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The assessment of daily dietary intake reveals the existence of a different pattern of bioaccumulation of chlorinated pollutants between domestic dogs and cats

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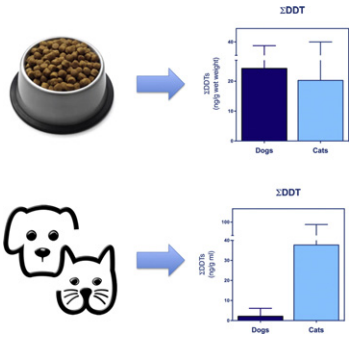
HIGHLIGHTS

- First assessment of the dietary intake of POPs in pet animals.
- Intake levels of pollutants are more than double in dogs than in cats.
- Proportionality between intake of PAHs and their plasma levels in both species.
- Lower levels of organochlorines in dog plasma, although their intake was higher.
- Dogs seem to be able of eliminating certain recalcitrant contaminants.

GRAPHICAL ABSTRACT



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ABSTRACT

Pet dogs and cats have been proposed as sentinel species to assess environmental contamination and human exposure to a variety of pollutants, including POPs. However, some authors have reported that dogs but not cats exhibit intriguingly low levels of some of the most commonly detected POPs, such as DDT and its metabolites. This research was designed to explore these differences between dogs and cats. Thus, we first determined the concentrations of 53 persistent and semi-persistent pollutants (16 polycyclic aromatic hydrocarbons (PAHs), 18 polychlorinated biphenyls (PCBs) and 19 organochlorine pesticides (OCPs)) in samples of the most consumed brands of commercial feed for dogs and cats, and we calculated the daily dietary intake of these pollutants in both species. Higher levels of pollutants were found in dog food and our results showed that the median values of intake were about twice higher in dogs than in cats for all the three groups of pollutants (ΣPAHs: 274.8 vs. 141.8; ΣOCPs: 233.1 vs. 83; ΣPCBs: 101.8 vs. 43.8 (ng/kg bw/day); respectively). Additionally, we determined the plasma levels of the same pollutants in 42 and 35 pet dogs and cats, respectively. All these animals lived indoors and were fed on the commercial brands of feed analyzed. As expected (considering the intake), the plasma levels of PAHs were higher in dogs than in cats. However, for organochlorines (OCPs and PCBs) the plasma levels were much higher in cats than in dogs (as much as 23 times higher for DDTs), in spite of the higher intake in dogs. This reveals a lower capacity of bioaccumulation of some pollutants in dogs, which is probably related with higher metabolizing capabilities in this species.

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1. Introduction

Certain environmental contaminants, including organochlorine pesticides (OCPs), and industrial products such as polychlorinated biphenyls (PCBs), are known for their toxicity and their resistance to degradation in the environment and biota. For these reasons they are included within the group of chemicals known as persistent organic pollutants (POPs) (El-Shahawi et al., 2010). Other compounds, such as polycyclic aromatic hydrocarbons (PAHs), strictly speaking cannot be considered as POPs because of their efficient metabolization. However, due to their high prevalence in the environment and their toxicity, they are frequently considered as such, and therefore are studied together (Lammel et al., 2013). It has been established that the ingestion of contaminated food contributes more than 90% to the total exposure to these compounds, and foodstuffs of animal origin are recognized as one of the main contributors (Almeida-González et al., 2012; Boada et al., 2014; Formigaro et al., 2014; Luzardo et al., 2012, 2013a; Malisch and Kotz, 2014; Rodríguez-Hernández et al., 2014; Schwarz et al., 2014; Zhou et al., 2012). As all these compounds are highly soluble in fat, their ingestion usually leads to bioaccumulation throughout the life and to biomagnification in the food chain (El-Shahawi et al., 2010; Safe, 1995). Numerous studies have revealed that many POPs, individually and in combination, may contribute to the development of severe health problems such as immune suppression, genotoxic effects, cancer, or endocrine disruption (Bergman et al., 2012; Boada et al., 2012; Kortenkamp et al., 2011; Lauby-Secretan et al., 2013; Valerón et al., 2009). For these reasons the majority of POPs have been banned or severely restricted (El-Shahawi et al., 2010).

Despite the time that has elapsed since the ban of many of these chemicals, today still relevant concentrations of many of them are detected, as witnessed by very recent studies (Boada et al., 2015; Henríquez-Hernández et al., 2011; Luzardo et al., 2014b; Storelli and Zizzo, 2014). Indeed, in some regions of the planet it has been reported that the levels of some compounds, such as PCBs, are even increasing (Garcia-Alvarez et al., 2014; Luzardo et al., 2014a). So, the monitoring of their levels in the environment remains a priority, especially, as regards to exposure of human populations (Diamond et al., 2015). This assessment of exposure to POPs can be done by directly measuring levels in biological samples donated by human volunteers. However, the assessment can be also performed by indirect estimates. Among these, calculations of the intake of pollutants in a given population to assess the exposure, or the employment of sentinel species are usually considered. Firstly, dietary intake estimations are made by combining food consumption data with the concentrations of contaminants found in food samples (Kesse-Guyot et al., 2013; Llobet et al., 2003; Luzardo et al., 2012; Zhou et al., 2012). These are studies that are usually linked to surveillance systems of human diseases in order to obtain quick and reliable information on the prevalence and occurrence of foodborne diseases and risks associated to food (Riviere et al., 2014; Veyrand et al., 2013). Additionally, this methodology has been also used to assess the exposure of animal species to pollutants (Formigaro et al., 2014). Secondly, all kinds of animals, which are convenient to sample, have been used to act as sentinels that allow the assessment of the environmental contamination status, and the estimation of the exposure of other species, including humans (Reif, 2011).

