on the RCRs, but it is not possible to evaluate whether the parameter leads to under- or overestimation of the RCRs.

Sensitivity assessment to effect selection of DNELs on RCRs

Uncertainties to the selected DNELs are discussed in section B.4.5.1. It may be put forward that the differences in the N(L)OAELs between the phthalates may be chiefly the result of experimental differences. Effects on the male mammary gland and delayed germ cell development in Lee et al. (2004) were the most sensitive effects seen with DBP. However, these endpoints have not been studied with DIBP, DEHP and BBP. Mechanistic evidence suggests equipotent or similar anti-androgenic potencies of the four phthalates (equipotent reduction in foetal testosterone production in e.g. Furr et al. 2014; Howdeshell et al. 2008; Hannas et al. 2011).

On request by RAC, a sensitivity scenario was constructed to show the effect on the RCRs when it is assumed that the DNELs for all four phthalates are equal to the DNEL of DBP (6.7 µg/kg bw/day). The results are presented in Table B73, Table B74 and Table B75. In comparison to the RCRs calculated for the 95th percentile combined exposure to the four phthalates as projected to 2014 (Table B52 and Table B54), the RCRs are about double when it is assumed that all four phthalates have the same DNEL.

In this sensitivity scenario, RCRs were >1 in all Member States based on the 95th percentile of combined exposure to the four phthalates in 2014. In several Member States (PL, ES, SK and CZ) RCRs based on the geometric mean combined exposure were >1 for children.

Approximately 25% of new born boys (640 000) were at risk through in utero exposure in 2014 and about 47% boys (1 200 000) were at risk from direct exposure in 2014.

Table B73 RCRs for exposure to the four phthalates as estimated from 95th percentile urinary biomonitoring exposure levels from DEMOCOPHES data projected to 2014, assuming that the DNELs for all four phthalates are equal to 6.7 µg/kg bw/day.

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
UK	21	0.4	0.1	0.0	0.3	0.8	21	0.8	0.3	0.1	0.3	1.5
SI	120	0.9	0.4	0.1	*NA*	1.3	120	1.0	0.4	0.1	*NA*	1.5
LU	60	0.7	0.2	0.1	0.3	1.3	60	0.6	0.2	0.1	0.8	1.7
CH	117	0.9	0.2	0.1	0.2	1.4	119	1.1	0.3	0.1	0.3	1.8
CY	59	2.2	0.2	0.0	0.5	2.9	60	1.2	0.2	0.1	0.5	1.9
DE	116	0.6	0.3	0.1	0.3	1.2	120	1.1	0.5	0.1	0.4	2.1
DK	143	0.8	0.2	0.1	0.4	1.5	142	1.1	0.3	0.1	0.7	2.2
PT	117	1.7	0.2	0.1	0.3	2.3	116	1.3	0.3	0.1	0.5	2.2
IE	120	1.0	0.2	0.1	0.4	1.7	120	1.5	0.2	0.1	0.5	2.4
HU	115	1.3	0.4	0.1	NA	1.8	117	1.9	0.6	0.1	NA	2.6
SE	96	0.9	0.7	0.3	NA	1.8	97	1.7	0.9	0.4	NA	2.9
SK	125	1.1	0.7	0.1	NA	1.8	127	2.1	1.0	0.1	NA	3.2
BE	125	0.7	0.4	0.1	0.7	1.9	125	1.8	0.4	0.1	1.2	3.5
CZ	117	1.2	0.7	0.2	NA	2.0	120	2.1	1.2	0.2	NA	3.5
ES	118	1.3	0.3	0.1	0.4	2.1	119	1.8	0.8	0.2	1.0	3.8
PL	119	1.8	0.8	0.1	0.8	3.5	115	2.6	1.0	0.2	1.4	5.2
RO	117	5.1	0.2	0.0	0.4	5.8	119	4.4	0.5	0.1	0.7	5.7

NA = not available

Table B74 Overall RCRs for exposure to the four phthalates as estimated from unweighted creatinine corrected urinary concentrations from DEMOCOPHES data extrapolated from 2011/2012 to 2014, assuming that the DNELs for all four phthalates are equal to 6.7 µg/kg bw/day.

Population		**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
Mothers	Min	0.0	0.0	0.0	0.0	0.0
	GM	0.3	0.1	0.0	0.1	0.6
	P50	0.3	0.1	0.0	0.1	0.5
	P90	0.8	0.2	0.1	0.3	1.4
	P95	1.2	0.3	0.1	0.4	2.0
	Max	18.2	8.9	1.8	1.6	30.4
Children	Min	0.0	0.0	0.0	0.0	0.0
	GM	0.5	0.1	0.0	0.2	0.8
	P50	0.5	0.1	0.0	0.2	0.8
	P90	1.3	0.3	0.1	0.5	2.2
	P95	1.8	0.5	0.2	0.7	3.1
	Max	37.9	3.4	2.3	6.6	50.2

Table B75 RCRs for exposure to the four phthalates as estimated from geometric mean (GM) urinary biomonitoring values projected to 2014, assuming that the DNELs for all four phthalates are equal to 6.7 µg/kg bw/day.

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
CH	117	0.2	0.1	0.0	0.1	0.3	119	0.3	0.1	0.0	0.1	0.5
CY	59	0.2	0.1	0.0	0.2	0.4	60	0.2	0.1	0.0	0.2	0.5
LU	58	0.2	0.1	0.0	0.1	0.3	60	0.2	0.1	0.0	0.1	0.5
SI	120	0.3	0.1	0.0	NA	0.4	120	0.4	0.1	0.0	NA	0.5
UK	21	0.2	0.1	0.0	0.1	0.3	21	0.4	0.1	0.0	0.1	0.6
DE	116	0.2	0.1	0.0	0.1	0.4	120	0.4	0.2	0.0	0.2	0.7
BE	125	0.2	0.1	0.0	0.2	0.5	125	0.3	0.1	0.0	0.2	0.7
PT	117	0.4	0.1	0.0	0.1	0.6	116	0.5	0.1	0.0	0.2	0.8
HU	115	0.3	0.1	0.0	NA	0.5	117	0.5	0.2	0.0	NA	0.8
DK	143	0.3	0.1	0.0	0.2	0.5	142	0.4	0.1	0.0	0.2	0.8
IE	120	0.3	0.1	0.0	0.1	0.5	120	0.5	0.1	0.0	0.2	0.8
SE	96	0.3	0.2	0.1	NA	0.6	97	0.5	0.3	0.1	NA	0.9
RO	117	0.5	0.1	0.0	0.1	0.8	119	0.7	0.1	0.0	0.2	1.0
CZ	117	0.4	0.2	0.0	0.0	0.7	120	0.7	0.4	0.0	0.0	1.1
SK	125	0.4	0.3	0.0	NA	0.6	127	0.7	0.4	0.0	NA	1.1
ES	118	0.5	0.1	0.0	0.2	0.8	119	0.7	0.2	0.1	0.2	1.2
PL	119	0.4	0.2	0.0	0.2	0.8	115	0.7	0.3	0.0	0.4	1.5

NA = not available

Appendix B1: Simulation indoor air using data from products on the Danish market

To get an estimate of the concentrations of the phthalates, DEHP, DIBP, DBP and BBP in indoor air, in realistic rooms furnished with real furniture/materials marketed in Denmark a simulation has been made. The simulation is based on data from tests of consumer products, performed by the Danish EPA (2001). Two types of rooms have been simulated: one children's play room and one bathroom. The data on concentration of phthalates in vinyl flooring and the wall paper is based on tests performed in 2001 (designated OLD) and 2010 (designated NEW). The test data on concentration of phthalates in the furniture/equipment placed in the rooms are based on the tests performed in 2010.

Method

The concentrations of DEHP, DIBP, DBP and BBP in the rooms, are estimated using the model described in Xu (Xu et al. 2009). Due to limitations in available data on model parameters the following simplifications are made on this model:

- only ceiling partitioning is included as a sink.
- only the material/air partition coefficient for DEHP in vinyl flooring is given (from Clausen et al., 2007). The material/air partition coefficients of other phthalates are estimated by a modified version of the default method suggested by Weschler et al., 2008. Weschler et al. suggest to use Raoults law to estimate air concentrations in a boundary layer above the emitting surface:

$$C_{air} = xP\frac{m}{RT} = \frac{C_{subst}}{C_{subst} + C_{rest}} P\frac{m}{RT} \approx \frac{C_{subst}}{C_{rest}} P\frac{m}{RT}$$

where

C_{air}	concentration of substance in a boundary layer above the emitting surface (µg/m^3)
X	mole fraction of substance in material
P	vapor pressure (mmHg)
M	molecular weight of the substance (in kg/mole)
R	R is the universal gas constant (8.314 Pa m^3 mol^{-1} K^{-1} or 0.0821 atmM^{-1} K^{-1})
T	temperature (K)
C_{subs}	concentration of the substance in the product (kg/m^3)
C_{rest}	concentration of the rest of the material in the product (kg/ m^3), all the non-phthalate molecules

From this, the material/air partition coefficient K is found as:

$$K = \frac{Pm}{C_{rest}RT}$$

As the C_{rest} is not exactly known, it is proposed to use the method as a relative method.

The material/air partition coefficient is assumed to scale as:

$$K_1 / K_2 = \frac{P_1 m_1}{P_2 m_2}$$

This equation is used to estimate the partition coefficients for DBP, BBP and DIBP between vinyl flooring and air from the partition coefficient of DEHP.

The value of the material/air partition coefficient for vinyl flooring is used as a surrogate for wallpaper/air, artificial leather/air and plastic/air in absence of relevant values/emission measurements.

The procedure for estimating material/air partition coefficients defined above yields the following values:

Phthalate	**mw (g/mol)**	**P (mmHg)**	**K**
DEHP	391	1.43e-7	**2.3e11***
DIBP	278	2.7e-5	3.1e13
DBP	278	7.5e-4	8.6e15
BBP	312	5.03e-6	6.5e12

*Xu, Y., et.al. 2010

The emission in each compartment (product/material) is proportional to the difference between the concentration in air and the concentration in the material/product. Second, the air concentration is determined by the removal due to ventilation q x C.

So for example, for just floor as a source:

$$\frac{dy}{dt} = -\frac{S_{floor} \times h_m}{V_r} \times (y - \frac{C_{floor}}{K_{floor}}) - q \times y$$

Where

Y	concentration of substance in air
S_{floor}	surface area of the source (flooring)
h_m	mass transfer rate (describes diffusion over a stagnant layer of air above the surface)
Vr	volume of the room
C_{floor}	concentration of the substance in the material (flooring)
K_{floor}	partition coefficient of the substance between material and air
Q	ventilation rate (air changes per hour)

For all the other sources similar terms to the one for flooring $(-\frac{S \times h_m}{V_r} \times (y - \frac{C_{floor}}{K}))$

enter the equations, so that:

$$\frac{dy}{dt} = -ddtC_{fl} - ddtC_{wp} - ddtC_s - ddtC_{bb} - ddtC_m - ddtC_w - q \times y$$

and

$$ddtC_{wp} = \frac{S_{wallpaper} \times h_m}{V_r}(y - \frac{C_{wp}}{K_{wallpaper}}) \equiv G_{wallpaper}(y - \frac{C_{wp}}{K_{wallpaper}})$$ in which

C_{wp}	concentration of substance in wallpaper
$S_{wallpaper}$	surface area if the source (wallpaper)
h_m	mass transfer rate (describes diffusion over a stagnant layer of air above the surface)
Vr	volume of the room
C_{floor}	concentration of the substance in the material (flooring)
$K_{wallpaper}$	partition coefficient of the substance between wall paper material and air
Q	ventilation rate (air changes per hour)
$G_{wallpaper}$	S_{floor} x hm/ Vr

and similar for the compartments chair covered by artificial leather (ddtCs), balance ball (ddtCbb), air mattress (ddtCm) and wall/floor/ceiling (ddtCw).

It is further assumed that all the product/air partition coefficients for the products (K_{floor}, K_{chair}, $K_{wallpaper}$, $K_{balance\ ball}$, $K_{mattress}$) are the same for each substance.

Simulation

The concentrations used in the calculations originate from material analyses performed by Danish EPA. Vinyl flooring and wall paper was analysed both in 2001 (designated OLD) and in 2010 (designated NEW). Other products like e.g. shower curtains were also analysed in 2001, but is not included in the scenarios designated OLD, as shower curtains is assumed to be changed frequently (once a year or every second year). In the old study, analyses were only performed for DEHP, DBP and BBP. Additionally, in the new studies DIBP were also included in the analysis.

Table B76 Concentration of phthalates in consumer products used in the calculations.

Furniture or material / Phthalate	**DEHP mg/kg**	**DBP mg/kg**	**BBP mg/kg**	**DIBP mg/kg**
Old vinyl flooring	150,000	-	900	-
New vinyl flooring	325	-	113	813
Old wall paper	100,000	-	-	-
New wall paper	24	-	-	19
Air mattress	192,000	-	-	-
Chair	391,500	11	-	41
Balance ball	442,000	20.5	-	693
Shower curtain	281,500	63.3	-	91.9

The two rooms that are simulated are a children's play room and a bathroom. The children's play room is sized 4 x 2.5 x 2.75 (L x W x H) = 27.5 m^3. The floor is covered with vinyl flooring and the walls are covered by wall paper, covered with a vinyl layer. It is assumed that the room is furnished with a chair, partly covered with artificial leather, a balance ball and an air mattress. The air exchange rate has been set to 0.2 times an hour (recommendation in ECHA Guidance R15, 2010).

In the other room, a bathroom, sized 2 x 2 x 2.75 (L x W x H) = 11 m^3, the floor is covered by vinyl flooring and the walls with wall paper, covered with a vinyl layer. Furthermore, there is a shower curtain made of vinyl. The air exchange rate has been set to 0.5 times an hour.

The output is presented in the table below.

In the table, the "steady-state" levels in the air, of the different phthalates, related to the different indoor simulations are found.

Table B77 "Steady-state" levels of the four phthalates.

Sources / Phthalate µg/m³	**DEHP**	**days**	**DIBP**	**days**	**DBP**	**days**	**BBP**	**days**	**TOTAL**
CHILDRENS PLAY ROOM Vinyl flooring, wall paper, air mattress, chair and balance ball **NEW** vinyl flooring and wall paper	0.16	150	7e-6	0	8.2e-11	1	3.2e-6	1	**0.16**
CHILDRENS PLAY ROOM Vinyl flooring, wall paper, air mattress, chair and balance ball **OLD** vinyl flooring and wall paper	0.81	150	1e-5	0	8.2e-11	1	4.4e-5	1	**0.81**
BATH ROOM Vinyl flooring, wall paper and shower curtain **NEW** vinyl flooring and wall paper	0.26	150	1e-5	0	1.5e-9	1	1.8e-6	1	**0.26**
BATH ROOM Vinyl flooring, wall paper and shower curtain **OLD** vinyl flooring and wall paper	0.8	150	6e-6	0	1.5e-9	1	2.5e-5	1	**0.8**

It should be stressed that the only difference between the OLD and the NEW scenarios is the concentration of phthalates in the vinyl flooring and in the wall paper. It may be concluded that the higher concentrations of phthalates in the air in the OLD scenarios are only caused by the higher concentrations of phthalates in the OLD vinyl floorings and OLD wall papers. The items, with which the rooms are furnished, are the same in both the OLD and the NEW scenarios

Scenario from EU Risk Assessment Report

As a comparison to the result from the previous simulation described above, a calculation has been made by applying the method developed in the EU Risk Assessment Report on DEHP (EU RAR, 2008). In the calculation described in EU RAR the only phthalate source in the room is DEHP emitted from the vinyl flooring. Based on the results presented in table A.2 above, it is also reasonable only to take DEHP into consideration in the following calculations. Applying the EU RAR method to the data from the children's play room, parameters like the room size, area of emitting sources, air exchange rate etc. is changed according to the following calculations:

At steady state the concentration (C) can be calculated as follows:

$$C = \frac{3{,}600 \times E \times A}{ach\, V}$$

(DEHP sources: wall paper, vinyl flooring, mattress, balance ball and chair)

E = DEHP emission rate (µg/m^2/s)	3 x 10^{-4} *
A = area of the PVC material (m^2)	48
ach = air change rate (air changes/hour)	0.2
V = volume of the room (m^3)	27.5

* (from EU RAR 2008a: Environ corporation (1988) Indoor DEHP Air Concentration Predicted after DEHP Volatilizes from vinyl Products. Prepared for Chemical Manufacturers Association). Volatilisation of DEHP based on an emission rate has been calculated by three different methods (see Environ Corporation, 1988). An emission rate of between 1.8 x 10-4 and 3 x 10-4 µg/m2/s at 25 °C was derived. The highest emission is used as a worst case in this risk assessment.

The floor is covered with vinyl flooring and the walls with wall paper containing PVC. The room is furnished with an air mattress, a balance ball and a chair partly covered with artificial leather containing PVC. The total area covered with PVC is 48 m2 and the room volume is 27.5 m^3. The air exchange rate is 0.2 (recommendation in ECHAs REACH Guidance on information requirements and chemical safety assessment R.15: Consumer Exposure Estimation) DEHP concentration in air is:

$$C = \frac{3{,}600 \times 3\,10^{-4} \times 48}{0.2 \times 27.5} = 9{,}4\ \mu g/m^3$$

This result is 10 fold higher than the concentrations found in the simulations and in the other references. In general, this may be due to the fact that the EU RAR calculation method is rough and based on very few data.

In the model used in our calculations, the emission rate (from the floor and the walls) is determined by:

hm * (Cfl/Kfloor-y1).

Note that the emission rate depends on the concentration in the room air, on time, sinks, material concentration etc., and is not a constant! This is in opposition to the assumption made

in the EU RAR method. In our case the initial value of the emission is approximately 0.0025 µg/m^2/s2 but it quickly drops during emission, when the room air becomes saturated. This saturation is not accounted for in the method described in the RAR, which may very well account for the fact that the calculated concentration level is 10 times higher.

In order to illustrate this, we have simulated the emission rate using our model. The emission rate is plotted on a log scale. It starts at about $10^{-3.5}$ µg/m^2/s2 but drops to less than $10^{-4.5}$ug/m^2/s2 after approximately 140 days, and even further after that.

Appendix B2: Environmental exposure assessment

The environmental releases, fate and occurrence of the phthalates DEHP, DBP and BBP are described in their respective EU Risk Assessment Reports (EU RAR 2008, EU RAR 2004, EU RAR 2007). Predicted no effect concentrations (PNECs) derived under these assessments for the aquatic compartment are reported below and in Annexes 2, 4, 7 and 9 to ECHA (2012a). In addition, the endocrine disrupting hazard properties of DEHP relevant to the environment and wildlife species are summarised in the support document accompanying the ECHA Member State Committee (MSC) opinion that concluded that DEHP meets the criteria for a substance of very high concern under REACH on the basis of "equivalent concern" (ECHA 2014).

Further information on the environmental effects of phthalates, including the four phthalates that are the subject of this restriction proposal, continues to be published in the scientific literature. For example, Martins et al. (2016) describe a negative correlation between increased tissue concentrations of phthalates in juvenile wild salmonids (including DEHP, DBP, BBP) and the expression of genes that play essential roles during immune cell maturation, activation and antibody production. The authors conclude that chronic exposure to phthalates dysregulates normal immune responses, including antibody production, which could lead to greater susceptibility to pathogens. In a recent review article, Mathieu-Denoncourt et al. (2015) highlight developmental and reproductive effects in mammalian and non-mammalian aquatic species exposed to plasticisers, including phthalates.

1. Summary of PNECs for the aquatic compartment

PNECs for DEHP, DBP and BBP in the aquatic compartment have been described in their respective EU Risk Assessment Reports (EU RAR 2008, EU RAR 2004, EU RAR 2007) and are presented in Table B78 alongside several other PNEC values.

Table B78 PNECs for in the aquatic compartment and other PNEC values

Phthalate	$PNEC_{aquatic}$	Other PNECs	Remarks
DEHP	-	$PNEC_{food}$: 16 mg/kg fresh food for fish exposed to DEHP via food only. Therefore, for fish an overall PNEC of >100 mg/kg dry weight for sediment organisms, mainly based on a study on amphibians (frogs) embryos/larvae, has been determined. PNEC for microorganisms in sewage treatment plants ($PNEC_{STP}$) was determined at >200mg/l.	No reliable short or long term studies below the 'apparent' water solubility indicating effects on aquatic organisms exposed to the water only. Hence a $PNEC_{aquat\ c}$ cannot be specified.
DBP	10µg/l	$PNEC_{sediment}$ = 1.2 mg/kg (wet weight) and 3.1 mg/kg (dry weight). $PNEC_{STP}$ = 0.22 mg/l.	Toxicity of DBP to sediment-dwelling organisms calculated by the equilibrium method as no experimental data available to derive the $PNEC_{sediment}$.
BBP	7.5 µg/l	$PNEC_{sediment}$ = 1.72 mg/kg (wet weight) $PNEC_{marine}$ = 0.75 µg/l with $PNEC_{marine\ sediment}$ = 0.172 mg/kg (wet weight).	A $PNEC_{STP}$ could not be derived, as only one study was available, showing no effect on respiratory activity in activated sludge at BBPs solubility limit of 2.8 mg/l.

Source: EU RAR (2008), EU RAR (2004), and EU RAR (2007)

2. Environmental hazard classification

DBP is classified in the category Aquatic Acute 1. BBP is classified in the categories Aquatic Acute 1 and Aquatic Chronic 1.

DEHP and DIBP do not have a harmonised classification for the environment, but the registrations classify these phthalates as Aquatic Acute 1 (DEHP and DIBP), and Aquatic Chronic 1 (DIBP).

In addition, DEHP is a Water Framework Directive (WFD; 2000/60/EC) Priority Hazardous Substance, which requires that discharges, emissions and losses should be ceased or phased out. An environmental quality standard (EQS) of 1.3 µg/L has been established under this Directive, which was based on a mammalian NOAEL value of 4.8 mg/kg bw/d[87]. Therefore, any revision of mammalian DNEL values (as discussed elsewhere in this background document) could prompt a re-evaluation of the WFD EQS, potentially resulting in a lower EQS value.

3. Endocrine disrupting properties

In addition to the environmental hazard classifications, the Member State Committee (MSC) unanimously agreed in December 2014 to identify DEHP as a substance of very high concern under REACH on the basis that it gives rise to an equivalent level of concern due to its endocrine disrupting properties to the human health and the environment, according to Article 57(f) of REACH.

In a recent RAC opinion on a restriction proposal for nonylphenol and nonylphenol ethoxylates, which are endocrine disruptors according to REACH article 57(f), risk assessment was based on a “traditional” aquatic PNEC. However, RAC noted that there was justification for substances with endocrine disrupting properties to be subject to particular scrutiny, principally as there is ongoing debate about how endocrine disrupting effects should be considered for regulatory action. On this basis, RAC considered that it was premature to give an opinion on whether or not it is possible to derive a safe exposure level for the endocrine disrupting effects of nonylphenol and nonylphenol ethoxylates (ECHA 2014b). A similar conclusion could be proposed for DEHP.

4. DEHP

4.1. Predicted environmental concentrations

The EU RAR considered production, industrial and consumer life stages related to the use of DEHP. Local, regional and continental PECs in different environmental compartments, including secondary poisoning, were calculated using EUSES (v1.0) in combination with TGD default emission values or, where available, measured data.

For production sites, local PECs for individual sites were calculated individually based mainly on site-specific data. For downstream uses (formulation/processing) generic local PECs for the

[87] EQS under WFD are derived to protect the most sensitive receptor exposed in or through the aquatic environment. For DEHP this was determined to be predators exposed through their food (secondary poisoning).

different types of uses (e.g. calendaring, extrusion, plastisol spread coating, sealants/adhesives, lacquers/paints, printing inks, and ceramics) were estimated.

Detailed summary tables of the PECs for different environmental compartments at local scale for each of the different use scenarios are presented in section 3.1.3.1 of the EU RAR. A summary of regional and continental PECs, reproduced from Table 3.56 in the EU RAR, is provided in Table B79, below

Table B79 Calculated regional and continental PECs for DEHP from EU RAR

Scenario	Air (mg/m³)	Surface water (µg/l)	Sediment (mg/kg dwt)	Agricultural soil (mg/kg dwt)	Soil pore water / ground water (µg/l)	Natural soil (mg/kg dwt)	Soil pore water (µg/l)	Urban/Ind. Soil (mg/kg dwt)
Regional	7.5 E-6	2.2	33.7	0.07	0.02	0.015	-	3.2
Continental	1.6 E-6	0.3	4	0.006	0.002	0.003	-	0.33

4.2. Measured data

A large number of studies on the occurrence and concentration of DEHP in environmental media, municipal and industrial wastewaters, sewage sludge and biota were summarised in the EU RAR.

In the aquatic compartment, the overall mean of detected concentrations in river waters was calculated to be 1.3 µg/L, whilst in lake water the concentration was lower, with a mean of 0.08 µg/L. DEHP concentrations in surface waters affected by diffuse pollution from industry or urban areas ranged from below the limit of detection to 21 µg/L. An extensive dataset from the Northrhein Westfalen region in Germany (Furtmann, 1993; Alberti et al, 2000) collected over the period 1993-2000 has a mean DEHP concentration of 0.83 µg/L. This concentration, considered to represent a worst-case regional situation, was used alongside the regional concentration of 2.2 µg/L predicted by EUSES in the EU RAR regional scale risk characterisation.

The mean measured concentration of DEHP in freshwater sediments from 58 sites was 5.2 mg/kg, which was used in preference to the EUSES calculation regional sediment PEC of 33.7 mg/kg dwt. In marine sediments associated with industrial and/or urban areas measured concentrations of DEHP were, on average, 1.5 mg/kg dwt.

Measured concentrations in untreated municipal wastewater (influent) from Sweden, Denmark, Norway and Germany varied between 4 and 250 µg/L.

In the UK, average WWTP influent concentrations of DEHP were reported to be 22.4 µg/L (UKWIR, 2004; Rule et al, 2006), although another study reported a 90%ile value in influent of

9 µg/L [88]. UKWIR (2004) also provided evidence that relatively greater concentrations of DEHP are found in wastewater from newer housing stock compared to older stock (Table B80).

Plastic products in roofing and plumbing materials, paints, sealants, adhesives and fillers may all contribute to this release. Dust and vapour deposited on clothing (and possibly also soft furnishings) that are subsequently laundered in washing machines has recently been identified as an additional source of DEHP to wastewater (Saini et al. 2016). Saini et al. (2016) estimate that a typical laundry machine may release about 300 mg of five phthalates (the four substances subject to the restriction proposal plus DINP) per laundry load to wastewater.

Table B80. DEHP concentration in wastewater from different sources.

Source	**DEHP concentration, µg/L**
New housing estates – wastewater	57
Old housing estates – wastewater	9.2
Town centre – wastewater	17.5 – 22.5
Light industrial estates – runoff	1 – 7.5
Housing estates – runoff	29 - 40

Corresponding concentrations in treated wastewater (effluent) varied between 0.07 and 28 µg/L. Recent monitoring data from the UK's Water Industry Research (UKWIR) Chemicals Investigation Programme[89] show that DEHP is widely found in wastewater treatment plant (WWTP) effluent in the UK, with a median concentration of 0.69 µg/L.

Monitoring studies on DEHP in municipal sewage sludge from Sweden, Denmark, Norway, the Netherlands and Germany report concentrations of DEHP up to 661 mg/kg dwt. Two Canadian monitoring studies report concentrations in the range of 33 to 440 mg/kg dwt.

Measured concentrations in a sample of 30 soils from the Netherlands ranged between <0.025 to 0.17 mg/kg. Agricultural soil in Germany fertilised with sewage sludge was reported to have a concentration of 5 mg/kg dwt after 10 years of application, whilst a Danish soil fertilised with sewage sludge for 25 year had a concentration of DEHP in the upper layers of 1 mg/kg dwt six years after application ceased. The terrestrial monitoring data reported in the EU RAR, whilst acknowledged to be limited, was considered to indicate that the regional concentrations predicted by EUSES could be underestimates and that calculated local PECs could be overestimated.

[88] Atkins (2006). Sources and fate of DEHP at wastewater treatment works and the risk of effluents failing the WFD EQS. Technical note. Cited by UK Environment Agency in their response to the public consultation on the restriction proposal.

[89] http://v-scheiner.brunel.ac.uk/bitstream/2438/8867/5/Fulltext.pdf. Further details about this programme can be found at https://www.ukwir.org/site/web/news/news-items/ukwir-chemicals-investigation-programme, and in the UK comments submitted for the nonylphenol ethoxylate textile restriction proposal.

Measured concentrations of DEHP in aquatic arthropods varied between 100 µg/kg (dwt) and 14 400 µg/kg (wet weight). The highest value was measured in the freshwater isopod *Asellus aquaticus* collected several kilometres upstream from a known industrial discharge. The upper end of this range agrees fairly well with the PECregional of 6 mg/kg WWT. The highest measured value from a local discharge site is 5 300 µg/kg (wet weight) in dragonfly larvae. Data on "mixed invertebrates" from the Netherlands range from 996 to 4 039 µg/kg WWT.

Measured levels of DEHP in molluscs vary between 10 µg/kg (wet weight) and 4 300 µg/kg (wet weight). The molluscs showing the highest values were collected in the River Elbe and are assumed to represent regional exposure. The highest values agree fairly well with the calculated regional concentration of 5.5 mg/kg WWT. Local calculated concentrations are in many cases similar to these levels but some are one to two orders of magnitude higher.

Measured levels of DEHP in fish vary between a few µg/kg and 19 000 µg/kg. Several studies report DEHP in the range of 2 600 to 7 200 µg/kg on a fresh weight basis in muscle. An Austrian study in 1997 reported a maximum value of 2 600 µg/kg (WWT). At five Austrian sites DEHP levels in a total of eight fish samples exceeded 1 000 µg/kg (WWT). The 90th percentile is 250 -500 µg/kg (WWT). In Dutch surveys the concentrations ranged from below the detection limit of 1 µg/kg (WWT) to 300 µg/kg (WWT). This range of measured values corresponds well with calculated regional levels of approximately 2 mg/kg and also with most of the local PECorals in fish.

4.3. Conclusions of the EU RAR

Based on generic scenarios and default emission data the EU RAR for DEHP (EU RAR 2008) concluded that there was concern for birds consuming mussels and mammals consuming earthworms that were exposed to DEHP near sites processing polymers with DEHP or sites producing printing inks, sealants and/or adhesives with DEHP (RCR >1) . No concern was identified for the limited number of sites that reported measured emission data or regional and continental scale risk characterisation.

5. DBP

5.1. Predicted environmental concentrations

The EU RAR (2004) considered production, industrial and consumer life stages related to the use of DBP. Local, regional and continental PECs in different environmental compartments, including secondary poisoning, were calculated using EUSES (v1.0) in combination with TGD default emission values or, where available, measured data.

For production sites, local PECs for individual sites were calculated individually based on site-specific data. For downstream uses (formulation/processing) generic local PECs for the different types of uses (e.g. plasticiser in PVC, adhesives, printing inks and fibres) were estimated.

Detailed summary tables of the PECs for different environmental compartments at local scale for each of the different use scenarios are presented in section 3.1.2.2.1 of the EU RAR. A summary of regional PECs, reproduced from Table 3.8 in the EU RAR, is provided in Table B80, below

Table B81 Calculated regional PECs for DBP from EU RAR

	Regional
PEC in water (µg/l)	0.4
PEC in sediment (µg/kg)	89
PEC in air (µg/m^3)	0.006
PEC in soil (mg/kg)	0.01

5.2. Measured data

The available studies on the occurrence and concentration of DEHP in environmental media, municipal and industrial wastewaters, sewage sludge and biota were summarised in the EU RAR. However, much fewer data were available at the time of the preparation of the EU RAR for DBP than were available for DEHP.

Mean measured DBP concentrations in surface waters from Germany. The Netherlands, UK, Norway and France range from 0.1 to 1 µg/l. This set of regional measured data of DBP in surface waters is considered to be reliable and representative of EU regional concentrations. Mean measured concentrations of DBP in aquatic sediments from The Netherlands, Sweden, Norway, Germany and Denmark range from 0.001 to 2.4 mg/kg (dry weight basis). Measured data for soil from the EU are not available. Concentrations from two Canadian studies range from 0.027 to 1.4 mg/kg.

Concentrations of DBP in European biota are available for aquatic invertebrates (0.3 – 0.8 mg/kg dry weight) and fish (0.2 – 0.5 mg/kg dry weight), both reported from the same study. Further data on freshwater fish are available from Canada (0.5 mg/kg) and the USA (<0.02 – 35.0 mg/kg wet weight). A single Canadian study reported concentrations of DBP in the egg yolk of cormorant and herring gull of 14.1 and 19.1 mg/kg (lipid basis), respectively.

5.3. Conclusions of the EU RAR

The initial EU RAR concluded that there was a need for further information to adequately characterise the risks to plants exposed via the atmosphere but that there no need for further information or testing or further risk reduction measures for the aquatic compartment (including sediment), soil and secondary poisoning. An addendum to the EU RAR published in 2004 took into account further information from a long-term plant toxicity text and concluded that there is a need to limit risks because of an anticipated risk to plants from atmospheric exposure at a local scale for sites using DBP for PVC production, adhesive production, use of printing inks and glass fibre production.

6. BBP

6.1. Predicted environmental concentrations

The EU RAR (2007) considered production, industrial and consumer life stages related to the use of DBP. The calculations of regional and continental PECs were performed using the EUSES

model (v1.0) in combination with TGD default emission values or, where available, measured data.

The EUSES WWTP defaults for production sites have been substituted with site-specific information with regard to release to surface water.

Detailed summary tables for different environmental compartments at the local scale for each of the different use scenarios are presented in section 3.1.3 of the EU RAR. A summary of the regional PECs, reproduced from Table 3.5 in the EU RAR, is provided in Table B82 below.

Table B82 Calculated regional PECs for BBP from EU RAR

	Regional
PEC in water (µg/l)	0.17
PEC in sediment (µg/kg)	0.07
PEC in air (µg/m^3)	0.0063
PEC in soil (mg/kg)	0.03

6.2. Measured data

The available studies on the occurrence and concentration of DEHP in environmental media, municipal and industrial wastewaters, sewage sludge and biota were summarised in the EU RAR.

There are no monitoring data available for production or formulation sites. BBP measurements in surface water are available for a variety of locations. Most of the samples show levels of BBP of less than 1 µg/l. The exceptions are the samples taken from industrial areas in Germany; both the Rhine and Emscher locations gave samples with BBP concentrations > 1 µg/l. However, a more recent monitoring survey show much lower BBP levels indicating that levels of BBP are no longer of concern here.

6.3. Conclusions of the EU RAR

The exposure scenarios for the production sites are based on site specific information and on default values. The PEC/PNEC ratios for the aquatic compartment are below 1.

Two use categories showed PEC/PNEC ratios > 1: use categories IIIa (flooring large and small sites) and IIIh (formulation of confidential use) based on BBP consumption data from 2004. However, in 2005 there were only two producers left.

Annex C: Baseline

The “baseline” scenario describes the tonnages of DEHP, DBP, DIBP, and BBP estimated to be contained in articles placed on the EU28 market in the absence of the proposed restriction. The scenario reflects foreseen regulatory changes and employs a set of assumptions taking into account the main factors impacting the projections of the estimated tonnages in articles. These factors include the long term market forces influencing the use of the four phthalates in article manufacturing in EU28 and the import of articles containing the four phthalates to the EU.

Use of the four phthalates in the production of articles in the EU28

The EU28 use of DEHP in article production is estimated on the basis of EuroStat orthophthalate production for 2004 to 2013.[90] The assumptions for the future trends of use in the EU28 largely draw on historic trends of substitution as a result of previous regulatory actions. The estimates are corroborated by confidential information (from applications for authorisation and market intelligence). This suggests a high degree of confidence in the estimates.

DEHP use by article group is estimated from confidential information from a survey of downstream users by applicants for authorisation. The data represents 2011 shares of DEHP use by application type (i.e., by end-use of DEHP in articles) reported by article groups. The 2011 shares of article groups are assumed throughout the study period. It is possible that DEHP is phased out in some end-uses faster than in others; however, there is no historical information which could suggest a more informed assumption. There is also no certainty that any historical trend would be applicable beyond 2015 because there is no information how the use of DEHP in EU28 article manufacturing has changed past its sunset date.[91] Furthermore, the applicants did not serve the whole EU DEHP market in 2011; therefore, there may be article groups that are misreported. The impact of these assumptions is low as the articles were included in the scope primarily on the basis of their contribution to risk (and as shown above, there is a high degree of confidence in the total DEHP tonnages used).

EU28 use of DBP, DIBP and BBP in the production of articles in the scope of this restriction proposal is estimated on the basis of EuroStat data, market intelligence and information from applications for authorisation. Due to the small tonnages used, individual statistics for these phthalates are rarely presented. In addition, there is high uncertainty related to how much these phthalates are used in soft PVC applications, as they are often used in applications such as sealants, adhesives, inks, etc. which may not fall within the scope of this proposed restriction, as they may not be used in articles. Therefore, there is uncertainty associated with the estimation of tonnages of the three phthalates in total and even greater related to the individual tonnages for BBP and those for DBP and DIBP combined. The tonnages of the three phthalates used in EU article manufacturing are estimated on the basis of the best available information. The uncertainty associated with these estimates impacts years 2006 – 2014, as the use of DBP, DIBP and BBP is assumed to be fully phased out in EU produced articles after

[90] EuroStat: 20143410 - Dibutyl and dioctyl orthophthalates and CN code: 29173200 Doctylphthalates (mainlyDEHP).

[91] This is because while an authorisation decision is pending, there is no obligation for downstream users of authorisation applicants to report their use of the substance under art. 66 of REACH.

21 February 2015 (as no authorisation applications were submitted for their use in articles within the scope of the restriction proposal).

The baseline projections of the EU28 use of the four phthalates in article manufacturing assume that past trends of domestic and international use of the four phthalates will continue from 2013 onward (the latest available historical statistics), with the following notable exceptions which take into account recent regulatory changes:

- the ban on the use of the four phthalates unless an authorisation is granted after their Annex XIV sunset date (21 February 2015);
- the modifications of the RoHS Directive on the use of the four phthalates in electrical and electronic equipment, such as wires, cables and moulded parts (to take effect in 2019 unless some exemptions will not be given before 2019).

These regulatory changes were taken into account in the baseline scenario as follows:

a) Impact of ban on the use of the four phthalates past their sunset date

The authorisation title of REACH impacts only the EU28 use of DEHP in the production of articles and excludes their exports or imports.

Currently, there are six pending authorisation decisions which impact articles within the scope of the proposal. The first group of three applications includes the formulation of DEHP in compounds, dry-blends and plastisol formulations and industrial use in polymer processing by calendering, spread coating, extrusion, and injection moulding to produce PVC articles. The applicants did not include the following uses in the scope of their applications for authorisation (in addition to any uses explicitly restricted under other EU legislation):

- Erasers;
- Adult toys (sex toys and other articles for adults with intensive contact with mucous membranes);
- Small (<10 cm) PVC items available in the home environment (without attachment to larger objects), which can be swallowed by small children;
- Textiles/clothing intended to be worn against the bare skin;
- Formulation and processing of rubber articles;
- Formulation of end product mixtures such as sealants, adhesives, and paints.[92]

The second group of three applications with a pending decision is for recycled material covering formulation of recycled soft PVC in compounds and dry-blends and industrial use of recycled soft PVC in polymer processing by calendering, extrusion, compression and injection moulding to produce PVC articles.

It is uncertain whether further phase out of DEHP has occurred past the sunset date of the substance because downstream users are not obliged to notify ECHA (under Art. 66) of their use until the authorisation decision is made.[93] Therefore, for the purpose of estimating the use of the four phthalates in articles in scope of the proposed restriction, the following assumptions

[92] For professional uses, as for consumer uses the substances are already restricted under Annex XVII entry #30.

[93] A decision by the European Commission on the authorisation applications is pending as of April 1, 2016.

were made for the use of the four phthalates in the EU28, which is subject to authorisation requirements under REACH:

- As no applications for authorisation were received for DBP, DIBP, and BBP in plasticised articles, these substances were assumed to be fully phased out in Europe as of 2015.
- Between 2013 and 2019, EU28 use of DEHP in articles in the scope of the restriction proposal, except wires and cables,[94] will continue to decline with the same rate of decline as between 2010 and 2013, i.e., about 13% annually on average as reported by EuroStat.
- Higher (than 13%) annual rates of decline were assumed for 2013-2015 as it is anticipated that the inclusion of DEHP on the Candidate list and the approach of the sunset date in 2015 have led to significant substitution of DEHP.
- Slightly lower than 13% decline rates were assumed between 2015 and 2019 as it is anticipated that substitution of DEHP will continue as not all downstream users in the supply chain of the authorisation holders will chose to take advantage of the authorisation. Thus, on balance, the projected decline in the DEHP use in articles manufacturing in EU28 is about 13% annually on average between 2013 and 2019. In effect, it is assumed that the regulatory pressure associated with the need to re-apply for authorisation is about the same as the combined pressure associated with regulatory actions such as their inclusion on Annex XIV, the approach of the latest application date and the requirements under the Food contact materials legislation.
- As a result of the above assumptions, in 2019, the EU28 use of DEHP in articles manufacturing within the scope of this proposal was projected under the baseline scenario to represent about one-third of the tonnages used in articles in 2011 (the reference point in applications for authorisations). This reflects the assumption that about 40% of downstream users' tonnages would be replaced prior to the sunset date (a decline since 2011) and by about another 35% during the four year authorisation period[95] (i.e., between 2015 and 2019).
- The DEHP content in recyclate is projected to decline by about 4.5% annually until 2015, due to decision not to apply for an authorisation and due to the declining DEHP content in incoming PVC waste. The rate of decline for 2016-2019 reflects primarily only the latter, therefore, it is lower: less than 1.5% annually.

From 2020 onward for the remainder of the study period (2039), it is assumed that:

- The holders of authorisation (or their downstream users) could restate their case for authorisation (under the adequate control or socio-economic route if it can be demonstrated that suitable alternatives for the applicant are not available) at the time of the review of the authorisation and the period of authorisation can be extended past 2019.
- No further regulatory pressures are anticipated past 2019; however, substitution trends driven by market forces are assumed to continue in the future: DEHP would continue to decline by 3.5% annually for the remainder of the temporal scope of the analysis.[96] This

[94] See point b) for assumptions impacting wires & cables.

[95] A decision by the European Commission on the authorisation applications is pending as of April 1, 2016.

[96] Applicants for authorisation assumed that DEHP would continue to be substituted with a rate of between [redacted] from 2011 levels under their continued use scenario. (AFA 2013a)

is equivalent to a further decline in the use of DEHP of another 50% from 2020 to 2039. The decline is assumed to be the balance of opposing forces anticipated to influence demand for plasticisers and DEHP in particular in the manufacture of articles. These are:

- an increase in DEHP tonnages used in EU28 article manufacturing due to higher demand for consumer articles (i.e., almost all in the scope of the restriction proposal) spurred by population and income growth;[97]
- a decline in DEHP tonnages used in EU28 article manufacturing due to general trends to outsource manufacturing activities of lower margin consumer products to lower cost jurisdictions;
- a decline in the tonnages of DEHP used in the EU28 due to awareness of suitable alternatives and their likely more competitive prices in comparison to DEHP in the future.

The uncertainties related to the assumed impact of authorisation applications on the conclusions on cost-effectiveness and risk reduction capacity of the proposed restriction are tested. For details see Annex E.

b) Impact of recent changes to the RoHS Directive

The recent changes in the RoHS Directive are relevant to the baseline scenario as wires and cables primarily for consumer use indoors are included in the scope of the proposed restriction to ensure consistency between the two regulations and to ensure that no exemptions for the use of the four phthalates are given under RoHS.[98] The changes in RoHS will impact the use of the four phthalates in electrical and electronic equipment (EEE), such as wires, cables and moulded parts. The cables included are those with a rated voltage of less than 250 volts that serve as a connection or an extension to connect EEE to the electrical outlet or to connect two or more EEE to each other. For simplicity reasons, it is assumed that 100% of the cables that qualify to be included in the scope of the proposed restriction (i.e., used indoor or outdoor with prolonged contact to skin) to assumed to be phased out by 2019 due to requirement of the RoHS legislation.

The assumptions are applied to estimates of the use of the four phthalates in manufacturing of both domestic and internationally produced wires and cables. No other regulatory actions are announced and therefore assumed to impact the articles categories included in this restriction proposal.

Table C1 below summarises the use of the four phthalates in EU28 manufacturing of articles in scope which take into account the assumed impact of the authorisation requirements and the recent changes in the RoHS Directive.

[97] EuroStat projects EU28 population growth at about 0.1% annually on average between 2013 and 2041. Real GDP growth is projected to just under 2% annually by 2020: http://knoema.com/mewdmh/european-union-gdp-growth-forecast-2015-2020-data-and-charts. It is assumed that this growth would continue throughout the temporal scope of the analysis.

[98] Currently there are no exemptions for the phthalates in Annex III of RoHS but industry can apply for exemptions once the changes come into force.

Table C1 Tonnes of DEHP, DBP, DIBP and BBP contained in articles in scope placed on the EU28 market – historical data and baseline projections

DEHP, DBP, DIBP and BBP content	**2011**	**2014**	**2020**	**2039**
Tonnes used in EU28 article manufacturing	92 403	62 612	13 828	9 663
% change from previous period		*-32%*	*-78%*	*-30%*
Tonnes contained in Exported articles	14 438	15 722	5 952	3 025
% change from previous period		*9%*	*-62%*	*-49%*
Tonnes contained in Imported articles	101 256	124 245	112 965	136 474
% change from previous period		*23%*	*-9%*	*21%*
Tonnes contained in articles placed on EU28 market*	**179 222**	**171 135**	**120 841**	**143 112**
% change from previous period		*-4.5%*	*-29%*	*18%*

Notes: * Tonnes contained in articles placed on EU28 market = Tonnes used in EU28 article manufacturing - Tonnes contained in Exported articles + Tonnes contained in Imported articles

Content of DEHP, DBP, DIBP and BBP contained in exported and imported articles

In summary, the tonnages of DEHP, DBP, DIBP and BBP contained in imported and exported articles was derived from EuroStat data by CN (Combined nomenclature) code on the volume of imported and exported articles (see Table C4 for codes included in the analysis). These statistics were adjusted to estimate:

- first, what portion of the tonnes per CN code is the plasticised material (see Table C2);
- second, what portion of the tonnes plasticised material are the plasticiser itself (see Table C2);
- third, what portion of the plasticiser tonnes could in fact be DEHP, DBP/DIBP or BBP (see Table C3).

Figure C1 Methodology for estimating the tonnages of DEHP, DBP, DIBP and BBP contained in exported and imported articles by CN code

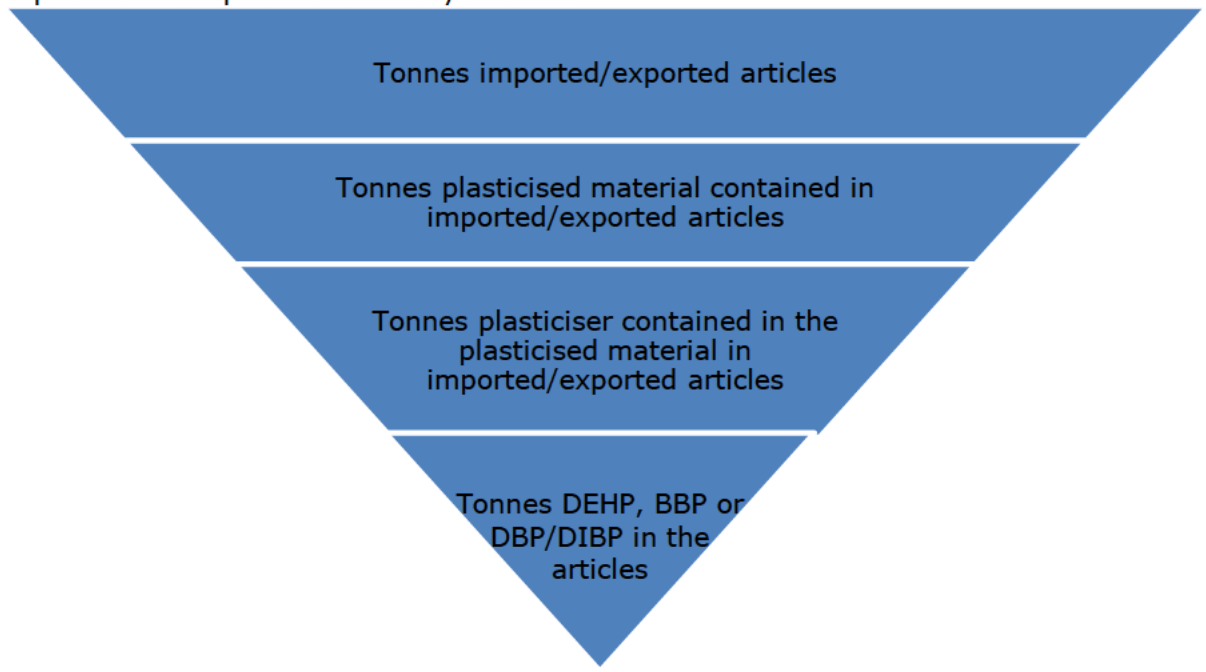


Figure C1 depicts the methodology for estimating the tonnages of DEHP, DBP, DIBP and BBP contained in exported and imported articles by CN code. The estimation begins with the selection of appropriate CN codes that capture the article types within the scope of the restriction proposal. In greater detail, the approach entails:

a) Selection of relevant article groups

The phthalate content in imported and exported articles is estimated on the basis of EuroStat export/import statistics by CN code. For the purpose of this estimation appropriate codes were selected representing the articles falling within the scope of the restriction proposal. The work in preparation of the ECHA 2012a restriction proposal was further refined to ensure the codes reflect the articles most likely to lead to the relevant exposure.

b) Estimation of plasticiser content in articles

EuroStat data of the volume traded by CN code (in kilograms) was adjusted to estimate:

- the content of soft PVC of articles within a CN code: the soft PVC content was estimated by applying an adjustment factor (in percentage) to the total tonnages of articles by CN code (estimates range from 1% to 100% of the weight of articles imported/exported by CN code);
- the plasticiser content in the soft PVC part of the article (estimated above): the plasticiser content was estimated by applying an adjustment factor (in percentage) to the estimated tonnages soft PVC in the articles by CN code (estimates range from 13% to 35% of the PVC imported/exported by CN code).

Separate adjustment factors were applied to each CN code on the basis of information gathered for the purpose of developing and monitoring the Danish PVC tax system. This

information represents the best available information regarding the PVC and plasticiser content by article group.

The model for estimating PVC and plasticiser content was first developed by Skaarup and Skytte (2003), which itself was based on a previous survey of the Danish consumption of PVC in 1995/1996 and formed the basis for the development of the Danish PVC tax system. The model was further enhanced and applied by Brandt and Hansen (2009), by ECHA 2009c to assess the import/export of DEHP in articles, by Høibye et al. (2011) in the background document for the Danish restriction proposal for the four phthalates, by Lassen et al. (2011) for estimating the content of DEHP in articles imported to Norway, by Hansen and Warming (2012) to update estimates of the content of phthalates in imported articles for the Danish EPA (2014). The model (and its subsequent iterations) does not differentiate between articles manufactured in the EU and outside the EU. The figures on PVC content and plasticiser content in the plasticised PVC are mainly based on information on articles produced in Denmark and the remaining EU. No data are available to distinguish between articles produced in the EU or outside the EU. Whereas the plasticiser content of plasticised PVC in the different commodity groups is not expected to differ significantly be region, some regional differences in the average content of plasticised PVC by CN code may be expected. However, no other studies of this scale on PVC and plasticiser content by CN code have been identified in Europe (and internationally).

ECHA 2015e used the conclusions of previous studies and further enhanced the adjustment factors for selected article groups using information from recent surveys of chemical products. These estimates (see Table C2) were applied to historical import and export statistics to estimate the content of the four phthalates for the purpose of this restriction proposal.

Table C2 Assumed phthalate concentration ranges by article group

Article group	**Includes:**			**Soft PVC of total weight of article**	**Plasticiser as % of soft PVC**
	DEHP	DBP/DIBP	BBP		
Flooring & heavy style wall covering	y	y	y	50-100%	~20%
Film & sheet	y	y	y	19-100%	19-30%
Bags	y			5-100%	25-30%
Clothing w coating	y	y	y	10-20%	15-42%
Mattresses	y	y	y	~20%	~25%
Footwear	y		y	~50%	~20%
Cables & wires	y			24-32%	~25%
Balls & bathing equipment	y		y	5-30%	15-35%
Other moulded products	y	y	y	3-50%	~30%
Paper/wallpaper	y	y	y	2-24%	25-30%

Source: ECHA 2015e

Note: Use of the four phthalates in articles included in the Miscellaneous group (see Annex A) is not quantified due to the diversity of the articles in the group and the limited information available.

This approach assumes that the individual articles reported under an article code in Denmark before introduction of the PVC tax system are representative of those in the EU28 and internationally. However, as some of the article groups are fairly large, it is possible that there is under or overrepresentation of some individual articles on the Danish market that are not typical for other markets. Furthermore, the (threat of) introduction of the Danish tax system

may have led to faster phase out of phthalates on the Danish market; therefore making it less representative of other EU or international markets. While there may be a degree of uncertainty related to the estimated phthalate content, this approach is based on best available information and methodology.

c) Estimation of tonnages of DEHP, DBP, DIBP and BBP in imported and exported articles

The tonnages of plasticiser in imported/exported articles (the output of stage b)) were further adjusted to estimate the amount of the four phthalates contained in a CN code. It is assumed that the proportion of the four phthalates in the plasticiser content would be equivalent to the tendency to choose one of the four phthalates instead of their alternatives. This tendency varies by geographic region worldwide and can be observed in the share of individual plasticiser use by major country or by geographical region for smaller trading partners. Market intelligence and information from the applications for authorisation was used to determine the share of these phthalates in the total plasticiser consumption by geographic region. Separate estimates were applied by geographic origin of the imports and exports. These are presented in Table C3 below:

Table C3 Assumed percent DEHP, DBP, DIBP, and BBP of the tonnages plasticiser imported and exported

Region	DEHP		DBP, DIBP, BBP	
	2004-2009	2010-2014	2004-2009	2010-2014
EU28 & EFTA	[redacted]			
Other European				
North America				
Central & South America				
China				
Japan				
Other Asia				
Africa				
Australia & Oceania				

Sources: IHS 2013, AFA 2013c

This approach captures the different prevalence of use of the four phthalates in article manufacturing worldwide. As mentioned previously, due to regulatory action, the use of the four phthalates has substantially declined in the EU and North America. The remaining markets, in particular China, where DEHP has traditionally been the dominant plasticiser in article manufacturing, continue to rely heavily on DEHP. Therefore, it was deemed important to ensure the analysis is sensitive to the tendency to use the four phthalates in the place of origin of the article manufacturing. This approach is innovative and significantly reduces the uncertainty in comparison to earlier estimates that made more generalised assumptions, including a flat percentage of phthalates for both EU and imported articles.[99]

[99] The Danish EPA (2014) applied an average concentration of the four targeted phthalates of 17% in European produced articles and 40% in imported articles (independent on origin). In Lassen et al. (2011), the share of DEHP in products was assumed to be different for articles imported from different regions and the following DEHP percentages was applied: Denmark and Sweden (5% of phthalate content), rest of EU (16%), the Americas (19%) and Asia (60%).

Two sets of phthalate content estimates were applied over the historic data on international trade: for 2004-2009 and 2010-2014. This was intended to capture the declining rate of use of DEHP, DBP (and DIBP) and BBP by different geographic region worldwide.

While there is fairly robust information on the consumption of DEHP by region, data on the DBP, DIBP and BBP is often presented in aggregation (usually with other phthalate plasticisers). Thus, there is a degree of uncertainty related to the estimated tonnages of these three phthalates in imported articles. The overall effect of this uncertainty on the estimation of the impacts of the restriction is considered to be minimal, because the tonnages of DBP, DIBP and BBP are small (projected at about 10% of all imports from 2015 onward).

The estimates for wires and cables take into account the foreseen regulatory changes under the RoHS directive. Therefore, it is assumed that all imported articles in this group will seize to contain the four phthalates by 2019.[100]

As no further regulatory changes impacting imported articles are anticipated, including in the jurisdictions of origin of these articles,[101] the phthalate content estimates are assumed to grow by about 1% annually from 2019 levels for the remainder of the study period. The forecast under the baseline scenario does not assume that the phthalate content would grow at the same rate as the increase in article import volumes[102] (see

Figure C2) as the following opposing forces are anticipated to lead to an overall more modest increase in the tonnages of the four phthalates in imported articles beyond 2019:

- an increase due to higher demand for consumer articles in EU28 (i.e., almost all in the scope of the restriction proposal) spurred by population and income growth;[103]
- an increase due to outsourcing manufacturing of lower profit margin products away from EU28 to lower cost jurisdictions which would lead to higher imports (whose relative content of the four phthalates is anticipated to remain much higher than the EU28 for the foreseeable future);
- a decline due to substitution as awareness of suitable alternatives increases.

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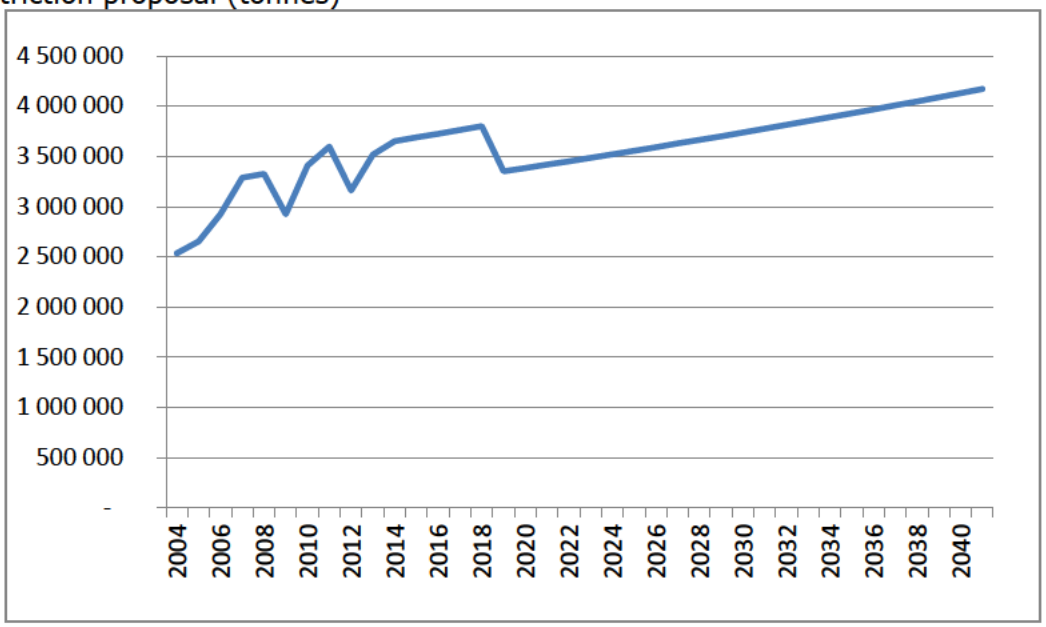
100 This is in addition to the approximately 20% wires and cables excluded from the analysis due to the assumption that they are used outdoors only without prolonged dermal (or mucous membrane) contact.

101 Source: ECHA/EC consultation with WTO members. (2015). The Commission communicated with TBT (Technical Barriers to Trade) contacts during July 29 to 31 October to gather information on future trends in the use of the 4 phthalates, additional uses not yet identified, phthalate content in articles, information on phthalate migration, information on the risk of the four phthalates, information on alternatives, and any legislation on these 4 phthalates used in articles. Information was received from Thailand, South Africa and Japan. Information included standards applicable in Thailand, information on hazards and legislation from Japan, and some trade statistics in South Africa.

102 Assuming article import volumes continue to grow in the future with the same rate as historically between 2004 and 2014, i.e., 4.2% average annual increase, higher in the last five years.

