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Labunska, Iryna; Abdallah, Mohamed Abou Elwafa; Eulaers, Igor; Covaci, Adrian; Tao, Fang; Wang, Mengjiao; Santillo, David; Johnston, Paul; Harrad, Stuart

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HUMAN DIETARY INTAKE OF ORGANOHALOGEN CONTAMINANTS AT E-WASTE RECYCLING SITES IN EASTERN CHINA

Iryna Labunska^{1,2} *, Mohamed Abou-Elwafa Abdallah^{2,3}, Igor Eulaers^{4,5}, Adrian Covaci⁴, Fang
Tao², Mengjiao Wang¹, David Santillo¹, Paul Johnston¹, Stuart Harrad²

*Author for correspondence

email: iryna.labunska@greenpeace.org; Tel: +44 1392 247923

¹Greenpeace Research Laboratories, Innovation Centre Phase 2, Rennes Drive, University of
Exeter, Exeter, EX4 4RN, United Kingdom

²School of Geography, Earth and Environmental Sciences, University of Birmingham,
Birmingham, B15 2TT, United Kingdom

³Department of Analytical Chemistry, Faculty of Pharmacy, Assiut University, 71526 Assiut,
Egypt

⁴Toxicological Center, University of Antwerp, 2610 Wilrijk, Belgium

⁵Ethology group, Department of Biology, University of Antwerp, 2610 Wilrijk, Belgium

Abstract

This study reports concentrations and human dietary intake of hexabromocyclododecanes (HBCDs), polychlorinated biphenyls (PCBs) as well as selected “novel” brominated flame retardants (NBFRs) and organochlorine pesticides, in ten staple food categories. Samples were sourced from areas in Taizhou City, Eastern China, where rudimentary recycling and disposal of e-waste is commonplace, as well as from nearby non-e-waste impacted control areas. In most instances, concentrations in foods from e-waste recycling areas exceeded those from control locations. Concentrations of 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EH-TBB) and bis-(2-ethylhexyl)-3,4,5,6-tetrabromophthalate (BEH-TBP) in samples from e-waste sites were 3.09 – 62.2 ng/g and 0.81 – 16.3 ng/g lipid weight (lw), respectively; exceeding consistently those in foods acquired from control sites by an order of magnitude in many cases. In contrast, while concentrations of HBCD in some foods from e-waste impacted areas exceed those from control locations; concentrations in pork, shrimp, and duck liver are higher in control samples. This highlights the potential significance of non-e-waste sources of HBCD (e.g. building insulation foam) in our study areas. While concentrations of DDT in all foods examined except pork were higher in e-waste impacted samples than controls; our exposure estimates were well below the provisional tolerable daily intake of 0.01 mg/kg bw/day derived by the Joint FAO/WHO Meeting on Pesticide Residues. Concentrations of Σ PCBs resulted in exposures (650 and 2340 ng/kg bw/day for adults and children respectively) that exceed substantially the Minimal Risk Levels (MRLs) for Σ PCBs of 20 ng/kg bw/day derived by the Agency for Toxic Substances & Disease Registry. Moreover, when expressed in terms of dioxin-like toxicity equivalency based on the four dioxin-like PCBs monitored in this study (DL-PCBs) (PCB-105, 118, 156, and 167); concentrations in e-waste impacted foods exceed limits set by the European Union in 6 of the 8

food groups studied and result in dietary exposures for children (10.2 pg TEQ/kg bw/day) that exceed the WHO tolerable daily intake of 1-4 pg TEQ/kg bw/day.

Keywords

Novel brominated flame retardants (NBFRs); hexabromocyclododecanes (HBCDs), polychlorinated biphenyls (PCBs); organochlorine pesticides; e-waste recycling in China; human dietary intake.

1 Introduction

Electrical and electronic waste (e-waste) can contain a wide range of hazardous chemicals, including brominated flame retardants (BFRs), and polychlorinated biphenyls (PCBs) (Robinson 2009). Moreover, improper e-waste treatment techniques, such as open burning of plastics to recover metals, introduce additional chemicals as by-products, including an additional source of dioxin-like PCBs (DL PCBs) (Frazzoli et al. 2010), over and above their presence in commercial PCB formulations. Such rudimentary recycling and disposal practices can result in high occupational and incidental exposure to a range of hazardous substances, and generate locally severe environmental contamination. Once released to the environment surrounding e-waste treatment facilities, BFRs and PCBs may accumulate in soils and sediments (Tang et al. 2010; Zhang et al. 2012; Labunska et al. 2013a; Liu et al. 2013) and become available for uptake by both terrestrial and aquatic organisms (Fu et al. 2010; Shang et al. 2013). Consequently, contaminants may enter the food chain with resultant pervasive and prolonged exposure to people residing in the vicinity of e-waste recycling areas.

A range of health effects has been reported in association with e-waste recycling activities in China, including decreased child height (Zheng et al. 2008), increases in adverse birth outcomes (Xu et al. 2012) and increased cancer risk in e-waste dismantling workers. Exposure to high concentrations of polychlorinated dibenzodioxins/furans (PCDDs/Fs), polybrominated diphenyl ethers (PBDEs) and PCBs may well play a contributing role in the incidence of these conditions (Wen et al. 2008). It has also been suggested that PBDE exposure through e-waste recycling operations may interfere with the thyroid hormonal system and cause genotoxic damage (Yuan et al. 2008).

Food has been reported as one of the main routes of human exposure to organohalogen chemicals (OHCs), including exposure to PCBs (Xing et al. 2009; Song et al. 2011), PBDEs (Ni et al. 2012, 2013; Chan et al. 2013), hexabromocyclododecane (HBCD) (Zheng et al. 2012; He et al. 2013) and “novel” BFRs (NBFRs) (Zheng et al. 2012). Despite this, data on human dietary exposure to BFRs and PCBs associated with e-waste recycling are still limited or – in the case of HBCD and NBFRs - are non-existent.

Relatively few studies exist that report human dietary exposure to multiple classes of OHCs, with even fewer addressing such exposures arising as a consequence of e-waste treatment. Some of those studies focus on exposure *via* a single type of food for multiple contaminants, e.g. monitoring a range of BFRs *via* consumption of domestic eggs originating from e-waste sites in South China (Zheng et al. 2012), while others assess exposure to a single class of contaminant (e.g. PBDEs), *via* consumption of a number of different food items, for example, chicken muscle and eggs from e-waste sites in Southeast China (e.g. Qin et al. (2011)). Our study estimates cumulative dietary exposures to selected OHCs *via* consumption of a range of animal-derived foodstuffs produced in an area in which rudimentary e-waste treatment is commonplace.

We have reported previously on human dietary exposure to PBDEs *via* consumption of duck eggs (Labunska et al. 2013b), and nine other staple foodstuff categories (Labunska et al. 2014) originating from e-waste recycling areas in Taizhou, eastern China. The current study was designed to investigate human dietary exposure to NBFRs and PCBs arising through consumption of the same foodstuffs, as well as to selected organochlorine pesticides (OCPs). While we did not expect e-waste treatment to be a source of OCPs, information on dietary exposure to these legacy contaminants permits better understanding of overall cumulative exposure to OHCs – many of which can exert toxicological effects. Moreover, the presence of NBFRs in food samples is of interest. Firstly, no regulations on the production or use of these alternative BFRs exist, so it is likely that their use as replacements for regulated BFRs will be increasing. Secondly, some NBFRs appear to be bioaccumulative, with bioaccumulation factors (Log BAFs) for 1,2 bis(2,4,6-tribromophenoxy)ethane (BTBPE) and hexabromobenzene (HBB) reported to range from 3.32 to 6.08 and from 3.31 to 5.04 respectively in aquatic species from a natural pond in an electronic waste recycling site in South China (Wu et al. 2011). Combined, these factors suggest that concentrations of NBFRs in food will likely rise substantially in the future. This is concerning, because although relatively little is known about their toxicity (Covaci et al. 2011; Stieger et al. 2014), a recent study has reported *in vitro* endocrine disruptive properties for both 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EH-TBB) and bis-(2-ethylhexyl)-3,4,5,6-tetrabromo-phthalate (BEH-TBP) (Saunders et al. 2013).

The specific OHCs investigated in the current study were: α -, β - and γ - HBCD, tri- to deca-chlorinated PCBs (33 congeners), dichlorodiphenyltrichloroethane (DDT) and its metabolites, hexachlorobenzene (HCB), and a range of NBFRs, comprising: pentabromoethylbenzene (PBEB), HBB, EH-TBB, BEH-TBP, BTBPE, and decabromodiphenyl ethane (DBDPE).

2 Methodology

2.1 Sample collection and preparation

A full description of the study area and sampling sites is provided in our earlier publications (Labunska et al. 2013b, 2014). Briefly, 127 samples of foodstuffs, comprising: fish, shrimps, meat, liver and eggs from chickens and ducks, and pork were purchased in Wenling and Luqiao districts located within Taizhou City, one of the largest e-waste recycling areas in Eastern China (Fig. S1). Control samples (n=62) were also procured, along with culinary vegetable oils (n=10) which were not directly impacted by e-waste treatment. While most control samples were purchased from supermarkets and local markets in Shanghai City and Nanjing City; chicken and duck liver control samples were obtained from a local market in Taizhou City, but were reported by vendors to originate from locations around the city that were not involved in any e-waste recycling.

Our earlier estimates of exposure to PBDEs were based on analysis of 189 individual food samples from e-waste and non-e-waste impacted areas. The wider range of contaminants measured in this study, necessitated the preparation of composite or ‘pooled’ samples in order to reduce the number of samples requiring analysis. To achieve this, portions of homogenised individual samples were combined according to food category (e.g. chicken eggs, duck meat etc.) and whether they originated from e-waste or non-e-waste impacted areas.

Treatment of the individual food samples used here for preparation of pooled samples, has been described previously (Labunska et al. 2014). In short, samples of meat, fish, shrimps and liver were cooked to reflect the condition of the foodstuffs as the point of consumption, then freeze dried, homogenised and stored at -20°C. Fat and liquids formed during cooking were discarded.

Chicken eggs were heat treated as has been described previously for duck eggs (Labunska et al. 2013b), and only yolks were subject to analysis. Pooled samples were then prepared by mixing equal amounts of each sample from each product type (2 g per sample for shrimps, 3 mL per sample for vegetable oils and 1 g per sample for all others). Pooled samples were then homogenised thoroughly using a Waring 32BL80 commercial blender, followed by further homogenisation using a mortar and pestle. Each homogenised pooled sample was sub-sampled into pre-cleaned extraction vials and stored at -20°C until analysis. Descriptions of the origins of these pooled samples are presented in Table S1. Pooled samples of meat, fish, shrimps and liver were prepared using individual samples from both Wenling and Luqiao districts combined. For chickens' eggs, separate pooled samples were prepared for Wenling and Luqiao district respectively using individual yolks from each site. In the case of duck egg yolk, one pooled sample was prepared per location for each of the five e-waste locations monitored in Wenling and Luqiao (Labunska et al. 2013b). One pooled control sample was prepared for each food category examined.

In addition to obtaining information on human exposure to a wide range of OHCs, we also took the opportunity to enhance understanding of how diastereomer and enantiomeric profiles of HBCDs vary between fish species, as well as how such profiles vary between muscle and liver in chickens and ducks. To do so, we analysed individual samples of 5 fish species - common carp (*Cyprinus carpio*), loach (*Misgurnus angullicaudatus*), snakehead (*Channa argus*), Chinese bream (*Megalobrama amblycephala*) and Chinese perch (*Siniperca chuatsi*) - as well as three paired samples (i.e. muscle and liver from the same three animals) for both chickens and ducks.

2.2 Sample extraction, clean up and analysis

2.2.1 HBCDs

Extraction was carried out using a pressurised liquid extraction technique utilising an Accelerated Solvent Extractor ASE 350 (Dionex). Diatomaceous earth (30/40 mesh) and Florisil (60/100 mesh) were purchased from Thames Restek (UK); anhydrous Na₂SO₄ - from Fisher Scientific (UK). Native α -, β -, and γ -HBCD standards (50 μ g/mL in toluene) were obtained from Cambridge Isotope Laboratories (Andover, MA, USA), while isotope-labelled ¹³C α -, β -, γ -HBCDs and α -HBCD-d₁₈ were purchased from Wellington Laboratories (Guelph, ON, Canada). Pentane (Rathburn Chemicals, UK) was HPLC grade; acetone and concentrated sulfuric acid (Fisher Scientific, UK) were AR grade.

