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Bu, Qingwei, MacLeod, Matthew, Wong, Fiona, Toms, Leisa-Maree, Mueller, Jochen, & Yu, Gang (2015) Historical intake and elimination of polychlorinated biphenyls and organochlorine pesticides by the Australian population reconstructed from biomonitoring data. *Environment International, 74*, pp. 82-88.

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https://doi.org/10.1016/j.envint.2014.09.014
Historical intake and elimination of polychlorinated biphenyls and organochlorine pesticides by the Australian population reconstructed from biomonitoring data

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A R T I C L E  I N F O

Article history:
Received 8 April 2014
Accepted 26 September 2014
Available online 21 October 2014

Keywords:
PCBs
OCPs
Pharmacokinetics
Modeling
Elimination
Intake

A B S T R A C T

Quantifying the competing rates of intake and elimination of persistent organic pollutants (POPs) in the human body is necessary to understand the levels and trends of POPs at a population level. In this paper we reconstruct the historical intake and elimination of ten polychlorinated biphenyls (PCBs) and five organochlorine pesticides (OCPs) from Australian biomonitoring data by fitting a population-level pharmacokinetic (PK) model. Our analysis exploits two sets of cross-sectional biomonitoring data for PCBs and OCPs in pooled blood serum samples from the Australian population that were collected in 2003 and 2009. The modeled adult reference intakes in 1975 for PCB congeners ranged from 0.89 to 24.5 ng/kg bw/day, lower than the daily intakes of OCPs ranging from 73 to 970 ng/kg bw/day. Modeled intake rates are declining with half-times from 1.1 to 1.3 years for PCB congeners and 0.83 to 0.97 years for OCPs. The shortest modeled intrinsic human elimination half-life among the compounds studied here is 6.4 years for hexachlorobenzene, and the longest is 30 years for PCB-74. Our results indicate that it is feasible to reconstruct intakes and to estimate intrinsic human elimination half-lives using the population-level PK model and biomonitoring data only. Our modeled intrinsic human elimination half-lives are in good agreement with values from a similar study carried out for the population of the United Kingdom, and are generally longer than reported values from other industrialized countries in the Northern Hemisphere.

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1. Introduction

Despite bans and phase-outs that began in the 1970s, persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCPs), are still detected in the environment due to their extensive use in the past in products with long lifetimes (Gasic et al., 2010) and persistence in the environment (Beyer and Biziuk, 2009; Namiki et al., 2013; Wang et al., 2013). POPs enter humans through diverse routes (e.g. inhalation, ingestion, dermal), but ingestion is often the dominant exposure pathway since POPs can bioaccumulate along the food chain (Kelly et al., 2007). Simultaneously, POPs are eliminated from the body by various pathways (e.g. metabolic conversion, and excretion through feces). The competing rates of intake and elimination determine the dynamic balance of POPs in the human body (Alcock et al., 2000). Quantifying these competing rates is thus of fundamental importance for understanding the levels and trends of POPs at a population level.

Ingestion of contaminated foods represents the most important exposure pathway for most POPs (Sweetman et al., 1999, 2000); therefore the intake can usually be assessed by measuring concentrations of POPs in various foodstuffs and multiplying by consumption rates (Caspersen et al., 2013). However, it is challenging to reconstruct past exposures in this way, as the food contamination by POPs has not been monitored continuously in the past and exposure estimates provide only an incomplete ‘snapshot’ of exposures that are changing over time. Models provide a powerful compliment to measurements that can help to interpolate or extrapolate from monitoring data (Cowan-Ellsberry et al., 2009). For example, Alcock et al. (2000) modeled dietary intakes of PCB-101 from contamination in the air. Models can also be used to explore alternative exposure scenarios that may arise due to the uncertainties in emission inventories and future use of POPs (Breivik et al., 2010).

