Chapter 4
Role of Pet Dogs and Cats as Sentinels of Human Exposure to Polycyclic Aromatic Hydrocarbons

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Abstract Polycyclic aromatic hydrocarbons (PAHs) are a large group of chemical contaminants, predominantly produced via fossil fuel combustion. They spread easily worldwide, so they are considered as semipersistent pollutants. Many of them are considered as carcinogenic or mutagenic compounds, for example, interacting directly with DNA. Benzo(a)pyrene (BaP) is the most important and well-known PAH. Living beings are exposed everyday through air, water, plastic stuff and smoke and almost by food intake, because they are highly lipophilic. In human risk assessments, monitoring these compounds, or their products, in environment, biological or food samples has attracted enormous interest. Pets commonly share habitat and routine life with humans. In this chapter, the possibility that pets were good sentinels of human exposure to PAHs is studied in detail. Concentrations of parental PAHs and some metabolites between human and pets have been compared. In the case of dogs, their concentrations and profiles of PAHs are very different to those of humans when compared. Dogs had lesser concentration of parental compounds and higher concentration of their metabolites than humans. Similarly, cats present different concentrations and detection frequencies than humans. Therefore, the scarce data available indicate that dogs and cats seem to have different sources of exposition to PAHs than humans. Although more studies are needed, pets do not seem to be good sentinels for human exposure to PAHs.

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M. R. Pastorinho, A. C. A. Sousa (eds.), *Pets as Sentinels, Forecasters and Promoters of Human Health*, https://doi.org/10.1007/978-3-030-30734-9_4

Keywords Benzo(a)pyrene · Dogs · Cats · Semipersistent organic pollutants · Sentinels · Carcinogenic chemicals · Polycyclic aromatic hydrocarbons · Pets · Sentinel · Biomonitoring

4.1 Introduction

Polycyclic aromatic hydrocarbons or polynuclear aromatic hydrocarbons (PAHs) are a large class of organic compounds (more than a hundred are known) made from carbon and hydrogen, formed by more than two benzene rings fused and organised on linear, angular or cluster structure. According to their molecular weight, they can be classified as low-molecular-weight PAHs (LMW-PAHs, up to three fused rings) or as high-molecular-weight PAHs (HMW-PAHs, minimum of four rings).

Generally, they are colourless, white or yellowish solid at room temperature; have low vapour pressure, high melting and boiling points and low water solubility; and are hence highly lipophilic (WHO 1998). The most harmful and best-known PAH is benzo(a)pyrene (BaP), but there are many other PAHs of concern (Fig. 4.1) because of their toxicity, human exposure, occurrence in the environment and scope of available information. According to the list of priority pollutants of the United States Environmental Protection Agency (USEPA), there are 16 priority PAHs, because of their occurrence and the fact that they are continuously emitted to the environment (ATSDR 1995). More important are those PAHs that have been identified as mutagenic/teratogenic/carcinogenic by the Joint FAO/WHO Expert Committee on Food Additives (JECFA) (Fig. 4.1). Based on data from oral bioassays conducted in mice with coal tar mixtures, the JECFA calculated margin of exposure values of 25,000 and 10,000 between the BMDL10 value of 100 μg of benzo[a]pyrene/kg bw/day and mean and 95th-percentile intake levels of 4 and 10 ng/kg bw/d, respectively.

According to their origin, PAHs can be classified as pyrolytic (high temperature), petrogenic (high pressure) or biological (synthesised by microorganisms). Besides, they can be disguised between anthropogenic (combustion of fossil fuels, principally) and natural (forest fires, volcanos, fossil fuel formation, vegetal matter decomposition) sources, although the latter have a minimal contribution to the total environment burden.

Granting that they have no utility per se, PAHs are used as intermediaries in different industries, namely, in the manufacture of pharmaceutical products, polyvinyl chloride (PVC) and plasticisers (naphthalene), pigments (acenaphthene, pyrene), dyes (anthracene, fluoranthene) and pesticides (phenanthrene) (WHO 1998). Nevertheless, production, processing and use of fossil fuels principally coal – and, to a lesser extent, oil and natural gas – for industries, heating or transportation in cities, are the main source of emission of these contaminants to the environment (Cabuk et al. 2014; Villar-Vidal et al. 2014). Concerning traffic, petrol-fuelled vehicles can emit greater amounts of fluoranthene and pyrene, whilst diesel-fuelled