An accurately weighed aliquot (~0.5 g) of each dried pooled sample was mixed with 5 g of anhydrous Na₂SO₄, transferred into a 33 mL stainless steel extraction cell half-filled with diatomaceous earth, spiked with 20 ng of each of ¹³C-labelled α -, β -, and γ - HBCD as internal (surrogate) standards, topped with diatomaceous earth, and extracted using the ASE 350 (temperature 90°C, pressure 1500 psi, heating time 5 min, static time 4 min, purge time 90 s, flush volume 50%, 3 static cycles, solvent pentane/acetone 3:1). A 2 mL aliquot of the resultant extract was used for gravimetric determination of lipid content. The remainder of the extract was evaporated to incipient dryness using a Turbovap system and reconstituted with 2 mL pentane.

Extracts were cleaned with 3 mL of concentrated sulfuric acid. After separation of phases, each extract was transferred to a glass column containing 2 g of Florisil topped with 0.5 g of anhydrous Na₂SO₄ and eluted with 25 mL of pentane/acetone (3:1 v/v). Cleaned extracts were evaporated to complete dryness using a gentle stream of nitrogen and reconstituted in 100 μ L of α -HBCD-d₁₈ (25 pg/ μ L in methanol) used as recovery determination (or syringe) standard for

QA/QC purposes. Sample analysis was carried out using an LC-MS/MS system composed of a dual pump Shimadzu LC-20AB Prominence liquid chromatograph equipped with a SIL-20A autosampler and a DGU-20A3 vacuum degasser, coupled to a Sciex API 2000 triple quadrupole mass spectrometer. Details of the methodology, LC columns and mass spectrometric conditions for both chiral and achiral separation and quantification of HBCDs can be found elsewhere (Abdallah and Harrad 2011). Regular analysis of method blanks (1 blank for every 5 samples) revealed no detectable interferences from target compounds. Method limits of detection (LOD) and quantification (LOQ) were estimated based on 3:1 and 10:1 signal:noise (S:N) ratios respectively (Table S2). Good recoveries (60 - 109 %) of the ^{13}C -labelled internal standards were obtained for α -, β -, and γ -HBCDs. The accuracy and precision of the analytical method for HBCDs was assessed via replicate analysis (n=4) of NIST SRM 2794A (Organics in Freeze-Dried Muscle Tissue). The results obtained compared favourably with the reference values.

2.2.2 NBRs

An accurately weighed aliquot (~0.5 g) of pooled sample was loaded into an ASE cell (half-filled with pre-extracted Hydromatrix), spiked with internal standards (15 ng of each of BDE 77, BDE 128, $^{13}\text{C}_{12}$ -BTBPE and 30 ng of $^{13}\text{C}_{12}$ -BDE 209), and extracted with hexane:acetone (3:1 v/v) using an ASE 350. After concentration to ~1 mL using the Turbovap system, extracts were eluted through activated Florisil (2 g) using 18 mL hexane (fraction 1) and 12 mL of DCM (fraction 2). Following evaporation to ~1 mL, fraction 1 was eluted through 44% acidified silica (3 g) with 20 mL hexane:DCM (1:1). Fraction 2 was reduced to 0.5 mL and solvent exchanged to hexane, prior to loading onto an isolute NH_2 cartridge and elution with hexane:DCM (1:1 v/v, 15 mL). The eluate from this cartridge was combined with that obtained by passing fraction 1

through Florisil. This combined eluate was concentrated under a gentle stream of nitrogen and solvent exchanged to iso-octane (200 μ L) containing PCB 129 as recovery determination (or syringe) standard. Target analytes were quantified using a TRACE™ 1310 Gas Chromatograph coupled to ISQ™ single quadrupole mass spectrometer (ThermoScientific, Austin, TX, USA) operated in negative ion chemical ionisation mode. Chromatographic resolution of NBFRs was achieved on a HP5-MS capillary column (15 m x 0.25 mm x 0.1 μ m; Agilent, CA, USA) according to a previously described method (Van den Eede et al. 2012). Recoveries of the internal standards ranged from 74 to 102 % in all samples. LOQs for the method are presented in Table S2.

2.2.3 PCBs and OCPs

Pooled samples (typically 0.5 g) were weighed accurately, homogenised, mixed with anhydrous Na₂SO₄, and spiked with PCB-143 as internal standard, prior to automated hot Soxhlet extraction for 2 h with hexane:acetone (3:1, v:v). Sample lipid content was determined gravimetrically by drying 10% of the crude extract for 1 h at 100 °C. The remaining extract was purified by elution through acidified silica (8 g; 44% H₂SO₄) with 20 mL of hexane followed by 15 mL of dichloromethane. Eluates were concentrated to incipient dryness under a gentle nitrogen flow and reconstituted in 150 μ L iso-octane. Analytes were quantified using a mass spectrometer (Agilent MS 5973, Palo Alto, CA, USA) coupled to a gas chromatograph (Agilent GC 6890, Palo Alto, CA, USA) equipped with a HT-8 capillary column (30 m x 0.22 mm x 0.25 μ m; SGE Analytical Science, Zulte, Belgium) operated in electron ionisation mode. Recoveries of the internal standard ranged from 81 to 106% (RSD =10%) in all samples. For every tenth sample, a procedural blank and certified reference material (SRM 1945 whale blubber) were analysed. The

LOQ (see Table S2) was defined either as three times the standard deviation of the procedural blanks, or for analytes not detectable in blanks, as S:N = 10:1.

2.3 Statistical analysis & daily intake calculations

Statistical analysis was performed using Microsoft Office Excel 2013. Exposure estimates were calculated using daily consumption rates for various food products derived from a number of publications, as described previously (Labunska et al. 2014). Adult body weights (BW_s; kg) employed in our calculations were the average Chinese male body weight of 63 kg (Zhou et al. 2012), with that for children equal to that for a standard Chinese 3 year old boy living in a rural area of 14.65 kg (MHC 2012). The total dietary intake (TDI; ng/kg bw/day) was calculated according to:

$$TDI = \sum_{i=1}^{10} \left(\frac{C_i \times CR_i}{BW} \right)$$

Where C_i is the concentration (ng/g ww) of OHC in a food sample and CR_i is the daily consumption rate of the foodstuff concerned (g/day) (see Table S5 for consumption rates of individual foodstuffs).

3 Results and discussion

3.1 Concentrations and patterns of contaminants in food

3.1.1 NBFRs

Concentrations of NBFRs in e-waste related samples in this study generally exceeded concentrations in the corresponding control samples by substantial margins (Table 1), indicating

a discernible influence of e-waste treatment on concentrations of these contaminants in locally produced foods. EH-TBB and BEH-TBP were detected in all e-waste related samples, with BTBPE detected in 86% of samples. Less frequently detected were PBEB and HBB (>LOQ in 36% of e-waste related samples) while DBDPE was detected only in shrimps. The highest concentration among NBFs was found for EH-TBB (fish: 62.2 ng/g lw) followed by DBDPE (shrimps: 45.3 ng/g lw). Consistent with previous findings for waterbirds from e-waste recycling sites in South China (Zhang et al. 2011), concentrations of EH-TBB, BEH-TBP, BTBPE and PBEB in avian livers in this study (chicken: 35.0, 10.6, 15.0, and 2.3 ng/g lw; duck: 38.4, 13.7, 11.7, and 3.4 ng/g lw, respectively) exceeded those in corresponding muscle samples (chicken: 24.7, 9.0, 1.5, and <0.17 ng/g lw; duck: 24.2, 7.2, 4.6, and 0.9 ng/g lw, respectively). In contrast, in both chickens and ducks, HBB was detected in muscle but not liver. Finally, while concentrations of PBEB, HBB, and DBDPE were all <LOQ in chicken and duck eggs, EH-TBB and BEH-TBP were detected in all e-waste related egg samples (average: 4.3 and 1.1 ng/g lw, respectively). No information related to the concentrations of BEH-TBP, EH-TBB, BTBPE, DBDPE, PBEB and HBB in food samples has been identified in a meta-analysis conducted by the European Food Safety Authority (EFSA 2012). However, we are aware of one study of chicken eggs from e-waste recycling sites in South China which reported concentrations of NBFs exceeding those in our study (Zheng et al. 2012). Specifically, this earlier study reported BTBPE, DBDPE, HBB and PBEB to be present at mean concentrations of 37.2–264, 5.97–37.9, 7.32–25.7, and 0.63–0.78, ng/g lw, respectively, compared to those in eggs in our study (range: <0.35 – 3.84 ng/g lw) (see Table 1). Another study (Fernandes et al. 2009) reported concentrations of three of our target NBFs (HBB, BTBPE and DBDPE) in a wide range of foodstuffs in the UK, with BTBPE being the only NBF detected above LODs and ranging from

1 **Table 1. Concentrations of OHCs in composite food samples from e-waste recycling (Taizhou) and control sites in Eastern**
 2 **China.**

Sample type	Site Category (Control/E-waste)	Lipid%	Concentration, ng/g lipid weight											
			∑HBCDs	∑PCBs	∑DDTs	HCB	PBEB	HBB	EH-TBB	BEH-TBP	BTBPE	DBDPE		
Vegetable oil	Control	100	0.1	<1	<2	<1	<0.17	<0.15	<0.20	<0.25	<0.35	<0.45		
Muscle	Fish	Control	23.4	5.9	79	100	8.3	<0.17	<0.15	4.00	1.90	2.10	<0.45	
		E-waste	5.7	310	75400	820	89.2	<0.17	6.49	62.2	15.5	6.83	<0.45	
	Shrimp	Control	4.5	5.7	240	54	5.4	<0.17	0.97	11.7	9.32	<0.35	9.19	
		E-waste	4.5	<0.06	1170	170	11.5	6.81	<0.15	36.8	16.3	9.07	45.3	
	Chicken	Control	19.0	0.09	<1	<2	6.3	<0.17	<0.15	2.66	1.78	<0.35	<0.45	
		E-waste	38.6	78.7	500	51	54	<0.17	0.41	24.7	8.97	1.46	<0.45	
	Duck	Control	18.2	2.4	33	<2	9.6	1.16	<0.15	2.74	<0.25	1.87	<0.45	
		E-waste	48.1	26.4	1320	70	18	0.87	0.56	24.2	7.23	4.57	<0.45	
	Pork	Control	24.6	3.1	<1	<2	2.4	<0.17	<0.15	2.14	1.37	2.69	<0.45	
		E-waste	18.4	<0.06	<1	<2	<1	<0.17	0.41	38.2	12.4	5.40	<0.45	
	Liver	Chicken	Control	38.6	78.7	500	51	54	<0.17	0.41	24.7	8.97	1.46	<0.45
			E-waste	20.9	42.5	410	200	49.9	2.30	<0.15	35.0	10.6	15.0	<0.45
Duck		Control	21.5	29.9	3	6	11.2	<0.17	<0.15	8.20	1.69	3.27	<0.45	
		E-waste	14.5	5.7	570	43	15.5	3.37	<0.15	38.4	13.7	11.7	<0.45	
Egg	Chicken	Control	29.6	1.7	<1	3	4.8	<0.17	<0.15	1.73	<0.25	<0.35	<0.45	
		E-waste	28.5	47.5	740	61	200	<0.17	<0.15	4.86	0.94	2.02	<0.45	
		E-waste	26.5	37.7	1180	32	37.4	<0.17	<0.15	4.73	1.37	3.84	<0.45	
	Duck	Control	32.7	0.5	<1	6	5.2	<0.17	<0.15	1.21	<0.25	<0.35	<0.45	
		E-waste	32.0	9.3	740	560	68.6	<0.17	<0.15	3.09	0.97	1.11	<0.45	
		E-waste	27.0	3.0	62	75.9	40.3	<0.17	<0.15	5.42	1.65	3.07	<0.45	
		E-waste	31.0	17.5	140	14	5.9	<0.17	<0.15	3.67	1.24	2.15	<0.45	
		E-waste	28.8	2.2	59	140	15.6	<0.17	<0.15	4.40	0.81	<0.35	<0.45	
E-waste	29.6	14.7	2860	110	21.9	<0.17	<0.15	3.56	0.88	<0.35	<0.45			

3

4 0.1 ng/g lw (in beef burger) to 3.33 ng/g lw (in lemon sole). As well as the presence of a much
5 wider range of NBFs; concentrations of BTBPE in our study exceeded those in UK foods. For
6 example, BTBPE was detected in our pooled chicken liver sample at 15.0 ng/g lw, compared to
7 0.75 ng/g lw in the UK study, while it was present at 6.83 and 9.07 ng/g lw in our fish and
8 shrimp samples respectively, exceeding the concentrations reported in UK fish (range: 0.26 –
9 3.33 ng/g lw).