Estimating human elimination half-lives of POPs presents several challenges and a range of different approaches that exploit different types of data have been explored. One approach is to use longitudinal
data from sequential measurements in the same individual. Many longitudinal data-based studies use individuals who experienced high exposure from the workplace or an accident (Masuda, 2001; Wolff et al., 1992) so that ongoing exposure could be considered negligible. However, half-lives derived from high exposure individuals or groups could be different from those for general population, as there is evidence showing that the elimination rates of POPs from the human body are concentration-dependent (Milbrath et al., 2009). An alternative is to combine longitudinal biomonitoring data with estimates of ongoing exposure and body weight changes to estimate elimination half-lives (Grandjean et al., 2008). Another alternative approach is to interpret one or more sets of cross-sectional data, which represents body burdens as a function of age in the entire population, using a population-level pharmacokinetic (PK) model. Steady-state (constant) intake has been assumed in several PK modeling approaches to estimate elimination pharmacokinetic (PK) model. Steady-state (constant) intake has been assumed in several PK modeling approaches to estimate elimination half-lives from cross-sectional data or population-averaged body burdens (Geyer et al., 2004; Ogura, 2004; Shirai and Kissel, 1996). However, in reality intake of POPs is likely to be variable over time.

Recently, Ritter et al. (2011b, 2009) introduced a dynamic population-level PK model (hereafter called the “Ritter model”) that can be fitted to cross-sectional data to quantitatively describe the levels and temporal evolution of human body burden measured in biomonitoring studies, and total intake. The Ritter model can be fit to the evolving body burdens and intakes by adjusting a rate constant for intrinsic elimination from the human body that eliminates the influence of ongoing exposure and changes in body condition. The intrinsic elimination rate constant is primarily a property of the chemical. Ritter et al. (2011b) modeled the intrinsic human elimination half-lives and historical intakes of PCBs in the United Kingdom (UK) population. Wong et al. (2013) further applied the model to study the dynamic balance between intake, elimination and human body burden of polybrominated diphenyl ethers (PBDEs) in the North American population. In the UK case for PCBs, the model was optimized using biomonitoring data with and without empirical intake estimates, and the authors argued that biomonitoring data contained sufficient information to reconstruct historical intakes and to estimate intrinsic half-lives. However, in Wong et al. (2013), the available biomonitoring data and intake estimates could not be simultaneously fitted by the model, possibly indicating that intakes had been underestimated in exposure pathway studies.

Here we applied the Ritter model to two sets of cross-sectional data describing levels of ten PCBs and five OCPs in the Australian population. Intrinsic human elimination half-lives of PCBs and OCPs in the Australian population are estimated and compared with estimates for other populations, and at the same time the historical intakes of PCBs and OCPs by the Australian population are reconstructed. The overall goal of this study is to further evaluate the possibility to extract information on intake and elimination from cross-sectional data by using a population level PK model.

2. Methods

2.1. Biomonitoring data

Two sets of cross-sectional data for PCBs and OCPs were obtained from studies by the Department of Environment, Australia conducted in 2003 and 2009 (Toms and Mueller, 2010). Pooled blood serum samples analyzed in the studies were stratified by age groups and gender. The youngest age group in 2003 was <16 years where the mean age was 10 years and in 2009 was 0–4 years (followed by 5–15 years) where the mean age was 2 years. For both 2003 and 2009 the remaining age groups were 16–30, 31–45, 46–60 and >60 years. Overall, the average age of individuals in the pools ranged from 10 to 76, and 2 to 74 years for the analysis in 2003 and 2009, respectively. As no significant difference between genders was observed for the chemicals of interest, we used concentrations measured in all pooled samples in our analysis. The most prevalent PCB congeners detected in 2003 and 2009 were studied: PCB-74, PCB-99, PCB-118, PCB-138, PCB-146, PCB-153, PCB-156, PCB-170, PCB-180, and PCB-187. Five OCPs studied were: hexachlorobenzene (HCB), β-hexachlorocyclohexane (β-HCH), p,p′-dichlorodiphenyldichloroethylene (p,p′-DDE), p,p′-dichlorodiphenyltrichloroethane (p,p′-DDT), and trans-nonachlor (TNONA). Detailed information on sample collection, analysis and measured concentrations in the pooled samples is presented in section SI-1 of the Supplementary material.