103 EuroStat projects EU28 population growth at about 0.1% annually on average between 2013 and 2041. Real GDP growth is projected to just below 2% annually by 2020: http://knoema.com/mewdmh/european-union-gdp-growth-forecast-2015-2020-data-and-charts. It is assumed that this growth would continue throughout the temporal scope of the analysis.

Figure C2 Historical and projected import volume to the EU28 of articles in the scope of the restriction proposal (tonnes)



As exports are not considered placing on the EU market, it is assumed that these would not be affected by the restriction proposal. The phthalate content in exported articles is projected to follow the same assumptions employed to project the use of the four phthalates in article manufacturing in the EU28.

The impact of these assumptions on the cost-effectiveness of the proposed restriction is tested in Annex E.

Figure C3 below depicts the tonnages of the four phthalates in articles assumed to be placed on the EU28 under the baseline scenario.

Figure C3 Baseline scenario: tonnes of DEHP, DBP, DIBP and BBP in articles placed on the EU28 market

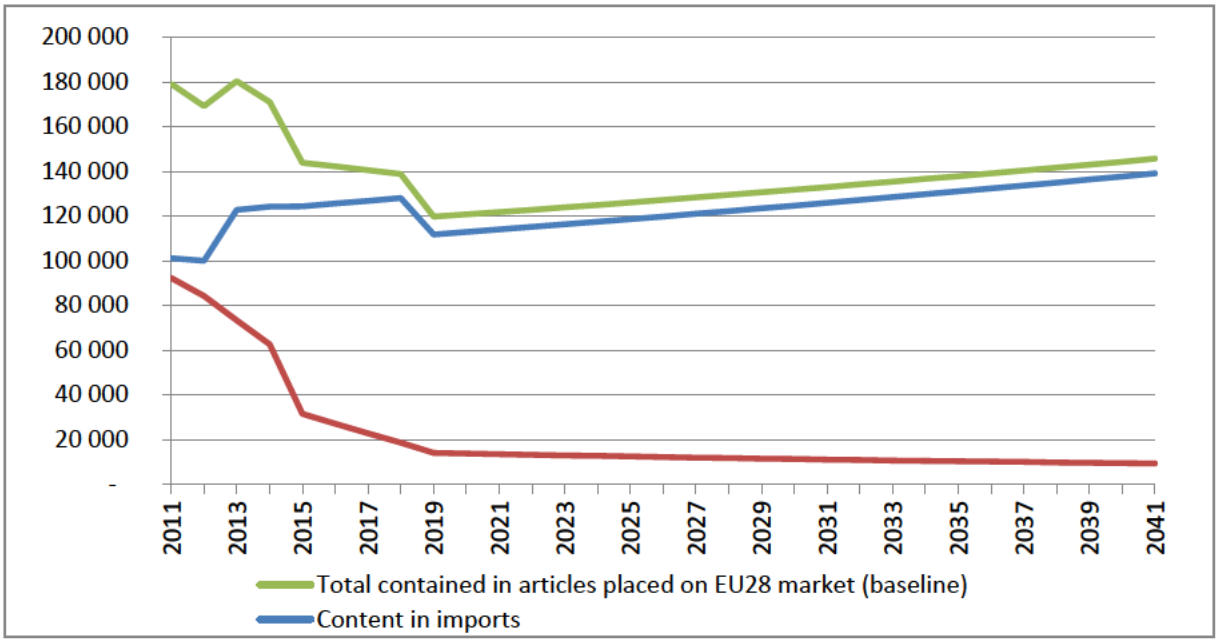


Table C4 Combined nomenclature (CN) codes used in the analysis

CN code	CN Title
39042200	Plasticised poly"vinyl chloride", in primary forms, mixed with other substances
39043000	Vinyl chloride-vinyl acetate copolymers, in primary forms
39044000	Vinyl chloride copolymers, in primary forms (excl. Vinyl chloride-vinyl acetate copolymers)
39153000	Waste, parings and scrap, of polymers of vinyl chloride
39159019	Waste, parings and scrap, of addition polymerization products (excl. that of acrylic polymers, polymers of ethylene, styrene and vinyl chloride and propylene)
39162000	Monofilament with any cross-sectional dimension of > 1 mm, rods, sticks and profile shapes, whether or not surface-worked but not otherwise worked, of polymers of vinyl chloride
39162010	=t("monofilament with any cross-sectional dimension of > 1 mm, rods, sticks and profile shapes, whether or not surface-worked but not further worked, of poly"vinyl chloride"")
39162090	=t("monofilament with any cross-sectional dimension of > 1 mm, rods, sticks and profile shapes, whether or not surface-worked but not further worked, of polymers of vinyl chloride (excl. poly"vinylchloride")")
39173300	Flexible tubes, pipes and hoses of plastics, not reinforced or otherwise combined with other materials, with fittings, seals or connectors
39173310	Flexible tubes, pipes and hoses, of plastics, not reinforced or otherwise combined with other materials, with fittings attached, for the piping of gases or liquids, for civil aircraft
39173390	Flexible tubes, pipes and hoses of plastics, not reinforced or otherwise combined with other materials, with fittings, seals or connectors (excl. Tubes for the piping of gases or liquids, for civil aircraft)
39181010	=t("floor coverings, whether or not self-adhesive, in rolls or in the form of tiles, and wall or ceiling coverings "in rolls with a width of >= 45 cm, consisting of a layer of plastics fixed permanently on a backing of any material other than paper
39181090	=t("floor coverings of polymers of vinyl chloride, whether or not self-adhesive, in rolls or in the form of tiles (excl. those on a backing coated, impregnated or covered with poly"vinyl chloride")")
39191011	=t("plastic strips of plasticised poly"vinyl chloride" or of polyethylene, coated with unvulcanised natural or synthetic rubber, self-adhesive, in rolls <= 20 cm wide")
39191012	=t("plastic strips of poly"vinyl chloride" or of polyethylene, coated with unvulcanised natural or synthetic rubber, self-adhesive, in rolls <= 20 cm wide")
39191013	=t("plastic strips of non-plasticised poly"vinyl chloride", coated with unvulcanised natural or synthetic rubber, self-adhesive, in rolls <= 20 cm wide")
39191051	Self-adhesive foil, film, tape, strip and other flat items of polymers of vinyl chloride, in rolls =< 20 cm wide (excl. Those of plastic tape 'strip' coated with unvulcanized natural or synthetic rubber)
39191061	=t("self-adhesive plates, sheets, film, foil, tape, strip and other flat shapes, of plasticised poly"vinyl chloride" or of polyethylene, in rolls <= 20 cm wide (excl. rolls of plastic strips coated with unvulcanised natural or synthetic rubber)")
39204291	Flexible plates, sheets, film, foil and strip, plasticized, of polymers of vinyl chloride, of a thickness =< 1 mm, (excl. Self-adhesive), non-cellular (not reinforced, laminated, supported or similarly combined with other materials)
39204299	Plates, sheets, film, foil, tape and strip of non-cellular polymers of vinylchloride, flexible, plasticized, not reinforced, laminated, supported or similarly combined with other materials, not worked or only surface-worked, or only cut to rectangular, incl. square, shapes, With a thickness of > 1 mm (excl. self-adhesive products, and floor, wall and ceiling coverings in heading 3918)
39204310	Plates, sheets, film, foil and strip, of non-cellular polymers of vinyl chloride, containing by weight >= 6% of plasticisers, of a thickness of <= 1 mm, not reinforced, laminated, supported or similarly combined with other materials, without backing, unworked or merely surface-worked or merely cut into squares or rectangles (excl. self-adhesive products, and floor, wall and ceiling coverings of heading 3918)
39204390	Plates, sheets, film, foil and strip, of non-cellular polymers of vinyl chloride, containing by weight >= 6% of plasticisers, of a thickness of > 1 mm, not reinforced, laminated, supported or similarly combined with other materials, without backing, unworked or

CN code	CN Title
	merely surface-worked or merely cut into squares or rectangles (excl. self-adhesive products, and floor, wall and ceiling coverings of heading 3918)
39204910	Plates, sheets, film, foil and strip, of non-cellular polymers of vinyl chloride, containing by weight < 6% of plasticisers, of a thickness of <= 1 mm, not reinforced, laminated, supported or similarly combined with other materials, without backing, unworked or merely surface-worked or merely cut into squares or rectangles (excl. self-adhesive products, and floor, wall and ceiling coverings of heading 3918)
39204990	Plates, sheets, film, foil and strip, of non-cellular polymers of vinyl chloride, containing by weight < 6% of plasticisers, of a thickness of > 1 mm, not reinforced, laminated, supported or similarly combined with other materials, without backing, unworked or merely surface-worked or merely cut into squares or rectangles (excl. self-adhesive products, and floor, wall and ceiling coverings of heading 3918)
39211200	Plates, sheets, film, foil and strip, of cellular polymers of vinyl chloride, unworked or merely surface-worked or merely cut into squares or rectangles (excl. Self-adhesive products, floor, wall and ceiling coverings of heading 3918 and sterile surgical or dental adhesion barriers of subheading 3006.10.30)
39232910	=t("sacks and bags, incl. cones, of poly"vinyl chloride"")
39233010	Carboys, bottles, flasks and similar articles for the conveyance or packaging of goods, of plastics, with a capacity of <= 2 l
39233090	Carboys, bottles, flasks and similar articles for the conveyance or packaging of goods, of plastics, with a capacity of > 2 l
39235090	Stoppers, lids, caps and other closures, of plastics (excl. Caps and capsules for bottles)
39261000	Office or school supplies, of plastics, N.E.S.
39262000	Articles of apparel and clothing accessories produced by the stitching or sticking together of plastic sheeting, incl. Gloves, mittens and mitts (excl. Goods of 9619)
39269010	Articles for technical use, of plastics or other materials of heading 3901 to 3914, for civil aircraft, N.E.S.
39269091	Articles of plastic sheeting N.E.S.
39269092	Articles made from plastic sheet, N.E.S.
39269097	Articles of plastics and articles of other materials of heading 3901 to 3914, N.E.S.
39269098	Articles of plastics and articles of other materials of heading 3901 to 3914, N.E.S.
39269099	Articles of plastics or other materials of heading 3901 to 3914, N.E.S. (excl. Articles made from sheet)
42021211	Executive-cases, briefcases, school satchels and similar containers, with outer surface of plastic sheeting
42021219	Trunks, suitcases, vanity cases and similar containers of leather, with outer surface of plastic sheeting (excl. Executive-cases)
42021250	Trunks, suitcases, vanity cases, executive-cases, briefcases, school satchels and similar containers, with outer surface of moulded plastic material
42022210	Handbags, whether or not with shoulder straps, incl. Those without handles, with outer surface of plastic sheeting
42023210	Wallets, purses, key-pouches, cigarette-cases, tobacco-pouches and similar articles carried in the pocket or handbag, with outer surface of plastic sheeting
42029211	Travelling-bags, toilet bags, rucksacks and sports bags, with outer surface of plastic sheeting
42029215	Musical instrument cases with outer surface of plastic sheeting
42029218	Shopping bags, map-cases, tool bags, jewellery boxes, cutlery cases, binocular cases, camera cases, musical instrument cases, gun cases, holsters and similar containers, with outer surface of plastic sheeting (excl. Trunks, brief-cases, school satchels and similar containers, bag or handbag articles, travelling-bags, toilet bags, sports bags, rucksacks and musical instrument cases)
42029219	Insulated food or beverage bags, shopping bags, map-cases, tool bags, jewellery boxes, cutlery cases, binocular cases, camera cases, gun cases, holsters and similar containers, with outer surface of plastic sheeting (excl. Travelling-cases, briefcases, satchels and

CN code	CN Title
	similar containers, bag or handbag articles, travelling-bags, toilet bags, sports bags, rucksacks and musical instrument cases)
48115100	Paper and paperboard, surface-coloured, surface-decorated or printed, coated, impregnated or covered with artificial resins or plastics, in rolls or in square or rectangular sheets, of any size, bleached and weighing > 150 g/m² (excl. Adhesives)
48115900	Paper and paperboard, surface-coloured, surface-decorated or printed, coated, impregnated or covered with artificial resins or plastics, in rolls or in square or rectangular sheets, of any size (excl. Bleached and weighing > 150 g/m², and adhesives)
48142000	Wallpaper and similar wallcoverings of paper, consisting of paper coated or covered, on the face side, with a grained, embossed, coloured or design-printed or otherwise decorated layer of plastics
48143000	Wallpaper and similar wallcoverings of paper, consisting of paper covered, on the face side, with plaiting material, whether or not bound together in parallel strands or woven
48149010	Wallpaper and similar wallcoverings of paper, consisting of grained, embossed, surface-coloured, design-printed or otherwise surface-decorated or covered with transparent protective plastics
59031010	=t("textile fabrics impregnated with poly"vinyl chloride" (excl. wallcoverings of textile materials impregnated with poly"vinyl chloride")")
59031090	=t("textile fabrics coated, covered or laminated with poly"vinyl chloride" (excl. wallcoverings of textile materials covered with poly"vinyl chloride"; floor coverings consisting of a textile backing and a top layer or covering of poly"vinyl chloride")")
61161010	Gloves, mittens and mitts, impregnated coated or covered with plastics, knitted or crocheted
61161080	Mittens and mitts, impregnated, coated or covered with plastics or rubber, knitted or crocheted, and gloves, impregnated, coated or covered with plastics, knitted or crocheted
62102000	Arments of the type described in subheading 6202,11 to 6202,19, rubberised or impregnated, coated, covered or laminated with plastics or other substances
62103000	Garments of the type described in subheading 6202,11 to 6202,19, rubberised or impregnated, coated, covered or laminated with plastics or other substances
62104000	Men's or boys' garments of textile fabrics, rubberised or impregnated, coated, covered or laminated with plastics or other substances (excl. Of the type described in subheading 6201,11 to 6201,19, and babies' garments and clothing accessories)
62105000	Women's or girls' garments of textile fabrics, rubberised or impregnated, coated, covered or laminated with plastics or other substances (excl. Of the type described in subheading 6202,11 to 6202,19, and babies' garments and clothing accessories)
63064000	Pneumatic mattresses of textile materials
63064100	Pneumatic mattresses of cotton
63064900	Pneumatic mattresses of textile materials (excl. Cotton)
64022000	Footwear with outer soles and uppers of rubber or plastics, with upper straps or thongs assembled to the sole by means of plugs (excl. Toy footwear)
64029931	Footwear with uppers of plastic and outer soles of rubber or plastics, with a vamp made of straps or which has one or several pieces cut out, with a maximum sole and heel height of > 3 cm (excl. With upper straps or thongs assembled to the sole by means of plugs
64029939	Footwear with uppers of plastic and outer soles of rubber or plastics, with a vamp made of straps or which has one or several pieces cut out, with a maximum sole and heel height of <= 3 cm (excl. With upper straps or thongs assembled to the sole by means of plugs
64029950	Slippers and other indoor footwear, with outer sole and upper of rubber or plastics (excl. Covering the ankle, footwear with a vamp made of straps or which has one or several pieces cut out, and toy footwear)
85442000	Coaxial cable and other coaxial electric conductors, insulated
85442010	Co-axial cable and other co-axial electric conductors, ready for connectors to be fitted or already provided with connectors

CN code	CN Title
85442091	Co-axial cable, for high frequency, insulated, neither ready for connectors to be fitted nor already provided with connectors
85442099	Co-axial cable and other co-axial electric conductors, insulated, neither ready for connectors to be fitted nor already provided with connectors (excl. High frequency cable)
85444110	Electric conductors of a kind used for telecommunications, for a voltage <= 80 v, insulated, with connectors (excl. Coaxial)
85444190	Electric conductors for a voltage <= 80 v, insulated, fitted with connectors, N.E.S.
85444210	Electric conductors of a kind used for telecommunications, for a voltage <= 1.000 v, insulated, fitted with connectors, N.E.S.
85444290	Electric conductors, for a voltage <= 1.000 v, insulated, fitted with connectors, N.E.S. (other than of a kind used for telecommunications)
85444910	Electric conductors, for a voltage =< 80 v, not fitted with connectors, insulated with plastic material, N.E.S.
85444911	Electric conductors for telecommunications, for a voltage =< 80 volts, insulated with plastic material, (excl. With connectors and coaxial)
85444919	Electric conductors (excl. For telecommunications), for a voltage =< 80 volts, insulated with plastic material, (excl. With connectors, excl. 8544.11-10 to 8544.30-90)
85444920	Conductors, electric, for a voltage <= 80 v, insulated, not fitted with connectors, of a kind used for telecommunications, N.E.S.
85444980	Conductors, electric, for a voltage <= 80 v, insulated, not fitted with connectors, N.E.S.
85444991	Electric wire and cables, for a voltage <= 1.000 v, insulated, not fitted with connectors, with individual conductor wires of a diameter > 0,51 mm, N.E.S.
85444993	Conductors, electric, for a voltage <= 80 v, insulated, not fitted with connectors, N.E.S. (excl. Winding wire, coaxial conductors, wiring sets for vehicles, aircraft or ships, and wire and cables with individual conductor wires of a diameter > 0,51 mm)
85444995	Electric conductors for a voltage > 80 v but < 1.000 v, insulated, not fitted with connectors, N.E.S. (excl. Winding wire, coaxial conductors, wiring sets for vehicles, aircraft or ships, and wire and cables with individual conductor wires of a diameter > 0,51 mm)
85444999	Electric conductors for a voltage 1.000 v, insulated, not fitted with connectors, N.E.S. (excl. Winding wire, coaxial conductors, wiring sets for vehicles, aircraft or ships, and wire and cables with individual conductor wires of a diameter > 0,51 mm)
85445100	Electric conductors, for a voltage > 80 v but =< 1 000 v fitted with connectors, N.E.S.
85445110	Electric conductors of a kind used for telecommunications, for a voltage > 80 v but <= 1.000 v, insulated, fitted with connectors, N.E.S.
85445190	Electric conductors, for a voltage > 80 v but <= 1.000 v, insulated, fitted with connectors, N.E.S. (other than of a kind used for telecommunications)
85445910	Electric wire and cable, for a voltage > 80 v but <= 1.000 v, insulated, not fitted with connectors, with individual conductor wires of a diameter > 0,51 mm, N.E.S.
85445920	Electric conductors for a voltage <= 80 v, insulated, not fitted with connectors, with individual conductor wires of a diameter <= 0,51 mm, N.E.S.
85445980	Electric conductors for a voltage > 80 v but < 1.000 v, insulated, not fitted with connectors, with individual conductor wires of a diameter <= 0,51 mm, N.E.S.
85445993	Lectric conductors, for a voltage > 80 v but =< 1 000 v, not fitted with connectors, with individual conductor wires of a diameter =< 0.51 mm, insulated with plastics other than elastomers or cross-linked materials, N.E.S.
95066200	Inflatable balls
95066210	Inflatable leather balls
95066290	Inflatable balls (excl. of leather)
95069990	Articles and equipment for sport and outdoor games N.E.S; swimming and paddling pools

Source: EuroStat

Notes: List includes CN codes used between 2004 and 2014. Therefore, old codes that are rolled into new codes are also included.

Annex D: Impact Assessment

D.1. Risk Management Options

D.1.1. Proposed options for restriction

The preparation of this restriction dossier on DEHP, DBP, DIBP and BBP in articles was initiated on the basis of the legal requirement specified in Article 69(2) of the REACH Regulation to examine whether the use in articles of Annex XIV substances whose sunset date has passed poses a risk to human health or the environment that is not adequately controlled. The scope of the proposal is limited to these four phthalates on Annex XIV whose sunset date has passed. As shown in Annex B, the conclusion of this examination is that the risk of these four phthalates in articles is not adequately controlled. Therefore, an analysis was conducted of diverse risk management options (RMOs) to identify the most appropriate to address these risks and to define its scope and conditions.

To identify the most appropriate RMO, the possibility to address the risks to human health and the environment from the four phthalates (see Annex B) under other REACH regulatory measures, existing EU legislation and other possible Union-wide RMOs was examined. However, these were deemed inappropriate to address all article categories contributing to risk as presented in section D.1.3.

Therefore, the possibility to impose a restriction under REACH was investigated further and the following restriction options were considered:

- Restriction on the placing on the market of all articles containing the four phthalates
- Restriction on the placing on the market of articles for indoor use and for outdoor use when there is a potential for contact with human mucous membranes or prolonged contact with human skin. This option is herein referred to as “the proposed restriction”. It includes selected derogations, FCMs being one of them.
- Proposed restriction without a derogation on FCMs
- Restriction on the placing of on the market of articles in the scope of the proposed restriction containing DEHP, DBP and DIBP only (i.e., excluding BBP from the scope of the proposed restriction)
- Proposed restriction with a derogation for DIBP in toys and childcare articles
- Restriction on the production as well as placing on the market of all articles

On the basis of the analysis of the effectiveness, practicality and monitorability[104] of these restriction options, as summarised in Section D.1.2., the following restriction is proposed:

[104] All discarded options were found monitorable.

Proposed restriction

Brief title: Restriction on articles containing the four phthalates for: i) indoor use and ii) outdoor use, if in contact with human skin or mucous membranes.

Table D1 Proposed restriction wording by the Dossier submitter

Bis(2-ethylhexyl) phthalate (DEHP) EC number: 204-211-0 CAS number: 117-81-7 Benzyl butyl phthalate (BBP) EC number: 201-622-7 CAS number: 85-68-7 Dibutyl phthalate (DBP) EC number: 201-557-4 CAS number: 84-74-2 Diisobutyl phthalate (DIBP) EC number: 201-553-2 CAS number: 84-69-5	1. Articles containing DEHP, DBP, DIBP, and BBP in a concentration, individually or in combination, greater than or equal to 0.1%[105] by weight of the plasticised material shall not be placed on the market. 2. Paragraph 1 shall apply three years from the entry into force of the restriction. Paragraphs 1 and 2 shall not apply to: a. articles only for outdoor use where the phthalate-containing material is not in prolonged contact with human skin or any contact with human mucous membranes "Prolonged contact with human skin" should in this context be understood as covering a daily overall contact with skin of more than 10 minutes continuously or 30 minutes discontinuously. “Only for outdoor use” should in this context be understood as articles which are not used or stored in the interior of dwellings where humans are present under normal and reasonably foreseeable conditions. b. articles only for use in industrial and agricultural workplaces. This derogation does not apply to articles where the phthalate-containing material is in prolonged contact with human skin by workers. c. measuring devices for laboratory use d. articles placed on the market in the European Union prior to the date in paragraph 2. Paragraph 1 and 2 shall not apply to articles covered under existing legislation: i. Food contact materials covered by Regulation (EC) No 1935/2004 and Regulation (EU) No 10/2011 on plastic materials. ii. Immediate packaging of medicinal products covered by Regulation (EC) No 726/2004, Directive 2001/82/EC or Directive 2001/83/EC, or to medical devices covered by Directive 90/385/EEC, Directive 93/42/EEC or Directive 98/79/EC. iii. Toys and childcare articles containing DEHP, DBP and BBP covered by existing restriction entry 51 in Annex XVII of REACH ‘Childcare article’ is defined as in the existing restriction entries 51 and 52 in Annex XVII.

[105] The current interpretation for the entry 51/52 states that “The limit value of 0.1% should therefore be applied for each group of phthalates combined, i.e. the concentration of DEHP, DBP and BBP combined should not be higher than 0.1% and the concentration of DINP, DIDP and DNOP combined should also not be higher than 0.1%.” EC (2011). *Questions and agreed answers concerning the implementation of Annex XVII to REACH on the restrictions on the manufacturing, placing on the market, and use of certain dangerous substances, mixtures and articles.* Version 4 –25 May 2011. Available at: http://ec.europa.eu/enterprise/sectors/chemicals/files/reach/restr-faq-may-2011_en.pdf

The wording of the proposed restriction was prepared on the basis of limited consultation with two separate enforcement authorities (see Annex F). It enacts a ban of all articles containing the four phthalates in concentration greater than 0.1%, individually or in combination, unless a derogation is provided.

The proposed restriction could also be defined as a positive description of articles that are within the scope, i.e.:

a) any (indoor or outdoor) articles whose phthalate containing material may be mouthed or is in prolonged contact with human skin or any contact with mucous membranes, and
b) any phthalate containing articles that are used (including stored) in an indoor environment where people are present under normal and reasonably foreseeable conditions and potentially exposed via inhalation. This does not apply to articles that are used only in industrial or agricultural workplaces by workers.

Both paragraph a) and b) do not apply to:

- articles placed on the EU market prior to the date of entry into force plus three years of transitional period
- articles covered under existing legislation on Food contact materials (Regulation (EC) No 1935/2004 and Regulation (EU) No 10/2011); immediate packaging of medicinal products (Regulation (EC) No 726/2004, Directive 2001/82/EC or Directive 2001/83/EC); medical devices (Directive 90/385/EEC, Directive 93/42/EEC or Directive 98/79/EC); toys and childcare articles containing DEHP, DBP and BBP (existing restriction entry 51 in Annex XVII of REACH)
- measuring devices for laboratory use

Prolonged contact with human skin, outdoor use and childcare articles are defined as in proposed restriction table above.

As defined in Table D1, the proposed restriction includes in its scope toys and childcare articles containing DIBP in concentration greater than 0.1% w/w; the other three phthalates are excluded as there is an exemption for entry 51 of Annex XVII. The rationale for a restriction is that DIBP has very similar hazard and risk as DBP and DIBP can replace DBP in all its uses. Therefore, from a risk perspective, there is no reason to differentiate DIBP from DBP, which in combination with DEHP and BBP is restricted in entry 51 in toys and childcare articles. Furthermore, this annex demonstrates that the proposed restriction, which includes in its scope DIBP in toys and childcare articles, is effective, practical and monitorable.[106] The most practical way of introducing such restriction is to revise the existing entry 51 of Annex XVII of REACH to include DIBP.[107] It is considered that the same restriction on all four phthalates in toys and childcare articles will ensure clarity for stakeholders in terms of requirements, type of

[106] The restriction costs per tonne DIBP replaced (i.e., the cost effectiveness) are expected to be equivalent to other articles in the scope of the proposed restriction, but the exposure of infants through mouthing of toys and childcare articles is considered to be the highest, thus leading to an improved benefit-cost ratio.

[107] The intention is that the limit applies to toys and childcare articles that contain DIBP in a concentration, individually or in combination with DEHP, DBP and BBP, greater than or equal to 0.1% by weight of the plasticised material. This is consistent to current interpretation of the entry. See footnote to Table D1.

articles covered, any testing or sampling methods, etc. The benefits and disadvantages of derogating DIBP in toys and childcare articles are further discussed in section D.1.2.

Revised restriction wording

The scope of the proposed restriction included wires & cables as these articles can cause dermal exposure or release phthalates to indoor air and thus, contribute to cumulative exposure and risk of the four phthalates. However, the relevant Commission services (DG GROW and DG ENV) requested following the submission of the dossier that the ECHA's Committees (RAC and SEAC), when adopting their opinions, exclude electric and electronic equipment (EEE), as defined in Article 3(1) of RoHS, from the scope of the proposal to restrict these four phthalates under REACH. As the changes to RoHS enter into effect in mid-2019, the Dossier Submitter incorporated the consequent phasing-out of the use of the four phthalates in wires & cables under the baseline scenarios. Therefore, the presented analysis of the effectiveness of the proposed restriction is not affected by the exclusion of wires & cables from the scope of the restriction. The proposed restriction wording was amended to introduce a derogation on EEE falling under RoHS.

During the public consultation, the following additional requests for derogations were received:

- an exemption for imported components for the manufacture of derogated medical devices;
- spare parts for vehicles (automotive and aircraft in particular) placed on the market prior to the entry into effect of the proposed restriction;
- aerospace articles used in the interior of aircraft: the rational for the request is that demonstrating equivalent performance of aerospace article to airworthiness authorities is a long process which can take two to seven years.
- wellingtons and boots made from recycled PVC
- materials that are hidden within, or below, assemblies in vehicles (automotive) would require more time to substitute to allow suitable testing and validation requiring typically 4-5 years.

The intention of the proposed restriction is to make available medical devices subject to the Directive 90/385/EEC, Directive 93/42/EEC or Directive 98/79/EC. Therefore, the import (and manufacturing) of any components required for such medical devices would also need to possible in the EU. However, the intent is not to exempt the import (and manufacturing) of such parts for other purposes.

Similarly, the intention of the restriction is to allow for the existing stock of articles on the EU market to be gradually reduced as older articles reach the end of their useful life. The intent of the restriction is not to make impossible the maintenance, repair, and overhaul to ensure safe use of this existing stock of articles during the course of its normal, useful life. Therefore, the placing on the market of spare parts for vehicles in the scope of the restriction (wagons, bicycles, motor vehicles (motorcycles, cars, trucks, buses), railed vehicles (trains, trams), watercraft (ships, boats), aircraft and spacecraft where humans are present and can be exposed via inhalation during normally foreseeable conditions) with long useful life would also need to be possible. Therefore, considering risk reduction and costs, on balance, the requested derogation for placing on the market of spare parts for vehicles is considered justified by the Dossier Submitter.

The rationale for the request is that development and implementation of alternatives in the aerospace industry is a lengthy process, which necessitates the demonstration of equivalent performance of aerospace articles to airworthiness authorities. The Dossier Submitter evaluated the information provided. There are no known uses for which there are no alternatives for the four phthalates and additional brief consultation with aviation industry representatives did not reveal specific cases for which recertification may be required. Therefore, there is no sufficient information to propose a derogation at this stage.

The Dossier Submitter evaluated the need for a derogation on boots and wellingtons at the time of the dossier preparation. The information provided during the public consultation was also available to the Dossier Submitter at that time. It helped establish that the DEHP containing recyclate is used mainly in industrial and agricultural applications (outside scope of the restriction proposal) and very few tonnages in boots and wellingtons manufacturing. While this information assisted with the justification of the derogations on industrial and agricultural applications, it was concluded that a derogation on boots and wellingtons will be problematic as it is difficult to differentiate those manufactured from virgin and those manufactured from recycled PVC. As very few tonnes of recyclate are to be affected, it is foreseeable to assume that this recyclate can be used for other applications outside the scope of the restriction, therefore, the total tonnes of recycled material would likely be unaffected. As manufacturing is not restricted, it is also possible that boots and wellingtons containing DEHP to be exported to international markets where such restriction is not in place at least in a short term if the transitional period of three years is insufficient to transition to DEHP-free source. The costs and benefits of a mixture of these strategies was taken into account in the estimation of the overall costs and benefits of the proposed restriction,[108] As shown in the Background Document, the proposed restriction, excluding a derogation on boots and wellingtons, is effective, practical and monitorable. It is therefore concluded that the transitional period gives sufficient time to manufacturers of boots and wellingtons to comply with the proposed restriction and the derogation is not justified.

The rationale for the time-limited derogation for “hidden within” parts is that more time would be required (typically 4-5 years) to allow suitable testing and validation of alternatives. Although industry has provided information that they have transitioned to alternatives and very few article types still contain the four phthalates, sufficient information (e.g., volume of phthalates used, number of vehicles impacted, definition of “hidden” articles, etc.) for an assessment of such a derogation was not provided. Therefore, the Dossier Submitter concluded that such a derogation cannot be justified at this stage.

As a result of the Forum advice on the enforceability of the Annex XV proposal for restriction on Four Phthalates (DEHP, BBP, DBP, DIBP) adopted on 12.09.2016, the following changes were introduces to the wording of the restriction:

- clarifications were made to ensure that parts of articles are also included in the scope of the proposed restriction;

[108] For example, if the boots and wellingtons are produced from a virgin material instead of recyclate, the Dossier Submitter estimated an increase in their raw material costs will be about 1-2% of their sales price. See section on impacts on recyclers in Annex D.

- more detailed definitions were introduced for agricultural and industrial workplaces, prolonged contact with skin, as well as for indoor/outdoor environment;
- clarifications were made to assist with the interpretation whether articles with dual use fall in the scope of the restriction;
- proposed that the restriction on DIBP in toys & childcare articles is introduced via an amendment of entry 51 of Annex XVII;
- editorial changes were made to improve clarity, e.g., paragraphs were numbered and all definitions were gathered in one paragraph that applies to the whole restriction entry;
- prepared a second wording of the restriction which on advice of Forum, attempts to define the restriction in terms of what's restricted (version B) rather than in terms of a total ban with derogations for the articles outside the scope (version A).

These changes to the original restriction proposal are presented in Table D2 (version B) below and Appendix D5 (version A):

Table D2 Proposed restriction: final proposal (Version B)

Bis(2-ethylhexyl) phthalate (DEHP) EC number: 204-211-0 CAS number: 117-81-7 Benzyl butyl phthalate (BBP) EC number: 201-622-7 CAS number: 85-68-7 Dibutyl phthalate (DBP) EC number: 201-557-4 CAS number: 84-74-2	1. The following articles or any parts thereof containing DEHP, DBP, DIBP, and BBP in a concentration, individually or in any combination, greater than or equal to 0.1% by weight of each plasticised material shall not be placed on the market: a. any articles whose phthalate containing material may be mouthed or is in prolonged contact with human skin or any contact with human mucous membranes, and b. any phthalate containing articles that are used (including stored) in an interior space where people are present under normal and reasonably foreseeable conditions and exposed via inhalation. This does not apply to articles that are used only in industrial or agricultural workplaces by workers. 2. Paragraph 1 shall not apply to: a. measuring devices for laboratory use or articles that form part of measuring devices for laboratory use[109], b. toys and childcare articles subject to entry 51 of this Annex, c. articles for which it can be demonstrated that they have been placed on the market for the first time in the European Union prior to the date in paragraph 6, d. parts, products or appliances of aircraft for which a type certificate in the meaning of Regulation No 748/2012 has been issued prior to entry into force of the restriction.* 3. Paragraph 1b shall not apply to articles for automotive vehicles, which are produced prior to the date in paragraph 6 plus 2 years as well as spare parts for these vehicles where the vehicle cannot function as intended without that spare part.* 4. Paragraphs 1 and 2 shall not apply to articles in the scope of:

[109] See ECHA Q&A#1179 for definition of measuring devices.

Diisobutyl phthalate (DIBP) EC number: 201-553-2 CAS number: 84-69-5	a. Food contact materials covered by Regulation (EC) No 1935/2004 and Regulation (EU) No 10/2011 on plastic materials. b. Immediate packaging of medicinal products covered by Regulation (EC) No 726/2004, Directive 2001/82/EC or Directive 2001/83/EC c. Medical devices covered by Directive 90/385/EEC, Directive 93/42/EEC or Directive 98/79/EC or components for such devices. d. Articles covered under Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (RoHS Directive). 5. The following definitions apply to this entry: a. "Prolonged contact with human skin" shall mean a daily overall contact with skin of more than 10 minutes continuously or 30 minutes discontinuously, under normal and reasonably foreseeable conditions of use. b. "Interior space" shall mean any space where people are present under normal and reasonably foreseeable conditions and potentially exposed via inhalation. Those may include buildings (residential: e.g., apartments, houses, mobile homes; or commercial areas: e.g., hospitals, restaurants, offices) or vehicles for transportation of people (e.g., railway cars, automobiles, airplanes). c. "Industrial or agricultural workplaces" shall mean any commercial activities performed by workers a workplace in in the following sectors: • agriculture, forestry and fishing [NACE A] • mining and quarrying [NACE B] • manufacturing [NACE C] • electricity, gas, steam and air conditioning supply [NACE D] • water supply; sewerage; waste management and remediation activities [NACE E] • construction [NACE F] d. "Childcare article" shall mean any product intended to facilitate sleep, relaxation, hygiene, the feeding of children or sucking on the part of children. 6. The restriction shall apply three years from its entry into force.
Amendment of entry 51 of Annex XVII of REACH	An amendment of the restriction entry to include DIBP in its scope.

Notes: * Amendments introduced following comments submitted during the public consultation on the SEAC draft opinion. See Appendix D6 for an assessment of the two derogations.

Justification for the selected scope of the proposed restriction

The proposed restriction aims to restrict the placing on the market of articles in whose production the four phthalates were used as plasticisers. These articles primarily include flooring, coated fabrics and paper, recreational gear and equipment, mattresses, footwear, office supplies and equipment, wires and cables, and other articles moulded from or coated with plastic (see Annex A for details).

The scope is defined to include only those articles that present risks to human health via the critical routes of exposure:

i. oral (due to mouthing) and dermal or mucous membrane in an indoor or outdoor environment, as well as
ii. oral (due to ingestion of dust) or inhalation route in an indoor environment.

This means that, for example, articles whose phthalate containing material does not come in contact with skin and mucous membranes, such as a phthalate containing plastic boots with inserts preventing contact with the skin of the foot, would be restricted as types of articles that lead to inhalation exposure in indoor environment. This is because they are present (i.e., stored) indoors and the phthalates from these articles are released to the indoor environment, thus contributing to air and dust levels of phthalates in the indoor environment (see Annex B.9.4.2.) or indoor-like environment such as vehicle interiors.[110]

However, the proposed restriction excludes (via specific derogations) articles whose use does not lead to high exposure situations under normal and reasonably foreseeable conditions for the general population and in particular for vulnerable groups (e.g., children). As these articles do not contribute to exposure to a significant extent, the costs of the substitution of the four phthalates in these articles would outweigh the benefits of the risk reduction. Examples of these are articles only for use in industrial or agricultural workplaces (in terms of being an occupational setting rather than, for example, a farm house, which is a domestic setting; agricultural workplaces could include storage for agricultural vehicles, commercial stables or commercial greenhouses). Another example are articles only present in building frames[111] or in (between) walls, which do not lead to contact with human skin and do not contribute to phthalate levels in the indoor environment to a significant extent.

The proposed restriction also defines prolonged contact with skin for enforcement purposes.

> "Prolonged contact with human skin" should in this context be understood as covering a daily overall contact with skin of more than 10 minutes continuously or 30 minutes discontinuously.

The specified duration is intended to signal that for outdoor articles where only short intermittent dermal contact occurs, it could be reasonably expected that the exposure is low and the articles should fall outside the scope of this restriction proposal, e.g., window blinds or shutters which are installed on the exterior wall of a house. Since such contact does not contribute significantly to the exposure; therefore, restricting these articles would not be necessary. At present, there is insufficient information available to set a specific contact limit for phthalates. Therefore, expert judgement was used to set the final value and should be seen in the context of the assumptions made in the exposure modelling where in the typical scenario for infants, children and women, it is assumed that the daily dermal contact time with articles containing one or more of the four phthalates is 30 min.

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110 KEMI 2015 argues that action on vehicle interiors is required. The report also identifies the four phthalates in articles in trucks and buses: seats and seat covering (DEHP), lamps (DEHP), loudspeaker (DEHP), doors (DIBP), cab roof (DIBP), etc.

111 The main supporting structure of a building - often steel, concrete or wood.

The proposed restriction also introduces a derogation on articles placed on the EU market for the first time prior to the entry into force of the proposed restriction. This is deemed necessary due to the large existing stock of diverse articles containing the four phthalates. The reason for the exemption is that it was concluded disproportionate to replace articles currently in use in the EU. Some of the articles, such as flooring, have long useful lives (normally upgraded on average every 10-15 years) and can require several thousand euro per dwelling to replace. The majority of the articles in the scope are consumer articles with brief lifespan, which are anticipated to be progressively replaced within a few years of the proposed restriction. This will gradually reduce the risk and manage the costs of the restriction.

In addition, a list of derogations is proposed for articles which fall under existing legislation. Due to the diverse list of articles in scope it is unavoidable that the use of some of these articles is already governed by other European legislation, given the long-standing investigation of the risks of the four phthalates. The derogations are included as it is recognised that sector-specific legislation, e.g., medical devices, food contact materials, etc., have effective measures (or effective risk management systems) in place to assess and prevent risk to human health and the environment from these articles. The derogations are also included to further clarify to stakeholders which legislation governs the use of these articles.

With that said, these derogations were first evaluated to conclude whether they adequately address (or can address) the risks. This restriction proposal argues that in the majority of article types in the scope of this proposal, a combined concentration of DEHP, DBP, DIBP, and BBP of less than or equal to 0.1% is required in order to adequately manage the risk to human health. This concentration limit is seen to effectively discourage any intentional use in articles within scope. Therefore, in cases where other EU legislation imposes a higher limit or where the scope of the limits do not align sufficiently, e.g., DIBP use regulated under the Toy Safety Directive, a proposal for restriction is included (see discarded restriction option on a derogation of DIBP in the next section).

A concentration limit is proposed as opposed to migration limit, as explained in Annex B, migration of phthalates varies depending on a number of factor such as type of contact, contact duration, temperature, plasticiser concentration difference, plasticiser concentration level, molecular weight and molecular structure. In addition, concentration limits are easier to enforce (according to previous advice of the Forum) and for companies, especially SMEs, to comply with.

The proposed restriction also includes an exemption for laboratory measurement devices. These may cover a wide range of electronic instruments such as signal generators, logic analysers, oscilloscopes, spectrum analysers, digital multimeters, chemical and biological analysers, etc. and their components. They are essential to the good functioning of electronic communications networks, heavy industrial processes such as steel manufacturing, the testing of vehicles for compliance with emissions standards, and the monitoring of complex systems of all types. The Dossier Submitter is also proposing that the definition of measuring devices in ECHA Q&A #1170 is used.

The proposed restriction anticipates that the market will be able to comply with the restriction within three years of its entry into force. It is assumed that this would occur in 2020 for the purpose of this analysis. It is anticipated that this will give sufficient time as substantial

substitution of these four phthalates in articles has already occurred due to ongoing regulatory action (e.g., substance classification and labelling, authorisations, RoHS, etc.) and as technically feasible alternatives with lower risk profile are available in the necessary quantities on the EU market and internationally at similar price levels. The proposed transitional period will specifically allow:

- EU articles manufacturers to transition to alternatives. These include only those currently operating under an authorisation on the use of DEHP in the formulation of plastisol and the incorporation of plastisol containing DEHP in articles (recommended date for review of authorisation:[112] February 2019). Alternatively, although less likely, the transitional period would give EU article manufacturers time to identify alternative markets domestically (i.e., manufacturing of articles outside the scope of the proposal) or internationally (i.e., all articles as the restriction bans the placing on the EU market and does not restrict exports).
- Recyclers[113] to focus on article production which fall outside the scope of the restriction. According to the recycling industry, the majority of the articles manufactured from recycled PVC are used in industrial or agricultural settings. The restriction is expected to impact primarily the production of wellingtons and other boots, which represent less than 10% of the volume of currently recycled soft PVC waste (EuPC 2016 and ECHA survey of recycling industry). The included derogations on articles for use in industrial and agricultural workplaces are expected to facilitate the recyclers' compliance with the proposed restriction within three years.
- EU importers to communicate to their international suppliers the requirements for phthalate content. Although, the supply chains of many of the articles in scope could be complex, it is anticipated that three years would be sufficient time as industry already has experience with ensuring compliance of phthalates in articles under the Candidate list or other regulatory action on phthalates in the EU or internationally.
- Non-EU manufacturers to transition to alternatives for the purpose of manufacturing of articles intended for the EU market. Given the availability of wide range of similarly priced alternative plasticisers and the familiarity with regulatory actions related to the four phthalates, three years is considered sufficient time for non-EU entities to comply with the proposed restriction.
- All actors to deplete existing supplies of articles produced under current EU regulatory requirements for phthalate content. Three years is deemed sufficient as the sales turnover of the majority of articles is understood as being shorter than three years.
- EU compounders using the four phthalates to substitute with alternative plasticisers or to identify alternative markets domestically (i.e., manufacturing of articles outside the scope of the proposal) or internationally (i.e., all articles as the restriction bans the placing on the EU market and does not restrict exports).
- EU manufacturers of the four phthalates to identify alternative markets (in terms of manufacturing alternative plasticisers or to identify international markets for the four phthalates they produce).

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[112] The decision of the European Commission on this authorisation is pending at the time of the writing of this report.

[113] An authorisation decision on an application by three recyclers is pending at the time of the writing of this report.

- The authorisations (if granted) to approach its recommended review dates.[114]
- Manufacturers of alternatives to make them available on the market in sufficient quantities. As discussed in section D.2.3.4, the alternatives are already available on the EU market and internationally. The transitional period is sufficient to avoid any price shocks on the market due to an increase in demand for alternative plasticisers, including on international markets where the four phthalates (DEHP in particular) currently dominate.

During the public consultation a number of competent authorities and non-governmental agencies (NGOs) argued that due to the significant substitution of the four phthalates by industry, a shorter review period is possible and for broader scope of the restriction capturing all uses of the four phthalates. Some industry representatives, primarily of plasticiser manufacturers argued for a longer review period and further derogations for authorised uses of DEHP.

The proposed restriction was concluded to be the most effective, practical and monitorable RMO. The argumentation for this conclusion is detailed in the remaining sections of this Annex.

D.1.2. Other evaluated restriction options

This section summarises the reasons for discarding the remaining restriction options which were considered during the formalisation of the proposed restriction. Each of these options was succinctly assessed against the main criteria for proposing a restriction.

Restriction on all articles containing the four phthalates (all-encompassing ban on the placing on the market of any articles containing the four phthalates). No derogations given.

The main rationale for restricting the placing on the market of all articles containing the four phthalates is that alternatives exist and that such a restriction would have a clear scope for compliance purposes (i.e., all articles containing the four phthalates). Furthermore, it could be argued that all articles would lead to release of these phthalates in the environment during and at the end of their service life and thus, cause exposure to the environmental compartments or humans through the environment. This potential exposure has not been quantified but the elimination[115] of all exposure sources would avoid any uncertainty related to the remaining risk as a result of the endocrine effects of the four phthalates and considerations on combined risks from other substances.

However, as shown in Annex B, there is a concern from exposure to the four phthalates in articles for human health and possibly also for the environment. Moreover, not all articles contribute to the risk to human health equally. For example, articles primarily for outdoor use with no potential for dermal or mucous membrane contact are expected to have lower

[114] Art. 61 of REACH specifies that authorisation holders are required to submit a review report at least 18 months before the expiry of the time-limited review period.

[115] When a ban on placing on the market of all articles containing the four phthalates is in place, there may (at least in theory) still be some remaining article production in the EU aimed at export. During the production occupational exposure and environmental releases would occur.

contribution to risk. Therefore, a restriction on these articles would have a lower risk reduction potential (which could also be seen as a proxy for the benefits of this restriction option) and benefit-cost ratio. The share of these articles (e.g., roofing, hoses and profiles, car undercoating, etc.) is estimated at 10-15% of all DEHP uses in article manufacturing in EU28 between 2011 and 2013.[116] However, as shown later, one of the significant exposure routes to phthalates, especially for DEHP, is from food. It is suspected that a large majority of this exposure comes from contamination from articles subject to the food contact materials legislation but some contamination may come from exposure of animals and plants to phthalates released from outdoor articles[117] (or from the disposal in landfills of all articles containing the four phthalates at the end of their service life).

In addition, separate legislation for many of the phthalate containing articles already exists (see Table D5 in Section D.1.3) and there would be confusion in the enforcement of potentially conflicting requirements. Derogations from the restriction measure would be more appropriate for those articles covered under other EU legislation.

Therefore, this option was discarded as it would be less net beneficial to society than the proposed restriction.

Restriction on the placing on the market of articles within the scope of the proposed restriction, including Food Contact Materials (FCMs)

This restriction option has the scope of the proposed restriction but it does not propose a derogation for articles that are FCMs, covered by Regulation (EC) No 1935/2004 and Commission Regulation (EU) No 10/2011 on plastic materials. This option was assessed as the risk assessment presented in Annex B showed that the majority of DEHP (up to 75%) and a sizeable proportion of exposure to DBP, DIBP and BBP (up to 25%) is attributable to food. The sources of the food contamination are:

- Food contact materials (FCMs)[118];
- Non-FCM articles that may come into contact with food; and
- The environment: environmental release of phthalates occurs from phthalate manufacturing plants (DEHP and DBP only), from downstream use of phthalates (DEHP and DBP only) and from the article service life[117] (including the waste stage). This may lead to contamination of plant and animal based food sources.

FCMs are considered the main cause of exposure via food due to the migration of the four phthalates into the food that comes into contact with the FCM articles containing them. FCMs

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116 This estimate excludes medical devices and packaging of medicinal products. There also is significant uncertainty surrounding this estimation as international trade data is not taken into account).

117 For example runoff water from roofing or from farm equipment that is not covered by FCM legislation may contribute to environmental contamination of food.

118 Food contact materials are either intended to be brought into contact with food, are already in contact with food, or can reasonably be brought into contact with food or transfer their constituents to the food under normal or foreseeable use. This includes direct or indirect contact. Examples include:
- containers for transporting food
- machinery to process food
- packaging materials
- kitchenware and tableware

include materials and articles containing plastics, which in their finished state are or are intended or can reasonably be expected to be in contact with food. These are materials and articles consisting exclusively of plastics, are coated with plastics, or have plastic layers in multi-material articles. These could be for industrial use, such as flexible hoses, conveyor belts, mixing tanks, mixing blades, gloves, etc. used in food manufacturing; or for packaging of food for sale to end consumers; or exclusively for consumer use for (repeated) storage of food.

Import of the relevant articles in this category to EU28 is estimated to be around 170 000 tonnes (2014), with EFTA trade partners accounting for more than a third of that. The value of EU28 consumption of these articles is about €11 billion with exports representing less than 15% of that (2013).[119] These statistics do not capture adequately the value and volumes of FCMs placed on the EU market as these articles are often merged with others in statistical codes, e.g., food packaging is often included (and therefore, reported) with the value of food; plastic components of food manufacturing equipment can be reported within the value of this equipment.

It is likely that the majority of FCMs placed on the EU28 market are manufactured in the EU but there is no adequate information on the tonnages of the phthalates used also, as the authorisation title of REACH is not applicable to these articles, in addition to the above mentioned challenges. The assumption that most FCMs originate within the EU is likely because it is often uneconomical to transport long-distance voluminous products such as those. It is also supported by a recent study in the Nordic countries, which showed that more than 80% of their sampled articles originated from within the EU (Norden 2015).

Commission Regulation (EU) No 10/2011 on plastic materials and articles intended to come into contact with food mandates that DIBP is not allowed but the other three phthalates could be used as plasticisers in the following applications and concentrations:

- DEHP: in repeated use materials and articles contacting non-fatty foods with a specific migration limit to food of 1.5 mg/kg food as well as a technical support agent in repeated use food contact materials contacting fatty food and single use food contact materials in concentrations up to 0.1 % in the final product.
- BBP: in repeated use materials and articles and in single-use materials and articles contacting non-fatty foods except for infant formulae and follow-on formulae as defined by Directive 2006/141/EC or processed cereal-based foods and baby foods for infants and young children as defined by Directive 2006/125/EC with a specific migration limit to food of 30 mg/kg food, as well as a technical support agent in concentrations up to 0.1 % in the final product.
- DBP: in repeated use materials and articles contacting non-fatty foods with a specific migration limit to food of 0.3 mg/kg food, as well as technical support agent in polyolefins in concentrations up to 0.05 % in the final product.

A recent study of FCM's compliance with phthalate limits in the Nordic countries revealed violations in 32% of the cases. Many of the non-compliant samples were of gloves, hoses and conveyor belts containing DEHP and DBP (Norden 2015). Section B.9.4.3 in Annex B contains some other examples of non-compliance with the FCM legislation. In addition to non-

119 EuroStat trade and production statistics.

compliance, it is important to note that the FCM legislation regulates the maximum migration of the three phthalates at each separate step of the chain from "field to fork". It does not take into account for example if several different (compliant) FCMs containing the three phthalates come into contact with the same food (or food ingredient). Furthermore, it does not consider the overall phthalate burden from repeated contact with FCMs and other sources of exposure (e.g., other articles or concentration of phthalates in food sources).

The following was taken into consideration in the decision whether to propose this restriction option:

- Targeting and risk reduction: As shown in Annex B, exposure to food contaminated with the four phthalates significantly contributes to the total human exposure to the four phthalates. Repeated contact with FCMs, whether compliant or not, results in significant phthalate levels in food. This further compounds the cumulative effect of the exposure of humans to the four phthalates from all articles in the scope of the proposed restriction. Thus, a restriction on the four phthalates in FCMs would significantly reduce the risks to human health.
- Restriction costs and other socio-economic impacts: A number of plasticisers are allowed to be used in FCM as per Commission Regulation (EU) No 10/2011. Previous studies have shown that ATBC, DEHT, DINCH, COMGHA and DINA are some of the most frequent substitutes in FCMs to DEHP, DBP and BBP (See Table D9). Some of these are similarly priced as the four phthalates. Therefore, it can be assumed that the costs of restricting these groups of articles per tonne phthalate replaced would be similar (although slightly higher) to the costs per tonne under the proposed restriction. The other socio-economic impacts are also expected to be similar to those associated with the proposed restriction as it impacts similar economic actors.
- Cost-effectiveness and benefit-cost comparison: As the restriction costs per tonne phthalate replaced in these articles groups are similar to those of the proposed restriction, it can be concluded that the cost-effectiveness of this restriction option is also similar. However, as these articles could potentially account for a large share of the risks the four phthalates are posing to human health, the ratio of benefits to costs of would be improved.
- Practicality and monitorability: There are specific EU regulations, with established risk management systems, that govern the FCM manufacturing and use. Regulating the use of the four phthalates under REACH in addition to the FCM regulations may result in a lack of clarity for economic actors in the food supply system. On the other hand, the current FCM's risk management system does not take into account the risks arising from repeated contact with FCMs and combined effects from other sources of exposure (e.g., the articles in scope of the proposed restriction).

Therefore, in summary, even though FCMs contribute substantially to human health risks from the four phthalates, it was concluded that the best course of action is to derogate FCMs in the proposed restriction on the grounds that a sector-specific legislation would lead to more efficient use of regulatory resources and would lead to improved clarity to stakeholders. However, this proposal highlights the need to address the risks associated with exposure from the four phthalates under the FCM legislation, ensuring that a combined assessment of risks is factored into the decision-making. During the public consultation several stakeholders argued for a restriction on the content of the four phthalates in FCMs.

Restriction on the placing on the market of articles containing DEHP, DBP, and DIBP (no ban on BBP)

BBP has more benign hazard profile in comparison to the other three phthalates. (See Table 7 in Annex B.) It is also more moderately used, leading to fewer opportunities for exposure and therefore, risk. This is supported by biomonitoring results. Exposure to BBP does not significantly contribute to the identified risk (see Annex B, RCRs are between 0.001 and 0.005). Therefore, it was examined whether a restriction on BBP would contribute to the effectiveness, practicality and monitorability of the restriction on DEHP, DBP and DIBP in articles.

The following was taken into consideration in the decision whether to propose this restriction option:

- Targeting and risk reduction:
    - BBP is included in Annex XIV as one of the substances aimed to be progressively replaced with suitable alternatives.
    - BBP is already substituted in the EU as shown by the lack of applications for authorisation (including for recycling).
    - BBP continues to be used internationally and if not restricted, its use could increase. Although its use statistics are usually reported in aggregate with other phthalate plasticisers, there is no reason to suspect that its use internationally has also been phased out: no international regulatory action has been taken against BBP specifically and the plasticiser is often reported as having particular advantages in flooring, artificial leather and seal applications (for isolating double glaze). (ECPI 2015) In fact, in many of the non-PVC applications both BBP and DBP are listed as common plasticisers, suggesting that at least in some applications they may be substitutes for each other (ECPI 2015).[120] The absence of a restriction on BBP may reverse the downward trend of its use.
    - BBP and DEHP, DBP and DIBP all are classified as reproductive toxicants and are all endocrine disruptors. There are uncertainties regarding the existence of a threshold for these substances which introduces uncertainties to the RCRs mentioned above (i.e., risk of BBP may be underestimated). All four phthalates have the same mode of action, i.e., anti-androgenic effects, which highlights the importance of considering their combined effects.
- Restriction costs and other socio-economic impacts: Alternatives to BBP (with lower risk, technically and economically feasible) are widely available at similar prices, as shown in Annex A and the section D.2(Table D9). Therefore, the restriction costs per tonne BBP replaced (i.e., the cost effectiveness) are likely similar to those of the proposed restriction. They would likely incur primarily to actors outside of the EU as the lack of applications for authorisations indicates that the substance has been phased out in the EU by its sunset date in 2015.
- Benefit-cost comparison: BBP has a lower contribution to human health risks in comparison to the other three phthalates but given its limited use, it also has a very low contribution to the total costs of the restriction. The benefits of risk reduction are shown

[120] In addition, as shown in Table A9 in Annex A, DIBP can also be used in flooring, where BBP is seen to have substantial advantages. Therefore, BBP might also be a substitute for DIBP in flooring applications.

to exceed the risks of the proposed restriction even with BBP included in its scope (see section D.3.7).

- Practicality and monitorability: Entry 51 of Annex XVII of REACH restricts DEHP, DBP and BBP together. Inclusion of BBP in the proposed restriction would ensure consistency with previous decisions.

Thus, it was concluded that the risks and the benefits of restricting BBP in articles should be viewed in combination with the other phthalates with the same mode of action. As demonstrated by the effectiveness, practicality and monitorability of the proposed restriction, a restriction on BBP would be beneficial. Therefore, this RMO was discarded from further evaluation.

Proposed restriction with a derogation on DIBP in toys and childcare articles

DIBP is not restricted in toys and childcare articles under entry 51 in Annex XVII but its concentration in toys is limited to 5% (it will be in the future further reduced to 0.3% w/w)[121] under the Toy Safety Directive. [122] However, there are notable differences in the definition of articles in the scope of entry 51 and the Toys Safety Directive.[123] Still, the possibility to derogate the use of DIBP in toys and childcare articles was examined given the partial overlap. Detailed information on the use of DIBP in articles is presented in Annex A.

The following was taken into consideration in the decision whether to propose this restriction option:

- Targeting and risk reduction: As described in Annex B, DIBP, DEHP, DBP and BBP are all classified as reproductive toxicants, have the same anti-androgenic mode of action and are included in Annex XIV. The hazard profile of DIBP is especially considered to be similar to DBP and exposure of DIBP to the general public is higher than DBP. Therefore, based on the hazard and risk of DIBP, there is no justification for a concentration limit that would differ from the other three phthalates (i.e., 0.1% w/w).
- Restriction costs and other socio-economic impacts:
    - Similar to BBP, alternatives to DIBP with lower risk that are technically and economically feasible are widely available at similar prices, as shown in section D.2 (Table D9). Therefore, the restriction costs per tonne DIBP replaced (i.e., the cost-effectiveness) are likely similar to those of the proposed restriction. All other

[121] The 9th Amendment to CLP was adopted by the Commission in July 2016 and will become effective on 1 March 2018 (http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=uriserv:OJ.L_.2016.195.01.0011.01.ENG&toc=OJ:L:2016:195:TOC). The concentration limit for DIBP will become 0.3%.

[122] Toy Safety Directive, i.e., Directive 2009/48/EC of the European Parliament and of the Council of 18 June 2009 on the safety of toys.

[123] Entry 51 does not define toys but defines childcare articles as "*any product intended to facilitate sleep, relaxation, hygiene, the feeding of children or sucking on the part of children.*" The Toy Safety Directive defines toys as products designed or intended, whether or not exclusively, for use in play by children under 14 years of age. Also, Annex I gives a list of articles that are not considered toys, such as, decorative objects for festivities and celebrations, sports equipment, including roller skates, inline skates, and skateboards intended for children with a body mass of more than 20 kg, aquatic equipment intended to be used in deep water, and swimming learning devices for children, such as swim seats and swimming aids, babies' soothers and fashion accessories for children which are not for use in play. Therefore, a number of articles included in the scope of the proposed restriction are not covered by the Toy Safety Directive, e.g., childcare article such as prams, pushchairs, car seats etc.

conclusions on costs and other socio-economic impacts for BBP are also valid for DIBP. The proposed restriction is shown to be net beneficial to society without a derogation on DIBP. In fact, the benefit-cost ratio would improve as while the substitution costs are similar, the benefits are larger as the exposure of infants through mouthing of toys and childcare articles is considered to be the highest.