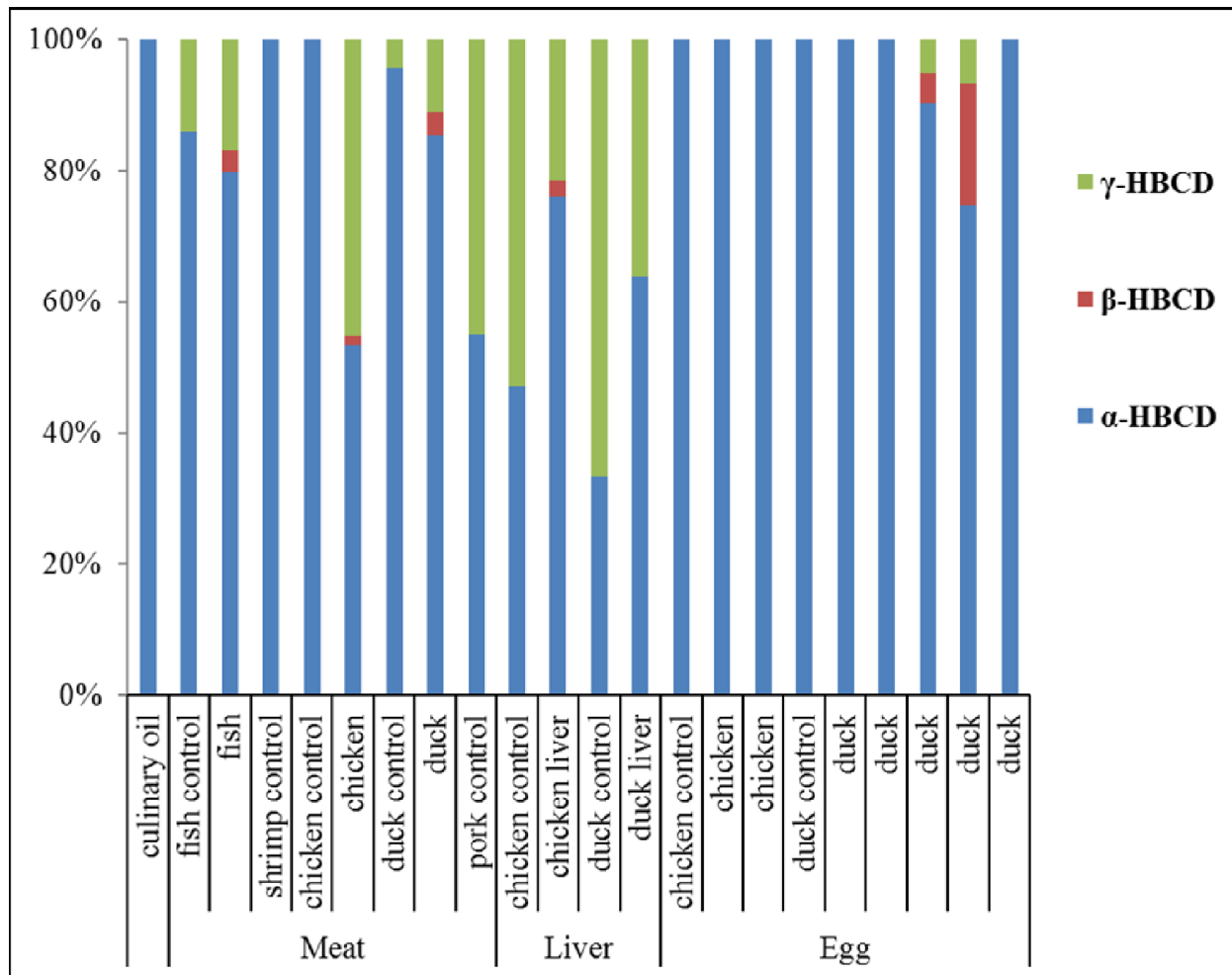
10

11 3.1.2 HBCD

12 Amongst pooled samples, the highest concentration of Σ HBCD was detected in fish muscle (310
13 ng/g lw) followed by chicken muscle, chicken egg, and chicken liver (79, 47 and 43 ng/g lw,
14 respectively) (Table 1). Concentrations of HBCD in all analysed samples are provided in Table
15 S4. Those in most of the e-waste related samples exceeded those in corresponding controls by 1-
16 3 orders of magnitude. However, HBCDs were <0.06 ng/g lw in e-waste related pork and shrimp
17 samples, while the corresponding control samples contained 3.1 and 5.7 ng/g lw Σ HBCDs,
18 respectively. Moreover, concentrations of Σ HBCDs in the duck liver control sample, exceeded
19 five-fold that detected in duck livers from e-waste-impacted locations. A previous study from the
20 Pearl River Delta in South China (Sun et al. 2012) reported higher concentrations of HBCDs in
21 the muscle and stomach of three species of passerine birds collected from urban sites than in
22 those from e-waste recycling sites. That difference was attributed to higher releases of HBCD
23 from textile and construction applications in urban areas. This may also help explain the higher
24 concentrations of Σ HBCDs in our control samples, which were all obtained from urban areas
25 which were not in the vicinity of e-waste recycling operations. Furthermore, as the main
26 application of HBCDs is in thermal insulation boards and construction materials (Covaci et al.

27 2006), a more widespread presence of this compound may be expected in urban areas, with no
28 particular association expected with proximity to e-waste recycling sites. Nevertheless, e-waste-
29 impacted samples of duck muscle, as well as of chicken muscle and liver displayed significantly
30 higher concentrations of \sum HBCDs than corresponding controls. We hypothesise this occurs
31 because chicken and duck housing at the Taizhou e-waste sites are often constructed from
32 potentially HBCD-containing recycled construction materials, as well as containing old furniture
33 and textiles that may also be treated with HBCD. This might also explain the higher
34 concentration of \sum HBCDs in chickens than in ducks from e-waste sites in our study as chickens
35 spend more time in their housing, though a significant contribution to the patterns from
36 differences in feeding habits between ducks and chickens cannot be ruled out. A similar
37 combination of factors may account for our observation that e-waste-impacted chicken eggs
38 contained higher concentrations of \sum HBCDs (47.4 and 37.7 ng/g lw for Wenling and Luqiao e-
39 waste sites respectively) than duck eggs from the same locations (range: 3.0 – 17.5 ng/g lw;
40 average: 9.35 ng/g lw). To the authors' knowledge, there are no previously reported data on
41 HBCD concentrations in domestic duck eggs. Interestingly, while the lowest \sum HBCD
42 concentration was detected in a pooled duck egg sample from a site in the Wenling district at
43 which e-waste activity has decreased substantially recently, the highest concentration was found
44 in a sample from a site in the same district where e-waste recycling had only recently ceased.
45 The farm on this former e-waste recycling area was the largest in our study with 2200 ducks
46 onsite, and which contained several large duck shelters thickly covered with recycled textile and
47 construction materials that potentially may contain substantial amounts of HBCDs (Alaee et al.
48 2003). This highlights the importance of sources of HBCDs other than e-waste treatment to our
49 samples.

50 Fig. 1. α -, β - and γ -HBCD diastereomer profiles in composite food samples containing
 51 concentrations above LOQ.



52
53

54 Diastereomer profiles of HBCDs in pooled samples are presented in Fig. 1. In general, α -HBCD
 55 was the predominant HBCD congener contributing on average $77 \pm 31\%$ Σ HBCDs in all our
 56 samples (from e-waste areas and controls). This accords with previous reports of the
 57 predominance of α -HBCD in biota as opposed to the higher contribution of γ -HBCD found in
 58 commercial formulations and abiotic matrices (Marvin et al. 2011). This has been attributed to
 59 several factors, including preferential biotransformation of the β - and γ -isomers (Abdallah and

60 Uchea 2014) in addition to higher bioavailability and higher bioaccumulation potential of α -
 61 HBCD (Szabo et al. 2011a; Abdallah et al. 2012; Fournier et al. 2012). Interestingly, while γ -
 62 HBCD comprised only 22% and 36% Σ HBCDs respectively in chicken and duck livers from e-
 63 waste related sites, in control samples of the same matrices, γ -HBCD contributed 53% and 67%
 64 of Σ HBCDs respectively. This suggests exposure of the control birds to a comparatively “fresh”
 65 HBCD source.

66

67 **Table 2. Concentrations of HBCDs (ng/g lw) in individual samples of fish, and paired**
 68 **samples (muscle and liver) of chickens and ducks.**

Sample Type		Location ^a	α -HBCD	β -HBCD	γ -HBCD	Σ HBCDs
Fish	carp	Wenling, Taizhou, Site 4	1530	56.0	350	1940
	loach	Luqiao, Taizhou, Site 5	490	30.0	190	710
	snakehead	Luqiao, Taizhou, Site 7	620	14.6	39.4	670
	bream	Shanghai supermarket, control sample	1.2	2.2	7.4	10.8
	perch	Nanjing supermarket, control sample	20.1	3.3	11.9	35.3
Duck	meat	Wenling, Taizhou, Site 1	19.2	0.5	5.6	25.3
	liver		3.1	3.5	11.9	18.5
	meat	Wenling, Taizhou, Site 2	1.6	0.1	0.9	2.6
	liver		0.2	0.4	2.5	3.1
	meat	Wenling, Taizhou, Site 2	1.7	0.2	0.9	2.8
	liver		0.7	0.2	2.7	3.7
Chicken	meat	Wenling, Taizhou, Site 2	7.6	1.2	5.1	14.0
	liver		1.9	3.5	12.8	18.3
	meat	Wenling, Taizhou, Site 1	490	9.0	150	640
	liver		190	4.4	190	390
	meat	Wenling, Taizhou, Site 2	14.4	1.4	5.4	21.1
	liver		5.4	1.8	5.7	12.9

69 ^a full description of sampling sites including assignment of sample codes is reported in Labunska et al.
 70 (2014)

71

72 Analyses also revealed different HBCD diastereomer profiles in liver and muscle tissues taken
73 from individual birds (Table 2), similar to those reported recently for predatory birds in Belgium
74 (Eulaers et al. 2014). Moreover, such intra-bird variation was exceeded still further by the
75 variation in profiles between duck and chicken tissues.

76 Notably, γ -HBCD was the dominant diastereomer in some avian liver samples compared to the
77 predominance of α -HBCD in muscle tissue from the same birds. We hypothesise this
78 predominance of γ -HBCD in some avian liver samples reflects the profile to which the bird is
79 exposed, as the liver is the first organ exposed after the gastrointestinal tract.

80 In contrast, the diastereomer profile in muscle tissue reflects more the chronic exposure after
81 metabolism and is thereby skewed towards α -HBCD, due to the latter's enhanced
82 bioaccumulation potential and resistance to enzymatic metabolism (Szabo et al. 2011b; Fournier
83 et al. 2012).

