(MBzP), mono-cyclohexyl phthalate (McHP), mono-ethyl phthalate (MEP), mono-isononyl phthalate (MINP) and mono-*n*-octyl phthalate (MOP), in 289 US adults (Blount et al., 2000). A subsequent study involving a group of 2541 individuals from participants of the National Health and Nutrition Examination Survey (NHANES) aged ⩾ 6 years in US provided similar findings to the previous study although urinary levels for MEP, MBP and MBzP were lower than the previously reported values (Silva et al., 2004a).

These urinary metabolite levels were used to calculate the ambient exposure levels for five PAEs, BBP, DBP, DEHP, di-*n*-octyl phthalate (DOP) and di-isononyl phthalate (DINP), in human populations (David et al., 2001; Kohn et al., 2000). The estimation of daily intake of phthalates was calculated by applying the following equation according to David et al. (2001):

$$\text{Intake } (\mu g/kg/day) = \frac{\text{UE } (\mu g/g) \times \text{CE } (mg/kg/day)}{f \times 1000\ (mg/g)} \times \frac{MW_d}{MW_m}$$

where UE is the urinary concentration of monoester per gram creatinine, CE is the creatinine excretion rate normalized by body weight, f is the ratio of urinary excretion to total elimination, and MW_d and MW_m are the molecular weights of the diesters and monoesters, respectively.

Table 2 shows the estimated ambient exposure to PAEs. As shown in Table 2, all estimated PAE intakes in the US population were lower than the tolerable daily intake (TDI) values settled by the EU Scientific Committee for Toxicity, Ecotoxicity and the Environment (BBP: 200 μg/kg/day, DBP: 100 μg/kg/day, DEHP: 37 μg/kg/day, DOP: 370 μg/kg/day, and DINP:150 μg/kg/day) (CSTEE, 1998), the reference dose (RfD) of the US EPA (BBP: 200 μg/kg/day, DBP: 100 μg/kg/day, and DEHP: 20 μg/kg/day) (US EPA, 2006) and the TDI values established by the Japanese Government (DEHP: 40–140 μg/kg/day and DINP: 150 μg/kg/day) (MHLW, 2002). Among these PAEs, DEHP is most commonly used plasticizer for flexible PVC formulations and is a widespread environmental contaminant (Kavlock et al., 2002c); however, the estimated daily intake level of DEHP was not high as expected.

Koch et al. (2004a, 2003) and Barr et al. (2003) cast doubt on the sensitivity of the biomarker MEHP for assessing DEHP exposure, and they explored mono- (2-ethyl-5oxo-hexyl) phthalate (5oxo-MEHP) and mono- (2-ethyl-5-hydroxyhexyl) phthalate (5OH-MEHP) as additional biomarkers for DEHP. After a single oral dose of DEHP in a male volunteer, peak concentrations of MEHP, 5OH-MEHP, and 5oxo-MEHP were found in the serum after 2 h, and in urine after 2 h (MEHP) and 4 h (5OH-MEHP and 5oxo-MEHP). The major metabolite was MEHP in serum and 5OH-MEHP in urine (Koch et al., 2004a). Barr et al. (2003) analyzed 62 urine samples for metabolites of DEHP, and the mean urinary levels of 5oxo-MEHP and 5OH-MEHP were 4-fold higher than MEHP.

Koch et al. (2003) determined a median DEHP intake of 13.8 μg/kg/day based on urinary oxidative metabolites of DEHP, 5OH-MEHP and 5oxo-MEHP, in male and female Germans ($n = 85$; aged 18–40). Twelve percent of the subjects exceeded the TDI of the EU-CSTEE (37 μg/kg/day) and 31% of the subjects exceeded the RfD of the US EPA (20 μg/kg/day). For DBP, BBP, DEP, and DOP, the 95th percentile intake values were estimated to be 16.2, 2.5, 22.1, and 0.42 μg/kg/day, respectively. Subsequently, urine samples from 254 German children aged 3–14 were also analyzed for concentrations of these three metabolites of DEHP. The geometric means for MEHP, 5OH-MEHP and 5oxo-MEHP in urine were 7.9, 52.1, and 39.9 μg/L, respectively (Becker et al., 2004). The median daily intake of DEHP in children was estimated to be 7.7 μg/kg. Four children exceeded the TDI of the EU-CSTEE (37 μg/kg/day) and 26 children also exceeded the RfD of the US EPA (20 μg/kg/day) (Koch et al., 2006).

Although these findings showed that German populations could be exposed to DEHP at a higher level than previously estimated values (David et al., 2001; Kohn et al., 2000), these results should be interpreted carefully. In the above-mentioned equation, Kohn et al. (2000) and David et al. (2001) applied the fractional urinary excretion value ($f = 0.106$: MEHP) determined by Peck and Albro (1982). On the other hand, Koch et al. (2003) applied the fractional urinary excretion values ($f = 0.074$: 5OH-

Table 2
Comparison of calculated intakes of phthalates based on the geometric mean values for urinary metabolites and the tolerable daily intake levels as well as the reference dose of phthalates (in μg/kg/day)

PAEs	Estimated by David et al. (2001) for 289 US individuals (Blount et al., 2000)		Estimated by Kohn et al. (2000) for 2541 US individuals (Silva et al., 2004a)		TDI (EU) (CSTEE, 1998)	RfD (US) (US EPA, 2006)	TDI (Japan) (MHLW, 2002)
	Geometric mean	95th percentile	Geometric mean	95th percentile			
BBP	0.73	3.34	0.88	4.0	200	200	Not established
DBP	1.56	6.87	1.5	7.2	100	100	Not established
DEHP	0.60	3.05	0.71	3.6	37	20	40–140
DOP	<LOD	—	0.0096	0.96	370	Not established	Not established
DINP	0.21	1.08	<LOD	1.7	150	Not established	150

LOD, limit of detection.

MEHP, 0.055: 5oxo-MEHP and 0.024: MEHP) determined by Schmid and Schlatter (1985). Using different fractional urinary excretion values can yield several fold differences in estimated values even if the levels of the urinary metabolites are the same.

Table 3 shows a comparison of the estimated median exposure levels of DEHP. Koo and Lee (2005) and Fujimaki et al. (2006) applied the same fractional urinary excretion values of Koch et al. (2003) for calculating daily DEHP intake. Koo and Lee (2005) estimated daily intake of DEHP in Korean children aged 11–12 years old ($n = 150$) and in Korean women aged 20–73 years old ($n = 150$) with a fractional urinary excretion value of 0.024 for MEHP. Median intake levels of DEHP were estimated to be 6.0 μg/kg/day in children and 21.4 μg/kg/day in adult women. TDI of the EU (37 μg/kg/day) was reached at the 56th percentile for women and the 95th percentile for children. Fujimaki et al. (2006) estimated the daily intake of DEHP in forty pregnant Japanese women. The median concentrations of MEHP, 5OH-MEHP and 5oxo-MEHP in the urine were 9.83, 10.4, and 10.9 μg/L, respectively. The median DEHP intake based on MEHP, 5OH-MEHP, and 5oxo-MEHP were estimated to be 10.4 (3.45–41.6), 4.55 (0.66–17.9), and 3.51 (1.47–8.57) μg/kg/day, respectively. These two studies showed higher exposure levels than the previously estimated values in the US population (David et al., 2001; Kohn et al., 2000). Koo and Lee (2005) also showed that a different estimation model can yield 10-fold lower values when estimating DEHP intake, indicating that methods for estimation of daily intake values of PAEs remain inconsistent.

Recently, other secondary oxidized metabolites of DEHP have been recognized (Koch et al., 2005b). Although 5OH-MEHP and 5oxo-MEHP in the urine reflect short-term exposure levels of DEHP, other secondary oxidized metabolites of DEHP such as mono-(2-ethyl-5-carboxypentyl) phthalate (5cx-MEPP) and mono-[2-(carboxymethyl)hexyl] phthalate (2cx-MMHP) are considered excellent parameters for measurement of the time-weighted body burden of DEHP due to their long half-times of elimination. Biological monitoring in a German population ($n = 19$) indicated that 5cx-MEPP is the major urinary metabolite of DEHP. Median concentrations of the metabolites of DEHP were 85.5 μg/L (5cx-MEPP), 47.5 μg/L (5OH-MEHP), 39.7 μg/L (5oxo-MEHP), 9.8 μg/L (MEHP) and 36.6 μg/L (2cx-MMHP) (Preuss et al., 2005). Furthermore, oxidized metabolites of DINP have been recently introduced as new biomarkers for measurement of DINP exposure (Koch and Angerer, 2007; Silva et al., 2006b). These new findings imply that more accurate methods for estimation of PAE exposure can be developed.

2.2. Exposure in fetuses and infants

PAE exposure to the fetus in utero is a great concern because some PAEs are considered to be developmental toxicants. Adibi et al. (2003) measured of urinary phthalate metabolites in pregnant women ($n = 26$) in New York. The median creatinine-adjusted concentrations of MEP, MBP, MBzP, and MEHP were 236, 42.6, 12.1, and 4.06 μg/g, respectively. Metabolites levels in pregnant women were comparable with those in US general population (Blount et al., 2000; Silva et al., 2004a). Another study in 24 mother–infant pairs confirmed DEHP and/or MEHP exposure during human pregnancies (Latini et al., 2003a). The mean DEHP concentrations in maternal plasma and cord plasma were 1.15 and 2.05 μg/mL, respectively, and the mean MEHP concentration was 0.68 μg/mL in both maternal plasma and cord plasma. The levels of phthalate metabolites in the amniotic fluid may reflect fetal exposure to PAEs. Only three metabolites, MEP, MBP, and/or MEHP, were detected in the amniotic fluid samples ($n = 54$). The levels of mono-methylphthalate (MMP), MBzP, McHP, MINP, MOP, 5OH-MEHP, and 5oxo-MEHP were under the limits of detection. Levels of MEP, MBP, and MEHP ranged from under the limits of detection to 9.0 ng/mL ($n = 13$), 263.9 ng/mL ($n = 50$), and 2.8 ng/mL ($n = 21$), respectively (Silva et al., 2004b). These studies suggest that human exposure to PAEs can begin in utero.