- DIBP can replace DBP in all its uses (ECPI 2015). Review of studies (ECHA (2016)) did not find data on the trend in the use of DIBP in toys, as the available number of surveys is too small to indicate any trend. Therefore, it cannot be confirmed that DIBP has replaced the use of DBP in toys and childcare articles but this cannot be excluded, given their structural and pricing similarities (ECHA 2016). Such substitution is not desirable, as DIBP has very similar hazard profile to DBP (see Annex B). During the public consultation on the restriction dossier, comments were received from the Norwegian and Swedish competent authority which suggest that DIBP's use in toys and childcare articles is being phased out.
- As demonstrated by the lack of applications for authorisation, DIBP is not used in article production in the EU; therefore, the ban on the use in the production of toys and childcare articles will lead to no costs to EU society, if no authorisations are requested and granted. The proposed restriction will prevent any undesirable future use of DIBP instead of DBP in the manufacturing of articles which would lead to the same occupational exposure concerns that led to the restriction on DBP in entry 51.

- Practicality and monitorability:
  - Regulating substances in use-specific legislation leads to greater clarity to stakeholders regarding their obligations under EU law. However, it is sometimes unavoidable to address the risks of the same substance under several legislations, especially if a substance has diverse uses. Even though, there are plans to reduce the concentration limit to 0.3% w/w, the Toy Safety Directive would not be applicable to all childcare articles as defined by entry 51. Therefore, by derogating DIBP from the scope of the proposed restriction, there would remain a notable number of articles which could potentially lead to exposure and risk to a vulnerable group: infants and young children, e.g., articles such as prams, pushchairs, car seats, baby soothers, etc.
  - The inclusion of DIBP in restriction entry 51 will ensure that the combined exposure to the four phthalates in toys and childcare articles is addressed by ensuring one concentration limit of 0.1% w/w for all, individually or in combination.[124] All other provisions put in place for DBP will (justifiably on the basis of similar hazard profile) be extended to DIBP. These will also include any clarification regarding the type of articles covered, any testing or sampling methods. It'll also lead to greater clarity to stakeholders that the risks from the use of DIBP are similar to that of DBP and the other two phthalates in entry 51.

[124] The current interpretation for the entry 51/52 states that "The limit value of 0.1% should therefore be applied for each group of phthalates combined, i.e. the concentration of DEHP, DBP and BBP combined should not be higher than 0.1% and the concentration of DINP, DIDP and DNOP combined should also not be higher than 0.1%." EC (2011). *Questions and agreed answers concerning the implementation of Annex XVII to REACH on the restrictions on the manufacturing, placing on the market, and use of certain dangerous substances, mixtures and articles.* Version 4 –25 May 2011. Available at: http://ec.europa.eu/enterprise/sectors/chemicals/files/reach/restr-faq-may-2011_en.pdf

- ECHA (2016) has shown that DIBP is used in toys and childcare articles (in 1-3% of toys with flexible PVC with an average concentration of 10-20% of PVC content),[125] often with phthalates such as DEHP, DINP, DBP or DIDP. Non-compliance with the Toy Safety Directive has also been demonstrated by ECHA (2016), RAPEX results and a recent PROSAFE (2016) survey, which found DIBP in about 4% of samples, more than half of which were non-compliant. The proposed concentration limit of 0.1% w/w would act as an effective ban on the use and placing on the market of any articles that contain DIBP. This will improve clarity to stakeholders.[126]

Thus, it was concluded that from the perspective of risk, there is no justification to deviate from the concentration limit of 0.1% w/w specifically for DIBP in toys and childcare articles. As demonstrated by the effectiveness, practicality and monitorability of the proposed restriction, a restriction on DIBP in toys and childcare articles would be beneficial. Therefore, this RMO was discarded from further evaluation.

Restriction on the production as well as placing on the market of all articles

This restriction option is a ban on the use of the four phthalates in the EU production and the placing on the market of all articles containing the four phthalates, with the exception of medical devices, packaging of medicinal products, and articles for military use. This restriction option would prevent the use of the four phthalates in FCMs, all outdoor articles, and all virgin or recycled PVC, in addition to the articles in the scope of the proposed restriction. Under this restriction option, future applications for authorisation for the use of the four phthalates in articles (other than the exempted uses) would not be possible[127].

Table D4 compares the impacts of this restriction option to the proposed restriction on the basis of a simplified analysis. It can be concluded that the advantages of this restriction option are:

- It would increase the health benefits of the restriction to EU consumers. The inclusion of FCMs into the scope of the restriction would bring most of the benefits (see restriction option discussion above on including FCMs in the scope of the proposed restriction). Other benefits would include reduced phthalate emissions to the environment, which could lead to reduced phthalate contamination of food sources.
- It would reduce further the occupational exposure during the production of articles using the four phthalates.
- It would avoid uncertainty related to the remaining risks, amongst others related to the endocrine effects of the four phthalates and combined risks from the four phthalates

[125] The actual concentration is likely much more variable, as shown in Table B31 of Annex B. Some of the recently sampled articles include school bags and children's risk watches, where DIBP was found in concentration of respectively 830-3 100 mg/kg and 70-50 000 mg/kg.

[126] Examining the RAPEX entries as well as personal communication with various stakeholders revealed that there appears to be misunderstanding about the existing concentration limit of DIBP. For example, reference A12/0916/14 states: "the tyres [of a toy] contain di-isobutyl phthalate (DIBP) above the permitted level (DIBP 1.3%)".

[127] Currently, there are two applications for authorisation for the use of DEHP in articles in the scope of this restriction option (see Annex A). The decision of the European Commission on these applications for authorisation is pending at the time of the writing of this report.

and other substances with similar anti-androgenic effects and any potential risks to the environment.

The disadvantages of this restriction option are:

- Including outdoor articles where there is no dermal contact is likely to bring only small benefits. The overall contribution of the outdoor articles to the phthalate body burden is considered to be low, in particular in comparison to the contribution of the articles in the scope of the proposed restriction. The reason is mainly because they do not lead to dermal, oral or inhalation exposure and their contribution to phthalate contamination of plant and animal food sources via the environment is considered to be small. Furthermore, these articles represent a small amount (estimated 10-15%) of the total DEHP use in EU manufacturing of articles. Thus, this restriction option would not be as effective as the proposed restriction, as it would not be targeted at the main sources of exposure that cause the identified risks.
- The estimated costs (Table D4) of this restriction option are higher than the proposed restriction and are likely underestimated. It could not be ruled out that this restriction option could have more severe impacts on selected companies, leading to loss of business and exit, which are not fully captured in the illustrative calculations. The highest costs would likely fall on the recycling sector which is accountable for less than 20 000 tonnes of DEHP in articles and plastisols placed on the EU market.
- Although the benefits for workers under this restriction option are higher than the proposed restriction, the authorisation requirement under REACH appears to be a more targeted regulatory instrument to address risks to workers from exposure to the four phthalates during the production process.

Thus, it was concluded that in comparison to this restriction option, the proposed restriction more effective and practical. Therefore, this restriction option was discarded and not evaluated further.

Table D3 Illustrative calculation of the potential impacts of a restriction on the production as well as placing on the market of all articles containing the four phthalates

	Analysed restriction option	**Proposed restriction**
Restriction scope	Placing on the market & EU article production	Placing on the market
Type of articles	As in proposed restriction, plus FCMs,[128] additional construction & automotive articles,[129] as well as plastisols, compounds and dry blends. Only limited derogations.[130]	Flooring, coated fabrics & paper, recreational gear, mattresses, footwear, office supplies, wires & cables, other moulded or coated with plastic
Tonnes removed from EU market	180 000 tonnes of four phthalates[131]	131 500 tonnes of four phthalates
Risks reduced	Reduction or risk characterisation ratios clearly below one, reduction of uncertainties (minimisation of possible remaining risks).	Reduction of risk characterisation rations at or below one. See Table D14 in Annex D.
Human health benefits	Higher than in the proposed restriction.	€33 m/yr for general public, plus benefits for workers and other non-quantified benefits[132]
Environmental benefits	Reduction of possible risks for the environment in terms of reduced releases of outdoor articles during useful life and at disposal	Reduction of possible impact on environment in terms of reduced releases due to disposal of articles
Actors impacted	Same type of stakeholders as in proposed restriction but also including those additional stakeholders in the construction & automotive industry.	Importers, article producers for EU market and converters of recycled PVC.
Substitution costs	€18.8 m/yr	€15.8 m/yr
Costs to recycling sector	€43.6m/yr[133]	€1.1 m/yr
Total costs	€62.4m/yr	€16.9m/yr
Impact on exports	Included in substitution costs	Exports not directly restricted
Impact on EU substance manufacturers	Same as in proposed restriction	Reduction of the manufacturing of four phthalates. Manufacturers of alternatives pick up the market.
Impact on compounders	Costs to compounders of recycled PVC are included in the costs to recycling sector.	Minimal. The costs are included in substitution costs
Impact on SMEs	Impact on SMEs is higher than the proposed restriction.	Some, primarily on converters of recycled PVC.

[128] It is estimated that the majority of food contact materials (close to 80%) originate from the EU or EEA.

[129] For outdoor use with no dermal contact.

[130] Derogations considered: medical (e.g., medical devices and packaging of medicinal products) and military uses (e.g., ammunition, aircraft seat propellants, rocket propellants).

[131] No analysis was conducted on the phthalate content in FCMs as well as of imported and exported construction, aerospace and automotive articles. The estimate assumes that EU manufacturing of construction and automotive articles is the same as the import. It is uncertain how much of the EU manufacturing of construction and automotive articles are exported.

[132] Estimate includes male infertility, cryptorchidism and hypospadias for the general public. The estimate does not include higher avoided risks for workers in comparison to the proposed restriction.

[133] It is assumed that only post-consumer recycled PVC is impacted and that: i) half the converters will transition to virgin PVC which would lead to them incurring substitution costs due to higher cost of PVC per tonne, ½ the currently recycled post-consumer PVC will not be recycled any more leading to externalities, profit loss of compounders of recycled PVC will be offset by gains of compounders of virgin PVC; and ii) half the converters will transition to post-industrial PVC or stay with post-consumer PVC with both options leading to substitution costs to converters due to higher costs of post-industrial PVC and the need for more frequent testing of post-consumer PVC, however, no externalities are foreseen as the amount of waste recycled will remain the same.

D.1.3. Other Union-wide risk management options than restriction

Possible Union-wide risk management measures other than a restriction are outlined in Table D5 below. However, it is concluded that none of these are realistic, effective and balanced means of solving the problem. As such, none of these other risk management options have been analysed further.

Table D4 Possible other Union-wide options discarded at this stage

Option	Reasons for discarding this option
Non-legislative measures	
Voluntary industry agreement to restrict the use of the four phthalates in articles.	The articles included in the proposal fall within numerous diverse industry sectors, which belong to different industry groups, often dominated by SMEs. There are several thousands of importers and European producers of articles that could contain the four phthalates that are not organised in European associations. VinylPlus is the new ten-year Voluntary Commitment of the European PVC industry. It is built on the Vinyl 2010 programme and outlines the next steps in addressing sustainability challenges for PVC. It represents the whole (organised) PVC industry; however, it does not represent importers of articles, which constitute the main source of the four phthalates in articles placed on the EU28 market. (See Annex C and Annex A for further details). The sheer number of stakeholders makes it difficult to negotiate a voluntary agreement and it cannot be effectively enforced. This will also likely affect the timelines for addressing the risks and the possibility to monitor the effectiveness of the proposed measure.
Voluntary agreement for industry to label articles.	Possible labelling options include: • To consumers – Use outdoors. Some articles intended for outdoor use could also be used indoors. For such products an easy and cheap solution could be labelling, where the label could specify that the article is only intended for outdoor use. However, this RMO was considered infeasible, according to consultation with FORUM, due to practical problems such as labelling (ECHA 2012a). It will also not address direct exposure to articles used outdoors from skin or mucous membrane contact. • To consumers – Ensure sufficient ventilation. This RMO requires an understanding from consumer what is sufficient ventilation and this may not be possible in different climates. It will also not cover direct exposure from articles from, e.g. skin contact. However, this RMO would address some of the risks from articles already in use. The RMO can be supplementary to a restriction in relation to existing articles. Such advice can be given on national or local level.

	This RMO will also share many of the disadvantages of the voluntary agreement to restrict substances such as enforcement and coverage (as above).
Information campaign to consumers to avoid buying the articles in question.	This RMO does not seem to be sufficiently effective. For the consumer, it will be difficult to identify the articles containing the substances. Even if the articles are labelled, it is a problem that some of the articles have a long useful life, e.g. PVC flooring. A house might change owners/tenants within the lifespan of the PVC flooring. Therefore, the person that is exposed might be different than the one taking the buying decision.

Legislation other than REACH

Control of emissions under the IED and/or Water Framework Directive and waste legislation	Articles containing the four phthalates have wide dispersive use. Exposure to the general public via emissions to indoor environment or direct dermal contact occurs during the use phase, not the production phase. Therefore, measures aimed at point sources would not address the risk of exposure and will not be an effective risk management measure.
Taxation on phthalate content	An example of such use of taxes is in Denmark. Since 2000, a tax is in place on phthalates and PVC in articles placed on the Danish market. The purpose of the tax is to create an incentive to phase out phthalates by doubling their effective price. It is assumed that the tax reflects the extra costs of using their alternatives. All phthalates (not only the DEHP, DBP, DIBP and BBP) are covered by the tax. Therefore, such a tax regime does not give incentive to replace the four phthalates with other phthalates. Taxation in general is not a harmonised measure across the EU. Therefore, whilst it might be effective in encouraging substitution, it is not likely that all Member States would introduce relevant taxes and thereby, not all EU citizens will be protected. This is likely to lead to a non-harmonised situation where different Member States apply different tax rates (if at all).
Legislative requirement for labelling, such as an amendment to Annex II of CLP	A labelling requirement in CLP will inform consumers that the articles should either be used outdoors of with sufficient ventilation. This option will suffer from some of the same disadvantages as above (voluntary agreement for labelling) and will take some time to implement.
Sector specific legislation	Uses within the scope of the proposal are varied and widely dispersed. It would be resource intensive to address the risks via a large number of sector specific legislation, which also does not exist for all relevant sectors. In addition, surveys have revealed that REACH restrictions are a convenient way to communicate all-encompassing regulatory measures related to chemicals. However, efforts have been made to derogate articles in the restriction proposal which are adequately covered by existing sector specific EU legislations (e.g., medical devices, FCMs, etc.) to avoid unnecessary overlap of regulatory actions and improve clarify for stakeholders.

Toy Safety Directive:

Includes restriction on the use of CMR chemicals. Substances that are classified as CMR of category 1A, 1B or 2 under CLP Regulation ((EC) No 1272/2008) shall not be used in toys, in components of toys or in micro-structurally distinct parts of toys. The four phthalates are not specifically restricted in the Toy Safety Directive. Instead chemicals that are susceptible to cause cancer, change genetic information, harm fertility or harm an unborn child (so-called CMR substances) are no longer allowed in the accessible parts of toys beyond the concentration limits set in the Regulation on Classification, Labelling and Packaging of substances and mixtures. These limits are the SCL for DIBP of 5%[134] and for the other three phthalates, there are no SCL, so the generic limit applies which for reproductive toxicity category 1b is 0.3%. In addition, DEHP, DBP and BBP are already restricted for use in toys in entry 51 of REACH at a concentration limit of 0.1%.

Construction Products Regulation:

Under this Regulation the information on the content of hazardous substances in the construction products should be included in the declaration of performance to reach all potential users. Information should initially be limited to substances referred to in Articles 31 and 33 of REACH, but all relevant information for other substances should eventually be taken into account. The Regulation refers to the obligations to label products according to the relevant legislation (e.g., CLP).

Medical Device Directive:

Three Directives deal directly with medical devices, either as the medical devices themselves, or as implantable medical devices or as in vitro diagnostics. According to these Directives, medical devices must be designed and manufactured taking into account the toxicity of materials used and minimising the risk for substances to leak out of the device.

Restriction of Hazardous Substances (RoHS) Directive:

Electrical and electronic products are regulated by the RoHS Directive. Recent revisions introduced a restriction on the use of DEHP, BBP, DBP and DIBP. The review is restricted to the use in electric and electronic devices and does not consider other articles and the combined exposure from all articles. The revisions enter into force in 2019.

Food Contact Materials:

Council and Parliament Regulation (EU) No 1935/2004 addresses risks from materials and articles intended to come into contact with food (FCM). Commission Regulation (EU) No 10/2011 on plastic materials and articles intended to come into contact with food is a specific measure for

[134] The 9th Amendment to CLP is likely to be adopted by the Commission in the second quarter of 2016 and will become effective 18 months after entry into force; this is estimated to be at the end of 2017. The concentration limit for DIBP will become 0.3%.

	plastic materials produced within the meaning of article 5 of Regulation 1935/20014. Under Regulation 10/2011, DIBP may not be used, while DEHP, DBP and BBP are authorised to be used in food contact materials with Specific Migration Limit of 1.5, 0.3 and 30 mg/kg food respectively. In addition, application of DEHP and BBP as technical support agents are allowed in concentrations up to 0.1% in the final product, DBP is allowed as a technical support agent in concentrations up to 0.05% in the final product. The regulations do not address risks related to the cumulative exposure from different articles containing the four phthalates. These regulations therefore address the risks only from part of the articles in the scope of this report.
Product Safety Directive 2001/95/EC	This Directive only addresses risks related to specific articles and not risks related to a cumulated exposure from different articles. It can be used to restrict articles but this needs annual renewal (similar to the old decision on phthalates in toys that was eventually made into a restriction).

Other REACH processes

REACH Authorisation process	All four phthalates have been included in the authorisation list (Annex XIV). The authorisation route only addresses the incorporation of the four phthalates in articles within the EU. This means that risks related to the placing on the market of imported articles are not addressed.
REACH Art. 68.2	REACH Article 68(2) stipulates that substances that are CMR categories 1 or 2 can be subject to a proposal from the Commission to inclusion in Annex XVII for consumer uses without using the procedures in article 69-73 in the REACH Regulation. The procedures in article 69-73 give an opportunity to investigate the human health and socio-economic implications of the combined exposure to consumers from various groups of articles containing the four phthalates. It is argued that, the investigations under art. 68(2) may require an equal degree of scrutiny by article group and thus, there may not be gains in terms of efficiency if this alternative regulatory route is explored.

D.2. Alternatives

Technically feasible alternative plasticisers are available at similar or lightly higher prices for all uses of DEHP, DBP, DIBP and BBP in the scope of the proposal; in fact these alternatives are already widely used in the EU and internationally. Their share of total plasticiser use in article production is increasing while that of the four phthalates has seen a steady decline over the past decades.

A number of studies have assessed the suitability of alternatives to the four phthalates. This section presents a summary of the latest reviews included as part of the Danish proposal for restriction of the four phthalates in articles (ECHA 2012a), and applications for authorisation for DEHP and DBP submitted to ECHA in 2013 (AFA 2013a,b,c).

D.2.1. Description of the use and function of the substances

A plasticiser or softener, as formally defied by the International Union of Pure and Applied Chemistry (IUPAC), is a substance or material incorporated into a material (usually a plastic or an elastomer) to increase its flexibility, workability or capability of swelling or stretching. A plasticiser may reduce the melt viscosity, lower the temperature of the second-order transition or lower the elastic modulus of the melt. When combined with PVC, plasticisers convert the rigid, intractable resins into workable compounds which can exhibit a wide range of properties depending on the type and concentration of plasticisers used.

The properties required for plasticisers, or the plasticised PVC products, generally include compatibility with the resins, non-volatility, non-flammability, good heat and light stability, good low temperature performance and non-toxicity (Titow 1984). However, the relative importance of these properties is different for different types of PVC processing:

- Plastisol processing (e.g., spread coating of wall covering, cushioned flooring, bags, and coated fabrics etc.) requires plasticisers with low heat viscosity, good gelation properties, low volatility and ensuring plastisol storage stability;
- Calendering process (e.g., shower curtain, tablecloth, batch equipment, tiles etc.) requires plasticisers that have low volatility, good processability (good solvators for PVC, not too viscous) and good resistance to extraction;
- Extrusion process requires very permanent plasticisers, good solvators for PVC, not too viscous and that the plasticiser can be processed and fused at a reasonable temperature (180ºC - 200ºC). (ECHA 2013)

According to applicants for authorisation, DEHP has reasonable plasticising efficiency, fusion rate, and viscosity and a number of technical criteria need to be taken into account in the selection of alternatives to DEHP in various applications:

Table D5 Criteria for assessment of alternatives of DEHP

Criteria category	**Specific technical feasibility & selection criteria**	**Criterion**		**Definition**
		Core	**Secondary /** Application specific	
Overarching criteria	PVC compatibility	x		Ability of two or more substances to mix without objectionable separation
Criteria relating to substance properties & manufacturing process (of flexible PVC)	Processability	x		Ease with which various processes combine the liquid plasticiser with the PVC polymer
	Efficiency	x		Ability to impact desired properties at low concentrations
	Melting/ freezing point		x	Esters with freezing point above -34°C may solidity during shipment or storage.
Criteria relating to the performance of the PVC end-product	Low temperature performance		x	Lower useful temperature limit of the finished product.
	Clarity		x	Clarity may be affected by the incompatibility with the resin (e.g., with polymeric plasticisers) or due to moisture absorption
	Elastic recovery		x	Fraction of a deformation that behaves elastically
	Odour		x	Odour of the end product. Important for such applications as FCMs
	Sterilisability		x	Compatibility with common sterilisation methods (e.g., of medical devices)
	Printability & adhesion		x	Any constrains the plasticiser may impart on the printability & adhesion properties of the PVC end-product
Criteria related to lifetime of PVC article	Permanence	x		Tendency of the plasticiser to be permanently retained once compounded with the PVC polymer

Source: AFA 2013a

DBP and DIBP are used in PVC for their viscosity reducing properties as processing aid for PVC plastisols and compounds typically in quantities of 5 to 10 wt% due to their higher polarity. BBP is used in PVC primarily as a fast fusing secondary plasticiser for foamed plastisols, e.g. used in flooring (ECHA 2013). One of the BBP's main benefits is that it enables manufacturing with less energy input than many similar plasticisers. The flooring industry makes use of BBP to add surface properties to flooring materials, minimising maintenance and prolonging the floor's longevity (ECPI 2015). Annex A contains further information about the use and function of the four phthalates.

D.2.2. Identification of alternatives fulfilling the function

According to the applicants for authorisation (AFA 2013a), in practice, the choice of plasticiser is generally a compromise between the processing technique, the end application of the plasticised material and economic factors. Their downstream users rated the two most important criteria for plasticiser selection as the cost of the plasticiser followed by compatibility with PVC. However, most applications rely on multiple criteria and can be expected that those applications that have more stringent requirements (indicated by a wide range of relevant comparison criteria) also have a smaller range of alternative plasticisers able to deliver the necessary specifications (AFA 2013a).

More than 30 000 different substances have been evaluated for their plasticising properties. Of these, only a small number – approximately 50 – are today in commercial use after meeting the rigorous performance, cost, availability, health and environmental requirements which are imposed by the market, users and regulators. The most common plasticisers include esters such as adipates, azelates, benzoates, citrates, cyclohexanoates, orthophthalates, sebacates, terephthalates and trimellitates (ECPI 2015). All plasticisers of these families could replace one or more of the four phthalates in one or several of the applications included in the scope of this restriction proposal.

Table D7 and Table D8 give a summary of the key performance characteristics of these plasticiser families as well as examples of key applications:

Table D6 Performance overview of plasticiser families

Family	**General Purpose**	**Performance plasticisers**			**Specialty plasticisers**		
		Strong Solvent	Low Temperature	Low Volatility	Low Diffusivity	Stability	Flame Resistance
Phthalates	xx	x	x	x	x		x
Trimellitates			x	xx	x		
Aliphatic dibasic esters			xx				
Polyesters				xx	xx		
Epoxides			x	x		xx	
Phosphates		x	x				xx
Extenders*	xx						
Miscellaneous		xx		xx	xx		

xx = Primary performance function x = Secondary performance function
Source: Wilkes 2005
* Shown in the general purpose plasticiser category because they are most commonly employed with phthalates to reduce costs in general purpose flexible PVC

Table D7 Overview of plasticiser families

Plasticiser family	**Short description**	**Key notes on applications**
Phthalates	The most commonly used plasticisers in the world. In Europe, ca. 1 million tonnes of phthalates are produced each year, of which approximately 93% are used to make flexible PVC. Manufactured by reacting PA with alcohol(s) which range from methanol and ethanol (C1/C2) up to tridecyl alcohol (C13), either as a straight chain or with some branching.	**PVC applications**: Electrical cables, Hoses, Flooring, Wall coverings, Coated textiles, Luggage, Sports equipment, Roofing, Pool liners, Footwear, Medical devices such as tubing and blood bags. **Non-PVC applications**: Coatings, Rubber products, Adhesives, Sealants
Low molecular weight phthalates	Include DEHP, DBP, DIBP and BBP and represent about 11% of the European market.	
High molecular weight phthalates	Include DINP, DIDP, DPHP, DIUP, and DTDP and represent about 85% of all the phthalates currently being produced in Europe	
Aliphatic dibasic acid esters	Based on aliphatic dibasic acids with carbon numbers ranging from C5 (glutaric) to C10 (sebacic).	
Adipates	Alcohols of similar chain length to those used in phthalate manufacturing (typically in the C8 to C10) range can be esterified with adipic acid, rather than PA, to produce a range of adipate plasticisers, e.g. di-2-ethylhexyl adipate (DEHA).	In PVC applications, adipates offer enhanced low temperature properties compared to phthalates. In plastisol applications, adipates impart low plastisol viscosities due to their lower neat viscosities
Sebacates & Azelates	Di-2-ethylhexyl sebacate (DOS) and di-2-ethylhexyl azelate (DOZ) are the most common members of this group, but di-isodecyl sebacate (DIDS) is also used.	These plasticisers impart low temperature performances superior to adipates but also command a significant premium, and their use is generally limited to extremely demanding low temperature flexibility specifications (e.g. underground cable sheathing in arctic environments)
Benzoate esters	Di-benzoate plasticisers are obtained by direct esterification of benzoic acid with glycols.	Used primarily in non PVC applications such as PVAc based adhesives, latex caulks and polysulphide
Citrates	Citric acid is the starting material for a number of citrate ester plasticisers, such as tributyl citrate, acetyl tributyl citrate, triethyl citrate, acetyl triethyl citrate and tri- 2-ethylhexyl citrate.	Toys, Pacifiers, Medical devices, Packaging films. 58% are used in food and beverage applications, 24% in household detergents and cleaners, 9% in pharmaceuticals and 9% in industrial applications.
Epoxy esters	Esters containing an epoxy group such as epoxidised soybean oil (ESBO) and epoxidised linseed oil (ELO). They are formed by the oxidation of an olefinic double bond to an oxirane structure.	Used to improve heat stability in the production of PVC articles by techniques such as extrusion, calendering, injection moulding, rotational moulding and spread coating. They are also used in rubbers, epoxy resins, paints and coatings. These can act as lubricants but also act as secondary stabilisers for PVC due to their epoxy content which can remove HCl from the degrading polymer.

Plasticiser family	Short description	Key notes on applications
Phosphate esters	Triaryl phosphates and alkyl diaryl phosphates are the two important categories of flame retardant phosphate plasticisers. Phosphate esters can help produce low smoke, low flammable flexible PVC.	The principal advantage of phosphate esters is their improved fire retardancy compared to phthalates. The fire performance of PVC, relative to other polymeric materials, is extremely good due to its high halogen content, but the addition of certain plasticisers may impair this property.
Terephthalates	Terephthalates are the other commercial isomeric form of phthalates. Terephthalates are esters of terephthalic acid and include the 1,4 benzenedicarboxilic acid ester often referred to as DEHT (di-(2ethylhexyl) terephthalate) or DOTP (di-octyl terephthalate).	Applications focused on low temperature properties, better resistance to soapy water extraction and lower volatility. In plastisols, DEHT provides lower initial viscosity and better viscosity stability but requires higher fusion and processing temperature.
Triglyceride plasticisers	Different types of glycerol esters have been proposed as alternatives to low phthalates, their limited availability and higher costs currently limit their use.	
Trimellitates	Trimellitates are produced by the esterification of C7-C10 alcohols with trimellitic anhydride (TMA), which is similar in structure to PA with the exception of a third functionality on the aromatic ring. Consequently, esters are produced in the ratio of three moles of alcohol to one mole of anhydride. Common esters in this family are Tris-2-ethyhexyl trimellitate (Tri-octyl trimellitate - TOTM), L79TM, an ester of mixed semi-linear C7 and C9 alcohols, and L810TM, an ester of mixed C8 and C10 linear alcohols.	Due to their low volatility, these plasticisers are used in the automotive industry (dashboard PVC skin produced by slush moulding) and in the insulation or sheathing of electrical cables.
Glycerol Acetylated esters	This plasticiser is made from fully hardened castor oil and acetic acid. Castor oil is extracted from the seeds of the castor oil plant, which is an annual plant grown in India, Brazil and China. The castor oil contains between 85% to 95% ricinoleic acid. The performance of castor oil is improved by modifying its structure (hardening) and replacing the longer chain acids with acetic acid. The resulting fully acetylated glycerol monoester has a lower molecular weight, improving the compatibility and processability of the plasticiser.	Expected main PVC applications for such esters are toys, bottle cap liners, screw cap liners for e.g. jam, teething rings, cling film, tubes and conveyor belts in the food industry and medical equipment.

Source: AFA 2013a

Table D8 Suitability and availability of selected alternatives: summary indicators

Abbreviation	IUPAC Name	Other common names/ acronyms	EC Number	CAS Number	Group [135]	Possible alternative for	Applications	Potential market share [136]	Risk summary [137]	Comparative costs [138]	Availability
ATBC	Tributyl o-acetylcitrate	Acetyltri-n-butyl citrate (ATBC, Citroflex A-4)	201-067-0	77-90-7	Citrate, SP	DEHP, BBP, DBP	Food packaging - cling wrap, toys, medical applications	[redacted]	HH: somewhat lower hazard Env: lower	Price: higher SF: NA [redacted]	Available, in use
ASE	Sulfonic acids, C10-21- alkane, Ph esters	Alkylsulphonic phenyl ester (ASE, Mesamoll)	293-728-5	91082-17-6	Alkyl sulphonic ester of phenol,	DEHP, BBP, DBP	Toys, waterbeds, coated fabrics		HH: potentially lower hazard Env: similar	Price: higher SF: NA [redacted]	Available, in use
DPHP	Bis(2-propylheptyl) phthalate	DPHP	258-469-4	53306-54-0	Orthohthalate, GP	DEHP	Flooring, wall coverings, cladding & roofing, cables & wires, film & sheet, automotive, tubes & hoses, coated fabrics		HH: lower but uncertainty (CoRAP) Env: similar (CoRAP)	Price: similar SF: NA [redacted]	Available, in use
DEHA/ DOA	Bis(2-ethylhexyl) adipate	Di-octyl adipate (DEHA) Di-octyl adipate (DOA)	203-090-1	103-23-1	Adipate, SP	DEHP	Flooring, wall coverings, cladding & roofing, film & sheet, automotive, tubes & hoses, coated fabrics, inks & waxes, food packaging - cling wrap, toys		HH: lower hazard (CoRAP) Env: lower/similar	Price: higher SF: 0.93 [redacted]	Available, in use
DEHT/ DOTP	Bis(2-ethylhexyl) terephthalate	DEHT, Dioctyl terephthalate (DOTP)	229-176-9	6422-86-2	Terephthalate, GP	DEHP	Flooring, Food packaging - Cling Wrap, Toys, Medical Applications		HH: somewhat lower hazard Env: similar	Price: similar SF: 1.03 [redacted]	Available, in use

[135] The following abbreviations are used: GP – general plasticiser, SP – specialty plasticiser.

[136] Responses of downstream users of DEHP (AFA 2013 a). The total exceeds 100% as several alternatives may be competing for DEHP's share in a given application.

[137] The following abbreviations are used: HH – human health, Env – Environment, CoRAP – Community Rolling Action Plan. The latter indicates the substance is currently on CoRAP for evaluation of certain properties: http://echa.europa.eu/information-on-chemicals/evaluation/community-rolling-action-plan/corap-table.

[138] The following abbreviations are used: SF – substitution factor (also referred to as efficiency differential or comparative loading), NA – not available.

Abbreviation	IUPAC Name	Other common names / acronyms	EC Number	CAS Number	Group [135]	Possible alternative for	Applications	Potential market share [136]	Risk summary [137]	Comparative costs [138]	Availability
DIDP	Di-isodecyl phthalate	DIDP	247-977-1	26761-40-0	Orthohthalate, GP	DEHP, DBP	Flooring, cladding & roofing, cables & wires, film & sheet, automotive, tubes & hoses, coated fabrics, inks & waxes	[redacted]	HH: somewhat lower hazard [139] Env: similar	Price: slightly higher SF: 1.1 [redacted]	Available, in use
DINP	Di-isononyl' phthalate	DINP	249-079-5	68515-48-0 and 28553-12-0	Orthohthalate, GP	DEHP	Flooring, wall coverings, cladding & roofing, cables & wires, film & sheet, automotive, tubes & hoses, coated fabrics, inks & waxes		HH: less potent for reprotox but more for liver effects [140] Env: lower	Price: similar SF: 1.06 [redacted]	Available, in use
DINCH	1,2-Cyclohexanedicarboxylic acid, 1,2-diisononyl ester	Di-iso-nonyl-1,2-cyclohexanedicarboxylate (DINCH, Hexamoll)	*605-439-7	EU166412-78-8, USA & Canada 474919-59-0	Cyclohexanoate, GP*	DEHP	Flooring, wall coverings, film & sheet, automotive, adhesives & sealants, tubes & hoses, coated fabrics, food packaging - cling wrap, toys, medical applications		HH: lower hazard Env: lower	Price: higher SF: NA [redacted]	Available, in use [141]
DEHS	Bis(2-ethylhexyl) sebacate	DEHS Dioctyl sebacate	204-558-8	122-62-3		DEHP	food packaging and storage, toys, mobile phones		HH: lower but uncertainty Env: lower or similar	Price: higher SF:	Available, in use
COMGHA	Glycerides, castor-oil mono-, hydro-genated, acetates	Glycerides, Castor-oil-mono-hydrogenated, acetates (COMGHA)	*616-005-1	736150-63-3	Acetylates Glycerol Ester, SP	DEHP, BBP, DBP	Food packaging - Cling Wrap, Toys, Medical Applications		HH: lower hazard Env: lower	Price: higher SF: 1	Available in EU
		Component A	N/A	330198-91-9		DEHP					
		Component B	N/A	33599-07-4		DEHP					

[139] Restricted in toys and childcare articles that could be placed in the mouth by children under entry 52 in Annex XVII of REACH.

[140] Restricted in toys and childcare articles that could be placed in the mouth by children under entry 52 in Annex XVII of REACH.

[141] One major manufacturer of DINCH (BASF) announced in May 2014 the doubling of its production capacity to 200 000 t/y in the EU (Source: AFA opinion – confidential)

Abbreviation	IUPAC Name	Other common names / acronyms	EC Number	CAS Number	Group [135]	Possible alternative for	Applications	Potential market share [136]	Risk summary [137]	Comparative costs [138]	Availability
TOTM/ TEHTM	Tris(2-ethylhexyl) benzene-1,2,4-tricarboxylate	Trioctyltrimellitate (TOTM) Tri(2-ethylhexyl) trimellitate (TEHTM)	222-020-0	3319-31-1	Trimellitate, SP	DEHP	Cables and wires, Film and sheet, Medical Applications	[redacted]	HH: lower hazard Env: CoRAP	Price: higher SF: 1.17 [redacted]	Available, in use
DINA	Diiso-nonyl adipate	Aliphatic dibasic esters	251-646-7	33703-08-1	Adipate, SP	DEHP	Adhesives & Sealants, Food packaging - Cling Wrap, Toys & childcare articles		HH: lower hazard Env: lower	Price: higher SF: 0.98	Available, in use
GTA	Triacetin	Glycerol Triacetate, 1,2,3-Propantriyl triacetate, Triacetin	203-051-9	102-76-1	Glycorol acetyl esters, SP	BBP, DBP	Adhesives, inks, coatings		HH: lower hazard Env: lower	Price: higher SF: NA	Available, in use
DEGD	Diethylene glycol dibenzoate	Oxydiethylene dibenzoate	204-407-6	120-55-8	Benzoate, SP	DEHP, BBP, DBP	Flooring, important substitute for BBP and DBP in non-polymer & spread coating applications		HH: lower hazard (CoRAP) Env: lower (CoRAP)	Price= BBP, ≥DEHP & DBP SF:NA	Available, in use
DGD	Oxydipropyl dibenzoate	Dipropylene glycol dibenzoate	248-258-5	27138-31-4	Benzoate, SP	DEHP, BBP, DBP	Flooring, important substitute for BBP and DBP in non-polymer & spread coating applications		HH: lower hazard (CoRAP) Env: lower (CoRAP)	Price: similar to DEHP SF: 0.98	Available, in use
INBP	1,2-Benzene dicarboxylic acid, benzyl C7-9-branched and linear alkyl esters	C7-C9 alkylbenzylphthalate	271-082-5	68515-40-2	Orthophthalate	BBP	Substitute for BBP in most polymer and non-polymer applications		HH: lower hazard (CoRAP) Env: lower	Effective price is +10% of BBP	Available, in use

Sources: AFA 2013a, ECHA 2013, ECPI 2015, Wilkes 2005, ECHA substance information, http://www.phthalate-free-plasticizers.com/

D.2.3. Summary of risk reduction, technical and economic feasibility, and availability of alternatives

D.2.3.1. Technical feasibility of alternatives

Due to the sheer number of possible alternatives as well as the number of previous assessments, it is impractical to discuss in detail all alternatives. A selection of alternatives to the four phthalates is presented below for the purpose of demonstrating the availability of suitable alternatives for the uses covered in this restriction proposal. It contains the most common alternatives from each plasticiser family that have already taken over a substantial market share of the four phthalates as well as some emerging alternatives or alternatives for niche applications. Table D9 lists these alternatives by chemical name, EC and CAS numbers. The selection is not an exhaustive list: it was prepared primarily taking into account the responses to consultations conducted by applicants for authorisation (AFA 2013a,b,c) and the Danish competent authority for the purpose of the preparation of a restriction proposal on the use of the four phthalates in articles (ECHA 2013a).

Key conclusions with respect to the technical feasibility of the shortlisted alternatives include:

- There are a large number of technically feasible plasticisers, approximately 50 of which are today used commercially. (ECPI 2015)
- The choice of plasticiser depends on the processing technique, the end application of the plasticised material and economic factors. For PVC articles, the two most important factors include cost of the plasticiser and compatibility of the plasticiser with PVC. (AFA 2013a)
- Companies engaged in formulating and compounding activities, as well as those producing flexible PVC articles, have tried and tested a range of alternative plasticisers. Many relevant substances have been investigated for plasticiser characteristics in early research and are already used to a great extent. There is a high degree of familiarity of the applicability of alternative plasticisers to the specific applications of the four phthalates in plastisols and soft PVC articles produced on the EU market (AFA 2013a). (See Table D10 and Table D9)
- Wide variety of technically feasible substitutes for DEHP exists for all uses within this restriction proposal and [redacted] The public consultations for ECHA 2013a, the call for evidence for the preparation of this dossier, as well as the applicants for authorisation did not pinpoint to a particular use of DEHP for which there is no technically feasible alternative. This is supported by other publicly available information, e.g., from a major producer of alternatives: http://www.phthalate-free-plasticizers.com/applications-plasticizers.html
- Technically feasible alternatives for DBP, DIBP and BBP exist for all applications of the substances and they have been fully replaced by EU based article manufacturers, as demonstrated by the absence of applications for authorisations for these three phthalates. [redacted]
- DINP and DIDP have become dominant alternatives to DEHP due to their closeness in performance to DEHP, their availability and their only moderately higher costs. In addition [redacted]

[redacted]

- In addition to these alternatives, a number of other plasticisers already have or have the potential to replace DEHP. (See Table D9 and Table D10) These are primarily non-orthophthalate and also specialty plasticisers which have advantages in particular, in more sensitive (e.g., food contact materials, medical, toys, etc.) applications. Their use has increased among others due to the increasing preference for non-phthalate plasticisers partially brought about by the increased scrutiny of the risks associated with phthalates.
- In some cases, blends of different alternative plasticisers may be needed to attain the desired technical characteristics. This is also a well-known practice with many DEHP uses. More blending may be needed with some of the non-phthalate alternatives to achieve general plasticiser characteristics and some of the marketed plasticiser products consist of several substances, pre-mixed to provide the desired performance characteristics. The products Soft-n-safe[142] and Benzoflex 2088[143] are examples of such mixed plasticiser products (ECHA 2012a).
- Plasticisers such as DINP and DIDP can often replace DEHP without any major process or equipment modifications. Other, in particular some non-phthalate plasticisers, may require reformulation and minor process modifications. The costs of these modifications have been reported to be minor in comparison to material costs. The extensive market experience with a number of alternative plasticisers suggests that the need for R&D is low and likely mainly needed for niche applications (see section D.3.1.1.2).
- Some plasticisers are less efficient than DEHP, i.e., a higher quantity of the alternative plasticiser is needed than DEHP in order to achieve the same softness (see section D.3.1.1), others are similar or better, e.g., DEHT, DEHA (see Table D9).

Table D9 summarises the conclusions of consultations with industry regarding their experience with alternative plasticisers across all possible uses of the four phthalates as well as the capability of alternative plasticisers to replace DEHP.

Table D10 demonstrates the market or production trial experience of industry with alternative plasticisers, such as the non-phthalate plasticisers ASE, ATBC, DGD, DEGD, COMGHA, GTA and TGD. It is evident that already in 2010 (the year the consultations took place), industry had extensive experience also with alternative, specialty plasticisers, across all soft PVC and non-

[142] GRINDSTED® SOFT-N-SAFE by Danisco. The main component is acetylated monoglyceride of 12-hydroxystearic acid and accounts for approximately 83-86% of total product composition (2 isomers, 12-acetoxy-octadecanoic acid 2,3-diacetoxypropyl ester and 12-acetoxy-octadecanoic acid 2-acetoxy-1-acetoxymethyl-ethyl ester). A second major component is fully acetylated monoglyceride of stearic and palmitic acid, approximately 10% of total product composition (4 isomers, octadecanoic acid 2,3-diacetoxy-propyl ester, octadecanoic acid 2-acetoxy-1-acetoxymethyl-ethyl ester, hexadecanoic acid 2,3-diacetoxy-propyl ester and hexadecanoic acid 2-acetoxy-1-acetoxymethyl-ethyl ester). According to Danisco (2011), commercial experience suggests that the product can be used in both 'sensitive' (food contact, medical, toys etc.) and technical areas alike, e.g., flooring, coated fabrics, carpet backing, wire & cabling, wallpaper, general purpose extrusion and calendaring, in particularly any application where internal air quality is of importance.

[143] It is a mixture of dibenzoates DEGD, DGD and triethylene glycol dibenzoate (CAS 120-56-9). It can substitute for fast fusing phthalate plasticisers such as BBP, DBP, and DIBP in vinyl applications, the largest of which is resilient flooring. Most plastisols were formulated with phthalates in mind, so utilizing an alternative chemistry, like benzoates, requires formulation adjustments (Genovique, 2009)

PVC applications of DEHP, DBP, DIBP and BBP. Information from applications for authorisation shows that this experience has expanded further (see Figure D1).

Table D9 Alternatives (beside DINP and other orthophthalates) to DEHP, BBP and DBP proposed by contacted manufactures, by application and with indication of market experience

Application	**ASE**	**GTA**	**DGD**	**Mix of DGD, DEGD, TGD**	**ATBC**	**COMGHA**
Substitute for DEHP						
Polymer applications:						
Calendering of film, sheet and coated products	2	2	4	4	3	3
Calendering of flooring, roofing, wall covering	4	2	3	3		3
Extrusion of hose and profile	2	2	3	3	3	3
Extrusion of wire and cable	2	2	3	3		3
Extrusion of miscellaneous products	2	2	2	2	2	3
Injection moulding of footwear and miscellaneous	?	2	2	2		3
Spread coating of flooring	2	2	2	2		2
Spread coating	2	2	2	2		3
Car undercoating	2		3	3		
PVC medical articles		2			2	
Toy and childcare articles		2			1	
Non polymer applications:						
Adhesives/sealant, rubber	2	2	1	1	2	4
Lacquers and paint	2	2	2	2		4
Printing ink	2	2	2	2	2	3
Production of ceramics						
Substitute for DBP (DIBP)						
Plasticiser in PVC	2		1	1	2	2
Plasticiser in other polymers	2					2
Adhesives	2	2		1	3	4
Printing inks	2	3			2	3
Miscellaneous:						
Sealants	2				3	4
PU foam sealants	2				4	
Nitrocellulose paints	2	3	2	2	2	
Film coatings	3				3	
Glass fibre production						4
Cosmetics						2
Substitute for BBP						
Polymer applications:						
General PVC (e.g. for moulded plastic parts)	2					4
Plastisol coating, for flooring	2		1	1		3
Extrusion or spread coating	2			2		2
Films, calendering	2		4	4		3
Non polymer applications:						
Sealants	2		1	1		
Coatings and inks)		2	1		3	
Adhesives	2			1		
Nail polish					1	

Notes: Interpretation of market experience categories: 1) Main alternative on market. 2) Significant market experience. 3) Examples of full scale experience. 4) Pilot/lab scale experience.
Source: ECHA 2012a

In summary, the information above clearly demonstrates that there are technically feasible alternatives for all applications of the four phthalates in the scope of this restriction proposal.

D.2.3.2. Risk reduction capacity of alternatives

Key conclusions with respect to the risk reduction capacity of the shortlisted alternatives in the latest review of suitable alternatives included as part of the Danish proposal for restriction of the four phthalates in articles (ECHA 2012a) and applications for authorisation for DEHP and DBP submitted to ECHA in 2013 (AFA 2013a,b,c) include:

- In general, the alternatives have more benign human health hazard and risk profile in comparison to the four phthalates, thus, replacement with these alternatives would be beneficial with regards to risks to human health, e.g., ASE, ATBC, DEGD, DGD, DEHT.
- None of the alternative substances have harmonised classification, or meet the criteria for PBP or vPvB, or are identified as SVHC, or are included in Annex XIV.
- With the exception of DINP, none of the presented alternatives exhibit anti-androgenic

effects. DINP has the same anti-androgenic mode of action but is significantly less potent than DEHP, DBP and DIBP (oral DNELs for reproductive toxicity are: for DINP 250 µg/kg bw/day (ECHA 2013a); for DEHP 34 µg/kg bw/day; for DBP 6.7 µg/kg bw/day; and for DIBP 8.3 µg/kg bw/day). A proposal to classify DINP as Repr. 1B has been submitted to ECHA[144].

- DNELs for repeated dose toxicity with DINP and DIDP are higher than the DNELs for reproductive toxicity for the four phthalates and ECHA (2013) concluded that no risks are to be expected from exposure to DINP and DIDP given the existing restriction on toys and childcare articles.[145]
- The applicants for DEHP (AFA 2013a,b,c) concluded that the alternatives have similar environmental effect profiles and comparable PNECs. Thus none of the alternatives would appear to introduce an environmental concern following substitution.

As with any assessment of alternatives, there are some uncertainties regarding the extent to which risks will be reduced following substitution. Some of the alternatives are not REACH registered (hence the body of evidence is limited), some have already raised concerns among the regulators and therefore, may be subject to Substance Evaluation following their listing on the Community Rolling Action Plan (CoRAP). Furthermore, some of the alternatives are subject to restrictions (DINP and DIDP) on their uses impacting vulnerable groups (i.e., in toys and childcare articles) and for others CLP notifications have been provided to ECHA (self-classification by manufacturers, importers and downstream users). (See Table D9 for summary information on risk from the alternatives.)

Overall, it can be concluded with sufficient confidence that alternatives will lead to overall risk reduction for workers and the general population in comparison to continued use of the four phthalates. This is consistent with RAC (2012) concluding that *"it can be assumed that using the alternatives instead of the four phthalates in question will result in an overall benefit."*

A summary of classification and labelling information is included in Table D39 in Appendix D3 to this Annex. The conclusions on the risk reduction capacity on the basis of hazard comparison are included in Table D9.

[144] CLH proposal received from Denmark on 18/11/2015: http://echa.europa.eu/registry-of-submitted-harmonised-classification-and-labelling-intentions/-/substance-rev/11718/term.

[145] Additional information: DINP and DIDP showed liver effects at around 15 mg/kg bw/day in rats with DNELs of 75 µg/kg bw/day (ECHA 2013). Thus the DNELs are higher than the DNELs for reproductive toxicity for the four phthalates and the severity of reproductive effects seen with the four phthalates is considered to be more severe in comparison with effects seen in the liver with DINP and DIDP. NOAELs for repeated dose toxicity with DINP and DIDP appear to be rather similar compared to the NOAEL for repeated dose toxicity with DEHP, but lower than DBP and BBP: DEHP showed kidney toxicity with a NOAEL of 28.9 mg/kg bw/day with a note the NAEL may be lower (EU RAR 2008a); a 90-day study with DBP suggested a NOAEL of 152 mg/kg bw; and a 90-day oral study in rats with BBP suggested a NOAEL at 151 mg/kg bw/day (EU RAR 2004, 2007).

D.2.3.3. Economic feasibility of alternatives

Key conclusions with respect to economic feasibility of the selected alternatives include:

- The price of plasticiser is a key decision criterion for selection of a suitable replacement for the four phthalates in the articles within the scope of this restriction proposal. (AFA 2013a)
- The prices of alternatives which have already replaced a large market share of the four plasticisers are similar to DEHP, e.g., that of DINP and DIDP. Prices of alternatives, such as DEHT, DPHP, and DINCH, which have in recent years began to take more significant market share, are approaching prices of DEHP. [redacted]
- The price of the plasticiser and its efficiency are the main factors that influence the change in the manufacturing costs of articles. As the substitution factors of DINP and DIDP in comparison to DEHP are on average 1.06 and 1.1 respectively (Wilkes 2005), it is thus anticipated that the transition to alternatives, including assuming substitution of the four phthalates with the most common alternatives takes place, will likely lead to additional material costs. However, these are anticipated to be minor in comparison to the total price of the articles. (See section D.3.7.1.) In addition, at present there are some alternatives, including general plasticisers, whose price and efficiency appear to be better or very similar to those of DEHP, [redacted] See section on Substitution costs under Economic impacts for further details.
- The public consultation has revealed that the plasticiser market is highly aggregated and that aggressive pricing strategies can be pursued by manufacturers in such markets. This situation applies to the four phthalates and its alternatives. For example, the market for DEHP in the EU is also highly aggregated with only two manufacturers applying for authorisation, one of which has already announced their intention to manufacture an alternative. [146]
- Prices of alternatives to DBP, DIBP, and BBP are difficult to obtain but ECHA 2013 showed that they are 5-15% higher than the three phthalates. DBP, DIBP, and BBP have been fully phased out in the EU by 2015. This suggests that the transition to their alternatives is not very costly.
- R&D, reformulation, process and plant modification (herein referred to in a total category as RDRPPM) costs are reported in ECHA 2013 to be relatively small in comparison to material cost (and have declined since, given the increased experience with alternative plasticisers). In addition, it has been reported that plasticiser manufacturers often conduct R&D and trials in various applications to promote alternative plasticiser sales. Thus, these R&D costs are already reflected in the price of the plasticiser. (See section on R&D, reformulation and process and plant modifications under Economic impacts for further information.)

It should be noted that pricing information is often confidential. Prices are a function of availability of low-cost feedstock (mainly alcohols), other raw materials and energy costs,

[146] More suppliers emerging for DOTP http://blog.phthalate-free-plasticizers.com/2012/01/05/more-suppliers-emerging-for-dotp/

complexity of synthesis, as well as overall supply (existing production capacity) and demand conditions. Thus, price differences between the four phthalates and their alternatives can vary across regions. For example, the current low utilisation of the DEHP production overcapacity may be a cause for the price differential to be larger on some Asian markets. The price differential could also further increase in the short term in regions where DEHP is dominant in the event regulatory pressures lead to a surge of demand for alternative plasticisers. However, prices of alternative plasticisers even in these regions can be expected to decrease as production capacity and competition increases. Different alternatives are however manufactured from different raw materials, and involve more or less complex and resource demanding chemical synthesis technologies. This of course sets limits to the minimum prices attainable even in a mature market, and some of the alternative plasticisers described may remain at higher price levels. However, as the function of all alternatives is the same, their prices are expected to equalise in the long run and the price differential to represent only the difference in quality they impart in the final product, efficiency, the need for RDRPPM, and customer preferences.

Table D9 presents summary indicators related to the evaluation of the economic feasibility of alternative plasticisers.

In summary, it has been demonstrated that a number of alternatives exist at a similar price level as the four phthalates for all uses in the scope of this restriction proposal. The transition to some of the alternatives may lead to additional costs primarily due to efficiency differences (with DEHP). These are estimated to be relatively small, and are anticipated to have minor impact on the final price of the articles.

D.2.3.4. Availability of alternatives

The alternatives profiled in this report are available and already in use. Key conclusions with respect to the availability of alternative plasticisers include:

- The main alternatives (including those considered in the substitution scenario of the proposed restriction) are produced in the EU28 and internationally.
- Production capacity of non-phthalate plasticisers has been increasing in Europe and internationally.
- Given the small tonnages of the phthalates to be substituted in EU manufacturing of articles and the availability of variety of alternatives, it is unlikely that in the event the restriction comes into force, shortages and price pressures would be experienced on the EU28 market. [redacted]
- Several non-EU markets are dominated by DEHP. Given the small tonnages of phthalates to be replaced, it is unlikely that even on these markets the entry into force of the restriction would create shortages and price pressures for alternative plasticisers in the medium and long-term. This is taken into account in the determination of the transitional period of the proposed restriction. [redacted]

[redacted]

D.2.3.5. Alternative materials

Substitution of phthalates may take place by substituting PVC-material by other materials that do not need to be plasticised with phthalates. In a review of various studies on alternative materials to plasticised PVC, it is demonstrated that for many applications of PVC, alternative materials exist at similar prices. Many of the materials seem to have equal or better environmental, health and safety, performance and cost profiles, but clear conclusions are complicated by the fact that not all aspects of the materials' lifecycle have been included in the assessments. (ECHA 2012a) It must also be mentioned, that the alternative materials also vary in terms of their properties that may give advantages or disadvantages in particular applications (end products), e.g., longevity, light weight, resistance to tear/breakage/humidity, etc.

Transitioning to alternative materials is associated with larger production changes, which could be considered higher than the costs associated with established alternative plasticisers for soft PVC. This could represent in some cases a substantial departure from the core business of the article manufacturers. [redacted] The fact that PVC with plasticisers is more likely to be replaced with PVC with other plasticisers is also indicated by the fairly stable consumption of plasticisers over time. Consultation with industry also suggested that the replacement of the four phthalates in EU manufacturing in particular has not resulted in market shift to other materials, and there is no reason to expect that the replacement of the four phthalates in imported articles would result in such substitution. The driver for substitution of PVC with alternative materials is to avoid halogens, e.g., in means of transport and in buildings.[147] (Larsen, pers. com.)

Although it can be anticipated that alternative materials could take over some of the share of the soft PVC articles in the event the four phthalates are restricted, it is assumed likely that the majority of existing article manufactures will transition to an alternative PVC plasticiser. Thus, this dossier focuses on these scenarios for the purpose of the estimating the regulatory impacts on industry. Alternative materials are only referenced for the purpose of completeness. The table below lists some possible alternative materials per application. Further information on alternative materials is available in section C.14.6 Alternative flexible polymers in the Danish restriction proposal submitted in 2011 (ECHA 2012a).

Table D10 Alternative materials to plasticised PVC

Alternative materials	Examples of main applications
Ethylene-vinyl-acetate (EVA)	Toys, hoses

[147] Halogen-free materials are marketed for use is buildings and means of transport in order to prevent the formation of toxic PCDD/PCDF (polychlorinated dibenzodioxins and polychlorinated dibenzofurans) and hydrocoric acid (HCl) in case of fire.

EPDM (ethylene propylene diene monomer) rubber	Hoses
Polyethylene (PE)	Toys
Polypropylene (PP)	Toys, office supplies
Elastomeric/thermoplastic polyolefins compounds (TPO or TPE-O compounds)	Film, tube and moulded medical and potentially non-medical uses*
Cardboard and paper	Office supplies
Leather	Shoes, office supplies
Polyurethane (PUR)	Waterproof clothing, shoes, boots
Nylon	Shoes
Neoprene rubber	Boots
Natural rubber	Shoes, boots
Wood	Furniture, flooring, wall covering**

Sources: ECHA 2012a, DEPA 2001, *Melitek 2015 and Saint Gobain 2015,**ECHA

D.3. Restriction scenario(s)

The following sections examine the impacts of the proposed restriction as well as its effectiveness, practicality and monitorability.

Proposed restriction

Brief tittle: Restriction on articles containing the four phthalates for: i) indoor use and ii) outdoor use, if in contact with human skin or mucous membranes.

The full definition of the proposed restriction is presented in section D.1.1.

Special considerations on FCMs: While FCMs contribute substantially to human health risks from the four phthalates, it was concluded that the best course of action is to derogate FCMs in the proposed restriction on the grounds that a sector-specific legislation would lead to a more efficient use of regulatory resources as well as improved clarity to stakeholders. However, this proposal highlights the need to address the risks associated with exposure from the four phthalates under the procedures outlined for Regulations (EC) No 1935/2004 and Commission Regulation (EU) No 10/2011, ensuring that a combined assessment of risks is factored into the decision-making.

D.3.1. Economic impacts

D.3.1.1. Substitution costs

Substitution costs are the costs article manufacturers will incur due to transition to alternatives in the event of a restriction on the four phthalates. According to previous studies, which draw on consultations with industry, these costs would consist primarily of material costs. Other substitution costs, such as R&D, reformulation, process and plant modifications (RDRPPM) and other costs, are reported to be minor in comparison (ECHA 2012a, ECHA 2013).

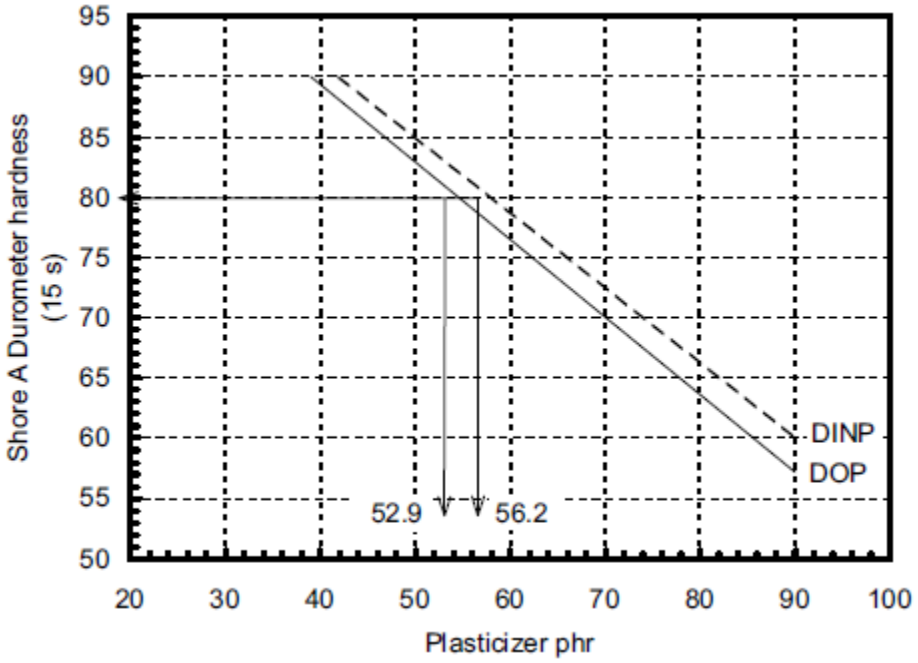
D.3.1.1.1. Material costs

Industry would bear additional material costs in the event the proposed restriction enters into force. These costs are driven by price and efficiency differences between the four phthalates and their alternatives.

Efficiency differential (comparative loading)

In simple terms, the efficiency differential, also referred to as efficiency or comparative loading factor, shows how much more of an alternative plasticiser needs to be added instead of DEHP (used as a reference) in order to achieve the same softness. For example, according to Wilkes 2005, about 6% more DINP needs to be added than DEHP (see Figure D2). Other plasticisers have similar or higher efficiency as compared to DEHP, e.g., DEHT, DEHA. See Table D9 for a summary of the comparative loading factors of selected common alternatives to the four phthalates.

