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Elevated Serum Polybrominated Diphenyl Ethers and Thyroid-Stimulating Hormone Associated with Lymphocytic Micronuclei in Chinese Workers from an E-Waste Dismantling Site

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In this study, we recruited 49 subjects from one village close to an electronic waste (e-waste) site (exposed group) and another located 50 km away from the e-waste site (control group). We found that serum levels of polybrominated diphenyl ethers (median PBDEs, 382 ng/g lipid weight; range, 77–8452 ng/g lipid weight) and thyroid-stimulating hormone (median TSH, 1.79 μIU/mL; range, 0.38–9.03 μIU/mL) and frequencies of micronucleated binucleated cells (MNed BNC; median, 5‰; range, 0–96‰) were significantly higher in the exposed group than in the control group (158 ng/g, range of 18–436 ng/g, and $p < 0.05$; 1.15 μIU/mL, range of 0.48–2.09, and $p < 0.01$; and 0‰, range of 0–5‰, and $p < 0.01$, respectively). A history of working with e-waste was significantly associated with increased MNed BNC frequencies (odds ratio (OR), 38.85; 95% confidence interval (CI) = 1–1358.71, $p = 0.044$), independent of years of local residence, a perceived risk factor. However, there was no association between PBDEs exposure and oxidative DNA damage. Therefore, the exposure to PBDEs at the e-waste site may have an effect on the levels of TSH and genetoxic damage among these workers, but this needs to be validated in large studies.

Introduction

With newer generations of technology and constant upgrade of the products, the quantities of discarded electronic equipment (so-called electronic wastes or e-wastes) increase rapidly. Certain chemicals such as polybrominated diphenyl ethers (PBDEs) from the e-wastes can pollute the surrounding environment of e-waste dumping sites and thereafter may cause certain adverse impacts on human health (*1–3*).

PBDEs as flame retardants are widely used in various consumer electronics and plastics (*4*). There is growing evidence that PBDEs persist in the environment (*5*) and accumulate in human tissues (*6, 7*). Recent studies found that PBDEs may cause some adverse effects on humans (*8*) such as thyroid hormone disruption, neurodevelopmental deficits, and potential carcinogenesis, due to their structural similarity to thyroid hormones and polychlorinated biphenyls (*9–11*).

Serum thyroid-stimulating hormone (TSH) is an important parameter of thyroid functions, because it can affect the level of other thyroid-stimulating hormones (such as free triiodothyronine and free thyroxine) by acting on the hypothalamus and/or the pituitary gland (*12, 13*). Studies in rats found that a short-term exposure to some commercial PBDE mixtures interfered with the thyroid hormone system via upregulation of UGTs (uridinediphosphate-glucuronosyltransferase) (*14, 15*). However, another study at an electronic recycling facility did not find a change of thyroid hormone levels among the workers exposed to PBDEs (*11*).

Oxidative stress in humans is characteristic of free radicals that can attack both DNA bases or deoxyribose residues and other macromolecules, such as lipids or proteins, leading to DNA adduct formation, mutations, or apoptosis (*16*). However, reactive oxygen species (ROS) are prevented by the antioxidant system, including antioxidants (ascorbic acid, glutathione, and tocopherols) in their active forms and ROS-interacting enzymes, such as superoxide dismutase (SOD), peroxidases, and catalases (*17*). Increasing evidence *in vivo* and *in vitro* suggests that oxidative stress etiologically relates to many diseases, including pulmonary diseases, cardiovascular diseases, and cancer (*18, 19*). Therefore, SOD, malondialdehyde (MDA), glutathione peroxidase (GSH-Px), and 8-hydroxy-2′-deoxyguanosine (8-OHdG) are often used to assess the oxidative stress status.