2.2. The Ritter population-based PK model

We calculated lipid-normalized whole-body concentrations of chemicals in representative individuals in the Australian population using the Ritter model. This approach implicitly assumes that the distribution of chemicals within body lipids is at equilibrium. Mass balance equations were established to describe the dynamic balance between intake, body burden and elimination of a chemical in each representative individual:

\[
dC(t_{age}) = I(t_{age} - t_{birth}) - \left[ k_e + \frac{dM_{lip}(t_{age})}{dt_{age}} \times \frac{1}{M_{lip}(t_{age})} \right] \times C(t_{age})
\]  

(1)

where \( t_{age} \) (years) and \( t_{birth} \) are the age and year of birth of the individual, respectively, \( C(t_{age}) \) (ng g lipid\(^{-1}\)) is the lipid-normalized concentration of a chemical in the body, \( M_{lip}(t_{age}) \) (kg lipid) is the mass of lipid as a function of age, \( k_e \) (year\(^{-1}\)) is the first order rate constant describing intrinsic elimination, and \( I(t_{age} - t_{birth}) \) (ng g lipid\(^{-1}\) year\(^{-1}\)) is the yearly intake of a chemical for the representative individuals born at \( t_{birth} \) and is a function of both \( t_{age} \) and \( t_{birth}\):

\[
I(t_{age} - t_{birth}) = \frac{E_{a} \times M_{bio}(t_{age}) \times P(t_{age}) \times I_{ref}(t) \times U}{M_{lip}(t_{age})}
\]

(2)

where absorption efficiency, \( E_{a} \) is the fraction of chemical in the gastrointestinal tract that is taken up into the body, and assumed to be 0.9 (Moser and McLachlan, 2001), \( M_{bio}(t_{age}) \) (kg) is the body weight as a function of age, which was interpolated by using the 2011–12 statistical data of the average weight of males and females from the Australian Bureau of Statistics (2012) and taking 80 years as a fixed life expectancy. \( P(t_{age}) \) (dimensionless) is a proportionality factor used to adjust the adult reference intake for people below 16 years old according to the intake of PCB-101 for the UK population (Alcock et al., 2000), \( I_{ref}(t) \) (ng kg bw\(^{-1}\) day\(^{-1}\)) is the adult reference intake at year \( t \) (= \( t_{age} + t_{birth} \)), and \( U \) (days \( \times \) year\(^{-1}\) \( \times \) kg lipid \( \times \) g lipid\(^{-1}\)) is a unit conversion factor.

The shape of \( I_{ref}(t) \) was defined according to the use history of PCBs and OCPs in Australia. Before 1940, \( I_{ref}(t) \) is assumed to be constant and have a low and negligible value. After their introduction to the environment, concentrations of PCBs and OCPs in the environment and human food would follow an increasing trend until regulated and then a decreasing trend. The year of peak intake was determined firstly by inspection of the historical use of PCBs (Connell et al., 1996; van Gelderen and Pettigrove, 2011) and OCPs (Australian Pesticides and Veterinary Medicines Authority, 2008) in Australia. Based on optimized fits of the model to the biomonitoring data, we assumed peak intake occurred in 1975 for both PCBs and OCPs. The rate of increase for the intake between 1940 and 1975 is assumed to be the same as the rate of decline which happens after the peak intake year. Thus, for PCBs as an example, the intake in 1940 is the same as the intake in 2010 (see SI-2 of the Supplementary material for details).

We modeled the human body burden for individuals born each year in the period 1900–2020. For individuals born between 1900 and 1924, no input from breast feeding was assumed. Beginning in 1925 the intake of chemicals for infants less than 6 months old was determined from...
the volume of breast milk consumed, the content of fat in the breast milk, and the lipid normalized concentration which is assumed to be equal to that in the serum of the mother. The median amount of breast milk consumed per day and the content of milk fat were 722 mL and 3.6%, respectively (Quinsey et al., 1995).