Figure D2 Durometer A hardness of DINP vs. DEHP (DOP)



Notes: Parts per hundred resin (phr)
Source: Wilkes 2005

Price differential

The substitution costs are highly dependent on the prices of alternatives. Price was also identified by downstream users as a key selection criterion of a plasticiser (AFA 2013a).

Plasticiser prices are driven by feedstock prices, e.g., phthalic anhydride and oxo-alcohols, such as n-butanol (NBA) used in the manufacture of DBP, isononyl alcohol for DINP, 2-ethylhexanol (2-EH) for DEHP and DEHT (reacted with either dimethyl terephthalate or terephthalic acid). Prices are also influenced by regional as well as global supply and demand conditions for individual plasticisers. Demand for plasticisers is highly dependent on demand for PVC and its end-use in articles, the demand for which tends to grow on average at about the same rate as GDP, although the possibility of expanding the use of PVC to new applications leads to additional growth. The PVC market, as well as the markets for many end-use PVC applications, is characterised as a highly competitive, low margin market. This influences the ability of suppliers upstream, including suppliers of plasticisers, to pass on increasing costs to their customers. Demand for plasticisers is also influenced by regulatory actions, e.g., the most recent being RoHS in the EU (2015) and DINP's addition to the list of substances with restricted use by the state of California, USA (2013).

As shown in Table D12 below, the average prices of DINP and DIDP on many geographic regions are about the same or slightly higher than that of DEHP, while those of non-phthalates tend to be higher. The price differential between DEHP and its most common alternatives such as DINP, DEHT, DPHP, and DIDP has been shrinking on most markets. It is however possible that the price gap between DEHP and its alternatives is slightly larger in the medium term on the Asian market, as there, and particularly in China, a substantial DEHP capacity expansion has recently occurred (BASF 2011).

An abatement cost study estimated that the costs of the main alternatives of DBP, DIBP and BBP (which could include benzoates and terephthalates) were higher by about 5-15% depending on the application (ECHA 2013). This information is dated, however, more precise pricing information of their alternatives is difficult to obtain to better estimate the price difference. Since 2015, all uses of these three phthalates are assumed to have been replaced in the EU, as no applications for authorisation for these three phthalates for articles in scope have been filed prior to their sunset date (21 February 2015).

Furthermore, based on past pricing trends, it can be assumed that in the long term, e.g., for the temporal scope of this analysis of 20 years following the entry into force of the restriction, the price differential between the four phthalates and their least cost alternatives would be based primarily on their comparative loading[148] as prices of less efficient alternatives would have to be lower in order to be competitive on the plasticiser market.

[148] This assumes that there are no other influences on the quality of the goods produced with different plasticisers. These are considered a different category of potential abatement costs.

Table D11 Prices of selected plasticisers

Substance	FC NWE* 09/2012		FC NWE* 09/2013		FC NWE* 11/2013		FC NWE* Average 11/13-11/14		
	European price range in €/tonne	Relative price to DEHP	European price range in €/tonne	Relative price to DEHP	European price range in €/tonne	Relative price to DEHP	11/2013 (€/tonne)	11/2014 (€/tonne)	Relative price to DEHP
DEHP	[redacted]								
DPHP									
DEHA/ DOA									
DEHT/ DOTP									
DIDP									
DINP									
TOTM/ TEHTM									

Substance	US$/kg		USc/lp Av 11/13-11/14			Asia** US$/t Av 11/13-11/14			FC NWE* 06/2015	
	Late 2012	Relative price to DEHP	Nov 2013	Nov 2014	Relative price to DEHP	Nov 2013	Nov 2014	Relative price to DEHP	European price range in €/tonne	Relative price to DEHP
DEHP	[redacted]									
DPHP										
DEHA/ DOA										
DEHT/ DOTP										
DIDP										
DINP										
TOTM/ TEHTM										

Notes: *FD - free delivered, NWE - Northwest Europe **CRF, east Asia – cost and freight
Source: ICIS 2015, IHS 2013

D.3.1.1.2. Research & Development, Reformulation, Process & Plant Modification costs

R&D, reformulation, process and plant modification (RDRPPM) could be necessary when it is required to replace the plasticiser used in the manufacturing process, in this particular case instigated by a potential entry into force of the proposed restriction. In general, these are costs that vary greatly depending on the plasticiser used, the type of applications, the number of applications and unique performance requirements, the process and equipment utilised, etc. They are one off costs per manufacturer, which make them difficult to estimate for imported articles as there is high uncertainty related to the number of entities affected, the number of heterogeneous uses each of them has and their unique requirements.

Previous studies have shown that RDRPPM costs are minor in comparison to material costs, as some of the main alternatives, such as DINP, DIDP, DPHP, DEHT and DINCH are reported drop-in alternatives to DEHP, i.e., requiring minimal plant and process adjustments (ECHA 2013). Similarly, INBP and DGD have been reported for BBP and dibutyl terephthalate (DBT) for DBP/DIBP (ECHA 2013).

Previous consultations with industry, for the preparation of ECHA 2012a and ECHA 2013 for example, did not provide information on potential RDRPPM costs. No such information was also received during the call for evidence circulated to assist with the preparation of this restriction proposal (ECHA 2015a), which was also circulated to EU international trade partners via the World Trade Organisation (WTO).

The long-term experience with substitution of DEHP as well as DBP, DIBP, and BBP domestically and internationally places these costs under significant uncertainty. As article manufacturers often produce more than one article type, it is likely that many of them have already faced the need to comply with restrictions on the four phthalates under already existing EU regulation (e.g., entry 51 and 52 of Annex XVII of REACH or the Toy Safety Directive) or other jurisdictions.

Furthermore, including RDRPPM costs in the compliance cost calculation bears the potential for double counting as they may already be (partially) reflected in the market price differential of alternative substances. This is so because:

- There are reports that plasticiser manufactures and formulators of compounds, plastisols and dry-blends often invest in R&D to support the marketing of their products to downstream users. The R&D costs would be included in the unit cost of the plasticiser (or plastisols) sold.
- Plasticisers associated with higher article production costs, due to e.g., necessary RPM by downstream users, would obtain a lower price on this competitive market of close substitutes (unless they possess superior characteristics with respect to the production process, end product quality, product “green” image, etc.).

Non-phthalate plasticisers require more blending with other plasticisers, therefore, more trials (and thus, higher R&D costs) may be required to identify the right plasticiser mix for end-products. Alternative specialty plasticisers currently used in limited applications are also more likely to require some process modifications. However, as stated previously, there are low cost, drop-in alternatives on the market. Thus, the transition to non-drop-in alternatives, may also

be associated with other trends (e.g., responding to consumer preference for non-phthalate articles) and not directly with the potential impacts of the restriction on the four phthalates.

RDRPPM are also highly uncertain on markets currently dominated by DEHP, as also there non-DEHP plasticisers already hold a significant market share: about 50% of worldwide plasticiser use (ECPI 2012). This is also characteristic of the Chinese market, for example, where the substantial use of DEHP is expected to continue in the foreseeable future (TOC 2012). [redacted]

Therefore, it can be concluded that given the significant market penetration of the alternative plasticisers, RDRPPM costs are anticipated to be negligible. These costs may be possible for some niche applications where DEHP is currently dominant; however, no such information has been received during the public consultations for the preparation of this (ECHA 2015a) and the previous restriction proposal (ECHA 2012a). Furthermore, the availability of drop-in alternatives to the four phthalates, even in international markets where their use is currently dominant, suggests that RDRPPM costs would also have minimal effect on the average prices of articles originating from these geographic regions.

These conclusions are consistent with cost estimates prepared for the existing restriction entry #51 in Annex XVII of REACH (EC 2000) and in the previous restriction proposal on phthalates (ECHA 2012a). No reformulation and RDRPPM costs were taken into account already in 2000 in estimating the impacts of a possible restriction on phthalates in toys and childcare articles due to the extensive experience with substitution. The estimates assumed substitution of phthalates with non-phthalates plasticiser ATBC. Similarly, the previous restriction proposal on phthalates (ECHA 2012a) also did not include RDRPPM costs in the estimation of the restriction costs, reflecting industry's high degree of familiarity with substitution of DEHP, DBP, DIBP and BBP. An additional 15 years of experience of substitution since EC 2000 and five years since ECHA 2012a has certainly made industry even more familiar with the available alternatives. Therefore, any RDRPPM costs to be borne by EU or international companies as a result of the proposed restriction are considered highly uncertain and if any, they are likely low, anticipated to have minimal impact on the average prices of articles within the scope of this restriction proposal.

D.3.1.1.3. Scenarios for the estimation of substitution costs

The scenarios employed in the estimation of the substitution costs society would bear due to the proposed restriction are described in this section. The scenarios take into account the factors influencing the substitution of the four phthalates and past market trends. They are formulated on the basis of the following main assumptions:

a) Main alternatives selected: DINP, DIDP, DEHT (or similar) and benzoates, terephthalates (or similar)

Previous studies have often selected DINP and DIDP as the main substitutes for DEHP for the purpose of estimating the compliance costs associated with the introduction of a restriction on its content in articles. It has been shown that DINP is technically feasible to substitute DEHP in the majority of its PVC applications (ECHA 2012a, AFA 2013a, also see Table D9), while DIDP

has particular technical advantages in wires and cables as well as automotive applications (including car interiors) due to its heat stability and electrical insulation properties (ECHA 2012a, ECPI 2015). However, other non-orthophthalate plasticisers, such as DEHT are gaining market share.

For the purpose of this analysis, it is also assumed that DINP, DIDP and DEHT (or a similar alternative) are going to substitute DEHP in all its uses, while DBP, DIBP and BBP are assumed to be substituted with a mix of plasticisers such as benzoates and terephthalates. This scenario was selected on the basis of technical feasibility criteria and demonstrated market trends, including historical price differences and supplier capability to provide alternatives in the necessary quantities (see Table D9, Table D10 and Table D12). It is seen to illustrate well the likely actions of industry in the event of a restriction as:

- Previous studies have shown that it is unlikely that one plasticiser would replace DEHP in all its current uses. Market reports have shown that the four phthalates are increasingly substituted by a large spectrum of plasticisers. This is likely due to increasing preference for non-orthophthalate plasticisers spurred by discussion in the public domain of the (potentially) hazardous properties of orthophthalates, including those with higher molecular weight.
- DINP has been the preferred substitute to DEHP but the market share of non-phthalate plasticisers has been increasing, influenced among others by the increased regulatory attention on phthalates.

- The prices of these selected alternatives are reported to be similar to DEHP (see Table D12); however, it is important to note that there are alternatives on the market with lower cost-efficiency substitution rates than DINP and DIDP. These are covered by selecting DEHT (or a similar plasticiser) to substitute part of DEHP's market.
- European production capacity of DEHT is expanding: In 2012, Zakłady Azotowe Kędzierzyn S.A. (a.k.a., ZAK, one of the applicants for authorisation) announced their

intention to produce DEHT.[149] In 2014, Oxea announced an increase of its European capacity by 50,000 tonnes by end of 2015 "in order to meet strongly growing customer demand".[150] Confidential information suggests that there are other plans for DEHT capacity expansion on the EU market [redacted]

- DBP, DIBP and BBP are assumed to be substituted with a mix of plasticisers such as benzoates and terephthalates. This approach is similar to the approach selected in ECHA 2013 for the purpose of estimation of the abatement costs curves for these substances.

b) Substitution (material) cost differences

- The costs of substituting DEHP with the main alternatives on the European market are anticipated to be driven primarily by their comparative to DEHP efficiency, as currently, their prices are similar. This is considered reasonable, as in the long-run the prices of less efficient plasticisers would decline in order to remain competitive on the market.[151]
- The price of the main alternatives of DEHP on non-EU markets is anticipated to be driven in the long run by their comparative to DEHP efficiency as well as their prices, which are assumed to be about 5% higher than the price of DEHP (taking into account historical price reports on non-EU markets, see Table D12). This suggests that in total the costs of substituting DEHP in imported articles would be approximately 8-16% higher than those of DEHP, depending on the selected alternative.[152] While it is fair to assume that in the long run the prices of less efficient plasticisers would decline in order for them to remain competitive on the market (whereby the substitution costs of imported articles would also begin to approach zero), this scenario was selected to reflect that, in the short to medium term, other minor costs, such as RDRPPM, could be incurred on markets where currently DEHP is dominant.
- The costs of benzoates or similar are assumed to be 10% higher than those of DBP, DIBP and BBP. This is in line with the mid-point estimate in the ECHA 2013 abatement cost study. This information is dated but due to the low volume use of these phthalates, more accurate public information is not available. This represents the best publicly available information.

Therefore, the scenario summarised in Table D13 was selected as the main scenario for the estimation of the substitution costs society would incur in the event the proposed restriction enters into force. Public consultation comments affirmed that the selected alternatives in Table D13 are the major alternatives for DEHP.

[149] More suppliers emerging for DOTP http://blog.phthalate-free-plasticizers.com/2012/01/05/more-suppliers-emerging-for-dotp/

[150] Promising future for DOTP http://blog.phthalate-free-plasticizers.com/2014/03/24/promising-future-for-dotp/; Oxea plans capacity increase for plasticizer Oxsoft GPO http://blog.phthalate-free-plasticizers.com/2014/03/24/oxea-plans-capacity-increase-for-plasticizer-oxsoft-gpo/; http://www.oxea-chemicals.com/uploads/tx_nfoxcnews/140324_EN_OXEA_DOTP_Expansion.pdf

[151] This ignores any potential quality impacts of the end-product due to specific plasticiser characteristics.

[152] Calculated equal to comparative loading times the price difference.

Table D12 Substitution of DEHP, DBP, DIBP, and BBP - summary assumptions

Alternative plasticisers	Uses of DEHP, DBP, DIBP & BBP to be replaced by alternatives	Comparative loading*	Price Differential**	
			Domestic	Imports
DINP	55% of all DEHP uses	1.06	1	1.05
DIDP	15% of all DEHP uses	1.1	1	1.05
DEHT/DPHP/similar	30% of all DEHP uses	1.03	1	1.05
Benzoates/similar	All uses of DBP, DIBP & BBP	1.1		

Notes: * Assumed difference in the required tonnage of the alternative in comparison to DEHP (and the other three phthalates)
** Assumed difference in the price of the alternatives in comparison to the price of the four phthalates.

As discussed in earlier, there are other alternatives with similar (or better) technical and economic characteristics which also have more benign risk profile than the selected alternatives. This is the scenario that appears to be most likely on the basis of publicly available information. Two additional scenarios are described in Annex E. These scenarios give an indication of the ranges of the substitution costs also on the basis of justifiable assumptions in the public domain. The confidentiality of information was one of the major deterrents to presenting more realistic substitution cost scenarios.

It is important to highlight that for the purpose of estimating the costs of the restriction the alternatives were selected on the basis of convenience due to less confidentiality issues related to the critical data required for the analysis. Therefore, the described scenario is believed to adequately illustrate the anticipated restriction costs (equal to substitution and other social and economic costs) as while the choice of alternatives influences them, it was shown above, that there are a number of alternatives the transition to which would likely lead to similar and even lower costs than these selected.

D.3.1.1.4. Total substitution cost - conclusion

On the basis of assumptions presented in Table D13 and the tonnages of the four phthalates in articles in scope projected in Annex C: Baseline, the substitution costs are estimated to €15.8 million annually from 2020 (the year of the assumed entry into force of the proposed restriction) onward.[153] They are calculated as the product of the price differential and comparative loading multiplied by the projected tonnes of the four phthalates. The assumed base price of DEHP is nearly €1 500 per tonne, while that for the other three phthalates - approximately 15% higher than DEHP's.

The substitution costs are considered an overestimate because:

- Confidential information implies that the least cost scenario for estimating material costs is closer to the lower end of their range: €8.4 million annually. (see Annex E). [redacted]

[153] 2014 was selected as the base year for the purpose of the analysis. All values are discounted or adjusted with CPI (EuroStat consumer price index) to 2014.

[redacted] This is a feasible least cost alternative scenario because:

- The alternatives to DEHP are technically feasible according to their manufacturers [redacted] their effective pricing (efficiency and price differential) appears to be very close to DEHP's (IHS 2013, AFA 2013a, also see section D.3.1.1.3.a)); they are apparently available in the required volumes to comply with the proposed restriction in the EU and internationally (IHS 2013).
- The assumption for the effective price differential between DBP, DIBP and BBP and their main alternatives is in line with the mid-point estimate in the ECHA 2013 abatement cost study. This information is dated but due to the low volume use of these phthalates, more accurate information is not available. The lack of applications for authorisation for their uses in articles in the EU suggests that the substances are fully phased out as of their sunset date in 2015. This implies that the substitution costs for these phthalates are lower than €40 000 per year and per substance; otherwise an authorisation would have been pursued.[155] [redacted]

- The analysis assumes for simplicity purposes that the base price for DEHP is the same globally. There is indication that the prices on the Asian market (where many of the imported articles originate from) are somewhat lower than on the EU market. Assuming a lower base price for DEHP (as shown for the Asian market in Table D12), the average annual material costs of the restriction would be less than €12.5 million.
- The estimates assume that the price and efficiency differences would exist throughout the selected study period of 20 years, while these would likely decline and approach zero in the long-run. This is because, in the long-run, the effective price differences between plasticisers are expected to disappear as the market would not be willing to pay a higher price for a plasticiser which is less efficient, unless the plasticiser offers other benefits such as improved end-use product for example.[156]
- The non-quantified RDRPPM costs are shown to be negligible and likely approaching zero in the long run as no plasticiser could obtain a higher price in a competitive market if it requires higher up-front costs. Therefore, including when accounting for the uncertainty regarding the value of RDRPPM costs, the estimated total annual substitution costs of €15.8 million as a result of the entry into force of the restriction are considered an overestimation.
- Lastly, the analysis assumes that all substitution costs for transitioning to the alternatives of imported articles (close to 97% of the €15.8 million annually) are fully passed on to EU entities (EU buyers or end-users) and are therefore, costs of the restriction to EU society. Given the high price competition on some article markets, this assumption is associated with considerable uncertainty. It is foreseeable to assume that

[redacted]

[155] Assuming average price of authorisation application of €225,000, one applicant per substance, and seven years of average authorisation period. The assumptions also ignore potential non-economic benefits of not applying, such as a "green image" of their products, i.e., not containing SVHCs.

[156] Such quality improvements are recognised but assumed negligible and discussed separately from substitution cost impacts for the purpose of simplifying the analysis. See section on Impacts on the quality of the good.

some of the substitution costs would be borne by international article manufacturers or other entities of the non-EU supply chain. This would likely lead to impacts on profits in non-EU jurisdictions.

D.3.1.2. Testing costs

Testing costs can be incurred by industry to ensure and self-monitor the compliance with a restriction measure. They include:

- the costs of laboratory tests in the importer, manufacturer or supplier's own laboratory or in independent, third party laboratories, as well as
- the price of the article tested as many of the tests of the phthalate content of articles are destructive tests.

There has been a debate in the past whether testing costs are true abatement (compliance) costs associated with a regulatory measure due to their voluntary nature, i.e., there is no regulatory requirement for testing of articles by industry to demonstrate compliance. Instead, the regulatory environment is such that enforcement authorities could require testing in the event compliance needs to be demonstrated on a one off basis. As stated in ECHA's case studies on abatement cost estimation (ECHA 2013), in a voluntary context where industry is not explicitly required by regulation, there is no obvious need for testing and testing costs should not appear in the abatement cost curve. On the other hand, if there are incentives to continue to use a restricted substance and it cannot easily be observed whether it is being used or not, testing might be needed to ensure regulatory abatement measures are being adopted upstream the supply chain (and to prevent any negative consequences such as litigation or fines). In this case, the costs of this monitoring should be included in the abatement cost estimates. Thus, ECHA 2013 concluded that the inclusion of testing costs in the abatement cost estimation is dependent on the context and should be examined on per case basis.

Testing costs have been estimated in recent regulatory actions.[157] However, these estimates often have been subject to high uncertainties due to the great variability of practices of companies to conduct testing, related to their frequency of testing (the number of articles tested annually) over time as well as the cost of testing to be allocated to a specific regulatory action (due to the applicability of tests to more than one substance, or to more than one regulatory action). Further uncertainties are introduced when the substances subject to the proposed restrictions are already subject to regulatory measures. This is because it is often ambiguous what portion of the testing costs could be attributed to the new measure only.

[157] For example, in the restriction proposal on NPE in textile articles (ECHA 2014a), testing costs were considered highly uncertain by SEAC and were not taken into account in the main scenario for assessing of the efficiency of the restriction. The case is considered similar to the four phthalates restriction not only because of the uncertainties related to the frequency of testing of articles but also because of uncertainties related to the percentage of incremental testing costs anticipated to be instigated by the entry into force of the NPE restriction. The latter was due to NPE's inclusion on the Candidate list shortly before the assessment of the restriction proposal.

The sections below attempt to address these unresolved methodological/conceptual issues by discussing the relevance of testing costs for the estimation of the total restriction costs and estimate the value of these costs. The sections draw on the results of a survey of compliance control strategies undertaken in preparation of the restriction proposal (see Annex F).

Relevance of testing costs for the estimation of compliance costs

a) Existing obligations and voluntary actions to ensure compliance control

As DEHP, DBP, DIBP and BBP are SVHCs and are included on the Authorisation list (and prior to that, on the Candidate list), actors in the supply chain have existing obligations under REACH:

- Under Article 7(2), “producers and importers” of articles must notify ECHA if an SVHC is present, totalling over one tonne per producer or importer per year, in a concentration higher than 0.1% by weight;
- Under Article 33, “suppliers” of articles containing an SVHC, in a concentration above 0.1% by weight, need to inform the recipient of the article, and to provide similar information in response to consumer enquiries within 45 days;
- Under Article 66, downstream users who use a substance under authorisation to an actor up his supply chain need to notify ECHA within three months of the first use of the substance.

Similar information requirements exist under other EU legislation as well as the legislation of other non-EU jurisdictions.[158]

Furthermore, as a result of a number of regulatory actions in the EU and internationally, and the ongoing public debate on the human health risks associated with exposure to phthalates, a number of companies have introduced voluntary measures on the four phthalates, under their own internal policies or under certification mechanisms (ecolabels) such as the Nordic Swan and Oekotex. Examples of the former include: restricting the use of substances on the Candidate list (e.g., MS 2014) or substances with specific classifications such as R60 or R61 (e.g., BCR 2015) or altogether imposing bans on use of PVC, e.g., in all articles (e.g., Greenpeace 2001) or only in packaging or FCMs (e.g., BCR 2015).[159]

These regulatory and voluntary industry actions have precipitated the need for industry to generate knowledge (information) on whether the four phthalates are used in their products. Discussions with industry have revealed that REACH has assisted in this respect and thus, has reduced the need for testing as the first course of action of ensuring compliance (pers. com. Toy Industries of Europe).

[158] For example, a declaration of compliance is required under the FCM legislation to exchange information between suppliers and customers at marketing stages up to but excluding the retailer to enable the customer to establish or confirm compliance of the FCM with the relevant legislation (EC 2013).

[159] Another example is Marks & Spencer limits phthalates and all other esters of ortho-phthalic acid (including DEHP, DBP, DIBP and BBP) to a combined limit of 250 ppm. It also specifies 1000 ppm combined total of the 6 legislated phthalates with 500 ppm maximum for each phthalate in the finished article (MS 2014).

In the recent survey of compliance control, 13 out of 19 respondents (the majority of which were large companies) answered that they already had to comply with some restrictions on the four phthalates beyond the EU and national restrictions. Ten (out of 19) answered that the listing of the substances on the Candidate List has had impacts on their avoidance of the substances in articles and several answered that they request articles without the four phthalates as a consequence of the listing. Therefore, it is highly uncertain whether the introduction of a restriction measure on the four phthalates will result in additional costs for these companies. The results are consistent with those examining compliance costs of NPE in textiles, where the fact that NPE was on the Candidate list also created pre-existing information requirements to the introduction of the restriction on NPE (ECHA 2014a). As by definition compliance control costs include only additional (incremental) costs due to the introduction of a new regulatory measure, not all costs related to testing or other information generating requirements of the phthalate content in articles can be associated with the proposed restriction on the four phthalates. Due to the low response rate to the survey despite active promotion, it is difficult to estimate with confidence what percentage of industry will incur incremental costs as a result of the proposed restriction on the four phthalates in articles.

b) Strategies industry uses to ensure compliance

The recent compliance control survey examined to what extent industry employs the following strategies to ensure their articles comply with EU regulations:

- Contractual procedures, i.e., explicitly specified in the purchase contract requirement for the international supplier to comply with EU legislation and/or the importer's internal chemical policies.
- Provision of information to suppliers regarding the requirements to meet EU legislation or the importer's chemical policies.
- Monitoring and control procedures. These are varied and may include:
    - a requirement the supplier to sign a declaration of compliance and/or to provide test documentation,
    - spot tests, audit/supervision, etc. carried out by an EU buyer (importer) to their international suppliers.

The survey revealed that contractual obligations in combination with information provision are the most frequent compliance management strategies used by respondents. This is consistent with previous surveys on compliance control costs, e.g., ECHA 2014a. Compliance testing is less frequently used and the majority of respondents rely on the provision of a declaration of compliance (with EU regulations or a list of restricted substances prepared by buyers). Less than 20% of the respondents conduct their own testing and about 20% require suppliers to provide testing documentation for all their shipments (which could very well be used for more than one buyer over the long period of time).[160] A limited number of respondents employ this strategy more frequently for new suppliers/articles or for point control (spot tests). In addition, the spot checks or requirements of suppliers to conduct tests are often dependent on the buyer's risk assessment of the suppliers, e.g., unknown suppliers or suppliers with track record of non-compliance are subject to higher frequency of tests (e.g., BCR 2015). The survey results (as well as the pre-survey interviews) imply that companies have very diverse practices

[160] For example, Marks & Spencer suggest testing articles at the following stages of manufacturing: preproduction, first bulk, production (once per style per season). (MS 2014)

when it comes to testing; therefore, it is very difficult to make further generalisations about the frequency and selection procedures for testing that could be applied to diverse lists of articles.

The pre-survey interviews also revealed that smaller companies often just define that the articles supplied should be in accordance with European and national legislation, or that imported articles should be in accordance with REACH, without further specifications, including requirements for testing. The conclusions were also supported by other stakeholders who mentioned that industry requires declaration of compliance first and foremost and some companies perform spot tests (pers. com. EuroCommerce).

Other reports from industry also support the conclusion that testing is not necessarily the only or the first/most common strategy employed by industry, given the large amount of information generated about the possible and quantified presence of the four phthalates in articles as a result of REACH. In an article regarding the new European Court of Justice ruling on the notification of SVHCs in articles (CW 2015), *some industry observers say this* [testing] *will not always be necessary, and it should often be possible for article importers to get the information they need on SVHCs from their suppliers*.

Conclusions

In summary, the following can be concluded regarding testing and other compliance control costs to be incurred by industry in the event the proposed restriction enters into force:

- Information about the presence of phthalates in articles is available via other means than testing, e.g., due to obligations under REACH or other legislation.
- The majority of companies ensure compliance with EU and national legislation primarily using contractual obligations and by providing information on the restricted substances to their suppliers.
- Compliance testing by the buyer is used in rare occasions, primarily for spot checks. This is practiced primarily by larger companies.
- The testing costs are dependent on the frequency of testing. Company practices are highly diverse and are often dependent on the track record of the international supplier and the variety of products supplied. Often, international suppliers are required to provide testing results, which could be used for multiple shipments and buyers.
- Many companies already have practices put in place (due to regulatory requirements or voluntary actions) regarding the presence of phthalates in their products. As these actions are part of the existing industry practices, they cannot be considered instigated by the proposed restriction and therefore, cannot be considered part of the costs of industry to ensure compliance with the proposed restriction.
- It is unlikely that these costs would occur indefinitely in the future. It is feasible to assume that the need for any testing for phthalates would decline over time with the increased familiarity with regulatory practices and the decreased incentive to use the four phthalates instead of their alternatives.

Thus, it can be concluded that although industry would likely continue to conduct testing to ensure compliance in the event the proposed restriction enters into force, these costs, whose magnitude is highly uncertain (due to diverse industry practices), are likely largely not attributable to the proposed restriction (due to existing practices to monitor the presence of phthalates in articles under regulatory obligation or voluntary policies). Any minor

uncertainties related to societal costs due to testing as a result of the restriction are already taken into account in the estimation of the substitution costs of imported articles. As stated there, a larger price differential was assumed for imported articles to account for such uncertainties. The impact of these uncertainties on the cost-effectiveness and the benefit-cost ratio of the proposed restriction is discussed in Annex E.

D.3.1.3. Costs for recycling sector

It is estimated that about 500 000 tonnes of PVC waste is recycled and of this around 200 000 tonnes is soft PVC (Vinyl Plus, 2014). Assuming that the average DEHP content of recyclates in soft PVC is 10%, it is estimated that in total about 20 000 tonnes of DEHP is recycled in the EU. The percentage is expected to decrease over time as the use of DEHP in virgin PVC is declining even without the restriction proposed in this report. It is expected that the proposed restriction will accelerate this decline.

The proposed restriction is likely to increase prices of those articles that can no longer use relatively cheap recycled material. This effect will be mitigated over time with reduction of the presence of the four phthalates in post-consumer waste which in turn will contribute to cleaner, less polluted waste to be available for recycled goods.

The articles in the scope of the restriction produced with PVC recyclates (mainly footwear) are likely to represent a small fraction of the total amount. The main concern centres on post-consumer waste as it tends to be less homogenous and has a higher level of DEHP concentration than post-industrial waste. Based on discussions with industry experts it is assumed that the restriction would impact about 4%[161] of currently recycled soft PVC waste. It is assumed that 50% of the recycled material - i.e. 4000 tonnes – would need to be replaced with the more expensive material.

Given the moderate share of the articles in the scope, it is likely that the recycled material that could no longer be used would be absorbed by the current markets. The companies affected would need to use (more expensive) virgin PVC (without the four phthalates) or higher quality recyclates, either way which would incur higher raw material costs.

Based on market research carried out through internet the cost difference between the low quality recyclates (made out of post-consumer PVC waste) and virgin material is assumed to be €350 per tonne of raw material. This is based on an assumption that the converters would mainly replace lower quality recyclates with higher quality ones based on post-industrial waste (and perhaps some additional virgin material), and accounting for some costs savings due to use of a more homogenous raw material.

[161] Plastics Recyclers Europe and EuPC have informed ECHA that the proposed restriction is expected to impact the volumes of current post-consumer PVC waste recycled, however the reduction should be less than 5%. For the post-industrial waste the percentage may be higher, however, also below 10%.

Table D14 gives the estimated cost increase of the proposed restriction because four phthalate containing recyclates are no longer available for certain uses.

Table D13. Estimate of the annual impact of the proposed restriction on recycled soft PVC containing four phthalates in 2020.

	amount	**unit**
Soft PVC recycled	200 000	tonnes of PVC
Articles covered due to restriction	8 000	tonnes of PVC
Substitution with more expensive PVC material	4 000	tonnes of PVC
Price increase due to higher quality raw material	€ 350	per tonne
Cost increase on due to restriction	€ 1.4	million (2020)
- discounted to 2014	€ 1.1	million (2014)[162]

Source: Vinyl plus (2014) (for the amount of soft PVC recycled), and own estimates

The costs are likely to decline in the future as the content of the four phthalates in the waste also declines. With the average DEHP concentration decreasing, more four phthalate free recyclates become available over time for the affected uses.

As an illustration, take Wellington boots[163], which cost between €15 and €50 a pair, depending on the quality, design etc. A pair weighs about 1 kg. If the price of raw material increased by €0.35/kg (i.e. €350/t) the material costs of the boots would increase by about 1-2%, if DEHP containing recycled PVC could not be used in the boots as a result of the proposed restriction. The price of boots using rubber or DEHP free (virgin) PVC would be unaffected.

The changes introduced by the restriction may affect converters. For instance, small and medium size converters may be affected, at least temporarily, from reduced possibilities to use those recyclates that contained DEHP. However, the total demand and supply of the end-products is not likely to be affected as a result of the restriction, as the price effect is small.

In conclusion, the proposed restriction is estimated to affect recycling mainly due to four phthalate containing recyclates no longer being available for some end products, such as footwear. Instead four phthalates free material would need to be used. The resulting price of the four phthalate free end product would be somewhat higher due to higher material cost.

[162] The additional industry cost would be €1.4million (= 4000 tonnes multiplied €350/t). When discounted from 2020 back to 2014, a total impact on recycling would be about €1.1 million annually.

[163] Wellington boots can be made of rubber or PVC. Also rubber may be incorporated into the PVC to increase softness. Rubber boots tend to be more expensive.

D.3.2. Other economic impacts

Impacts on compounders (on producers of PVC in primary forms)

Placing on the EU market of PVC in primary (semi-final) forms (pellets, plastisols, compounds) is not directly impacted by the proposed restriction because these articles are not used in indoor environment or in outdoor environment that leads to contact with skin or mucous membrane before further processing.[164] Insofar PVC in primary forms is not further converted into products that fall within the scope of the proposed restriction, there would be no impacts on producers of PVC compounds or plastisols containing one or more of the four phthalates.

In the event of a restriction, compounders could continue to place DEHP containing compounds, dry-blends or plastisol formulations on the EU market or for exports if they are within the supply chain of holders of authorisation. However, they would no longer be used by downstream users who produce articles within the scope of the restriction for the EU market (although it would be possible to produce articles for export purposes if they are members of the supply chain of authorisation holders). Therefore, these compounders would have to consider identifying domestic markets not impacted by the restriction (e.g., manufacturing of roofing: out of scope as roofing is not anticipated to lead to dermal or inhalation exposure), international markets, or replacing DEHP. If the former, the compounders could incur some transaction costs.[165] If the latter, the substitution costs of compounders are assumed to be fully passed on to downstream users and these costs are taken into account in the restriction compliance cost insofar the products in primary forms are further converted into articles in the scope of the restriction. These are reported as substitution costs by end-use article group.

Any possible profit losses as a result of lower demand for DEHP containing plastisol are likely to occur only in the short term due to transitioning to alternatives and will likely be offset by profit gains of compounders of plastisol containing alternatives plasticisers who are able to respond immediately to increased demand for DEHP free plastisols.

Impacts on articles outside the scope of the restriction

Anecdotal information suggests that following the entry into force of the restriction on phthalates in toys and childcare articles (entry #51 of Annex XVII), some producers choose to transition to alternatives for all their product lines. This could be explained by their seeking to realise economies of scale for plasticiser purchasing or other procurement and manufacturing efficiencies, or by their pursuing marketing strategies (e.g., “green” image), or by optimising article production on one manufacturing line (when multiple articles are produced on the same line). It is possible that similar manufacturing trends could lead to the replacement of the four phthalates also in articles that fall outside of the scope of the proposed restriction if it impacts article manufacturers with mixed product lines. For example, producers of roofing (out-of-scope as roofing is not anticipated to lead to dermal or inhalation exposure) and flooring (in the scope of the proposed restriction), could consider replacing the four phthalates in both

[164] Use in the EU of compounds, dry-blends or plastisol formulations containing DEHP, DBP, DIBP, or BBP requires an authorisation under title VII of REACH.

[165] Transaction cost is a cost incurred in making an economic exchange (restated: the cost of participating in a market). Transaction costs can be divided into three broad categories: search and information costs, bargaining costs, policing and enforcement costs [of e.g., a contract]. Source: Wikipedia.org

product lines (or stop producing either flooring or roofing membranes if that is economically more sensible). However, it is uncertain to what extent the past substitution of the four phthalates could be attributed to these (inadvertent) consequences of a restriction or to other forces (e.g., other regulatory pressures which began with the introduction of the restriction on toys and childcare articles and the classification of the four phthalates). Therefore, these potential impacts are noted but not quantified for the purpose of the assessment of the proposed restriction.

Impact on exports

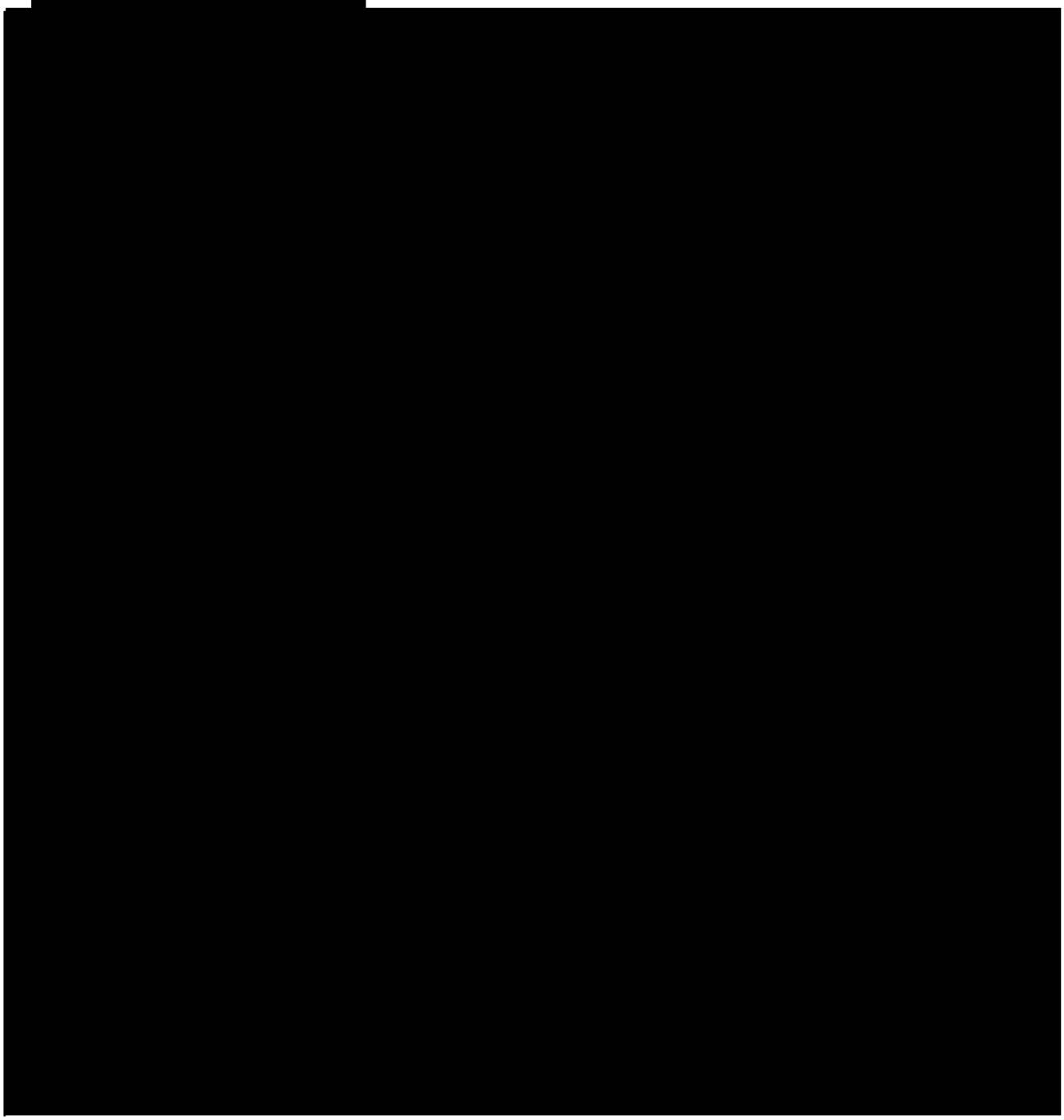
The proposed restriction bans the placing on the EU market of articles containing the four phthalates; therefore, export of these articles is not directly affected by the restriction. EU manufacturing of DEHP containing articles[166] used indoors or outdoors with prolonged dermal or mucous membrane contact could continue for the purpose of exports, provided these EU producers are within the supply chain of authorisation holders. As it is uncertain what percentage of exports would cease as a result of the restriction (and in fact it is theoretically possible that exports increase as a result of the restriction), the costs of transitioning to alternatives for exported articles are not included in the restriction compliance costs. These costs are anticipated to be fairly minor in importance (i.e., the tonnages of the four phthalates contained in exported articles in scope are projected to represent less than 5% of their tonnages contained in EU article production and imports in 2020). As shown in Annex E, if the costs to transition to alternatives for exports are included in the costs of the restriction, the annualised restriction costs would increase from €16.9 million to €17.2 million, while the cost-effectiveness would decline by 1.5%.

Impacts on the quality of the goods

Many of the similarly priced plasticisers have very similar performance characteristics as DEHP, DBP, DIBP and BBP. Some of the alternatives have advantages in particular applications (e.g., extreme temperature resistance, improved permanency) potentially leading to increased quality of the goods. By and large, it is assumed that the quality of end products is similar or marginally improved and therefore, difficult to differentiate by consumers. Thus, these benefits to consumers are not quantified for the purpose of the estimation of the restriction compliance costs. (See Table D15 in Confidential Annex to Analysis of Alternatives in AFA (2013a).)

[166] Only exports of DEHP containing articles are relevant as there are no applications for authorisation for DBP, DIBP and BBP for their use in articles within the scope of the proposed restriction.

Table D14: [redacted]



Impacts on substance manufacturers and their upstream supply chain

Two[167] manufacturers of DEHP currently await a decision by the EU Commission on the granting of authorisation for its use in articles which fall within the scope of the restriction. It is uncertain whether and to what extent these manufacturers have already refocused to export markets, to replace imports or to manufacture of some of DEHP's alternatives. As mentioned previously, one of the applicants for authorisation already in 2012 announced that they will produce DEHT.[168] [redacted] DEHP tonnages produced in the EU in 2013 would be impacted by the entry into force of the restriction. Although DEHP production capacity in the EU has since declined due to Arkema's plant closure, it is possible that this would lead to a profit loss for EU manufacturers of DEHP. The remaining manufacturer of DEHP in the EU would likely also experience costs if they choose to transition manufacturing to one of the alternatives. These costs will also depend on their ability access precursor raw materials such as alcohols, which may be dominated by existing manufacturers of alternatives. However, information from the public consultation revealed that some alternatives, such as DEHT, are of potentially lower cost as a replacement because it does not require investment in new alcohols and raw materials.

At the same time, the introduction of the restriction would encourage substitution of DEHP with alternative plasticisers, many of which, including those assumed in the substitution scenario, are currently produced within the EU. Therefore, it is assumed that EU manufacturers of alternatives are anticipated to increase their profits as a result of the restriction.

For the purpose of estimating the restriction costs it is assumed that the profit margin of all plasticiser producers is similar; therefore, any negative impacts on profits of DEHP manufacturers are anticipated to be offset by gains in profits by manufacturers of alternatives due to the restriction. As DEHP manufacturers are located in Central Europe (Poland and the Czech Republic), while manufacturers of alternatives are primarily in Western European Member States, the proposed restriction may create distributional impacts (See section D.3.3).

Enforcement costs

Enforcement costs are administrative costs incurred by Member States enforcement agencies to ensure that economic actors on the EU28 market comply with the Union regulations. It is estimated that on average Member States spent approximately €55 600 per restriction per year (in 2014 values) to ensure compliance with Annex XVII of REACH. This is estimated on the basis of reported number of controls for the period 2010-2014 (ECHA 2015b) based on the data collected from the Member States (reporting under REACH art. 117 / CLP art.46). The calculation is based on an average cost per control (inspection) and an average number of controls per restriction. The report argues that due to learning and economies of scale and due to generally low inflation there is no reason to expect large increases in the costs per-control over the study period. Furthermore, while the average enforcement costs may remain fairly similar over time, as they are driven by budgetary constraints, the costs for individual restrictions would likely vary. It is often the practice that enforcement campaigns focus on

[167] Following the application for authorisation, the third applicant, Arkema France, closed manufacturing facilities.

[168] More suppliers emerging for DOTP http://blog.phthalate-free-plasticizers.com/2012/01/05/more-suppliers-emerging-for-dotp/

newer restrictions or high risk restrictions, and less resources are allocated to restrictions industry is familiar with. Therefore, it can be anticipated that enforcement costs for the proposed restriction would be larger in the first years after the entry into force and close to zero towards the end of the study period. The development of enforcement costs over time depends on the general volumes of phthalates used, the non-compliance rate, and the Member State enforcement priorities.

Impacts on SMEs

The proposed restriction is expected to have some impact on different actors in the supply chain. It can impact importers of articles containing the four phthalates, however, the effect should be quite limited given: the availability of similarly priced substitutes, long-term experience with substitution, no barriers to transitioning to alternatives such as high up-front investment or proprietary technology, long-standing knowledge of regulatory action on the four phthalates, substantial share of DEHP use remaining outside the scope, etc. It can also have distributional impact in the manufacturing as some actors currently relying on the four phthalates may exit the industry whereas others working with or moving to alternatives are expected to take their place as the total demand for articles phthalates are used for is not supposed to largely change. The decisions to exit or adapt depend on several firm and product specific characteristics and are difficult to forecast at this point. At this point, there is no evidence, that certain type or size companies, e.g. SMEs, would be more affected than others.

A large proportion of manufactures of plastic products are SMEs using EuroStat 2007 figures: 99% of 58 300 companies employ 249 persons or less, with 90% employing less than 50 persons. However, these statistics have some uncertainties linked to them due to their age but these are thought to be minor as the total number of companies in this sector has remained stable: 58 300 (2007) vs 55 000 in 2013. It is assumed that the larger companies produce more different types of articles and may face fewer challenges adapt to the restriction. It is assumed that all importers of phthalates are SMEs.

In the recycling sector industry claims the SME's to be potentially disproportionately impacted by the proposed restriction. This may be true, given that some companies may have based their activities on certain type of (recycled) material or as it can be more difficult for an SME to move to an alternative substance. On the other hand, according to the analysis of recycling sector in this dossier, the total impact should be only moderate as most of the articles based on recyclates are not in the scope of the proposed restriction.

The public consultation also revealed that SMEs are more likely to have limited production lines and therefore, are less likely to shift away from DEHP as a general purpose plasticiser due to its advantages in diverse applications. However, it is noted that there are other general purpose plasticisers which have been shown to be technically feasible across diver applications (e.g., DINP, DEHT).

Therefore, the availability of alternatives for all uses of phthalates in the articles included in the scope of this restriction (and the proposed derogations of the restriction which accommodate the majority of the articles manufactured from recyclate) and the transitional period of three years is expected to minimise the impact of the proposed restriction on SMEs.

D.3.3. Social, wider economic and distributional impacts

Social impacts

It is possible that as a result of the proposed restriction employment of DEHP manufacturers is impacted negatively. The size of the social impacts would depend on the degree to which the two manufactures are able to take over the share of DEHP import to the EU or new export markets or to diversify into production of DEHP alternatives. As mentioned in the section Impacts on substance manufacturers and their upstream supply chain, the latter is already afoot as a result of the inclusion of DEHP on the Authorisation list.

Any possible employment losses to DEHP manufacturers will likely be offset by employment gains in manufacturing of alternative plasticisers, whose EU sales are anticipated to increase as a result of the proposed restriction. However, as DEHP manufacturers are located in Central Europe (Poland and the Czech Republic), while manufacturers of alternatives are in other European member states, the restriction could lead to a temporary loss of employment income in small communities in Poland and the Czech Republic. Such communities tent to have less diversified regional economies which can be highly dependent on large employers of the calibre of chemical manufacturers. It is often the case in such communities that a downsizing of a major employer is associated with long-term negative effects on the regional economic development.

It should also be noted that employment in the chemical sector is often associated with higher wages and benefits. Therefore, another aspect of a loss of employment in small, less diversified communities is that it would be difficult for many workers to find similarly high-paying employment without relocating and thereby, incurring additional transaction costs to obtain alternative employment.

Wider economic impacts

As shown in the section D.3.7, the proposed restriction would have minor impacts on article prices; therefore, international trade flows are likely to remain unchanged and no substantial wider economic impacts can be anticipated as a result of the restriction.

Distributional impacts

Currently, EU manufacturers could use DEHP, DBP, DIBP, and BBP in articles within the scope of the restriction proposal if they apply for an authorisation, while importers are not required to apply (as authorisation requirements do not apply to imported articles). This creates extra costs for EU manufacturers in comparison to importers to access the EU market. The entry into force of the restriction will level the playing field for EU article manufacturers and importers.

Negative impacts of the restriction on DEHP manufacturers (profits and employment) are anticipated to be compensated by gains by manufacturers of alternatives. However, as DEHP manufacturers are located in Central Europe (Poland and the Czech Republic), while manufacturers of alternatives are in other European member states (or potentially outside the EU28), the proposed restriction would give rise to distributional impacts.

Other impacts

DEHP is a Water Framework Directive (2000/60/EC) Priority Hazardous Substance and as required by the directive, requires discharges, emissions and losses to be ceased or phased-out. According to information submitted during the public consultation, recent monitoring data from the UK's Water Industry Research (UKWIR) Chemicals Investigation Programme show that DEHP is widely found in wastewater treatment plant (WWTP) *effluent* in the UK, with a median concentration of 0.69 µg/L[169] and that the concentration in a small proportion of effluents exceeds the Environmental Quality Standard (EQS). There is evidence that DEHP is emitted in wastewater from households, with higher concentrations arising from newer housing stock than old. Plastic products in roofing and plumbing materials, paints, sealants, adhesives and fillers may all contribute to this release. However, an additional source has recently been shown by Saini et al. (2016) to be dust/vapour deposited on clothing (and possibly also soft furnishings) that are subsequently laundered in washing machines. This paper provides an estimate that a typical laundry machine may release about 300 mg of five phthalates (the four substances subject to the restriction proposal plus DINP) per laundry load to wastewater. Restricting the content of these four phthalates in articles in scope will therefore remove one of the remaining sources of DEHP to the environment, and thereby help to ensure that the aims of the WFD are achieved in the future.

D.3.4. Total restriction costs of the proposed restriction

The total costs to EU society from the introduction of the proposed restriction are estimated at €16.9 million annually. (See Table D16) These net compliance costs are estimated to occur annually for 20 years from the entry into force of the proposed restriction: assumed as 2020 for the purpose of this analysis. The NPV of these future costs over the next 20 years is less than €230 million in total (using 4% discount rate). These costs are less sensitive to the chosen discount rate (in comparison to benefits): applying 2% discount rate, the NPV of the total restriction costs is € 311.5 million or €19.1 million annually.

[169] http://v-scheiner.brunel.ac.uk/bitstream/2438/8867/5/Fulltext.pdf. Further details about this programme can be found at https://www.ukwir.org/site/web/news/news-items/ukwir-chemicals-investigation-programme, and in the UK comments submitted for the nonylphenol ethoxylate textile restriction proposal

Table D15 Summary of Net compliance costs of the proposed restriction, annual, 2014 - base year

Net costs from 2020 onward	**Estimates (annual)**
Substitution costs	
- Material costs	€15.8 million euro
- RDRPPM costs	Not estimated, likely negligible
Testing costs	Uncertain, addressed in material costs
Costs of recycling sector	€1.1 million euro
Enforcement costs	€0.06 million
Costs to compounders (i.e., on producers of PVC in primary forms)	Included in material costs
Costs to substance manufacturers	Assumed €0 but potential benefits for manufacturers of alternatives are not estimated
Impacts of higher quality of the good containing the alternatives	Assumed €0 but likely on balance represent benefits (and not net costs) of the restriction
Costs to SMEs	Not estimated, likely negligible
Social impacts	On balance, likely €0
Wider economic costs	On balance, likely €0
Distributional costs	Assumed €0 but likely on balance represent benefits (and not net costs) of the restriction
Total restriction costs	**16.9 million euro**

Notes: * R&D, reformulation, process & plant modifications

The total restriction costs of €16.9 million annually are believed to adequately illustrate the anticipated costs to EU society as some costs are overstated in order to account for any uncertainties related to the non-quantified negative impacts of the restriction. In summary, the estimates overstate the costs to EU society because:

- Material costs are overestimated because:
  - Confidential information supports that the costs are closer to the lower range of estimates of €8.4 million annually.
  - The main assumption in the analysis that the effective price differences between the four phthalates and their alternatives would exist throughout the sturdy period of 20 years is highly uncertain. These differences would likely approach zero in the long run. (See section D.3.1.1.4 for further details on the reasons for considering material costs overestimated)
- Enforcement costs and the costs of the recycling sector are assumed to remain constant throughout the study period for simplicity, while it is likely that these would be incurred in the short to medium term of entry into force of the restriction. (See sections D.3.1.3 and D.3.2 for further information.)
- The majority of costs are associated with transitioning to alternatives of imported articles: €15.3 million annually or more than 90% of total costs. The assumption that all these costs are passed on to EU entities (EU buyers or end-users) is highly uncertain. Given the high price competition on some article markets, it is foreseeable to assume that some of these costs are borne by international article manufacturers or other entities in the non-EU supply chain. This would likely lead to impacts on profits in non-EU jurisdictions.

Annex E tests the impact of the main uncertainties on the conclusions of the analysis.

D.3.5. Human health and environmental impacts

D.3.5.1. Human health impacts

All four phthalates show effects on reproductive organs and fertility in experimental animals exposed prenatally and are all classified as toxic to reproduction in category 1B according to the CLP Regulation. The cause for the effects has been shown to be their anti-androgenic properties. For that reason it has been unanimously agreed in the Member State Committee that the four substances have endocrine disrupting properties.

Experimental animals

As described in Annex B, the spectrum of adverse effects observed in the male rat following gestational exposure to the four phthalates known as the rat phthalates syndrome includes:

- reduced semen quality;
- testicular changes including decreased testes and epididymides[170] weight, tubular atrophy and Leydig cell hyperplasia;
- decreased anogenital distance[171];
- increased incidence of hypospadias;
- increased incidence of cryptorchidism;
- delayed puberty onset (delayed preputial separation[172]);
- increased nipple retention[173];
- multinucleated gonocytes and changes in early germ cell differentiation.

Tubular atrophy is a diagnosis for the absence of most or all germ cells from affected tubules (Creasy et al. 2012). The loss of germ cells is an important observation since it is the germ cells that divide and differentiate into mature spermatozoa in the testis. Decreased testis weight can be caused by damage to the seminiferous tubules. There is a low inter-animal variability in testis weight and therefore a significantly decreased testis weight is an indicator for an adverse effect (OECD 2008). Similarly, decreased epididymal weight may be a sensitive indicator of decreased sperm production (Sellers et al. 2007). Leydig cell hyperplasia may develop to evolve to a neoplastic proliferative lesion (tumour) (Creasy et al. 2012). Reduced semen quality has also been observed following gestational exposure to these. All these observations in experimental animals confirm that the exposure of experimental animals leads to testicular injury.

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[170] Plural from “epididymis”, the tube that connects a testicle to a vas deferens (the vas deferens transports sperm from the epididymis to the ejaculatory duct). Sperm from the testis matures during transit in the epididymis.

[171] Anogenital distance (AGD) is considered to be a marker for diminished androgen action and males with significant decreased AGD are considered to be “feminised” males (Kortenkamp et al. 2011).

[172] Preputial separation is the separation of the prepuce (foreskin) from the glans penis (head or tip of the penis) during development and has been shown to be androgen dependent (Korenbrot et al. 1977). Yamasaki et al. (2001) and Korenbrot et al. (1977) observed that in the rat normal separation happens from post-natal day 39 and is completed by post-natal day 45. This is around the puberty in male rats (Korenbrot et al. 1977).

[173] Normally male rats don’t have nipples.

In addition, decreased anogenital distance, increased (permanent) nipple retention, increased incidence of hypospadias, increased incidence of cryptorchidism and delayed puberty onset have been observed following exposure to the four phthalates.

The spectrum of effects is known as the "phthalate syndrome" (Foster 2006; NRC 2008; Kortenkamp et al. 2011; CHAP 2014; Health Canada 2015). The syndrome is characterised by *"malformations of the epididymis, vas deferens, seminal vesicles, prostate, external genitalia (hypospadias), cryptorchidism and testicular injury together with permanent changes (feminization) in the retention of nipples/areolae (sexually dimorphic structures in rodents) and demasculinization of the growth of the perineum resulting in a reduced anogenital distance (AGD)"* (Foster 2006). It is well understood that the cause for the rat phthalate syndrome is suppression of foetal androgen action (Kortenkamp et al. 2011). Suppression of foetal testosterone synthesis, reduced semen quality, testicular changes, retained nipples and changes in AGD appear to be the most sensitive effects in the rat after gestational exposures. At higher dose levels, malformations of the genitalia (e.g., cryptorchidism and hypospadias) and delayed preputial separation were seen. Younger animals are more sensitive than older ones to the phthalate syndrome: the testis is most sensitive during the prenatal period, and prepubertal and pubertal rats are more sensitive than adults (NCR 2008).

In addition to the anti-androgenic effects, DBP, DIBP and DEHP induce multinucleated gonocytes and changes in early germ cell differentiation in experimental animals[174]; effects which are considered to be independent of foetal testosterone reduction (Borch et al. 2006, Gaido et al. 2007; Lambrot et al. 2009) and may correspond to precursors of testicular germ cell cancer in humans (Ferrara et al. 2006). However, findings by Saffarini et al. (2012) do not appear to lend support to this hypothesis.

Humans

Table D17 below summarises important elements in the evidence for human impacts resulting from exposure to DEHP, DBP, DIBP and BBP.

Several epidemiological studies show associations between maternal phthalate exposure and anogenital distance, congenital malformations of the male genitalia, semen quality and pubertal timing in children. Development of testicular cancer shares some of the same risk factors but human epidemiological studies investigating phthalate exposure and testicular cancer development are lacking. The associations reported are not always consistent and are generally associated with uncertainties due to methodological limitations, amongst others related to the difficulties to study associations between foetal exposure and effects later in life, limited study populations which limit the power of the studies, uncertainties in the back-calculation of urine concentrations to estimated daily exposure and influence from the fact that humans are exposed to many substances at the same time and that genotype and life style factors such as diet, smoker/non-smoker, weight, behaviour and societal background. Associations from epidemiological studies do not prove causality, and conversely, the absence of an association does not imply the absence of a causal relationship. The epidemiological studies on the four phthalates do not allow for derivation of a dose-response relationship in humans. It can be concluded that in isolation the epidemiological evidence is generally rather

[174] Multinucleated gonocytes and multinucleated germs cells are synonyms for the same lesion observed in fœtal or neonatal rats, mice and marmosets. The lesion should not be confused with 'multinucleated giant cells'.

weak but it provides further support to the default assumption that observations in experimental animals are relevant to humans and the evidence that effects occur in the population following exposure to the four phthalates.

The effects of the phthalate syndrome observed in rats have also been observed in humans and it has been suggested to have a human counterpart known as the "testicular dysgenesis syndrome" (TDS). Cryptorchidism, hypospadias and poor sperm quality are risk factors for each other in humans (Kortenkamp et al. 2011; Skakkebaek et al. 2016). These conditions are also predictive of testicular germ cell cancer, in fact, the majority of men with testicular germ cell cancer also have some or all of the other disorders (Kortenkamp et al. 2011). Increasing evidence also link reduced AGD in humans to this group of risk factors (Skaekkebaek et al. 2016). The single symptoms and combinations thereof are also risk factors for reduced fecundity (Skaekkebaek et al. 2016).

Semen quality is determined by several parameters (e.g., sperm counts, sperm motility, sperm concentration, ejaculation volume and other parameters). Effects on individual sperm parameters in experimental animals may provide important information about effects on semen quality because in humans even a slight reduction in sperm quality/count may be critical for fertility (ECHA guidance Chapter R.7a). Rodents are different than humans with regard to semen quality: as much as 90% reduction in sperm count may be needed to affect the fertility index in rodents. Human males on the contrary have highly variable sperm counts, generally lower than in rodents, and many men have sperm concentrations near or below WHO reference values for fertility (OECD 2008). Thus, in a case of human subfertility even a small change in sperm counts or sperm motility may lead to infertility. For this reason, a statistically significant change in sperm count in a rodent study is considered to be indicative of a potential effect on fertility in humans (OECD 2008).

The interlinked testicular changes observed in experimental animals are all relevant to humans and contribute to the evidence that humans are at risk of reduced fertility caused by exposure to the four phthalates.