In animals, PBDEs have toxic effects on the sperm functions as a result of PBDE 209-induced oxidative stress (*20*), PBDE metabolites and their parent pollutants may also contribute to contaminant-related stress (*21*), and PBDE congeners may alter thyroid hormone and glutathione metabolism (*22*). In addition, low PBDE concentrations ($\leq 10^9$ M) were genotoxic in human MCF-7 cells (*23*). The highly reactive hydroxyl radical (•OH) is believed to cause genotoxicity (*24, 25*) or oxidative stress (*18, 26, 27*) that threaten the stability of the genome.

There is limited evidence of the impacts of PBDEs on human health. The aims of the present study were to explore possible factors that may influence selected biomarkers for exposure to the e-wastes. We determined whether PBDE exposure was associated with levels of serum PBDEs of the subjects (internal exposure), thyroid-stimulating hormone (host response), and urinary 8-hydroxydeoxyguanosine (8-OHdG) and with frequencies of the cytokinesis-block mi-

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cronucleus (CBMN; biological effects). Levels of 8-OHdG and the CBMN assay are often used to assess the DNA damage (*28–30*).

Materials and Methods

Study Population. For this study, we recruited 23 workers as the exposure group, who manually dismantled and "recycled" electronic goods such as personal computers and mobile phones on a daily basis in a village close to a decade-long e-waste recycling site in southeast China. We recruited another 26 farmers as the control group in a village located 50 km away from the recycling site without other pollutions. These subjects had similar age and sex distributions, social status, and residential areas. Having provided an informed consent, all subjects were interviewed by an occupational physician using a questionnaire including the information about personal medical history, smoking history, alcohol consumption, and occupational history. The study was approved by the Tongji Medical College Ethnics and Human Subject Committee.

Sample Collection. From each subject, we collected a 5 mL venous blood sample with an anticoagulant-free tube for separating the serum by centrifugation, which was stored at −20 °C before further analysis. Another 1 mL venous blood sample was drawn into a heparinized tube for performing the CBMN assay. The urine samples were collected in polypropylene specimen containers and stored at −20 °C until used for further analysis.

PBDEs Levels. The serum PBDE levels were analyzed according to the method described by Thomas et al. (*31*). Briefly, the blood samples were first denatured with hydrochloric acid and propan-2-ol (BDH Laboratory Supplies, Poole, Dorset, U.K.; purity > 99.7%), followed by extraction with a hexane:methyl-*tert*-butyl ether (MTBE; Aldrich, Stenheim, Germany; purity > 99.8%) mixture. Samples were then cleaned using concentrated sulfric acid, followed by gel permeation chromatography (Biobeads S-X3; Bio-Rad Laboratories, Inc., CA). ^{13}C-labeled C_{12}-PCBs served as the recovery standards. The internal standards included $^{13}C_{12}$-labeled CB208, BDE-69, and BDE-181 (Cambridge Isotope Laboratories, Inc., Andover, MA). Serum PBDE levels were analyzed using either gas chromatography–mass spectrometry (GC-MS) or gas chromatography−high-resolution mass spectrometry (GC-HRMS) according to the bromine number of PBDEs.

TSH Levels. A 150 μL serum sample was used to determine the TSH level by the chemiluminescent microparticle immunoassay with the commercial ARCHITECT TSH test kit (Abbott Laboratories, Abbott Park, IL) following the manufacturer's instructions. The results were expressed as μIU/mL.

SOD, MDA, and GSH Levels. To evaluate the oxidative stress status in the subjects, the levels of SOD, MDA, and GSH in serum samples were measured using the commercial SOD, MDA, and GSH test kits (Jiancheng Bioengineering Ltd., Nanjing, China), following the manufacturer's instructions. The results were expressed as U/mL for SOD, mM for MDA, and mg/L for GSH.