2.3. Fitting procedure

To produce the best estimates for both the intakes and intrinsic elimination half-lives, we fit the model to two sets of empirical cross-sectional data in the general Australian population by changing three adjustable parameters: the adult reference intakes in the peak intake year and in 2000, and the intrinsic elimination rate constant, $k_0$. These three parameters were optimized to minimize the value of the objective function $(OF)$ representing the difference between empirical and modeled data:

$$OF = \left( \ln C_{\text{mea}} - \ln C_{\text{model}} \right)^2 \tag{3}$$

where $C_{\text{mea}}$ and $C_{\text{model}}$ are empirical and modeled concentrations, respectively.

The model was implemented in Microsoft Excel 2013 and optimized using the Solver add-in. Historical intake trends and intrinsic elimination rates are modeled. The reduction half-life for intake is calculated using the Solver add-in. Historical intake trends and intrinsic elimination half-lives for each chemical are calculated as $\ln(2) / k_0$. Three indicators, i.e. coefficients of determination ($R^2$), residues weighted by number of empirical data points ($OF/n$), and 95% confidence factor around the fit ($CF$), were used to evaluate the goodness of fit of the model to the empirical data and to verify that there was no bias introduced by our model fitting procedure.

3. Results

Values of the three indicators that we used to evaluate the performance of the model and the reliability of our estimates are reported in Table 1. These results are also demonstrated graphically in SI-3 (see Supplementary material). For most PCBs and OCPs, the empirical cross-sectional data can be explained by our model with $R^2$ higher than 0.7, and $OF/n < 0.13$. In these cases, the modeled concentrations fall within a 95% $CF$ of less than 2.16. However, there are three exceptional cases where the model fits to the biomonitoring data are not as good: $\beta$-HCH, HCB, and $p,p'$-DDT (bold entries in Table 1).

High $OF/n$ values for $\beta$-HCH and HCB indicated a relatively large discrepancy between the modeled and empirical cross-sectional data.

The measured values of $\beta$-HCH are highly variable in pooled samples of people of the same age (see Supplementary material, Fig. S1-I). The model cannot explain the variability adequately, leading to a poor correlation and large $CF$. This high variability might represent a high degree of inter-individual variability in body burdens in the underlying population. As a result, very long half-lives of over 5000 years were modeled for $\beta$-HCH, which are not plausible. In contrast, the low $R^2$ and relatively high $OF/n$ values for the model fit to empirical data for HCB are due to an apparent outlying group of older people who had higher body burdens than expected from the model fit (see Supplementary material, Fig. S1-k). The intrinsic elimination half-life (6.4 years) and intake trend for HCB calculated by the optimized model are not sensitive to the inclusion of this outlying datum.

For $p,p'$-DDT, despite the relatively low $R^2 = 0.377$, the modeled data fall within a narrow confidence interval ($CF = 2.14$) and a small discrepancy between the model and the measurements was indicated by the relatively small residues ($OF/n = 0.210$). Visual inspection of empirical cross-sectional data in 2003 and 2009 shows no obvious decreasing trend between 2003 and 2009, and especially in cross-sectional data of 2009, no significant difference for the human body burdens in pools of different age groups, except for the youngest cohort that is likely exposed by breastfeeding (see Supplementary material, Fig. S1-n). The modeled intrinsic elimination half-life for $p,p'$-DDT is about 115 years, which is much longer than that estimated by Ritter et al. (2009). However, the intrinsic elimination half-life could be unresolvable by our model fitting procedure because of ongoing low-level exposure to fresh DDT, which was also indicated by their modeled trend of $p,p'$-DDT showing an almost unchanged intake levels (see Supplementary material, Fig. S1-n). Mueller et al. (2008) speculated that the decline in DDT contamination in human milk in Australia slowed after the 1980s due to new input via long range transport or via consumption of food imported from more polluted countries. A significant increase of the intake of total DDT in 1990s was also observed in the total dietary studies in Australia (Connell et al., 2007). However, a declining trend over the past decade in estuarine urban water measured using passive samplers in Australia was observed by Mueller et al. (2011). Therefore, an ongoing exposure to fresh DDT may be due to changing exposure pathways as the use pattern changed (Ritter et al., 2011a).