Hypospadias is a common birth defect of the urethra in the human male where the urinary opening is not at the usual location on the head of the penis. Kortenkamp et al. (2011) gave the following description: *"Androgen action in foetal life is crucially important to ensure proper location of the urethral opening at the tip of the glans penis. If androgen action is diminished, the urethra opens on the underside of the glans penis (mild, so-called glanular hypospadias). In severe case, the opening is positioned on the shaft of the penis, or even near the scrotal sack."*.

Cryptorchidism is the absence of one or both testes from the scrotum and is the most common birth defect in infants (undescended testes). However, the majority of these cases in humans resolve within three months after birth. Kortenkamp et al. (2011) gave the following description of cryptorchidism: *"Normal testis descent occurs in two phases. Early in gestation, the foetal testes migrate from their point of origin near the kidneys into the pelvis (transabdominal phase). Later, towards the end of gestation, descent from the pelvis into the scrotum is accomplished (transinguinal phase). The second, transinguinal phase is androgen dependent, and disruption of this phase appears to be the most common cause of cryptorchidism at birth. The burst of androgen synthesis that occurs during the "mini puberty" in the first 3-5 months after birth probably helps to revert many cryptorchidisms diagnosed at*

birth. Approximately 50% of cryptorchidisms diagnosed at birth resolve themselves within the first 3 months of life. The earlier, transabdominal phase of descent is triggered by insulin-like factor 3 (Insl3), a peptide hormone, and disruption of Insl3 action causes complete failure of testicular descent.".

No good animal model for human testicular cancer seems to exist. Leydig cell derived testicular tumours are seen in rats, whereas those in humans are germ-cell derived (NRC 2008). However, unusual clusters of undifferentiated germ cells and multinucleated germ cells have been observed across species in foetal rats, mice, marmosets, and also humans (van den Driesche et al. 2015, Fisher et al. 2003, Barlow and Foster 2003, Gaido et al. 2007, McKinnell et al. 2009), and it has been proposed that these developmentally impaired germ cells might correspond to precursors of testicular germ cell cancer in humans (Ferrara et al. 2006). Testicular germ cell cancer is considered to be part of the human "testicular dysgenesis syndrome" (TDS): the conditions of TDS are predictive of testicular germ cell cancer and testicular germ cell cancer is believed also to be of foetal origin and is caused by disturbances in germ cell programming (Skakkebaek et al. 2016). Overall, it is unclear whether exposure to the four phthalates has a role in testicular germ cell cancer in humans.

The biological significance of nipple retention is not fully known and a difference between humans and rats is evident – in human males two nipples are always retained during development whereas in male rats nipples are absent. If one or more nipples are present in male rats after birth they have what is called "increased nipple retention". Permanent nipples in males have been observed (Barlow et al. 2004; Gray et al. 2009) and are considered a malformation (OECD 2008). Nipple retention is part of the rat phthalate syndrome and is considered to be an indicator of foetal androgen suppression. Foetal androgen suppression in experimental animals is biologically relevant to humans and thus nipple retention in the rat is considered a relevant biomarker to humans. Similarly, decreased anogenital distance in the male rat is considered a relevant biomarker to humans. Decreased AGD in male offspring and nipple or areola retention are sensitive measures to exposure to anti-androgens and have been shown to be predictive of other effects such as hypospadias and undescended testes (OECD 2008).

It is reasoned that the disorders of TDS share a common foetal origin (NRC 2008). For example regarding poor semen quality, it is thought that reduced foetal testosterone affects the functioning of germ cell supporting Sertoli cells and the androgen synthesising Leydig cells (Kortenkamp et al. 2011). It is known that androgen action in foetal development is a crucial factor in the proliferation and programming of Sertoli cells (Kortenkamp et al. 2011; NRC 2008). If not enough Sertoli cells are available to support germ cells, it may result in low sperm counts in adulthood and may explain how diminished androgen action in foetal life may negatively impact on semen quality later in life (Kortenkamp et al. 2011).

As mentioned above, there is some epidemiological evidence in support of TDS and specifically of the contribution of the four phthalates. The disorders of TDS are observed with higher frequency in some countries in comparison to others and appear to be increasing over time (Kortenkamp et al. 2011; Skakkebæk et al. 2001; Carlsen et al. 1992; Swan et al. 2000). This could be seen as further evidence that in addition to genetic and life-style factors, environmental factors such as exposure to anti-androgenic chemicals may have a role in the disease incidences (Hu et al. 2009).

Population at risk

The RCR for combined exposure to DEHP, DBP, BBP and DIBP at the 95th percentile exposure level was at or above 1 in 14 out of 15 Member States. In Member States with a combined RCR at or below 1 at the 95th percentile exposure level there was still a risk for individuals with the highest exposure levels in the study population. It can be concluded that there was a risk in all Member States in 2014. The number of boys at risk in the population was estimated based on the number of live births in each EU28 country and the GM and 95th percentile RCR values projected for 2030[175].

The number of boys at risk due to foetal exposure is estimated to be 54 000 per year, or 2.1% of new born boys. This corresponds to **1.1 million boys** over the time span of 20 years.

Although the foetus is thought to be more sensitive to the effects of the four phthalates, neonates, infants and children are still considered to be among the sensitive population because their reproductive system is still developing (David 2006; Foster et al. 2001; den Hond and Schoeters 2006; Jacobson-Dickman and Lee 2009). Using the exposure values from children, it is estimated that 175 000 boys per year are at risk, or **3.5 million boys** over the time span of 20 years. This corresponds to a share of new born boys of 6.8%.

There are several uncertainties to these estimates. As concluded in Annex B, evaluation of the uncertainties to the RCRs generally point to possible underestimation of the RCRs[176]. Furthermore, the projected RCRs were used in the estimation, but the projections of future tonnages placed on the EU market (baseline) and the relationship between the baseline projections and the risk level are uncertain (see section D.3.5.3). Moreover, other effects described in Table D17 were evaluated to occur (e.g., immunological effects, reduction of semen quality from exposure in adult men, delayed onset of puberty in boys and girls). These effects may increase the number of persons at risk since also girls, adult men and adult women may be affected. In particular the moderate to strong relationship between exposure estimates and adjuvant effects in humans suggests that the population at risk may be underestimated. The available information suggests that phthalate exposure could lead to immunological disorders (allergy, asthma and eczema), possibly at levels lower than reproductive toxicity, however, no DNEL could be derived[177].

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[175] In short, the calculations are as follows. Using the GM and 95th percentile RCR values projected for 2030 the standard deviation was calculated, assuming a lognormal distribution. This allowed to calculate the percentile where the RCR equals 1 for the 15 EU Member States (the results from the UK were excluded because of the small sample size (n=21) and CH is not part of the EU). This percentile was used to derive the annual number of male births per country that are at risk, assuming the birth rate is constant. The fraction of boys at risk from the 15 EU countries was subsequently applied to the male births in the remaining 13 Member States to derive the total number of boys at risk. It is assumed that the entry into force of the restriction will be 1 January 2020 and the temporal scope is 20 years. Thus the population at risk is assessed for the period 2020-2039.

[176] Amongst others, using volume based method of intake calculation instead of the creatinine method we used possibly doubles the RCRs; children younger than 6 are likely to have higher exposure.

[177] The available information does not allow to determine a DNEL nor a specifically sensitive period for the adjuvant effects of the four phthalates. As a matter of illustration, if it were assumed that the DNELs for immunotoxicity were equal to the DNELs for reproductive toxicity and that the DNEL applicable to the whole population (males, females, children and adults), and that the exposure level estimated for either mothers or children is applicable to the entire population, the population at risk might be estimated to be somewhere between 2% (based on exposure levels of mothers) and 7% (based on exposure levels of children).

In comparison with the above figures estimated for 2030, the estimated number of boys at risk in 2011 (when urinary samples were taken) is 6% of new born boys from foetal exposure and 18% of new born boys from exposure during early life. Considering the uncertainties on exposure from biomonitoring, the uncertainties related to future risk projections, and uncertainty that the population at risk is limited to boys, a scenario can be considered where the projected risks are assumed to be twice as high. In this scenario 5.4 million boys would be at risk over the time span of 20 years from foetal exposure or 13 million from exposure during early life.

It should be noted that individuals in the population at risk have an increased probability to the disorders discussed above. It is unknown what the increased disease incidence rates of the disorders in the population at risk would be as a result of exposure to the four phthalates.

In addition, as described in Annex B, workers are exposed to DEHP during manufacturing and formulation of DEHP and the production of articles. Workers are furthermore exposed to the four substances during formulation of recycled soft PVC containing DEHP in compounds and dry-blends. RAC concluded that the applicants did not demonstrate adequate control. The number of exposed workers was claimed confidential by the applicants, but considering the number of downstream users of the applicants, the number is of considerable size.

Conclusion

Biologically relevant findings seen in experimental animals should be considered relevant to humans unless convincing evidence exists to the contrary (ECHA guidance Chapter R.7a). All of the effects observed in experimental animals are considered to be biologically relevant since the conditions also exist in human males[178]. In addition, there is supporting epidemiological evidence and it has been hypothesised that the phthalate syndrome observed in rats has a human counterpart known as the “testicular dysgenesis syndrome”.

Reproductive risks are of obvious concern for the general population and similarly, to the individual, an impairment of the ability to reproduce and the occurrence of developmental disorders are self-evidently serious health constraints (ECHA guidance Chapter R.7a). Thus, since a risk is identified for combined exposure to DEHP, DBP, BBP and DIBP in the majority of European countries (14 out of 15 Member States), there is a risk in the European population that the phthalates cause a spectrum of serious and interlinked developmental effects in males, including with high probability reduction of semen quality, testicular changes, decreased anogenital distance, decreased foetal testosterone and with moderate likelihood at the estimated exposure levels, hypospadias, cryptorchidism and germ cell changes. The population of male children at risk is estimated to be in the range of 1.1 – 3.5 million over a time span of 20 years.

In addition, there is a moderate to strong probability for children suffering from immunological effects from exposure to the four phthalates and a moderate evidence for reduction of semen quality from exposure in adult men.

[178] Except increased nipple retention, but as explained earlier nipple retention is an indicator of foetal androgen suppression which is biologically relevant to humans.

Furthermore, there is a weak probability that the four phthalates cause delayed onset of puberty in boys and girls as well as delayed mammary gland development in women. Moreover, there is weak evidence for effects on female reproductive development, neurodevelopment and metabolism from exposure to the four phthalates during gestation, as well as weak evidence for liver carcinogenesis from exposure during adulthood.

These effects may increase the number of persons at risk as it includes additional populations since it not only encompasses boys but also girls, adult men and adult women.

Table D16 Summary of important elements in the evidence for human impacts resulting from exposure to DEHP, DBP, DIBP and BBP

Human health effects of concern	Are the effects observed in animal studies?	Are the effects observed in animal studies adverse & relevant to humans?	Epidemiological studies on phthalate exposure and human health effects of concern	Strength of relationship b/n exposure & human health impacts[179]	Indication of the monetary value of the human health concern
Effects from exposure during development: male reproductive effects					
Reduced semen quality	Reduced postnatal spermatocyte development quality (starting at 2 mg/kg bw/d for DBP, Lee et al. 2004)	Adverse and relevant. Part of the testicular dysgenesis syndrome (TDS) believed to have a common origin in foetal life.	It is estimated that 20-30% of young men today have sperm concentration associated with reduced fecundity which are thus at risk for prolonged waiting time to pregnancy, 10-15% with a sperm count so low they might require fertility treatment (Skakkebaek et al. 2016). It has been observed that there are important geographical differences in semen quality and in temporal trends of quality (e.g., decrease of 20% in sperm counts in Finish men between 1998-2006 but a 12-14% increase in Danish men between 1996-2010), which could be seen as evidence that environmental factors such as exposure to anti-androgenic chemicals may have a role in low semen quality. The sperm counts in Danish men (1996-2010) are 25 % lower than among infertile couples in the 1940s (Skakkebaek et al. 2016). It is very difficult to study associations with foetal exposure to phthalates and thus epidemiological evidence is scarce. Axelsson et al. (2015) found that prenatal exposure to phthalates was negatively associated with reproductive	Strong evidence from animal studies supported by epidemiological studies. Strong evidence based on exposure considerations. Overall strength: strong	Male infertility: estimated, see table D20

[179] The overall strength of the relationship between exposure estimates and human health impacts is rated as either weak, moderate or strong (≈likelihood or probability for human health impacts). The overall rating “strong” is given when both the evidence from animal studies and the evidence based on exposure considerations are strong. The overall rating “moderate” is given when (1) the evidence from animal studies is strong, but exposure considerations are moderate or weak; or (2) the evidence of both animal studies and exposure considerations are moderate. The overall rating “weak” is given in other cases where some evidence for effects from animal studies or epidemiology exists.

			function of young men, testicular size, semen quality, reproductive hormones. In adults several studies have found negative associations between phthalate exposure and semen quality, see "Effects from exposure during adulthood" below.		
Increased incidence of cryptorchidism Spermatogenic cells disappear in early childhood unless testes are in the proper position in the scrotum (Skakkebaek et al. 2016). Testes descend is hormonally regulated by testosterone and INSL3 secreted by Leydig cells (Skakkebaek et al. 2016).	Cryptorchidism starting at 5 mg/kg bw/d for DEHP (Andrade et al. 2006) and more clearly at higher dose levels starting at 250 mg/kg bw/d with DBP (Jiang et al. 2007)	Adverse and relevant. Part of the testicular dysgenesis syndrome (TDS) believed to have a common origin in foetal life.	Rising incidences in cryptorchidism were observed in EU countries, e.g. in DK from 1.8% (1959-1961) to 8.8% (1997-2001) and in the UK from 2.7% (end 50s) to 3.8% (end 80s), but not in FI (Skakkebaek et al. 2016). Main et al. (2006) found an association between phthalate levels in maternal milk and reproductive hormone profiles in infant males, indicating testicular function of the more exposed boys were affected. Swan et al. (2005) did not find an association between prenatal exposure based on maternal urinary phthalate levels and cryptorchidism but boys with decreased AGD were more likely to be cryptorchid than the boys with longer AGD, also indicating decreased virilisation. Cryptorchidism is a risk factor to infertility, testicular cancer and hypospadias (Skakkebaek et al. 2016).	Strong evidence from animal studies. The epidemiological studies examining phthalate exposures and cryptorchidism have not clearly been able to confirm the evidence from animal studies, but provide indirect support for the TDS hypothesis. Moderate-weak evidence based on exposure considerations. Overall strength: moderate	Cryptorchidism: estimated, see table D24 and Appendix D1
Increased incidence of hypospadias Hypospadias affect around 0.2 – 4% of boys at birth. Penile development is regulated by	Hypospadias starting at 250 mg/kg bw/d for DBP (Gray et al. 1999)	Adverse and relevant. Part of the testicular dysgenesis syndrome (TDS) believed to have a common origin in foetal life.	Rising incidences have been reported in EU countries since the 1970s (Toppari et al. 2001; Skakkebaek et al. 2016). Ormond et al. (2009) observed a 3 -fold increased risk of hypospadias among children of mothers that were exposed to phthalates in the workplace during pregnancy.	Strong evidence from animal studies. Limited support from epidemiological studies. Weak evidence based on exposure considerations.	Hypospadias: estimated, see table D27 and Appendix D1

dihydrotestosterone that is produced locally from testosterone (Skakkebaek et al. 2016).				Overall strength: moderate	
Testicular changes Reduced reproductive organ weight, tubular atrophy and reproductive tract malformations contribute to reduced male fertility (Skakkebaek et al. 2016).	Decreased testes and epididymides weight and testis seminiferous tubular atrophy (starting at 14 mg/kg bw/d for DEHP, Wolfe and Layton 2003) Malformed/degenerated seminiferous chords (starting at 15 mg/kg bw/d for DEHP, Andrade et al. 2006)	Adverse and relevant. Part of the testicular dysgenesis syndrome (TDS) believed to have a common origin in foetal life.	Axelsson et al. (2015) found that prenatal exposure to phthalates was negatively associated with reproductive function of young men, testicular size, semen quality, reproductive hormones.	Strong evidence from animal studies. Limited support from epidemiological studies. Strong evidence based on exposure considerations. Overall strength: strong	Male infertility: estimated, see table D17
Decreased foetal testosterone Decreased foetal testosterone is believed to contribute to reduced male fertility (Skakkebaek et al. 2016).	Decreased foetal testosterone starting at 50 mg/kg bw/d for DBP (Lehmann et al. 2004)	Adverse and relevant. Part of the testicular dysgenesis syndrome (TDS) believed to have a common origin in foetal life.	One study has found that elevated phthalate levels in breastmilk were associated with lower testosterone levels in newborn males (Main et al. 2006)	Strong evidence from animal studies, epidemiological studies not available. Strong evidence based on exposure considerations. Overall strength: strong	Decreased foetal testosterone is considered to affect several health outcomes, some of which have been monetised (male fertility, hypospadias, cryptorchidism, testicular cancer)
Decreased anogenital distance (AGD)	Decreased AGD (starting at 10/14 mg/kg bw/day for DEHP, Christiansen et al. 2010/Wolfe & Layton 2003)	Effects on AGD are believed to be early marker effects for other adverse reproductive effects observed for phthalates. Relevant to humans.	Several studies have found associations between reduced AGD in infants and prenatal maternal phthalates exposure in boys (Swan et al. 2005, 2015, Suzuki et al. 2012, Bustamante-Montes et al. 2013, Huan et al. 2009). A recent study from DK found no associations (Jensen et al. 2015).	Strong evidence from animal studies, some evidence from epidemiological studies. Strong evidence based on	Decreased AGD is considered a sensitive marker to exposure to anti-androgens and has been shown to be predictive of other effects, some of which have been monetised (male fertility,

		Considered to be adverse since it is a sensitive marker of androgen deficiency and effects on the male reproductive system and is part of the rat phthalate syndrome. Possibly part of the testicular dysgenesis syndrome (TDS) believed to have a common origin in foetal life.	Increasing evidence links reduced AGD in humans to the group of risk factors for TDS including testicular cancer, cryptorchidism, hypospadias and reduced semen quality (Skakkebaek et al. 2016). For example, Mendiola et al. (2011) showed a significant correlation between shorter AGD and poorer semen quality after in young men.	exposure considerations. Overall strength: strong	hypospadias, cryptorchidism, testicular cancer)
Germ cell changes / Increased incidence of testicular germ cell cancer Testicular cancer, cryptorchidism, hypospadias and reduced semen quality are risk factors for each other at an individual level and at the population level. Increasing evidence also link reduced AGD in humans to this group of risk factors (Skakkebaek et al. 2016). These risk factors often show evidence of dysgenesis in parts of the testicular tissue, including clusters of incompletely differentiated Sertoli	Multinucleated germ cells (MNGs) starting at 14 mg/kg bw/d for DEHP (Wolfe and Layton 2003)	Adverse (not fully elucidated). Relevant as also seen in human testes explant studies (van den Driesche et al. 2015). Developmentally abnormal germ cells might correspond to precursors of testicular germ cell cancer in humans (Ferrara et al. 2006) MNGs are hypothesized also to be associated to development of seminiferous chords (Gaido et al. 2007; Toppari et al. 2010). Rodents are not a good model for testicular cancer as they do not seem to develop the same kind of testicular	Incidences rising in all measured EU countries from 1980-2010, highest in DK and NO. Hereditary genetic factors can explain <25% of cases. Epidemiological evidence supports the foetal origin of TGCC, incidence peaks at 20-45 years suggesting an early onset of malignant process and immigration studies have shown that young men develop TGCC with same incidence as in their home countries whereas their sons born abroad acquired the risk of the host country. Further, testes show comprised development and function of foetal Leydig and Sertoli cells(Skakkebaek et al. 2016). Although the testicular germ cell cancer observed in humans is not seen in rodent studies, the other effects on male reproductive development as well as the dysgenesis of Sertoli, Leydig and germ cells in the testes suggest that the increasing incidences of testicular germ cell cancer in humans is part of the TDS of common foetal origin in both experimental animals and humans (Skakkebaek et al. 2016).	Strong evidence from animal studies for MNGs. Since no good rodent models appear to exist for studying human TGCC, it is unclear whether testicular carcinogenicity in humans may result from exposure to the four phthalates. Weak epidemiological evidence. Strong evidence based on exposure considerations. Overall strength: Strong for germ cell changes Weak for testicular germ cell cancer	Testicular cancer: €81 000 of direct, indirect and intangible costs of one testicular cancer case, estimated by Norden (2014). (See Appendix D2 for social impacts attributable to phthalates in articles.) ECHA (2014e) estimates (2012 values): Value of statistical life= €3.5 million, Value of statistical case of cancer = €350 000, Value of cancer morbidity = €410 000

cells and clustered Leydig cells. In animal studies, unusual clusters of undifferentiated germ cells and multinucleated germ cells have been observed across species (Skakkebaek et al. 2016).These developmentally impaired germ cells are proposed to correspond to precursors of testicular germ cell cancer in humans (Ferrara et al. 2006).		cancer as the prevalent form in humans, the testicular germ cell cancer (TGCC). Testicular cancer is part of the testicular dysgenesis syndrome (TDS) believed to have a common origin in foetal life. Adverse and relevant.			
Effects from exposure during development: effects in males and females					
Delayed age at puberty onset for girls and boys	Few experimental studies. In males, delayed preputial separation seen starting at 15 mg/kg bw/d for DEHP in Andrade et al. (2006). In females, delayed vaginal opening starting at 15 mg/kg bw/d for DEHP (Grande et al. 2006), see also delayed mammary gland development below.	Adverse. Relevant.	Ferguson et al. (2014) studied relationships between prenatal and childhood exposure to phthalates (as measured through urinary metabolites) with pubertal onset and sex hormones in boys (8-14 year). Prenatal exposure to several phthalates was associated with decreased dehydroepiandrosterone sulfate (DHEAS) and inhibin B levels, and with increased sexhormone-binding globulin (SHBG). Prenatal exposure to most phthalates was associated with greatly reduced odds of adrenarche and slightly reduced odds of puberty. Childhood exposure was associated with increased SHBG levels and decreased total and free testosterone levels. On the other hand, a trend to earlier onset of puberty in boys has been observed (decreasing age at testicular volume>3 ml) (Skakkebaek et al. 2016),	Overall, studies on phthalate exposure and puberty onset in animals and humans are equivocal. In rat studies prenatal phthalate exposure is associated with delayed – not advanced - puberty in female offspring, while exposure of prepubertal rats have been shown to advance the age of vaginal opening (Ma et al. 2006) so the effect of phthalate exposure may depend on the timing of exposure.	Hormone therapy may be required although more severe cases may lead to long term physical (including infertility) and behavioural or social problems.

			and also in girls (in DK breast development was observed to occur one year earlier in 2006/08 compared to 1991/93, see Aksglaede et al. 2009)	Overall strength: weak	
Persistent mammary gland changes	Alveolar atrophy and vacuolar degeneration of alveolar cells in adult males starting at 2 mg/kg bw/d for DBP (Lee et al. 2004)	Adverse. Relevance to humans unknown. Considered to be adverse since it is a sensitive marker of androgen deficiency. Alveolar atrophy may result from a decreased level of serum testosterone (OECD 2009). The mammary gland changes support the studies showing decreased testosterone levels induced by phthalates.	/	Some evidence from animal studies. Relevance to human males unknown. Strong evidence based on exposure considerations. Overall strength: Weak	Not monetised separately. Persistent mammary gland changes in male rats are considered a marker to exposure to anti-androgens. It may be considered to be predictive of other effects, several of which have been monetised (male fertility, hypospadias, cryptorchidism, testicular cancer)
Delayed mammary gland development	Hypoplasia of alveolar buds in prepubertal female mammary gland starting at 2 mg/kg bw/d for DBP (Lee et al. 2004)	Adverse. Relevant. Not examined for all phthalates. Available for DEHP and DBP.	Changes in onset of breast development can be examined in human epidemiological studies. Phthalates have been shown to delay puberty onset as described under 'Delayed puberty onset', and a delayed mammary gland development around puberty can be related to such changes in puberty onset.	Some evidence from animal studies. Overall, epidemiological studies are equivocal. Strong evidence based on exposure considerations. Overall strength: Weak	Hormone therapy may be required although more, severe case may lead to long term physical (including infertility) and behavioural or social problems.
Effects on female reproduction Effects on female reproduction are well described, though mechanistic studies are not as abundant as for male reproductive effects.	DEHP and DBP are classified for toxicity to reproduction in both males and females. Effects on female fertility are seen with adult exposure or in prenatally exposed females and are well documented in EU	Adverse and relevant to humans. (pregnancy, menopause, Polycystic Ovary Syndrome (PCOS), endometriosis, fibroids, preterm birth)	Overall, studies on female reproductive system show associations between phthalates and impaired ovarian development and impaired ovarian steroidogenesis. Increasing evidence of adverse consequences.	Some evidence from animal studies, limited support from epidemiological studies, data not always consistent between experimental and epidemiological studies (Gore et al. 2015). Weak	Female infertility: WTP value for statistical infertility €29 700/case. Preterm birth: WTP of statistical case of very low birth weight (2012

	RARs, effects starting at 500 mg/kg bw/d for DBP in adult females and 250 mg/kg bw/d for DBP in female offspring (Gray et al. 1999). New knowledge from mechanistic studies may show effects at lower dose levels. E.g. Meltzer et al. (2015) show effects of DEHP on steroidogenesis and ovarian Theca cells in female offspring starting from 50 mg/kg bw/d and Hannon et al. (2015) show effects on oestrous cyclicity and primordial cell counts in adult female mice starting from 20 mg/kg bw/d.			evidence based on exposure considerations. Overall strength: Weak	value) = €126 200 (ECHA 2014b)[180] <u>Endometriosis</u>: Social damage per case= €10,524 (2019 value) used in ECHA 2015, €8 620 (2010 value) weighted average per case used by Hunt et al (2016) <u>Fibroids</u>: Hospital costs for fibroid treatment average over €3 000. Health and lost productivity cost for fibroids and endometriosis[181] per woman in the EU = €8 000. Both quoted by Hunt et al (2016). <u>PCOS association with infertility</u> (see above), <u>diabetes, heart disease</u>): Average direct costs per case of adult diabetes as estimated by (Legler 2015): €29 600 (in 2010

[180] ECHA (2015c) refers to Rautava et al. (2009) which reports the results of a national study of all VLBW infants born in Finland between 2000 and 2003. 1,169 (900 live-born) children were compared against 368 full-term controls. Compared with the controls, 1.3 QALYs had been lost by each VLBW by age 5. This implies a discounted cost per case of around €75,000 based on the NewExt median VOLY. Given that VLBW is likely to result in negative health implications throughout the individual's life, the total cost would likely be higher than this figure.

[181] Together, endometriosis and fibroids represent the most common female reproductive disorders with an estimated combined incidence of up to 70% of women overall (Hunt et al 2016).

					values) (see Appendix D2 for the potential impact in the EU) WTP for a 1 in 1 000 000 risk reduction of heart disease= $4.82-$9.05 (2009 US$)[182]
Neurodevelopmental effects Exposure to phthalates may contribute to increasing incidences of autism spectrum disorders, ADHD, learning disabilities, altered play behaviour (Braun et al. 2013).	Animal studies examining behavioural effects of phthalate exposure have shown some effects that may be related to altered sex differentiation, whereas other behavioural effects are not clearly linked with disruption of sex hormones (reviewed in Miodovnik 2014)	Adverse. Relevant. Not fully clarified. Animal studies examining behavioural effects have shown some effects that may be related to altered sex differentiation, others have not found a link with disruption of sex hormones.	Autism spectrum disorders, attention deficit hyperactivity disorder (ADHD), learning disabilities, altered play behaviour have been associated with higher phthalate exposure in humans (Braun et al. 2013, Miodovnik 2014). Ejedar et al. 2015 (review) found associations between urinary phthalate concentrations and children's neurodevelopment (adverse cognitive and behavioural outcomes in children, including lower IQ, and problems with attention, hyperactivity and poorer social communication).	Some epidemiological evidence. The effects on neurodevelopment have not been a priority area in animal studies. Overall strength: weak	Autism: Average costs per case = €630 000 (in 2010 value) as estimated by Bellanger et al (2015) (see Appendix D2 for potential impact in the EU) ADHD: Average costs per case = €90 000 (in 2010 value) as estimated by Bellanger et al (2015) (see Appendix D2 for potential impact in the EU) Cognitive outcomes: WTP per IQ point = $466 (2007 US$) (Von Stackelberg et al 2009)
Effects on the metabolism	Metabolism Obesity, type 2 diabetes (T2D)	Adverse. Relevant. Not fully clarified. In vitro studies show some phthalates are PPARγ agonists, making them potential obesogenics e.g. by promoting differentiation and accumulation of lipid in lipid cells. Few in vivo	Few studies have been able to clarify the role of prenatal exposure to phthalates in the obesity epidemic (reviewed by Gore et al. 2015). Gestational diabetes (diabetes in the pregnant woman occurring during pregnancy) has become more frequent with increasing trends in obesity. In women with gestational diabetes, the	Emerging area of concern. Obesity and T2D have overlapping pathologies, but can also be affected separately. Current evidence suggests phthalates to act as obesogens and thereby generate	Diabetes: average direct costs per case of adult diabetes as estimated by Legler et al (2015): €29 600 (in 2010 value) (see Appendix D2 for potential impact in the EU)

[182] Valuation scenario is defined as 10 year latency, sick for 5 years, then death of a person with $42 000 income at 35, 40, and 65 years of age. Cameron et al (2009).

		studies are available to support the in vitro findings. In vivo, perinatal DEHP exposure has been shown to affect glucose homeostasis without affecting lipid accumulation. Reviewed in Gore et al. 2015.	risk of delivering a cryptorchid son is four-fold compared with non-diabetics.	insulin resistance and glucose intolerance increasing the risk of developing T2D. Overall strength: weak	Obesity: average direct & indirect costs per case of adult diabetes: €290 000 (in 2010 values) estimated by Legler et al (2015) (see Appendix D2 for potential impact in the EU)
Effect other than effects during development					
Immunological effects Exposure to phthalates may contribute to increasing incidences of allergy, asthma and eczema in both sexes.	In all studies with direct oral exposure (7), DEHP displayed adjuvant effects on airway hyperresponsivenes, atopic dermatitis or liver response. Confirmed with evidence from other routes and other phthalates. Current data show effects of DEHP in juvenile rats on immune parameters already at around 1 mg/kg bw/day (Guo et al. 2012; Tonk et al. 2012; Han et al. 2014).	Adverse. Relevant.	Case reports and epidemiological data show clear associations between PVC materials or phthalate exposure and increased immunological symptoms (asthma, other respiratory symptoms, rhinitis and eczema) (e.g., Braun et al. 2013, Bornehag et al. 2004, Hsu et al. 2012, Kolarik et al. 2008).	Fairly strong evidence from animal studies. Further robust animal data is needed for DNEL setting. Some epidemiological evidence. Strong evidence based on exposure considerations. Overall strength: moderate/strong	Asthma: WTP to avoid asthma discomfort = €50/episode in 2012 euro (ECHA 2014f) Allergy: WTP to avoid respiratory sensitisation = €17.5/episode in 2012 euro (ECHA 2014f) Eczema: WTP to avoid mild dermatitis = €227/episode in 2012 euro (ECHA 2014f)

Liver carcinogenesis	Activation of PPARα is an important mode of action for DEHP carcinogenicity, but the data suggest that multiple pathways in several cell types contribute to cancer in rats and mice (Rusyn and Corton 2012).	Unclear. IARC classification of DEHP 'possibly carcinogenic to humans (Group 2B)'	Overall body of evidence on human cancer hazard of DEHP remains inconclusive (Rusyn and Corton 2012).	Some evidence from animal studies. Overall body of evidence on human cancer hazard of DEHP remains inconclusive. DEHP is 'possibly carcinogenic to humans (Group 2B)' (Grosse et al. 2011; IARC 2012) Overall strength: weak	Liver cancer: ECHA (2014e) estimates (2012 value): Value of statistical life= €3.5 million, Value of statistical case of cancer = €350 000, Value of cancer morbidity = €410 000
Effects from exposure during adulthood					
Reduced semen quality from exposure during adulthood	Direct effects on adult fertility are described in the EU RARs, starting at 200 mg/kg bw/d for BBP. Reduced male fertility in F0/P0 generation is seen for BBP starting at 200 mg/kg bw/d (NTP 1997), DEHP starting at 592 mg/kg bw/d (Wolfe 2003), DBP starting at 500 mg/kg bw/d (Gray 1999)	Direct effects on male (and female) fertility are adverse effects in rats with relevance to humans. Direct effects on male (and female) fertility in rats occur at higher dose levels than adverse effects on male fertility when exposed during development.	In adults several studies have found negative associations between phthalate exposure and semen quality (Duty et al. 2003, Hauser et al. 2006, Pant et al. 2008, Pant et al. 2011, Jensen et al. 2015b, Huang et al. 2011, 2014). A recent meta-analysis by Cai et al. (2015) strengthens the evidence that the phthalates of concern adversely affect semen quality from exposure during adulthood. Relatively few of the epidemiological studies on semen quality in adult males represent the general population, as most are conducted among men being treated for infertility. The effect in the general population can thus be difficult to assess from these studies.	Strong evidence from animal studies, supported by epidemiological studies. Weak evidence based on exposure considerations. Overall strength: moderate	Adult male infertility in EU attributable to the four phthalates in articles = €7 630 per case (in 2010 euro) in terms of direct costs for ART, estimated on the basis of Hauser et al (2015) (see Appendix D2 for potential impacts in the EU)
Low testosterone levels in adult men While individual men with poor sperm concentration might have testosterone levels within normal range, as a group,	Lower testosterone levels in adults (may also be due to exposure of foetus) Direct effects on male fertility in rats occur at higher dose levels than adverse effects on male	Adverse. Relevant.	Secular trends consistently report declining total testosterone levels in adult men, after adjusting for covariates like overweight, type 2 diabetes and a sedentary life style (Skakkebaek et al. 2016). Several studies exist showing negative associations between phthalates	Some epidemiological evidence. Relatively few of the epidemiological studies on semen quality and testosterone levels in adult males represent	Low testosterone leading to increased mortality in EU attributable to the four phthalates in articles = €320 700 per case (in 2010 values) in terms of loss of economic productivity, estimated on the basis of Hauser et

subfertile men have lower serum testosterone than fertile men, indicating impaired Leydig cell function (Skakkebaek et al. 2016).	fertility when exposed during development and at higher dose levels at adulthood compared with juvenile rats. EU RARs describe effects of phthalates on decreasing testosterone levels in adult rats, for DEHP starting at 100 mg/kg bw/d for prepubertal rats, but with no effect in adult rats (Akingbemi et al. 2001).		exposures and testosterone levels in adult men (Mendiola et al. 2012, Mendiola et al. 2011, Meeker er al. 2009, Pan et al. 2006).	the general population, as most are conducted among men being treated for infertility. The effect in the general population can thus be difficult to assess from these studies, but overall there may be an association between phthalate exposure and changes in male sex hormones from the general population. Overall strength: weak	al (2015) (see Appendix D2 for potential impacts in the EU)

D.3.5.2. Environmental impacts

The Member State Committee (MSC) unanimously agreed in December 2014 to identify DEHP as a substance of very high concern under REACH on the basis that it gives rise to an equivalent level of concern due to its endocrine disrupting properties to the human health and the environment, according to Article 57(f) of REACH. The MSC opinion states that scientific evidence shows that exposure during sensitive time windows of development may cause irreversible developmental programming effects leading to severe effects on development and reproduction, regarded as particularly serious in relation to human health and wildlife species, also because these adverse effects may first manifest themselves in later life stages as a consequence of exposure during early life stages.

Exposure to DEHP is reported to affect steroidogenesis (e.g. decreased foetal testosterone production) resulting in adverse effects in the male reproductive system (e.g., effects on sex ratio, ovo-testis) in a range of species across taxonomic groups representative of both aquatic and terrestrial environmental compartments. DEHP appears to act via relatively weak anti-androgenic or oestrogenic mechanisms. However, effects that could be mediated by the thyroid axis have also been noted by some authors for some species of fish and amphibians (ECHA 2014).

The ECHA support document (ECHA 2014) outlines that DEHP may adversely affect the reproductive ability of fish populations by changing male fish into female fish and may, according to some studies, directly reduce fish fecundity. Such reproductive effects are considered an adverse and serious effect with population level relevance associated to the long-term sustainability of fish populations, particularly because of the apparent irreversibility of effects (e.g. changes in sex ratio). The developmental and reproductive effects of DEHP observed in rats are also considered to be of particular concern in relation to mammalian wildlife including top predators (including endangered species), where the described reproductive effects are expected to cause serious effects at the population level because of a natural low reproductive output of such taxa (ECHA 2014).

These potential impacts are not monetised but just presented here qualitatively.

D.3.5.3. Risk reduction capacity

It is assumed that the entry into force of the restriction will be 1 January 2020 and the temporal scope is 20 years. Thus the risk reduction capacity for the proposed restriction is assessed for the period 2020-2039.

The level of risk is based predominantly on urinary samples taken in September 2011 until February 2012. RCRs were at or above 1 at the 95th percentile exposure level in 13 out of 15 Member States (86%) in 2014 (projected risk). In Member States with an RCR = 1 or an RCR < 1 at the 95^{th} percentile exposure level, there was still a risk for individuals with the highest exposure levels in the study population in 2014. It can be concluded that there was a risk in all Member States in 2014.

As concluded in section B.9.3 of Annex B, evaluation of the uncertainties to the RCRs generally point to possible underestimation of the RCRs[183]. It is concluded that a risk has been identified that is not adequately controlled. The proposed restriction avoids placing on the market of an estimated 131 560 tonnes per year of the four phthalates in articles in the period 2020-2039 (min 120 840 tonnes/year, max 143 510 tonnes/year).

As a result of the effect of authorisation, the volumes are projected to decline until 2019. It is thus expected that future risks will be lower than those observed in 2011-2012. However, as a consequence of increased imports the volumes and risks are projected to gradually increase again from 2020 onwards.

The risk in the absence of the restriction in 2020 and 2039 may be projected based on the estimates of the future market of the four phthalates (the baseline). Any results of such exercise needs to be interpreted with great caution since first, the market volumes are projections themselves and associated with significant uncertainty. Second, as RAC (2012) remarked, there is no simple one-to-one relationship between volumes placed on the market and exposure levels. In other words, the percentage decline in volumes does not translate in an equal percentage decline in exposure. Possible reasons for the relationship between marketed volumes and exposure to be blurred are:

- the volume decline may not be uniform across all market segments;
- articles from certain market segments may lead to higher exposures in proportion to their volume compared with other articles[184]; and
- the length of the service-life influences the relationship between marketed volumes and exposure levels.

Bearing in mind the above caveats, a projection of future risks was attempted. Since the DEMOCOPHES biomonitoring samples used in the exposure assessment were taken in the period September 2011 until February 2012, the reference year for the risk assessment can be assumed to be 2011. It can furthermore be assumed that exposure via food is not affected by the declining baseline because the authorisation requirements do not apply to food contact materials (FCMs). It is assumed that FCMs such as food packaging and articles that are used during the processing of food (e.g., tubes, gloves, tools, recipients, etc.) are the principle source of food contamination[185]. Section B.8.3.2 of Annex B concluded that 75% of the exposure to DEHP is from food intake and 25% from other sources that are considered to be covered by the scope of the restriction. For DBP, DIBP and BBP the situation is inverse and only 25% of the exposure is from food intake and 75% from other sources included in the scope of the restriction. In other words, under the above assumptions, the impact of the baseline projections will be lower for DEHP in comparison to the other three phthalates, in

[183] Amongst others, using volume based method of intake calculation instead of the creatinine method we used possibly doubles the RCRs; children younger than 6 are likely to have higher exposure.

[184] As shown in Annex B, e.g., erasers, sext toys and sandals are examples of articles that may lead to high exposure. Also for example extensive use of mobile phones may lead to extensive dermal exposure to phthalates in cell phone covers.

[185] Non-FCM articles may come into contact with food and environmental contamination may contribute to food contamination as well, but are thought to be minor sources of food contamination.

particular DBP and DIBP which together are responsible for the highest contribution to the combined risks (see section B.9.1).

If under the above assumptions a one-to-one relationship between the baseline volumes and the proportion of risk from articles in the scope is assumed, the projected risks for 2020 and 2039 in the main baseline scenario are as presented in Table B52 and Table D19. The projected risks in the low and high tonnage baseline scenarios are not substantially changing the picture (see section E). Similarly, the projected RCRs are not very sensitive to the assumptions taken regarding the contribution of food (results not presented[186]). During the public consultation a biomonitoring study from Germany showed that the projections of the RCRs for the year 2015 are reasonable[187].

The number of boys at risk in the period 2020 - 2039 due to foetal exposure to the four phthalates is estimated to be 54 000 per year, or 2.1% of new born boys (1.1 million boys over 20 years). Using the exposure values from children, it is estimated that 175 000 boys per year are at risk, or 6.8% of new born boys (3.5 million boys over 20 years).

[186] As an indication, if it would be assumed that the contribution of food to DEHP exposure is only 25% and that of DBP, DIBP and BBP 10%, the RCR for Polish children in the low tonnage baseline scenario is 1.3 in both the 2020 and 2039 projections (risk in 3 countries).

[187] The impact of the predicted decline in exposure from 2011 to 2015 on the RCRs are not explicitly shown in the BD but the predicted RCRs for 2015 and 2020 are nearly equal and indicate there is still a risk. The BD assumes a decline of 1.8% for DEHP of the 95th percentile of exposure from 2011 to 2015 whereas the data in Koch et al. (2016) suggests a decline of 11%. Similarly, for DBP, DIBP and BBP the BD assumed a decline of 50% of the 95th percentile of exposure whereas the data in Koch et al. (2016) suggests a decline of about 17.5%. Thus, the projections in the BD underestimate the decline in exposure to DEHP but might overestimate decline in exposure for DBP, DIBP and BBP. It should be stressed that the exposure to DBP and DIBP has the highest impact on the RCRs and thus, overall, based on the data in Koch et al. (2016), the RCRs for combined exposure to the four phthalates is underestimated rather than overestimated. In fact, the data suggests that for Germany, the BD underestimates the combined RCR with 42%. On average (all EU countries in DEMOCOPHES) the BD underestimated the RCRs on average by 37%. Moreover, the trend in DE is not necessarily representative of the whole EU and the population in Koch et al. (2016) is very homogeneous (students of 20-29y from 4 university cities) in contrast to the EU population. Some of the trend between 2011 and 2015 may also be due to statistical fluctuation in the relatively small sample size (n=60) in Koch et al. (2016). In conclusion, the projections made in the BD are reasonable and the need for a restriction is not challenged by the new study by Koch et al. (2016).

Table D17 RCRs for four phthalates as estimated from 95th percentile urinary biomonitoring values projected to 2020 *in the main baseline scenario (no restriction)*

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.2	0.0	*NA*	0.4	120	0.2	0.2	0.0	*NA*	0.4
UK	21	0.1	0.1	0.0	0.1	0.3	21	0.1	0.1	0.0	0.1	0.4
CH	117	0.2	0.1	0.0	0.1	0.4	119	0.2	0.1	0.0	0.1	0.5
CY	59	0.4	0.1	0.0	0.2	0.7	60	0.2	0.1	0.0	0.2	0.6
LU	60	0.1	0.1	0.0	0.1	0.4	60	0.1	0.1	0.0	0.4	0.6
PT	117	0.3	0.1	0.0	0.2	0.6	116	0.2	0.2	0.0	0.2	0.6
IE	120	0.2	0.1	0.0	0.2	0.5	120	0.3	0.1	0.0	0.2	0.7
DE	116	0.1	0.2	0.0	0.1	0.4	120	0.2	0.3	0.0	0.2	0.7
DK	143	0.1	0.1	0.0	0.2	0.4	142	0.2	0.2	0.0	0.3	0.7
HU	115	0.2	0.2	0.0	NA	0.5	117	0.4	0.3	0.0	NA	0.7
SE	96	0.2	0.4	0.0	NA	0.5	97	0.3	0.5	0.0	NA	0.8
SK	125	0.2	0.4	0.0	NA	0.6	127	0.4	0.6	0.0	NA	1.0
CZ	117	0.2	0.4	0.0	NA	0.6	120	0.4	0.7	0.0	NA	1.1
BE	125	0.1	0.2	0.0	0.3	0.6	125	0.3	0.2	0.0	0.5	1.1
ES	118	0.2	0.2	0.0	0.2	0.6	119	0.3	0.5	0.0	0.4	1.2
RO	117	1.0	0.1	0.0	0.2	1.3	119	0.8	0.3	0.0	0.3	1.4
PL	119	0.3	0.4	0.0	0.4	1.1	115	0.5	0.6	0.0	0.6	1.7

NA = not available

Yellow marking highlights RCR levels above 1 or equal to 1 when rounded

Table D18 RCRs for four phthalates as estimated from 95th percentile urinary biomonitoring values projected to 2039 *in the main baseline scenario (no restriction)*

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.2	0.0	*NA*	0.4	120	0.2	0.2	0.0	*NA*	0.4
UK	21	0.1	0.1	0.0	0.1	0.3	21	0.2	0.2	0.0	0.2	0.5
CH	117	0.2	0.2	0.0	0.1	0.4	119	0.2	0.2	0.0	0.1	0.5
CY	59	0.4	0.1	0.0	0.2	0.8	60	0.2	0.1	0.0	0.2	0.6
LU	60	0.1	0.1	0.0	0.2	0.4	60	0.1	0.1	0.0	0.4	0.7
PT	117	0.3	0.1	0.0	0.2	0.6	116	0.3	0.2	0.0	0.2	0.7
IE	120	0.2	0.1	0.0	0.2	0.5	120	0.3	0.1	0.0	0.3	0.7
DE	116	0.1	0.2	0.0	0.1	0.4	120	0.2	0.3	0.0	0.2	0.7
DK	143	0.2	0.1	0.0	0.2	0.5	142	0.2	0.2	0.0	0.3	0.7
HU	115	0.2	0.3	0.0	NA	0.5	117	0.4	0.4	0.0	NA	0.8
SE	96	0.2	0.4	0.0	NA	0.6	97	0.3	0.5	0.0	NA	0.9
SK	125	0.2	0.4	0.0	NA	0.7	127	0.4	0.6	0.0	NA	1.0
CZ	117	0.2	0.4	0.0	NA	0.7	120	0.4	0.7	0.0	NA	1.2
BE	125	0.1	0.2	0.0	0.3	0.7	125	0.3	0.2	0.0	0.6	1.2
ES	118	0.2	0.2	0.0	0.2	0.6	119	0.3	0.5	0.0	0.5	1.3
RO	117	1.0	0.1	0.0	0.2	1.3	119	0.9	0.3	0.0	0.3	1.5
PL	119	0.4	0.5	0.0	0.4	1.2	115	0.5	0.6	0.0	0.7	1.8

NA = not available

Yellow marking highlights RCR levels above 1 or equal to 1 when rounded

Under the above assumptions, the proposed restriction would remove all exposure to articles in the scope of the restriction proposal from 2020 onwards. In other words, the risk from 2011 would be reduced by 25% for DEHP and 75% for DBP, DIBP and BBP. As can be seen in Table D20, the proposed restriction would be able to reduce the combined risk to projected levels below 1, except in Romania and Poland where RCRs are projected to be around 1 following the entry into force of the proposed restriction.

Table D19 RCRs for four phthalates as estimated from 95th percentile urinary biomonitoring values *in case of a restriction (2020 and onwards)*

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.1	0.1	0.0	*NA*	0.2	120	0.2	0.1	0.0	*NA*	0.3
UK	21	0.1	0.0	0.0	0.1	0.2	21	0.1	0.1	0.0	0.1	0.3
CH	117	0.1	0.1	0.0	0.0	0.2	119	0.2	0.1	0.0	0.1	0.3
LU	60	0.1	0.1	0.0	0.1	0.2	60	0.1	0.1	0.0	0.2	0.3
CY	59	0.3	0.0	0.0	0.1	0.5	60	0.2	0.1	0.0	0.1	0.3
PT	117	0.3	0.1	0.0	0.1	0.4	116	0.2	0.1	0.0	0.1	0.4
DE	116	0.1	0.1	0.0	0.1	0.2	120	0.2	0.1	0.0	0.1	0.4
DK	143	0.1	0.0	0.0	0.1	0.3	142	0.2	0.1	0.0	0.1	0.4
IE	120	0.1	0.1	0.0	0.1	0.3	120	0.2	0.1	0.0	0.1	0.4
HU	115	0.2	0.1	0.0	NA	0.3	117	0.3	0.2	0.0	NA	0.5
SE	96	0.1	0.2	0.0	NA	0.3	97	0.2	0.2	0.0	NA	0.5
SK	125	0.2	0.2	0.0	NA	0.4	127	0.3	0.3	0.0	NA	0.6
BE	125	0.1	0.1	0.0	0.2	0.4	125	0.3	0.1	0.0	0.3	0.6
CZ	117	0.2	0.2	0.0	NA	0.4	120	0.3	0.3	0.0	NA	0.6
ES	118	0.2	0.1	0.0	0.1	0.4	119	0.3	0.2	0.0	0.2	0.7
RO	117	0.8	0.1	0.0	0.1	0.9	119	0.7	0.1	0.0	0.2	1.0
PL	119	0.3	0.2	0.0	0.2	0.7	115	0.4	0.3	0.0	0.3	1.0

NA = not available

Yellow marking highlights RCR levels above 1 or equal to 1 when rounded

Conclusion

It can be concluded that the proposed restriction is capable of significantly reducing the risks to human health of combined exposure (RCRs are expected to be reduced to levels equal to or below 1 at the 95th percentile) within a reasonable period of time, starting from 2020, although with some delay is caused by the service-life of articles in use. Considering the important contribution of food consumption to exposure to the four phthalates, in addition to the proposed restriction, the relevant authorities in the EU are encouraged take the necessary measures to reduce the risks relating to the four phthalates from food consumption. Any associated risks for the environment from the articles in scope would also be reduced as a result of the proposed restriction (see Appendix B.2 of Annex B). The proposed restriction may furthermore reduce occupational risks due to substitution of DEHP in the production of articles in the EU.

If it is concluded that no threshold exists for the endocrine properties of the four phthalates, there would be a remaining risk following the entry into force of the proposed restriction. In this case, the restriction would contribute to reducing the exposure and thus the remaining risk.

D.3.5.4. Quantification and monetisation of impacts

As concluded in section D.3.5.1, male children are at risk of suffering irreversible damages to the male reproductive system due to exposure (*in utero* or after birth) to the four phthalates. The risk for reduced semen quality, testicular changes and feminisation[188] is particularly pronounced, but also cryptorchidism and hypospadias, albeit with lower probability. Possibly,

[188] Feminisation in this context refers to permanent retention of nipples/areolae in male rats and reduced AGD (see section D.3.5.1)

some male children may be at increased risk for testicular germ cell cancers and delayed puberty as well. Further effects might also result from phthalate exposure, but the current scientific evidence is inconclusive (see Table D17).

The following section estimates the damage to society of male infertility associated with exposure to the four phthalates in articles for the purpose of demonstrating that the benefits of risk reduction outweigh the costs of the proposed restriction. This damage would be avoided as a result of the proposed restriction, i.e., it represents the benefits of the proposed measure. Detailed results of the quantification and monetisation of the impacts of the proposed restriction are included for cryptorchidism and hypospadias in Appendix D1. Appendix D2 to this Annex presents the results of other valuation studies of phthalates and endocrine disrupting chemicals (EDCs). Table D17 gives an indication of the benefits to society if some of those other, non-monetised impacts can be avoided. Further quantification and monetisation of those benefits was not pursued due to considerable uncertainty related to the estimation of the number of cases associated with exposure to the four phthalates.

a) Damage to society due to male infertility

Reduced semen quality can lead to infertility, which can lead to significant emotional anguish and to financial costs in the event a couple pursues assisted reproductive treatment (ART).

Infertility is defined by the World Health Organisation as *"the inability of a sexually active, non-contraception couple to achieve spontaneous pregnancy in one year"*. A male infertility associated factor is found responsible in 50% of involuntary childless couples (about 15% of couples) (EAU 2015). Male fertility can be affected by a host of factors such as genetic abnormalities, urogenital tract infections, malignancies, endocrine disturbances, lifestyle and others. About 30-40% of cases however are idiopathic in nature (EAU 2015). One of the most common ART for male infertility is intracytoplasmic sperm injection (ICSI) performed by *in vitro* fertilisation (IVF).

The social damage of male infertility due to exposure to the four phthalates in articles is estimated on the basis of the number of cases, derived from current incidence rates and monetised using direct and indirect costs per case gathered by Norden (2014) and intangible costs presented in terms of the willingness to pay value (WTP) of statistical infertility, estimated by ECHA (2013b). The approach is similar to that used by Norden 2014 (see Appendix D2) and AFA 2013a, for example.

As there is no published aetiological fraction[189] of the number of male infertility cases associated with exposure to phthalates, this fraction was estimated on the basis of the incidence of male infertility due to causes that could be associated with exposure to chemicals, e.g., idiopathic infertility, testicular tumours, cryptorchidism and some forms of hypogonadism (see step c in Table D21). As it is uncertain whether all these cases can be attributed to exposure to chemicals, this fraction is reduced and the assumption is tested in the sensitivity analysis (see step d in Table D21). The last step in the estimation of the aetiological fraction (see step e in Table D21) is to account of the fact that: i) in addition to phthalates, there are other chemicals with effects on the male reproductive system; and ii) the articles in the scope

[189] It indicates the proportion of new cases of a disease within a population that can be said to be due to (i.e., attributable to) a particular exposure.

of this dossier are not the only source of exposure to the four phthalates. (See the note in Table D21 for detailed explanation of the derivation in step e.)

Table D20 Infertility incidence related to exposure to DEHP, DBP, DIBP, and BBP in articles (EU28)

Step in analysis		Percent	Cumulative	Source
a	Couples who do not achieve pregnancy within 1 year and seek medical treatment for infertility	15%		EAU 2015
b	Of a), those with male-infertility-associated factor together with abnormal semen parameters	50%	7.5%	EAU 2015
c	Of b), those whose diagnosis may be associated with exposure to chemicals with anti-androgenic mode of action and other unknown causes (e.g., idiopathic infertility, hypogonadism - primary of unknown cause, constitutional delay of puberty, infertility of possible causes - testicular tumour, maldescended testes, other)	54%	4.1%	EAU 2015
d	Of c), those that can be associated with exposure to chemicals only Sensitivity - low estimate Sensitivity - high estimate	50% 25% 75%	2.0% 1.0% 3.0%	AFA 2013a, Norden 2014, HEAL 2014, WHO/UNEP 2012
e	Of d), those that can be attributed to exposure to the four phthalates in articles* - mid-point estimate Sensitivity - low estimate Sensitivity - high estimate	4%	0.08% 0.04% 0.12%	*

Notes: * The 4% is a composite percentage equal to:

- the share of cases due to exposure to phthalates: 11%. This assumption is on the basis of a report to the European Commission DG Environment (Kortenkamp et al., 2011) which identified nine groups of chemicals of concern in the EU28 that are associated with adverse impacts on male reproductive health.
- the fraction of cases attributable to exposure to DEHP, DBP, DIBP, and BBP: 90%. It is estimated on the basis of world tonnages usage of all phthalates that have the same mode of action, weighed by their relative hazard (on the basis of their oral DNEL). This estimate accounts for the possibility that part of the four phthalate tonnages would be replaced by DINP (about 55%), as DINP has similar anti-androgenic mode of action (but at 7-8 times time higher doses than DEHP on the basis of DNELs).[190]
- the fraction of cases attributable to exposure to the four phthalates in articles: 40%, estimated on the basis of modelling of exposure to articles and the estimated share of non-food sources of exposure to the four phthalates (25% for DEHP and 75% for DBP, DIBP and BBP) weighed by their tonnages.

Therefore, on the basis of the estimated aetiological fraction in Table D21, the number of male infertility cases associated with exposure to the four phthalates in articles is estimated at more than 2 110 (see Table D23). This represents the estimated number of male children that experienced diminished androgen activity during critical foetal development (i.e., less than 4% of the population at risk due to foetal exposure) or early childhood (i.e., less than 1.5% of the population at risk due to exposure during infancy and early childhood) due to exposure to

[190] It is important to remind that for the purpose of estimating the costs of the restriction the alternatives were selected on the basis of convenience (due to less confidentiality issues related to the critical data required for the analysis). As discussed in the section on Alternatives and Substitution costs, there are other alternatives with similar (or better) technical and economic feasibility which also have more benign risk profile than DINP.

phthalate containing articles in the scope of the restriction. These male children would experience direct, indirect or intangible costs from their desired age of fatherhood and onward. These social costs would be avoided in the event of the proposed restriction enters into force.

To estimate the direct, indirect and intangible costs associated with ART, we assume, similar to Norden (2014), three scenarios: successful and unsuccessful outcome of ICSI treatment and no ICSI treatment. Table D22 shows how these costs are varied in each scenario.

Table D21 Scenarios for estimating the number of cases and total social damage related to male infertility (EU28)

	ICSI treatment*			
Assumptions	**Outcome: live birth**	**Outcome: no live birth**	**No treatment**	**Source**
Proportion of infertile cases	40%	18%	42%	Norden 2014
Average number of cycles of treatment	2.53	5.00	0	Norden 2014
Cost per ICSI cycle (€)	2 830	2 830	0	Norden 2014
Time spent on treatment (hrs)	35	35	0	Norden 2014
Average hourly labour cost in the EU28 in euro (costs for wages and salaries plus non-wage costs such as employers' social contributions)	24.60	24.60	N/A	EuroStat, 2014 data
Proportion with intangible costs (WTP value of statistical case of infertility)	0%	100%	50%	**
Intangible costs per infertility case (€)	0	29 710	29 710	ECHA 2014b[191]

Notes: All costs in 2014 euro

* Indirect health care costs attributable to ART (assisted reproductive treatment) of €0.3 million were also included in the ICSI (intracytoplasmic sperm injection) treatment scenario in accordance with Connolly 2010 which estimates that ART can account for 0.25% of public health service budgets. The rate was applied to EuroStat 2012 data and adjusted to 2014 values and the share of infertility cases attributable to the four phthalates in articles.

** It is assumed that all who do not have a baby as a result of ART will suffer intangible costs. Those who have a child as a result of the treatment are assumed not to suffer emotional damages. This is an underestimation as ECHA 2014b shows that couples are willing to pay to conceive faster. Only 50% of those who do not pursue treatment are assumed to suffer psychological pain, as it is uncertain whether they do not pursue treatment because they do not value having a child or because of other motives.

Assuming that the proposed restriction enters into force in 2020, the cases of male infertility avoided as a result would occur between 2020 and 2039 (the assumed end of the study period). However, as mentioned, the benefits would begin to materialise only at the time of their desired age of fatherhood, assumed to be 30.[192] The nominal value of the social damage that would be avoided as a result of the introduction of the proposed restriction is more than

[191] ECHA 2014b refers to other studies which higher WTP values: e.g., in the study by Neumann and Johannesson (1994), the WTP per statistical baby ranged from $40 640 ($63 156 in USD 2010) to $1 730 000 ($2 688 461 in USD 2010). Several other studies also derive higher WTP values than ECHA 2014b.

[192] According to respondents from EU25 in Eurobarometer 2006, 27 years was the ideal age of having children for men. However, the mean actual number of children per age group was as follows: 0 for men below 24 years of age, 1 for 25-39, 1.7 for 40-54, and 2 for 55+. Thus, 30 was chosen as the age for desired fatherhood for the purpose of this analysis.

€40 million annually on average from 2050 onward as presented in Table D23. The present value of this average annual social damage is €9.8 million annually after discounting with the standard social time preference rate of 4%, also used for the costs of the proposed restriction. However, this does not take into account that the income elasticity of the value of health is one; therefore, an increase in wealth in the future would lead to an equivalent increase in the value of health. To address this issue, the UK Health and Safety Executive recommends that the value of preventing a health outcome is uprated in real terms each year by real GDP per capita growth, i.e., by about 2% per year, [193] which is also consistent with past long-term growth. Taking this into account, the discounted value of the social benefits of avoided male infertility due to the proposed restriction is €19.6 million annually.