Urinary 8-OHdG Levels. The urinary 8-hydroxy-2′-deoxyguanosine levels were examined according to a previously described method (*32*) with some minor modifications. Briefly, urine samples were first centrifuged at 1500*g* at 4 °C for 10 min to remove precipitates; 2 mL of the supernatant liquid was adsorbed on a pretreated C_{18}−OH cartridge (10 mL/500 mg; Bond Elut LRC; Varian, Palo Alto, CA) with first 10 mL of pure methanol, 5 mL of deionized water, and then 10 mL of 0.1 M KH_2PO_4 (pH 6.0), and last with 3 mL of deionized water. The adsorbed substances were eluted by using the pressed air flow, first with 3 mL of 0.1 mM potassium dihydrogen phosphate (KH_2PO_4; pH 6.0), then with 3 mL of 5% methanol solution, and last with 1 mL of pure methanol. Subsequently, each of the eluated samples was evaporated with a vacuum speed at 45 °C for 2 h. The dried sample was dissolved in 1 mL of KH_2PO_4 (pH 6.0) and then was filtrated by Millipore microinjector (0.22 um, Millipore Corp, CA).

Each 20 μL sample was analyzed by a HPLC system with an electrochemical detector (Varian-Prostar model 370; Varian Inc., CA). The detection conditions was as follows: (a) a reverse-phase C_{18} column was used to separate 8-OHdG; (b) 30 mM sodium hydroxide, 12.5 mM citric acid, 25 mM sodium acetate, 10 mM acetic acid, 0.05 mM ethylene diamine tetraacetic acid, and 5% methanol solution were used as the mobile phase. The system was run isocratically at 25 °C at a flow rate of 1.4 mL/min; (c) 0, 28, 70, 140, 350, 700, and 1400 nM of standard solutions of 8-OHdG were used to establish the retention time. The concentrations of 8-OHdG in the samples were interpolated from a log-transformed standard curve. Urinary 8-OHdG levels were subsequently adjusted by urinary creatinine levels, presented as micromoles per mole of creatinine.

CBMN Assay. The CBMN assay is a reliable method to assess genotoxic effects (*28, 33*). In the present study, the CBMN assay was performed according to the method previously described (*34*) with some modifications. Briefly, 0.5 mL of the fresh whole blood was added to 4.5 mL of medium supplemented with 15% fetal calf serum and 100 IU/mL penicillin and the final concentration of 20 μg/mL phytohemagglutinin (Sigma, St. Louis, MO). A 30 μL aliquot of cytochalasin B (1 μg/μL, Sigma) was added at a final concentration of 6 μg/mL after 44 h of incubation at 37 °C. After incubation for 72 h, the cells were centrifuged, fixed, and stained with 10% Giemasa solution. The number of micronucleated binucleated (MNed) cells in 1000 binuclealed lymphocytic cells for each sample was scored.

Statistical Methods. The comparisons of two group means were carried out using the Student's *t* test. Group differences in frequencies of selected variables were evaluated by the Pearson χ^2 test. The comparisons of group biomarkers were analyzed with the Mann–Whitney test. The Pearson correlation coefficient was used to evaluate correlations among the biomarkers. Multivariate logistic regression models were performed to assess the association-selected variables or biomarkers. All statistic tests were two-sided with a significance level of 0.05. All statistical analyses were performed using the SPSS 12.0 package (SPSS, Chicago, IL).

Results

General Characteristics. The demographic characteristics of 49 subjects are shown in Table 1. The 23 workers had a shorter resident time than 26 controls with a median residence time of 3 years (range, 1–12 years) for the exposed group and 40 years (range, 2–45 years) for the control group ($p < 0.01$). However, no differences in other variables, including age, sex, body mass index (BMI), and smoking status, were detected between the control and the exposed group, although the difference in alcohol consumption between the groups was borderline significant ($p = 0.052$; with a power of 0.955).

Clinical Parameters. As shown in Table 2, the median level of total PBDEs was 158 ng/g lipid weight (range, 18–436 ng/g lipid weight) in the control group and 382 ng/g lipid weight (range, 77–8452 ng/g lipid weight) in the exposed group, and the difference was statistically significant ($p < 0.05$). In addition, the median level of serum TSH (median, 1.79 μIU/mL; range, 0.38–9.03 μIU/mL) was significantly higher in the exposed group than that in the control group (median, 1.15 μIU/mL; range, 0.48–2.09 μIU/mL; $p < 0.01$). Furthermore, the median MNed BNC frequency was 5‰ (range, 0–96‰) for the exposed group and 0.00‰ (range, 0–5‰) for the control group, and the difference was also

