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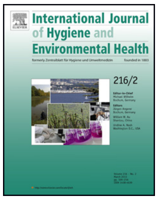

Is the fact of parenting couples cohabitation affecting the serum levels of persistent organohalogen pollutants?

Katarzyna Góralczyk[a,*], Paweł Struciński[a], Bogdan Wojtyniak[b], Daniel Rabczenko[b], Christian H. Lindh[c], Bo A.G. Jönsson[c], Gunnar Toft[d], Virissa Lenters[e], Katarzyna Czaja[a], Agnieszka Hernik[a], Jens Peter Bonde[f], Henning S. Pedersen[g], Valentyna Zvyezday[h], Jan K. Ludwicki[a]

[a] *Department of Toxicology and Risk Assessment, National Institute of Public Health – National Institute of Hygiene, Chocimska 24, 00-791 Warsaw, Poland*
[b] *Centre of Monitoring and Analyses of Population Health, National Institute of Public Health – National Institute of Hygiene, Chocimska 24, 00-791 Warsaw, Poland*
[c] *Division of Occupational and Environmental Medicine, Department of Laboratory Medicine, Sweden Lund University, S-221 85 Lund, Sweden*
[d] *Danish Ramazzini Centre, Department of Occupational Medicine, Aarhus University Hospital, Norrebrogade 44, Building 2c, 8000 Aarhus C, Denmark*
[e] *Division of Environmental Epidemiology, Institute for Risk Assessment Sciences, Utrecht University, Yalelaan 2, 3584 CM Utrecht, The Netherlands*
[f] *Department of Occupational and Environmental Medicine, Copenhagen University Hospital, Bispebjerg Bakke 23, 2400 Copenhagen NV, Denmark*
[g] *Primary Health Care Clinic, Postbox 570, DK-3900 Nuuk, Greenland*
[h] *Department of Social Medicine and Organization of Public Health, Kharkiv National Medical University, 61022 Kharkiv, Ukraine*

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ABSTRACT

Organohalogen compounds constitute one of the important groups of persistent organic pollutants (POPs). Among them, due to their long-term health effects, one should pay attention on organochlorine pesticides, polychlorinated biphenyls (PCBs) and perfluoroalkylated substances (PFASs). This paper is an attempt to answer the question about relation between the fact of cohabitation by couples expecting a child and the level of the organohalogen compounds in the blood serum of both parents. The study was done on a population of parent couples from Greenland, Poland and Ukraine, from whom blood samples were collected in order to establish the levels of marker organohalogen compounds. We selected, as the representative of these compounds, the most persistent metabolite of DDT, i.e. *p,p′*-DDE, the most frequently detected PCB congener – CB-153, and PFOS and PFOA as the representatives of PFASs. The results show that in case of all compounds under study the highest concentrations were present always in men in relation to the levels detected in the blood serum of their female partners, regardless of the country of origin of the couple. A positive correlation was noted between the concentrations of the studied compounds in the blood serum of men and women in parenting couples. In some cases these correlations were statistically significant, e.g. for concentrations of *p,p′*-DDE in pairs from Greenland and Ukraine, of CB-153 in pairs from Poland and Ukraine, and of PFOS for parents from Greenland and Poland, while for PFOA – only for couples from Greenland. The concentrations of the compounds included in the study were similar to the levels found in general population in other countries. Our results show that the exposure to POPs resulting from cohabitation plays a role in the general exposure to these compounds.

Abbreviations: BMI, Body Mass Index; DDT, 1,1,1-trichloro-2,2-bis-(p-chlorophenyl) ethane; EFSA, European Food Safety Authority; HCH, hexachlorocyclohexane; ICC, intraclass correlation coefficient; LC, liquid chromatography; LC/MS/MS, liquid chromatography tandem mass spectrometry; LOD, limit of detection; OCs, organochlorine compounds; PCBs, polychlorinated biphenyls; *p,p′*-DDE, 1,1-dichloro-2,2-bis-(p-chlorophenyl) ethylene; PFASs, perfluoroalkylated substances; PFOA, perfluorooctanoic acid; PFOS, perfluorooctane sulfonate; POPs, persistent organic pollutants; SPE, solid phase extraction; TTP, time to pregnancy.

[*] Corresponding author. Tel.: +48 225421361; fax: +48 228497441.