It seems obvious that the more suitable species to act as sentinels of human exposure are the pets, because they closely share the habitat with their owners. So, there are numerous authors that have explored the potential of dogs and cats in this sense (Andrade et al., 2010; Baker et al., 2005; Calderón-Garciduenas et al., 2001; Heyder and Takenaka, 1996; Rabinowitz et al., 2008; Reif, 2011). However, in the case of exposure to POPs the results have been variable, because although some authors have suggested that cats seem to be adequate sentinels of human exposure to these contaminants (Ali et al., 2013; Dirtu et al., 2013; Guo et al., 2012), the role of dogs as such does not seem so clear (Ruiz-Suárez et al., 2015; Sévère et al., 2015). One reason is that several authors have reported that, intriguingly, dogs and other canines exhibit extremely low levels of some of the more abundant POPs in most mammals (including cats and humans), such as DDE and DDT, which suggests a higher metabolic capacity of these animals (Georgii et al., 1994; Kunisue et al., 2005; Ruiz-Suárez et al., 2015; Sévère et al., 2015; Shore et al., 2001; Storelli et al., 2009). This is what led us to design the present investigation, to explore these differences between dogs and cats.

In light of the above, the objectives of the present study were the following: (1) To determine the levels of selected POPs (OCPs, PCBs, and PAHs) in commercial feed for dogs and cats; (2) to estimate the daily dietary intake of these POPs by dogs and cats on the basis of the recommended consumption of these feeds; (3) To analyze the plasma samples collected from two groups of domestic dogs and cats fed on these commercial feeds; and (4) to evaluate the potential differences in contaminant levels between both species in relation with their respective intakes.

2. Material and methods

2.1. Sampling

Blood samples of pet dogs (n = 42, 24 females and 18 males) and cats (n = 35, 19 females and 16 males) were collected during 2013–2014 through cephalic vein puncture. All samples were collected in the Veterinary Hospital of the University of Las Palmas de Gran Canaria (ULPGC, Canary Islands, Spain) during a routine care. Only clinically normal animals were included in the study after owner's consent. All the dogs and cats were adults. The mean age of dogs was 5.2 y.o. (range = 2–14), and the mean age of cats was 4.8 y.o. (range = 2–11). No statistically significant differences in age were observed between males and females. All the animals included in this study were healthy, lived inside the houses with their owners, and were fed with commercial feed. Samples of blood were collected in heparinized tubes and maintained at 4 °C. Plasma was separated after centrifugation and kept frozen at −20 °C in the Laboratory of Toxicology of the ULPGC until sample preparation for chemical analysis.

In addition, we made a random purchase of different brands of commercial feed for dogs and cats in supermarkets and specialty stores from Gran Canaria (Canary Islands, Spain). The feed brands were chosen having into account their composition, and were matched between species according to raw materials they contain. Samples were acquired in triplicate, and chosen among the top selling brands (7 brands of dog feed, and 9 brands of cat feed). All the samples were individually processed as described below, and we used the mean values of the triplicate samples of each brand in the calculations of dietary intake.

2.2. Chemicals, reagents and analytes of interest

All the organic solvents (dichlorometane, hexane, ethyl acetate, and cyclohexane) were of mass spectrometry grade (VWR International, PA, USA). Ultrapure (UP) water was produced in the laboratory using a Milli-Q Gradient A10 apparatus (Millipore, Molsheim, France). The inert desiccant (Celite ® 545) was purchased from Sigma-Aldrich (St. Louis, USA). Bio-Beads SX-3 were purchased from BioRad Laboratories (Hercules, USA). Standards of OCPs, PCB congeners, and internal standards (ISs, PCB 202, tetrachloro-m-xylene, p,p′-DDE-d8, heptachloro epoxide cis, and phenanthrene-d10), were purchased from Dr Ehrenstorfer, Reference Materials (Augsburg, Germany). Standards of PAHs were purchased from Absolute Standards, Inc. (Connecticut, USA). All standards were neat compounds. Stock solutions of each compound at 1 mg/ml were prepared in cyclohexane and stored at −20 °C. Diluted solutions from 0.05 ng/ml to 40 ng/ml were used for calibration curves (9 points).

We determined the levels of 53 organic compounds in plasma samples and commercial feed for dogs and cats: (a) 19 OCPs: methoxychlor;

dicofol; four isomers of hexachlorocyclohexane (α-, β-, γ-, and δ-HCH); p,p′-dichloro-diphenyl-trichloroethane (p,p′-DDT) and its metabolites (p,p′-DDE, and p,p′-DDD); hexachlorobenzene (HCB); aldrin; dieldrin; endrin; chlordane (cis- and trans-isomers); mirex; endosulfan (α- and β-isomers) and endosulfan-sulfate; (b) we also determined the levels of 6 marker (M-PCBs) and 12 dioxin-like PCBs (DL-PCBs) which were numbered according to the International Union of Pure and Applied Chemistry (IUPAC): IUPAC numbers # 28, 52, 77, 81, 101, 105, 114, 118, 123, 126, 138, 153, 156, 157, 167, 169, 180, 189; and (c) the 16 most environmentally relevant PAHs listed by the United States Environmental Protection Agency (US-EPA): naphthalene, acenaphthylene, acenaphtene, fluorene, anthracene, phenanthrene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo [b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno [1,2,3,-cd]pyrene, dibenzo[a,h]anthracene, and benzo[ghi]perylene.