Breast milk and infant formula can be routes of PAE exposure for infants. Table 4 shows phthalate monoesters levels in human milk, infant formula, and consumer milk. Levels of phthalate monoesters in pooled breast milk

Table 3
Comparison of estimated mean daily intake of DEHP (μg/kg/day)

Metabolites	German[a] adults ($n = 85$) (Koch et al., 2003)	Korean[a] (Koo and Lee, 2005)		Japanese[a] pregnant women ($n = 40$) (Fujimaki et al., 2006)	US[b] adults ($n = 289$) (David et al., 2001)	US[b] aged ⩾6 years ($n = 2541$) (Kohn et al., 2000)
		Adults (women) ($n = 150$)	Children ($n = 150$)			
MEHP	10.3 (38.3)	21.4 (158.4)	6.0 (37.2)	10.4	0.60 (3.05)	0.71 (3.6)
5OH-MEHP	13.5 (51.4)	No data	No data	4.55	No data	No data
5oxo-MEHP	14.2 (52.8)	No data	No data	3.51	No data	No data
Oxidative DEHP metabolites[c]	13.8 (52.1)	No data	No data	No data	No data	No data

Figures in parentheses show the 95th percentile.
[a] Applying the equation of David et al. (2001) and the fractional urinary excretion value determined by Schmid and Schlatter (1985).
[b] Applying the equation of David et al. (2001) and the fractional urinary excretion value determined by Peck and Albro (1982).
[c] Average of estimated intakes of DEHP based on 5OH-MEHP and 5oxo-MEHP.

Table 4
Phthalate monoester levels (μg/L) in human milk, infant formula and consumer milk

Monoester	Diester	Three pooled breast milk samples (Calafat et al., 2004b)	Thirty-six samples of Danish mother's milk (Mortensen et al., 2005)	Ten samples of infant formula (Mortensen et al., 2005)	Seven samples of consumer milk (Mortensen et al., 2005)
MMP	DMP	<LOD	0.17 ± 0.26[a]	<LOD	<LOD
MEP	DEP	<LOD	1.78 ± 2.74	<LOD	<LOD
MBP	DBP/ BBP	1.3 ± 1.5[a]	359 ± 1830	0.6–3.9[b]	1.4–2.8[b]
MBzP	BBP	<LOD	1.2 ± 1.6	<LOD	<LOD
MEHP	DEHP	7.8 ± 6.8	13 ± 11	5.6–9.1	7.1–9.9
MINP	DINP	15.9 ± 7.7	114 ± 69	<LOD	<LOD

LOD, limit of detection.
[a] Values are given as mean ± standard deviation.
[b] Values are given as range.

($n = 3$) were reported by Calafat et al. (2004b). A subsequent study for 36 individual human milk samples provided higher values for all metabolites; in particular, levels of MBP were two magnitudes higher (Mortensen et al., 2005) than that in the previous study by Calafat et al. (2004b). Phthalate metabolites in breast milk were detected in their free forms unlike the metabolites found in urine and blood. Therefore, infants may receive active PAE metabolites from breast milk on a daily basis. Only MBP and MEHP were detected in consumer milk and infant formula (Mortensen et al., 2005).

The levels of PAEs were determined for 27 infant formulae sold in several countries, and DEHP and DBP were found (Yano et al., 2005). The amounts of DEHP (34–281 ng/g) were much higher than DBP (15–77 ng/g). DEHP, DBP, and DEP were also found in a total of 86 human milk samples collected from 21 Canadian mothers over a 6-month postpartum period. DEHP was the major ester with a mean value of 222 ng/g (8–2920 ng/g), followed by DBP with a mean of 0.87 ng/g (undetectable to 11.39 ng/g). DEP with a mean of 0.31 ng/g (undetectable to 8.1 ng/g) was detected in only a small number of samples. Dimethyl phthalate (DMP), BBP, and DOP were not detected in any samples (Zhu et al., 2006). Table 5 presents estimated maximum daily intakes of PAEs in infants, which was calculated by assuming that the body weight of infants is 7 kg and the daily intake of milk is 700 mL. Although the total estimated maximum daily intake of DEHP in infants was generally less than in general adults (Koch et al., 2003), the estimated maximum daily intake per body weight was higher than adults due to the low body weight of the infants. Assuming that milk was the only exposure route for PAEs in the infants, it is likely that infants had less exposure to DBP and DEP than the general adult population (Koch et al., 2003). These studies suggest that some infants may also be exposed to DEHP at higher levels than the established safe standard levels.

2.3. Possible variation of PAE exposure

Some humans may be exposed to PAEs at higher level than the established safe standard levels. Measurements of urinary metabolites of PAEs have revealed notable differences in concentrations of specific metabolites based on age, gender and race (Blount et al., 2000; Silva et al., 2004a). Concentrations of MBP, MBzP, and MEHP were higher in the youngest age group (6–11 years) and decreased with age. Non-Hispanic blacks tended to have higher levels of phthalate metabolites than non-Hispanic whites or Mexican Americans. Females tended to have a higher level of phthalate metabolites than did males (Silva et al., 2004a). Blount et al. (2000) also indicated that women of reproductive age (20–40 years) had significantly higher levels of MBP than other age/gender groups. Measurement of the three urinary metabolites MEHP, 5OH-MEHP and 5oxo-MEHP in male and female children ($n = 254$) aged 3 to 14 showed that boys had higher concentrations of these three metabolites of DEHP than girls (Becker et al., 2004). The higher levels of PAE metabolites in the young age group may be due to a different food category, dairy products, or the use of PVC toys (CSTEE, 1998), and the higher levels of MBP in females may be

Table 5
Estimated maximum daily intake (μg/kg/day) of PAEs in infants and general German population

Compounds	Human milk ($n = 21$) (Zhu et al., 2006)	Infant formula ($n = 27$) (Yano et al., 2005)	General population ($n = 85$) (Koch et al., 2003)
DEHP	301 (41.1)	6.9	166 (52.1)
DBP	1.21 (0.12)	1.07	22.6 (16.2)
DEP	0.87	Not measured	69.3 (22.1)

Daily PAE intake levels were calculated by assuming that the average daily milk consumption is 700 mL (722 g; specific gravity of human milk = 1.031) and average body weight is 7 kg.
Figures in parentheses show 95th percentile.

due to use of cosmetic products that contain high levels of DBP (Koo and Lee, 2004).

Koo et al. (2002) approached this issue from a different point of view. Their statistical examination concluded that higher levels of MBP in urine were associated with a lower level of education (only a high school education) and/or lower family income (less than $1500) in the month before sampling. Slightly higher levels of MEP were found in urban populations, low income groups, and males. PAE exposure occurred from food, water, and indoor air, although dietary intake of PAEs from contaminated food was likely to be the largest source (Schettler, 2006). Education level and family income may therefore influence the dietary pattern.

It is still unknown whether the variations in these metabolites represent differences in the actual exposure levels. Metabolism of PAEs may vary by age, race, or sex; for example, the ratios of 5OH-MEHP/5oxo-MEHP and 5oxo-MEHP/MEHP decrease with increasing age (Becker et al., 2004). The mean relative ratios of urinary MEHP to 5OH-MEHP to 5oxo-MEHP were 1 to 7.1 to 4.9 in German male and female children and 1 to 3.4 to 2.1 in German male and female adults. This might indicate enhanced oxidative metabolism in children (Koch et al., 2004b). The ratios for urinary MEHP, 5OH-MEHP and 5oxo-MEHP in Japanese pregnant women were reported to be approximately 1 to 1 to 1 (Fujimaki et al., 2006). The variation seen in these three populations may be due to differences in the analytical methods; however, these variations in human populations are still not negligible for accurate risk assessment. Because the current estimates of PAE intake in humans can be imprecise and ADMEs of PAEs in each subpopulation are not clear, the significance of exposure to PAEs with regard to health effects is yet unknown.

2.4. Exposure from medical devices

DEHP has been used for a wide variety of PVC medical devices such as i.v. storage bags, blood storage bags, tubing sets, and neonatal intensive care units (NICUs), and known treatments that involve high DEHP exposures include blood exchange transfusions, extracorporeal membrane oxygenation and cardiovascular surgery.

Serum concentrations of DEHP were significantly increased in platelet donors and receptors (Buchta et al., 2005, 2003; Koch et al., 2005c). A median increase of 232% of serum DEHP was detected after plateletpheresis in healthy platelet donors (Buchta et al., 2003). Mean DEHP doses for discontinuous-flow platelet donors and continuous-flow platelet donors were 18.1 and 32.3 μg/kg/day on the day of apheresis, which were close to or exceeded health standard levels such as the TDI or RfD (Koch et al., 2005c).

Premature infants who experience medical procedures may have a higher risk of exposure to DEHP than the general population. Because the same size of each medical device is used for all ages, infants may receive a larger dose of PAEs on a mg/kg basis than adults due to their smaller size. Calafat et al. (2004a) provided the first quantitative evidence confirming that infants who undergo intensive therapeutic medical interventions are exposed to higher concentrations of DEHP than the general population. They assessed exposure levels of DEHP in 6 premature newborns (23–26 weeks old) by measuring levels of urinary MEHP, 5OH-MEHP and 5oxo-MEHP. The geometric mean concentrations of MEHP (100 μg/L), 5oxo-MEHP (1617 μg/L), and 5OH-MEHP (2003 μg/L) were found to be one or two orders of magnitude higher than German children aged 3–5 (MEHP: 6.96 μg/L, 5OH-MEHP: 56.7 μg/L and 5oxo-MEHP:42.8 μg/L). Koch et al. (2005a) estimated DEHP exposure due to medical devices by using five major DEHP metabolites. Forty-five premature neonates (2–31 days old) with a gestational age of 25–40 weeks at birth were exposed to DEHP up to 100 times over the RfD value set by the US EPA depending on the intensity of medical care (median: 42 μg/kg/day; 95th percentile: 1780 μg/kg/day).

3. Health effects of PAEs in human populations

In the late 20th century, a few studies reported a relationship between environmental exposure of PAEs and human health. For example, Murature et al. (1987) reported that there was a negative correlation between DBP concentration in the cellular fraction of ejaculates and sperm production. Fredricsson et al. (1993) reported that human sperm motility was affected by DEHP and DBP. In females, decreased rates of pregnancy and higher levels of miscarriage in factory workers were associated with occupational exposure of DBP (Aldyreva et al., 1975). More recent studies in human males, females and infants are summarized below.