E-mail addresses: kgoralczyk@pzh.gov.pl (K. Góralczyk), pstrucinski@pzh.gov.pl (P. Struciński), bogdan@pzh.gov.pl (B. Wojtyniak), daniel@pzh.gov.pl (D. Rabczenko), Christian.Lindh@med.lu.se (C.H. Lindh), bo_a.jonsson@med.lu.se (B.A.G. Jönsson), gunntoft@rm.dk (G. Toft), V.C.Lenters@uu.nl (V. Lenters), kczaja@pzh.gov.pl (K. Czaja), ahernik@pzh.gov.pl (A. Hernik), Jens.Peter.Ellekilde.Bonde@regionh.dk (J.P. Bonde), HSP@MIL.AU.DK (H.S. Pedersen), dmusic@ic.kharkov.ua (V. Zvyezday), k.ludwicki@pzh.gov.pl (J.K. Ludwicki).

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Introduction

Organohalen compounds, being persistent organic pollutants (POPs), are a subject of keen interest of scientists, especially in the domain of chemical safety. Organohalen POPs include organochlorine compounds, such as polychlorinated biphenyls (PCBs) or DDT and its metabolites, as well as perfluoroalkylated substances (PFASs). Due to their similar physical and chemical properties, similar interaction with various elements of the environment, including the human organism, as well as their prevalence in the environment, they are a subject of environmental studies, monitoring of human exposure to food and air contaminants and studies of direct impact on the human health at various stages of development (Barr et al., 2007; Darnerud et al., 2006; Hernik et al., 2014; Góralczyk et al., 2010; Kucharska et al., 2011; Struciński et al., 2000, 2006; Tan et al., 2009).

Organochlorine compounds (OCs) found broad application as pesticides in crop protection. The most widely known organochlorine insecticides were: DDT and its metabolites, hexachlorocyclohexane (HCH), chlordane, endosulfan, metoxychlor, heptachlor and cyclodien pesticides – aldrin, dieldrin, endrin (McKinlay et al., 2008); their presence is still confirmed in biological monitoring (Czaja et al., 2006; Fernandez et al., 2008; Góralczyk et al., 2010; Hernik et al., 2014; Knutsen et al., 2011; Pulkrabová et al., 2009). Next to the above-mentioned pesticides, one should also list polychlorinated biphenyls (PCBs) which were widely used as dielectrics in large-capacity transformers and condensers, heat exchangers, hydraulic systems, as a part of lubricating oils and cooling/lubricating liquids, plasticizers for paints, inks, printing inks and copy paper, as flame retardants for plastics, as well as pesticide vectors (Ziets et al., 2008). Humans are exposed to these compounds throughout their whole life. The main exposure pathways are food and air. The share of these routes in the general exposure profile differs depending on the physical and chemical properties of the compound. Around 90% of organochlorine compounds (pesticides, PCBs) are taken in by humans with food, especially of animal origin, of which 70–80% of e.g. PCBs intake comes from fish and seafood, despite the withdrawal of these compounds from the market in Europe many years ago. Scandinavian research showed that daily intake of PCBs and organochlorine pesticides in food is respectively 615 and 523 ng (Darnerud et al., 2006). Due to their lipophilicity, these compounds may accumulate in human tissues. For example, in research conducted in the Czech Republic (Pulkrabová et al., 2009), samples of adipose tissue contained an average ΣDDT 615.6 ng g^{-1} lipid with a dominant *p,p′*-DDE metabolite, which confirms the distant and long-term exposure to this pesticide. Similar concentrations were found by Struciński et al. (2002) in the tissue of the mammary gland of women from Poland. This caused the concentration of *p,p′*-DDE to be accepted in numerous studies as the indicator of the exposure of humans to organochlorine pesticides (Crisp et al., 1998; Jönsson et al., 2004). In case of ΣPCBs, Fernandez et al. (2008) quantified the average concentration of seven indicator congeners of PCBs (CB-28, 52, 101, 118, 138, 153 and 180) in human adipose tissue to 625.5 ng g^{-1} lipid. The total concentration of these seven indicator congeners has been accepted in Europe as the marker of PCBs dietary exposure (Wingfors et al., 2006). The dominating congeners in the adipose tissue of general populations are always CB-138, 153 and 180. Similar results, together with a defined profile of occurrence of individual congeners, were achieved also by other authors (Covaci et al., 2008; Knutsen et al., 2011; Thomas et al., 2006). That is why in many studies it was accepted that the levels of congener CB-153 in the blood serum constitute a marker for human exposure to polichlorinated biphenyl (Crisp et al., 1998; Jönsson et al., 2004).