2.3. Extraction and clean-up procedure

Plasma samples were subjected to solid-phase extraction using Chromabond® C18ec cartridges (Macherey-Nagel, Germany) that yielded recoveries in the range of 89–107% for the analytes studied. Before application of the plasma samples, the cartridges were cleaned and conditioned with three 1-ml volumes of methanol followed by four 1-ml volumes of ultrapure water. The samples were then passed through the cartridge by gravity flow. The plasma tubes were rinsed with four 1-ml aliquots of ultrapure water, and each aliquot was applied to the corresponding cartridge by gravity flow. The cartridges were rinsed with three 1-ml portions of ultrapure water and dried for 20 min under gentle vacuum (~15 mm Hg). The adsorbed analytes were eluted with 1 ml of methylene chloride by gravity flow. Gentle vacuum was then used to elute the residual methylene chloride from the cartridges. Solvent of the extracts was then evaporated under gentle nitrogen stream, and the extracted analytes were solubilized in 200 μl of cyclohexane that was transferred to 1.8-ml GC vials with 250 μl inserts (Chromatographic Research Supplies, Inc., USA). No additional purification was necessary for the plasma samples that were subjected to chromatographic analysis.

Because the contaminants included in this study are totally lipid-soluble and therefore found bound to the lipid fraction, when we extracted contaminants from the fish, we first extracted the fat of the fish. The mean fat content was 17.8% for Pacific herring, 0.7% for whiting and 17.9% for capelin. A total of 10 g of the homogenated fish were spiked with the 10-ppm surrogate's mix in acetone to yield a final concentration of 100 ppb and mixed with 30 g of diatomaceous earth to absorb all the humidity. The method of extraction and purification followed that recommended by the European Standard for the determination of pesticides and PCBs in fatty food (EN, 1996a,b), whose validity has been previously proven in our laboratory for fatty foods (Almeida-González et al., 2012; Hernández et al., 2015; Luzardo et al., 2012, 2013a). This method combines an automated Soxhlet extraction method (FOSS Soxtec Avanti 2055) with a purification step using gel permeation chromatography (GPC). This method gives acceptable recoveries that ranged between 74.5% and 104.7%. No additional purification steps were required, and the 1-ml extracts in cyclohexane obtained at the end of the GPC were used for the gas chromatography/triple quadrupole mass spectrometry (GC–MS/MS) analysis.

2.4. Procedure of chemical analysis

Gas chromatography analyses of 53 contaminants plus ISs were performed in a single run on a Thermo Trace GC Ultra equipped with a TriPlus Autosampler and coupled to a Triple Quadrupole Mass Spectrometer Quantum XLS (Thermo Fisher Scientific Inc., Waltham, MA, USA), as previously described (Luzardo et al., 2013b), and identifications were done using an electron ionization (EI)-MS/MS based on the retention time and the relative ion transition ratios of each of the analytes. Quantifications were performed against calibration curves as mentioned above. The limit of quantification was set at 0.01 ng/ml for all of the analytes.

2.5. Quality of analyses and quality control (QA/QC)

All of the measurements were performed in triplicate, and we used the means for the calculations. In each batch of samples, four controls were included for every 18 vials (6 samples): a reagent blank consisting of a vial containing only cyclohexane; a vial containing 2 ng/ml of each of the pollutants in cyclohexane; and two internal laboratory quality control samples (QCs) consisting of: a) blank serum (lyophilized human serum, Medidrug Basis Line, Medichem, Germany), and b) melted meat fat, both spiked at 20 ng/ml of each of the analytes. These QCs were processed using the same method as the plasma and feed samples, respectively. The results were considered to be acceptable when the concentration of the analytes determined in the QC sample was within 15% of the deviation of the theoretical value.

2.6. Dietary intake estimates and calculations

The exposure assessment of dogs and cats through feed consumption was calculated by multiplying the respective concentrations of the contaminants in the extracted feed fat (mean values) by the amount of fat contained in the average daily feed consumption, as recommended by the manufacturer for either dogs or cats, and divided by the body weight (bw) of each animal. Thus we obtained the consumption of each one of the pollutants expressed in ng/kg bw/day. The exposures were assessed for all of the contaminants, both individually considered and also grouped in different forms. For the calculations, when the concentration of a given contaminant was below the limit of quantification (LOQ) but above the limit of detection (LOD) of the technique, the value was assumed to be ½ LOQ. Otherwise the value was considered to be 0.

In this work, we expressed the total value of the OCP residues (ΣOCPs) as the sum of the 19 OCPs and metabolites measured; the total value of the DDTs (ΣDDT) was expressed as the sum of the measured values of p,p′-DDT, p,p′-DDE and p,p′-DDD; the total value of the HCH residues (ΣHCH) was expressed as the sum of the 4 HCH isomers measured (α-, β-, δ- and γ-HCH); and the total value of the cyclodiene residues (Σcyclodienes) was expressed as the sum of aldrin, dieldrin, endrin, cis-chlordane, trans-chlordane and heptachlor. PCB congeners: 6 (M-PCB = PCBs #28, 52, 101, 138, 153 and 180) and 12 dioxin-like PCBs (DL-PCB = PCBs #77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169 and 189). Finally, we considered the total content of the PAHs (ΣPAHs) as the sum of the values of the 16 US-EPA compounds included in this study.