TABLE 1. General Characteristic of the Exposed and Control Groups

variables	control ($n = 26$)	exposed ($n = 23$)	p-value
age, year (mean ± SD)	37.15 ± 6.27	34.61 ± 7.78	0.178[b]
sex (*n*, %)			
male	15 (57.7)	16 (69.6)	0.390[c]
female	11 (42.3)	7 (30.4)	
median years of residence (range), years	40.0 (2–45)	3.0 (1–12)	0.000[d]
BMI[a] (kg/m^2 (*n*, %))			
≤24	15 (57.7)	18 (78.3)	0.125[c]
>24	11 (42.3)	5 (21.7)	
smoking (*n*, %)			
yes	11 (42.3)	13 (56.5)	0.321[c]
no	15 (57.7)	10 (43.5)	
alcohol drinking (*n*, %)			
yes	3 (11.5)	8 (34.8)	0.052[c]
no	23 (88.5)	15 (65.2)	

[a] BMI: body mass index. [b] Student's *t* test for comparisons between the exposed and control groups. [c] Pearson χ^2 test for comparisons between the exposed and control groups. [d] Mann–Whitney *U* tests for comparisons between the exposed and control groups.

TABLE 2. Levels of the Medical Parameters of Subjects in the Exposed and Control Groups

	control ($n = 26$)		exposed ($n = 23$)		
parameters	mean ± SD	median (range)	mean ± SD	median (range)	p-value
serum total PBDEs[a] (ng/g lipid weight)		158 (18–436)		382 (77–8452)	0.045[h]
serum MDA[b] (nmol/mL)	3.32 ± 0.6	3.42 (1.64–4.25)	2.75 ± 1.3	2.88 (1.02–5.08)	0.052[i]
serum SOD[c] (U/mL)	97.74 ± 9.5	99.96 (76.92–116.47)	101.73 ± 15.9	104.59 (61.67–128.27)	0.287[i]
serum GSH[d] (mg/L)	197.87 ± 36.7	197.87 (88.74–266.23)	212.63 ± 35.6	203.86 (127.12–307.00)	0.161[i]
serum TSH[e] (μIU/mL)	1.12 ± 0.5	1.15 (0.48–2.09)	2.27 ± 1.9	1.79 (0.38–9.03)	0.004[i]
urinary 8-OHdG[f](μmol/(mol of creatinine))	229.97 ± 210.1	156.30 (13.52–733.70)	69.04 ± 222.2	82.06 (6.54–1057.03)	0.200[i]
MNed BNC[g]		0.00 (0–5.00)		5 (0–96)	0.000[i]

[a] PBDEs: polybrominated diphenyl ethers. [b] MDA:mMalondialdehyde. [c] SOD: supreroxide dismutase. [d] GSH: glutathione. [e] TSH: thyroid-stimulating hormone. [f] 8-OHdG: 8-hydroxy-2′-deoxyguanosie. [g] MNed BNC: micronucleated binucleated cells. [h] Mann–Whitney *U* test for comparisons between the exposed and control group. [i] Student's *t* test for comparisons between the exposed and control groups.

statistically significant ($p < 0.01$). In addition, a significantly increased frequency of MNed BNC was observed in the exposed group compared with the control group ($p < 0.01$). However, no differences were observed among other markers, including levels of serum SOD, MDA, GSH, and urinary 8-OHdG levels, between these two groups ($p > 0.05$).