Perfluoroalkylated substances have high chemical and thermal stability. For this reason, they found application in many industries as oleophobic substances, textile impregnation coatings, sliding substances, protective anti-adhesive coatings, electric cable insulators, high-capacity fire suppressing foams, hydro-gels applied on open wounds, emulsifiers for cosmetics, as well as ingredients of fat-resistant food packaging (EFSA, 2012; Kucharska et al., 2011). Such a wide application of perfluoroalkylated substances resulted in their global emission and distribution to various parts of the environment, including the human organism. Among the PFASs, the most commonly applied ones were the perfluorooctanoic acid (PFOA) and the perfluorooctane sulfonate (PFOS). Due to their toxicological properties the restrictions on the use of PFOS and PFOA in Europe were introduced by Commission Regulation (2010). Similarly as in the case of OCs and PCBs, the basic source of intake of these substances by humans is the food (EFSA, 2012; Guerranti et al., 2013; Noorlander et al., 2011). In case of PFOS, for people from various age groups, the main source of exposure are fish and seafood (50–80%), fruits and processed fruit products (8–27%), as well as meat and meat products (5–8%). In case of PFOA, the main source of exposure are fruits and processed fruit products (18–39%), and fish and seafood (7.6–27%) (EFSA, 2008). Similar exposure profiles are mentioned also by other authors studying the exposure of various populations to these compounds (Guerranti et al., 2013; Yamaguchi et al., 2013; Vestergren et al., 2008). In contrast to OCs, the magnitude of exposure of humans to PFASs through diet, according the studies of some research centres, is influenced also by the drinking water (Noorlander et al., 2011). In Sweden, studies were carried out in the years 1987–2007 for the levels of PFOS and PFOA, among others, in the blood serum of healthy women born in the years 1934–1967 (Axmon et al., 2014). The median of PFOS concentrations in individual years were in the range between 10.2 and 35.5 ng mL^{-1}, with slight falling trend since 1998, when the highest level of this compound was noted. PFOA concentrations were found in the range between 1.78 and 5.51 ng mL^{-1}, without any discernible trend (Axmon et al., 2014). Other authors studying the levels of PFOA and PFOS in blood serum obtained varying results, depending on the geographical location (place of residence) of the sample donors. In the study of Liu et al. (2012), for the whole population studied a median of concentrations for PFOS was found to be 1.92 $\mu g\,L^{-1}$ (for women = 1.20 $\mu g\,L^{-1}$, for men = 2.39 $\mu g\,L^{-1}$), and for PFOA all results were below 0.03 $\mu g\,L^{-1}$. Meanwhile, in similar studies performed in Greece, the detected levels were higher, while retaining the relation of the higher concentrations in the blood serum of men over women. For women, the PFOS median was equal to 7.03 ng mL^{-1}, and for men – 13.69 ng mL^{-1}, in case of PFOA the values were, respectively 1.70 ng mL^{-1} and 3.14 ng mL^{-1} (Vassiliadou et al., 2010). Much higher levels were found in Germany, where the range of concentrations for the general population was 6.2–130.7 $\mu g\,L^{-1}$ with the median of 22.3 $\mu g\,L^{-1}$, and for PFOA – between 1.7 $\mu g\,L^{-1}$ and 39.3 $\mu g\,L^{-1}$, with the median of 6.8 $\mu g\,L^{-1}$. In all samples studied both compounds were found (Midasch et al., 2006).

Numerous publications confirm that people who start to live together as a couple change their eating habits. These changes take various directions: either one of the partners adapts to the eating habits of the other, or a common diet is developed based on eating habits of both (Bove et al., 2003; Kremmer et al., 1998; Lewis et al., 2006; Louk et al., 1999; Nasuti et al., 2014). Additionally, the diet of couples expecting a baby tends to change. During this period couples avoid products which are popularly deemed as unhealthy, while increasingly and more frequently eating healthier products and those that are less burdensome for the organism. (Laroche et al., 2012; Verbeke and de Bourdeaudhuij, 2007). This approach varies depending on whether the couple expects their first or another child. In case of the first child, the change in diet is more pronounced and the couple tries to introduce as many products which according to contemporary dietary knowledge are beneficial to health as