Table D22 Summary of estimated social damage related to male infertility due to exposure to DEHP, DBP, DIBP and BBP in articles in scope (EU28)

Steps in analysis	**Low**	**Mid-point**	**High**
Average annual male births (EuroStat, 2020-2050)	2 600 000	2 600 000	2 600 000
Fraction of cases of infertility attributable to DEHP, DBP, DIBP, and BBP in articles	0.04%	0.08%	0.12%
Annual number of cases of infertility due to DEHP, DBP, DIBP, and BBP in articles	1 050	2 110	3 160
Direct costs*	5 780 000	11 560 000	17 340 000
Indirect costs**	2 288 000	4 046 000	5 805 000
Intangible (WTP)***	12 224 000	24 447 000	36 671 000
Total annual social costs of male infertility (from 2050 onward)	20 292 000	40 053 000	59 816 000
weighted average per case	19 230	18 980	18 900
Total annual social costs of male infertility (discounted to 2014 with 4% discount rate)	4 944 000	9 760 000	14 575 000
weighted average per case	4 690	4 630	4 610
Total annual social costs of male infertility (discounted to 2014 with 2% effective discount rate)	9 947 000	19 635 000	29 323 000
weighted average per case	9 430	9 310	9 270

Note: 2014 values, average, representative year analysis.
* Direct costs in this case include costs per treatment for an average number of ICIS cycles (see Table D22).
** Indirect costs in this case include productivity loss of patient as well as overhead public health case spending attributable to ART (assisted reproductive spending) (see Table D22).
*** Intangible costs presented in terms of the WTP value of statistical infertility, estimated by ECHA (2014b). (See Table D22 for scenarios for estimating intangible costs.)

b) Summary and uncertainties of estimating damage to society of male infertility

In addition, despite being comparable to other studies (see Appendix D2), the analysis presented in Table D23 may be underestimating the damage to society of male infertility because:

- Impacts on the male reproductive system lead to a number of health conditions which are closely associated (or lead) to male infertility. These could entail years of

[193] http://www.hse.gov.uk/risk/theory/alarpcba.htm#footnotes

mental anguish and financial cost for diagnosis and treatment prior to the date of desired fatherhood. These are not captured in the presented estimates.

- Not all males who have experienced infertility are captured in the statistics used to derive the incidence rate of exposure to the four phthalates. For example, a fertile partner may compensate for the infertility of a man (EAU 2015) and couples may achieve spontaneous pregnancy in more than one year. If these couples have not sought treatment, they are not captured in the incidence rates used in the analysis. In this case, the costs associated primarily with the mental anguish of not being able to conceive for an extended period of time are not presented above. Those costs could be considerable, as ECHA 2014b shows individuals are willing to pay to reduce the time to pregnancy.
- Other reasons why the direct, indirect and intangible costs may not fully capture the total social damage associated with male infertility is because, e.g., couples may wish to have more than one child. In this case, these may be further direct and indirect costs for ART and if unsuccessful, the couple would suffer intangible costs. In addition, ART is a long process and even if successful, the couple may suffer mental anguish for the duration. For simplicity, the analysis assumes that these would be zero (see Table D22).
- The standard social time preference rate of 4% does not take into account that the income elasticity of the value of health is one; therefore, an increase in wealth in the future would lead to an equivalent increase in the value of health. As explained above, the 2% may be more appropriate for discounting human health benefits.

At the same time, a considerable uncertainty is associated with the estimated aetiological fraction of infertility cases due to exposure from the four phthalates. The analysis has reduced this uncertainty by making a number of informed assumptions, in particular by selecting only these cases of infertility whose cause may be exposure to chemicals. However, what percentage of those could be associated solely with chemicals remains unknown. The analysis uses estimates that have been applied elsewhere, e.g., Norden 2014, AFA 2013a, HEAL 2013. However, these studies also recognise the high uncertainty associated with the share of incidence due to exposure to chemicals. Therefore, the scenarios presented here and in Annex E, may not show accurately the upper and lower bound of the value of social damage due to the four phthalates in articles.

c) Other human health and environmental impacts

In addition to reduced semen quality and in severe cases infertility, among the most pronounced damages are cryptorchidism and hypospadias. They are often risk factors for each other (including testicular cancer) and together they are hypothesised to comprise the testicular dysgenesis syndrome (TDS) (Norden 2014).

Appendix D1 presents supplementary information on avoided social damage of cryptorchidism and hypospadias with the entry into force of the restriction. These effects are established in experimental animals and are considered relevant and adverse to humans. Based on the current evidence in animals, these additional effects might be expected to occur in the population at higher exposure levels than those exposure levels estimated on the basis of biomonitoring. However, mild incidences of cryptorchidism were in fact seen at dose levels corresponding to the DNEL for DEHP (Andrade et al. 2006). This, and the fact that these malformations are a part of the TDS, casts doubt on this conclusion. It may therefore be

necessary to extend the conclusion that a risk has been identified to the whole spectrum of effects in the rat phthalate syndrome observed in animals. Furthermore, the many uncertainties in hazard and exposure assessment need to be kept in mind, including the uncertainty whether a threshold exists for these substances as endocrine disruptors. For these reasons, it was considered important to provide estimates of the potential social damage of cryptorchidism and hypospadias.

The analysis in Appendix D1 employs a similar approach to the methodology used to estimate the social damage of male infertility. It shows that approximately 480 cases of cryptorchidism and 540 of hypospadias can be associated with exposure to the four phthalates in articles in the scope of this restriction proposal. Their direct, indirect and intangible costs are estimated to more than €13.9 million and €9.1 million annually. The total damage to society from male infertility, cryptorchidism and hypospadias due to exposure to the four phthalates in articles in the scope of this proposal are in excess of €32.8 million annually (Table D24). The results are comparable with the results of other studies. For example, if the benefits are derived on the basis of the impacts estimated by Norden (2014), the total social damage due to exposure from the four phthalates in articles would be €23.7 million (although this estimate does not include the WTP to avoid infertility).[194] See Appendix D2 to this Annex for the results of this and other valuation studies of phthalates and endocrine disrupting chemicals (EDCs).

Table D23 Damage to society from male infertility, cryptorchidism and hypospadias due to exposure to DEHP, DBP, DIBP and BBP in articles in scope: summary, EU28

2014 euro - annual, million	Low estimate	Mid-point estimate	High estimate
Male infertility	4.9	9.8	14.6
Cryptorchidism	1.2	13.9	39.7
Hypospadias	0.9	9.1	22.8
Total	**7.1**	**32.8**	**77.1**

Notes: All values discounted to 2014 with 4% social time preference rate. Average, representative year analysis. See Appendix D1 for details on estimation of impacts related to cryptorchidism and hypospadias.

Appendix D1 outlines the reasons the monetary value of the benefits of avoided cases in Table D24 can be underestimated. Many of them are already raised regarding the monetisation of male infertility in section D.3.5.4.b). Others relate to the selection of incidence rate as the starting point of the analysis and the indirect costs included in the estimates. Annex E discusses the impact of other uncertainties affecting the estimation of benefits and their impact on the overall conclusions on the effectiveness of the proposed restriction.

As mentioned, in addition to male infertility, cryptorchidism and hypospadias, exposure to the four phthalates in articles might be associated with a number of other human health and environmental conditions that are considerably more difficult to estimate. In the event of entry into force of the proposed restriction, it can be expected that a considerable other social impacts would be avoided, e.g., sexual development such as delay in puberty, as well as behavioural changes, metabolic disorders, and hormonally-related cancers (see Table D17). Studies that have attempted to estimate some of these suggest that the total damage to the EU society may be as high €6.7 billion annually, i.e., Trasande et al (2015), Legler et al (2015), Hauser et al (2015), Bellanger et al (2015) and Hunt et al (2016) presented in

[194] Norden 2014 recognises that psychological (intangible) costs of infertility exist but does not estimate them.

Appendix D2. Table D17 gives an indication of the benefits to society if some of the human health and environmental impacts due to exposure from the four phthalates can be avoided as a result of the proposed restriction.

In conclusion, it is plausible that the benefits of the restriction are at the minimum comprised of the mid-point estimates of avoided cases of male infertility, cryptorchidism and hypospadias, i.e., in excess of €32.8 million.

Note on health impacts arising from transitioning to alternatives

Although the alternatives of the four phthalates are generally of lower risk, it is recognised that some of the main alternatives can lead to similar human health impacts albeit at much higher level of exposure (e.g., the DNEL-oral for reprotoxic effects for DINP is 0.25 mg/kg bw/day, or more than seven times higher than that for DEHP: 0.035 mg/kg bw/day). As stated in Annex D of the Background document (see Notes to Table D21), the quantitative assessment of human health benefits recognises that there may be some (limited) negative human health impacts as a result of transitioning to alternatives. Therefore, for the purpose of assessing the benefits of the proposed restriction, the analysis assumes that all four phthalates would be replaced with the most hazardous of the alternatives. Thus, the share of the human health cases attributable to the four phthalates is reduced with those that would arise if all four phthalates are replaced with their most hazardous alternative. This is likely underestimating the benefits of the proposed restriction as already presented there are other alternatives with much lower risk which could also be selected by industry to replace the four phthalates (e.g., non-orthophthalates).

D.3.6. Practicability and monitorability

Practicality

For practicality, an RMO must be implementable, manageable and enforceable. All these aspects are discussed below.

Implementability and manageability

A portion of the market has already phased out the four phthalates discussed in this dossier. Especially use of DEHP has been reducing in recent years. Alternative substances are available for all uses, either phthalate or non-phthalate alternatives.

The implementation of the proposed restriction by switching to alternative substances or techniques) is clear and understandable to all actors involved. The use of alternative plasticisers may of course imply some changes in processing and material composition and may require some research and development. However, the R&D cost of substituting e.g. DINP for DEHP is assumed to be low, and the main cost comes from slightly higher use and costs of the alternatives.

For manageability, an RMO should take into account the characteristics of the sectors concerned and be understandable to affected parties. It is expected that small and medium sized enterprises (SMEs) will not be affected less that the general industry in the sectors in question with respect to the technical compliance. However, it is expected that the suppliers

offering the alternatives are large companies, and they serve as general customer advisers when it comes to adjusting polymer formulations and production setup. This will help SMEs in adoption of alternatives. As a consequence, this restriction is implementable and manageable.

Some articles might be used outdoors as well as indoors. To avoid unintended indoor use of such articles they are covered by the proposal even if the actual intended use of a specific article is for outdoor use. This should further support the manageability of the restriction proposal prepared. The additional cost from this is minor and not separately addressed here.

Enforceability

For enforceability the authorities responsible for enforcement need to be able to check the compliance of relevant actors with the RMO in a reasonable manner.

Enforcement activities should cover the import of the four phthalates in articles, and the production of articles in the EU. However, production of articles covered by derogations could continue. Therefore, import of the four phthalates in articles, as well as the production of articles in the EU (and subsequent placing on the market) should be permitted only if the final article is used according to the terms of the derogations proposed.

The derogations proposed should also be enforceable. A definition is given for prolonged contact with skin and a definition is given for outdoor use to facilitate the understanding of those complying with the restriction and enforcement bodies. Workplaces are excluded, using terminology consistent with occupational and health legislation. Exemptions are also given for articles placed on the market before entry into force of the restriction and this will need to be justified to enforcing authorities but wording has been used that is consistent with other recent restrictions.

The placing on the market of articles containing one or more of the four phthalates will be enforced mainly by inspecting the producers, importers or retailers. The enforcement authorities could check documentation from the supply chain confirming that the articles are within the limits given. In addition, in cases where enforcement action will be taken, the need to verify whether the articles contain those phthalates by testing can be foreseen.

Analytical methods to verify the concentration of phthalates in articles exist and are well established. Three of the four phthalates are already restricted in other articles (toys and childcare articles) with the same content limit. Although, there are no harmonised methods for the measurements, the test method(s) for the existing restrictions are available. The same methods can be used for the four phthalates. The content of the four plasticisers when deliberately added will generally be much higher than the detection limit. An applicable test method is CPSC-CH-C1001-09.2 used by U.S. Consumer Product Safety Commission's (CPSC) testing laboratory (LSC) for the analysis of phthalate content in children's toys and child care articles covered by the standard set forth in the Consumer Product Safety Improvement Act Section 108. This contains both a sampling and analytical methodology.[195]

[195] Compendium of analytical methods Recommended by the Forum to check compliance with Reach annex xvii restrictions March 2016 Version 1.0

The analysis is carried out by means of gas chromatography with mass spectrometric detection (GC-MS), which is a well-established and common technique. Standardisation of the methods through CEN would be preferable. There has been work in relation to developing an ISO-standard for “Determination of phthalate plasticiser in toys and children’s products”. The content of phthalates in articles can be measured with detection limits around 10 mg/kg or 0.001 % (m/m). The relative uncertainty of the method is generally estimated to be 10-15%.

For the presence of DEHP, BBP, DBP and DIBP sampling, extraction and laboratory analysis is required. Sampling, sample preparation and analysis increase control costs to be covered by the manufacturers, importers and the authorities. For importers it might be necessary to require testing of products to ensure compliance of imported products of PVC. The frequency of the need to require testing depends of the market situation and the relation between the importer and the supplier. In practice the importer and retailers may often rely on information from suppliers of the articles.

The control activities should ensure that articles containing plasticised materials do not contain the four phthalates above the concentration limit. The types of articles are very inhomogeneous. Some groups of articles like vinyl flooring and cables are homogeneous articles often imported in large batches. Other groups of articles may be heterogeneous, may contain many different materials, and may be imported in smaller quantities. In any case the number of imported articles will be very large and organisation and costs of control can vary greatly and largely depend on the article type. The testing cost is relevant for enforcement authorities and for importers. Again, taking the high cost of testing into consideration, the importer, retailers, etc. would normally rely on supplier information.

The final restriction proposal addresses all comments in Forum advice on the enforceability of the Annex XV proposal for restriction on Four Phthalates (DEHP, BBP, DBP, DIBP) adopted on 12.09.2016 to further improve the enforceability of the restriction.

Monitorability

The results of the implementation of the RMO must be monitored. The monitoring of the restriction will be done through enforcement. No additional monitoring activities are envisaged. In addition to national reporting of enforcement success, notifications of any violation of the restrictions could be reported to the RAPEX system and which in that way would support monitoring of the implementation of the proposed restriction.

The proposed restriction is considered to be feasible as the risk from the phthalates can be addressed by substituting harmful substances with less harmful alternatives.

It is possible to enforce and monitor. As regards to toys three of the phthalates (DEHP, DBP and BBP) are already covered by restrictions in REACH, Annex XVII, entry 51. The proposed restriction will not impact the ban on articles already covered by this entry in REACH Annex XVII.

The restriction proposal prepared does not harm recycling of PVC as recyclates are in large extend used for production of articles not in the scope of the proposal. In the long run recycling may gain from the proposed restriction as the future waste would be less polluted and thus easier to use in the recycling process.

D.3.7. Affordability, cost-effectiveness and benefit-cost comparison

The last stage of the assessment against the criteria for a restriction is an analysis of whether the proposed restriction is a sound regulatory measure. According to the *ECHA Guidance on the preparation of Annex XV dossier for a restriction*, this entails among others:

- An analysis of whether the efforts from the actors to implement and enforce the proposed restriction correspond in amount or degree to the adverse effects that are to be avoided
- An analysis of whether the proposed restriction ensures a good balance between costs and benefits and is cost-effective.

The following sections demonstrate that the proposed restriction is a sound regulatory measure by examining its affordability, cost-effectiveness and the benefit-cost ratio.

D.3.7.1. Market evidence and affordability

One of the arguments that the restriction is justified arises from past trends of substitution. From one of the most used plasticiser in the 1990s, DEHP today is accountable for less than 10% of plasticiser use in Western Europe. Market intelligence shows that there are readily available, comparatively priced, drop-in alternatives with very similar technical performance characteristics to DEHP. In fact, these alternatives today account for the majority in the EU and about half of the worldwide plasticiser use (ECPI 2012). The use of the alternatives is growing even on markets where DEHP is currently dominant.

DBP, DIBP and BBP have even higher rate of replacement that DEHP. The substances have been fully phased out in the EU for uses within the scope of this proposal, as demonstrated by the lack of applications for authorisation. Their use is continuing to decline internationally.

This clearly evident trend in substitution suggests that the restriction would likely not exert disproportionate costs to companies required to comply with it. As demonstrated in section D.3.4 of this Annex and the scenarios presented in the Annex E, the total restriction costs of €16.9 million annually adequately reflect the anticipated costs from the proposed restriction. The majority of these costs (more than 90%) would be incurred to replace the four phthalates in articles to be imported to the EU. Assuming that all costs are passed on the EU consumers, the proposed restriction would lead to an increase in their price per tonne of articles of about 2%. However, this assumption is associated with considerable uncertainty, as it is possible that these costs are shared among the many members of the supply chain, some of which might be outside the EU.

Less than 10% of the total restriction costs, or about €1.5 million, would be borne directly by EU producers of plastic articles. This represents about 0.02% of the value added at factor cost that can be attributed to producers of plastic products using DEHP.[196] Assuming there are approximately 5 600 companies who use DEHP in their production, each company would have to bear additional costs as a result of the restriction of less than €300 annually. Although, it is

[196] EuroStat values for manufacturing of plastic products adjusted for the share of DEHP of total plasticiser use.

important to note that some companies, such as those currently using recyclate PVC to produce articles, would bear the largest share of these costs.

The cost increases due to transition to the alternatives would likely lead to an increase in the EU PVC production costs by about 2.2%. These costs would likely be passed on to EU consumers. The impact on consumers can be seen as minimal if the increase is expressed intuitively: the price increase to consumers will be 22 cents assuming the article is made entirely of PVC, containing 25% DEHP and its original price was €10.

All these statistics suggest that on average the proposed restriction would be affordable to the impacted supply chains. Although, affordability does not imply that a measure has a net benefit for society, this analysis suggests that the proposed restriction likely would not exert disproportionate costs to industry and society as a whole.

D.3.7.2. Cost-effectiveness analysis

The proposed restriction is anticipated to replace on average more than 131 500 tonnes annually of DEHP, DBP, DIBP, and BBP in articles in the scope of this restriction. The burden to EU society is estimated at €16.9 million annually (see section D.3.4). The NPV of these future costs over the next 20 years is less than €230 million in total. This suggests that the cost to society per tonne phthalates replaced is less than €130 (Table D25).

Table D24 Cost effectiveness of the proposed restriction

	Proposed Restriction
Total restriction costs (annual, million euro)	€16.9
Total tonnes substituted due to proposed restriction	131 560
Cost effectiveness (euro/tonne)	€130

Note: All values discounted to 2014 with 4% social time preference rate.

The costs per tonne of the proposed restriction are compared to those of other measures on phthalates: the restriction on phthalates in toys and childcare articles (i.e., restriction entry 51 and 52 in Annex XVII). However, it must be noted that a direct comparison is not possible due to some differences in methodologies and target populations.[197]

The ex-ante compliance costs of the previous restriction on selected phthalates are presented in Table D26. They were estimated on the basis of the costs to industry to transition to an alternative plasticiser: ATBC. Already in 1999-2000, it was assumed that due to the industry's considerable experience with substituting phthalate plasticisers, the compliance costs would comprise primarily of the price difference between ATBC and the phthalate plasticiser (approximately 3.3 times the price of DEHP at that time). It was assumed that reformulation costs would be negligible and there would be no efficiency loss due to comparative loading (i.e., a substitution factor of 1 was applied). Depending on the scope of the then discussed RMOs for restriction on phthalates in toys and childcare articles, its ex-ante cost-effectiveness

[197] Restriction entries 51 and 52 of Annex XVII of REACH is targeted at a vulnerable group: young children, while the proposed restriction targets risks to the general population and vulnerable groups: pregnant mothers and young children.

ranged between €2 270 and €2 630 in 2014 euro (or €1 780 and €2 070 in 1999 values). Therefore, the proposed restriction is 17-20 times more cost-efficient than the existing restriction on phthalates in toys.

Table D25 Estimated compliance costs of a restriction on phthalates in toys (ex-ante)

Compliance cost item	Unit	Phthalates	ATBC	Incremental costs of ban	Cost effectiveness 2014 €/tonne
Products intended to be placed in the mouth of children under 3yrs old					
Plasticiser used	tonne	2.9	2.9		
Cost of plasticiser	euro	2 300	7 500		
Raw material costs increase	euro	-	5 200		
Costs of finished products	euro	136 000	142 000	6 000	2 630
Soft PVC toys for children under 3yrs old					
Plasticiser used	tonne	7 100	7 100		
Cost of plasticiser	euro	5.7 million	19 million		
Raw material costs increase	euro	-	13.3 million		
Costs of finished products	euro	675 million	688 million	13 million	2 325
All soft PVC toys					
Plasticiser used	tonne	28 000	28 000		
Cost of plasticiser	euro	23 million	75 million		
Raw material costs increase	euro	-	52 million		
Costs of finished products	euro	2.7 billion	2.75 billion	50 million	2 270

Notes: The calculation assumes that all tonnages are substituted within one year
Source: EC 2000

In summary, the proposed restriction is a cost-effective measure of addressing the risks of exposure to the four phthalates in articles in the scope of this restriction. This conclusion remains even when uncertainties are taken into account and a sensitivity analysis is performed. The details of this analysis are presented in Annex E.

D.3.7.3. Cost-benefit analysis

Section D.3.4 shows that the total restriction costs of €16.9 million annually adequately illustrate the anticipated costs to EU society as the monetised costs are overstated in order to account for any uncertainties related to the non-quantified negative impacts of the restriction (Table D16).

Considering the many uncertainties in hazard and exposure assessment discussed in section D.3.5, it is plausible that the benefits of the restriction are not only associated with the estimated 2 110 cases of infertility (€9.8 million annually of avoided damage to society, see Table D23) but also with other avoided human health damages such as cryptorchidism and hypospadias (respectively 480 and 540 cases or €23.1 million annually of avoided damage, see

Table D30 and Appendix D2) and even cases associated with behavioural changes, metabolic disorders, and hormonally-related cancers (see Table D17 for an indication of the value of these potential benefits). Therefore, it can be concluded that the benefits of the restriction outweigh its costs.

For the proposed restriction to break even, it is necessary to prevent about 3 655 cases of male infertility annually.[198] This represents about 0.1% of the average annual male births projected in the EU28 or less than 7% of the population at risk due to foetal exposure or about 2% of the population at risk due to infant and early childhood exposure. These cases would be prevented with the entry into force of the proposed restriction from 2020 onward.[199]

Taking into account other health impacts that are associated with exposure to phthalates, for the proposed restriction to break even, it is necessary to prevent about 2 110 cases of male infertility (mid-point estimate for male infertility) and 250 cases of cryptorchidism (or less than 0.01% of the projected male births) or 420 cases of hypospadias (or less than 0.1% of the projected male births). This is less than 5% of the population at risk due to foetal exposure or less than 1.5% of the population at risk due to infancy and early childhood exposure. See Appendix D1 for further details.

Thus, in summary, a modest number of cases show that the benefits of the proposed restriction would exceed its costs. Therefore, it can be concluded that the proposed restriction is an effective measure to address the risks of the four phthalates in articles as its benefits outweigh its costs. This is even more pronounced when a 2% discount rate is applied to both benefits and costs of the proposed restriction: the total benefits of male infertility alone are estimated in excess of €19.6 million annually (see Table D23), while the total restriction costs are less than 19.1 million annually (see Table D16).

This conclusion that the benefits of risk reduction under the proposed restriction outweigh the costs is reinforced when uncertainties are taken into account. Detailed testing of the impacts of uncertainties of this analysis is presented in Annex E. Table D27 attempts to combine the effects on the benefit-cost ration of the proposed restriction of quantified and non-quantified impacts. It clearly shows that the benefits outweigh the costs even when non-monetised impacts are taken into account.

[198] Calculated assuming about €4 600 / case in 2014 values (discounted by 4% social preference rate)

[199] Appendix D1 discusses that for the purpose of showing that the benefits of the proposed restriction exceed its costs, an exposure to the four phthalates in articles as a unique or contributing cause of cryptorchidism and hypospadias would have to be demonstrated in a very limited number of cases (about 380 cases of cryptorchidism and 430 cases of hypospadias) for the benefits of the restriction to outweigh the costs. The estimated cases would represent less than 0.03% of the projected average number of male children borne in the EU28.

Table D26 Summary of uncertainties impacting the benefit-cost (B/C) ratio of the proposed restriction

Impact	Description	Direction B/C ratio is likely affected
Social damage of male infertility	Likely higher than estimated leading to increased value of benefits and improved B/C ratio of the proposed restriction (see section D.3.5.4.b)	+
Social damage of hypospadias & cryptorchidism	Likely higher than estimated (see Appendix D1.c)	+
Other human health impacts to be avoided (general population) – non-monetised	Not estimated. Their estimation will lead to increase in the value of benefits, resulting in an improved B/C ratio of the proposed restriction. An indication of their value is provided in Table D11.	+++
Other human health impacts to be avoided (worker exposure) – non-monetised	Not estimated. Their estimation will lead to increase in the value of benefits, resulting in an improved B/C ratio of the proposed restriction.	+
Environmental benefits, e.g., effects on mammals similar to those of humans	Not estimated. Their estimation will lead to increase in the value of benefits, resulting in an improved B/C ratio of the proposed restriction. (See section D.3.5.2.)	+
4% standard social time preference rate	The standard discount rate of 4% does not take into account that the income elasticity of the value of health is one; therefore, an increase in wealth in the future would lead to an equivalent increase in the value of health. A 2% effective rate, which reflects historical long term income growth, may be more appropriate for discounting human health benefits.	++
Substitution costs	Likely lower than estimated leading to lower overall costs of the proposed restriction, resulting in an improved B/C ratio of the proposed restriction (see section D.3.1.1.4.)	+
Testing costs	Not estimated in main restriction scenario. Their inclusion will lead to higher total restriction costs, eroding the B/C ratio of the proposed restriction (see Annex E)	-
Costs of recycling sector	Unlikely to occur as assumed annually throughout the study period. This will decrease the total restriction costs, resulting in an improved B/C ratio of the proposed restriction (see section D.3.1.3)	+
Enforcement costs	Unlikely to occur as assumed annually throughout the study period. This will decrease the total restriction costs, resulting in an improved B/C ratio of the proposed restriction (see section D.3.2)	+
Costs to compounders (i.e., on producers of PVC in primary forms)	Cost to compounders using DEHP are assumed to be fully passed downstream, i.e., they are included in the estimated substitution costs (see section D.3.2). The potential benefits of the proposed restriction to compounders using alternative plasticisers are not estimated.	+
Costs to substance manufacturers	Not estimated. It is likely that the gains of manufacturers of alternatives are larger than the costs of DEHP manufacturers as some applicants for authorisation have already begun to transition their manufacturing to alternatives of DEHP (see section D.3.2). This would result in higher benefits and an improvement of the B/C ratio of the proposed restriction.	+
Costs to SMEs	Not estimated. It is possible that some SMEs have higher costs to transition to alternative (see section D.3.2).	-
Social impacts	Not estimated. It assumed that all employment losses of DEHP manufacturers are offset by employment gains of alternatives manufacturers (see section D.3.2).	+/-
Impacts of higher quality of the good containing the alternatives	Not estimated but likely positive, leading to lower total restriction costs and improved B/C ratio of the proposed restriction (see section D.3.3).	+
Wider economic impacts	Not estimated, likely negligible (see section D.3.3).	+/-
Distributional costs	Not estimated but it is likely that on balance these represent benefits of the restriction in terms of eliminating the effects of authorisation requirements on EU industry and likely diffused impacts of the restriction along the EU and international supply chains (see section D.3.3).	+

Legend:
Direction in which the B/C ratio is affected: "+" denotes an improvement and "-" a deterioration
Degree of improvement/deterioration of the B/C ratio: "+/-" denotes minor, "++/--": moderate and "+++/---": significant improvement/deterioration.

Appendix D1 Monetisation of additional health benefits

Social damage of malformations

As mentioned previously, *in utero* or after birth exposure to phthalates during the sexual differentiation period of embryos (12-20 weeks of pregnancy) could cause irreversible damages to the male reproductive system. In addition to reduced semen quality and in severe cases infertility, among the most pronounced damages are cryptorchidism and hypospadias. They are often risk factors for each other (including testicular cancer) and together they are hypothesised to comprise the testicular dysgenesis syndrome (TDS) (Norden 2014).

This Appendix presents supplementary information on the monetisation of health benefits in addition to those used in the main assessment, i.e., those associated with avoided social costs of cryptorchidism and hypospadias with the entry into force of the restriction. These effects are established in experimental animals and are considered relevant and adverse to humans. Based on the current evidence in animals, these additional effects might be expected to occur in the population at higher exposure levels than those exposure levels estimated on the basis of biomonitoring. However, mild incidences of cryptorchidism were in fact seen at dose levels corresponding to the DNEL for DEHP (Andrade et al. 2006). This and the fact that these malformations are a part of the TDS, cast doubt on this conclusion and it may therefore be necessary to extend the conclusion that a risk has been identified to the whole spectrum of effects in the rat phthalate syndrome observed in animals. Furthermore, the many uncertainties in hazard and exposure assessment need to be kept in mind, including the uncertainty whether a threshold exists for these substances as endocrine disruptors. For these reasons, it was considered important to provide estimates of the potential social damage of cryptorchidism and hypospadias.

As shown in Table D28 and subsequent sections of this Appendix, for the purpose of demonstrating comparing the benefits to the costs of the proposed restriction, exposure to the four phthalates in articles as a unique or contributing cause of cryptorchidism and hypospadias would have to be demonstrated in a very limited number of cases (less than 1% of the projected average number of male children borne in the EU28 with either cryptorchidism or hypospadias) for the benefits of the restriction to outweigh its costs.

Table D27 Break-even analysis on the basis of estimated number of cases and social damages of male infertility, cryptorchidism, and hypospadias in EU28

Minimum number of cases for Benefits to ≥ Costs	2 110 cases of Male infertility & 250 cases of Cryptorchidism	2 110 cases of Male infertility & 420 cases of Hypospadias	380 cases of Cryptorchidism & 430 cases of Hypospadias
Equivalent to population at risk of foetal exposure (54 000 male children/yr)	<4.5%	<5%	<1.5%
Equivalent to population at risk due to infancy & early childhood exposure (175 000 male children/yr)	<1.5%	<1.5%	<0.5%
Equivalent to percent of projected annual male births (2.6 million/yr)	<0.1%	<0.1%	<0.03%

Notes: Number of cases estimated on the basis of the weighted average social damage per case (mid-point estimate) and total projected annual male equal to 2.6 million presented in Table D23, Table D30 and Table D33. The estimation of the population at risk is as shown in section Risk reduction capacity.

The benefits of avoiding health outcomes such as cryptorchidism and hypospadias are estimated using the same approach as male infertility: the number of cases are derived from current incidence rates and monetised using direct and indirect costs per case gathered by Norden (2014) and intangible costs presented in terms of the willingness to pay (WTP) value of statistical case of a healthy child (minor birth defects, internal or external) estimated by ECHA (2013b).

a) Social damage of cryptorchidism

Cryptorchidism is a birth defect where one or two testes have not descended into the scrotum but remain in the abdomen. Many of the cases spontaneously descend within the first year of birth, but in a small percentage of children with undescended, ascended or retractile testes this defect remains until adulthood. As the position of the testes impacts the development of germ cells, which determine the long term male fertility, cases where there is no spontaneous descend are often corrected surgically with orchidopexy. Currently, the procedure is performed at six to 12 months of age in many centres, as from about 15 months of age the germ cells may be permanently affected. Evidence suggests that an orchidopexy in infancy would improve eventual fertility but the outcome also depends on the position of the testes in relation to the scrotum and whether the cryptorchidism is unilateral or bilateral, the latter being associated with higher risk of infertility.[200]

Cryptorchidism is statistically associated with 1.6-19% of testicular cancer cases (Taran et al 2006). The risk of developing testicular cancer is not eliminated by orchidopexy but it appears to decrease the risk. While testicular cancers are largely treated today, still between 2–6% of the patients in the Nordic countries have not survived five years after the incidence of the disease (Norden 2010).

[200] Taran et all (2006) reports that in adult men with a history of unilateral orchiopexy, fertility is nearly normal, but is significantly reduced following bilateral orchiopexy.

The incidence of cryptorchidism is difficult to determine as the definition of the condition varies in scientific publications. Kortenkamp et al. (2011) states that *depending on country and geographical location, it affects 2 – 4% of boys, but according to recent estimations this can be as high as 9% in some countries.* Incidence reported in the literature in data from hospital-based or central registers (with diagnosis performed from birth to 1 year of age) rates range from less than 1% to 10% (Thonneau et al 2003). Orchidopexy rates have been reported between 2.4% and 3.8% (Jones et al (1998), Campbell et al (1987), Tamhne et al (1990)). HEAL (2014) quotes an incidence rate of 6% and uses the rate of 3% for the purpose of their analysis. Therefore, for the purposes of estimating the aetiological fraction associated with exposure to the four phthalates in articles, we pick as the starting point of the analysis the mid-point between the lower bound of the undescended testes incidence rate and the higher bound of orchidopexy rates, excluding 4% of cases for which studies have shown hereditary causes (Kolon 2015). The analysis of the etiological fraction is detailed in Table D29.

Table D28 Cryptorchidism incidence related to exposure to DEHP, DBP, DIBP, and BBP in articles (EU28)

	Step in analysis	Percentage	Cumulative	Source
a	Incidence of cryptorchidism post 1 year of age	2.4%		Thonneau et al (2003), Jones et al (1998), Campbell et al (1987), Tamhne et al (1990), Kolon (2015)
b	Assumed hereditary	4%	2.30%	Kolon 2015
c	Fraction attributable to exposure to chemicals - mid-point estimate	20%	0.46%	
	Sensitivity - low estimate	2%	0.05%	AFA 2013a, Norden 2014,
	Sensitivity - high estimate	50%	1.15%	HEAL 2014, WHO/UNEP 2012
d	Of c), those that can be attributed to exposure to the 4 phthalates in articles - mid-point	4%	0.018%	*
	Sensitivity - low estimate		0.002%	
	Sensitivity - high estimate		0.046%	
e	Of c) those, who may develop cancer as a result of cryptorchidism induced by exposure to the four phthalates in articles - mid-point estimate	5%	0.001%	
	Sensitivity - low estimate	0%	0.000%	Average of results of five studies
	Sensitivity - high estimate	10%	0.005%	reported by Taran et al (2006)

Notes: * The 4% is a composite percentage equal to:

- the share of cases due to exposure to phthalates: 11%. This assumption is on the basis of a report to the European Commission DG Environment (Kortenkamp et al., 2011) which identified nine groups of chemicals of concern in the EU28 that are associated with adverse impacts on male reproductive health, with phthalates being one of them: polychlorinated biphenyls (PCBS), polychlorinated dibenzodiozins (PCDDS) and polychlorinated dibenzofurans (PCDFS), polybrominated biphenylethers (PBDES), perfluorinated compounds (PFCS), pesticides, heavy metals, bisphenol A, parabens.
- the fraction of cases attributable to exposure to DEHP, DBP, DIBP, and BBP: 90%. It is estimated on the basis of world tonnages used of all phthalates that have the same mode of action, weighed by their relative hazard (on the basis of their oral DNEL). This estimate accounts for the possibility that part of the four phthalate tonnages would be replaced by DINP (about 55%), as DINP has similar

anti-androgenic mode of action (but at 7-8 times time higher doses than DEHP on the basis of DNELs).[201]

- the fraction of cases attributable to exposure to the four phthalates in articles: 40%, estimated on the basis of modelling of exposure to articles and the estimated share of non-food sources of exposure to the four phthalates (25% for DEHP and 75% for DBP, DIBP and BBP) weighed by their tonnages.

The number of cases of cryptorchidism and their associated social damage are presented in Table D30. Approximately, 480 male children are estimated to be borne every year with cryptorchidism because their mothers were exposed to articles containing the four phthalates during their pregnancy. Exposure during infancy may influence spontaneous descend of cryptorchidism cases. The costs of this exposure are associated with direct costs of surgery, indirect of hospital stay, and intangible costs associated with the treatment and potential long term effects that remain untreated (see Table D31). These cases would be prevented with the entry into force of the proposed restriction from 2020 onward. Estimated at €13.9 million, this damage represents a proxy for the annual benefits to society of the proposed restriction.

To show that the benefits of the proposed restriction exceed its costs, it is sufficient to prevent more than 460 cases of cryptorchidism or less than 0.02% of the average number of male children projected to be borne in the future in the EU28. As shown in Table D30, this is significantly less than the estimated number of cases of cryptorchidism in the EU28 that could be associated with exposure to the four phthalates in articles as a unique or contributing cause.

[201] It is important to remind the reader that for the purpose of estimating the costs of the restriction the alternatives were selected on the basis of convenience (due to less confidentiality issues related to the critical data required for the analysis). As discussed in the sections D.2.3.3 and D.3.1.1.3, there are other alternatives with similar (or better) technical and economic feasibility which also have more benign risk profile than DINP.

Table D29 Summary of estimated social damage related to cryptorchidism due to exposure to DEHP, DBP, DIBP and BBP in articles (EU28)

	Low estimate	**Mid-point estimate**	**High estimate**
Average annual male births (EuroStat, 2020-2050)	2 600 000	2 600 000	2 600 000
Fraction of cases of cryptorchidism attributable to DEHP, DBP, DIBP, and BBP in articles	0.002%	0.02%	0.05%
Number of cases of cryptorchidism due to DEHP, DBP, DIBP, and BBP in articles	50	480	1 200
Direct costs*	210 900	2 109 000	5 272 000
Indirect costs**	56 600	566 000	1 415 000
Intangible***	1 246 000	14 940 000	43 550 000
Total annual social damage of cryptorchidism	1 513 000	17 615 000	50 236 000
weighted average per case (€)	31 600	36 800	41 900
Total annual costs (discounted to 2014 by 4%)	1 196 000	13 921 000	39 702 000
weighted average per case (discounted by 4%)	25 000	29 000	33 100
Total annual costs (discounted to 2014 by 2%)	1 344 000	15 641 000	44 609 000
weighted average per case (discounted by 2%)	28 000	32 600	37 200

Note: 2014 values, average, representative year analysis
* Direct costs in this case include costs per treatment with orchidopexy (see Table D28). Cancer treatment costs are not assessed.
** Indirect costs in this case include productivity loss due to time spent for treatment (see Table D28). Indirect costs associated with cancer treatment are not assessed.
*** Intangible costs presented in terms of the WTP to avoid having a child with external birth defects as a proxy for 95% of the cases of cryptorchidism and WTP to avoid having a child with internal birth defects as a proxy for the remaining cases which are assumed to lead to testicular cancer. WTP values estimated by ECHA (2014b).

Table D31 presents the assumptions and sources used in the estimation of the social benefits from the restriction associated with avoided cryptorchidism cases. We use the willingness to pay to avoid having a child with external birth defects as a proxy for 95% of the cases of cryptorchidism. It is assumed that this value captures all associated suffering and minor long term effects that a child borne with mal-descended testes might have. For 5% of cases, where we assumed that the cryptorchidism may lead to more serious health outcomes such as testicular cancer, the willingness to pay (WTP) to avoid having a child with internal birth defects is used as a proxy. This value was assumed to imply the parents' willingness to pay to avoid more serious congenital anomalies, that may have long lasting health consequences, and in very rare cases, premature death (ECHA 2014b).

Table D30 Assumptions for estimating the number of cases and total social damage related to cryptorchidism due to exposure to the four phthalates in articles (EU28)

Assumptions	**Unit**	**Value**	**Source**
Treatment costs per case	2014 euro	4 400	Norden 2014
Time spent on treatment	hours	48	Norden 2014
Average hourly labour cost in the EU28 (costs for wages and salaries plus non-wage costs such as employers' social contributions)	2014 euro/hour	24.60	EuroStat, 2014 data
Intangible costs per case (WTP - external birth defects)	2014 euro	26 000	ECHA 2014b
Intangible costs per case (WTP - internal birth defects)	2014 euro	129 500	ECHA 2014b

b) Social damage of hypospadias

Hypospadias is a birth defect of the urethra in males manifested by an abnormally placed urinary opening (Norden 2014). The degree of this congenital defect is defined in terms of how close to the junction between the penis and the scrotum is the urinary opening. Surgery may succeed at correcting the condition. However, regardless of the success of the correction, it may in the future lead to the necessity for additional treatment for urethrocutaneous fistula (often surgery) and for uretra stricture, in respectively 15% and 10% of cases (Norden 2014).

Incidence of hypospadias is hard to estimate due to limited number of studies and the difference in defining the condition in different countries. It is estimated to be of hereditary nature in 4% to 25% of cases (Table D32). It is often associated with other congenital abnormalities, such as cryptorchidism (although this is not the most common); therefore it is uncertain to what extent a causal effect on infertility (or other long-term effects) could be uniquely associated with hypospadias.

Table D31 Estimation of hypospadias incidence related to exposure to DEHP, DBP, DIBP, and BBP in articles (EU28)

Step in analysis		Percentage	Cumulative	Source
a	Incidence in EU population	3%		*
b	Assumed hereditary	15%	2.57%	**
c	Fraction attributable to exposure to chemicals - mid-point estimate Sensitivity - low estimate Sensitivity - high estimate	20% 2% 50%	0.51% 0.05% 1.28%	AFA 2013a, Norden 2014, HEAL 2014, WHO/UNEP 2012
d	Of c), those that can be attributed to exposure to the four phthalates in articles - mid-point estimate Sensitivity - low estimate Sensitivity - high estimate	4%	0.021% 0.002% 0.051%	See Note in Table D29
e	Those who may develop additional health conditions, e.g., urethrocutaneous fistula, uretra stricture	25%		Norden 2014

Notes:
*On the basis of Norden 2014, HEAL 2014 and taking into account significant underreporting of hypospadias cases and a trend towards increase in incidence (Toppari 2001).
** Fredell et. all (2002), Stoll et al. (1990), Monteleone et al. (1981), Bauer et al. (1981), Czeizel et al. (1979), Bauer et al. (1979), Sweet et al. (1974) reported by Thorup et al. (2010)

The number of cases of hypospadias and their associated social damage are presented in Table D33. Approximately, 540 male children are estimated to be borne every year with hypospadias because their mothers were exposed to articles containing the four phthalates during their pregnancy. The social damage of this exposure are associated with direct costs of surgery, indirect of hospital stay and intangible costs associated with the treatment and potential long term effects that remain untreated (See Table D31). These cases would be prevented with the entry into force of the proposed restriction from 2020 onward. The estimated €9.1 million annually represent a proxy for the annual benefits to society of the proposed restriction.

Table D32 Summary of estimated social damage related to hypospadias due to exposure to DEHP, DBP, DIBP and BBP in articles (EU28)

	Low estimate	**Mid-point estimate**	**High estimate**
Average annual male births (EuroStat, 2020-2050)	2 600 000	2 600 000	2 600 000
Fraction of cases of hypospadias attributable to DEHP, DBP, DIBP, and BBP in articles	0.002%	0.021%	0.051%
Number of cases of hypospadias due to DEHP, DBP, DIBP, and BBP in articles	50	540	1 340
Direct costs*	551 100	5 511 000	13 778 000
Indirect costs**	82 100	821 000	2 053 000
Intangible (WTP)***	522 400	5 223 700	13 059 400
Total annual social damage of hypospadias	1 156 000	11 556 000	28 891 000
weighted average per cost case (€)	21 600	21 600	21 600
Total annual costs (discounted to 2014 by 4%)	913 000	9 133 000	22 833 000
weighted average per case (discounted by 4%)	17 100	17 100	17 100
Total annual costs (discounted to 2014 by 2%)	1 026 000	10 262 000	25 655 000
weighted average per case (discounted by 2%)	19 200	19 200	19 200

Note: 2014 values, average, representative year analysis

* Direct costs in this case include costs per treatment – surgery (see Table D28).

** Indirect costs in this case include productivity loss due to time spent for treatment (see Table D28).

*** Intangible costs presented in terms of the WTP to avoid having a child with minor birth defects as a proxy for 75% of the cases of hypospadias. For the remaining, where it is assumed that the hypospadias may lead to additional health complications (see Table D32), the WTP to avoid having a child with external birth defects is used as a proxy. WTP values estimated by ECHA (2014b).

Table D34 presents the assumptions and sources used in the estimation of the social benefits from the restriction associated with avoided hypospadias cases. We use the willingness to pay to avoid having a child with minor birth defects as a proxy for 75% of the cases of hypospadias. It is assumed that this value captures all associated suffering and minor long term effects that a child borne with hypospadias might have. For 25% of cases, where it is assumed that the hypospadias may lead to additional health complications (see Table D32), the willingness to pay to avoid having a child with external birth defects is used as a proxy. This value was assumed to imply the parents' willingness to pay to avoid more serious health effects associated with further treatment or surgery (ECHA 2014b).

Table D33 Assumptions for estimating the number of cases and total social damage related to hypospadias

Assumptions	**Unit**	**Value**	**Source**
Treatment costs per case	2014 euro	10 300	Norden 2014
Time spent on treatment	hours	62	Norden 2014
Average hourly labour cost in the EU28 (costs for wages and salaries plus non-wage costs such as employers' social contributions)	2014 euro/hour	24.60	EuroStat, 2014 data
Intangible cost per case (WTP - minor birth defects)	2014 euro	4 350	ECHA 2014b
Intangible costs per case (WTP - external birth defects)	2014 euro	26 000	ECHA 2014b

c) Summary of social damage of cryptorchidism and hypospadias and uncertainties

In summary, the many uncertainties in hazard and exposure assessment discussed in section Human health and environmental impacts of Annex D indicate that the benefits of the restriction are not only associated with the estimated 2 110 cases of infertility (€9.8 million annually of avoided damage to society, see Table D23) but also with other avoided human health damages such as cryptorchidism and hypospadias (respectively 480 and 540 cases or €23.1 million annually of avoided damage, see Table D30 and Table D33) and even cases associated with behavioural changes, metabolic disorders, and hormonally-related cancers (see Table D17 for an indication of the value of these potential benefits. To show that the benefits of the proposed restriction exceed its costs, it is sufficient to prevent approximately 380 and 430 of the estimated cryptorchidism and hypospadias presented. These would represent about 0.03% of the average number of male children projected to be borne in the EU28 in the future and less than 1.5% of the population at risk of *in utero* exposure.

In addition to the uncertainties related to hazard and exposure which underpin the estimates of cryptorchidism and hypospadias in Table D30 and Table D33, a number of factors related to the approach of quantifying and monetising of the impacts may be understating the total social benefits because:

- Several studies place the incidence rate of cryptorchidism much higher, e.g., Skakkebaek et al. (2016).
- It is uncertain whether the intangible costs fully capture all costs associated with the pain and suffering of all impacted by the medical condition. Impacts on the male reproductive system are complex and lead to a number of health conditions which are closely associated (or lead) to e.g., cryptorchidism. These could entail years of mental anguish and financial cost for diagnosis and treatment prior to the date of desired fatherhood. These are not captured in the presented estimates.
- The direct and indirect costs do not fully capture all costs of treatment that are associated with hypospadias and cryptorchidism (e.g., for those cases where testicular cancer is a secondary diagnosis).
- It is uncertain to what extend indirect costs such as overhead costs of the public health system are taken into account in the collected cost of treatment data.

At the same time, a considerable uncertainty is associated with the estimated aetiological fraction of malformation cases due to exposure from the four phthalates. The analysis has reduced this uncertainty by making a number of informed assumptions. However, what percentage of the total cases of cryptorchidism and hypospadias in society could be associated solely with chemicals remains unknown. The analysis uses estimates that have been applied elsewhere, e.g., Norden (2014), AFA (2013a), HEAL (2013). However, these studies also recognise the high uncertainty associated with the share of incidence due to exposure to chemicals. Therefore, the scenarios presented here and in Annex E, may not show accurately the upper and lower bound of the value of social damage due to the four phthalates in articles.

d) Social damage of immunological effects

Animal studies, epidemiological data and exposure levels of the four phthalates suggest that immunological effects such as allergy, asthma and eczema are adverse and relevant effects for humans following exposure to the four phthalates. Data from over 30 animal studies show that phthalates have adjuvant properties leading to development or enhancement of immunological

disorders (allergy, asthma and eczema), possibly at levels lower than reproductive toxicity. Epidemiological studies show that allergy, asthma, and eczema have been associated with exposure to phthalates, e.g. in children living in homes with high concentration of phthalates in dust (Braun et al. 2013, Bornehag et al. 2004, Hsu et al. 2012, Kolarik et al. 2008).

The overall strength of association between exposure to the four phthalates and immunological effects in humans is evaluated as moderate/strong on the basis of fairly strong evidence from animal studies, some epidemiological evidence and strong evidence based on exposure considerations.

WHO/UNEP (2012) gives an overview of the available evidence of phthalate-related exposure and the incidence of asthma: An epidemiological study examining the association between PVC products in the home and the incidence of airway symptoms determined that the presence of PVC materials increased the risk for bronchial obstruction in young children (Oie et al., 1999; Chalubinski & Kowalski, 2006). Additional studies from Sweden, Russia and Finland supported this finding and showed that exposure to PVC flooring and/or PVC wall covering material was correlated with airway symptoms in children (Jaakkola, Verkasalo & Jaakkola, 2000; Bornehag et al., 2005). Moreover, two case-control prevalence studies from Sweden and Bulgaria describe an association between the concentration of DEHP in indoor dust and asthma and wheezing in children (Bornehag et al., 2004; Kolarik et al., 2008). Taken together, these studies are in accord with published animal studies (Bornehag & Nanberg, 2010). WHO/UNEP (2012) also reports that the prevalence of paediatric asthma has more than doubled over the past 20 years, and is now the leading cause of hospitalisations and school absenteeism (Landrigan & Goldman, 2011).

The social damage of immunological effects caused by exposure to the four phthalates has been partially quantified here, i.e., the social damage of asthma was quantitatively estimated, but not eczema and other allergies.

Social damage of asthma

Asthma is a chronic lung disease that inflames and narrows the airways. Asthma causes recurring periods of wheezing, chest tightness, shortness of breath, and coughing. The coughing often occurs at night or early in the morning. Asthma affects people of all ages, but it most often starts during childhood. Among children, more boys have asthma than girls, but among adults, more women have the disease than men. It's not clear whether or how sex and sex hormones play a role in causing asthma. (NIH 2016)

It is estimated that 3.3% of the total population of the Netherlands (2007 data) has asthma although a higher proportion of children under 10 are affected: 3.9% (Suijkerbuijk et al, 2012). Van den Akker-van Marle et al. (2005) reports total EU prevalence of 7.2%.

Many people who have asthma also have other health conditions (such as allergies) (NIH 2016). Rijk at al. (2016) report that respiratory conditions for which a pathophysiologic link to asthma is believed to exist are allergic rhinitis, sinusitis and otitis media. More than 25% of children have one or more of these comorbidities vs less than 10% of nonasthmatic children (Grupp-Phelan et al. 2001).

Asthma is treated with two types of medicines: long-term control and quick-relief medicines. Long-term control medicines help reduce airway inflammation and prevent asthma symptoms.

Quick-relief, or "rescue," medicines relieve asthma symptoms that may flare up. Some medications are possibly associated with side effects, e.g., inhaled corticosteroids with slow growth of children of all ages. (NIH 2016)

To quantify the benefits of avoiding social damage as a result of the proposed restriction, the number of asthma cases associated with exposure to the four phthalates in articles in scope is derived following the same approach as infertility and malformations. I.e., it is derived on the basis of assumptions of the aetiological fraction of cases that can be attributed to exposure to chemicals, EDCs in this case, to exposure to all phthalates, and to the four phthalates in articles in scope (taking into account the substitution with phthalates that may lead to similar health effects) (see notes in Table D29 and Table D21 for the explanation of the assumptions). The main notable difference between the assumptions applied here compared with infertility is the lower rate assumed for the fraction of cases attributable to exposure to chemicals (i.e., chemicals with similar endocrine disrupting properties as the four phthalates): 2.5% for mid-point estimates. While the rate used for infertility and malformations was derived by a panel of experts which weighed the strength of evidence of the association between exposure to chemicals (EDCs) and the health effects comprising testicular dysgenesis syndrome (TDS), the asthma analysis uses the rate applied by Rijk et al (2016), which is the latest published study examining asthma impacts. The latter rate, which is almost 10 times lower, is selected by Rijk et al (2016) on the basis of a review of studies examining the effects of EDCs (see step c in Table D32).

Table D34 Estimation of asthma incidence related to exposure to DEHP, DBP, DIBP, and BBP in articles (EU28)

Step in analysis		**Percentage**	**Cumulative**
a	Prevalence rate	3.3%	
b	Of a), those that can be associated with exposure to EDCs only – mid-point estimate	2.5%	0.08%
	Sensitivity - low estimate	1.0%	0.03%
	Sensitivity - high estimate	10.0%	0.33%
c	Of b), those that can be attributed to exposure to the four phthalates in articles - mid-point estimate	4.0%	0.003%
	Sensitivity - low estimate		0.001%
	Sensitivity - high estimate		0.013%

Source: Rijk et al (2016), Suijkerbuijk et al (2012)

Direct and indirect costs to EU society due to asthma are estimated on the basis of Suijkerbuijk et al (2012), similar to Rijk et al (2016). The analysis presented below also estimates intangible or psychological costs[202] related to the pain and inconvenience of asthma symptoms on quality of life. These are estimated on the basis of willingness to pay values presented in ECHA 2014f (approximately €50/asthma episode in 2012 values) and the number of health care consultations presented by Suijkerbuijk et al (2012).

As shown in Table D33, the total costs to society due to exposure to the four phthalates in articles leading to asthma effects are approximately €45 million annually. As with the quantification of the other health effects, a considerable uncertainty is associated with the estimated aetiological fraction of asthma cases due to exposure from the four phthalates. The

[202] Intangible costs are not estimated by Rijk et al (2016).

analysis has reduced this uncertainty by making a number of informed assumptions and by assuming a conservative percentage of case attributable to the four phthalates in articles in scope. However, what percentage of the total cases of asthma in society could be associated solely with chemicals remains unknown.

Table D35 Summary of estimated social damage related to asthma due to exposure to DEHP, DBP, DIBP and BBP in articles (EU28)

	Low estimate	**Mid-point estimate**	**High estimate**
Average annual population (EuroStat, 2020-2040)	507 162 571	507 162 571	507 162 571
Fraction of cases of asthma attributable to 4 phthalates in articles	0.001%	0.003%	0.013%
Number of cases of asthma due to 4 phthalates in articles	6 720	16 800	67 210
Direct & Indirect costs*	7 618 129	19 045 323	76 192 629
Intangible (psychological costs)**	15 186 950	37 967 374	151 892 097
Total annual costs (2020 onward)	22 805 079	57 012 697	228 084 725
weighted average / case	3 390	3 390	3 390
Total annual costs (4% discount)	18 023 000	45 058 000	180 259 000
weighted average / case (discounted)	2 680	2 680	2 680
Total annual costs (2% discount)	20 250 000	50 626 000	202 533 000
weighted average / case (discounted)	3 010	3 010	3 010

Source: Rijk et al (2016), Suijkerbuijk et al (2012)
Notes:
* Direct & indirect costs include: direct medical costs (the costs of consultations with specialists, nurses, general practitioners, physiotherapy, hospitalisation, medication, influenza vaccinations) and indirect non-medical costs such as occupational disability and sick leave.
** Intangible costs are estimated on the basis of the willingness to pay values presented in ECHA 2014f (approximately €50/asthma episode in 2012 values) and the number of health care consultations presented by Suijkerbuijk et al (2012).

Aside from the uncertainties related to the estimation of the aetiological fraction of cases attributable to exposure to the four phthalates in articles, a number of factors related to the approach of quantifying and monetising the impacts point to underestimation rather than overestimation of the total damage to EU society due to asthma:

- The rate of asthma applied in the analysis of 3.3% for the whole population is lower than reported in other studies: e.g., total EU prevalence of 7.2% Van den Akker-van Marle et al, 2005
- Suijkerbuijk et al (2012) point out that the methodology for gathering cost data for asthma may have led to an underestimation.
- The use of cost and prevalence rates for the Netherlands may not be representative for the EU-total. However, Suijkerbuijk et al (2012) is a very detailed study for asthma, Chronic Obstructive Pulmonary Disease (COPD), and respiratory allergy and similarly detailed data is not available for the EU28.
- The analysis assumes that asthma sufferers visit medical providers each time they experience an asthma episode. In reality, some asthma episodes are resolved with previously prescribed medication and medical consultation may not be necessary. This points to an underestimation of the number of asthma episodes per year and thus, the intangible costs of asthma. However, the results of this analysis are similar to the annual willingness to pay values derived by other studies reported in ECHA 2014f.

- Indirect costs include only absences and disability if they affect paid employment. Volunteer work, school absenteeism, or leave to care for a sick child are not included in Suijkerbuijk et al (2012).
- The estimates do not take into account premature mortality due to asthma.
- The cost data in Suijkerbuijk et al (2012) is from 2007. Although the results presented in Table D36 are adjusted to 2014 price levels using CPI, it is not clear whether the value of medical costs have increased at a higher rate than the basket of goods underlying the CPI.

Appendix D2 Review of recent valuation studies of the impacts of the four phthalates

Norden (2014) presents the number of cases and social costs associated with exposure to endocrine disrupting chemicals for the Nordic countries (Denmark, Norway, Sweden, Iceland and Finland). To illustrate these costs, the study estimates the number of cases of testicular cancer, cryptorchidism, male infertility, and hypospadias. The study extrapolates the number of cases and costs to EU28 but warns of problems of using Nordic incident rates for the EU as a whole. For example, the study uses the incident rates for Sweden to estimate the number of cases of cryptorchidism (1%), while several other studies show that this is at the lower end of incidence. It also does not take into account the significantly larger number of corrective surgeries which suggests that less severe cases of mal-descended testes also require treatment to minimise impacts on long term fertility. Other concerns relate to the fact that the study does not always take into account intangible costs associated with some health conditions (e.g., male infertility) and it likely underestimates the indirect costs of treatment.

Norden 2014 estimates the total annual cost of exposure to endocrine disrupting chemicals to be close to €600 million (Table D37). As phthalates are one of several endocrine disrupting chemical groups, we could derive, on the basis of Norden (2014) estimates, the costs to society due to exposure to the four phthalates in articles in the scope of the proposed restriction. If the fraction of the exposure that can be associated with the four phthalates in articles is applied (equal to 4%, see notes in Table D29 and Table D21 for the explanation of the assumptions behind this estimate), the costs to society that would be avoided in the event of entry into the force of the proposed restriction are approximately €23.7 million annually.

Table D36 Estimation of the costs to EU society due to exposure to DEHP, DBP, DIBP and BBP in articles on the basis of Norden (2014) (2014 euro)

Health outcome	Annual cost of EDCs	Annual costs of four phthalates
Male infertility	72 328 000	2 893 000
Cryptorchidism	181 485 000	7 259 000
Testicular cancer	249 213 000	9 969 000
Hypospadias	88 917 000	3 557 000
Total	**592 000 000**	**23 678 000**

Notes: The Annual costs of EDCs (column two) estimated by Norden (2014) were adjusted to arrive to Annual costs of four phthalates in column three with the share of the cases that can be attributed to exposure to the four phthalates in articles. See notes in Table D21 for details on the estimation of the share of the four phthalates in articles.

Trasande et al (2015), Legler et al (2015), Hauser et al (2015), Bellanger et al (2015) and Hunt et al (2016) quantified a range of health and economic costs attributed to endocrine disrupting chemicals in the EU. An expert panel identified low epidemiological and strong toxicological strength of evidence for male infertility (adult male infertility, as opposed to infertility due to in utero or infancy exposure presented by Norden and in table D20 of this report) attributable to DBP and BBP exposure, with a 40–69% probability of causing 618 000 additional assisted reproductive technology procedures, costing €4.71 billion annually. Similarly, 24 800 associated deaths annually and lost economic productivity of €7.96 billion with 40-69% probability were estimated as a result of lower testosterone concentrations in 55- to 64-year-old men due to phthalate exposure. (Hauser et al 2015) The group of researchers

identified a 40% to 69% probability of phthalate exposure causing 53 900 cases of obesity in older women with €15.6 billion in associated costs as well as 20 500 new-onset cases of diabetes in older women with €607 million in associated costs annually. (Legler et al 2015) 145 000 cases of endometriosis was found to be associated with 20-39% probability of phthalate exposure leading to €1.25 billion annually of economic and health care costs (Hunt et al 2016).[203]

The study assigned causation of Autism spectrum disorder and Attention-Deficit Hyperactivity Disorder (ADHD) by multiple EDCs (phthalates being one of them). Autism was assigned a 20–39% probability, leading to 316 (sensitivity analysis, 126–631) attributable cases at a cost of €199 million annually (sensitivity analysis, €79.7 million to €399 million), while ADHD: with 20–69% probability to be associated with 19 300 to 31 200 cases at a cost of €1.21 billion to €2.86 billion annually (Bellanger et al 2015).[204]

On the basis of Trasande et al. (2015), Legler et al. (2015), Hauser et al. (2015), and Bellanger et al. (2015), it can be estimated that the costs to EU society due to exposure of the four phthalates in articles is approximately €6.7 billion annually. Table D38 summarises the results of these studies.