Associations between Risk Factors and Serum TSH and MNed BNC Frequencies. As shown in Table 3, among other factors, a history of working with e-wastes was the only significant predictor of the MNed BNC frequencies (odds ratio (OR) = 38.85; 95% confidence interval (CI), 1.11–1358.71; $p < 0.05$), and no factors were associated with serum TSH levels. However, working with e-wastes potentially increased serum TSH levels (OR = 11.86; 95% CI, 0.46–308.39; $p = 0.137$). Further stepwise multivariate logistic regression analysis suggested that a history of engaging in e-wastes was an independent predictor of the MNed BNC frequencies (OR = 28.00; 95% CI, 5.90–132.83; $p < 0.000$). Both a history of engaging in e-wastes (OR = 6.12; 95% CI, 1.58–23.72; $p < 0.009$) and sex (OR = 4.35; 95% CI, 1.06–17.80; $p < 0.041$) were independent predictors of serum TSH levels (Table 4). Furthermore, there was no correlation found between these two markers ($p > 0.05$) (data not shown). The OR and CI values were calculated and adjusted by including several covariates in the same model with the stepwise approach. The difference suggests that the association between levels of enhanced MNed BNC frequencies and history of engaging e-wastes may be independent to the year of local residence (Table 5).

Correlation between Serum TSH Levels and MNed BNC Frequencies. There was no correlation found between serum TSH levels and MNed BNC frequencies ($p > 0.05$, data not shown).

Discussions

There is an increasing concern about environmental pollution resulting from released pollutants from the e-wastes and their potential adverse effects on human health (*3, 4, 35, 36*). In modern China, popular use of electronic appliances and frequent upgrades of electronic products have produced a large volume of e-wastes. At the meantime, as a recycle place of used electronic appliances, plentiful e-wastes have been continually transported to China from some western countries. For these reasons the environmental burden in certain areas in China has been aggravated (*2, 3, 35, 37*). For example, the exposure area in the present study is just one of many e-waste dismantling sites that release many pollutants including polycyclic aromatic hydrocarbons (PAHs) and heavy metals during the improper process of dismantling and recycling of these e-wastes (*2, 36*). However, exposure levels of PAHs in the subjects were not detected because of limited sample quantity. No association was found between the clinical parameters and concentrations of heavy metals including lead, copper, and cadmium in the body fluid in this study population.

PBDEs are known to have some characteristics of persistent organic pollutants (POPs) such as bioaccumulation and biomagnification in the environment (*38, 39*). However, evidence of their effects on human health is lacking, because the exposure was rather low or scant (*40*). Previous studies reported that PBDEs were detectable in some aquatic organisms, diet, and breast milk (*10, 41, 42*). Furthermore, a recent study suggested that levels of PBDEs were also obviously increasing in the environment and in humans (*10*),

TABLE 3. Multivariate Logistic Regression Analysis[a] of Risk Factors, MNed BNC, and Serum TSH[b]

	frequency of MNed BNC[c] (‰)		serum TSH level (μIU/mL)	
variable	OR[d]	95% CI[e]	OR[d]	95% CI[e]
age	1.01	0.88–1.17	0.94	0.84–1.06
sex				
male		referent		referent
female	2.15	0.34–13.58	4.48	0.88–22.87
year of residence	1.01	0.91–1.13	1.04	0.94–1.145
BMI (kg/m^2)				
≤24 (0)		referent		referent
>24 (1)	0.57	0.09–3.42	0.66	0.13–3.29
smoking				
no (0)		referent		referent
yes (1)	1.55	0.25–9.53	0.42	0.08–2.17
alcohol drinking				
no (0)		referent		referent
yes (1)	1.30	0.16–10.81	6.24	0.88–44.23
history of engaging e-waste				
no (0)		referent		referent
yes (1)	38.85[f]	1.11–1358.71	11.86	0.46–308.39

[a] Multivariate logistic regression model with enter approach. [b] TSH: thyroid-stimulating hormone: >1.41 (1), ≤1.41 (0) (1.41: median levels of all subjects). [c] MNed BNC: Micronucleated binucleated cells: >4 (1), ≤ 4 (0) (4: median levels of all subjects). [d] OR, odds ratio. [e] CI, Confidence interval. [f] $p \leq 0.05$.