3.1. Studies of the male reproductive system

Table 6 shows a summary of studies of the male reproductive system in human populations. Two studies are available for 168 male subjects who were members of subfertile couples (Duty et al., 2003a,b). Eight urinary PAE metabolites, MEP, mono-methyl phthalate (MMP), MEHP, MBP, MBzP, MOP, MINP and McHP, were measured with a single spot urine sample. Urinary MEHP, MOP, MINP, or McHP showed no relevance to sperm parameters or DNA damage (Duty et al., 2003a,b). Urinary MBP was associated with lower sperm concentration and lower motility, and urinary MBzP was associated with lower sperm concentration. There was limited evidence suggesting an association of increased MMP with poor sperm morphology (Duty et al., 2003a). A neutral comet assay revealed that urinary MEP levels were associated with increased DNA damage in sperm (Duty et al., 2003b). This result was confirmed by a recent study in 379 men from an infertility clinic in which sperm DNA damage was associated with MEP (Hauser et al., 2007).

Table 6
Male reproductive effects in human populations

Compounds	Number of subjects	Related effects	Reference
Total PAEs[a]	$n = 21$	↓Sperm normal morphology, ↑Percent of single-stranded DNA in sperm	Rozati et al. (2002)
Phthalic acid	$n = 234$	↑Large testis[c], ↑Sperm motility[c]	Jonsson et al. (2005)
DEHP	$n = 37$	↓Semen volume, ↑Rate of sperm malformation	Zhang et al. (2006)
MEHP	$n = 187$	↓Straight-line velocity and curvilinear velocity of sperm[d]	Duty et al. (2004)
	$n = 74$	↓Plasma free testosterone	Pan et al. (2006)
%MEHP[b]	$n = 379$	↑Sperm DNA damage	Hauser et al. (2007)
MEP	$n = 168$	↑DNA damage in sperm	Duty et al. (2003b)
	$n = 234$	↑Large testis[c], ↓Sperm motility, ↓Luteinizing hormone	Jonsson et al. (2005)
	$n = 379$	↑DNA damage in sperm	Hauser et al. (2007)
	$n = 187$	↓Sperm linearity[d], ↑Straight-line velocity and curvilinear velocity of sperm[c,d]	Duty et al. (2004)
DBP	$n = 37$	↓Semen volume	Zhang et al. (2006)
MBP	$n = 168$	↓Sperm concentration, ↓Sperm motility	Duty et al. (2003a)
	$n = 463$	↓Sperm concentration, ↓Sperm motility	Hauser et al. (2006)
	$n = 187$	↓Straight-line velocity and curvilinear velocity of sperm[d]	Duty et al. (2004)
	$n = 74$	↓Plasma free testosterone	Pan et al. (2006)
	$n = 295$	↑Inhibin B level[c,d]	Duty et al. (2005)
MBzP	$n = 168$	↓Sperm concentration	Duty et al. (2003a)
	$n = 463$	↓Sperm concentration[d]	Hauser et al. (2006)
	$n = 187$	↓Straight-line velocity and curvilinear velocity of sperm[d]	Duty et al. (2004)
	$n = 295$	↓Follicle-stimulating hormone[c]	Duty et al. (2005)
MMP	$n = 168$	↑Poor sperm morphology[d]	Duty et al. (2003a)

[a] Total level of DMP, DEP, DBP, DEHP and DOP.
[b] The urinary concentrations of MEHP divided by sum of MEHP, 5OH-MEHP and 5oxo-MEHP concentrations and multiplied by 100.
[c] Data do not support the association of PAEs with reproductive adverse effects in male human populations.
[d] Only suggestive association was observed (statistically not significant).

In another study, semen volume, sperm concentration, motility, sperm chromatin integrity and biochemical markers of epididymal and prostatic function were analyzed together with MEP, MEHP, MBzP, MBP, and phthalic acid levels in urine in 234 young Swedish men (Jonsson et al., 2005). Urinary MEP level was associated with fewer motile sperm, more immotile sperm, and lower serum luteinizing hormone (LH) values. However, higher phthalic acid levels were associated with more motile sperm and fewer immotile sperm; therefore, the results for phthalic acid were opposite what had been expected.

A similar study was conducted in 463 male partners of subfertile couples (Hauser et al., 2006). Phthalate metabolites were measured in a single spot urine sample. There were dose–response relationships of MBP with low sperm concentration and motility. There was suggestive evidence of an association between the highest MBzP quartile and low sperm concentration. There were no relationships between MEP, MMP, MEHP or oxidative DEHP metabolites with any of the semen parameters.

Although there were associations between some metabolites of PAEs and sperm count, motility, or morphology, no statistically significant associations between MEP, MBzP, MBP, MEHP, or MMP and sperm progression, sperm vigor, or swimming pattern were observed in 187 subjects. There were only suggestive associations as follows: negative associations between MBzP with straight-line velocity (VSL) or curvilinear velocity (VCL), between MBP with VSL and VCL and between MEHP with VSL and VCL. MEP was positively associated with VSL and VCL but negatively associated with linearity (Duty et al., 2004).

Duty et al. (2005) explored the relationship between urinary phthalate monoester concentrations and serum levels of reproductive hormones in 295 men. In their previous studies (Duty et al., 2003a,b), MBP and MBzP were associated with sperm parameters, and the investigators had hypothesized that inhibin B, a sensitive marker of impaired spermatogenesis (Uhler et al., 2003), would be inversely associated with MBP and MBzP. However, MBP exposure was associated with increased inhibin B, although this was of borderline significance. Additionally, MBzP exposure was significantly associated with a decrease in serum follicle-stimulating hormone (FSH) level. The serum FSH level has been used as a marker of spermatogenesis for infertile males in clinical evaluation (Subhan et al., 1995), and it is increased in comparison to normal males (Sina et al., 1975). Therefore, the hormone concentrations did not change in the expected patterns.

DEHP is known to cause adverse effects on the male reproductive system in rodents (Gray et al., 2000), and DNA damage in human lymphocytes is also induced by DEHP and MEHP (Anderson et al., 1999). A Hershberger assay with DEHP or MEHP showed anti-androgenic effects in castrated rats (Stroheker et al., 2005; Lee and Koo, 2007). However, only a few studies have suggested that DEHP could be a reproductive toxicant in humans. Urine and blood samples from 74 male workers at a factory producing unfoamed polyvinyl chloride flooring exposed to DBP and DEHP were compared with samples from 63 unexposed male workers. The exposed workers had significantly elevated concentrations of MBP (644.3 vs. 129.6 μg/g creatinine) and MEHP (565.7 vs. 5.7 μg/g

creatinine) in their urine. The plasma free testosterone level was significantly lower (8.4 vs. 9.7 μg/g creatinine) in the exposed workers than in the unexposed workers. Free testosterone was negatively correlated to MBP and MEHP in the exposed worker group (Pan et al., 2006). Another recent study showed that although the urinary MEHP concentration was not associated with sperm DNA damage, the percentage of DEHP metabolites excreted as MEHP (MEHP%) was associated with increased sperm DNA damage. It is of interest that the oxidative metabolites had inverse relationships with sperm DNA damage (Hauser et al., 2007).

Unlike other studies, the following two studies used diester concentrations for measurement of PAEs. Rozati et al. (2002) reported that the concentration of total PAEs (DMP, DEP, DBP, BBP, DEHP, and DOP) in the seminal plasma was significantly higher in infertile men ($n = 21$) compared to controls ($n = 32$). Correlations were observed between seminal PAEs and sperm normal morphology ($r = -0.769$, $p < .001$), in addition to the % of single-stranded DNA in the sperm ($r = 0.855$, $p < .001$). This study examined only total PAEs, and relationships between individual PAEs and sperm parameters were not identified. Another study in a human male population was carried out by measurement of semen parameters and DEHP, DBP, and DEP in human semen ($n = 37$) (Zhang et al., 2006). The three PAEs were detected in most of the samples, and mean levels of DEHP, DBP, and DEP were 0.28, 0.16, and 0.47 μg/L, respectively. There was a negative correlation between semen volume and concentration of DBP or DEHP. There was also a positive association between the rate of sperm malformation and DEHP concentrations. These diester concentrations may directly reflect PAE exposure levels.

Animal data have suggested that mature exposure to DBP and DEHP affects sperm parameters (Agarwal et al., 1986; Higuchi et al., 2003). Dietary exposure of mature male F344 rats (15–16 weeks old) to DEHP (0–20,000 ppm) for 60 consecutive days resulted in a dose dependent reduction in testis, epididymis and prostate weights at 5000 and 20,000 ppm (284.1 and 1156.4 mg/kg/day). Epididymal sperm density and motility were also reduced and there was an increased occurrence of abnormal sperm at 20,000 ppm (Agarwal et al., 1986). Exposure of BBP from adolescence to adulthood showed changes in reproductive hormones in CD(SD)IGS rats at 100 and 500 mg/kg/day (Nagao et al., 2000). In Dutch-Belted rabbits, exposure of DBP during adolescence and in adulthood decreased the amount of normal sperm whereas in utero exposure of DBP decreased the amount of normal sperm, sperm counts, ejaculated volume, and accessory gland weight (Higuchi et al., 2003). Preadolescent male rats appear to have a greater sensitivity to the adverse testicular effects of DEHP than older rats. Akingbemi et al. (2001) demonstrated that preadolescent male rats (21 days old) were more sensitive than young adult animals (62 days old) to 14- or 28-day DEHP exposures that induced decreases in Leydig cell production of testosterone. PAE effects on male reproductive organs could be influenced by the stage of development, but the data also support the possibility that mature animals are susceptible to PAE exposure. The studies in human populations were in accord with these animal data.

Some studies in human populations have suggested associations between MEP, a metabolite of DEP, and changes in sperm; however, these results regarding to MEP are not supported by animal studies. According to Foster et al. (1980), oral dosing of DEP (1600 mg/kg/day) for 4 days did not damage the testes in young SD rats. In another study, male and female CD-1 mice were given diets with DEP (0–2.5%) for 7 days prior to and during a 98-day cohabitation period. There were no apparent effects on reproductive function in animals exposed to DEP (Lamb et al., 1987).