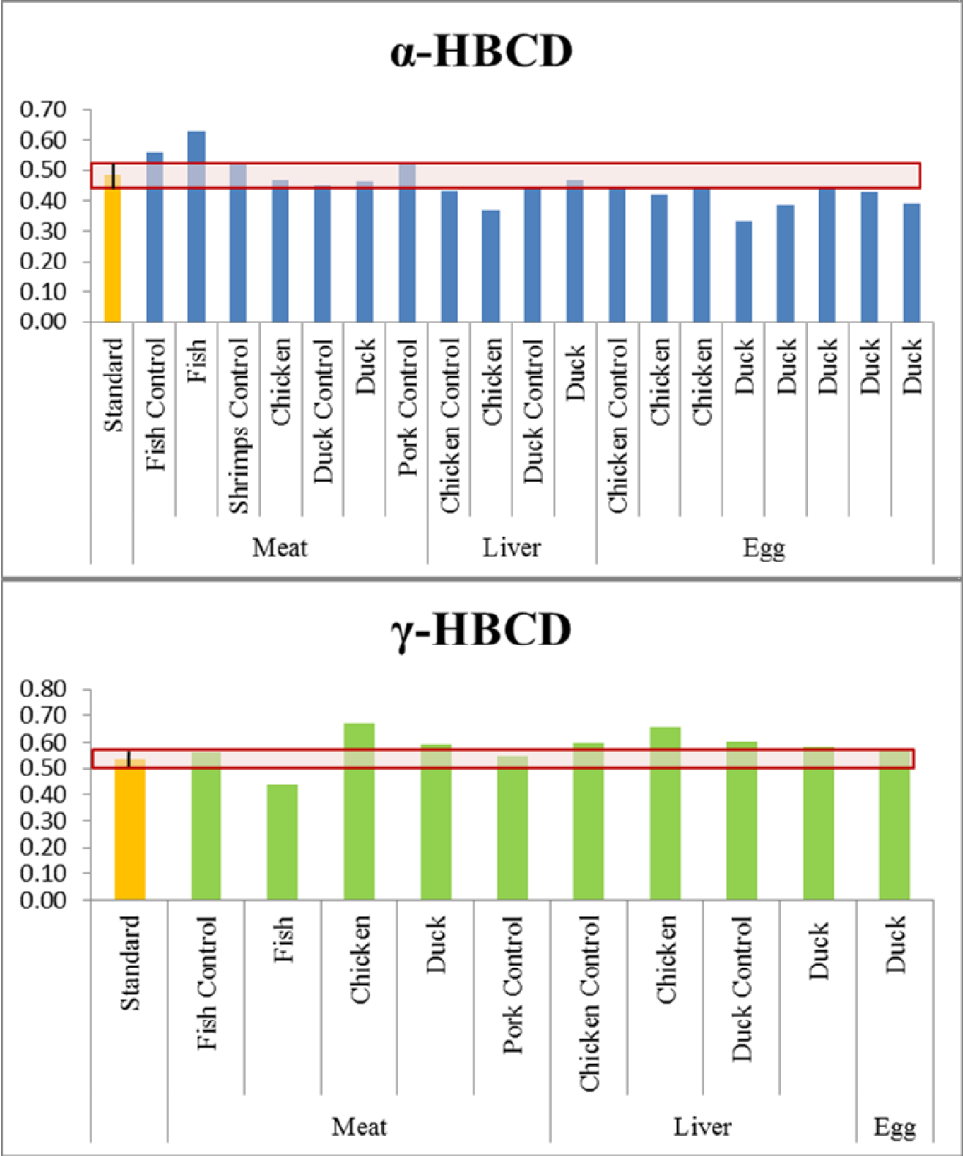
84 HBCD enantiomer fractions (EFs) are presented in Fig. 2. Generally, muscle and eggs from
85 ducks and chickens displayed significant enrichment of the (-)- α -HBCD and (+)- γ -HBCD
86 enantiomers. This is in agreement with previous reports for terrestrial bird samples from e-waste,
87 urban and rural locations in South China (He et al. 2010; Sun et al. 2012) and eggs of predatory
88 birds from Northern Europe (Janak et al. 2008).

89 Fish samples showed variable EF values with enrichment of (+)- α -HBCD in loach, carp and
90 bream, (-)- γ -HBCD in loach, and (+)- γ -HBCD in bream. Our results are consistent with previous
91 studies reporting species-specific variation in EFs in fish (Janak et al. 2005; Köppen et al. 2010;
92 He et al. 2013). These findings indicate potential enantioselective processes associated with

93 absorption, biotransformation and/or excretion of HBCDs in the fish and bird species studied,
 94 and that such processes are species-specific.

95

96 **Fig 2. Enantiomer fractions (EF) of α -HBCD and γ -HBCD in pooled food samples**
 97 **containing concentrations above LOQ. Error bar – SD for a racemic standard.**



98

99

100 3.1.3 PCBs

101 With the exception of pork and chicken liver, concentrations of Σ PCB concentrations in our e-
102 waste related food samples were all higher than in corresponding control samples (see Table 1).
103 Results for individual PCB congeners in all samples are presented in Table S3. PCBs can arise in
104 environmental samples from both primary and secondary sources and, because of their
105 persistence and long-range transport, are widely distributed environmental contaminants.

106 The relatively high levels in many of the pooled food samples from e-waste areas are, however,
107 suggestive of primary contributions from local point sources, such as recycling and disposal of
108 obsolete transformers, an activity widely practised in Luqiao (Xing et al. 2011; Zhanz et al.
109 2012) as well as handling of other electrical and electronic waste.

110 In view of the history of transformer treatment at Luqiao, it is thus unsurprising that
111 concentrations of Σ PCB from samples obtained from this site were the highest in this study. Of
112 the species considered, concentrations were highest in fish (75416 ng/g lw or 1168 ng/g ww),
113 followed by those in duck eggs (site 5, 2857 and 846 ng/g ww), duck muscle (1322 and 276 ng/g
114 ww) and chicken eggs (1179 and 312 ng/g ww). Consistent with previous studies (Rajaei et al.
115 2010; Xing et al. 2010) and possibly attributable to higher metabolic activity in liver than
116 muscle, concentrations of Σ PCB in livers of both ducks and chickens were lower than in the
117 corresponding muscle samples. It is also interesting to note that, while the concentration of
118 Σ PCB in duck muscle exceeded approximately three-fold that in chicken muscle (276 and 88
119 ng/g ww, respectively), concentrations were similar in both duck and chicken livers at \sim 30 ng
120 Σ PCB/g ww. This may reflect species-specific differences in metabolism.

121 Concentrations of Σ PCB in duck meat (276 ng/g ww), duck liver (32 ng/g ww) and eggs
122 (average in chicken and duck eggs was 109 and 91.2 ng/g ww respectively) substantially

123 exceeded those reported previously for e-waste-related sites in Taizhou (13.8, 2.1 and 7.9 ng/g
 124 ww, respectively) (Xing et al. 2010).

125

126 **Table 3. Concentrations of indicator PCBs, Σ NDL-PCBs and Σ DL-PCBs in food**
 127 **originating from Taizhou, China; and maximum levels (ML) for these contaminants in**
 128 **foodstuffs (EU 2011).**

	Σ PCBs, indicator ^c	Σ NDL- PCBs ^d	ML, Σ NDL-PCBs ^e	Σ DL-PCBs- TEQ ^f	ML, Σ DL- PCBs-TEQ ^g
	ng/g			pg/g TEQ	
Fish	34456 ^a	441 ^b	125 ^b	4.32 ^b	3.0 ^b
Shrimps	569 ^a	5.82 ^b		0.09 ^b	
Chicken muscle	270 ^a	206 ^a	40 ^a	3.07 ^a	1.25 ^a
Duck muscle	698 ^a	530 ^a		7.74 ^a	
Chicken liver	180 ^a	137 ^a	40 ^a	2.09 ^a	5.5 ^a
Duck liver	288 ^a	225 ^a		3.15 ^a	
Chicken egg	518 ^a	388 ^a	40 ^a	6.41 ^a	2.5 ^a
Duck egg	428 ^a	336 ^a		5.38 ^a	

129 ^a expressed on a lipid weight basis;

130 ^b expressed on a wet weight basis;

131 ^c sum of PCB28, PCB52, PCB101, PCB118, PCB138, PCB153 and PCB180;

132 ^d sum of PCB28, PCB52, PCB101, PCB138, PCB153 and PCB180;

133 ^e maximum levels for sum of PCB28, PCB52, PCB101, PCB138, PCB153 and PCB180;

134 ^f sum of PCB105, PCB118, PCB156 and PCB167;

135 ^g maximum levels for sum of twelve dioxin-like PCBs derived by subtraction of value for the
 136 sum of dioxins (WHO-PCDD/F-TEQ) from the sum of dioxins and DL-PCBs (WHO-PCDD/F-
 137 PCB-TEQ).

138

139 The homologue pattern of PCBs in avian-related samples originating from our e-waste sites (see
 140 Fig. S4) were dominated by hexa-PCBs (from 28% Σ PCBs in duck meat to 41% Σ PCBs in duck
 141 eggs), followed closely by penta-PCBs. In contrast, tetra- and penta-PCBs were more prominent

142 in aquatic species. Penta-PCBs comprised 36% Σ PCBs in fish and 26% Σ PCBs in shrimps, with
143 tetra-PCBs contributing 27% Σ PCBs in fish and 29% Σ PCBs in shrimps. While these homologue
144 patterns are consistent with previous observations for Taizhou (Xing et al. 2010); those in our e-
145 waste impacted fish samples differed from patterns observed in Europe, in which hexa-PCBs
146 (mostly PCB-153 and PCB-138) are dominant in fish (Roosens et al. 2008; Malarvannan et al.
147 2014). The different patterns seen in our e-waste-associated samples may reflect influences from
148 multiple local sources at e-waste recycling sites. These include releases from PCB-containing
149 transformers and/or capacitors as well as unintentional formation of some PCBs during poorly-
150 controlled combustion processes (Shen et al. 2009).

151 Concentrations of non-dioxin-like (NDL-) and dioxin-like (DL-) PCBs in comparison to
152 maximum levels (ML) promulgated by the European Commission for certain contaminants in
153 foodstuffs (EU 2011) are presented in Table 3. The sum of seven indicator PCBs (PCB-28, 52,
154 101, 118, 138, 153 and 180) in fish (534 ng/g ww) in our study exceeded the Chinese maximum
155 residue limit of 500 ng/g ww for aquatic organisms and aquatic products (MOH 2012).

156 Additionally, with the exception of chicken liver, all samples of avian origin from e-waste sites
157 in our study (Table 3) exceeded the maximum level of 200 ng/g lw set by the European
158 Commission (EC 1999), for the sum of these seven indicator PCBs in products including eggs
159 and poultry.

160 Another EC Regulation No. 1881/2006 (EC 2006) established maximum levels (ML) for dioxins
161 and DL-PCBs in a range of foodstuffs. This was amended recently (EU 2011) to include limits
162 for the sum of six marker NDL-PCBs (see Table 3). This ML for Σ NDL-PCBs was exceeded by
163 at least an order of magnitude for foodstuffs of avian origin in our study and 3.5 times in case of
164 fish. Moreover, while only four DL-PCBs were measured in this study (PCB-105, 118, 156, and

165 167), Table 3 shows that based on the contribution of these four congeners alone, the
166 corresponding EU ML was exceeded for all foodstuffs studied except for livers.

167

168 3.1.4 Organochlorine pesticides

169 Table 1 shows that concentrations of \sum DDTs in our e-waste related food samples were all higher
170 than in the corresponding control samples, with the exception of pork. Concentrations of DDT
171 and its metabolites are presented in Table S3. DDT is a significant contaminant group in food
172 samples from Taizhou, despite the ban on most uses of this pesticide in China in 1983 (Fu et al.
173 2003). However, an exemption was made for the use of DDT as an intermediate for dicofol
174 production, which has been shown to be a major source of DDT to the Chinese environment over
175 recent decades (Qiu et al. 2005). Concentrations of \sum DDT in our study were highest in fish (823
176 ng/g lw), followed by duck eggs from Wenling site 1 (558 ng/g lw) and chicken liver (201 ng/g
177 lw). On a fresh weight basis, duck eggs contained the highest \sum DDT concentrations (range 1.7 -
178 70.2 ng/g), followed by chicken liver (15.0 ng/g ww) and duck muscle (14.6 ng/g ww).

179 Interestingly, while duck muscle contained a higher fresh weight concentration of \sum DDTs than
180 duck liver, the concentration in chicken liver was almost twice that in chicken muscle. Neither
181 control nor e-waste related pork samples contained any DDT above detection limits. This is
182 consistent with earlier findings that pork accumulates relatively low loadings of DDT (range:
183 0.09 – 0.89 ng/g ww) in comparison to other animal-based food products (Yu et al. 2012). Most
184 of the duck eggs in our study contained higher \sum DDT concentrations than reported previously
185 for chicken eggs (5.62 and 5.09 ng/g ww for Taizhou and Guiyu respectively) (Man et al. 2013).
186 This was particularly evident for duck eggs from Wenling, site 1, which, at 70.2 ng/g (ww)
187 \sum DDT was close to the Chinese maximum residue limit in eggs of 100 ng/g ww (MOH 2012).