The modeled adult reference intakes in 1975 for PCB congeners ranged from 0.89 to 24.5 ng/kg bw/day, which were slightly lower than the daily intakes for $p,p'$-DDE and much lower than those for HCB (Tables 2 and 3). After the bans of PCBs and OCPs, a sharp decline of the total human intake was expected. The modeled reduction half-lives of intake range from 1.1 to 1.3 years with the exception of PCB-156 and PCB-99, which are declining more slowly than other congeners. The reduction of intake of OCPs is comparable to PCBs, with the reduction half-lives of 0.83–0.97 years. The PCB congeners considered in this study are generally eliminated more slowly from the human body than the OCPs considered here. The shortest intrinsic human elimination half-life is 6.4 years for HCB, and the longest is 30 years for PCB-74.

Fig. 1 illustrates the modeled age-concentration structures at different sampling years for PCB-156 and TNONA, with empirical cross-sectional data available at 2003 and 2009. Graphical results for the other PCBs and OCPs are shown in SI-3 (see Supplementary material). Similar age-concentration trends were observed for PCB-156 and TNONA at different sampling years. The difference between human body burdens of PCB-156 and TNONA becomes smaller as the post-ban period increases, which is because TNONA has a shorter elimination half-life (9.7 years) than PCB-156 (18 years).

Strong growth dilution effects for most compounds were observed at ages less than 16 years, which is in good agreement with the empirical measurements for other persistent organic pollutants (Toms et al., 2009). Individuals who were older than 16 years in the peak intake year are marked as the “pre-ban group” in Fig. 1. Because the pre-ban group experienced high exposure without the benefit of growth dilution to reduce concentrations, they all have similar concentrations of

| Compound   | $R^2$ | $CF$ | $OF/n$ |
|------------|-------|------|--------|
| PCB-74     | 0.675 | 1.93 | 0.156  |
| PCB-99     | 0.789 | 1.79 | 0.106  |
| PCB-118    | 0.725 | 1.83 | 0.120  |
| PCB-138    | 0.855 | 1.72 | 0.092  |
| PCB-146    | 0.861 | 1.75 | 0.085  |
| PCB-153    | 0.885 | 1.80 | 0.088  |
| PCB-156    | 0.844 | 1.97 | 0.110  |
| PCB-170    | 0.845 | 2.01 | 0.114  |
| PCB-180    | 0.879 | 2.16 | 0.115  |
| PCB-187    | 0.864 | 1.84 | 0.114  |
| HCB        | 0.478 | 4.51 | 0.506  |
| $\beta$-HCH | 0.494 | 3.34 | 0.409  |
| $p,p'$-DDE | 0.725 | 2.03 | 0.163  |
| $p,p'$-DDT | 0.377 | 2.14 | 0.210  |
| TNONA      | 0.778 | 1.94 | 0.124  |

* $CF$, 95% confidence factor around the fit.
* $OF/n$, residues weighted by number of empirical data points.
PCB-156 (Fig. 1) and other POPs with long elimination half-lives. This phenomenon has been termed “the memory effect” of past exposure (Ritter et al., 2011b). Within the “post-ban group” that reaches the age of 16 after the peak intake year, body burdens are higher in older individuals (Fig. 1), and are determined by exposure history and elimination simultaneously.