Furthermore, studies in rodents may have little relevance to humans for the reason that DEHP and DINP do not cause reproductive effects in non-human primates. Pugh et al. (2000) showed no evidence of testicular lesions in young adult cynomolgus monkeys (~2 years old) gavage dosed with 500 mg/kg bw/day DEHP and DINP for 14 days. A study with matured marmosets (12–15 months old) showed that repeated dosing of DEHP at up to 2500 mg/kg bw/day for 13 weeks resulted in no differences in testicular weight, prostate weight, blood testosterone levels, blood estradiol levels or any other aspect of the reproductive system (Kurata et al., 1998). DEHP treatment up to 2500 mg/kg bw/day in marmosets from weaning (3 months old) to sexual maturation (18 months old) produced no evidence of testicular damage. Sperm head counts, zinc levels, glutathione levels and testicular enzyme activities were also not affected (Tomonari et al., 2006). In contrast to data from rabbits and rodents, no testicular effects of DEHP or DINP were found in non-human primates at any ages. The current understanding of how PAEs affects semen parameters, sperm DNA damage, and hormones in human populations is limited and further investigation is required.

3.2. Studies of the female reproductive system

Studies of adult female humans are less numerous than those of adult males. Cobellis et al. (2003) compared plasma concentrations of DEHP and MEHP in endometriotic women ($n = 55$) with control women ($n = 24$), and higher plasma DEHP concentrations were observed in endometriotic women. Similar results were observed in a recent study reported by Reddy et al. (2006). The investigators collected blood samples from 49 infertile women with endometriosis (the study group), 38 infertile women without endometriosis (control group I) and 21 women with proven fertility (control group II). Women with endometriosis showed significantly higher concentrations of DBP, BBP, DOP, and DEHP when compared to both control groups. Upon analysis of cord blood samples of 84 newborns, Latini et al. (2003b) revealed that MEHP-positive

infants had a lower gestational age (38.16 ± 2.34 weeks) than MEHP-negative infants (39.35 ± 1.35 weeks). Intrauterine inflammation due to DEHP and/or MEHP exposure may be a risk factor for prematurity because intrauterine infection/inflammation is a major cause of premature labor. These studies suggest that DEHP may play a role in inducing the intrauterine inflammatory process.

Thelarche, premature breast development, is the growth of mammary tissue in young girls without other manifestations of puberty. Colon et al. (2000) analyzed serum samples from 41 Puerto Rican thelarche patients and 35 age matched controls. Significantly higher levels of DMP, DEP, DBP, DEHP, and MEHP were found in 28 (68%) samples from thelarche patients. This study suggested a possible association between PAEs and premature breast development. However, McKee et al. (2004) stated that the association between PAE exposure and thelarche seems highly unlikely for two reasons. First, the reported exposure levels of PAEs may have reflected contamination since they were very high when compared to recent exposure information. Second, toxicological evidence from the laboratory studies described below do not support any influence on female sexual development.

DEHP exposure at 2000 mg/kg/day for 1–12 days in mature SD rats resulted in decreased serum estradiol levels, prolonged estrous cycles and no ovulation (Davis et al., 1994). A two generation reproductive study in SD rats revealed that oral doses of 500 mg/kg/day BBP caused atrophy of the ovary in one female and significant decreases in absolute and relative ovary weights. However oral doses of up to 500 mg/kg/day BBP did not affect estrous cycles in SD rats (Nagao et al., 2000). Similarly, when DEHP was administered to rats over two generations at up to 9000 ppm (about 900 mg/kg/day) in the diet, there were no effects on the pattern and duration of the estrous cycle in F0 female rats (Schilling et al., 1999). Histological changes in female reproductive organs were also not observed after exposure to di-*n*-propyl phthalate, DBP, di-*n*-pentyl phthalate, DHP, or DEHP (Heindel et al., 1989; Lamb et al., 1987). Although some PAEs have been reported to be weakly estrogenic in estrogen-responsive human breast cancer cells (Jobling et al., 1995; Sonnenschein et al., 1995; Soto et al., 1995; Zacharewski et al., 1998) and/or in a recombinant yeast screen (Coldham et al., 1997; Harris et al., 1997), no PAEs showed any estrogenic response upon in vivo uterotrophic or vaginal cornification assay (Zacharewski et al., 1998). Thus, there is no evidence that PAEs influence the timing of female sexual development in laboratory studies.

3.3. Studies in human infants

Anogenital distance (AGD) is a developmental landmark for the differentiation of the external genitalia and is commonly used as a hormonally sensitive parameter of sex differentiation in rodents. AGD in male rats is normally about twice that in females, and a similar sex difference is observed in humans (Salazar-Martinez et al., 2004). Many studies in male rodents reported a reduction of AGD after prenatal exposure to PAEs (Table 7). Chemicals that adversely affect human sex differentiation (Schardein, 2000) also produce predictable alterations of this process in rodents (Gray et al., 1994). In a Hershberger assay, significant decreases in seminal vesicles, ventral prostate, levator ani/bulbocavernosus muscles weights were observed in animals treated with DEHP, DBP, DINP, di-isodecyl phthalate or MEHP, which suggest that some phthalates possess anti-androgenic activity (Lee and Koo, 2007). Swan et al. (2005) presented the first study of AGD and other genital measurements in relation to PAE exposure in a human population. AGD data were obtained for 134 boys of 2–36 months of age. Mother's urine during pregnancy was assayed for phthalate metabolites. Urinary concentrations of four phthalate metabolites, MEP, MBP, MBzP, and mono-isobutyl phthalate (MiBP), were negatively related to the anogenital index (AGI) which is a weight-normalized index of AGD [AGD/weight (mm/kg)].

In rats, undescended testes were observed in male pups after maternal dosing of BBP, MBzP, DEHP, DBP, or MBP (Table 7). Main et al. (2006) investigated whether phthalate monoesters in human breast milk had any relation to cryptorchidism in newborn boys (1–3 months of

Table 7
Decreased AGD and undescended testes observed in experimental animals

Compounds	Animals	Days of administration	Route	Dose (mg/kg/day)	Decreased male AGD	Undescended testes	Reference
BBP	Wistar rat	GDs 15–17	Gavage	500	+	+	Ema and Miyawaki (2002)
				1000	+	+	
MBzP	Wistar rat	GDs 15–17	Gavage	250	+	+	Ema et al. (2003)
				375	+	+	
DEHP	SD rat	GD 2-PND 21	Gavage	750	+		Moore et al. (2001)
				1500	+	+	
DBP	Wistar rat	GDs 11–21	Diet	555	+	+	Ema et al. (1998)
				661	+	+	
DBP	Wistar rat	GDs 15–17	Gavage	500	+	+	Ema et al. (2000)
				1500	+	+	
MBP	Wistar rat	GDs 15–17	Gavage	250	+	+	Ema and Miyawaki (2001)
				500	+	+	

age). The median levels of MMP, MEP, MBP, MBzP, MEHP, and MINP in breast milk were 0.10, 0.95, 9.6, 1.2, 11, and 95 μg/L, respectively. No association was found between phthalate monoester levels and cryptorchidism. However, there were positive correlations for MEP and MBP with sex hormone-binding globulin, MMP, MEP, and MBP with the ratio of LH/free testosterone, and MINP with LH. MBP was negatively correlated with free testosterone. These mother–son cohort studies provided evidence that testicular and genital development may also be vulnerable to perinatal exposure to PAEs.

Although these two studies of human infants indicate possible associations between PAE exposure and the development of the human reproductive system, two follow-up studies of adolescents exposed to DEHP from medical devices as neonates showed no significant adverse effects on their maturity or sexual activity. A comparison of very low birth weight infants who had undergone neonatal intensive care and infants with normal birth weights showed that there were no differences in the rates of sexual intercourse, pregnancy, or live births when the infants became young adults (Hack et al., 2002). Another study indicated that adolescents exposed to DEHP as neonates showed no significant adverse effects on physical growth and pubertal maturity. Thirteen male and 6 female subjects of 14–16 years of age who had undergone extracorporeal membrane oxygenation as neonates had a complete physical examination to evaluate the long-term toxicity of DEHP in infants. Thyroid, liver, renal, and male and female gonadal functions tested were within normal ranges for the given age and sex distribution (Rais-Bahrami et al., 2004).

4. Overall conclusions

In conclusion, exposure data in human populations indicate that the current methodology of estimation of PAEs is inconsistent. It is important to obtain improved data on human PAE exposure and a better understanding of the toxicokinetics of PAEs in each subpopulation. Oxidized metabolites of DEHP and DINP were recently recognized as the major urinary metabolites in humans (Barr et al., 2003; Koch and Angerer, 2007; Koch et al., 2004a, 2005b). These findings could be useful to establish new hypotheses for laboratory studies. Hauser et al. (2007) found that oxidative metabolites of DEHP had a negative association with sperm DNA damage, suggesting that the oxidation of MEHP to 5OH-MEHP and 5oxo-MEHP is protective against sperm DNA damage. However, in an in vitro study, 5OH-MEHP and 5oxo-MEHP, but not DEHP or MEHP, were antiandrogenic (Stroheker et al., 2005). The relevance of this in vitro study to findings in human populations is not clear. Therefore, further studies are required to facilitate accurate risk assessments for human health.

Studies of health effects of PAEs in humans have remained controversial due to limitations of the study designs. Some findings in human populations are consistent with animal data suggesting that PAEs and their metabolites produce toxic effects in the reproductive system. However, it is not yet possible to conclude whether phthalate exposure is harmful for human reproduction. Studies in humans have to be interpreted cautiously because they are conducted in a limited number of subjects. Spot samples only reflect recent phthalate exposure due to the short half-life and it has not yet been confirmed whether point estimates are representative of patterns of long exposure, although reproducibility was found for urinary phthalate monoester levels over two consecutive days (Hoppin et al., 2002). The timing of exposure is a critical factor for decreased AGD in animal studies (Ema and Miyawaki, 2001); however, the stage of fetal development was unknown at the time of urine sampling in the study of Swan et al. (2005). Further studies need to be conducted to confirm these results in human populations and identify the potential mechanisms of interaction.

The studies in human populations reviewed in this paper are useful for showing the strength of associations. Evidence from human studies is preferred for risk assessment as long as it is obtained humanely. It is sometimes claimed that the use of animal data for estimating human risk dose not provide strong scientific support. However, because it is difficult to find alternative methods to test the direct toxic effects of chemicals, continuance of studies in animals is required for risk assessment of chemicals including PAEs.