2.7. Statistical analysis

Database management and statistical analysis were performed with PASW Statistics v 19.0 (SPSS Inc., Chicago, IL, USA). To determine whether the data were normally distributed, we used the Kolmogorov–Smirnov test. Because the levels of organic pollutants did not follow a normal distribution, we used a non-parametric test. The differences between two independent groups were tested with the Mann–Whitney *U*-test. The results were reported as medians and ranges (or interquartile ranges). Probability levels of less than 0.05 (two tailed) were considered statistically significant.

3. Results and discussion

As expected, our results showed that many of the studied pollutants were present in all of the commercial feed samples, as well as in cat and dog plasma samples, indicating that dogs and cats are continuously exposed to POPs through their diets. Following the objectives of this research we present the levels of the studied pollutants (OCPs, PCBs,

and PAHs) in feed in Table 1; the estimated intake of these contaminants in Table 2; and the plasma levels found in dog and cat plasma in Table 3. Moreover, in all the three tables we show the results of the statistical comparison and, additionally we present a graphical representation of the most relevant differences in Figs. 1 to 3. There are very few references in the literature that have examined the levels of PCBs, OCPs and PAHs in the blood and tissues of pets (Ali et al., 2013; Kunisue et al., 2005; Ruiz-Suárez et al., 2015; Storelli et al., 2009), and even less in commercial feed for these species (Kunisue et al., 2005). This paper studies a wide range of these compounds in two groups of domestic dogs and cats, and in their food, comparing between them. It also presents two characteristics that confer its originality: as far as we know, a) this is the first time that dietary intake of contaminants in these species is evaluated, b) is also the first time that levels of PAHs are reported in cats. In the following sections we present and discuss the results in detail.

Table 1
Concentrations of POPs in commercial feed for dogs and cats. All the results are presented as median, percentiles 25th–75th, and frequency of detection, and expressed in ng/g (fresh product weight).

Name	Commercial feed for dogs			Commercial feed for cats			*p*
	Median	P25–P75	Freq.	Median	P25–P75	Freq.	
	(ng/g w.w.)		(%)	(ng/g w.w.)		(%)	
Polycyclic aromatic hydrocarbons (PAHs)							
Naphtalene	0.50	0.42–1.24	100.0	0.22	0.13–0.55	100.0	n.s.
Acenaphthene	0.31	0.09–0.37	100.0	0.14	0.12–1.12	100.0	n.s.
Acenaphthylene	0.38	0.0–0.69	57.1	0.0	0.0–0.20	22.2	n.s.
Fluorene	0.18*	0.08–0.50	100.0	0.03	0.0–0.10	55.5	0.011*
Phenanthrene	12.1*	6.33–14.53	100.0	3.75	2.61–6.79	100.0	0.010*
Anthracene	–	–	n.d.	–	–	n.d.	–
Fluoranthene	3.75**	3.40–4.22	100.0	2.15	1.15–2.97	100.0	0.003**
Pyrene	3.84	3.71–5.21	100.0	2.44	2.10–4.53	100.0	n.s.
Benzo[a]anthracene	0.0	0.0–0.0	14.3	0.0	0.0–0.07	33.3	n.s.
Chrysene	0.0	0.0–0.27	42.8	0.0	0.0–0.08	22.2	n.s.
Benzo[b]fluoranthene	0.0	0.0–0.0	14.3	0.0	0.0–0.15	22.2	n.s.
Benzo[k]fluoranthene	0.0	0.0–0.16	28.6	0.0	0.0–0.07	33.3	n.s
Benzo[a]pyrene	–	–	n.d.	–	–	n.d.	–
Dibenzo[a,h]anthracene	–	–	n.d.	–	–	n.d.	–
Benzo[g,h,i]perylene	–	–	n.d.	–	–	n.d.	–
Indeno[1,2,3-c,d]pyrene	–	–	n.d.	–	–	n.d.	–
Organochlorine pesticides (OCs)							
Hexachlorocyclohexane (alpha)	0.02	0.0–0.10	71.4	0.0	0.0–0.1	77.7	n.s.
Hexachlorocyclohexane (beta)	0.33	0.02–0.66	85.7	0.06	0.03–0.29	100.0	n.s.
Hexaclorocyclohexane (gamma)	0.27	0.01–0.57	71.4	0.12	0.05–0.023	88.8	n.s.
Hexachlorocyclohexane (delta)	0.01	0.0–0.09	14.3	0.0	0.0–0.1	22.2	n.s.
Heptachlor	–	–	n.d.	–	–	n.d.	–
Aldrin	0.04	0.0–0.21	57.1	0.01	0.0–0.03	66.6	n.s.
Dieldrin	13.22*	6.14–17.25	100.0	4.68	3.05–9.18	100.0	0.0229*
Endrin	–	–	n.d.	–	–	n.d.	–
Chlordane (cis)	–	–	n.d.	–	–	n.d.	–
Chlordane (trans)	–	–	n.d.	–	–	n.d.	–
Endosulfan (alpha)	–	–	n.d.	–	–	n.d.	–
Endosulfan (beta)	–	–	n.d.	–	–	n.d.	–
Endosulfan sulfate	–	–	n.d.	–	–	n.d.	–
p,p′-DDT	0.67	0.53–0.81	71.4	0.53	0.33–0.89	88.8	n.s.
p,p′-DDE	0.43	0.29–1.31	100.0	0.32	0.24–1.76	100.0	n.s.
p,p′-DDD	0.05	0.02–0.08	14.3	0.04	0.0–0.13	22.2	n.s.
Dicofol	–	–	n.d.	–	–	n.d.	–
Metoxychlor	–	–	n.d.	–	–	n.d.	–
Mirex	–	–	n.d.	–	–	n.d.	–
Polychlorinated biphenyls (PCBs)							
PCB 28	5.54*	2.38–5.79	100.0	1.82	0.69–2.63	100.0	0.0079*
PCB 52	1.81**	0.61–1.90	100.0	0.41	0.26–0.92	100.0	0.0115**
PCB 77	0.03	0.01–0.04	85.7	0.01	0.0–0.04	66.6	n.s.
PCB 81	0.01	0.0–0.02	71.4	0.01	0.0–0.02	66.6	n.s.
PCB 101	0.56**	0.28–0.59	100.0	0.19	0.11–0.29	100.0	0.0079**
PCB 105	0.03	0.02–0.04	100.0	0.02	0.01–0.03	88.8	n.s.
PCB 114	0.0	0.0–0.01	28.6	0.0	0.0–0.0	33.3	n.s.
PCB 118	0.15	0.08–0.16	100.0	0.06	0.04–0.18	100.0	n.s.
PCB 123	0.0	0.0–0.01	14.3	0.0	0.0–0.01	33.3	n.s.
PCB 126	–	–	n.d.	–	–	n.d.	–
PCB 138	0.05	0.04–0.06	100.0	0.06	0.03–0.09	100.0	n.s.
PCB 153	0.07	0.05–0.17	100.0	0.09	0.03–0.14	100.0	n.s.
PCB 156	–	–	n.d.	–	–	n.d.	–
PCB 157	–	–	n.d.	–	–	n.d.	–
PCB 167	–	–	n.d.	–	–	n.d.	–
PCB 169	–	–	n.d.	–	–	n.d.	–
PCB 180	0.02	0.0–0.07	100.0	0.02	0.0–0.05	88.8	n.s.
PCB 189	–	–	n.d.	–	–	n.d.	–