4. Discussion

4.1. Intakes

Only two studies were identified in which total intakes of PCBs ($\sum$ PCBs) were reported for the Australian population. As no measured data was available for most PCB congeners, we back-calculated $\sum$ PCBs by assuming that the 10 congeners we studied represent about 40% of the dietary intake of $\sum$ PCBs (MAFF, 1996). Our estimates are in good agreement with those made by the Australian Market Basket Survey (AMBS) (National Advisory Body on Scheduled Wastes, 1998), but about 2 orders of magnitude lower than calculated by Kannan et al. (1994) (Table 2).

Kannan et al. (1994) also reported a higher empirical intake than our modeled intake for HCB in 1990. The initial AMBS conducted in 1970 reported an estimated daily intake for HCB from 700 to 1400 ng/kg bw/day, with an average of 600 ng/kg bw/day for 15–18 year old males (Connell et al., 2007). It is reasonably higher than our estimates of adult reference daily intakes as younger individuals are expected to have a higher daily intake (Alcock et al., 2000). The empirical intake for p,p'-DDE estimated by AMBS was much higher than our model estimate.

This discrepancy between modeled and empirical intakes could be due to overestimation of intakes by previous total dietary studies, overestimation of intrinsic elimination half-lives, or both. To assess the plausibility of our model results, we fit the biomonitoring data to our model by constraining the intake at 1990 to be equivalent to those estimated from Kannan et al. (1994). The modeled elimination half-lives were 2 orders of magnitude lower than those from Grandjean et al. (2008), which is not plausible. As well, a greater discrepancy between the modeled and measured cross-sectional data was observed (see Supplementary material, Table S4). Therefore we believe that the empirical intake of PCBs reported by Kannan et al. (1994) is too high to plausibly explain the PCB body burdens in the Australian population.

Overestimation of the intake could be due to uncertainties in dietary exposure estimation. First, the food samples analyzed may not be representative because dietary habit differs between people. In Kannan et al. (1994), the total dietary intakes are estimated based on PCBs and OCPs in foodstuffs collected from different locations in Australia and may not reflect the dietary intakes of individuals that participated in the biomonitoring studies. Second, intakes were always estimated based on short-term food consumption surveys, such as 24-h records (EFSA, 2006).

We also considered the study of Ritter et al. (2011b) that modeled intakes of PCBs in the UK population using the same model that we used in this study. The peak intake in our study occurred 5 years later in Australia compared to the UK and the values of the peak intake for the Australian population are generally lower than those in the UK by factors of up to 25 for PCB-180 (Table 2). The lower intakes of PCBs in the Australian population likely reflect the lower use and contamination by PCBs in various matrices of Australia than in other places worldwide (Kalantzi et al., 2001; Meijer et al., 2003; Pozo et al., 2006). A faster reduction trend in PCB intake in Australia relative to the UK is also indicated (Table 2).

In our study biomonitoring data were obtained from measured POP concentrations in pooled serum samples. Pooled samples have several advantages relative to individual samples, and also some limitations (Heffernan et al., 2013). One important property in the current context is that pooled samples reflect the arithmetic mean concentration of individual samples in the pool (Heffernan et al., 2013). In the case of PCBs in the UK population, the biomonitoring data were categorized by age and the geometric mean was calculated for different age groups. To characterize the bias due to geometric versus arithmetic means, we estimated the geometric mean of PCB concentrations for the Australian population.

Table 2

| Reference          | Year | Country | PCB-74 | PCB-99 | PCB-118 | PCB-138 | PCB-146 | PCB-153 | PCB-156 | PCB-170 | PCB-180 | PCB-187 | $\sum$ PCBs* |
|--------------------|------|---------|--------|--------|---------|---------|---------|---------|---------|---------|---------|---------|--------------|
| Ritter et al. (2011b) | 1970 | UK      | 0.89   | 1.53   | 12.3    | 22.7    | 2.8     | 23.7    | 3.8     | 10.7    | 24.5    | 6.4     | 62.1         |
| This study         | 1975 | Australia | 0.088 | 0.029 | 0.0026 | 0.0034 | 3.5 × 10^-4 | 0.0035 | 3.9 × 10^-4 | 0.0024 | 0.0029 | 0.0017 | 0.34 | 0.7  |