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ORIGINAL ARTICLE

Evaluation of reproductive and developmental toxicity of the rubber accelerator N,N-dicyclohexyl-2-benzothiazolesulfenamide in rats

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ABSTRACT **Male and female Crl:CD(SD) rats were fed a diet containing the rubber accelerator N,N-dicyclohexyl-2-benzothiazolesulfenamide (DCBS) at 0, 1500, 3000, 6000 or 10 000 p.p.m. (0, 83, 172, 343 or 551 mg/kg bw/day in males and 0, 126, 264, 476 or 707 mg/kg bw/day in females) for a total of 57 days beginning 16 days before mating in males, and a total of 61–65 days from 16 days before mating to day 21 of lactation in females. Body weight gains and food consumption were reduced in males at 6000 p.p.m. and higher and in females at 3000 p.p.m. and higher. The weights of the spleen at 6000 and 10 000 p.p.m. and of the thymus at 10 000 p.p.m. were decreased in females. No changes in estrous cyclicity, copulation index, fertility index, gestation index, delivery index, precoital interval or gestation length were observed at any dose of DCBS. Numbers of implantations at 6000 and 10 000 p.p.m. and pups delivered at 10 000 p.p.m. were reduced. There were no changes in the sex ratio or viability of pups. The body weights of male and female pups were lowered at 6000 p.p.m. and higher. Decreased weight of the spleen in weanlings was also observed in males at 1500 p.p.m. and higher and in females at 3000 p.p.m. and higher. The data indicate that DCBS possesses adverse effects on reproduction and development in rats.**

Key Words: developmental toxicity, N, N-dicyclohexyl-2-benzothiazolesulfenamide, rat, reproductive toxicity, rubber accelerator

INTRODUCTION

Sulfenamide accelerator compounds are widely used in the manufacture of automotive compartments and industrial rubber products such as tires, hoses, conveyer belts, bushings seals, gaskets and windshield wiper blades (EPA 2001). N,N-Dicyclohexyl-2-benzothiazolesulfenamide (DCBS, Fig. 1) is a sulfenamide accelerator. The annual production level of DCBS in Japan was approximately 1000 tons in 1990–1993 and 1900 tons in 2000–2003. Most of this amount was sold and handled domestically (OECD 2007). DCBS is used as an accelerator of vulcanization and is completely reacted in the vulcanizing process (OECD 2007). DCBS is regulated in Germany for use in articles that contact food, but is not regulated by the United States Food and Drug Administration for use in food contact applications (Flexsys 2000). Exposure of workers handling sulfenamide accelerator materials is likely to be highest in the area of materials packaging. During material packout at the manufacturing site, and to a lesser degree during weigh-up activities at the consumer site, there is a possibility of skin and inhalation exposure. Although consumer exposure should be minimal, the most likely route of consumer exposure is skin contact with rubber or latex articles (EPA 2001).

Only up to 6% biodegradation has been determined for DCBS in a ready biodegradability test, and a measured log Kow value of 4.8 suggests that DCBS may have a high bioaccumulation potential (OECD 2007). The possibility of such a chemical compound entering biological systems has aroused great concern regarding its toxicological potential. Generally, biological effects of chemicals should be studied in laboratory animals to investigate their possible influences on human health, and the results of animal tests of chemical toxicity relevant to humans (Clayson & Krewski 1990). However, very little information on the toxicity of DCBS has been published. The toxic effects of DCBS have been briefly summarized by the European Chemical Bureau (2000) and US EPA (2001). It was reported that the oral LD50 values were 1077–10 000 mg/kg bw in rats, the oral NOAEL for 44-day repeated dose toxicity was higher than 100 mg/kg bw/day in rats, and no effects on reproduction were observed at doses up to 400 mg/kg bw/day in rats (EPA 2001). The oral LD50 value was 8500 mg/kg bw in male mice, and repeated daily inhalation exposure of male rats for 15 days at 2 h/day and 350–400 mg/m^3 caused mucous membrane irritation (Vorobera 1969).

The Japanese Government (MHW 1998) conducted toxicity studies for DCBS, including acute toxicity, *in vitro* genotoxicity and repeat dose toxicity combined with reproductive/developmental toxicity as a part of the Safety Examination of Existing Chemical Substances and Chemical Safety Programmes. These toxicity studies are summarized in the IUCLID Data Sets (EPA 2006), OECD Screening Information Data Sets (OECD 2007) and the Hazard Assessment Sheet (CERI 2002). We previously reported the results of a screening test for repeat dose toxicity combined with a reproductive/developmental toxicity in rats, where DCBS at 400 mg/kg bw/day had a deleterious effect on reproduction and development and caused a marked decrease in the number of live pups as well as a total loss of pups by postnatal day (PND) 4 (Ema *et al.* 2007). The primary effects may be on the gestation index for dams and live birth index for pups, both of which appear to be affected at multiple points along the female reproductive process. The viability of neonatal pups may also be affected. To examine the adverse effect of dietary DCBS on survival and growth of pups, a reproductive and developmental toxicity study was performed in rats given DCBS during an extended administration period up to the weaning of pups.

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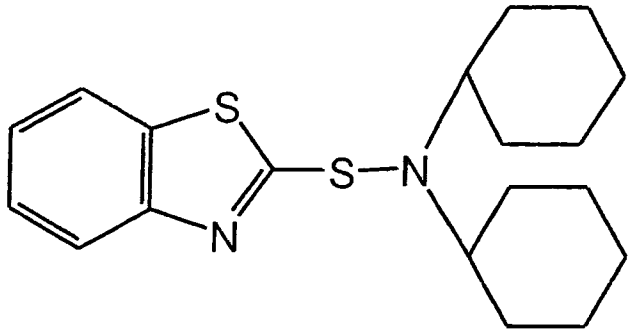

Fig. 1 Structural formula of N,N-dicyclohexyl-2-benzothiazolesulfenamide.

MATERIALS AND METHODS

This study was performed in 2005–2006 at the Safety Research Institute for Chemical Compounds (Sapporo, Japan) in compliance with *Law for the Humane Treatment and Management of Animals* (Law no. 105, October 1, 1973, revised December 22, 1999, Revised Law no. 221; revised June 22, 2005, Revised Law no. 68), *Standards Relating to the Care, Management and Refinement of Laboratory Animals* (Notification no. 88 of the Ministry of the Environment, Japan, April 28, 2006) and *Fundamental Guidelines for Proper Conduct of Animal Experiment and Related Activities in the Testing Facility under the Jurisdiction of the Ministry of Health, Labour and Welfare* (Notification no. 0601005 of the Health Sciences Division, Ministry of Health, Labour and Welfare, Japan, June 1, 2006).

Chemical and dosing

DCBS (CAS no. 4979-32-2) was obtained from Ouchishinko Chemical Industrial (Tokyo, Japan). DCBS in the form of off-white to tan granules is very slightly soluble in water and methanol but soluble in oil. Its melting point is 100–105°C, density is 1230 kg/m^3 and molecular weight is 347 (Flexsys 2000). DCBS (Lot no. 508001) used in this study was 99.7% pure and was kept in a sealed container under cool (1–8°C) and dark conditions. The purity and stability of the chemical were verified by analysis using high-performance liquid chromatography before and after the study. Rats were given dietary DCBS at a concentration of 0 (control), 1500, 3000, 6000 or 10 000 p.p.m. Males were fed a diet containing DCBS for a total of 57 days beginning 16 days before mating. Females were fed a diet containing DCBS for a total of 61–65 days from 16 days before mating to day 21 of lactation throughout the mating, gestation and lactation periods. Control rats were fed a basal diet only.

The dosage levels were determined based on the results of a previous study in rats that were given DCBS by gavage at 0, 6, 25, 100, or 400 mg/kg bw/day for a total of 44 days from 14 days before mating in males and a total of 40–51 days beginning 14 days before mating to day 3 of lactation throughout the mating and gestation periods in females (Ema *et al.* 2007). In that study, toxicologically significant changes were observed only at 400 mg/kg bw/day. Three of 10 females died during parturition. An increased incidence of females showing decreased locomotor activity, soil of the lower abdominal fur and reddish tears was observed. Decreased body weights were found in males and females. Decreased weight of the thymus in both sexes was noted. Decreases in the gestation index, numbers of corpora lutea, implantations, pups born and pups born alive, live birth index and viability index were detected.

Dosed diet preparations were formulated by mixing DCBS into an appropriate amount of a powdered basal diet (CRF-1; Oriental Yeast, Tokyo, Japan) for each dietary concentration. Chemical analysis showed that DCBS in the diet was stable for at least 21 days at room temperature and the formulations were maintained in a room temperature for no more than 21 days. Generally, the diet was replaced once a week.

Animals and housing conditions

Sprague–Dawley (Crl:CD[SD]) rats were used throughout this study. Rats of this strain were chosen because they are the most commonly used in reproductive and developmental toxicity studies and historical control data are available. Male and female rats at nine weeks of age were purchased from the Tsukuba Breeding Center (Charles River Laboratories Japan, Yokohama, Japan). The rats were acclimated to the laboratory for six days prior to the start of the experiment. Male and female rats found to be in good health were selected for use. Rats (F0) were randomly distributed into five groups of six males and six females each, and all animals were assigned a unique number and tattooed on the ear prior to the start of the experiment. Animals were housed individually in suspended aluminum/stainless steel cages except during the acclimation, mating and nursing periods. From day 17 of pregnancy to the day of weaning, individual dams and litters were reared using wood chips as bedding (White Flake; Charles River Laboratories Japan,).

Animals were reared on a basal diet or a diet containing DCBS and filtered tap water *ad libitum* and maintained in an air-conditioned room at 22 ± 3°C with a humidity of 50 ± 20% and a 12-h light (8:00–20:00)/dark (20:00–8:00) cycle. The room was ventilated 10–15 times/h.

Observations

All rats were observed twice a day for clinical signs of toxicity. The body weight was recorded once a week for males and once a week during the premating period, on days 0, 7, 14 and 20 of pregnancy, and on days 0, 4, 7, 14 and 21 of lactation for females. Food consumption was recorded once a week for males, and once a week during the premating period, on days 0, 7, 14 and 20 of pregnancy and on days 0, 7, 14 and 21 of lactation for females.

Rats were euthanized by exsanguination under ether anesthesia. Males were euthanized at 17 weeks and females at 18 weeks on day 21 of lactation. The external surfaces of the rats were examined for abnormalities. The abdomen and thoracic cavities were opened and gross internal examination was performed. In females, the number of implantation sites was recorded. The brain, pituitary, thymus, thyroid, liver, kidney, spleen, adrenal gland, testis, epididymis, seminal vesicle, ventral prostate, ovary and uterus were weighed. The thyroid and seminal vesicle were weighed after fixation with 10% neutral buffered formalin.