Table D37 Estimation of the costs to EU society due to exposure to DEHP, DBP, DIBP and BBP in articles on the basis of Trasande et al. (2015), Legler et al. (2015), Hauser et al. (2015), Bellanger et al. (2015) and Hunt et al. (2016)

Human health outcome (million euro)	**Chemicals**		**Base estimates (2010 €): total impact**	**Probability of causation**	**Annual costs (2014 values): attributable to phthalates in articles**
	All EDC	**Phthalates only**			
Adult male infertility		x	4 714	40–69%	1 031
Low testosterone leading to increased mortality		x	7 959	40–69%	1 833
Adult obesity		x	15 611	40–69%	3 414
Adult diabetes		x	607	40–69%	133
ADHD	x		1 743	20–69%	35
Autism	x		199	20–39%	3
Endometriosis		x	1 250	20–39%	156
Total annual costs to EU society due to exposure to the four phthalates in articles (million euro)					6 689

Notes: Annual costs are estimates of Trasande et al. (2015), Legler et al. (2015), Hauser et al. (2015), Bellanger et al. (2015) and Hunt et al. (2016)adjusted with i) the mid-point probability of causation, ii) the share of the cases that can be attributed to exposure to the four phthalates in articles, and iii) 2014 values. See notes in Table D21 for details on the estimation of the share of the four phthalates in articles.

Following the submission of the restriction dossier, Rijk et al. (2016) published a review of recent studies estimating costs that may be associated with exposure to EDCs and estimated EDC attributable socio-economic costs for neural tube defects, endometriosis and asthma for the Netherlands. The study extrapolated from these costs the total potentially EDC-attributable costs for EU28. Table D39 presents these estimates. Similarly to above, as phthalates are one

[203] All values as of 2010.

[204] All values as of 2010.

of several endocrine disrupting chemical groups, we could derive, on the basis of Rijk et al (2016), the damage to society due to exposure to the four phthalates in articles in the scope of the proposed restriction. These costs, i.e., approximately €103 million annually, consist of direct and indirect healthcare and non-healthcare costs. Intangible costs associated with these human health outcomes are not estimated.

Table D38 Estimation of the costs to EU society due to exposure to DEHP, DBP, DIBP and BBP in articles on the basis of Rijk et al. (2016)

Human health outcome (annual, million euro)	**Chemicals contributing to health outcome**	**EDC attributable costs (reported yr)**	**Four phthalate attributable costs (2014 values)**
Endometriosis	phthalates; dioxins; OCP: β-HCH γ-HCH Mirex; PCBs; DES	1 940 (2009)	83.9
Neutral tube defects	pesticides, POPs, PCBs, dioxins, BFRs, perchlorate, BPA, PFCs, phthalates, UV	19.1 (2005)	0.9
Asthma	phthalates, triclosan, parabens, PCBs, dioxins, BPA	432 (2007)	18.5
Total costs to EU28 society:			**103.2**

Notes: Annual costs are estimates of Rijk et al. (2016) adjusted by: i) the share of the cases that can be attributed to exposure to the four phthalates in articles, and iii) 2014 values. See notes in Table D21 for details on the estimation of the share of the four phthalates in articles.

Using similar methodology to Rijk et al. 2016 but also including the intangible costs to society, the total social damage of asthma only, associated with exposure to the four phthalates in articles in scope is about €65.7 million annually to EU society. (See Appendix D1)

Appendix D3 Available information on risks of alternatives

There is no available information on risk, so information on hazard was taken from registration dossiers or from the classification and labelling inventory[205]. Where information is taken from the inventory it should be understood this is from manufacturers, importers and downstream users and is not a harmonised classification agreed by authorities, nor has the Dossier Submitter verified any of the underlying justifications for the information.

Table D39 Hazard classification and labelling and conclusions of hazard/risk profile of the alternatives

Substance Name	Regulatory and CLP status	Harmonised classification and notifications to the classification and labelling inventory					
		Harmonised classification		Notifications to the classification and labelling inventory			
		Hazard Class and Category Code(s)[1]	Hazard Statement Code(s) (labelling)	Hazard Class and Category Code(s)[1]	Hazard Statement Code(s) (labelling)	Number of notifiers	Additional comments
ASE Sulfonic acids, C10-21- alkane, Ph esters (EC 293-728-5)	REACH Registered (10 000 - 100 000 tonnes per annum) Dossier evaluation carried out and information on effects to the terrestrial environment requested by 1 June 2016. Not fulfilling PBT or vPvB criteria. Previously the subject of a temporary TDI by EFSA.	Not classified	-	Not classified	-	76	
ATBC	REACH Registered (2 registrations:	Not classified	-	-	-	1216	

[205] The information in this table is valid as of 3 March 2016.

Tributyl o-acetylcitrate (201-067-0)	joint registration at 10000-100000 tonnes and single registration at 100-1000 tonnes) RMOA being developed by France on the basis of Endocrine Disruption effects to the environment.					58	Data lacking
				Flam. Gas 1	H220 (Extremely flammable gas)	19	12 companies notified the substance with a classification of Muta. 1B (H340) and Carc. 1B (H350), accompanied by Note K. Note K states that the classification as a carcinogen or mutagen need not apply if it can be shown that the substance contains less than 0.1% w/w 1,3-butadiene (EINECS No 203-450-8). If the substance is not classified as a carcinogen or mutagen, at least the precautionary statements (P102-) P210-P403 or the S- phrases (2-) 9-16 should apply. Therefore, it is reasonable to assume that under certain circumstances ATBC may be accompanied by impurities (1,3- butadiene), which could lead to a Carc. 1B and Muta. 1B classification.
				Muta. 1B	H340 (May cause genetic defects)		
				Carc. 1B	H350 (May cause cancer)		
				Skin Irrit. 2	H315 (Causes skin irritation)		
				Eye Irrit. 2	H319 (Causes serious eye irritation)		

				Aquatic Chronic 2	H412 (Harmful to aquatic life with long lasting effects)		Other companies notified for skin and eye irritation (3 and 6 respectively) and there was a single company who notified for chronic aquatic toxicity.
COMGHA **EC:** 616-005-1 **Component A (CAS:** 330198-91-9**)** **Component B (CAS:** 33599-07-4**)**	Not REACH registered	-	-	-	-	-	No notifications were available for COMGHA, Component A, or Component B.
DEHA **Bis(2-ethylhexyl) adipate** **EC:** 203-090-1	REACH registered; 10 000 - 100 000 tonnes per annum TDI established by EFSA; Entered in CoRAP list update (2013-2015) because of CMR concerns	-	-	Not classified	-	638	The lead registrant and a further 767 notifiers did not classify the substance.
				-	-	28	Data lacking.
				Skin Irrit. 2	H315 (Causes skin irritation)	27	A further 27 notifiers classified the substance in 9 distinct notifications. Most of the companies who notified a hazard for DEHA have mentioned irritating properties (skin irrit.2, eye irrit.2) and aquatic toxicity (mainly Acute 1 but also Chronic 1).
				Eye Irrit. 2	H319 (Causes serious eye irritation)		
				Aquatic Acute 1	H400 (Very toxic to aquatic life)		
				Aquatic Chronic 1	H410 (Very toxic to aquatic life with long lasting effects)		

				Acute Tox. 4	H302 (Harmful if swallowed)		There were single cases that mentioned Acute toxicity to humans, Carcinogenicity 2 and Reproductive toxicity 2, but the reason for these classifications are not clear and may be
				Acute Tox. 2	H332 (Harmful if inhaled)		Companies had in their product.
				Carc.2	H351 (Suspected of causing cancer)		
				Repr. 2	H361 (Suspected of damaging fertility or the unborn child)		
				Aquatic Chronic 2	H411 (Toxic to aquatic life with long lasting effects)		
					H319		1 notifier didn't fill the rest of the information.

DEHS Bis(2-ethylhexyl) sebacate **EC:** 204-558-8	REACH registered: joint registration (1 000 - 10 000 tonnes per annum) and individual registration (100-1000).	-	-	No classification	-	306	The lead registrant and a further 261 notifiers did not classify the substance.
				Acute Tox 4	H302 (Harmful if swallowed)	3	There were 3 companies that notified the substance with a single hazard, i.e. Acute Toxic 4 (oral). No other notifications were
DEHT Bis(2-ethylhexyl) terephthalate **EC:** 229-176-9	Not REACH registered; Some differences in CLP classifications notified by various parties (particularly regarding reprotoxic classification) RMOA france	-	-	No classification	-	207 (joint)	The lead registrant and a further 164 notifiers did not classify the substance.
				-	-	34	Data lacking.
		-	-	Aquatic Chronic 4	H413 (May cause long lasting harmful effects to aquatic life)	1	
		-	-	Repr. 2	H361 (Suspected of damaging fertility or the unborn child)		

DPHP Bis(2-propylheptyl) phthalate **EC:** 258-469-4	REACH registered; 100 000 - 1 000 000 tonnes per annum Not classified under CLP CoRAP Potential endocrine disruptor			No classification	-	110 (joint)	
DIDP 1[2206] 1,2-Benzenedicarboxylic acid, di-C9-11-branched alkyl esters, C10-rich **EC:** 271-091-4	REACH Registered100 000 - 1 000 000 tonnes per annum; TDI established by EFSA; Only minor differences in CLP classifications notified by various parties			No classification	-	355	substance.
				Skin Irrit. 2	H315 (Causes skin irritation)	25	A further 32 companies classified the substance in 2 distinct notifications. 25 of them classified it as skin and eye irritant 2 and 7 as just eye irritant 2.
				Eye Irrit. 2	H319 (Causes serious eye irritation)		
				Eye Irrit. 2	H319 (Causes serious eye irritation)	7	
DIDP 2[3] Di-''isodecyl'' phthalate **EC:** 247-977-1	-	-	-	No classification	-	95	

[206] The various hazards that were notified are collectively presented for each hazard. For most of them, overlaps may exist among the different notifications, because some classifications were present in more than one of them.

				Aquatic Chronic 2	H411 (Toxic to aquatic life with long lasting effects)	85	A further 86 companies classified the substance in 4 distinct notifications. 42 of them classified it as acute toxic and 67 as aquatic chronic (43 as aquatic chronic cat. 2 and 24 as aquatic chronic cat. 1). There was a single notification where the company classified it as skin irritant 2 and eye irritant 2.
				Aquatic Acute 1	H400 (Very toxic to aquatic life)		
				Aquatic Chronic 1	H410 (Very toxic to aquatic life with long lasting effects)		
				Skin Irrit. 2	H315 (Causes skin irritation)		
				Eye Irrit. 2	H319 (Causes serious eye irritation)		
DINCH **EC:** 605-439-7	REACH Registered; TDI established by EFSA; Not classified under CLP	-	-	No classification	-	109	-
DINP 1[4]				No classification	-	244	

1,2-Benzenedicarboxylic acid, di-C8-10-branched alkyl esters, C9-rich **EC:** 271-090-9	REACH Registered; 100 000 - 1 000 000 tonnes per annum TDI established by EFSA; Some differences in CLP classifications notified by various parties (particularly regarding reprotoxic classification); Recent opinion by RAC on human health hazard identifies concerns					3 joint plus 56	The lead registrant and a further 69 notifiers found that data was lacking.
				Aquatic Acute 1	H400 (Very toxic to aquatic life)	56	A further 29 companies classified the substance in 4 distinct notifications.
				Repr. 2	H361 (Suspected of damaging fertility or the unborn child)		
				Skin Irrit. 2	H315 (Causes skin irritation)		
				Eye Irrit. 2	H319 (Causes serious eye irritation)		
DINP 2[5] **EC:** 249-079-5	100 000 - 1 000 000 tonnes per annum			No classification	-	603	The lead registrant and a further 793 notifiers did not classify the
				-	-	85	Data lacking.
					H400	1 1	
				Aquatic Tox 4	H332 (Harmful if inhaled)		
				Aquatic Acute 1	H400 (Very toxic to aquatic life)		

TOTM **Tris(2-ethylhexyl) benzene-1,2,4-tricarboxylate** **EC:** 222-020-0	REACH Registered; 10 000 - 100 000 tonnes per annum Some differences in CLP classifications notified by various parties (particularly regarding reprotoxic classification); Entered in CoRAP list (2012-2014) due to environmental PBT concern PBT corapSuspected PBT/vPvB			No classification	-	162	The lead registrant and a further 353 notifiers did not classify the substance.
				Repr. 2	H361 (Suspected of damaging fertility or the unborn child)	91	A further 70 companies classified the substance in 4 distinct classifications. Reproductive toxicity cat.2 was notified by 55 of the companies as the single hazard of the substance. There were 10 who classified it as skin, eye and respiratory irritant (STOT SE 3). There were also 5 companies that classified it as Aquatic Chronic 4.
				Acute Tox. 4	H312 (Harmful in contact with skin)		
				Eye Irrit. 2	H319 (Causes serious eye irritation)		
				Skin Irrit. 2	H315 (Causes skin irritation)		
				STOT SE 3	H335 (May cause respiratory irritation)		

				Aquatic Chronic 4	H413 (May cause long lasting harmful effects to aquatic life)		
DINA **EC:** 251-646-7	-	-	-	-	-	123 (joint)	Not classified
GTA **EC:** 203-051-9	- -	-	-	-	-	1064 (joint)	Not classified
				Flam. Liq. 3	H226 (flammable liquid and vapour)	71	
				Skin Irrit. 2	H315 (causes skin irritation)		
				Skin Sens. 1	H317 (may cause an allergic skin reaction)		
				-	H226 (flammable liquid and vapour)	2	
				-	H315 (causes skin irritation)		
				-	H317 (may cause an allergic skin reaction)		

				-	-	42	Data lacking
DEGD **EC:** 204-407-6	-	-	-	-	-	204 (joint)	Not classified
				-	-	72	Data lacking
				-	H411 (toxic to acquatic life with long lasting effects)	11	
				Aquatic Chronic 2	H411 (toxic to aquatic life with long lasting effects)	1	
				Eye Irrit. 2	H319 (Causes serious eye irritation)	1	
DGD **EC:** 248-258-5	-	-	-	Aquatic Chronic 3	H412 (harmful to aquatic life with long lasting effects)	69 (joint)	
				-	-	241	Not classified
				Aquatic Chronic 2	H411 (toxic to aquatic life with long lasting effects)	137	

	-	-	-	-	-	67	Data lacking
				Eye Irrit. 2	H319 (causes serious eye irritation)	47	
				Aquatic Chronic 2	H411 (toxic to aquatic life with long lasting effects)		
				Repr. 2	H361 (Suspected of damaging fertility or the unborn child)	16	
				-	H412 (harmful to aquatic life with long lasting effects)	11	
INBP EC:	-	-	-	-	-	-	-

Source: AfA 2013a, ECHA

Please note: Notifications from the classification and labelling inventory relate to the substance which is placed on the market, including all impurities. These impurities can also contribute to final hazard profile of marketed substance.

Appendix D4 Additional valuation of health benefits

The Dossier Submitter prepared the following analysis on request of SEAC.

Valuation of infertility and malformations using alternative Willingness to pay values

On request of SEAC, the Dossier Submitter used the higher wiliness to pay (WTP) values in ECHA 2016b. Table D41 summarises the values used for the new estimates of the social damage from infertility and malformations due to exposure to the four phthalates in articles in scope of this restriction proposal. All remaining assumptions are as presented in the preceding sections.

Table D40 High Willingness to pay values uses in Table D42

Willingness to pay value	2012 values (euro)	2014 values (euro)
Value of a statistical case of Healthy Child: MINOR birth defects	43 000	43 450
Value of a statistical case of Healthy Child: defects in INTERNAL organs	712 000	719 460
Value of a statistical case of Healthy Child: defects on EXTERNAL body parts	330 000	333 460
Value of statistical infertility (in vitro fertilisation treatment)*	55 810	56 390

Source: ECHA 2016b

Notes: * Estimated using the Value of statistical pregnancy, reflecting that not all pregnancies result in a life birth.

Table D41 Estimation of the social damage from infertility and malformations due to exposure to the four phthalates in articles in scope, using high Willingness to pay values presented in Table D41 (Euro)*

Total costs – representative year	Low estimate	Mid-point estimate	High estimate
Infertility (2050)	31 268 399	62 007 437	92 746 475
Cryptorchidism (2020)	16 247 755	162 479 258	406 215 193
Hypospadias (2020)	6 837 657	68 376 571	170 941 427
Total	54 353 811	292 863 266	669 903 095
Total costs (2014) – 4% discount factor	**Low estimate**	**Mid-point estimate**	**High estimate**
Infertility	7 619 100	15 109 300	22 599 400
Cryptorchidism	12 840 800	128 409 700	321 037 800
Hypospadias	5 403 900	54 039 000	135 097 500
Total	25 863 800	197 558 000	478 734 700
Total costs (2014) – 2% discount factor	**Low estimate**	**Mid-point estimate**	**High estimate**
Infertility	15 328 500	30 397 500	45 466 500
Cryptorchidism	14 427 500	144 276 900	360 707 500
Hypospadias	6 071 600	60 716 400	151 791 100
Total	35 827 600	235 390 800	557 965 100

Notes: * Low/Mid-point/High estimate refers to the estimation of the number of cases of infertility or malformations assuming different fraction of these health impacts that can be

attributed to chemicals such as the four phthalates. See Table D21, Table D29, and Table D32 for details on this fraction.

Break-even analysis

This section presents the combination of the number of cases of human health impacts to be avoided as a result of the proposed restriction that demonstrate that its benefits exceed its costs. Twelve different scenarios are presented, which vary:

- the WTP values used to monetise benefits (low – as shown in Annex D and Appendix D1, and high – as shown in Table D41)
- the discount rate: 4% and 2%, the latter taking into account that the income elasticity is one, therefore, with increased income, the WTP values will increase by the same rate (assumed 2% in accord with recent forecasts of GDP growth for EU28)
- the fractions of the human health cases attributable to chemicals similar to the four phthalates: See Table D21, Table D29, Table D32 and Table D36 for details on this fraction

Table D43 presents the number of cases and the weighted average per case as calculated in the preceding sections. Using the Solver function in MS Excel, the number of break-even cases was calculated while minimising the difference between the Total benefits (= weighted average per case * number of break-even cases) and Total restriction costs (=€16.9 million). Other combinations of the number of infertility, malformation or asthma cases are also possible, although they may not optimally minimise the difference between Total benefits and Total costs.

Table D42 Estimation of the combination of the number of cases of human health impacts to be avoided which demonstrate the benefits of the proposed restriction exceed its cost – 12 sensitivity scenarios varying WTP values, discount rates and attributable fractions

4%, low WTP, mid-case	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	2 110	67	4 626	311 023	3.2%	0.12%	0.04%	0.00%	
Cryptorchidism	480	422	29 049	12 265 597	88.0%	0.78%	0.24%	0.02%	
Hypospadias	540	248	17 069	4 234 923	45.9%	0.46%	0.14%	0.01%	
Asthma	16 800	39	2 680	104 401	0.2%				0.000%
Total				16 915 944		1.366%	0.419%	0.028%	
Male infertility only	2 110	3 657	4 626			6.77%	2.08%	0.14%	
4%, low WTP, low # cases	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	1 050	979	4 687	4 586 434	93.2%	1.81%	0.56%	0.038%	
Cryptorchidism	48	48	24 959	1 198 021	100.0%	0.09%	0.03%	0.002%	
Hypospadias	54	54	17 069	921 718	100.0%	0.10%	0.03%	0.002%	
Asthma	6 720	3 810	2 680	10 209 772	56.7%				0.001%
Total				16 915 944		2.001%	0.614%	0.042%	
Male infertility only	1 050	3 609	4 687			6.68%	2.05%	0.14%	
4%, low WTP, high # cases	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	3 160	55	4 605	253 030	1.7%	0.10%	0.03%	0.002%	

Cryptorchidism	1 200	395	33 139	13 101 388	32.9%	0.73%	0.22%	0.015%	
Hypospadias	1 340	204	17 069	3 475 838	15.2%	0.38%	0.12%	0.008%	
Asthma	67 210	32	2 680	85 688	0.0%				0.000%
Total				16 915 944		1.211%	0.372%	0.025%	
Male infertility only	3 160	3 673	4 605			6.80%	2.09%	0.14%	
2%, low WTP, mid-case	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	2 110	103	9 310	959 072	4.9%	0.19%	0.06%	0.00%	
Cryptorchidism	480	361	32 638	11 786 958	75.2%	0.67%	0.21%	0.01%	
Hypospadias	540	212	19 178	4 069 663	39.3%	0.39%	0.12%	0.01%	
Asthma	16 800	33	3 010	100 250	0.2%				0.000%
Total				16 915 944		1.253%	0.384%	0.026%	
Male infertility only	2 110	1 817	9 310			3.36%	1.03%	0.07%	
2%, low WTP, low # cases	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	1 050	1 048	9 430	9 884 605	99.8%	1.94%	0.60%	0.04%	
Cryptorchidism	48	48	28 043	1 346 057	100.0%	0.09%	0.03%	0.00%	
Hypospadias	54	54	19 178	1 035 612	100.0%	0.10%	0.03%	0.00%	
Asthma	6 720	1 545	3 010	4 649 670	23.0%				0.0003 %
Total				16 915 944		2.130%	0.654%	0.044%	
Male infertility only	1 050	1 794	9 430			3.32%	1.02%	0.07%	

2%, low WTP, high # cases	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	3 160	85	9 270	786 127	2.7%	0.16%	0.05%	0.00%	
Cryptorchidism	1 200	341	37 233	12 682 285	28.4%	0.63%	0.19%	0.01%	
Hypospadias	1 340	175	19 178	3 364 648	13.1%	0.32%	0.10%	0.01%	
Asthma	67 210	28	3 010	82 883	0.0%				0.000%
Total				16 915 944		1.113%	0.341%	0.023%	
Male infertility only	3 160	1 825	9 270			3.38%	1.04%	0.07%	
4%, high WTP, mid-case	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	2 110	6	7 161	42 965	0.3%	0.01%	0.00%	0.00%	
Cryptorchidism	480	55	267 949	14 852 578	11.5%	0.10%	0.03%	0.00%	
Hypospadias	540	20	100 992	2 019 838	3.7%	0.04%	0.01%	0.00%	
Asthma	16 800	0	2 680	563	0.0%				0.000%
Total				16 915 944		0.151%	0.046%	0.003%	
Male infertility only	2 110	2 362	7 161			4.37%	1.34%	0.09%	
4%, high WTP, low cases	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	1 050	15	7 256	108 982	1.4%	0.03%	0.01%	0.00%	
Cryptorchidism	48	47	267 945	12 681 205	98.6%	0.09%	0.03%	0.00%	
Hypospadias	54	41	100 992	4 118 722	75.5%	0.08%	0.02%	0.00%	

Asthma	6 720	3	2 680	7 036	0.0%				0.000%
Total				16 915 944		0.191%	0.059%	0.004%	
Male infertility only	1 050	2 331	7 256			4.32%	1.32%	0.09%	
4%, high WTP, high cases	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	3 160	9	7 152	64 365	0.3%	0.02%	0.01%	0.00%	
Cryptorchidism	1 200	55	267 960	14 825 510	4.6%	0.10%	0.03%	0.00%	
Hypospadias	1 340	20	100 992	2 019 838	1.5%	0.04%	0.01%	0.00%	
Asthma	67 210	2	2 680	6 231	0.0%				0.000%
Total				16 915 944		0.156%	0.048%	0.003%	
Male infertility only	3 160	2 365	7 152			4.38%	1.34%	0.09%	
2%, high WTP, mid-case	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	2 110	9	14 406	129 658	0.4%	0.02%	0.01%	0.00%	
Cryptorchidism	480	55	301 059	16 559 344	11.5%	0.10%	0.03%	0.00%	
Hypospadias	540	2	113 471	226 942	0.4%	0.00%	0.00%	0.00%	
Asthma	16 800	0	3 010	0	0.0%				0.000%
Total				16 915 944		0.122%	0.038%	0.003%	
Male infertility only	2 110	1 174	14 406			2.17%	0.67%	0.05%	
2%, high WTP, low cases	# cases estimated	# break-	weighted	Total value (in euro)	Break even cases as percentage of				

Human health impacts	in main analysis	even cases	average €/case		# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	1 050	5	14 599	73 102	0.5%	0.01%	0.00%	0.00%	
Cryptorchidism	48	48	301 055	14 334 935	99.2%	0.09%	0.03%	0.00%	
Hypospadias	54	22	113 470	2 498 876	40.8%	0.04%	0.01%	0.00%	
Asthma	6 720	3	3 010	9 032	0.0%				0.000%
Total				16 915 944		0.138%	0.042%	0.003%	
Male infertility only	1 050	1 159	14 599			2.15%	0.66%	0.04%	
2%, high WTP, high cases	# cases estimated in main analysis	# break-even cases	weighted average €/case	Total value (in euro)	Break even cases as percentage of				
Human health impacts					# cases in main analysis	population at risk due to foetal exposure	population at risk due to early childhood exposure	males born annually	total population
Male infertility	3 160	1	14 388	14 388	0.0%	0.00%	0.00%	0.00%	
Cryptorchidism	1 200	55	301 071	16 558 133	4.6%	0.10%	0.03%	0.00%	
Hypospadias	1 340	3	113 471	340 413	0.2%	0.01%	0.00%	0.00%	
Asthma	67 210	1	3 010	3 010	0.0%				0.000%
Total				16 915 944		0.109%	0.034%	0.002%	
Male infertility only	3 160	1 176	14 388			2.18%	0.67%	0.05%	

Appendix D5 Additional valuation of health benefit

As a result of the Forum advice on the enforceability of the Annex XV proposal for restriction on four phthalates (adopted on 21.09.2016), the Dossier Submitter prepared two revised versions of the proposed restriction: version A in the table below and version B in Table D2. In the opinion of RAC, SEAC and the Forum, version B is more enforceable and better captures the intended scope of the restriction. Version A is included for solely reverence purposes.

Table D43 Proposed restriction: interim proposal (Version A)

Bis(2-ethylhexyl) phthalate (DEHP) EC number: 204-211-0 CAS number: 117-81-7 Benzyl butyl phthalate (BBP) EC number: 201-622-7 CAS number: 85-68-7 Dibutyl phthalate (DBP) EC number: 201-557-4 CAS number: 84-74-2 Diisobutyl phthalate (DIBP) EC number: 201-553-2 CAS number: 84-69-5	1. Articles or any parts thereof containing DEHP, DBP, DIBP, and BBP in a concentration, individually or in any combination, greater than or equal to 0.1% by weight of each plasticised material shall not be placed on the market. 2. Paragraph 1 shall not apply to: a. articles for outdoor use where the phthalate-containing material is not in prolonged contact with human skin or in any contact with human mucous membranes, e.g. through mouthing b. articles for use in industrial or agricultural workplaces. This derogation does not apply to articles where the phthalate-containing material is in prolonged contact with human skin by workers c. measuring devices for laboratory use or articles that form part of measuring devices for laboratory use[207] d. toys and childcare articles subject to entry 51 of this Annex e. spare parts for the maintenance of vehicles for which it can be demonstrated that they have been_placed on the market for the first time in the European Union prior to the date in paragraph 5 f. articles for which it can be demonstrated that they have been_placed on the market for the first time in the European Union prior to the date in paragraph 5. 3. Paragraphs 1 and 2 shall not apply to articles in the scope of: a. Food contact materials covered by Regulation (EC) No 1935/2004 and Regulation (EU) No 10/2011 on plastic materials. b. Immediate packaging of medicinal products covered by Regulation (EC) No 726/2004, Directive 2001/82/EC or Directive 2001/83/EC c. Medical devices covered by Directive 90/385/EEC, Directive 93/42/EEC or Directive 98/79/EC or components for such devices. d. Articles covered under Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (RoHS Directive). 4. The following definitions apply to this entry: a. "Prolonged contact with human skin" shall mean a daily overall contact with skin of more than 10 minutes continuously or 30 minutes discontinuously, under normal and reasonably foreseeable conditions of use. b. “Only for outdoor use” shall mean articles which are not used (including stored) in the interior of buildings and vehicles where humans are present

[207] See ECHA Q&A#1179 for definition of measuring devices.

	and potentially exposed via inhalation under normal and reasonably foreseeable conditions. c. "Industrial or agricultural workplaces" shall mean any commercial activities performed by workers in a workplace in the following sectors: • agriculture, forestry and fishing [NACE A] • mining and quarrying [NACE B] • manufacturing [NACE C] • electricity, gas, steam and air conditioning supply [NACE D] • water supply; sewerage; waste management and remediation activities [NACE E] • construction [NACE F] d. "Childcare article" shall mean any product intended to facilitate sleep, relaxation, hygiene, the feeding of children or sucking on the part of children. 5. The restriction shall apply three years from its entry into force.
Amendment of entry 51 of Annex XVII of REACH	An amendment of the restriction entry to include DIBP in its scope.

Appendix D6 Derogations on articles in vehicles

The Dossier Submitter ran an extensive public consultation prior to the preparation of the restriction dossier (public consultations to the 2012 Danish restriction proposal and call for evidence ran by ECHA in 2015). No information was submitted regarding difficulties to replace the four phthalates in automotive vehicles and aircraft.

1. Aircraft

Background:

At the closing of the public consultation (PC) on the submitted Annex XV restriction dossier, a representative of the aircraft industry [redacted] submitted information on the difficulty in replacing articles in the interior of the aircraft and spare parts for aircraft already in operation (see submission #1506). The Dossier Submitter followed up with the industry association, Aerospace and Defence Industries of Europe (ASD-Europe), who surveyed their members on the potential impacts of the industry. [redacted] Therefore, there was insufficient information to justify a derogation on articles in the interior of aircraft. This conclusion was supported by both RAC and SEAC. However, there appeared to be a need for derogation for spare parts for aircraft for which a type certificate is issued (this also relates to the discussion for automotive vehicles and will not be repeated in that section), as aircraft have long useful lives and may be rendered inoperable in the absence of an essential spare part. Such derogation was not supported by RAC in their final opinion (FO) on the grounds that there is insufficient information on the volumes and risk due to exposure to these articles. Therefore, it was proposed that further information would be gathered on derogations during the PC on the SEAC draft opinion (DO).

[redacted] No information on possible type of spare parts was provided but based on historical use of the four phthalates, the Dossier Submitter assumes that at least some of these articles can lead to dermal and inhalation exposure to the general population and personnel involved in the operations and maintenance of the aircraft.

In addition, the submission outlined issues with the following remaining uses part of an ongoing production that cannot be substituted by 2020: [redacted]

[redacted]

Proposed derogation:

It is proposed that parts, products or appliances of aircraft for which a type certificate in the meaning of Regulation No 748/2012 has been issued prior to entry into force of the restriction are derogated. As the industry has largely replaced the four phthalates with suitable alternatives, it is important to derogate only those articles without which the airworthiness of the aircraft is jeopardised and for which it can be demonstrated that the use of the four phthalates cannot be readily substituted under the terms of the type certificate requirements, as it is recognised that time is necessary to fully qualify, certify and implement an alternative under these requirements. This will ensure that unnecessary prolongation of the phase out of the four phthalates by the industry is avoided, in particular since viable alternatives exist.

Assessment:

- Targeting and risk reduction capacity: Parts under type certificate requirements may lead to dermal or inhalation exposure of passengers and aircraft operations and maintenance personnel. As the restriction is targeted at the combined exposure of diverse types of articles, the derogation on selected aircraft articles would diminish the risk reduction capacity of the proposed restriction. As the derogation is aimed at only essential articles for the airworthiness of the already operational aircraft and aircraft articles which receive type certificate prior to the entry into force of the proposed restriction, it is anticipated that a limited number of articles would be derogated. Their number is expected to decline in the future as older aircraft is retired.
- Restriction costs and other socio-economic impacts:
    - Alternatives: [redacted] Therefore, it can be concluded that alternatives for all current uses exist.
    - Time to transition to alternatives: While alternatives exist, those parts, products and appliances with issued type certificate would need to undergo requalification. This would involve demonstration of an equivalent or superior performance in the environment where the part is used. According to the PC submissions, this is a long and costly process taking more than a decade in some cases [redacted]
    - Overall proportionality: Aircraft have long service lives, typically 20-40 years. To comply with the airworthiness requirements throughout the lifespan of the product, certified parts must be available for maintenance and repair to ensure safe operations. Changing a design and recertifying individual articles take financial and other resources. These costs have not been taken into account in the compliance costs of the proposed restriction. Therefore, if no derogation is proposed, the costs of the proposed restriction would increase (with the value of

recertification and if recertification is not complete before the entry into force of the proposed restriction, with the replacement costs of aircraft which was prematurely retired due to lack of compliant spare parts). As the derogation would impact the risk reduction capacity, the benefits of the restriction are expected to be lower than the estimated in Annex D. The impact of this on the benefit-cost ratio is uncertain: substantial substitution has been taking place (therefore, the impact on risk reduction and benefits could be expected to be limited), on the other hand, there is insufficient information on the volume of the phthalates in these spare parts and likely exposure. Overall, while no sufficient information was submitted to quantify the impacts of a derogation on articles for aircraft, it appears that without an exemption, the costs of the restriction will be higher and the proportionality of the overall restriction may be eroded.

- Practicality and monitorability: It is anticipated that the derogation is enforceable via the terms of the type certificate, which clearly delineates the requirements for aircraft manufacturers. The clear wording of the restriction is paramount to avoid expanding its scope to non-essential articles, not covered by type certificate requirements.

2. Automotive vehicles

Background:

During the PC on the dossier, the European Automotive Manufacturers Association (ACEA) requested two derogations. (see submission #1478) The first one related to vehicles placed on the EU market for the first time lasting up to 3 years after entry into force of the restriction. It was requested on the grounds that components and material testing for vehicles currently in the engineering pipeline has been completed, contracts with suppliers have been signed and volumes and delivery of parts have been agreed. Changes in the specification of materials and parts late in the development timeline could delay the launch of a new model, costing €1 million per day.

The second requested derogation related to spare parts and remanufactured parts for vehicles on the grounds that sustainable supply of spare parts is required to enable vehicles currently in operations to be maintained and repaired. According to ACEA:

- it is normal to offer spare parts for 15 years or longer after the end of mass production;
- type approval legal requirements and minimum warranty obligations must be fulfilled;
- after end of mass production, spare part manufacturing is often outsourced to SMEs, which continue production in very low volumes based on the original specification and
- stockpiling is not always possible due to material degradation among others.

In addition, to reduce the risk related to exposure via inhalation in the interior of a vehicle, the concentration limit of the four phthalates in air was proposed by ACEA to be restricted to 120µg/m^3.

As the information submitted was not sufficient to quantify the impacts of the derogations, additional information was requested from ACEA. The association responded by clarifying that problematic articles are only those that may lead to inhalation exposure, i.e., hidden within, or below, assemblies such as harnesses, hoses, rubbers, seals and tapes. (see submission #1506_2)

Therefore, in addition to the spare parts derogation for vehicles (aircraft and automotive), a derogation on articles in the interior of the vehicle, hidden within or below assemblies was discussed in plenary. Such targeted derogation was seen as a more suitable way to introduce a derogation on such parts as opposed to an interior concentration limit (which could also capture articles that lead to dermal exposure). Such derogation was not supported by RAC in their FO due to the absence of information on the degree of inhalation exposure, the contribution to the risk and definition of such parts. Therefore, it was discussed that further information would be gathered on derogations during the PC on the SEAC DO.

As part of the PC on the SEAC DO, ACEA narrowed their request to derogations on articles leading to inhalation exposure for:

- automotive vehicles and spare parts that were produced before the entry into force of the proposed restriction plus five years, and
- legacy spare parts for vehicles that have ceased mass production prior to the entry into force of the proposed restriction plus five years. (see submission #320)

Proposed derogation:

It is proposed to derogate articles leading to inhalation exposure for automotive vehicles, which are produced prior to the date of entry into effect[208] plus 2 years (or entry into force plus five years) as well as spare parts for these vehicles where the vehicle cannot function as intended without that spare part. Such articles, however, can continue to be used in industrial workplaces if they do not lead to dermal exposure.

Assessment:

- Targeting and risk reduction capacity: According to ACEA, there are about 80 million operational cars in the EU, divided almost equally in three groups in terms of age: less than five-years-old, between five and 10 years, and older than 10 years. In addition, industry has stated that the use of the four phthalates has significantly declined over the past five years, as they have largely replaced the four phthalates in EU manufactured carts. Therefore, the derogation would likely largely apply to older EU made or imported cars. With that said, it is difficult to estimate the impact of the derogation on the risk reduction capacity. According to ACEA, there are 4 000 – 9 000 main components and assemblies in a single vehicle. How many of those may contain phthalates and their typical concentration is uncertain as, according to ACEA, the supply chain comprises of several thousand (often) SMEs, there is no steady demand for individual spare parts and the accuracy of reported data may vary. Overall, it can be concluded that the derogation would diminish the risk reduction capacity of the overall restriction, while older vehicles remain operational.
- Restriction costs and other socio-economic impacts
    - Alternatives: Industry has largely replaced the four phthalates in vehicles and many of the remaining applications concern imported articles only.

[208] I.e., entry into force plus transitional period of three years.

- Time to transition to alternatives: Typically, around 75% the vehicles original equipment components and technology are sourced from automotive suppliers. Original equipment manufacturers (OEM) on average have 1 500 – 4 500 tier one suppliers who in turn have 500 – 1 500 tier two supplier, which in turn have their own suppliers. Therefore, according to ACEA, due to the global nature of the industry, the complexity of the supply chain and the time required for development, validation and testing (in the original vehicle), three to five years may be required to transition to alternatives.
- Overall proportionality: Redesigning parts for vehicles currently in the engineering pipeline can lead to delays costing estimated €1 million per day to OEMs in addition to potential reputational damage due to possible delays or recalls. Redesigning parts for vehicles no longer in manufacturing can be costly (€20 000 - €150 000 plus system/vehicle level validations). In some cases, it may not be economically feasible due to the low production volumes of service parts, often manufactured by SMEs. These SMEs may not have the necessary resources to transition to alternatives and absorb the costs and could lead to increased prices of these parts. Furthermore, the redesign may be technically challenging as the altered parts need to be validated in the original vehicle which may not be available to the spare part manufacturer. Car manufacturers commit to replacing parts to their models (10-20 years after their manufacturing). Without the derogation on parts that lead to inhalation exposure, society would incur costs in terms of industry costs related to redesign, testing and validation of vehicle parts or associated with the premature retiring of currently operational cars, if essential parts for their safety and proper functioning are not available. These costs have not been included in the total compliance costs for the restriction. Therefore, it can be expected that without the derogation, the total restriction costs estimated in Annex D would increase and the cost-effectiveness would decline. As the derogation would impact the risk reduction capacity, the benefits of the restriction are expected to be lower than the estimated in Annex D. The impact of this on the benefit-cost ratio is uncertain: substantial substitution has been taking place (therefore, the impact on risk reduction and benefits could be expected to be limited), on the other hand, there is insufficient information on the volume of the phthalates in these articles and likely exposure. Overall, while no sufficient information was submitted to quantify the impacts of a derogation on articles for automotive vehicles, it appears that without an exemption, the costs of the restriction will be higher and the proportionality of the overall restriction may be eroded.

- Practicality and monitorability: The derogation is consistent with the principle “repair as produced” taken up by other EU legislation, e.g., Commission decision 2005/438/EC. The clear wording of the restriction is paramount to avoid expanding its scope to non-essential to the functioning of the vehicle articles.

Note: The analysis above was developed primarily on the basis of information submitted during the PC on the dossier and the PC on the SEAC DO. Due to time constraints no additional validation of industry claims has been performed.

Annex E: Assumptions, uncertainties and sensitivities

This Annex discusses the impact of key assumptions on the risk reduction capacity, cost-effectiveness and the benefit-cost ratio of the proposed restriction.

E.1. Impacts of assumptions regarding the Baseline scenario on risk reduction capacity and cost-effectiveness

Two extreme case baseline scenarios were prepared: Scenario 1 (High tonnage) and Scenario 3 (Low tonnage) in addition to Scenario 2: Baseline (Main) which is used for the purpose of presenting the impacts of the proposed restriction in the main report and other annexes. These scenarios test the impact of the assumptions on the risk reduction capacity and cost-effectiveness of the proposed restriction. They relate to the future substitution of DEHP, DBP, DIBP, and BBP as a result of the requirements for authorisation as well as the growth in imports. The variables tested are described in Table E1.

Table E1 Baseline scenario assumptions – description

Assumptions for:	**From 2019 to Assumed entry into effect + 20 years**		
	Scenario 1 (High tonnage)	**Scenario 2: Baseline (Main)**	**Scenario 3 (Low tonnage)**
DEHP			
Use in EU production of articles (including exports)	2013-2019: approximately 13% annual decline of DEHP tonnages used in EU article manufacturing, plus all wires & cables phased out by 2019 – the year of entry into force of RoHS amendments. See Annex C: Baseline for further details.		
	0% change from 2019 levels in DEHP use in EU article manufacturing - no further regulatory pressure for substitution of DEHP (authorisations granted for 2019 volumes); Opposite but equal forces at work: increased use because of higher demand for end-products due to population & income growth, which is balanced out by decreased use of DEHP in the EU article production due to relocating production outside the EU	3.5% annual phase out of DEHP use in EU manufacturing due to substitution from 2019 onward; Authorisations are granted for the remaining volumes; Opposite but equal forces at work: increased use because of increased demand for end-products due to population & income growth , which is balanced out by decreased use of DEHP in the EU article manufacturing due to relocating production outside the EU (outsourcing)	0 tonnes of DEHP used in EU manufacturing in 2019, i.e., 100% phase out of no authorisations are requested and granted
Import	Wires & cables containing the four phthalates will no longer be imported past 2019.		
	2% annual growth of the DEHP contained in imported articles since 2014 explained with similar but stronger demand forces as described in Scenario 2	1% annual growth since 2014 explained with opposite but unequal forces: Increase of DEHP contained in imported articles because of increased demand for end-products due to population & income growth and due to higher outsourcing (to e.g., China). This force is larger than the decline in the DEHP in imports due to substitution as no further regulatory action is anticipated.	0% change in annual growth since 2014, explained with: Increase of DEHP contained in imported articles because of increased demand for end-products due to population & income growth and due to higher relocating outside the EU (to e.g., China). This growth is equal to the decline in the DEHP in imports due to substitution internationally.
DBP, DIBP and BBP			
Use in EU production of articles	Full phase out by 2015. No exports containing DBP, DIBP and BBP		
Import	Same as DEHP	Same as DEHP	Same as DEHP

Notes: The Scenario 2: Baseline (Main) is used to develop the main analysis of the impacts in Annex D and the main report.

Scenario 1 (High tonnage) is seen as the upper bound of what could be expected on the EU market in the future, as it is likely that at least part of the tonnages still in use in EU article production are phased out (either due to outsourcing of production internationally, due to the aversion to work in an environment where regular reviews of the granted authorisations are required, or due to not being able to obtain an authorisation).

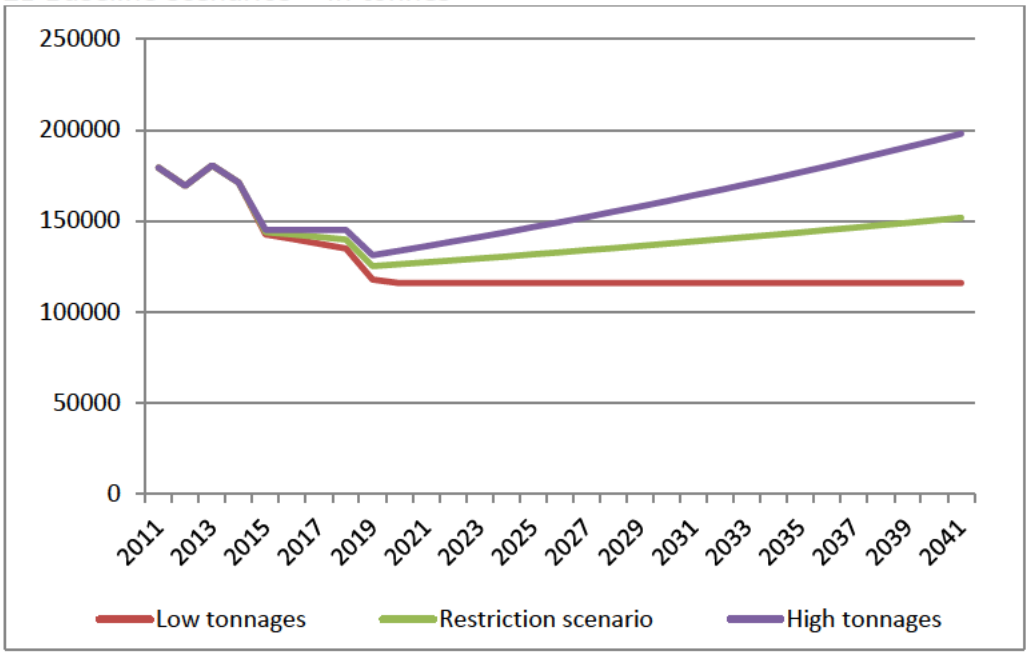
Scenario 1 assumes 2% annual increase in the tonnage of the four phthalates contained in imported articles. It could be argued that this is modest, given the business and regulatory environment of some of the EU's international trading partners. For example, China, the place of origin of majority of imported articles, has a cost competitive environment which continues to attract investment in article manufacturing. Furthermore, the market is experiencing substantial DEHP overcapacity (BASF 2011, TOC 2012). Therefore, this coupled with anticipated increased demand in the EU28 due to population and income growth, it is possible that the increase in tonnages of the four phthalates in articles originating in China is larger than 2% annually. However, it was deemed inappropriate to assume that these trends would persist throughout the next 20-25 years; therefore, a moderate growth of 2% was selected.

Scenario 3 (Low tonnage) describes the low tonnage boundary of the baseline. It also is not considered to be a realistic scenario because:

- As explained above, the tonnages of the four phthalates in imported articles is likely to increase in the future
- It is possible that some EU manufactures obtain authorisations at least for some niche applications.

Figure E1 depicts the tonnages of the four phthalates in articles consumed within the EU28 under the three baseline scenarios.

Figure E1 Baseline scenarios – in tonnes



Impacts of Baseline assumptions on risk reduction capacity

The projected risk levels for the Low tonnage and High tonnage scenarios for the years 2020 and 2039 are presented in Table E2 - Table E5. The projected risk levels for the Main scenario are presented in Tables D12-D13 in Annex D. It is clear from these tables that the projected risks are not very sensitive to the baseline assumptions. The High tonnage scenario leads to 2% and 12% higher RCRs compared with the Main scenario in 2020 and 2039 respectively. The High tonnage scenario leads to 2% and 10% lower RCRs compared with the Scenario 2: Baseline (Main) in 2020 and 2039 respectively.

The level of risk following an entry into force of the proposed restriction is given in Table D14 in Annex D and remains the same regardless of the baseline projections[209].

It can thus be concluded that the risk reduction capacity of the proposed restriction is not very sensitive to the baseline assumptions.

[209] Under the simplifying assumption of a one-to-one relationship between the baseline volumes and the risk from articles in the scope of the restriction, there will be no exposure from articles in the scope of the proposed restriction once it enters into force. The remaining exposure will be from articles outside the scope, especially from exposure via food. Since these articles are not assumed to be impacted by the baseline, the exposure levels remain indifferent to the baseline assumptions.

Table E2 RCRs for four phthalates as estimated from 95th percentile urinary biomonitoring values projected to 2020 *in the Low tonnage baseline scenario (no restriction)*

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.2	0.0	*NA*	0.4	120	0.2	0.2	0.0	*NA*	0.4
UK	21	0.1	0.1	0.0	0.1	0.3	21	0.1	0.1	0.0	0.1	0.4
CH	117	0.2	0.1	0.0	0.1	0.4	119	0.2	0.1	0.0	0.1	0.5
CY	59	0.4	0.1	0.0	0.2	0.7	60	0.2	0.1	0.0	0.2	0.5
LU	60	0.1	0.1	0.0	0.1	0.4	60	0.1	0.1	0.0	0.4	0.6
PT	117	0.3	0.1	0.0	0.2	0.6	116	0.2	0.2	0.0	0.2	0.6
IE	120	0.2	0.1	0.0	0.2	0.5	120	0.3	0.1	0.0	0.2	0.6
DE	116	0.1	0.2	0.0	0.1	0.4	120	0.2	0.3	0.0	0.2	0.7
DK	143	0.1	0.1	0.0	0.2	0.4	142	0.2	0.1	0.0	0.3	0.7
HU	115	0.2	0.2	0.0	NA	0.5	117	0.4	0.3	0.0	NA	0.7
SE	96	0.2	0.4	0.0	NA	0.5	97	0.3	0.5	0.0	NA	0.8
SK	125	0.2	0.4	0.0	NA	0.6	127	0.4	0.6	0.0	NA	0.9
CZ	117	0.2	0.4	0.0	NA	0.6	120	0.4	0.7	0.0	NA	1.0
BE	125	0.1	0.2	0.0	0.3	0.6	125	0.3	0.2	0.0	0.5	1.1
ES	118	0.2	0.2	0.0	0.2	0.6	119	0.3	0.4	0.0	0.4	1.2
RO	117	0.9	0.1	0.0	0.2	1.2	119	0.8	0.3	0.0	0.3	1.4
PL	119	0.3	0.4	0.0	0.4	1.1	115	0.5	0.6	0.0	0.6	1.6

Table E3 RCRs for four phthalates as estimated from 95th percentile urinary biomonitoring values projected to 2020 *in the High tonnage baseline scenario (no restriction)*

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.2	0.0	*NA*	0.4	120	0.2	0.2	0.0	*NA*	0.4
UK	21	0.1	0.1	0.0	0.1	0.3	21	0.2	0.2	0.0	0.1	0.5
CH	117	0.2	0.1	0.0	0.1	0.4	119	0.2	0.1	0.0	0.1	0.5
CY	59	0.4	0.1	0.0	0.2	0.8	60	0.2	0.1	0.0	0.2	0.6
LU	60	0.1	0.1	0.0	0.1	0.4	60	0.1	0.1	0.0	0.4	0.6
PT	117	0.3	0.1	0.0	0.2	0.6	116	0.2	0.2	0.0	0.2	0.6
IE	120	0.2	0.1	0.0	0.2	0.5	120	0.3	0.1	0.0	0.2	0.7
DE	116	0.1	0.2	0.0	0.1	0.4	120	0.2	0.3	0.0	0.2	0.7
DK	143	0.2	0.1	0.0	0.2	0.5	142	0.2	0.2	0.0	0.3	0.7
HU	115	0.2	0.3	0.0	NA	0.5	117	0.4	0.4	0.0	NA	0.7
SE	96	0.2	0.4	0.0	NA	0.6	97	0.3	0.5	0.0	NA	0.8
SK	125	0.2	0.4	0.0	NA	0.6	127	0.4	0.6	0.0	NA	1.0
CZ	117	0.2	0.4	0.0	NA	0.6	120	0.4	0.7	0.0	NA	1.1
BE	125	0.1	0.2	0.0	0.3	0.7	125	0.3	0.2	0.0	0.5	1.1
ES	118	0.2	0.2	0.0	0.2	0.6	119	0.3	0.5	0.0	0.4	1.3
RO	117	1.0	0.1	0.0	0.2	1.3	119	0.8	0.3	0.0	0.3	1.5
PL	119	0.3	0.4	0.0	0.4	1.2	115	0.5	0.6	0.0	0.6	1.7

Table E4 RCRs for four phthalates as estimated from 95th percentile urinary biomonitoring values projected to 2039 *in the Low tonnage baseline scenario (no restriction)*

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.2	0.0	*NA*	0.4	120	0.2	0.2	0.0	*NA*	0.4
UK	21	0.1	0.1	0.0	0.1	0.3	21	0.1	0.1	0.0	0.1	0.4
CH	117	0.2	0.1	0.0	0.1	0.4	119	0.2	0.1	0.0	0.1	0.5
CY	59	0.4	0.1	0.0	0.2	0.7	60	0.2	0.1	0.0	0.2	0.5
LU	60	0.1	0.1	0.0	0.1	0.4	60	0.1	0.1	0.0	0.4	0.6
PT	117	0.3	0.1	0.0	0.2	0.6	116	0.2	0.2	0.0	0.2	0.6
IE	120	0.2	0.1	0.0	0.2	0.5	120	0.3	0.1	0.0	0.2	0.6
DE	116	0.1	0.2	0.0	0.1	0.4	120	0.2	0.3	0.0	0.2	0.7
DK	143	0.1	0.1	0.0	0.2	0.4	142	0.2	0.1	0.0	0.3	0.7
HU	115	0.2	0.2	0.0	NA	0.5	117	0.4	0.3	0.0	NA	0.7
SE	96	0.2	0.4	0.0	NA	0.5	97	0.3	0.5	0.0	NA	0.8
SK	125	0.2	0.4	0.0	NA	0.6	127	0.4	0.6	0.0	NA	0.9
CZ	117	0.2	0.4	0.0	NA	0.6	120	0.4	0.7	0.0	NA	1.0
BE	125	0.1	0.2	0.0	0.3	0.6	125	0.3	0.2	0.0	0.5	1.1
ES	118	0.2	0.2	0.0	0.2	0.6	119	0.3	0.4	0.0	0.4	1.2
RO	117	0.9	0.1	0.0	0.2	1.2	119	0.8	0.3	0.0	0.3	1.4
PL	119	0.3	0.4	0.0	0.4	1.1	115	0.5	0.6	0.0	0.6	1.6

Table E5 RCRs for four phthalates as estimated from 95th percentile urinary biomonitoring values projected to 2039 *in the High tonnage baseline scenario (no restriction)*

		Mother						Child				
Country	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**	**N**	**DEHP**	**DBP**	**BBP**	**DIBP**	**SUM**
SI	120	0.2	0.3	0.0	*NA*	0.4	120	0.2	0.3	0.0	*NA*	0.5
UK	21	0.1	0.1	0.0	0.2	0.3	21	0.2	0.2	0.0	0.2	0.5
CH	117	0.2	0.2	0.0	0.1	0.5	119	0.2	0.2	0.0	0.2	0.6
CY	59	0.5	0.1	0.0	0.3	0.9	60	0.2	0.1	0.0	0.3	0.7
LU	60	0.2	0.1	0.0	0.2	0.5	60	0.1	0.2	0.0	0.5	0.8
PT	117	0.4	0.1	0.0	0.2	0.7	116	0.3	0.2	0.0	0.3	0.8
IE	120	0.2	0.2	0.0	0.2	0.6	120	0.3	0.2	0.0	0.3	0.8
DE	116	0.1	0.2	0.0	0.1	0.5	120	0.2	0.4	0.0	0.2	0.8
DK	143	0.2	0.1	0.0	0.3	0.5	142	0.2	0.2	0.0	0.4	0.8
HU	115	0.3	0.3	0.0	NA	0.6	117	0.4	0.4	0.0	NA	0.8
SE	96	0.2	0.5	0.0	NA	0.7	97	0.3	0.6	0.0	NA	1.0
SK	125	0.2	0.5	0.0	NA	0.7	127	0.4	0.7	0.0	NA	1.2
CZ	117	0.2	0.5	0.0	NA	0.7	120	0.4	0.9	0.0	NA	1.3
BE	125	0.1	0.3	0.0	0.4	0.8	125	0.4	0.3	0.0	0.7	1.3
ES	118	0.3	0.2	0.0	0.2	0.7	119	0.4	0.6	0.0	0.6	1.5
RO	117	1.1	0.2	0.0	0.2	1.4	119	0.9	0.4	0.0	0.4	1.7
PL	119	0.4	0.5	0.0	0.5	1.4	115	0.5	0.7	0.0	0.8	2.1

Impacts of Baseline assumptions on cost-effectiveness

Table E6 shows that the changes in the baseline assumptions have almost no impact on the cost-effectiveness of the proposed restriction. Its cost-effectiveness would be reduced only by about 2.5% even in the Low tonnage scenario that assumes that no authorisations will be granted after 2020 and the content of the four phthalates in imported articles would remain unchanged at 2014 levels for the remainder of the study period. The cost-effectiveness of the restriction would improve by just over 2% in the High tonnage scenario that assumes that everyone who has not phased out the four phthalates by 2019 would receive an authorisation for unchanged volumes and the four phthalate tonnages in imported articles would increase by 2% annually.

As Scenarios 1 and 3 represent the upper and lower bound of future tonnages of the four phthalates placed on the EU28 market, the proposed restriction would remain cost-effective under all reasonably foreseeable baseline scenarios.

Table E6 Impact of changes of baseline assumptions on the cost-effectiveness of the proposed restriction

Costs	**Baseline scenarios**		
	Scenario 1: High Tonnage	**Baseline (Main)**	**Scenario 3: Low Tonnage**
Tonnes replaced - 2020 EiF	153 690	131 562	111 717
Total restriction costs - 2020 EiF (annual, million euro)	19.4	16.9	14.7
Cost-effectiveness (euro/tonne) - 2020 EiF	126	129	132
Tonnes replaced - 2022 EiF	159 578	133 936	111 717
Total restriction costs - 2022 EiF (annual, million euro)	18.6	15.9	13.6
Cost-effectiveness (euro/tonne) - 2022 EiF	116	119	122

Notes:
Baseline (Main) with 2020 entry into force (EiF) is the scenario used to develop the main analysis of the impacts in Annex D and the main report. Scenarios 1 and 3 vary the phase out assumptions as described in Table E1.

2022 EiF denotes how the costs or tonnages phased out will change if the restriction enters into force in 2022. All other assumptions are as defined in the respective scenario.

E.2. Impact of pricing of alternatives on cost-effectiveness

As shown in Annex D, the described scenario used to estimate the substitution costs is believed to adequately illustrate the anticipated restriction costs as confidential information suggests that the material costs are much lower and closer to those estimated in the Low cost scenario. Therefore, as explained in Annex D, the estimated substitution costs (under the Scenario 2: Baseline (Main) assumptions) address any uncertainties related to the long term price differential between the four phthalates and their least cost alternatives, and in particular for markets such as Asia where the price differential may be larger in the medium term. It also captures any uncertainties related to testing costs, research & development, reformulation and plant and process modifications (RDRPPM) to be incurred by industry as a result of the entry into force of the proposed restriction. Therefore, there is a relatively low level of uncertainty associated with this scenario.

In brief, in the Scenario 2: Baseline (Main) it was assumed that:

- The costs of substituting DEHP with the main alternatives on the European market are anticipated to be driven in the long run primarily by their comparative to DEHP efficiency, as currently, their market prices are similar (or even lower).
- The costs of transitioning to the main alternatives of DEHP on international markets is anticipated to be driven in the long run by their comparative to DEHP efficiency as well as their prices, which are assumed to be about 5% higher than DEHP (taking into account historical price reports on non-EU markets, see Table D8 in Annex D).
- Substituting DBP, DIBP and BBP is assumed to result in a 10% increase in total costs. This is in line with the mid-point estimate in the abatement cost study recently carried out. (ECHA 2013)

In this Annex, we test the impact of these assumptions on the cost-effectiveness with the following two scenarios described in Table E7.