TABLE 4. Multivariate Logistic Regression Analysis[a] of Risk Factors, MNed BNC and Serum TSH

	frequency of MNed BNC[a] (‰)		serum TSH[b] level (μIU/mL)	
variable	crude OR (95% CI)	adjusted OR (95% CI)	crude OR (95% CI)	adjusted OR (95% CI)
history of engaging e-wastes	28 (5.90–132.83)[c]	28 (5.90–132.83)[c,d]	2.77 (0.82–9.31)[e]	6.12 (1.58–23.72)[c,d]
sex			4.32 (1.30–14.34)	4.35 (1.06–17.80)[e,f]

[a] MNed BNC, micronucleated binucleated cells: >4 (1); ≤4 (0) (4, median levels of all subjects). [b] TSH, thyroid-stimulating hormone: >1.41 (1); ≤ 1.41 (0) (1.41, median levels of all subjects). [c] $p < 0.01$. [d] OR (odds ratio) adjusted by age, sex, years of residence, body mass index, smoking, and alcohol drinking using multivariate logistic regression model with backward stepwise approach. [e] $p < 0.05$. [f] OR (odds ratio) adjusted by age, years of residence, BMI, smoking, alcohol drinking, and history of engaging e-wastes using multivariate logistic regression model with Backward stepwise approach.

TABLE 5. Logistic Regression Analysis of Risk Factors for Frequency of MNed BNC, Serum TSH, and Year of Local Residence

	frequency of MNed BNC				serum TSH level			
	OR[a]		95% CI[b]		OR[a]		95% CI[b]	
variable	W[c]	W/O[d]	W[c]	W/O[d]	W[c]	W/O[d]	W[c]	W/O[d]
sex								
male			referent	referent			referent	referent
female	2.15	2.18	0.34–13.58	0.38–13.60	4.48	4.69	0.88–22.87	0.93–23.60
age	1.01	1.02	0.88–1.17	0.90–1.16	0.94	0.96	0.84–1.06	0.87–1.07
year of residence	1.01	–	0.91–1.13	–	1.04		0.94–1.15	–
BMI(kg/m^2)								
≤24 (0)			referent	referent			referent	referent
>> (1)	0.57	0.57	0.09–3.42	0.10–3.41	0.66	0.68	0.13–3.29	0.14–3.31
smoking								
no (0)			referent	referent			referent	referent
yes (1)	1.55	1.53	0.25–9.53	0.25–9.38	0.42	0.42	0.08–2.18	0.08–2.14
alcohol drinking								
no (0)			referent	referent			referent	referent
yes (1)	1.30	1.30	0.16–10.81	0.16–10.93	6.24	5.84	0.88–44.23	0.85–39.94
history of engaging e-wastes								
no (0)			referent	referent			referent	referent
yes (1)	38.85[e]	28.7[f]	1.11–1358.71	4.78–172.40	11.86	4.10	0.46–308.39	0.89–18.82

[a] OR, odds ratio; [b] CI, confidence interval. [c] W, adjustment for other factors with years of residence. [d] W/O, adjustment for other factors without years of residence. [e] $p < 0.05$. [f] $p < 0.01$.

although the levels were still low compared with other pollutants such as polychlorinated biphenyls.

In the present study, subjects from the exposure group had a high probability of exposure to PBDEs in the environment. Because the median concentration of total PBDEs in the atmosphere of the e-waste recycling site was 7149 pg/m^3 (range, 2858–19815 pg/m^3), which was 47 times the concentration in the atmosphere of the control site (median, 150 pg/m^3; range, 80–209 pg/m^3). Furthermore, the median concentration of total PBDEs in the exposed group was markedly increased by more than 2-fold compared with that of the control group. This finding implies that these workers

may have been exposed to PBDEs in the environment polluted by e-wastes. It is possible that the detected PBDE concentrations may come from both the workplace where the e-wastes were processed and their living environment which was in a close vicinity of the workplace, because these workers were employed by the owners of the waste sites, who also provided the workers with shelters close to the waste sites. It is notable that the median PBDE level of the exposed group was much higher than those reported from studies in Spain (median, 12 ng/g lipid (*43*)), New Zealand (4 congeners; mean, 7.17 ng/g lipid (*44*)), and Japan (13 congeners, median, 2.89 ng/g lipid (*45*)). These data imply that some active measures of environmental and occupational protection should be reinforced in the local areas in China by introducing advanced processing methods, improving the workplace environment, and biomonitoring of the exposed populations.