n.d. non detected.
n.s. not significant.
* $P < 0.05$ ** $P < 0.01$.

Table 2
Values of daily dietary intakes of POPs (ng/kg b.w./day) in domestic dogs and cats which are feed on commercial feed. All the results are presented as mean ± SD, median, and range.

	Daily intake (dogs)			Daily intake (cats)			
Name	Mean ± SD	Median	Range	Mean ± SD	Median	Range	*p*
Polycyclic aromatic hydrocarbons (PAHs)							
∑ c-PAHs[a]	8.5 ± 16.4	0.0	0.0–44.3	6.0 ± 12.7	1.23	0.0–39.5	n.s.
∑ PAHs	**323.6 ± 102.1**	**274.8**	**244.4–511.8**	183.6 ± 104.9	141.8	71.6–409.2	0.023*
Organochlorine pesticides (OCs)							
∑ HCH	**10.5 ± 8.9**	**8.0**	**1.0–27.3**	2.8 ± 2.9	1.5	0.3–9.7	0.031*
∑ DDTs	24.3 ± 14.3	17.4	13.6–53.1	20.3 ± 19.7	15.6	3.4–71.6	n.s.
∑ cyclodienes	**179.3 ± 59.3**	**207.0**	**84.2–255.0**	99.8 ± 73.3	68.4	24.3–265.3	0.031*
∑ OCPs	**214.0 ± 60.5**	**233.1**	**119.0–299.7**	122.9 ± 92.7	83.0	29.6–340.6	0.030*
Polychlorinated biphenyls (PCBs)							
∑ M-PCBS	**99.9 ± 28.0**	**98.5**	**56.0–132.8**	48.0 ± 29.3	42.7	22.2–106.8	0.008**
∑ DL-PCBS	3.3 ± 0.5	3.2	2.5–4.2	3.3 ± 3.5	1.9	0.9–12.2	n.s.
∑ $TEQ_{DL\text{-}PCBs}$[b]	0.4 ± 0.1	0.4	0.24–0.5	0.3 ± 0.3	0.3	0.1–1.1	n.s.
∑ PCBs	**103.2 ± 27.7**	**101.8**	**59.4–136.3**	41.4 ± 32.1	43.8	24.1–119.0	0.012*

Figures in bold indicate the group of samples having a significantly higher value.
n.d. non detected.
n.s. not significant.
* $P < 0.05$ ** $P < 0.01$.
[a] Carcinogenic PAHs = the sum of the 8 compounds for which there are evidences of carcinogenicity: benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[ghi]perylene, chrysene; dibenz[a,h]anthracene; and indeno[1,2,3-cd]pyrene.
[b] Expressed in pg/g wet weight.

3.1. Levels of PAHs, OCPs and PCBs in feed for dogs and cats, and estimation of their daily intake through diet

3.1.1. PAHs

Eleven out of the sixteen priority USEPA PAHs were detected in the commercial feed samples analyzed in this work. The compounds detected were the same in feed for dogs than in feed for cats. Only anthracene, benzo[a]pyrene, dibenzo[a,h]anthracene, benzo[*ghi*]perylene and indeno[1,2,3,-*cd*]pyrene were not detected in any of the samples. On the contrary, naphthalene, acenaphthene, phenanthrene, fluoranthene, and pyrene were detected in 100% of the samples of feed for both species, and in addition fluorene was also detected in 100% of the samples of feed for dogs (Table 1). When we considered the concentrations of these contaminants (and not the frequencies), we found that the levels were in general higher in feed for dogs than in feed for cats, and for fluorene, phenanthrene, and fluoranthene these differences reached statistical significance (Table 1). Thus, the ΣPAHs was also significantly higher in dog food (median = 21.86 vs. 7.58 ng/g w.w., in feed for dogs and cats, respectively; $P < 0.05$). According to the levels found in the food, it was expected that the daily intake levels of these contaminants were also higher in dogs than in cats, as indeed our estimates demonstrated. So, intake of ΣPAHs was 274.8 ng/kg bw/day in dogs and 141.8 ng/kg bw in cats ($p < 0.05$) (Table 2 and Fig. 1, left).