Table E7 Material cost scenarios – description of pricing assumptions

Alternative to:	Low cost scenario	Main scenario	High cost scenario
DEHP	The prices of alternatives are the same as DEHP's for EU manufactured and imported articles	The prices of alternatives are: - the same as DEHP's for EU manufactured articles - 5% higher than DEHP's for imported articles	The prices of alternatives are 5% higher than DEHP's EU manufactured and imported articles
DBP/DIBP/BBP*	5% higher	10% higher	15% higher

Notes: * The assumptions represent the high- and the low-point estimate in the abatement cost study recently carried out. (ECHA 2013)

The Low costs scenario is considered likely as shown in Table D8 in the Annex D, the prices of the many alternatives are similar to DEHP's price even in markets such as Asia where DEHP currently is dominant. Thus, prices on all markets are anticipated to be driven in the long run primarily by their comparative to DEHP efficiency. This is a fair assumption, as in fact in the long-run, it can even be expected that the prices of less efficient plasticisers would decline to

remain competitive on the market (whereby the substitution costs would also begin to approach zero).[210] Also, as DBP/DIBP and BBP have been fully substituted in the EU28 in all applications in scope of this restriction proposal, the cost differential for their alternatives is likely also approaching zero. Therefore, the Low cost scenario also provides some buffer for minor costs such as RDRPPM and testing costs which might occur in the short run on markets where DEHP is currently dominant. This conclusion is supported by confidential information.

The High costs scenario is considered highly unlikely. As shown in Table D8 in the Annex D, the prices of many alternatives are similar to DEHP's even on markets such as Asia where DEHP currently is dominant. Furthermore, DBP, DIBP and BBP are fully phased out in the EU (no applications for authorisation) which suggests that their substitution costs are rather low. The High cost scenario is presented here for the sole purpose of demonstrating that the proposed restriction would remain cost-effective even if the four phthalates are replaced by higher cost alternatives for example due to the public preference for non-phthalate plasticisers in some niche, specialised applications. This gives an indication of the costs of the combined factors (restriction and public awareness) which are relevant for industry.

Table E8 Substitution (material) cost scenarios – assumptions

Alternative plasticisers	Applications	Comparative loading	Scenarios			
			Low	Baseline (Main)		High
			All	Domestic	Imports	All
DINP	55% of all DEHP uses	1.06	1	1	1.05	1.05
DIDP	15% of all DEHP uses	1.1	1	1	1.05	1.05
DEHT/DPHP/similar	30% of all DEHP uses	1.03	1	1	1.05	1.05
Benzoates/similar	All uses of DBP/DIBP/BBP	1	1.05	1.1		1.15

Table E9 shows that the relaxation of the assumptions on the pricing of the alternatives would lead to the cost-effectiveness nearly doubling under the assumptions of the Low cost scenario; thereby reinforcing the claim that the Scenario 2: Baseline (Main) presented in Annex D is sufficiently conservative and provides sufficient buffer to capture other costs, such as RDRPPM and testing costs.

The impact of the High cost scenario is minor (Table E9). The cost-effectiveness of the restriction options would decline by less than 7.5%. Even if the pricing assumptions in the High cost scenario are applied, the proposed restriction remains almost 16-19 times more cost-effective than the existing restriction on toys and childcare articles (€2 270 -2 630/tonne phthalates avoided), i.e., restriction entries 51 and 52 in Annex XVII of REACH.

Therefore, the impact of the assumptions regarding the pricing of alternatives and total material costs are deemed to be insignificant.

[210] This ignores any potential quality impacts of the end-product due to specific plasticiser characteristics.

Table E9 Impact of material costs on the cost-effectiveness of the restriction

Costs	Substitution cost scenarios		
	Low	Baseline (Main)	High
Tonnes replaced - 2020 EiF	131 562	131 562	131 562
Total restriction costs - 2020 EiF (annual, million euro)	9.6	16.9	18.2
Cost-effectiveness - 2020 EiF (euro/tonne)	73	129	139
Tonnes replaced - 2022 EiF	133 936	133 936	133 936
Total restriction costs - 2022 EiF (annual, million euro)	9.0	15.9	17.2
Cost-effectiveness - 2022 EiF (euro/tonne)	67	119	128

Notes:
Scenario 2: Baseline (Main) with 2020 entry into force (EiF) is used to develop the main analysis of the impacts in Annex D and the main report. Scenarios Low and High vary the pricing assumptions as described in Table E7 and Table E8.
2022 EiF denotes how the costs or tonnages phased out will change if the restriction enters into force in 2022. All other assumptions are as defined in the respective scenario.

E.3. Impact of testing costs on cost-effectiveness

The discussion in Annex D section D.3.1. concluded that although industry would likely continue to conduct testing to ensure compliance in the event the proposed restriction enters into force, these costs, whose magnitude is highly uncertain (due to diverse industry practices), are likely largely not attributable to the proposed restriction (due to existing practices to monitor the presence of phthalates in articles under regulatory obligation or voluntary policies). Any minor uncertainties related to societal costs due to testing as a result of the restriction are already taken into account in the estimation of the substitution costs of imported articles. As stated there, larger price differential was assumed for imported articles to account for such uncertainties.

Table E10 presents testing cost estimates for the purpose of illustrating what impact of these costs could have on the cost-effectiveness of the restriction. This exercise shows that the impact on the price of the article could vary between less than 0.0001% and 0.03% depending on:

- the tendency to choose testing of imported articles as a strategy to ensure compliance for any substance under regulation or company policy (as opposed to other strategies such as contractual obligations and/or information provision),
- the frequency of testing articles, and
- the share of the new stakeholders who choose to test because of the restriction. This last point is trying to isolate the testing done due to the proposed restriction versus any other existing legislation or voluntary company policy.

It is important to note that there are significant uncertainties related to the parameters above. There is a great diversity of practices even among companies who may choose to test. For example, the frequency to test very much depends on how large the shipments, how often and how homogeneous. For example a large buyer (e.g., a wholesaler) that purchases only one homogenous product would likely receive regular large shipments, unless they practice just in time supply where smaller shipments from various locations are received. In general, the

larger and the more homogeneous the shipments are, the fewer items are tested, even if the practice is to test every shipment received. (Further information about the uncertainties related to testing costs is included in Annex D.)

It is also important to note that there are other parameters that significantly influence the results, e.g., the total number of imported articles. EuroStat (as well as Statistics Denmark) data on the number of articles by CN code is not available for the majority of codes included in the scope of the proposed restriction. Therefore, the approach in Table E10 estimates the total number on the basis of EuroStat statistics on import volume and an expert estimate of the average weight of a typical article in scope. This expert opinion is based on the results of the survey of phthalate content by Klif, presented in ECHA 2015b. However, this survey focused on consumer articles and did not take into account articles with higher weight per unit, such as flooring, hoses, pipes, etc. Therefore, the assumed average weight per article might be overstating the number of articles and therefore, testing costs, as the higher the assumed average weight, the lower the testing costs. Only testing parameters are varied in the analysis in Table E10, as the focus of this sensitivity assessment is to show the impact of testing costs on the total restriction costs.

Because of the described uncertainties above, the testing costs presented in Table E10 should not be viewed as ranges of the testing costs that might be expected to be incurred by industry but rather as scenarios that estimate the costs given different set of assumptions regarding possible company practices.

Table E10 Testing cost scenarios

Step	Description of assumption	Testing scenarios			Sources / Assumptions
		Low	Mid-point	High	
a	Total import of articles in tonnes	4 000 000	4 000 000	4 000 000	EuroStat, ECHA projections
b	Assumed average weight of article, kg	0.9	0.9	0.9	ECHA 2015b
c	Total number of articles	4 444 444 000	4 444 444 000	4 444 444 000	=a/b
d	Assumed tendency to test articles (vs other compliance strategy)	2%	11%	20%	ECHA 2015b*
e	Number of articles which could be selected for testing	84 444 436	486 666 618	888 888 800	=c*d
f	Assumed frequency of test, one out of	25000	15 000	5000	ECHA 2015b
g	Number of tested articles	3 378	90 578	177 778	=e/f
h	Price per test, €	80	100	120	ECHA 2015b
i	Total cost of tests, €	270 222	10 801 777	21 333 331	=g*h
j	Average price per articles	4.50	4.50	4.50	EuroStat, value of imported articles (2004-14)/a
k	Destructive test - incl value of articles tested, €	15 200	407 600	800 000	=g*j
l	Total cost of testing,€	285 422	11 209 377	22 133 331	=i+k
m	Testing costs allocated to the proposed restriction	10%	25%	40%	ECHA 2015b**
n	Share of testing costs allocated to the restriction (in 2014 €)	22 557	3 509 737	6 996 917	=l*m
o	Testing costs allocated to each product	0.000	0.001	0.002	=n/c
p	Increase in price due to testing	0.0001%	0.02%	0.03%	=(m+o)/m-1

Notes:

*Less than 20% of respondents replied that they conduct their own testing or require suppliers to provide testing documentation for all their shipments. 60% of those are large companies (>250 employees). Percentages presented as rounded numbers.

**Not all projected costs can be considered to be instigated by the restriction as in the testing survey for example 13 out of 19 respondents answered that they already had some restrictions on the four phthalates beyond the EU and national restrictions, and 10 (out of 19) answered that the listing of the substances on the Candidate List has had impacts on their company's avoidance of the substances in articles and several answered that they request articles without the four phthalates as a consequence of the listing.

Table E10 shows that if the industry has high tendency to test any articles for which there is an EU regulation rather than using other strategies (step d) such as information provision, a lot more new companies will adopt this strategy as a result of the proposed restriction. In this case, the testing costs are about €7 million annually. If the industry tends to prefer other compliance strategies (and conduct only spot testing), then very few companies will pursue testing as a result of the restriction. In this case, the testing costs are about €22 600 annually. When assuming the industry adopts the mid-point estimates, testing costs are about €3.5 million annually.

If testing costs are included in addition to the substitution costs already described to be overestimated, the cost-effectiveness of the proposed restriction declines by about 20% in the Mid-point testing scenario and by about 40% in the High testing scenario (Table E11). Still, it is much more cost-effective than the ex-ante estimates for restriction entries 51 and 52 of Annex XVII of REACH (between €2 270 and €2 630 in 2014 euro, depending on the RMO option), although a direct comparison is inappropriate as the latter analysis did not consider testing costs.

As described in section D.3.1.2. in Annex D, the inclusion of testing costs in the total restriction costs estimated in the main scenario would result in further overestimation of the costs. Therefore, these impacts do not represent realistic estimates of the total compliance costs of the proposed restriction.

Table E11 Estimated testing costs and their impact on the cost-effectiveness of the restriction

Costs (annual, euro)	**Baseline (Main) +Testing cost scenarios**		
	Low	**Mid-point**	**High**
Testing costs (million)	0.02	3.5	7.0
Total restriction costs - 2020 EiF (million)	16.9	20.4	23.9
Total restriction costs - 2022 EiF (million)	15.9	19.2	22.4
Cost-effectiveness - 2020 EiF (euro/tonne)	129	155	182
Cost-effectiveness - 2022 EiF (euro/tonne)	119	143	167

Notes:
The testing costs scenarios are the sum of the estimated testing costs under each of the three scenarios described in Table E10 and the Total restriction costs estimated in the Scenario 2: Baseline (Main) used to develop the main analysis of the impacts in Annex D and the main report. See Table E6 in this annex for the Total costs and cost-effectiveness under the Main scenario with 2020 and 20220 entry into force (EiF)).
2022 EiF denotes how the costs will change if the restriction enters into force in 2022. All other assumptions are as defined in the respective scenario.

E.4. Impact of phasing out exports on cost-effectiveness

As stated in the Annex D, the proposed restriction bans the placing on the EU market articles containing the four phthalates and not manufacturing and export of these articles. Therefore, the costs to transitioning to alternatives for the purpose of exports are not included in the restriction compliance costs under the Scenario 2: Baseline (Main) presented in Annex D and the Main report. EU manufacturing of DEHP containing articles[211] used indoors or outdoors with prolonged dermal or mucous membrane contact could continue for the purpose of exports, provided these EU manufacturers are within the supply chain of authorisation holders to use DEHP in the PVC formulation and article manufacturing. This section tests this assumption and presents the impacts on cost-effectiveness if industry substitutes DEHP in all exports of articles in scope.

Table E12 shows that the costs to transition to alternatives for exports are minor (approximately €0.3 million annually). They have less than 1.5% impact on the cost-effectiveness of the proposed restriction.

Table E12 Impact of transitioning to alternatives of exported articles on the cost-effectiveness of the restriction

Costs	Baseline (Main) (exports excluded)	Exports included
Tonnes replaced - 2020 EiF	131 562	135 895
Total restriction costs - 2020 EiF (annual, million euro)	16.9	17.2
Cost-effectiveness - 2020 EiF (euro/tonne)	129	127
Tonnes replaced - 2022 EiF	133 936	137 971
Total restriction costs - 2022 EiF (annual, million euro)	15.9	16.3
Cost-effectiveness - 2022 EiF (euro/tonne)	119	118

Notes:
Baseline (Main) scenario does not include substation for the purpose of exports. This scenario with 2020 entry into force (EiF) is used to develop the main analysis of the impacts in Annex D and the main report.
“Exports included” represents the changes in the Main Scenario if it is assumed that EU producers of articles in scope transition to alternatives as a result of the restriction.
2022 EiF denotes how the costs or tonnages phased out will change if the restriction enters into force in 2022. All other assumptions are as defined in the respective scenario.

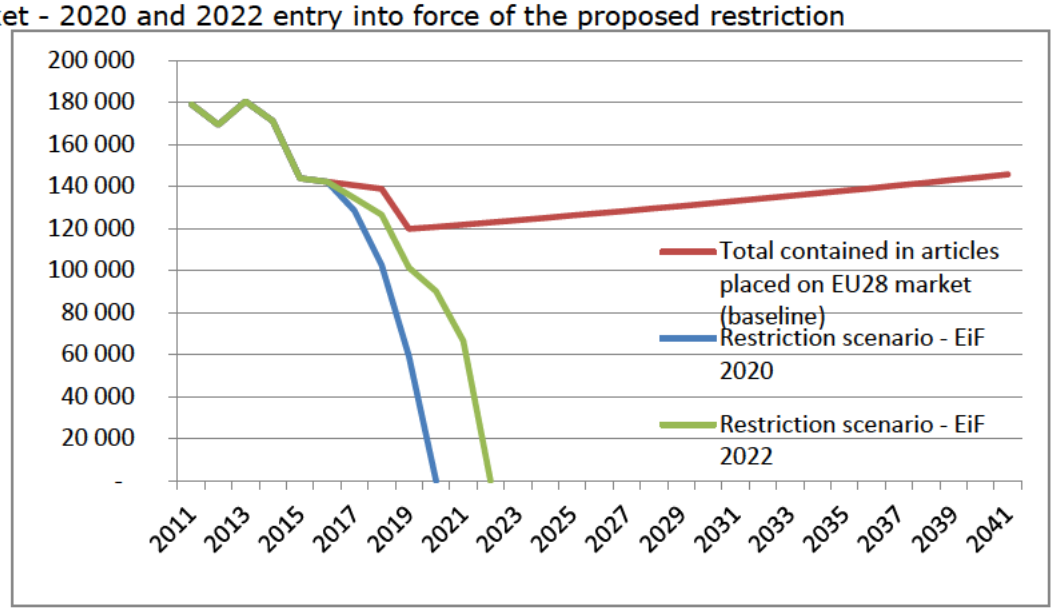
[211] Only exports of DEHP containing articles could be impacted as there are no applications for authorisation for DBP, DIBP and BBP for their use in articles within the scope of this restriction.

E.5. Impact of longer transitional period on risk reduction capacity and cost-effectiveness

Figure E2 shows how the tonnages of the four phthalates in articles placed on the EU28 market would change over time under the baseline and the Scenario 2: Baseline (Main) with 2020 and 2022 as years for its entry into force.

The combined impact of the transitional period and other uncertainties is presented in the remaining sections of this annex.

Figure E2 Historical and projected content of DEHP, DBP, DIBP and BBP in articles placed on the EU market - 2020 and 2022 entry into force of the proposed restriction



A five year transitional period delays the risk reduction capacity of the restriction by two years. This would result in an estimated 124 000 boys at risk due to foetal exposure or 444 000 boys at risk due to exposure in early life. This risk would be avoided with the shorter transitional period of the proposed restriction.

Table E13 shows what intuitively is expected in terms of the impacts of a longer transitional period on cost-effectiveness: the longer the transitional period, the lower is the present value of the restriction costs associated with its coming into force. The cost-effectiveness of the restriction would be higher by 7% if the proposed restriction enters into force in 2022 instead of 2020.

Table E13 Impact of longer transitional period on the cost-effectiveness of the proposed restriction

Summary of Restriction costs	2020 EiF (Baseline (Main))	2022 EiF
	million euro per yr (2014 values)	
Total Restriction costs	16.9	15.9
Tonnes substituted	131 562	133 936
Cost-effectiveness (euro/tonne)	129	119

Notes: 2022 EiF denotes how the costs or tonnages phased out will change if the restriction enters into force in 2022. All other assumptions are as defined in the respective scenario.

Annex D already demonstrates that the market will be able to comply with the restriction within three years of its entry into force. It is anticipated that this will give sufficient time as substantial substitution of these four phthalates in articles has already occurred due to ongoing regulatory action (e.g., classification and labelling, authorisations, RoHS, food contact material legislation, etc.) and as technically feasible alternatives with lower risk profile exist at similar price levels. Actors in the supply chain, including those located outside the EU, are familiar with the ongoing action to address risks from the four phthalates in the EU and other international jurisdictions. There is an awareness of which articles may contain these phthalates, what the best alternatives are for specific applications, as well as how to transition to these alternatives. Furthermore, as demonstrated in section D.3.7 in Annex D, the proposed restriction with entry into force in 2020 would not exert unsurmountable costs to the supply chains required to comply with it.

The main advantage of a longer transitional period is to give recyclers more time to comply with the proposed restriction.[212] However, as discussed in section D.3.1.3 in Annex D, more than 90% of uses of recyclate are for articles outside the scope of the proposed restriction and the average concentration of DEHP in incoming PVC waste is projected to decline substantially past 2020.

E.6. Impact on cost-effectiveness of the combined uncertainties related to the assumptions in the analysis

As the assumptions may partially compensate for each other's impact on the total restriction costs, it is not reasonable to assume that the worst and best case scenario are a result of the additions of the estimates presented in the tables above (i.e., Table E6, Table E9, Table E11,

Table E12, Table E13). After running all possible combinations of assumptions, it was determined that the cost-effectiveness of the restriction is the lowest when the baseline assumptions of the Low tonnages (Scenario 3 in Table E1), High Material costs (Scenario 3 in Table E8) and the High testing costs (Scenario 3 in Table E10) scenarios are combined. As shown in Table E14, in this case, the cost-effectiveness is lower by about 60% in comparison to the main scenario used in the main report and the remaining annexes (i.e., Scenario 2: Main baseline scenario/Scenario 2: Baseline (Main) and no testing costs whose assumptions are presented in Table E1 and its costs and tonnages phased out are reported in Table E6). Still, it is more cost-effective than the ex-ante estimates for restriction entries 51 and 52 of Annex XVII of REACH (between €2 270 and €2 630 in 2014 euro, depending on the RMO option), although a direct comparison is inappropriate as the latter analysis did not consider testing costs.

[212] The application for authorisation of recyclers is with pending decision at the time of the writing of this dossier. Its scope includes the formulation of recycled soft PVC containing DEHP in compounds and dry-blends and use of recycled soft PVC containing DEHP in polymer processing by calendering, extrusion, compression and injection moulding to produce PVC articles

Table E14 Combined impact on the cost-effectiveness: scenarios with largest impact

Costs	Low substitution costs & Baseline (Main)	Baseline (Main)	High testing costs & Baseline Scenario 3: Low Tonnages & High material costs
Tonnes replaced - 2020 EiF	131 562	131 562	111 717
Total restriction costs - 2020 EiF (annual, million euro)	9.6	16.8	22.8
Cost-effectiveness - 2020 EiF (euro/tonne)	73	129	204
Tonnes replaced - 2022 EiF	133 936	133 936	111 717
Total restriction costs - 2022 EiF (annual, million euro)	9.0	15.9	21.1
Cost-effectiveness - 2022 EiF (euro/tonne)	67	119	189

Notes:
"Scenario 2: Baseline (Main)" denotes the scenario used in the main report and the remaining annexes (i.e., Scenario 2: Main baseline scenario and no testing costs whose assumptions are presented in Table E1 and its costs and tonnages phased out are reported in Table E6).
"High testing costs & Baseline Scenario 3: Low tonnages" the cost-effectiveness of the restriction is the lowest when the baseline assumptions of the Low tonnages (Scenario 3 in Table E1) and the High testing costs (Scenario 3 in Table E10) scenarios are combined.
"Low substitution costs & Baseline Scenario 2: Main" is the Substitution cost scenario: Low presented in Table E9. It shows how the costs change if the we assume no price differential of alternatives for DEHP and 5% increase in costs for the other three phthalates. The remaining assumptions are as presented in the Baseline (Main) scenario used in the main report and the remaining annexes.
2022 EiF denotes how the costs or tonnages phased out will change if the restriction enters into force in 2022. All other assumptions are as defined in the respective scenario. Due to the estimation of fixed (e.g., costs of recycling sector, enforcement costs) and variable costs driven by the tonnages replaced, the total costs (and cost-effectiveness) of these complex scenarios cannot be compared to the sum of total costs under each individual scenario, i.e., "High testing costs & Baseline Scenario 3: Low tonnages" is not equal to Total restriction costs of High testing costs scenario + Total restriction costs of Baseline Scenario 2: Low tonnages.

As explained earlier, neither the Baseline Scenario 3: Low tonnages nor the High testing cost scenarios are reasonably foreseeable. This analysis shows that even under these unlikely circumstances the proposed restriction remains proportionate a low cost, efficient risk management measure.

Conversely, the cost-effectiveness of the proposed restriction almost doubles if we assume that there will not be a price difference between DEHP and its alternatives and the substitution of the other three phthalates will lead to 5% increase in the production costs, i.e., Substitution cost scenario: Low in Table E9. Confidential information suggests that this scenario is likely.

E.7. Impact of assumptions of benefit estimates on the conclusions on the effectiveness of the proposed restriction

Impact of assumptions used in quantification of benefits

As explained in Annex D, section 3.5.4, a number of educated assumptions were made to estimate the aetiological fraction, i.e., the fraction of cases that can be attributable to exposure to the four phthalates in articles within the scope of this restriction proposal. The biggest unknown, however, remains the fraction of cases that can be attributed to exposure to chemicals (in this case, those chemicals that have similar endocrine disrupting properties as phthalates).

WHO/UNEP 2012 stated that in general for human diseases and disorders globally, as much as 24% are estimated to be due at least in part to environmental factors. For cryptorchidism, for example, studies have reported that about 4% of cases are of hereditary nature, while for hypospadias this is between 4 and 25% (see Table D26 in Appendix D1 of Annex D). In general, environmental factors could include exposure to chemicals but also, injury, lifestyle (smoking, sun exposure, alcohol consumption, etc.), side-effects of an illness or its treatment, and others. HEAL 2014, Norden 2014, AFA 2013a all assumed a share of the incidence rate attributable to chemicals respectively 2%, 20%,[213] and 50% (for their central scenario).

Rijk et al (2016) also identifies as one of the major challenges to identify a reliable attributable fraction to estimate the fraction of total cost that could be related to endocrine disrupting chemicals (EDC) exposure. The study states that: “To establish an attributable or etiological fraction for a single cause to a disease, is not only a challenge for EDCs and their attribution to health effects, but is a general scientific challenge for all factors influencing development of diseases. Diseases usually have a multifactorial origin, and the exact onset of disease remains unexplained in most of the cases. An estimation of socio-economic cost based on a single factor, whether this is exposure to EDCs or another cause, remains a simplification of reality. For quantification of the socio-economic costs, the EDC-attributable fraction remains a very influential parameter that highly influences the final outcome of a socio-economic cost evaluation. A substantial over- or underestimation of EDC-attributable cost due to a wrong estimate of the attributable fraction is therefore realistic. […] EDC-attributable fractions can be calculated from selected exposure-response relations (ERRs) in epidemiological studies. This approach was followed in the studies of Trasande and co-authors. Unfortunately, not for all of the health effects and suspected compounds is epidemiological evidence available. The estimates are highly dependent on availability of exposure and (human) effect studies, general quality, representativeness to the desired population, and selection (bias) to determine an ERR. […] Furthermore, there is currently no (legislative) framework that requires studies on the mode of action of a chemical when it is brought to the market. At present, the required toxicological data within regulatory frameworks is not sufficient to establish an attributable fraction of a chemical to a disease. […] Therefore, assumptions will have to be made to establish an etiological fraction for the contribution of EDCs to disease. Using a best estimate of a predefined etiological fraction could be criticized as being a “wild guess” but is a

[213] Norden 2014 presents the analysis on the basis of 20% attributable to exposure to chemicals such as phthalates. The percentage was developed on the basis of expert opinion that took into account the probability association of the health outcome with exposure to chemicals such as phthalates.

transparent approach, easy to interpret and modify if needed (e.g. as new insights on EDCs arise providing arguments for adjustments upwards or downwards), aggregates the effects of EDC mixtures, and is widely applicable to all diseases in absent of better information.

Therefore, after reducing the reported incidence rates for hereditary cases,[214] this analysis also applied percentages in the valuation scenarios for cryptorchidism and hypospadias to estimate the fraction of the incidence rate attributable to exposure to chemicals (see step c in Tables D23 and D26 in Appendix D1). The percentages chosen were in line with the three available studies[215] at the time of the dossier preparation: Low estimate of 2% after HEAL 2014, Mid-point of 20% after Norden 2014 and High estimate of 50% after AFA 2013a. The analysis in the main report and Annex D was presented on the basis of the Mid-point values as the attributable fraction used in Norden 2014 was developed using expert panel who looked into the association between the exposure to EDCs (including phthalates) and the health effects comprising the testicular dysgenesis syndrome: infertility, cryptorchidism, and hypospadias which are the primary effects of relevance for phthalates exposure.

For male infertility these percentages are slightly different: respectively 13.5%, 27%, and 40.5% (a composite percentage of steps c and d in Table D15 in section D.3.5.4 in Annex D). Less variability between the scenarios was seen appropriate in this case as EAU 2015 (the main source for incidence data on male infertility) provides very detailed statistics regarding the conditions that may lead to infertility (i.e., infertility due to congenital condition, injury, illness or other conditions). Therefore, a lot of non-environmental, non-chemical related factors were excluded at step c of the analysis. The resulting share of cases attributable to exposure to phthalates (step e of the analysis) of less than 2% is much lower than the effective share used in Hauser et all (2016) of 5% for adult male infertility[216] associated with DBP and BBP.

Table E15: Damage to society from male infertility, cryptorchidism and hypospadias due to exposure to DEHP, DBP, DIBP and BBP in articles in scope: summary, EU28

2014 euro - annual, million	Low estimate	Mid-point estimate	High estimate
Male infertility	4.9	9.8	14.6
Cryptorchidism	1.2	13.9	39.7
Hypospadias	1.0	9.1	22.8
Total	7.1	32.8	77.1

Notes: All values discounted to 2014 with 4% social time preference rate.

All other factors leading to uncertainties or underestimation of benefits described in Annex D remain valid for all three scenarios presented in Table E15.

Despite the fact that the benefits were estimated on the basis of best available information, the scenarios, whose results are presented in Table E15, are associated with high uncertainty. Therefore, the scenarios presented, may not show accurately the upper and lower bound of the value of social damage due to exposure to the four phthalates in articles. They are deemed however adequate to show that the proposed restriction is an efficient regulatory measure as

[214] Step omitted by HEAL 2013, Norden 2014, AFA 2013a.

[215] Rijk et al (2016) was published after the submission of the restriction proposal.

[216] Table D14 evaluates the evidence for association between exposure to the four phthalates during adulthood as moderate. In comparison, the health effect that is monetised for the purpose of showing the proportionality of the proposed restriction is infertility due to in utero and early childhood exposure. Its association is rated as high in table D14.

the prevention of exposure to a small fraction of the population at risk would lead to the benefits of the restriction to exceed the costs. This is also demonstrated in the unlikely worse case situation when the highest costs scenario (High testing costs, High material costs & Baseline scenario for Low tonnages in Table E14) is compared to the lowest benefits valuation scenario (Low estimate in Table E15).

Table E16 shows that if the combinations of cases are avoided due to the entry into force of the proposed restriction, its benefits would exceed the unlikely worst case restriction costs of €22.8 million (Table E14). These cases represent less than 7% of the population at risk of foetal exposure and less than 2% of the population at risk due to infancy and early childhood exposure. Therefore, very few cases demonstrate that the proposed restriction is an efficient regulatory measure even in the worst case scenario.

Table E16 Break-even analysis on the basis of estimated number of cases and social damages of male infertility, cryptorchidism, and hypospadias in EU28

Minimum number of cases for Benefits to ≥ Costs	**3 160 cases of Male infertility & 185 cases of Cryptorchidism & 210 cases of Hypospadias**	**1 050 cases of Male infertility & 405 cases of Cryptorchidism & 455 cases of Hypospadias**
Equivalent to population at risk of foetal exposure (54 000 male children/yr)	<7%	<3.5%
Equivalent to population at risk due to infancy & early childhood exposure (176 000 male children/yr)	<2%	<1.1%
Equivalent to percent of projected annual male births (2.6 million on average/yr)	<0.15%	<0.1%

Notes: The number of cases is estimated on the basis of the weighted average damage to society per case in the Low estimate scenario of benefit estimation (see section 3.5.4 and Appendix D1 in Annex D).

Impact of assumptions used in monetisation of benefits

The analysis presented in the main report and Annex D uses WTP values from ECHA 2015c. Some of the deficiencies of the methodology and results are evaluated in ECHA 2016b but overall, these values tend to be lower than the values used in similar studies. For example, the effective per case value applied in the mid-point (main case) analysis is about €19 000 (from 2050 onward) per case of infertility.[217] (See Table D23). In comparison, AFA 2013a uses ranges that are even higher: from €32 500 to €77 500 per fertility impairment case, combining both assisted reproductive treatment (ART) costs and WTP for having a baby with ART. SEAC reviewed this assessment and proposed a similar range of €58 200 to €67 200 per case. Therefore, we here present the results of the monetisation of benefits using values recommended in ECAH 2016b for sensitivity analysis.

[217] Or €4 350/case discounted to 2014 using 4% discount rate.

Table E17 Willingness to pay values used

Health outcome	Main analysis	Sensitivity analysis
Value of a statistical case of Healthy Child: MINOR birth defects	€4 300	€43 000
Value of a statistical case of Healthy Child: defects in INTERNAL organs	€128 200	€712 000
Value of a statistical case of Healthy Child: defects on EXTERNAL body parts	€25 700	€330 000
Value of statistical infertility (in vitro fertilisation treatment)	€29 400	€55 806

Notes: all values in 2012€
Source: ECHA 2016b

Table E18 below shows the results of the sensitivity analysis of the valuation of benefits. By using the values presented in Table E18 above for sensitivity purposes, the values of the benefits associated with avoided cases of infertility, cryptorchidism and hypospadias is estimated at €200 million annually (discounted to 2014 using 4% discount rate). This is about six times higher than the estimated value of benefits in the main analysis of 32 million annually. (See Table E15)

Table E18 Valuation of benefits – sensitivity analysis

Total costs (representative year)	Low estimate	Mid-point estimate	High estimate
Infertility (2050)	31 268 399	62 007 437	92 746 475
Cryptorchidism (2020)	16 247 755	162 479 258	406 215 193
Hypospadias (2020)	6 837 657	68 376 571	170 941 427
Total	54 353 811	292 863 266	669 903 095

Total costs (2014) - 4% discount rate	Low estimate	Mid-point estimate	High estimate
Infertility	7 619 100	15 109 300	22 599 400
Cryptorchidism	12 840 800	128 409 700	321 037 800
Hypospadias	5 403 900	54 039 000	135 097 500
Total	25 863 800	197 558 000	478 734 700
% difference from main case	367%	602%	621%

Total costs (2014) - 2% discount rate	Low estimate	Mid-point estimate	High estimate
Infertility	15 328 500	30 397 500	45 466 500
Cryptorchidism	14 427 500	144 276 900	360 707 500
Hypospadias	6 071 600	60 716 400	151 791 100
Total	35 827 600	235 390 800	557 965 100

Annex F: Stakeholder consultation

F.1. Call for evidence

A call for evidence was launched on ECHA's website in order to gather information on phthalates in articles. Specifically, information was sought relating to the current uses of the four phthalates in articles on the EU market, the phthalate content and migration rates, as well as other relevant information on exposure and alternatives for the preparation of an Annex XV restriction dossier.

Consultation started on 24 April 2015 and ended on 24 June 2015. In total 17 comments were received. Respondents included Member States, companies, industry or trade associations, and international NGOs. Received comments were taken into account in the development of the report.

More information is available in the background note for the call for evidence:
http://echa.europa.eu/documents/10162/80e55162-c072-4f47-9337-0b0aa6a1170f

F.2. Survey on compliance control costs

In addition to the call for evidence, ECHA appointed Amec Foster Wheeler and COWI A/S to collect up-to-date information on compliance control costs for the four phthalates. A questionnaire was used to collect data on industry practices to ensure compliance with REACH restrictions on substances in articles, and in particular the phthalates.

The questionnaire (see Annex) was developed on the basis of information obtained through interviews with several stakeholders and by reviewing existing assessments of compliance control costs. Interviews were conducted with:

- Trade associations: The European Council for Plasticisers and Intermediates (ECPI), Eurocommerce, Danish Chamber of Commerce, British Retail Consortium and Toy Industries of Europe (TIE).
- Laboratories: Eurofins, Intertek.
- Selected importers: Three major Danish importers, one major UK importer.

The questionnaire was distributed via the following channels:

a) Publication on ECHA's website as part of the call for evidence;
b) A request for distribution to relevant member companies sent to Eurocommerce, Toy Industries of Europe (TIE) and the 35 national member associations of Eurocommerce;
c) A request sent via email to 860 relevant importers, which were identified via the business-to-business database Compass.

The survey was opened from 4 May to 24 June 2015. In total, responses were received from 19 representatives of large, medium and small companies from 8 Member States.

Please see the questionnaire in the Appendix.

F.3. Survey on the possible impacts of potential restriction on soft PVC recycling

The survey was conducted to assess the possible impacts of a potential restriction on soft PVC recycling. It was carried out by ECHA in collaboration with the European Plastics Convertors (EuPC), which acted as contact point with their members.

The survey was opened from 9 December 2015 to 15 January 2016. In total 10 responses were received. The answers received were taken into account in the development of the restriction proposal.

Please see the questionnaire in the Appendix.

F.4. Consultation with international organisations and non EU Countries

To assist with understanding of the situation in non-EU countries, the Commission communicated with TBT (Technical Barriers to Trade) contacts during July 29 to 31 October to both inform about a potential future restriction on the 4 phthalates and to gather information on future trends in their use, additional uses not yet identified, phthalate content in articles, information on phthalate migration, information on the risk of the four phthalates, information on alternatives, and any legislation on these 4 phthalates used in articles.

Information was received from Thailand, South Africa and Japan. Information included standards applicable in Thailand, information on hazards and legislation from Japan, and some trade statistics in South Africa.

F.5. ECHA consultation with EU authorities

To assess the enforceability of the proposal, the draft Annex XVII entry was shared with the Finnish enforcement authority and The Chemical Inspection Service in Denmark.

The advice received was mainly regarding the comprehensibility of the derogations and how to avoid having a really long list of derogations that makes a lot of confusion in the enforcement stage. Instead to use restriction entries 51 and 52 as inspiration.

F.6. Information gathered during the preparation of the Annex XV restriction report on four phthalates by Denmark

During the preparation of the Annex XV restriction report on four phthalates (DEHP, BBP, DBP and DIBP) by Denmark, a number of European trade organisations were contacted (ECHA 2012a). These organisations included both importers and producers and covered a broad range of products.

The purpose of this work was to obtain information about the market for the different uses of the four phthalates in the different product groups. The organisations were asked to assist the DK CA with gathering the required information amongst their members.

Overall, the information gathered related to wallpaper, vinyl flooring and waterbeds. In some cases, further information on overall trends data and indicative data for specific product types were provided. However, some of this information was protected by confidentiality. None of the organisations were able to provide the detailed data types requested in the study questionnaire.

Individual Danish and international companies were also approached for the majority of the application areas investigated. Both producers and importers of articles were contacted.

Appendix F1

Survey of compliance control and costs of possible restriction on four classified phthalates

ECHA and the Danish authorities are currently considering a restriction on the four phthalates DEHP, BBP, DBP and DIBP in articles. The scope of this restriction, if any, has not yet been decided. However, if it is introduced, companies in the EU would need to comply with the restriction.

As part of the decision-making process, this survey aims to better understand how companies would ensure that imported articles comply with a restriction, and what the associated costs will be. The information you provide will help ECHA assess whether a restriction proposal is proportionate and appropriate.

Please note that there will be a later formal public consultation on any restriction proposal. ECHA expect to submit their formal proposal (Annex XV Restriction Dossier) by 8 January 2016, and subsequently it will be possible to submit general comments to the proposal.

This survey has been developed by COWI and Amec Foster Wheeler for the European Chemicals Agency (ECHA) and is executed by ECHA. If you have any questions relating to this survey, please contact Elina Liopa (elina.liopa@echa.europa.eu, +358 9 6861 8777).

You may reply anonymously, but if you give your contact details in the end of the survey, we will send the summary of the survey to you and we can also check any issue with you. Your individual responses will be kept confidential. All results will be presented at the aggregate level only.

Please note that the term "phthalate-containing articles" in this questionnaire designates articles, or parts of articles, that contain any phthalate, unless it is specified that the question concern the four classified phthalates only.

Questions marked with an asterisk are obligatory.

If you would like to print the questionnaire before filling it out, please press here. However, only answers submitted electronically can be used.

Please use the <--Previous button for navigating back and not the back step button in

the browser.

A: Company and product information

1. I respond on behalf of a company, which is:
a) *

☐ Retail company with import of possible phthalate-containing articles

☒ Wholesale company with import of possible phthalate-containing articles

☐ Manufacturer of possible phthalate-containing articles with import of semi-manufacture and final articles from subcontractors

☐ Other: (please specify)

_____

b) *

◯ More than 2,500 employees

◯ 250-2,500 employees

◯ 50-250 employees

◯ 10-50 employees

◯ Less than 10 employees

c) Is located in: *

◯ Austria
◯ Belgium
◯ Bulgaria
◯ Croatia
◯ Cyprus
◯ Czech Republic
◯ Denmark
◯ Estonia
◯ Finland
◯ France
◯ Germany
◯ Greece

◯ Hungary
◯ Iceland
◯ Ireland
◯ Italy
◯ Latvia
◯ Liechtenstein
◯ Lithuania
◯ Luxembourg
◯ Malta
◯ Netherlands
◯ Norway
◯ Poland
◯ Portugal
◯ Romania
◯ Slovakia
◯ Slovenia
◯ Spain
◯ Sweden
◯ United Kingdom

2. Which phthalate containing (or potentially phthalate-containing) products (or product components) does your company import or manufacture?

	Import from countries outside the EU	Import from EU Member States	Manufacture	Some of the products are of own brand
Articles of plasticised PVC:				
Flooring and heavy wall covering	☐	☐	☐	☐
Hoses and tubes of PVC	☐	☐	☐	☐
Bathing equipment: Pools, swimbelts, and similar items	☐	☐	☐	☐
Training balls for physical ex-ercises	☐	☐	☐	☐

Water and air mattresses	☐	☐	☐	☐
Footwear: Sandals and flip/flops, thermo boots, waders, and similar items	☐	☐	☐	☐
Conveyer belts of PVC	☐	☐	☐	☐
Articles containing PVC coatings and PVC films:				
Tablecloths, curtains, tarpaulins, and similar items	☐	☐	☐	☐
Plastic coated wallpaper/tapestry	☐	☐	☐	☐
Furniture with PVC coating or PVC print	☐	☐	☐	☐
Clothing with PVC coating or PVC print	☐	☐	☐	☐
Accessories: Bags, suitcases, belts, and similar items with thin PVC film, PVC coated fabric or PVC marks or print	☐	☐	☐	☐
Office supply: laminated PVC sheets, portfolios, plastic foil, etc.	☐	☐	☐	☐
Packaging materials of PVC: film, foil, bags, etc.	☐	☐	☐	☐
Other articles with PVC parts:				
Electrical and electronic devices and accessories (e.g. in wires, hoses or handles)	☐	☐	☐	☐
Wires and cables with PVC insulation	☐	☐	☐	☐
Bicycles	☐	☐	☐	☐
Automotive PVC components	☐	☐	☐	☐

Carpets (any kind) with PVC back coating	☐	☐	☐	☐
Footwear with PVC soles	☐	☐	☐	☐
Sporting equipment: Diving masks and flippers, googles, golf clubs, and other items with PVC parts	☐	☐	☐	☐
Tools with PVC handles or other parts	☐	☐	☐	☐
Garden furniture e.g. with plastic meshwork	☐	☐	☐	☐
Toys and other childcare articles	☐	☐	☐	☐
Toys for pets	☐	☐	☐	☐

Other articles, please specify:

_____

_____

_____

3. Please indicate what percentage of the total number of articles (products) imported by your company are phthalate-containing articles: *

- ◯ More than 25 %
- ◯ 5-25 %
- ◯ 1- 5 %
- ◯ Less than 1 %
- ◯ I have no idea

4. Does your company have an internal policy to exclude some or all of the four classified phthalates beyond EU and national restrictions?

- ◯ No
- ◯ Yes, the four phthalates must be excluded from some products (apart from the products targeted by EU and national restrictions on the four phthalates)
- ◯ Yes, the four phthalates are excluded from all products

5. Has the inclusion of the four phthalates in the REACH Candidate List had any impacts in your company's avoidance of the four phthalates in articles?

- ◯ No
- ◯ Yes. Please indicate how

_____

5a. Is your internal policy/strategy for phthalates any different from the strategy for other substances (chemicals) listed on the Candidate list or addressed in other EU or national legislation which may be present in your products?

- ◯ No
- ◯ Yes. Please indicate how

_____

5b. Will your current policy/strategy for ensuring compliance change if a new restriction on the phthalate content in articles (of e.g., 0.1% by weight of the plasticised material) comes into force?

- ◯ No
- ◯ Yes. Please indicate how

_____

B: Contractual obligations and information provided

6. Below are examples of strategies for contractual obligations that could be used to ensure that the articles you import comply with the existing EU and national chemicals regulations. The questions concern not only the phthalates

but all restricted substances. Please indicate which strategies your company has used in recent years for those substances that may contain restricted chemicals: *

	We use this strategy for all shipments we receive	We use this strategy for 50% of the shipments we receive	We use this strategy for only a few of the shipments we receive	We haven't used this strategy until now because the articles do not contain restricted chemicals
Contractual obligations:				
Our suppliers are required by contract to supply articles that comply with EU and national regulations on chemicals. These regulations are not explicitly specified in the contract.	☐	☐	☐	☐
We have a list of "Restricted substances" with concentration limits which the suppliers are required by contract to comply with	☐	☐	☐	☐
Our suppliers are required by contract to provide laboratory tests results to confirm meeting their contractual obligations. The following information may be specified in the contract: frequency, laboratory, and test method	☐	☐	☐	☐
Information provision:				

We provide the suppliers with information to make them aware of the existing EU/national restrictions on chemicals	☐	☐	☐	☐
We provide the suppliers with information on restricted substances of particular relevance for the specific articles	☐	☐	☐	☐

We use another strategy:

○ Yes. Please specify:

_____

C: Monitoring of chemicals contained in imported articles

7. Below are examples of monitoring strategies that could be used to ensure that the articles you import comply with the restrictions as defined in the contracts. The questions concern not only the phthalates but all restricted substances. Please indicate which strategies you use or plan to use for articles that potentially may contain restricted substances: *

	We use this strategy for all shipments we receive	We use this strategy only for shipments of a new type of products and/or for shipments from new suppliers	We use this strategy for only a few of the shipments we receive (point control)	We do not use this strategy

Our suppliers are requested to provide a declaration of compliance	○	○	○	○
Our suppliers are requested to provide test documentation from an independent laboratory	○	○	○	○
We monitor compliance by conducting/contracting our own tests on articles	○	○	○	○

8. If you request that the supplier provide documentation from an independent laboratory, which laboratories may be used?

○ Any laboratory with an accreditation for the analysis concerned

○ A laboratory appointed by your company

○ A laboratory from a list of laboratories in the country appointed by your company

○ Other. Please specify:

_____

9. If you request that the suppliers provide documentation from an independent laboratory, are the requirements the same for all suppliers?

○ The requirements are the same for all suppliers.

○ More tests are required on articles from new suppliers (e.g. the first year).

○ We have a ranking system where the suppliers by demonstrating compliance can advance to a higher level where fewer tests are required.

○ We have a risk assessment system where the suppliers are assessed and the requirements are depending on the outcome of the assessment.

10. If you monitor compliance by conducting/contracting your own tests, what would be the most common to use?

○ An independent laboratory outside the EU

○ An independent laboratory within the EU

○ Testing in our own laboratory

11. Does your company have experience with compliance control for the four phthalates? *

◯ Yes

◯ No

11b. Regarding articles that may contain the four phthalates: What is the average frequency of articles tested either by the suppliers or by you (best estimate)? *

◯ More than one article out of 5,000 articles

◯ One article out of 5,000-25,000 articles

◯ Less than one article out of 25,000 articles

◯ I have no idea

11c. Regarding articles that may contain the four phthalates: How many articles are received and tested? *

◯ a) In 2014, how many of these articles did you receive from suppliers? (approximate numbers)

_____

◯ b) How many of these articles did you test at your own or contracted laboratory? (approximate numbers)

_____

◯ c) How many of these articles did you require your suppliers to test at their expense? (approximate numbers)

_____

◯ I have no idea

11d. Did the requirement for testing impact the price of the article?

◯ Yes, by less than 0.1% on average

◯ Yes, by about 0.1-1% on average

◯ Yes, by about 1-10% on average

◯ Yes, by more than 10% on average

◯ No

11e. Are other restricted substances typically tested in the same articles?

◯ Yes

◯ No

D: Time used for compliance control, training and information distribution

12. How much time does your company use annually for compliance control, training and dissemination of information with respect to chemicals (not only phthalates) restrictions and obligations under REACH as concern imported products?

	Changes of contractual documents incl. restriction lists	Compliance control*	Training and dissemination of information within the company (incl. time for keeping updated on new legislation)	Training and dissemination of information to suppliers or other actors within the supply chain
Less than one man-day	◯	◯	◯	◯
One man-day to one man-week	◯	◯	◯	◯
One man-week to one man-month	◯	◯	◯	◯
One to four man-months	◯	◯	◯	◯

Four man-months to one man-year	○	○	○	○
One to four man-years	○	○	○	○
More than four man-years	○	○	○	○

* Compliance control: time used for requesting and controlling compliance documentation

You have answered that your company has some experience with compliance control for the four phthalates - due to either EU or national restriction or your company's own policy.

13. How much time (best estimate) did your company use the first year after a new restriction on the four phthalates came into force?

	Changes of contractual documents	Compliance control	Training and dissemination of information within your company	Training and dissemination of information to suppliers or other actors within the supply chain
Less than 1 man-day	○	○	○	○
1 man-day to 1 man-week	○	○	○	○

1-3 man-weeks	○	○	○	○
More than 3 man-weeks	○	○	○	○

14. How much time (best estimate) did your company use annually for administration of the restrictions on the four phthalates after the first year?

	Changes of contracts and list of restricted chemicals	Compliance control	Training and information dissemination of information within your company	Training and dissemination of information to suppliers or other actors within the supply chain
Less than 1 man-day	○	○	○	○
1 man-days to 1 man-week	○	○	○	○
1-3 man-weeks	○	○	○	○
More than 3 man-weeks	○	○	○	○

D: Time used for compliance control, training and information distribution

12. How much time does your company use annually for compliance control, training and dissemination of information with respect to chemicals restrictions and obligations under REACH as concern imported products?

	Changes of contractual documents incl. restriction lists	Compliance control*	Training and dissemination of information within the company (incl. time for keeping updated on new legislation)	Training and dissemination of information to suppliers or other actors within the supply chain
Less than one man-day	○	○	○	○
One man-day to one man-week	○	○	○	○
One man-week to one man-month	○	○	○	○
One to four man-months	○	○	○	○
Four man-months to one man-year	○	○	○	○
One to four	○	○	○	○

man-years				
More than four man-years	○	○	○	○

* Compliance control: time used for requesting and controlling compliance documentation

You have answered that your company has no experience with compliance control for the four phthalates.

13. Will your current total costs for ensuring compliance (for all substances) change significantly if a new restriction on the phthalate content of 0.1% by weight of the plasticised material in articles comes into force?

○ It may increase by less than 1%

○ It may increase by 1-5%

○ It may increase by 5-25%

○ It may increase by more than 25%

○ I have no idea

14. How much time (best estimate) do you expect your company would need to spend in the first year after a possible restriction on the four phthalates enters into force (i.e. prohibiting the presence of phthalates above 0.1% by weight of the plasticised material in the articles you import)?

	Changes of contractual documents	Compliance control	Training and information dissemination of information within your company	Training and dissemination of information to suppliers or other actors within the supply chain
Less than 1	○	○	○	○

man-day				
1 man-day to 1 man-week	○	○	○	○
1-3 man-weeks	○	○	○	○
More than 3 man-weeks	○	○	○	○

E: Feedback with summary

15. Would you like to receive a summary of the results of the survey (after the responces have been analysed)? *

○ Yes

○ No

Please give your contact details. Contact details will be used only for submitting the survey results to you, the company details will be kept confidential by ECHA. *

Your name

_____

Company

_____

Email

_____

16. Please feel free to give any additional thoughts in the text box below:

_____

_____

_____

The personal information gathered in this form is subject to data privacy laws. ECHA is committed to user privacy. The policy in relation to protection of individuals with regard to the processing of personal data by the Community institutions is based on Regulation (EC) No 45/2001 of the European Parliament and of the Council of 18 December 2000.

100% completed (0 of 14 pages)

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ECHA
EUROPEAN CHEMICALS AGENCY

Survey of the impact of a potential restriction on four classified phthalates in articles

ECHA and the Danish Competent Authority (CA) are currently investigating the need for a restriction on the placing on the market of articles containing the following four phthalates listed on Annex XIV with a sunset date of 21 February 2015:

- **Bis(2-ethylhexyl) phthalate (DEHP)**
- **Benzyl butyl phthalate (BBP)**
- **Dibutyl phthalate (DBP)**
- **Diisobutyl phthalate (DIBP)**

The work is instigated under Article 69(2) of REACH, according to which ECHA shall

after 21 February 2015 consider whether the use of these phthalates in articles poses a risk to human health or the environment that is not adequately controlled. If ECHA considers that the risk is not adequately controlled, it shall prepare an Annex XV dossier taking into account the latest scientific evidence, in order to initiate the restriction procedure. For further details, please see the Registry of Intentions on ECHA's website:http://www.echa.europa.eu/web/guest/registry-of-current-restriction-proposal-intentions

As part of this work, ECHA and the Danish CA are seeking your assistance in understanding the possible impacts of this potential restriction. Your answers will be taken into account in the restriction proposal.

For the purpose of the survey, please assume a hypothetical entry into force of a restriction with the scope and transitional period described in the survey. However, please note that the restriction wording, if any, has not yet been defined. Similarly, if your company is awaiting a decision on authorisation, please assume a hypothetical situation where this decision is granted with the conditions defined in the opinion on your application of ECHA's Committees on Risk Assessment (RAC) and Socio-economic Analysis (SEAC). Please note that this is suggested to facilitate your answers to the survey. At this point ECHA does not have information on when the European Commission will reach a decision on these authorisations and their conditions if granted.

The survey is carried out by ECHA in collaboration with the European Plastics Convertors (EuPC). If you have any questions relating to this survey, please contact Geoffroy Tillieux, Director of the Technical Department of EuPC (geoffroy.tillieux@eupc.org) or Mark Blainey, Restrictions Coordinator, Risk management implementation unit of ECHA (mark.blainey@echa.europa.eu).

You may reply anonymously but if you give your contact details in the end of the survey we could contact you with follow-up questions if such emerge. Your individual responses will be kept confidential. All results will be presented at the aggregate level only. Aggregate results will also be shared with the Danish CA and EuPC.

Questions marked with an asterisk are mandatory.

Please use the <--Previous button for navigating back and not the back step button in the browser.
Thank you for your participation!

1. I respond on behalf of a: *

○ Company producing articles from recycled waste containing DEHP/DBP/DIBP/BBP (integrated recycler)

○ Company producing and selling PVC regrind/pellets from waste containing DEHP/DBP/DIBP/BBP (non-integrated recycler)

○ Company producing (PVC) articles from recycled PVC compounds/pellets (downstream user of recyclate)

1.1. Our company recycles on average *

tonnes/year post-industrial PVC

☐ _____

tonnes/year post-consumer PVC

☐ _____

1.2. Our company recycles on average *

tonnes/year post-industrial PVC

☐ _____

tonnes/year post-consumer PVC

☐ _____

1.3. Our company uses on average *

tonnes/year compound from post-industrial PVC

☐ _____

tonnes/year compound from post-consumer PVC

☐ _____

2. Our current incoming PVC waste streams contain the following concentrations of phthalates. Please provide best estimates of average percentage of the phthalate content in values between 0 and 100% weight by weight of the plasticised material.

[Please include "0" if any of the phthalates are not present in the PVC waste stream.]

a. DEHP*

i. In post-industrial PVC waste streams _____

ii. In post-consumer PVC waste streams _____

b. DBP*

i. In post-industrial PVC waste streams _____

ii. In post-consumer PVC waste streams _____

c. DIBP*

i. In post-industrial PVC waste streams _____

ii. In post-consumer PVC waste streams _____

d. BBP*

i. In post-industrial PVC waste streams _____

ii. In post-consumer PVC waste streams _____

3. The PVC regrind/pellets our company purchases or produces from waste containing DEHP/DBP/DIBP/BBP are used in the production of the following articles: *

◯ Sold in the European Union

◯ For Export

◯ Both

◯ I don't know

3.1. Sold in the European Union: *

1. Articles for industrial use

☐ a. Foils

b. Road equipment – please specify

☐ _____

c. Flooring – please specify

☐ _____

d. Other indoor products than flooring– please specify

☐ _____

e. Outdoor products – please specify

☐ _____

f. Other – please specify

☐ _____

2. Articles for consumer or professional use

g. Flooring – please specify

☐ _____

h. Other indoor products than flooring – please specify

☐ _____

i. Outdoor products – please specify

☐ _____

j. Footwear – please specify

☐ _____

k. Other – please specify

☐ _____

3.2. For Export: *

1. Articles for industrial use

☐ a. Foils

b. Road equipment – please specify

☐ _____

c. Flooring – please specify

☐ _____

d. Other indoor products than flooring– please specify

☐ _____

e. Outdoor products – please specify

☐ _____

f. Other – please specify

☐ _____

2. Articles for consumer or professional use

g. Flooring – please specify

☐ _____

h. Other indoor products than flooring – please specify

☐ _____

i. Outdoor products – please specify

☐ _____

j. Footwear – please specify

☐ _____

k. Other – please specify

☐ _____

3.1. Sold in the European Union: *

1. Articles for industrial use

☐ a. Foils

b. Road equipment – please specify

☐ _____

c. Flooring – please specify

☐ _____

d. Other indoor products than flooring– please specify

☐ _____

e. Outdoor products – please specify

☐ _____

f. Other – please specify

☐ _____

2. Articles for consumer or professional use

g. Flooring – please specify

☐ _____

h. Other indoor products than flooring – please specify

☐ _____

i. Outdoor products – please specify

☐ _____

j. Footwear – please specify

☐ _____

k. Other – please specify

☐ _____

3.2. For Export: *

1. Articles for industrial use

☐ a. Foils

b. Road equipment – please specify

☐ _____

c. Flooring – please specify

☐ _____

d. Other indoor products than flooring– please specify

☐ _____

e. Outdoor products – please specify

☐ _____

f. Other – please specify

☐ _____

2. Articles for consumer or professional use

g. Flooring – please specify

☐ _____

h. Other indoor products than flooring – please specify

☐ _____

i. Outdoor products – please specify

☐ _____

j. Footwear – please specify

☐ _____

k. Other – please specify

☐ _____

4. Will your company strategy change in the event of a restriction on the placing on the market of the articles containing DEHP/DBP/DIBP/BBP in concentration equal or greater than 0.1% of each or in combination by weight of the plasticised material? For the purpose of your response, please assume that the restriction would cover the following articles containing phthalates and would enter into force as of 1 January 2020:

• Articles for indoor use (including storage), including articles in vehicle interiors
• Articles for outdoor use that may lead to prolonged contact with skin or any contact with mucous membranes
• Exempted articles include: *

i. used articles (in circulation prior to entry into force)
ii. articles covered under other EU legislations:

- medical devices covered by Regulation (EC) No 726/2004, Directive 2001/82/EC or Directive 2001/83/EC, or to medical devices covered by Directive 90/385/EEC, Directive 93/42/EEC or Directive 98/79/EC.
- articles covered by Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (RoHS Directive)
- toys and childcare articles covered by restriction entry 51 in Annex XVII of REACH and Directive 2009/48/EC on the Safety of Toys
- food contact meterial articles covered by Regulation (EC) No 1935/2004 and Regulation (EU) No 10/2011 on plastic materials and articles intended to come into contact with food

iii. aerospace and defence articles or equipment
iv. articles strictly used in industrial settings (excluding articles that may lead to prolonged contact with skin or mucous membranes)
v. Flooring strictly used in stables and greenhouses
vi. Measuring devices for laboratory use

Please select from the list of possible strategies (multiple answers possible) to comply with the restriction:

☐ The restriction will not affect our market. Our products fall outside the scope of this potential restriction

☐ We will produce products that fall outside its scope

☐ We will export our products outside the EU

☐ We will refuse some incoming PVC waste streams. Please specify whether you'd refuse post-industrial or post-consumer PVC waste streams containing the above mentioned phthalate plasticisers

_____

☐ We will have to invest in extraction of the phthalates from the incoming PVC waste streams. Please specify:

_____

☐ We will have to increase the frequency (and scope) of our sampling to assess the content of the incoming waste streams

☐ We will relocate outside of the EU

☐ We will undertake other strategies to remain in business. Please specify:

_____

5. Would your strategy change if the restriction enters into force as of 1 January 2022? *

○ No

Yes. Please explain how:

○ _____

6. What would be the impact on your company if the potential restriction enters into force as of 1 January 2020: *

○ Significantly reduced profits (greater than 50% of our current total profits)

○ Moderate reduction of profits (between 10% and 50% of our current total profits)

◯ Low reduction of profits (less than 10% of our current total profits)

◯ No change in profits

◯ Difficult to say

Please explain why:

_____

_____

7. What would be the impact on your company if the potential restriction enters into force as of 1 January 2022? *

◯ Significantly reduced profits (greater than 50% of our current total profits)

◯ Moderate reduction of profits (between 10% and 50% of our current total profits)

◯ Low reduction of profits (less than 10% of our current total profits)

◯ No change in profits

◯ Difficult to say

Please explain why:

_____

_____

8. The potential restriction includes a number of exemptions. Please indicate if an exemption of other types of articles could significantly reduce the impact of the restriction on your business?

Please describe and explain how much:

_____

_____

9. Do you currently test the incoming waste or your output for phthalates in order to inform your customers, ensure a consistent product or to aid your manufacturing process? How much do you spend per year on this testing? *

◯ No

Yes. How much do you spend per year on this testing?

○ _____

10. How do you ensure your PVC regrind/pellets meets your customer specifications for the production of finished articles (for integrated recyclers, the customer requirements refer to the requirements of the article producing units)? *

☐ We purchase waste from the same customer to ensure consistent materials for our process

☐ We conduct test throughout the recycling process and adjust the process to ensure consistent final product

☐ We don't ensure consistent output. Our product depends on the type of waste we receive. This is acceptable for our customers

Other, please specify

☐ _____

11. Could we contact you with further questions? (Please note that your individual answers will be kept confidential and the results of the survey will be presented in aggregate form only.) *

○ Yes

○ No

Please give your contact details: *

Your Name _____

Company / Organization _____

Email _____

12. Do you have any additional comments?

_____

_____

100% completed (0 of 15 pages)

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