To the best of our knowledge, the impact of PBDEs on human health has not been reported, and our finding of possible adverse health effects of PBDE exposure on humans is an important addition to the literature. Previous studies with a short-time rat experiment indicated that some commercial PBDE mixtures can interfere with the thyroid hormone system (*8, 14*). However, another study suggested that the variation in plasma thyroid hormone levels was not related to the exposure level of total PBDEs among the workers who worked at an electronic recycling facility (*11*). In the present study, exposure to PBDEs may have an effect on serum TSH levels because there was a trend in the associations as the levels of exposure increased. It is likely that the pollutants other than PBDEs from the e-waste dismantling process may also affect the balance of thyroid hormone homeostasis. Because chemical structures of PBDEs and their metabolites are similar to specific thyroid hormone metabolites thyroxine (T4) (*8*), alteration in glucuronidation capacity of T4 in the liver can be influenced (*46*), resulting in increasing TSH and T4 levels from the thyroid gland by controlling of feedback mechanisms within the hypothalamic−pituitary−thyroid axis. In the present study, the influence of T4 on thyroid hormone homeostasis was not measured, let alone many endogenous and exogenous factors that can affect the balance of thyroid hormone in the human body but which were not measured in the present study. Continuous exposure to various released chemicals from the primitive treatment process of the e-wastes may lead to an accumulative biological effect, which is a time-dependent event.

Numerous studies suggested that oxidative stress may cause damage to macromolecules (i.e., DNA, lipids, and proteins) such as 8-OHdG that can lead to many diseases including cancer and heart diseases (*18*). Urinary 8-OHdG levels generally reflect the extent to which DNA damage in the target tissues is induced by oxidative stressors. However, in the present study, we did not detect any significant changes in the indicators of cellular oxidative damage such as MDA, SOD, and GSH in serum samples as well as urinary 8-OHdG in the exposed group. No differences in the levels of SOD, GSH, and 8-OHdG were found between the exposed and control groups, which may due to a low statistical power of this study, although the sample size was large enough for detecting the MDA level (power = 0.754 as calculated by the Power Sample Size software available at http://biostat.mc.vanderbilt.edu/twiki/bin/view/Main/PowerSampleSize). It is also likely that balances between oxidants and antioxidants in those subjects from the exposed group were still maintained because of the relatively short-term or low levels of exposure. Obviously, larger studies are needed to further test the hypothesis that PBDEs may cause oxidative stress in the exposed population.

The CBMN assay can assess cytotoxicity and mitotic activity in response to exposure to genotoxic agents that predominantly induce excision-repairable lesions (*28, 47*). Accumulated evidence suggests that the micronucleus induction can lead to cancer (*48, 49*). In the present study, we found that the subjects with a history of working with e-wastes had 28-fold increased risk (OR, 28; 95% CI, 5.90–132.83) for the MNed BNC frequencies, compared with the subjects without a history of working with e-wastes. Two in vitro study suggested that PBDEs can cause DNA damage in HepG2 cells (*50*) and that Ultrafine TiO_2 (often exist in plastic products) can increases frequencies of MNed BNC (*51*). Therefore, these workers may have also been exposed to PAHs and unknown metals that need to be further investigated.

In conclusion, our findings showed that levels of serum PBDEs and TSH and MNed BNC frequencies among subjects exposed to e-wastes in the workplace were significantly increased, suggesting that PBDEs exposure may interfere with the thyroid hormone system and cause the genotoxic damage. However, there was no evidence of oxidative DNA damage that may be caused by PBDE exposure in the study population. Nevertheless, this study provides the first evidence of an adverse health effect of working with e-wastes, although the mechanisms underlying the observed cytogenetic toxicity remain to be determined.

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