3.1.2. OCPs

Within this group of organochlorine compounds we detected nine out of the nineteen compounds analyzed (Table 1) in the analyzed feed samples, and the compounds detected were the same in both types of feed. Heptachlor, endrin, cis-chlordane, trans-chlordane, α-endosulfan, β-endosulfan, endosulfan-sulfate, dicofol, metoxychlor, mirex were not detected in any of the samples. The frequencies of detection of each of the OCPs were pretty similar between the samples of feed for dogs and feed for cats. However, the median concentrations were found to be higher for all of them in the group of feeds for dogs (although only statistically significant for dieldrin). Therefore, the median levels of these compounds reached statistical significance when summed (median ΣOCPs = 14.84 vs. 6.24 ng/g in feed for dogs and cats, respectively, $p < 0.05$). Subsequently, as also occurred with PAHs, our estimations showed that the daily intake of OCPs were also higher in dogs than in cats (Table 2). As observed in this table, dogs consume statistically significant higher levels of ΣHCHs, Σcyclodienes, and ΣOCPs ($p < 0.05$ in all cases), whereas the intake of ΣDDTs can be considered almost the same in both species.

As far as we know, only one previous study has analyzed the levels of organochlorine contaminants (OCPs and PCBs) in food for domestic animals (canned food for dogs and cats). In this study the residues detected were similar in both types of food, either in frequency and in concentration (Kunisue et al., 2005). Although no estimation of intake was performed in that work, given the results it is probable that the dietary exposure would be also the same in both species. Other authors have studied the levels of pollutants in pet food, but these other studies have focused in food contaminants as the cause of toxic outbreaks (Dobson et al., 2008; Rumbeiha and Morrison, 2011), and therefore do not have considered the food as a source of exposure to environmental pollutants.

3.1.3. PCBs

All the M-PCBs and 6 out of the 12 DL-PCBs were detected in both types of feed samples. The congeners #126, 156, 157, 167, 169 and 189 were not detected in any of the samples. As occurred with the other two groups of chemicals, the frequencies of detection of PCBs were similar between samples of feed for dogs and samples of feed for cats, but the concentrations were higher in samples of feed for dogs. Indeed, the levels detected were more than three times higher, when we considered all the congeners as a group (8.49 vs. 2.37 ng/g, $p < 0.05$), and also individually for PCB 28 ($p < 0.01$), PCB 52 ($p < 0.05$), and PCB 101 ($p < 0.01$) (Table 1). It is remarkable that all of the M-PCBs were detected in 100% of the samples analyzed, and that this group of congeners accounted for >95% of ΣPCBs in both types of feed. Thus, also similar to that described above, the estimated dietary exposure to these pollutants is much higher in dogs than in cats (median ΣPCBs intake = 101.8 vs. 43.8 ng/kg bw/day, $p < 0.05$) (Table 2). Probably due to the enormous difference of concentration between ΣM-PCBs and ΣDL-PCBs in feed, the higher exposure in dogs seems to be only attributable to the intake of M-PCBs (no statistically significant differences in the intake of ΣDL-PCBs nor of $\Sigma TEQs_{DL\text{-}PCBs}$ (Van den Berg et al., 2006) were found).

3.2. Plasma levels of PAHs, OCPs, and PCBs in dogs and cats

3.2.1. PAHs

As shown in Table 3, and consistently with the levels found in feed and the estimation of daily intake, the plasma levels of ΣPAHs were significantly higher in dogs than in cats (422.5 vs. 253, respectively;

Table 3
Concentrations of POPs (mean ± SD, median and range) in plasma samples of dogs and cats, which are feed on commercial feed. Results are expressed in ng/g lw.

	Plasma levels (dogs)			Plasma levels (cats)			
Name	Mean ± SD	Median	Range	Mean ± SD	Median	Range	*p*
Polycyclic aromatic hydrocarbons (PAHs)							
∑ c-PAHs[a]	1.8 ± 2.5	1.0	0.0–9.1	4.6 ± 20.2	0.0	0.0–120.0	n.s.
∑ PAHs	**447.3 ± 189.9**	**422.5**	**230.0–995.0**	346.3 ± 211.4	253.0	52.0–962.0	0.025*
Organochlorine pesticides (OCs)							
∑ HCH	2.7 ± 7.9	0.0	0.0–25.1	**3.1 ± 3.7**	**2.0**	**0.0–14.0**	<**0.001****
∑ DDTs	2.1 ± 3.9	0.0	0.0–15.0	**49.2 ± 90.4**	**12.0**	**1.0–441.0**	<**0.001****
∑ cyclodienes	26.8 ± 25.4	25.0	0.0–125.4	40.5 ± 43.3	26.3	0.0–174.2	n.s.
∑ OCPs	31.6 ± 29.9	25.1	0.0–152.3	**92.7 ± 122.7**	**47.9**	**3.1–618.0**	**0.022***
Polychlorinated biphenyls (PCBs)							
∑ M-PCBS	67.1 ± 61.6	50.1	0.0–225.7	**87.7 ± 43.5**	**87.0**	**24.1–216.4**	0.040*
∑ DL-PCBS	6.1 ± 8.3	2.5	0.0–30.9	3.3 ± 3.7	2.0	0.0–21.2	n.s.
∑ $TEQ_{DL\text{-}PCBs}$[b]	3.9 ± 6.7	0.4	0.0–25.9	1.8 ± 5.5	0.4	0.1–30.7	n.s.
∑ PCBs	73.1 ± 69.2	50.8	0.0–255.1	**90.9 ± 45.7**	**89.2**	**22.3–221.6**	0.039*

Figures in bold indicate the group of samples having a significantly higher value.
n.d. non detected.
n.s. not significant.
[a] Carcinogenic PAHs = the sum of the 8 compounds for which there are evidences of carcinogenicity: benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[ghi]perylene, chrysene; dibenz[a,h]anthracene; and indeno[1,2,3-cd]pyrene.
[b] Expressed in pg g^{-1} wet weight.