Daily vaginal lavage samples from each female were evaluated for estrous cyclicity for two weeks of the premating period. Females with repeated 4–6 day estrous cycles were judged to be normal. Each female rat was mated overnight with a single male rat of the same dosage group until copulation occurred. During the mating period, daily vaginal smears were examined for the presence of sperm. The presence of sperm in the vaginal smear and/or a vaginal plug was considered evidence of successful mating (day 0 of pregnancy). Copulated females were checked for signs of parturition three times a day on days 21–23 of pregnancy.

The females were allowed to deliver spontaneously and nurse their pups until PND 21. The day on which parturition was

completed by 13:00 was designated as PND 0. Total litter size and the numbers of live and dead pups were recorded. Live pups were counted, sexed, examined grossly and individually weighed on PND 0, 4, 7, 14 and 21. On PND 4, litters were randomly adjusted to eight pups comprised of four males and four females. No adjustment was made for litters with fewer than 8 pups. Selected pups were assigned a unique number and tattooed on a limb on PND 4. Unselected pups were necropsied on PND 4. Weanlings were necropsied on PND 21 and the brain, thymus, liver, spleen and uterus were weighed.

Statistical analysis

Statistical analysis of the offspring was carried out using the litter as the experimental unit.

Body weight, body weight gain, food consumption, length of estrous cycle, precoital interval, gestation length, number of implantations and pups delivered, delivery index, organ weight, organ/body weight ratio (relative organ weight) and the viability of pups were analyzed for statistical significance in the following way. Bartlett's test of homogeneity of variance was used to determine if the groups had equivalent variances. If the variances were equivalent, the groups were compared by one-way analysis of variance (ANOVA). If significant differences were found, Dunnett's multiple comparison test was performed. If the groups did not have equivalent variances, the Kruskal–Wallis test was used to assess the overall effects. Whenever significant differences were noted, pairwise comparisons were made by Mann–Whitney *U*-test. The incidence of females with normal estrous cycles, copulation index, fertility index, gestation index and neonatal sex ratio was analyzed by the χ^2 test or Fisher's exact test.

The 0.05 level of probability was used as the criterion for significance.

RESULTS

Clinical observations, body weight and food consumption (F0 males and females)

No deaths were found in F0 males and females. In males, there were no compound related clinical signs of toxicity at any doses. Hematuria and soil of perigenital fur were each observed at 10 000 p.p.m. in one female.

Table 1 shows body weight gain in F0 males and females during dosing. In males, body weight gain on days 0–7 of the dosing period at 6000 p.p.m. and higher was significantly lowered. In females,

Table 1 Body weight gains of F0 parental male and female rats given N,N-dicyclohexyl-2-benzothiazolesulfenamide

Dose (p.p.m.)	0 (Control)	1500	3000	6000	10 000
No. males	6	6	6	6	6
Initial body weight (g)†	367 ± 7	367 ± 6	366 ± 7	366 ± 8	366 ± 7
Body weight gain during dosing period (g)†					
Days 0–7	48.0 ± 10.4	36.8 ± 14.5	36.3 ± 4.8	26.7 ± 8.5**	25.2 ± 6.5**
Days 7–14	38.2 ± 9.2	33.7 ± 13.4	34.5 ± 8.7	35.2 ± 5.6	29.5 ± 5.5
Days 14–21	21.7 ± 9.2	27.3 ± 7.1	24.3 ± 5.2	23.0 ± 12.6	21.8 ± 3.1
Days 21–28	26.8 ± 10.2	25.7 ± 8.3	22.3 ± 10.5	23.5 ± 6.7	25.2 ± 5.0
Days 28–35	21.2 ± 7.7	20.8 ± 6.5	28.5 ± 12.6	24.0 ± 3.7	19.2 ± 4.1
Days 35–42	14.8 ± 6.9	15.3 ± 6.5	20.3 ± 6.9	17.3 ± 6.3	20.5 ± 5.2
Days 42–49	13.8 ± 8.3	19.5 ± 4.2	13.5 ± 2.7	19.8 ± 5.5	17.2 ± 2.9
Days 49–56	14.8 ± 7.6	19.5 ± 6.3	16.7 ± 5.0	20.5 ± 6.2	17.0 ± 3.5
No. females	6	6	6	6	6
Initial body weight (g)†	238 ± 6	239 ± 7	237 ± 5	238 ± 6	237 ± 7
Body weight gain during premating period (g)†					
Days 0–7	6.5 ± 7.7	8.8 ± 8.8	6.8 ± 4.6	−6.5 ± 9.7*	−19.3 ± 9.3**
Days 7–14	15.7 ± 8.5	16.3 ± 6.9	14.2 ± 5.9	12.2 ± 8.0	13.0 ± 8.7
Body weight gain during pregnancy (g)†					
Days 0–7	45.3 ± 6.5	42.5 ± 4.2	32.8 ± 5.4*	31.2 ± 8.6*	19.5 ± 12.0**
Days 7–14	38.3 ± 6.0	35.7 ± 5.4	36.8 ± 7.0	35.2 ± 6.2	31.2 ± 8.6
Days 14–20	76.7 ± 14.6	68.3 ± 4.3	75.8 ± 12.4	68.7 ± 7.7	62.5 ± 12.2
Body weight gain during lactation (g)†					
Days 0–4	28.0 ± 15.7	10.8 ± 24.3	28.0 ± 15.7	8.2 ± 8.7	−2.5 ± 14.6*
Days 4–7	6.5 ± 2.7	12.0 ± 9.0	10.0 ± 10.2	5.3 ± 7.3	0.5 ± 10.9
Days 7–14	1.3 ± 10.7	10.2 ± 7.3	4.2 ± 8.1	14.2 ± 11.1	6.0 ± 12.5‡
Days 14–21	−19.0 ± 14.7	−31.7 ± 9.9	−17.0 ± 8.3	−8.8 ± 9.8	2.6 ± 14.3‡*

*Significantly different from the control, $P < 0.05$; **significantly different from the control, $P < 0.01$.

†Values are given as mean ± SD; ‡data were obtained from five females because one female was excluded (total litter loss on day 9 of lactation).

body weight gains were decreased on days 0–7 of the premating period at 6000 p.p.m. and higher, on days 0–7 of pregnancy at 3000 p.p.m. and higher, and on days 0–4 of lactation at 10 000 p.p.m. Body weight gain on days 14–21 of lactation was significantly increased at 10 000 p.p.m.

In F0 males, food consumption was significantly decreased during the first week at 6000 p.p.m. and higher and during the second week at 10 000 p.p.m. In F0 females, food consumption was significantly decreased throughout the premating, pregnancy and lactation periods at 6000 and 10 000 p.p.m., except on days 7–14 and 14–20 of pregnancy at 6000 p.p.m. A tendency towards decreased food consumption was observed on days 0–7 of pregnancy at 3000 p.p.m.

The mean daily intakes of DCBS were 83, 172, 343 and 551 mg/kg bw in F0 males, and 126, 264, 476 and 707 mg/kg bw in F0 females for 1500, 3000, 6000 and 10 000 p.p.m., respectively.

Estrous cyclicity (F0 females)

All F0 females showed normal estrous cycles in all groups, and the length of the estrous cycles was not significantly different between the control and DCBS-treated groups.

Reproductive and developmental effects (F0 parents/F1 offspring)

The reproductive and developmental parameters for F0 parents/F1 offspring are presented in Table 2. In F0 parent animals in all groups, all pairs copulated, all male and female rats were fertile and all females delivered live pups. All rats of all groups mated within four days. There were no significant differences between control and DCBS-treated groups in copulation index, fertility index, gestation index, precoital interval, gestation length, delivery index, sex ratio of F1 pups, or viability of F1 pups during lactation. Significantly lower numbers of implantations at 6000 and 10 000 p.p.m. and pups delivered at 10 000 p.p.m. were observed. Body weights of male pups were significantly lowered on PND 4, 7 and 21 at 6000 p.p.m. and on PND 7, 14 and 21 at 10 000 p.p.m. In female pups, significantly lower body weights were observed on PND 7, 14 and 21 at 6000 p.p.m. and higher. No malformed pups were detected in any groups.

Necropsy and organ weights (F0 males and females)

Atrophy of the thymus was found in two females at 10 000 p.p.m. No compound-related gross lesions of the reproductive organs were noted in F0 males and females. In males, significantly increased relative weights of the liver and kidney were observed at 10 000 p.p.m.

The organ weights of F0 females are shown in Table 3. The body weight at the scheduled terminal sacrifice was significantly lowered at 6000 and 10 000 p.p.m. The absolute weight of the ovary was significantly lowered at 10 000 p.p.m. Significantly increased relative weights were found for the pituitary at 3000 p.p.m., the liver at 6000 p.p.m., and the brain, kidney and adrenal gland at 10 000 p.p.m. The absolute and relative weights of the thymus at 10 000 p.p.m. and the spleen at 6000 p.p.m. and higher were significantly decreased.

Necropsy and organ weights (F1 weanlings)

No compound related gross lesions were observed in F1 weanlings.

The organ weights of F1 male weanlings are presented in Table 4. The body weight at the scheduled sacrifice was significantly reduced at 6000 and 10 000 p.p.m. The absolute weights of the brain at 6000 and 10 000 p.p.m. and the liver at 10 000 p.p.m. were also significantly reduced. The relative weights of the liver at 1500 and 6000 p.p.m. and of the brain at 10 000 p.p.m. were significantly increased. Significantly decreased absolute and relative weights of the spleen, except for the relative weight at 3000 p.p.m., were noted at 1500 p.p.m. and higher.

The organ weights of F1 female weanlings are presented in Table 5. The body weight at the scheduled sacrifice was significantly reduced at 6000 p.p.m. and higher. Significantly reduced absolute weights of the brain at 6000 and 10 000 p.p.m., the liver at 10 000 p.p.m., and the uterus at 3000 p.p.m. and 10 000 p.p.m. were also observed. The relative weight of the brain was significantly increased at 10 000 p.p.m. The absolute and relative weights of the spleen were significantly reduced at 3000 p.p.m. and higher.