Table 3

| Reference          | Year | Country | HCB | p,p'-DDE | TNONA |
|--------------------|------|---------|-----|----------|-------|
| Connell et al. (2007) | 1990 | Australia | 3.37 | 2.81 | 1.7 × 10^-5 | 3.0 × 10^-7 |
| This study         | 1993 | Australia | 0.078 | 0.084 | 2.5 × 10^-4 | 2.1 × 10^-4 |
| AMBS (2001)        | 1998 | Australia | 9.6  | 20     | 1.1   | 6.3   | 19    | 335    | 773    | 0.0078 | 0.0084 | 2.5 × 10^-4 | 2.1 × 10^-4 |
| This study         | 1999 | Australia | 0.97 | 0.89 | 0.83 | 0.97 | 0.89 | 0.83 |

Table 4

| Reference          | Year | Country | HCB | p,p'-DDE | TNONA |
|--------------------|------|---------|-----|----------|-------|
| Kangen et al. (1994) | 1990 | Japan   | 6.3  | 19    | 335    | 773    | 0.0078 | 0.0084 | 2.5 × 10^-4 | 2.1 × 10^-4 |
population. The procedure is described in detail in Supplementary material, and followed the approach recommended by Aylward et al. (2014). Briefly, we used the degree of variability in the National Health and Nutrition Examination Survey biomonitoring data in 2003 and 2004 (NHANES, 2005) to estimate the variability in the Australian population. The model was fit to the estimated geometric mean of the biomonitoring data and modeled intakes and intrinsic elimination half-lives are listed in Table S6 (see Supplementary material). When fitting the model using the geometric mean, no bias was observed for the intrinsic elimination half-lives, but estimates for peak intakes were lower than when using the arithmetic mean by a factor of around 2. Hence the difference between intake estimates for the UK and Australian populations is even larger, especially for PCB-180 differing by 2 orders of magnitude. Estimates of intakes from model fitting using the geometric mean indicated an even larger discrepancy between modeled intakes and empirical measurements from exposure pathway studies.

We reconstructed intakes of the PCBs and OCPs by fitting the Ritter model to biomonitoring data. The use of modeled data has been encouraged to compensate for the absence of measurements of contamination in various exposure pathways (Sahmel et al., 2010). Although a discrepancy was observed between our modeled intakes and empirical measurements, our modeled intakes adequately explain human body burdens in the biomonitoring data that are considered to be the gold standard in studies. Overall, our results demonstrate the effectiveness of reconstructing historical exposure of a population by using a population-based PK model and biomonitoring data only. However, we emphasize that uncertainties in our reconstructed historical intake trend and in our intrinsic elimination half-lives (reported below) are high and remain unquantified. More refined model estimates of intake and elimination and a quantitative treatment of uncertainty will be feasible when more cross-sectional datasets are added to the biomonitoring database in the future.

4.2. Elimination half-lives

The intrinsic elimination half-lives estimated for PCBs in the Australian population are similar to those derived from cross-sectional data from the UK population based on the same model by Ritter et al. (2011b) (Table 2).

We also considered the study of Ogura (2004) that takes ongoing exposure and change in body size into account by using a PK model. However, different PCB congeners were studied by Ogura (2004) than our study, except for PCB-118 and PCB-156. Ogura (2004) reported the intrinsic elimination half-life for PCB-118 as 6.3 years, which is a factor of 1.5 shorter than that estimated by Ritter et al. (2011b), and a factor of 1.7 shorter than our value. Our estimated intrinsic elimination half-life of 18 years for PCB-156 is very similar to Ogura’s estimate of 19 years.