188 To the authors' knowledge, there are no previously published data on concentrations of DDT and
189 its metabolites in Chinese duck eggs, with exception of one study which included duck eggs as
190 part of a pooled sample of eggs that also comprised chicken eggs and egg products (Zhou et al.
191 2012). Our data suggest that these so-called "legacy" contaminants remain prevalent in our
192 study area.

193 The differences in the relative abundance of DDT and its metabolites between bird tissues, livers
194 and eggs in our study compared to studies of other avian species, may be attributed to a variety
195 of factors. Among other influences, these include: the composition of the birds' diet, their age
196 and weight, and the levels and pattern of Σ DDT contamination in the areas they inhabit (Covaci
197 et al. 2004).

198 The principal contributor to concentrations of Σ DDT in all samples in our study, was *p,p'*-DDE
199 (see Fig. S5). Concentrations of *p,p'*-DDE ranged from <2 ng/g lw in pork to 546 ng/g lw in fish.
200 In contrast, *o,p'*-DDT was not present above detection limits in any of our samples, while *p,p'*-
201 DDT was present in 42% of samples. This predominance of *p,p'*-DDT over its *o,p'*-isomer is
202 consistent with historical contamination of the investigated area by technical DDT which mostly
203 comprises *p,p'*-DDT (up to 77% Σ DDTs). In addition, the ratio of DDT/DDE+DDD (where it
204 was possible to calculate), was always < 1 in our study, with *o,p'*-DDT undetected. This is also
205 consistent with historical contamination by technical DDT, as *o,p'*-DDT is the primary
206 contaminant in dicofol formulations accounting for 46% of the four DDTs (*o,p'*-DDT, *o,p'*-DDE,
207 *p,p'*-DDT, and *p,p'*-CI-DDT), while *p,p'*- DDT contributes only 7% to this sum (Qiu et al. 2005).
208 In agreement with an earlier study on lake fish in China (Guo et al. 2012), *p,p'*-DDD contributed
209 significantly to Σ DDTs in both control and e-waste related fish samples in our study (25 % and
210 19% respectively). Similar DDT isomer patterns were observed for chicken and duck livers,

211 though they displayed slightly lower contributions of *p,p'*-DDD to Σ DDT than observed in fish
212 (at 5% and 16% in avian liver and fish muscle respectively), while in chicken muscle and
213 shrimps, only *p,p'*-DDE was detected.

214 Control chicken and duck eggs similarly contained only *p,p'*-DDE, and then at relatively low
215 concentrations, while those from e-waste related areas contained also *p,p'*-DDT (in chicken
216 eggs) or both *p,p'*-DDT and *p,p'*-DDD (in duck eggs). Reassuringly, none of the muscle samples
217 in our study exceeded either the Chinese maximum residue limit for DDTs of 2000 ng/g lw
218 (MOH 2012) or the 1000 ng/g lw ML set by the EC (EEC 1986).

219 Including controls, HCB was detected in 92% of the samples considered in this study, with the
220 exception of vegetable oils and pork (see Table 1). All control samples showed lower
221 concentrations of HCB than the corresponding e-waste related samples, with the exception of
222 duck liver and pork. Concentrations of HCB (based on lipid weight) were also relatively high in
223 fish (89.2 ng/g), chicken muscle and liver (54.0 ng/g and 49.9 ng/g respectively) and in duck
224 eggs (up to 68.6 ng/g). Overall, food products of avian origin yielded the highest concentrations
225 of HCB in our study, exceeding values reported in a previous total diet study conducted in China
226 in 2007 (Zhou et al. 2012) for both muscle (mean: 1.58 ng/g ww) and chicken eggs (mean: 0.59
227 ng/g ww).

228

229 3.2 Estimation of daily intakes

230 Our estimated exposures of both adults and children to halogenated organic contaminants via the
231 consumption of a range of frequently-consumed local foodstuffs in e-waste recycling sites of
232 Taizhou are presented in Table 4. The contributions to total dietary exposures for organic
233 contaminants from different types of food are presented in Fig. 3. Comparison of total dietary

234 exposures to HBCD, PCBs, DDT and metabolites, HCB, NFRs, and our previously reported
235 data for PBDEs (Labunska et al. 2014) are presented in Fig. 4.

236

237 3.2.1 NFRs

238 Estimated total dietary exposure to NFRs considered in our study decreased in the following
239 order: EH-TBB > BEH-TBP > BTBPE > DBDPE > PBEB > HBB (Table 3).

240 EH-TBB had the highest estimated exposures for both adults and children (8.03 and 18.9 ng/kg
241 bw/day respectively) followed by BEH-TBP (2.6 and 6 ng/kg bw/day respectively). The main
242 source of both EH-TBB and BEH-TBP to dietary exposure was via pork consumption (54 and
243 34% for adults and children respectively). Consumption of pork was also the principal
244 contributor to dietary exposure to BTBPE (44 and 23% for adults and children respectively). For
245 HBB, fish was the main contributor for children (44%), while for adults, both pork and fish each
246 contributed 36% of dietary exposure. Duck muscle and duck liver combined contributed over
247 50% of PBEB dietary exposure for both adults and children, while shrimps were the only food
248 type in our study found to result in exposure to DBDPE. To our knowledge, this study is the first
249 report of dietary exposures to NFRs in Eastern China. One study has reported daily intakes of
250 several NFRs via chicken eggs from e-waste sites in South China (Zheng et al. 2012), but
251 reported their dietary estimates as the sum of all monitored NFRs, in contrast to our compound-
252 specific approach. Moreover, the main contributor to exposure to NFRs in the Zheng et al study
253 was Dechlorane Plus (DP), which was not investigated in our study. Hence, it was not possible to
254 make a direct comparison between the two studies. Moreover, to our knowledge - with the
255 exception of HBB - no health based limit values (HBLVs) with legislative standing exist for the

256 **Table 4. Estimated adult and child exposure to selected organohalogen contaminants (ng/kg bw/day) via different types of food**
 257 **at e-waste recycling sites in Taizhou, China. n/e – not estimated.**

		Meat					Liver		Egg		TDI
		Fish	Shrimps	Chicken	Duck	Pork	Chicken	Duck	Chicken	Duck	
∑HBCDs	Adult	2.18	n/e	3.67	1.1	n/e	0.28	0.03	2.44	0.71	10.4
	Child	7.49	n/e	9.86	3.95	n/e	1.08	0.11	10.5	3.06	36.1
∑PCBs	Adult	460	1.23	22.3	52.6	n/e	2.7	2.76	51.9	53.5	650
	Child	1590	22.0	60	190	n/e	10.54	10.8	220	230	2340
∑DDTs	Adult	5.06	0.81	2.27	2.78	n/e	1.31	0.21	2.57	11.0	26.0
	Child	17.4	3.16	6.11	9.95	n/e	5.13	0.83	11.0	47.2	100
HCB	Adult	0.55	0.06	2.4	0.72	n/e	0.33	0.08	6.57	1.93	12.6
	Child	1.88	0.22	6.46	2.57	n/e	1.27	0.3	28.3	8.29	49.2
PBEB	Adult	n/e	0.03	n/e	0.04	n/e	0.01	0.02	n/e	n/e	0.10
	Child	n/e	0.12	n/e	0.13	n/e	0.06	0.06	n/e	n/e	0.37
HBB	Adult	0.05	n/e	0.02	0.02	0.05	n/e	n/e	n/e	n/e	0.14
	Child	0.16	n/e	0.05	0.08	0.07	n/e	n/e	n/e	n/e	0.36
EH-TBB	Adult	0.43	0.16	1.15	1.01	4.34	0.23	0.18	0.27	0.25	8.02
	Child	1.49	0.62	3.09	3.62	6.22	0.89	0.71	1.18	0.32	18.1
BEH-TBP	Adult	0.11	0.07	0.42	0.3	1.41	0.07	0.06	0.07	0.08	2.59
	Child	0.37	0.28	1.12	1.08	2.02	0.27	0.25	0.28	0.33	6.00
BTBPE	Adult	0.05	0.04	0.07	0.19	0.61	0.1	0.06	0.16	0.1	1.38
	Child	0.16	0.15	0.18	0.68	0.88	0.38	0.22	0.71	0.44	3.80
DBDPE	Adult	n/e	0.2	n/e	n/e	n/e	n/e	n/e	n/e	n/e	0.20
	Child	n/e	0.77	n/e	n/e	n/e	n/e	n/e	n/e	n/e	0.77

258

259 NBFRs considered in our study. For HBB, our exposure estimates are reassuringly well below
260 the U.S. Environmental Protection Agency's (US EPA 1988) reference dose (RfD) of 2000
261 ng/kg bw/day.

262

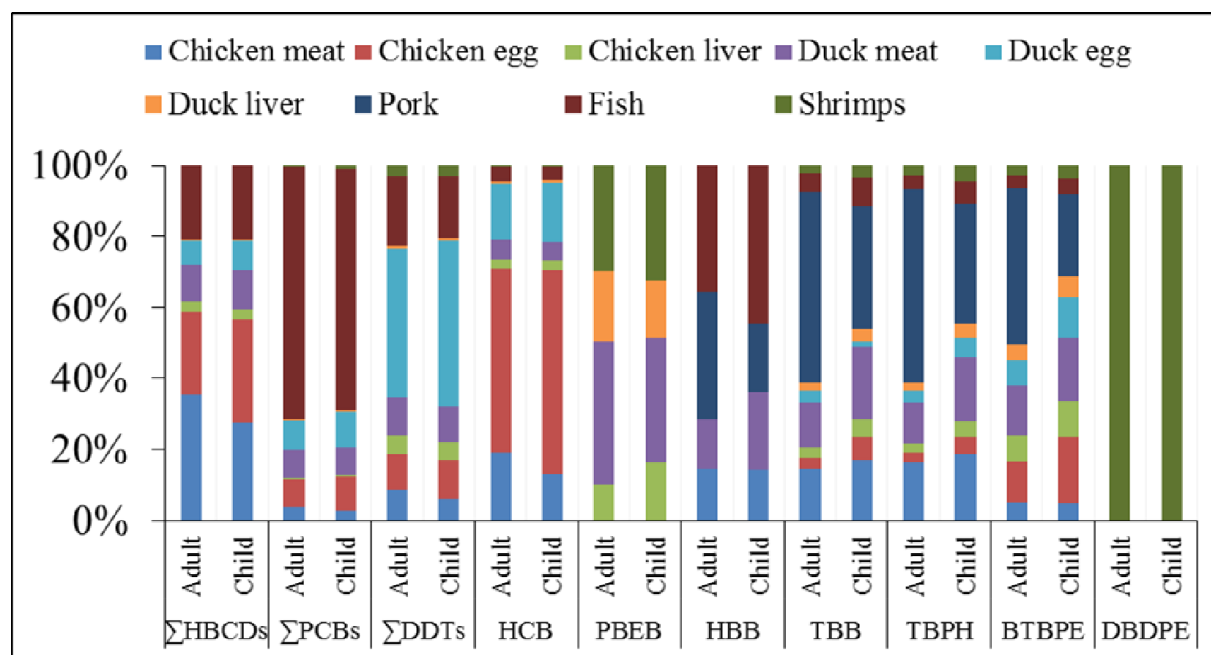
263 3.2.2 HBCD

264 As with the NBFRs, we are unaware of any HBLV of legislative standing for HBCD. However,
265 we were able to compare our dietary exposure estimate with those reported in other studies.