DISCUSSION

This study was designed to assess the effects of DCBS on continuous parameters such as body weight and food consumption, as well as endpoints for reproductive and developmental toxicity.

Significant decreases in body weight gain and food consumption were observed at 6000 p.p.m. and higher in F0 males and females. In females at 3000 p.p.m., body weight gain was significantly decreased during early pregnancy. Food consumption also decreased, but not significantly. The data indicate that changes in body weight gain were associated with changes in food consumption and that DCBS adversely affects body weight gain and food consumption at 6000 p.p.m. in male rats and 3000 p.p.m. in female rats. The higher relative weights of the liver and kidney at the highest dose in F0 males seem to be due to secondary effects of lowered body weight rather than direct effects of DCBS on the organs. More pronounced effects on organ weights were noted in females. Lower absolute and relative weights of the thymus at 10 000 p.p.m. and spleen at 6000 p.p.m. and higher were detected. In our previous study, histopathological examination revealed atrophy of the thymus and spleen at 400 mg/kg bw/day (Ema *et al.* 2007). Other changes in female organ weight such as the relative weights of the brain, pituitary, liver, kidney and adrenal gland, as well as the absolute weight of the ovary are unlikely to be due to the toxic effects of DCBS because the degree of changes was relatively small, no dose-dependency was shown and no changes were noted in absolute or relative weight. These findings suggest that the immune system may be a target of DCBS toxicity, and that female rats have a higher susceptibility to the toxicity of DCBS than male rats. These findings are consistent with our previous study (Ema *et al.* 2007). The higher susceptibility to DCBS toxicity in females may be explained by the stress of pregnancy and lactation. DCBS is likely to be not reproductively toxic in male rats because DCBS caused neither pathological changes in male reproductive organs nor changes in male reproductive parameters.

In our previous study, DCBS given by gavage to rats at 400 mg/kg bw/day from 14 days before mating to day 3 of lactation caused significant decreases in the gestation index, number of corpora lutea, implantations, pups born and pups born alive, live birth index and viability index (Ema *et al.* 2007). This dose also caused severe maternal toxicity and a total loss of pups by PND 4. No maternal or reproductive/developmental toxicity was detected at 100 mg/kg bw/day in our previous study. In the present study, no serious reproductive difficulties were noted even at the highest dose of 10 000 p.p.m., and necropsy of the reproductive organs revealed no evidence of reproductive failure. Although decreased numbers of implantations and pups delivered were noted at the highest dose, the viability of pups until weaning was not significantly decreased. In the present feeding study, the mean daily intakes of DCBS at the

Table 2 Reproductive and developmental findings for F0 parents/F1 offspring of rats given N,N-dicyclohexyl-2-benzothiazolesulfenamide

Dose (p.p.m.)	Control	1500	3000	6000	10 000
No. pairs	6	6	6	6	6
Copulation index[b]					
Male/female (%)	100/100	100/100	100/100	100/100	100/100
Pre-coital interval (days)[a]	2.2 ± 0.8	2.3 ± 1.2	3.2 ± 0.8	3.0 ± 0.9	2.7 ± 1.2
Fertility index[c]					
Male/female (%)	100/100	100/100	100/100	100/100	100/100
Gestation index (%)[d]	100	100	100	100	100
Gestation length (days)[a]	22.2 ± 0.4	22.2 ± 0.4	22.2 ± 0.4	22.0 ± 0.0	22.2 ± 0.4
No. implantations[a]	16.0 ± 1.8	15.0 ± 0.9	16.3 ± 1.2	13.5 ± 2.0*	12.8 ± 1.2**
Delivery index (%)[a,e]	95.8 ± 8.0	96.7 ± 3.7	95.8 ± 5.3	95.6 ± 8.1	86.7 ± 21.1
No. pups delivered[a]	15.3 ± 2.2	14.5 ± 1.0	15.7 ± 1.8	13.0 ± 2.6	11.2 ± 3.1*
Sex ratio of F1 pups[f]	0.467	0.448	0.564	0.526	0.463
Viability index (%)[a]					
PND 0[g]	100 ± 0	100 ± 0	100 ± 0	100 ± 0	91.2 ± 12.9
PND 4[h]	99.1 ± 2.3	97.9 ± 3.3	95.9 ± 5.3	90.6 ± 12.2	72.1 ± 40.8
PND 21[i]	97.9 ± 5.1	97.9 ± 5.1	100.0 ± 0.0	89.6 ± 25.5	83.3 ± 40.8
Male pup body weight during lactation (g)[a]					
PND 0	6.8 ± 0.4	6.7 ± 0.7	6.3 ± 0.4	6.2 ± 0.6	6.5 ± 0.7
PND 4	10.6 ± 0.9	10.3 ± 0.8	9.6 ± 0.6	9.1 ± 0.7**	9.1 ± 2.2[j]
PND 7	18.7 ± 1.3	17.7 ± 1.3	17.6 ± 1.3	14.5 ± 2.2**	13.3 ± 3.7[j]**
PND 14	39.2 ± 3.0	36.2 ± 3.0	37.3 ± 2.9	33.0 ± 4.0	26.3 ± 7.2[j]**
PND 21	67.0 ± 4.6	61.1 ± 6.1	62.8 ± 3.2	55.7 ± 7.6*	44.1 ± 9.9[j]**
Female pup body weight during lactation (g)[a]					
PND 0	6.4 ± 0.4	6.4 ± 0.5	6.0 ± 0.3	5.8 ± 0.6	6.2 ± 0.5
PND 4	10.1 ± 1.1	9.9 ± 0.7	9.0 ± 0.6	8.7 ± 0.7	8.5 ± 1.9
PND 7	18.2 ± 2.0	17.4 ± 0.7	16.0 ± 1.2	13.8 ± 1.3**	11.7 ± 4.2*
PND 14	38.6 ± 3.5	36.1 ± 2.1	35.0 ± 2.4	31.5 ± 4.9*	25.3 ± 7.2[k]**
PND 21	65.1 ± 5.2	60.1 ± 3.7	58.2 ± 3.3	53.5 ± 9.0*	42.5 ± 9.9[k]**

*Significantly different from the control, $P < 0.05$; **significantly different from the control, $P < 0.01$.

[a]Values are given as mean ± SD; [b]copulation index (%) (number of animals with successful copulation/number of animals paired) × 100; [c]fertility index (%) (number of animals that impregnated a female or were pregnant/number of animals with successful copulation) × 100; [d]gestation index (%) (number of females that delivered live pups/number of pregnant females) × 100; [e]delivery index (%) (number of pups delivered/number of implantations) × 100; [f]sex ratio (total number of male pups/total number of pups delivered); [g]viability index on PND 0 (number of live pups on PND 0/number of pups delivered) × 100; [h]viability index on PND 4 (number of live pups on PND 4/number of live pups on PND 0) × 100; [i]viability index on PND 21 (number of live pups on PND 21/number of live pups selected on PND 4) × 100; [j]data were obtained from five litters because one female experienced total male litter loss by day 4 of lactation; and [k]data were obtained from five litters because one female experienced total litter loss by day 9 of lactation.

PND, post natal day.

highest dose were 551 and 707 mg/kg bw in F0 males and females, respectively. One possible explanation for the discrepancy in the degree of reproductive and developmental toxicity between the present and previous studies may be the difference in administration method. Some studies have shown that gavage and feed administration result in different toxicokinetics for various chemicals (Yuan *et al.* 1994, 1995). Further studies are needed to clarify the difference in DCBS toxicokinetics between gavage and feed administrations.

Regarding the development of offspring, decreases in the numbers of implantations and pups delivered and lowered body weights of male and female pups were noted at 6000 p.p.m. and higher. These findings indicate that the dose level of 6000 p.p.m. used in this study was potent enough to adversely affect the survival and growth of pups. Reduced weight of the spleen was also observed in male and female weanlings. These findings also suggest that the immune system may be a target of DCBS toxicity. Other changes in the weights of organs, such as the brain and liver in male weanlings and the brain, liver and uterus in female weanlings are unlikely to be due to the toxic effects of DCBS because the degree of changes was relatively small, no dose-dependency was shown, no changes were noted in the absolute or relative weight, and also

Table 3 Absolute and relative organ weights of F0 female rats given N,N-dicyclohexyl-2-benzothiazolesulfenamide

Dose (p.p.m.)	Control	1500	3000	6000	10 000
No. females	6	6	6	6	5
Body weight (g)†	331 ± 18	316 ± 16	320 ± 11	306 ± 14*	274 ± 20**
Brain (g)†	2.10 ± 0.05‡	2.11 ± 0.08	2.10 ± 0.05	2.06 ± 0.10	2.06 ± 0.03
	0.63 ± 0.03§	0.67 ± 0.04	0.66 ± 0.02	0.67 ± 0.04	0.76 ± 0.05**
Pituitary (mg)†	13.3 ± 1.6‡	13.4 ± 2.4	15.4 ± 0.9	13.9 ± 1.9	12.9 ± 2.6
	4.03 ± 0.44§	4.24 ± 0.65	4.81 ± 0.32*	4.53 ± 0.46	4.70 ± 0.66
Thyroid (mg)†	18.3 ± 3.6‡	17.6 ± 3.5	17.7 ± 4.3	18.8 ± 2.7	17.5 ± 3.6
	5.52 ± 0.87§	5.55 ± 0.99	5.51 ± 1.18	6.15 ± 0.94	6.39 ± 1.02
Thymus (mg)†	255 ± 47‡	205 ± 63	237 ± 45	186 ± 89	116 ± 60**
	77.1 ± 14.4§	65.0 ± 19.6	74.2 ± 13.1	60.1 ± 26.5	41.7 ± 19.9*
Liver (g)†	13.03 ± 0.83‡	12.51 ± 0.71	13.42 ± 1.18	13.69 ± 0.68	12.18 ± 1.60
	3.94 ± 0.21§	3.97 ± 0.23	4.20 ± 0.27	4.48 ± 0.09**	4.46 ± 0.59
Kidney (g)†	2.34 ± 0.16‡	2.38 ± 0.13	2.35 ± 0.10	2.20 ± 0.12	2.51 ± 0.41
	0.71 ± 0.04§	0.75 ± 0.05	0.74 ± 0.04	0.72 ± 0.03	0.92 ± 0.18**
Spleen (mg)†	682 ± 74‡	589 ± 68	600 ± 89	493 ± 24**	459 ± 46**
	206 ± 20§	187 ± 19	188 ± 31	161 ± 5**	168 ± 15**
Adrenal (mg)†	75.5 ± 11.0‡	81.8 ± 12.9	77.0 ± 8.8	72.0 ± 8.8	88.2 ± 8.3
	22.9 ± 3.2§	26.0 ± 3.9	24.1 ± 2.7	23.5 ± 2.8	32.4 ± 3.8**
Ovary (mg)†	109 ± 18‡	113 ± 17	101 ± 5	101 ± 10	75 ± 23**
	32.9 ± 3.8§	36.1 ± 6.8	31.6 ± 2.4	32.9 ± 3.9	27.1 ± 6.4
Uterus (mg)†	513 ± 68‡	465 ± 73	489 ± 101	414 ± 71	369 ± 183
	156 ± 24§	148 ± 26	153 ± 32	135 ± 22	132 ± 56

*Significantly different from the control, $P < 0.05$; **significantly different from the control, $P < 0.01$.
†Values are given as the mean ± S.D; ‡absolute organ weight; §relative organ weight (organ weight [g or mg]/100 g body weight).