Grandjean et al. (2008) estimated the intrinsic elimination half-lives using longitudinal data from a cohort of children from 4.5 to 14 years old. They used a regression approach to explain these longitudinal data by considering body mass index and the number of whale dinners as covariates. Estimates of intrinsic elimination half-lives from Grandjean et al. (2008) usually differ by a factor of 2 from Ritter et al. (2011b) and ours (Table 2).

We are only able to identify one study (To-Figueras et al., 2000) which reported the elimination half-life of HCB. The literature reported value is 6 years, similar to our estimate of 6.4 years. Again, our estimates of the intrinsic elimination half-life for pp’-DDE differ from previously reported values by a factor of 2 or less (Table 3). For TNONA, the intrinsic elimination half-life in the Australian population is estimated as 9.7 years. To the best of our knowledge, it is the first report on the elimination of TNONA in humans.

The difference in intrinsic half-lives between our estimates and the literature reported values may be due to inter-study variability. However, other factors may contribute to the relatively high elimination half-lives, such as concentration-dependent elimination process (Ritter et al., 2011b). Here we mean that the elimination half-lives may be concentration range dependent. The human body burdens of PCB congeners in our study are compared to the cross-sectional data from the UK used in Ritter et al. (2011b) and longitudinal data for children in Grandjean et al. (2008) in Table S7 (see Supplementary material). Geometric means of all PCB congeners in our cross-sectional data are lower than those in the UK by a typical factor of 6, and much lower than those of longitudinal data for children (usually by a factor of 20 or more).
Further, the peak concentrations in Australians are much lower than the lowest concentrations in Grandjean et al. (2008). Therefore, the relatively lower range of human body burdens in our study may be another factor that is linked to longer intrinsic half-lives.

Literature evidence has shown that elimination of POPs in humans depends, to some extent, on the absolute level of body burdens (Leung et al., 2007; Milbrath et al., 2009). For example, Aylard et al. (2004) investigated the elimination of 2,3,7,8-tetrachlorodibenzo-p-dioxin in humans with different initial body burdens using sequential measurement. They found longer elimination half-lives for those individuals with lower initial body burdens. This phenomenon can be explained by the decreased metabolic activity for POPs at lower concentrations (Sorg et al., 2009). A similar observation has been reported in other studies (Kerger et al., 2006; Leung et al., 2005; Michalek et al., 2002).

Previous studies have speculated that the longest plausible intrinsic human elimination half-life for POPs is approximately 15 years (Kreuzer et al., 1997; Ritter et al., 2011b; Shirai and Kissel, 1996). Our results do not contradict this inference when considering the uncertainty in model estimation. However, our results highlight the possible importance of the absolute level of body burden on the elimination of POPs in humans, which requires further study.

5. Conclusions

For PCBs and OCPs in the Australian population, we are able to reconstruct intake levels and trends that are adequate to explain the time evolution of cross-sectional data representing the age-concentration structure. Plausible intrinsic half-lives that are in good agreement with other studies were derived using the Ritter model and biomonitoring data for the Australian population. Our results demonstrated the feasibility of using the Ritter population-level PK model to reconstruct intakes and to estimate intrinsic elimination half-lives from biomonitoring data. The possible importance of the absolute level of body burdens on the intrinsic elimination of POPs in humans was highlighted by our model results.

Acknowledgments

This research was funded by the European Union Seventh Framework Programme (FP7/2007–2013) under grant agreement #295138: Synergising International Studies of Environmental Contamination with Organic Flame Retardant Chemicals (INTERFLAME), (FP7-People-ITN-2010), project no. 264600. Indoor Contamination with Flame Retardant Chemicals: Causes and Impacts (INFLAME), the Department of the Environment, Australian Government, the Program for Changjiang Scholars and Innovative Research Team in University (IRT1261), and the Collaborative Innovation Center for Regional Environmental Quality. Jochen Mueller is funded by an ARC Future Fellowship (FF 120100546). Entox is a joint venture of the University of Queensland and Queensland Health. The National Research Centre for Environmental Toxicology is co-funded by Queensland Health.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/envint.2014.09.014.

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