$P < 0.05$). On the contrary, when we considered only those PAHs that are classified as probable carcinogens for animals and humans by the USEPA (benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene and indeno[1,2,3,-cd]pyrene), no differences were observed between dog and cat plasma samples, but this was also consistent with the intake levels in both species (Table 2). As shown in Fig. 1, proportionality was found between the intake and the plasma levels in both species. Unlike the rest of pollutants included in this study, the PAHs are readily metabolized in vertebrates (Walker et al., 2006), and although high differences in metabolizing capabilities among species have been reported between dogs and cats (Saengtienchai et al., 2014), in this case the differences found seem to be mainly attributable to the higher intake in dogs (Fig. 1).

According to the literature only one previous study has reported the plasma levels of PAHs in dogs (Ruiz-Suárez et al., 2015), and none in cats. The levels reported in dogs by Ruiz-Suárez et al. (2015) were similar to those found in this series.

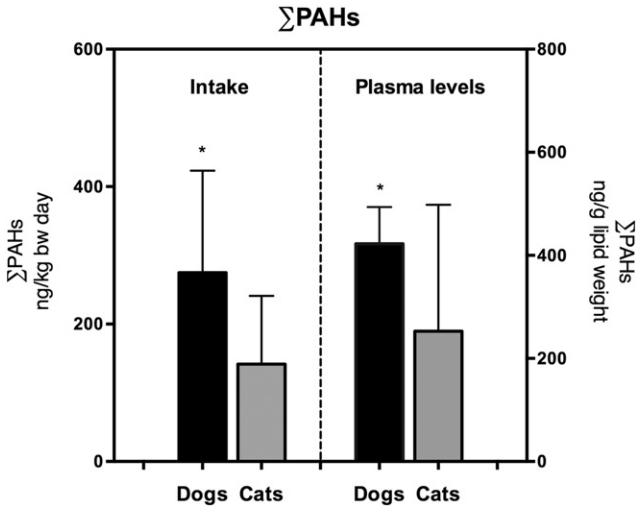

Fig. 1. Comparison of the intake of ΣPAHs through food in dogs and cats (left), and of the plasma levels of ΣPAHs in both species (right). The bar height represents the median and the lines extending vertically from the bar indicate the interquartile ranges.

3.2.2. OCPs

Although, as described above, levels of OCPs in feed and their dietary intake were significantly higher in dogs than in cats; the plasma levels found in cats showed an inverse pattern, with median values almost doubling those in dogs (Table 3 & Fig. 2). The greatest difference found was for ΣDDTs, whose mean plasma levels in cats were about 23 times higher than in the dogs (Fig. 2, inset), even though the similar

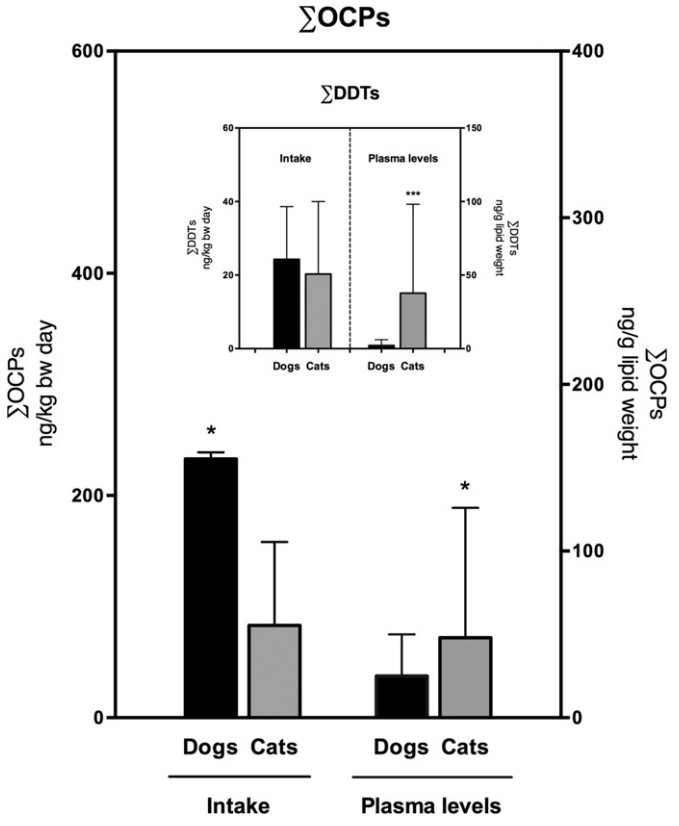

Fig. 2. Comparison of the intake of ΣOCPs through food in dogs and cats (left), and of the plasma levels of ΣOCPs in both species (right). The bar height represents the median and the lines extending vertically from the bar indicate the interquartile ranges. Inset: representation of the detailed comparison of the subgroup of DDTs.