Table 4 Absolute and relative organ weights for F1 male weanlings of rats given N,N-dicyclohexyl-2-benzothiazolesulfenamide

Dose (p.p.m.)	Control	1500	3000	6000	10 000
No. males	6	6	6	6	5
Body weight (g)†	67.1 ± 6.7	62.5 ± 4.5	63.8 ± 4.2	55.3 ± 8.9*	43.8 ± 10.6**
Brain (g)†	1.70 ± 0.05‡	1.63 ± 0.12	1.59 ± 0.04	1.51 ± 0.05**	1.45 ± 0.11**
	2.55 ± 0.21§	2.61 ± 0.24	2.50 ± 0.15	2.78 ± 0.42	3.44 ± 0.74*
Thymus (mg)†	257 ± 44‡	219 ± 33	265 ± 45	246 ± 36	190 ± 65
	382 ± 50§	351 ± 57	415 ± 59	449 ± 60	424 ± 50
Liver (g)†	2.56 ± 0.35‡	2.65 ± 0.29	2.69 ± 0.38	2.37 ± 0.38	1.72 ± 0.49**
	3.80 ± 0.17§	4.22 ± 0.20*	4.20 ± 0.37	4.30 ± 0.33*	3.90 ± 0.22
Spleen (mg)†	372 ± 63‡	276 ± 53**	296 ± 32*	250 ± 45**	148 ± 36**
	556 ± 84§	442 ± 80*	466 ± 56	452 ± 32*	337 ± 31**

*Significantly different from the control, $P < 0.05$; **significantly different from the control, $P < 0.01$.
†Values are given as mean ± S.D; ‡absolute organ weight; §relative organ weight (organ weight [g or mg]/100 g body weight).

because the changes seem to be secondary effects of the lowered body weight. In the present study, external and internal morphological examinations of offspring were performed, but no skeletal examinations were conducted. To accurately evaluate prenatal developmental toxicity including teratogenicity, it is necessary to interrupt pregnancy 12–24 h before the expected term either by hysterectomy or the necropsy of maternal animals (Wilson 1965).

The adverse effects of DCBS on reproduction and development noted in the present feeding study are almost consistent with the findings of our previous gavage study (Ema *et al.* 2007), which showed decreased numbers of implantations and pups delivered and decreased body weight of the pups at higher doses. These endpoints appear to be affected at multiple points of the female reproductive and developmental process. The decreased number of implantations

Table 5 Absolute and relative organ weights for F1 female weanlings of rats given N,N-dicyclohexyl-2-benzothiazolesulfenamide

Dose (p.p.m.)	Control	1500	3000	6000	10 000
No. females	6	6	6	6	5
Body weight (g)†	65.7 ± 7.2	61.1 ± 3.4	59.9 ± 4.6	54.0 ± 9.6*	42.8 ± 9.6**
Brain (g)†	1.60 ± 0.09‡	1.56 ± 0.07	1.53 ± 0.03	1.50 ± 0.05*	1.37 ± 0.08**
	2.46 ± 0.25§	2.56 ± 0.16	2.57 ± 0.18	2.84 ± 0.39	3.34 ± 0.78**
Thymus (mg)†	272 ± 46‡	253 ± 33	252 ± 27	243 ± 51	216 ± 82
	415 ± 56§	415 ± 57	422 ± 37	456 ± 101	491 ± 92
Liver (g)†	2.58 ± 0.31‡	2.47 ± 0.27	2.42 ± 0.42	2.27 ± 0.43	1.71 ± 0.49**
	3.93 ± 0.14§	4.03 ± 0.22	4.02 ± 0.41	4.19 ± 0.13	3.96 ± 0.29
Spleen (mg)†	360 ± 57‡	296 ± 16	267 ± 60*	247 ± 50**	163 ± 59**
	548 ± 66§	484 ± 9	442 ± 72*	456 ± 37*	371 ± 58**
Uterus (mg)†	44.7 ± 6.6‡	41.3 ± 6.1	35.7 ± 2.1*	42.0 ± 6.9	32.4 ± 4.8**
	68.9 ± 14.0 Temp.§	67.7 ± 9.8	60.0 ± 7.4	78.5 ± 10.8	77.3 ± 10.3

*Significantly different from the control, $P < 0.05$; ** significantly different from the control. $P < 0.01$.
†Values are given as the mean ± S.D; ‡absolute organ weight; §relative organ weight (organ weight [g or mg]/100 g body weight).

was the most striking effect in the present study. In our previous study, a decreased number of corpora lutea was noted in female rats given DCBS (Ema *et al.* 2007). Therefore, it is likely that the decreased number of implantations can be attributed to the decreased number of corpora lutea. The present study does not provide complete information on all aspects of reproduction and development due to the relatively small numbers of animals in the dose groups and selectivity of the endpoints. To further evaluate the reproductive and developmental toxicity of DCBS in rats, a two-generation reproductive toxicity study should be performed.

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有機スズ化合物の生殖発生毒性

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Reproductive and Developmental Toxicity of Organotin Compounds

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Reproductive and Developmental Toxicity of Organotin Compounds

Makoto Ema

Organotin compounds are chemicals widely used in agriculture and industry. Widespread use of organotins has caused increasing amounts to be released into the environment. Organotins show many aspects of toxicity, such as immunotoxicity, neurotoxicity, and reproductive/developmental toxicity. However, the reproductive and developmental toxicity of organotins is not well understood. The findings of the studies on reproductive and developmental effects of organotin compounds in mammals were summarized in this review.

Keywords: Organotin, reproductive toxicity, developmental toxicity, implantation failure, teratogenicity

1. はじめに

有機スズ化合物は農業や工業の分野で広く使われている[1, 2]．四価のスズ化合物は主に他の有機スズ化合物生産の中間体として使用されている．三価の有機スズ化合物は殺生物作用を有しており，防黴剤，ダニ駆虫剤，ネズミ駆散剤，軟体動物駆除剤等として，また，船底防汚剤として広く用いられている．特に，トリフェニルスズ（TPT）とトリブチルスズ（TBT）は藻類駆除剤，軟体動物駆除剤として，防汚剤製品中によく使われてきた．二価の有機スズ化合物は商業上で最も重要な誘導体であり，主にプラスティック工業の分野でポリマーの劣化を防止するためにポリ塩化ビニル（PVC）プラスティックの熱，光安定剤として使われている．一価の有機スズ化合物はPVCの安定剤として使用されている．有機スズ化合物の生産量をTable 1に示した。

近年の有機スズ化合物の広範な使用により有機スズ化合物による環境汚染の懸念が高まっている．農薬としての使用以外の有機スズ化合物の環境汚染の経路は，PVCプラスティックの安定剤として使われた有機スズ化合物の水中への溶出であり[3]，また，船底防汚剤としての使用が水環境汚染の原因となっている[4]．海棲生物[5-7]や海産物[8-12]からTBTやTPTが検出されており，カキ[13]，泥ガニ[14]，ムールガイ[15]，チヌークサーモン[16]，イルカ，マグロ及びサメ[17]における食物連鎖によるTBTの生物濃

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Table 1. スズ化合物の生産量

物 質 名	CAS	生産量（トン）
Dibutyltin dichloride	683-18-1	10,000 - 15,000
Dibutyltin dilaurate	77-58-7	1000 - 5000
Dibutyltin malcate	78-04-6	500 - 1000
Dibutyltin oxide	818-08-6	1000 - 5000
Dibutyltin bis (2-ethylhexylmercap-acetate)	10584-98-2	7,500 - 12,500
Dibutyltin bis (isooctyl mercap-acetate)	25168-24-5	Not available
Dimethyltin dichloride	753-73-1	1,000 - 5,000
Dimethyltin bis (2-ethylhexyl mercap-acetate)	57583-35-4	5,000 - 10,000
Dimethyltin bis (isooctyl mercap-acetate	26636-01-1	Not available
Dioctyltin dichloride	3542-36-7	5,000 - 10,000
Dioctyltin bis (2-ethylhexyl mercap-acetate)	15571-58-1	7,500 - 12,500
Dioctyltin bis (isooctyl mercap-acetate)	26401-86-5	Not available
Monobutyltin trichloride	1118-46-3	10,000-15,000
Monobutyltin tris (2-ethylhexyl mercap-acetate)	26864-37-9	2,500-7,500
Monobutyltin tris (isooctyl mercap-acetate)	25852-70-4	Not available
Monomethyltin trichloride	993-16-8	1,000 - 5,000
Methyltin Reverse Ester Tallate	201687-57-2	7,500 - 10,000
Monomethyltin tris (2-ethylhexylmercap-acetate)	57583-34-3	5,000 - 10,000
Monomethyltin tris (isooctylmercap-acetate)	54849-38-6	Not available
Mono-octyltin trichloride	3091-25-6	1,000 - 5,000
Mono-octyltin tris (2-ethylhexylmercap-acetate)	27107-89-7	2,500 - 7,500
Mono-octyltin tris (isooctylmercap-acetate)	26401-86-5	Not available
Tributyltin chloride	1461-22-9	2500 - 3000
Tetrabutyltin	1461-25-2	10,000 - 12,500
Tetraoctyltin	3590-84-9	2,500 - 7,500
Tin Tetrachloride	7646-78-8	20,000 - 25,000

出典：ORTEP Association. 2004. Global production data