266 Estimated dietary exposure of both adults and children to Σ HBCD in our study (10.4 and 36.1
267 ng/kg bw/day) was significantly higher than reported previously for a Swedish market basket
268 study (0.14 ng/kg bw/day) (Törnkvist et al. 2011), as well as for the Belgian population (0.37
269 ng/kg bw/day) (Gosciny et al. 2011). Chicken muscle and eggs were the main contributors to
270 Σ HBCDs exposure for both adults and children at e-waste sites in Taizhou accounting for over
271 50% of the dietary exposure combined (Fig. 3). This contrasted with two studies from Sweden
272 (Remberger et al. 2004; Törnkvist et al. 2011) in which fish was the main contributor to dietary
273 intake of HBCD. Such differences likely reflect both differences in local dietary preferences and
274 variations in environmental contamination of the study areas. Our estimate of daily Σ HBCD
275 intake from chicken egg consumption (154 ng/day) was similar to that reported for chicken eggs
276 at e-waste sites in South China (range: 80 – 490 ng/day) (Zheng et al. 2012), but exceeded
277 significantly that reported in Romania (0.02 ng/day) (Dirtu and Covaci 2010).

278 Our exposure estimates for adults and children to Σ HBCD *via* consumption of chicken eggs
279 (2.44 and 10.5 ng/kg bw/day respectively), exceeded significantly those reported for the Belgian
280 population (0.004 ng/kg bw/day) (Gosciny et al. 2011).

281 **Fig. 3. Contributions to total dietary exposures for organic contaminants from different**
 282 **types of food originating from Taizhou, China.**

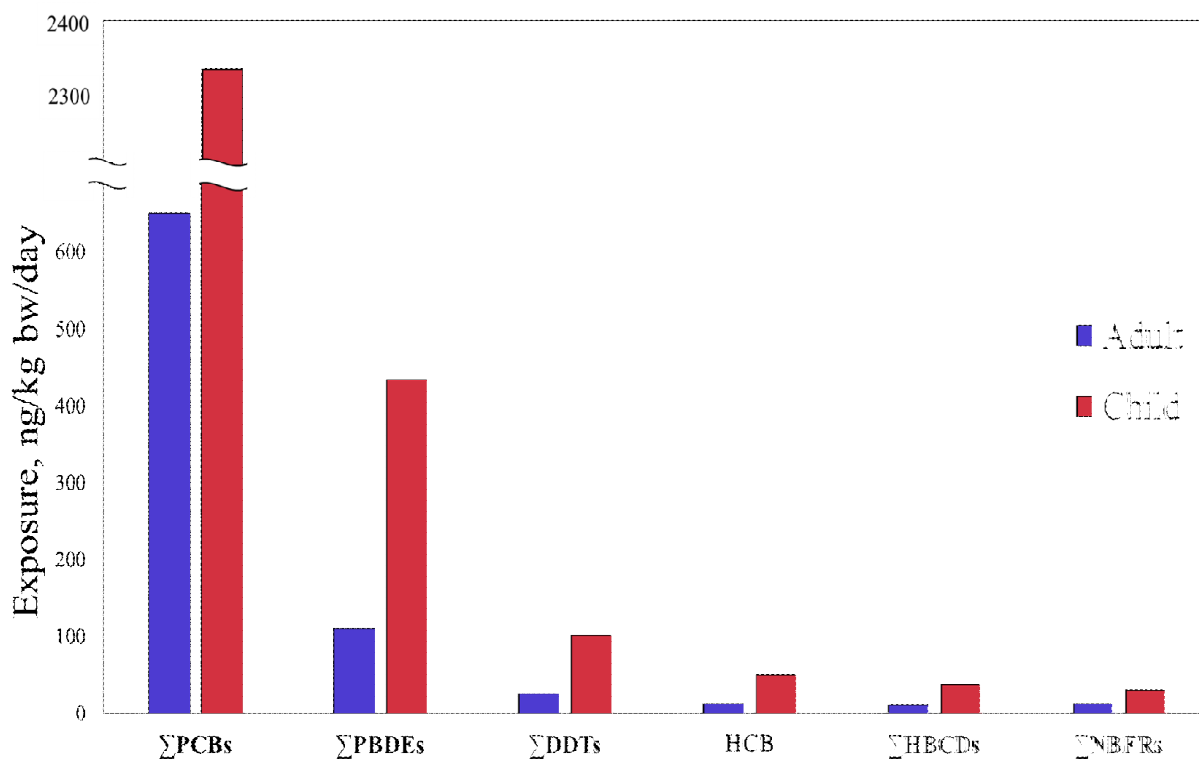


283
 284
 285 Fish contributed a further 21% to our estimate of dietary exposure to Σ HBCD for both adults and
 286 children, at 2.18 and 7.49 ng/g bw/day, respectively; values that exceeded those reported for the
 287 populations of nine Chinese coastal cities (range: 0.004 – 1.00 ng/kg bw/day) (Xia et al. 2011).
 288 Interestingly, while chicken muscle and eggs were the main contributors to Σ HBCD exposure
 289 (adults: 35 and 23 % respectively), duck muscle and eggs contributed greatly to Σ PBDE
 290 exposure (adults: 56 and 3 % respectively) as was previously reported (Labunska et al. 2014).
 291 Our estimate also exceeded that of rural residents of Tianjin, China (1.64 ng/kg bw/day) but was
 292 similar to that reported for urban residents (6.16 ng/kg bw/day) (Zhang et al. 2013). In
 293 comparison to other countries, our estimate of exposure to Σ HBCDs *via* fish consumption was in
 294 line with the average estimate for Japanese adults (range: 1.3 – 3.7 ng/g bw/day) (Nakagawa et

295 al. 2010). The contribution of duck muscle and eggs to Σ HBCDs exposure was lower than that
296 from chicken muscle and eggs. Consistent with other studies (e.g. Goscinnny et al. 2011), of the
297 avian tissues studied, liver made the lowest contribution to our estimated dietary intake of
298 Σ HBCD.

299

300 **Fig. 4. Estimated total dietary exposures to PCBs, DDTs, HCB, HBCDs, NBFRs, and**
301 **previously reported data on PBDEs (Labunska et al. 2014).**



302

303

304 3.2.3 PCBs

305 Dietary exposure to PCBs (650 and 2340 ng/kg Σ PCBs bw/day for adults and children

306 respectively) was the highest of all of the contaminants considered in this study and exceeded

307 that for PBDEs in our previous study of the individual samples from which the pooled samples
308 analysed here were taken (see Fig.4). Moreover, it exceeded substantially the Minimal Risk
309 Levels (MRL) for Σ PCBs of 20 ng/kg bw/day derived by the Agency for Toxic Substances &
310 Disease Registry (ATSDR 2000). Fish was the major contributor to total dietary exposure to
311 PCBs (Fig. 3) followed by eggs (both duck and chicken) and duck muscle. Additionally, we
312 estimated dietary intake of the four DL-PCBs measured in our study (i.e. PCB-105, 118, 156 and
313 167) based on their dioxin-like toxicity equivalence (TEQ) (Van den Berg et al. 2006; EU 2011)
314 (see Table 5). While adult intake (2.83 pg TEQ/kg bw/day) was within the range of tolerable
315 daily intake (TDI) for dioxin-like chemicals of 1 – 4 pg TEQ/kg bw/day established by the
316 World Health Organisation (WHO) (van Leeuwen et al. 2000); estimated intake for children at
317 10.22 pg TEQ/kg bw/day was 2.5 times the WHO TDI upper limit for all 12 DL-PCBs
318 combined. Moreover, DL-PCBs-TEQ monthly intakes in our study (84.9 and 307 pg TEQ/kg
319 bw/month for adults and children respectively) exceeded the provisional tolerable monthly intake
320 (PTMI) of 70 pg TEQ/kg bw, which was established by Joint FAO / WHO Expert Committee on
321 Food Additives (JECFA 2002).

322 We compared our estimates with those reported elsewhere. Specifically, a study at an e-waste
323 recycling site in Luqiao, Eastern China (Song et al. 2011) reported the estimated dietary intake of
324 adults to PCDD/Fs and DL-PCBs combined to be 402 pg TEQ/kg bw/month, *via* consumption of
325 six types of local food (rice, vegetable, chicken, chicken eggs, duck, and crucian carp). Although
326 this exceeded our exposure estimate, it is important to emphasise that our DL-PCB-TEQ
327 exposure estimates are based on exposure to only 4 out of the 12 PCB congeners deemed to
328 display dioxin-like toxicity. Moreover, the exposure estimates of Song et al (2011) are calculated
329 using WHO-1988 TEFs (Van den Berg et al. 1998), compared to the current re-evaluated WHO-

330 2005 TEFs (Van den Berg et al. 2006) used in our study. The use of the WHO-1998 TEFs to our
 331 data raises our estimated monthly dietary intake of DL-PCBs to 332 and 1197 pg TEQ/kg
 332 bw/month for adults and children respectively. Our data thus suggest dietary exposure to PCBs
 333 and, potentially, other related contaminants in the study area, is of substantial health concern,
 334 particularly for children.

335

336 **Table 5. Concentrations of four DL-PCBs (pg TEQ/g ww) and dietary intake (pg TEQ/kg**
 337 **bw/day) by adults and children from e-waste recycling sites in Eastern China.**

	Concentrations (pg TEQ/g ww)					Intake, pg TEQ/kg bw/day	
	PCB 105	PCB 118	PCB 156	PCB 167	Σ PCBs	Child	Adult
Fish	1.16	2.79	0.24	0.13	4.32	5.89	1.71
Shrimps	0.02	0.06	0.00	0.00	0.09	0.12	0.03
Chicken meat	0.14	0.34	0.03	0.03	0.54	0.37	0.14
Duck meat	0.41	1.05	0.10	0.05	1.62	1.10	0.31
Chicken Liver	0.04	0.10	0.01	0.00	0.16	0.05	0.01
Duck Liver	0.06	0.11	0.01	0.00	0.18	0.06	0.02
Chicken Egg	0.45	0.20	0.03	0.06	0.74	1.51	0.35
Duck Egg	0.34	0.14	0.02	0.03	0.54	1.11	0.26
Total						10.22	2.83

338

339 3.2.4 Organochlorine pesticides

340 The highest exposure to Σ DDTs in our study was via consumption of duck eggs (11.0 and 47.2
 341 ng/kg bw/day for adults and children respectively) followed by fish (5.1 and 17.4 ng/kg bw/day
 342 for adults and children respectively) (Table 3). Consumption of these two products accounted for
 343 over 60% of dietary exposure to Σ DDTs for both adults and children (Fig. 3). Exposure to
 344 Σ DDTs *via* fish in our study was similar to that reported for adults in South China (range: 5.74 –
 345 5.84 ng/kg bw/day) (Guo et al. 2010), higher than that reported in another recent study (Man et

346 al. 2013) around e-waste sites in Taizhou (3.39 ng/kg bw/day), but lower than that in Guiyu (6.88
347 ng/kg bw/day).

348 Exposure to Σ DDTs *via* consumption of chicken and duck muscle as well as chicken eggs in our
349 study was between 2.27 and 2.78 ng/kg bw/day and 6.11 and 11.04 ng/kg bw/day for adults and
350 children, respectively. Exposure of adults to Σ DDTs via eating chicken eggs in our study (2.57
351 ng/kg bw/day) was similar to the 2.35 ng/kg bw/day reported for Taizhou previously (Man et al.
352 2013). However, exposure to Σ DDTs via consumption of duck eggs in our study was over four-
353 fold that received via eating chicken eggs. Total dietary exposure to Σ DDTs in our study (11.0
354 and 47.2 ng/kg bw/day for adults and children respectively) exceeded substantially that reported
355 for 12 provinces in China (range: 0.008 – 0.13 ng/kg bw/day) (Zhou et al. 2012). In contrast, our
356 exposure estimates fell well below both the provisional tolerable daily intake (PTDI) for DDT of
357 0.01 mg/kg bw/day derived by the Joint FAO/WHO Meeting on Pesticide Residues (FAO/WHO
358 2001), and the WHO's proposed acceptable daily intake of 20000 ng Σ DDTs/kg bw/day. While
359 we did not expect e-waste treatment to represent an important source of DDTs to the local
360 environment, our study nevertheless reveals exposures well above those reported elsewhere in
361 China, and moreover demonstrates that duck eggs are an important pathway of human exposure
362 to DDT and its metabolites. This is important as a recent study attributed elevated levels of
363 DDTs in milk, placenta and hair of mothers from Taizhou to high seafood intake, but did not
364 mention consumption of duck eggs (Man et al. 2014).