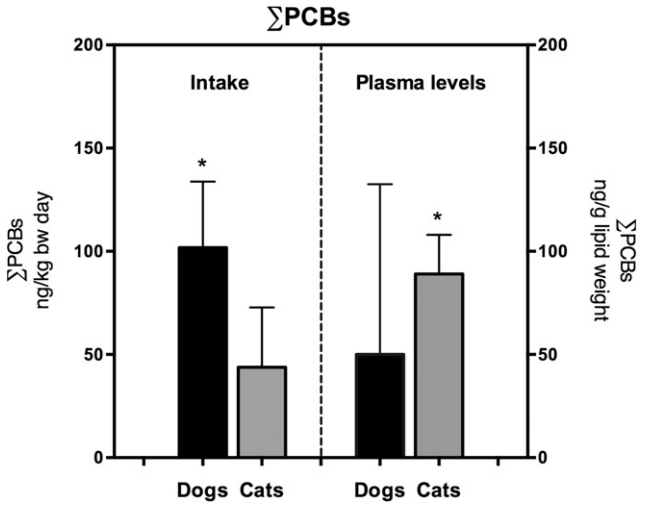

Fig. 3. Comparison of the intake of ΣPCBs through food in dogs and cats (left), and of the plasma levels of ΣPCBs in both species (right). The bar height represents the median and the lines extending vertically from the bar indicate the interquartile ranges.

dietary exposure to these contaminants in both species. This was no an unexpected result, as other authors have also found low or even almost undetectable levels of these pesticides in dogs (Ali et al., 2013; Kunisue et al., 2005; Ruiz-Suárez et al., 2015), other canines (Shore et al., 2001), and some other species of vertebrates, such as foxes or polar bears (Hoshi et al., 1998; Polischuk et al., 2002). Some of these authors have pointed to the possibility that dogs (and maybe other species) would have an uncommon capacity of metabolizing these pesticides. However, it was also possible that these animals were exposed to lower levels of contaminants through diet. In this work we have demonstrated that these animals do not accumulate OCPs in the same manner than cats (or other vertebrates), although the dietary exposures were comparable to each other. Furthermore, according to our results this does not seem to be only a particularity of DDT. So, we also report here that dogs, although they are 5 times more exposed to HCHs through diet than cats, they exhibit much lower plasma levels (Table 3), which would also suggest a higher metabolic capability for these other pesticides in dogs.

3.2.3. PCBs

Similarly to what was observed with the group of OCPs, also plasma levels of PCBs were significantly higher in cats than in dogs ($p < 0.05$) (Table 3), although dietary intake was calculated over twice in dogs than in cats (Table 2). These differences were observed both, for ΣPCBs and ΣM-PCBs, but not for ΣDL-PCBs, nor $\Sigma TEQ_{DL\text{-}PCBs}$. In Fig. 3 we graphically show this inverse relationship. Interestingly, Ali et al. (2013) reported higher levels of PCB metabolites (OH-PCBs) in dogs than in cats, and Kunisue et al. (2005) suggested that cats hardly metabolize and eliminate PCBs. Elaborating on this, other authors have shown that dogs are capable of metabolizing some PCB congeners by means of the cytochrome P-450 (CYP2B) (Ariyoshi et al., 1992; Duignan et al., 1987), so that they can eliminate them more quickly and more efficiently than other mammalian species (Sipes et al., 1982a,b). Reversely, in the study of Ali et al. (2013) the authors reported lower levels for 4OH-PCB 146 (a major metabolite of PCB 153) in cats compared with dogs, suggesting that cats hardly metabolize PCB 153, and Kunisue and Tanabe (2009) indicated that cats may preferentially metabolize lower chlorinated OH-PCBs and retain these metabolites in their blood. Unfortunately, due to technical limitations we were not able to determine the levels of OH-PCBs in these groups of animals. However, other authors have used the concentration ratio of PCB 153/PCB 180 as an indicator of specific metabolism of PCB 153 (Storelli et al., 2009), and found a big difference between dogs and cats (0.3 vs. 1.8), which suggests the ability of dogs to metabolize this highly persistent congener. When we applied this indirect indicator of metabolism our results were consistent with those of Storelli et al. (2009), with significantly lower ratios in dogs than in cats (median = 0.52 vs. 2.33, $p < 0.001$).

Taken together, our results indicate that there are great disparities between the capacities to metabolize organochlorine contaminants between dogs and cats. Thus, we agree with previous authors in that cats may serve as better sentinels of human exposure to environmental organochlorine compounds and for the control of geographical variations of these pollutants than canine species (Dirtu et al., 2013; Kunisue et al., 2005; Storelli et al., 2009). However, we reinforce here the findings of Ruiz-Suárez et al. (2015), who indicated that dogs are not good sentinels of the exposure to POPs, since we add evidence that this species pose higher metabolic capabilities to deal with these compounds than the majority of mammals.

4. Conclusions

The findings of this study reveal that dogs exhibit lower plasma levels of OCs than cats, even though it was estimated that this species is exposed to higher dietary levels when fed on commercial feed. All the analytes detected in the feed were detected in the plasma samples of both, dogs and cats, but the levels of many organochlorine compounds were from 2 to 23 less abundant in dog plasma samples than in cat plasma samples. Taken into account that it has been reported that more than 90% of exposure to POPs is through food in vertebrates, our results suggest that dogs, unlike the majority of mammals, seem to be able to efficiently metabolize and eliminate some POPs. Therefore domestic cats, instead of dogs, may represent a better model to assess human exposure to these chemicals.

Conflict of interest

The authors declare no conflict of interest.

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