365 Total dietary exposure to HCB (12.6 and 49.2 ng/kg bw/day for adults and children respectively)
366 (Table 3) was well within the corresponding US EPA RfD of 800 ng/kg bw/day (IRIS 1991).
367 However, as with DDTs, dietary exposures in our study exceeded substantially those reported
368 elsewhere. For example, human adult exposures to HCB were: in Catalonia, Spain, 1.0 ng/kg

369 bw/day (Martí-Cid et al. 2008), in Northwest Russia, 3.1 ng/kg bw/day (Polder et al. 2010), and
370 in 12 provinces in China, 9 ng/kg bw/day on average (Zhou et al. 2012). Consumption of chicken
371 eggs (6.57 and 28.3 ng/kg bw/day for adults and children) and chicken muscle (2.4 and 6.46
372 ng/kg bw/day for adults and children) combined contributed around 70% of our estimates of total
373 dietary exposure to both adults and children (Fig. 3) followed by duck eggs and muscle (about
374 20% total exposure). Chicken eggs from Wenling showed the highest HCB concentration on
375 both a lipid (197.4 ng/g) and wet weight basis (23.5 ng/g), with the concentration detected falling
376 just below the 200 ng/g lw maximum residue level promulgated by the EC (EEC 1986).

377 This study reveals substantially elevated concentrations of a wide range of OHCs in foods
378 originating from an area where unregulated e-waste treatment is practised. Even though only 4 of
379 the 12 designated dioxin-like PCB congeners were measured in this study; concentrations of
380 dioxin-like PCBs expressed as TEQ, exceeded the EU limit value in 6 out of the 8 food groups
381 studied. Combined with our earlier report based on the same samples, of substantially elevated
382 exposure of young children to PBDEs 47 and 99; the data reported here add to growing concerns
383 about the adverse health impacts of unregulated e-waste treatment. Efforts to discontinue such
384 activities should be prioritised in order to minimise further release of toxic chemicals into the
385 environment. Meanwhile, educational programs are recommended in areas where rudimentary e-
386 waste treatment is conducted, in order to help the local population to minimise their exposure to
387 chemical contaminants – for example, by restricting access of their livestock to e-waste
388 contaminated land.

389

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397

398 **5 Supplementary Materials**

399 The following materials are available as supplementary information: a description of pooled
400 samples (Table S1); locations of sampling sites (Fig. S1); limits of quantification (LOQs) of
401 target compounds (Table S2); comparison of reference values for α - and β -HBCD from NIST
402 SRM 2794A (Organics in Freeze-Dried Mussel Tissue) and values obtained in our study (Fig.
403 S2); concentrations of HBCD isomers in pooled food samples from e-waste and control sites
404 (Table S3); HBCD diastereomer profiles in selected individual food samples (Fig S3);
405 concentrations of PCBs and DDTs in pooled food samples (Table S4); PCB and DDT
406 congener/isomer profiles in pooled food samples (Fig. S4 and Fig S5); and a Table summarising
407 published data and data used in this study on daily consumption rates for food products (Table
408 S5).

409

410 6 References

- 411 Abdallah M, Uchea C. Enantioselective Biotransformation of Hexabromocyclododecane by in
412 Vitro Rat and Trout Hepatic Sub-Cellular Fractions. *Environ Sci Technol.*
413 2014;48(5):2732–40.
- 414 Abdallah MA, Harrad S. Tetrabromobisphenol-A, hexabromocyclododecane and its degradation
415 products in UK human milk: relationship to external exposure. *Environ Int.*
416 2011;37(2):443–8.
- 417 Abdallah MA-E, Tilston E, Harrad S, Collins C. In vitro assessment of the bioaccessibility of
418 brominated flame retardants in indoor dust using a colon extended model of the human
419 gastrointestinal tract. *J Environ Monit.* 2012;14(12):3276–83.
- 420 Alae M, Arias P, Sjödin A, Bergman A. An overview of commercially used brominated flame
421 retardants, their applications, their use patterns in different countries/regions and possible
422 modes of release. *Environ Int.* 2003;29(6):683–9.
- 423 ATSDR. Agency for Toxic Substances & Disease Registry. Toxicological profile for PCBs
424 [Internet]. 2000 [cited 2014 Aug 12]. Available from:
425 <http://www.atsdr.cdc.gov/toxprofiles/tp.asp?id=142&tid=26>
- 426 Van den Berg M, Birnbaum L, Bosveld AT, Brunström B, Cook P, Feeley M, et al. Toxic
427 equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environ*
428 *Health Perspect.* 1998;106(12):775–92.
- 429 Van den Berg M, Birnbaum LS, Denison M, De Vito M, Farland W, Feeley M, et al. The 2005
430 World Health Organization reevaluation of human and Mammalian toxic equivalency
431 factors for dioxins and dioxin-like compounds. *Toxicol Sci.* 2006;93(2):223–41.
- 432 Chan JKY, Man YB, Wu SC, Wong MH. Dietary intake of PBDEs of residents at two major
433 electronic waste recycling sites in China. *Sci Total Environ.* 2013;463-464:1138–46.
- 434 Covaci A, Gerecke AC, Law RJ, Voorspoels S, Kohler M, Heeb N V, et al.
435 Hexabromocyclododecanes (HBCDs) in the environment and humans: a review. *Environ*
436 *Sci Technol.* 2006;40(12):3679–88.
- 437 Covaci A, Gheorghe A, Schepens P. Distribution of organochlorine pesticides, polychlorinated
438 biphenyls and alpha-HCH enantiomers in pork tissues. *Chemosphere.* 2004;56(8):757–66.
- 439 Covaci A, Harrad S, Abdallah MA-E, Ali N, Law RJ, Herzke D, et al. Novel brominated flame
440 retardants: a review of their analysis, environmental fate and behaviour. *Environ Int.*
441 2011;37(2):532–56.
- 442 Dirtu AC, Covaci A. Estimation of daily intake of organohalogenated contaminants from food
443 consumption and indoor dust ingestion in Romania. *Environ Sci Technol. American*
444 *Chemical Society;* 2010;44(16):6297–304.
- 445 EC. Commission Decision of 3 December 1999 on protective measures with regard to
446 contamination by dioxins of certain products of porcine and poultry origin intended for
447 human or animal consumption. *OJ.* 1999;L310:62–70.
- 448 EC. Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels
449 for certain contaminants in foodstuffs. *OJ.* 2006;L364:5–24.
- 450 EEC. Council Directive of 24 July 1986 on the fixing of maximum levels for pesticide residues
451 in and on foodstuffs of animal origin (86/363/EEC). *OJ.* 1986;L221:43–7.

452 Van den Eede N, Dirtu AC, Ali N, Neels H, Covaci A. Multi-residue method for the
453 determination of brominated and organophosphate flame retardants in indoor dust. *Talanta*.
454 2012;89:292–300.

455 EFSA. Scientific Opinion on Emerging and Novel Brominated Flame Retardants (BFRs) in
456 Food. EFSA Panel on Contaminants in the Food Chain. *EFSA J.* 2012;10(10):2908.

457 EU. Commission Regulation (EU) No 1259/2011 of 2 December 2011 amending Regulation (EC)
458 No 1881/2006 as regards maximum levels for dioxins, dioxin-like PCBs and non dioxin-
459 like PCBs in foodstuffs. *OJ.* 2011;L320:18–23.

460 Eulaers I, Jaspers VLB, Pinxten R, Covaci A, Eens M. Legacy and current-use brominated flame
461 retardants in the Barn Owl. *Sci Total Environ.* 2014;472:454–62.

462 FAO/WHO. Pesticide residues in food - 2000. Evaluations. Part II — Toxicology. Joint
463 FAO/WHO Meeting on Pesticide Residues. [Internet]. 2001. Available from:
464 http://www.inchem.org/documents/jmpr/jmpmono/v00pr03.htm#_00032000

465 Fernandes S, Smith F, Petch R, Panton S, Carr M, Rose M. Investigation of the Occurrence of
466 Brominated Contaminants in Selected Foods. *FD 08/07.* 2009.

467 Fournier A, Feidt C, Marchand P, Venisseau A, Le Bizec B, Sellier N, et al. Kinetic study of
468 gamma-hexabromocyclododecane orally given to laying hens (*Gallus domesticus*) “Transfer
469 of HBCD in laying hens.” *Environ Sci Pollut Res.* 2012;19(2):440–7.

470 Frazzoli C, Orisakwe OE, Dragone R, Mantovani A. Diagnostic health risk assessment of
471 electronic waste on the general population in developing countries’ scenarios. *Environ*
472 *Impact Assess Rev.* 2010;30(6):388–99.

473 Fu J, Mai B, Sheng G, Zhang G, Wang X, Peng P, et al. Persistent organic pollutants in
474 environment of the Pearl River Delta, China: an overview. *Chemosphere.* 2003;52(9):1411–
475 22.

476 Fu Z, Wu F, Amarasiriwardena D, Mo C, Liu B, Zhu J, et al. Antimony, arsenic and mercury in
477 the aquatic environment and fish in a large antimony mining area in Hunan, China. *Sci*
478 *Total Environ.* Elsevier B.V.; 2010;408(16):3403–10.

479 Gosciny S, Vandevijvere S, Maleki M, Van Overmeire I, Winald I, Hanot V, et al. Dietary
480 intake of hexabromocyclododecane diastereoisomers (α -, β -, and γ -HBCD) in the Belgian
481 adult population. *Chemosphere.* 2011;84(3):279–88.

482 Guo J, Wu F, Shen R, Zeng EY. Dietary intake and potential health risk of DDTs and PBDEs via
483 seafood consumption in South China. *Ecotoxicol Environ Saf.* 2010;73(7):1812–9.

484 Guo J-Y, Wu F-C, Zhang L, Liao H-Q, Tang Z, Zheng C, et al. Characteristics of DDTs in fish
485 from Lake Taihu: an indicator of continual DDTs input in China. *Sci Total Environ.*
486 2012;437:196–9.

487 He M-J, Luo X-J, Yu L-H, Liu J, Zhang X-L, Chen S-J, et al. Tetrabromobisphenol-A and
488 hexabromocyclododecane in birds from an e-waste region in South China: influence of diet
489 on diastereoisomer- and enantiomer-specific distribution and trophodynamics. *Environ Sci*
490 *Technol.* 2010;44(15):5748–54.

491 He M-J, Luo X-J, Yu L-H, Wu J-P, Chen S-J, Mai B-X. Diastereoisomer and enantiomer-specific
492 profiles of hexabromocyclododecane and tetrabromobisphenol A in an aquatic environment
493 in a highly industrialized area, South China: vertical profile, phase partition, and
494 bioaccumulation. *Environ Pollut.* 2013;179:105–10.

495 IRIS. Hexachlorobenzene (CASRN 118-74-1). US EPA Integrated Risk Information System
496 [Internet]. 1991. Available from: <http://www.epa.gov/iris/subst/0374.htm>

497 Janak K, Covaci A, Voorspoels S, Becher G. Hexabromocyclododecane in marine species from
498 the Western Scheldt Estuary: diastereoisomer- and enantiomer-specific accumulation.
499 *Environ Sci Technol.* 2005;39(7):1987–94.

500 Janak K, Sellstrom U, Johansson AK, Becher G, de Wit CA, Lindberg P, et al. Enantiomer-
501 specific accumulation of hexabromocyclododecanes in eggs of predatory birds.
502 *Chemosphere.* 2008;73(1 Suppl):S193–200.

503 JECFA. Evaluation of certain food additives and contaminants: Fifty-seventh report of the Joint
504 FAO/WHO Expert Committee on Food Additives. Geneva; 2002 p. 186pp.

505 Köppen R, Becker R, Esslinger S, Nehls I. Enantiomer-specific analysis of
506 hexabromocyclododecane in fish from Etnefjorden (Norway). *Chemosphere.*
507 2010;80(10):1241–5.

508 Labunska I, Harrad S, Santillo D, Johnston P, Brigden K. Levels and distribution of
509 polybrominated diphenyl ethers in soil, sediment and dust samples collected from various
510 electronic waste recycling sites within Guiyu town, southern China. *Environ Sci Process*
511 *Impacts.* 2013 a;15(2):503–11.

512 Labunska I, Harrad S, Santillo D, Johnston P, Yun L. Domestic duck eggs: an important pathway
513 of human exposure to PBDEs around e-waste and scrap metal processing areas in Eastern
514 China. *Environ Sci Technol.* 2013 b;47(16):9258–66.

515 Labunska I, Harrad S, Wang M, Santillo D, Johnston P. Human dietary exposure to PBDEs
516 around e-waste recycling sites in Eastern China. *Environ Sci Technol.* American Chemical
517 Society; 2014;48(10):5555–64.

518 Van Leeuwen FXR, Feeley M, Schrenk D, Larsen JC, Farland W, Younes M. Dioxins: WHO's
519 tolerable daily intake (TDI) revisited. *Chemosphere.* 2000;40(9-11):1095–101.

520 Liu M, Huang B, Bi X, Ren Z, Sheng G, Fu J. Heavy metals and organic compounds
521 contamination in soil from an e-waste region in South China. *Environ Sci Process Impacts.*
522 2013;15(5):919–29.

523 Malarvannan G, Belpaire C, Geeraerts C, Eulaers I, Neels H, Covaci A. Assessment of persistent
524 brominated and chlorinated organic contaminants in the European eel (*Anguilla anguilla*) in
525 Flanders, Belgium: Levels, profiles and health risk. *Sci Total Environ.* 2014;482-483C:222–
526 33.

527 Man YB, Chan JKY, Wang HS, Wu SC, Wong MH. DDTs in mothers' milk, placenta and hair,
528 and health risk assessment for infants at two coastal and inland cities in China. *Environ Int.*
529 2014;65C:73–82.

530 Man YB, Chan JKY, Wu SC, Wong CKC, Wong MH. Dietary exposure to DDTs in two coastal
531 cities and an inland city in China. *Sci Total Environ.* 2013;463-464:264–73.

532 Martí-Cid R, Llobet JM, Castell V, Domingo JL. Human dietary exposure to hexachlorobenzene
533 in Catalonia, Spain. *J Food Prot.* 2008;71(10):2148–52.

534 Marvin CH, Tomy GT, Armitage JM, Arnot JA, McCarty L, Covaci A, et al.
535 Hexabromocyclododecane: Current Understanding of Chemistry, Environmental Fate and
536 Toxicology and Implications for Global Management. *Environ Sci Technol.*
537 2011;45(20):8613–23.

538 MHC. Chinese Health Statistics 2012. Ministry of Health of China. 2012.

539 MOH. National food safety standard - The Limits of Pollutants in Foods. GB 2762-2012 (2012).
540 Published by Ministry of Health of the People's Republic of China. 2012.

541 Nakagawa R, Murata S, Ashizuka Y, Shintani Y, Hori T, Tsutsumi T. Hexabromocyclododecane
542 determination in seafood samples collected from Japanese coastal areas. *Chemosphere*.
543 2010;81(4):445–52.

544 Ni H-G, Ding C, Lu S-Y, Yin X-L, Samuel SO. Food as a main route of adult exposure to
545 PBDEs in Shenzhen, China. *Sci Total Environ*. 2012;437:10–4.

546 Ni K, Lu Y, Wang T, Kannan K, Gosens J, Xu L, et al. A review of human exposure to
547 polybrominated diphenyl ethers (PBDEs) in China. *Int J Hyg Environ Health*.
548 2013;216(6):607–23.

549 Polder A, Savinova TN, Tkachev A, Løken KB, Odland JO, Skaare JU. Levels and patterns of
550 Persistent Organic Pollutants (POPS) in selected food items from Northwest Russia (1998-
551 2002) and implications for dietary exposure. *Sci Total Environ*. 2010;408(22):5352–61.

552 Qiu X, Zhu T, Yao B, Hu J, Hu S. Contribution of Dicofol to the Current DDT Pollution in
553 China. *Environ Sci Technol*. American Chemical Society; 2005;39(12):4385–90.

554 Rajaei F, Bahramifar N, Esmaili Sari A, Ghasempouri SM, Savabieasfahani M. PCBs and
555 organochlorine pesticides in ducks of Fereydoon-kenar wildlife refuge in Iran. *Bull Environ*
556 *Contam Toxicol*. 2010;84(5):577–81.

557 Remberger M, Sternbeck J, Palm A, Kaj L, Strömberg K, Brorström-Lundén E. The
558 environmental occurrence of hexabromocyclododecane in Sweden. *Chemosphere*.
559 2004;54(1):9–21.

560 Robinson BH. E-waste: an assessment of global production and environmental impacts. *Sci Total*
561 *Environ*. 2009;408(2):183–91.

562 Roosens L, Dirtu AC, Goemans G, Belpaire C, Gheorghe A, Neels H, et al. Brominated flame
563 retardants and polychlorinated biphenyls in fish from the river Scheldt, Belgium. *Environ*
564 *Int*. 2008;34(7):976–83.

565 Saunders DM V, Higley EB, Hecker M, Mankidy R, Giesy JP. In vitro endocrine disruption and
566 TCDD-like effects of three novel brominated flame retardants: TBPH, TBB, & TBCO.
567 *Toxicol Lett*. 2013;223(2):252–9.

568 Shang H, Wang P, Wang T, Wang Y, Zhang H, Fu J, et al. Bioaccumulation of PCDD/Fs, PCBs
569 and PBDEs by earthworms in field soils of an E-waste dismantling area in China. *Environ*
570 *Int*. 2013;54:50–8.

571 Shen C, Chen Y, Huang S, Wang Z, Yu C, Qiao M, et al. Dioxin-like compounds in agricultural
572 soils near e-waste recycling sites from Taizhou area, China: chemical and bioanalytical
573 characterization. *Environ Int*. 2009;35:50–5.

574 Song Y, Wu N, Han J, Shen H, Tan Y, Ding G, et al. Levels of PCDD/Fs and DL-PCBs in
575 selected foods and estimated dietary intake for the local residents of Luqiao and Yuhang in
576 Zhejiang, China. *Chemosphere*. 2011;85(3):329–34.

577 Stieger G, Scheringer M, Ng CA, Hungerbühler K. Assessing the persistence, bioaccumulation
578 potential and toxicity of brominated flame retardants: Data availability and quality for 36
579 alternative brominated flame retardants. *Chemosphere*. 2014 Mar 20; *In press*.

580 Sun Y-X, Luo X-J, Mo L, He M-J, Zhang Q, Chen S-J, et al. Hexabromocyclododecane in
581 terrestrial passerine birds from e-waste, urban and rural locations in the Pearl River Delta,

582 South China: levels, biomagnification, diastereoisomer- and enantiomer-specific
583 accumulation. *Environ Pollut.* 2012;171:191–8.

584 Szabo DT, Diliberto JJ, Hakk H, Huwe JK, Birnbaum LS. Toxicokinetics of the Flame Retardant
585 Hexabromocyclododecane Alpha: Effect of Dose, Timing, Route, Repeated Exposure, and
586 Metabolism. *Toxicol Sci.* 2011 a;121(2):234–44.

587 Szabo DT, Diliberto JJ, Huwe JK, Birnbaum LS. Differences in Tissue Distribution of HBCD
588 Alpha and Gamma between Adult and Developing Mice. *Toxicol Sci.* 2011 b;123(1):256–
589 63.

590 Tang X, Shen C, Shi D, Cheema S a, Khan MI, Zhang C, et al. Heavy metal and persistent
591 organic compound contamination in soil from Wenling: an emerging e-waste recycling city
592 in Taizhou area, China. *J Hazard Mater.* 2010;173(1-3):653–60.

593 Törnkvist A, Glynn A, Aune M, Darnerud PO, Ankarberg EH. PCDD/F, PCB, PBDE, HBCD
594 and chlorinated pesticides in a Swedish market basket from 2005--levels and dietary intake
595 estimations. *Chemosphere.* 2011;83(2):193–9.

596 US EPA. Hexabromobenzene; CASRN 87-82-1; 03/01/1988 [Internet]. 1988 [cited 2014 Aug
597 12]. Available from: <http://www.epa.gov/iris/subst/0161.htm>

598 Wen S, Yang F-X, Gong Y, Zhang X-L, Hui Y, Li J-G, et al. Elevated levels of urinary 8-
599 hydroxy-2'-deoxyguanosine in male electrical and electronic equipment dismantling
600 workers exposed to high concentrations of polychlorinated dibenzo-p-dioxins and
601 dibenzofurans, polybrominated diphenyl ethers, and polychlorinat. *Environ Sci Technol.*
602 2008;42(11):4202–7.

603 Wu J-P, Guan Y-T, Zhang Y, Luo X-J, Zhi H, Chen S-J, et al. Several current-use, non-PBDE
604 brominated flame retardants are highly bioaccumulative: evidence from field determined
605 bioaccumulation factors. *Environ Int.* 2011;37(1):210–5.

606 Xia C, Lam JCW, Wu X, Sun L, Xie Z, Lam PKS. Hexabromocyclododecanes (HBCDs) in
607 marine fishes along the Chinese coastline. *Chemosphere.* 2011;82(11):1662–8.

608 Xing GH, Chan JKY, Leung AOW, Wu SC, Wong MH. Environmental impact and human
609 exposure to PCBs in Guiyu, an electronic waste recycling site in China. *Environ Int.*
610 2009;35(1):76–82.

611 Xing GH, Liang Y, Chen LX, Wu SC, Wong MH. Exposure to PCBs, through inhalation, dermal
612 contact and dust ingestion at Taizhou, China--a major site for recycling transformers.
613 *Chemosphere.* 2011;83(4):605–11.

614 Xing GH, Wu SC, Wong MH. Dietary exposure to PCBs based on food consumption survey and
615 food basket analysis at Taizhou, China--the world's major site for recycling transformers.
616 *Chemosphere.* 2010;81(10):1239–44.

617 Xu X, Yang H, Chen A, Zhou Y, Wu K, Liu J, et al. Birth outcomes related to informal e-waste
618 recycling in Guiyu, China. *Reprod Toxicol.* 2012;33(1):94–8.

619 Yu Y, Li C, Zhang X, Zhang X, Pang Y, Zhang S, et al. Route-specific daily uptake of
620 organochlorine pesticides in food, dust, and air by Shanghai residents, China. *Environ Int.*
621 2012;50:31–7.

622 Yuan J, Chen L, Chen D, Guo H, Bi X, Ju Y, et al. Elevated serum polybrominated diphenyl
623 ethers and thyroid-stimulating hormone associated with lymphocytic micronuclei in
624 Chinese workers from an E-waste dismantling site. *Environ Sci Technol.* 2008;42(6):2195–
625 200.

- 626 Zhang W-H, Wu Y-X, Simonnot MO. Soil Contamination due to E-Waste Disposal and
627 Recycling Activities: A Review with Special Focus on China. *Pedosphere*. 2012
628 Aug;22(4):434–55.
- 629 Zhang X-L, Luo X-J, Liu H-Y, Yu L-H, Chen S-J, Mai B-X. Bioaccumulation of several
630 brominated flame retardants and dechlorane plus in waterbirds from an e-waste recycling
631 region in South China: associated with trophic level and diet sources. *Environ Sci Technol*.
632 American Chemical Society; 2011 Jan 15;45(2):400–5.
- 633 Zhang Y, Sun H, Liu F, Dai Y, Qin X, Ruan Y, et al. Hexabromocyclododecanes in limnic and
634 marine organisms and terrestrial plants from Tianjin, China: diastereomer- and enantiomer-
635 specific profiles, biomagnification, and human exposure. *Chemosphere*. 2013
636 Nov;93(8):1561–8.
- 637 Zheng L, Wu K, Li Y, Qi Z, Han D, Zhang B, et al. Blood lead and cadmium levels and relevant
638 factors among children from an e-waste recycling town in China. *Environ Res*. 2008
639 Sep;108(1):15–20.
- 640 Zheng X-B, Wu J-P, Luo X-J, Zeng Y-H, She Y-Z, Mai B-X. Halogenated flame retardants in
641 home-produced eggs from an electronic waste recycling region in South China: levels,
642 composition profiles, and human dietary exposure assessment. *Environ Int*. 2012 Sep
643 15;45:122–8.
- 644 Zhou P, Zhao Y, Li J, Wu G, Zhang L, Liu Q, et al. Dietary exposure to persistent
645 organochlorine pesticides in 2007 Chinese total diet study. *Environ Int*. 2012 Jul;42:152–9.
646