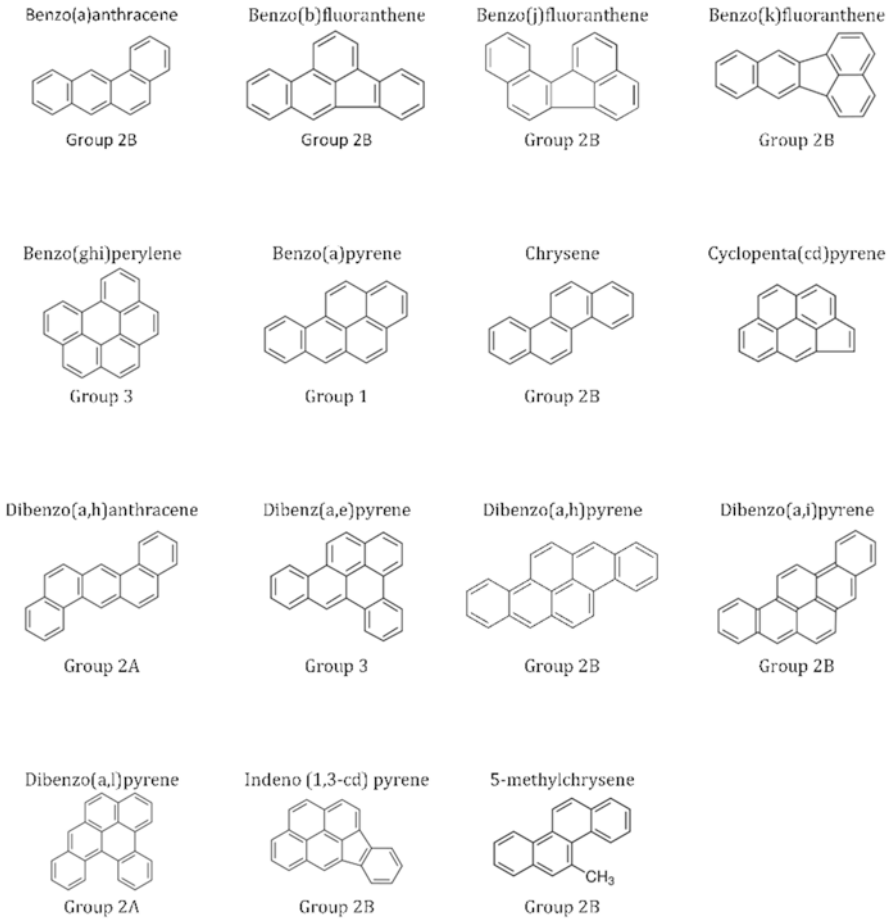

Fig. 4.1 Polycyclic aromatic hydrocarbons for which there is clear evidence of mutagenicity/genotoxicity in somatic cells in experimental animals in vivo and, with the exception of benzo(ghi)perylene, which have also shown clear carcinogenic effects in various types of bioassays in experimental animals

vehicles emit naphthalene and acenaphthene. In the case of smoking, cooking or burning (of stubble, garbage, tyres or other types of waste), a great variety of different compounds are emitted, including the ones already mentioned.

After being formed, these hydrocarbons are dispersed in the environment according to their molecular weight and climate conditions (Kozak et al. 2003). Thus, the HMW-PAHs can be adsorbed into the organic matter of the soil, water or air, whilst the LMW-PAHs will become a part of the gas phase in the atmosphere (Li et al. 2015). Both can be transported over long distances in several weeks until they are precipitated and/or degraded by solar light or microorganisms in the soil or sediments (Walgraeve et al. 2010). Along the way, they can react with different airborne

compounds, namely, sulphur oxides, nitrogen oxides or ozone, resulting in no less toxic combinations (Li et al. 2015; Walgraeve et al. 2010), like nitro-/oxy-PAHs and radicals formation.

4.2 Sources of Exposure and Health Effects

Humans and other living beings can be exposed to PAHs through inhalation or dermal/mucosa contact or mainly through water and food intake (Boada et al. 2016; Henriquez-Hernandez et al. 2017b; Hernandez et al. 2015, 2017; Luzardo et al. 2013a; Rodríguez-Hernández et al. 2015b, 2016, 2017). Inhalation is an important source in smokers and people who live near or in big cities or industrialised zones, where ten times higher concentrations of PAHs than in rural areas can be found (de la Gala Morales et al. 2015; Srogi 2007). Several authors have described higher concentration of PAHs in winter than in summer because of increased use of domestic heating (de la Gala Morales et al. 2015; Li et al. 2015; Villar-Vidal et al. 2014).

It has also been described that dermal exposure may be relevant, mainly when prolonged or continued contact with products made of petroleum derivatives occurs. Recently, the European Union, through the REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals) regulation, established new restrictions about PAHs in several day-to-day stuffs made of plastics or rubber, which are in direct, prolonged or short-term repetitive contact with human skin or mucosa. These items should not contain more than 10 mg/kg of the sum of benzo(a)pyrene (BaP), benzo(e)pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(j)fluoranthene, benzo(k)fluoranthene and dibenzo(a,h)anthracene, or no more than 1 mg/kg of BaP alone (ECHA 2017).

Although parental PAHs are generally inert, once in the organism, PAHs can be metabolised to be eliminated, generally via urine (LMW-PAHs) or faeces (HMW-PAHs). By the way, the process may result in active PAH metabolites (m-PAHs) capable of forming adducts with the DNA (Boada et al. 2016; Ramesh et al. 2004; Rodríguez-Hernández et al. 2015a, Hernandez et al. 2017). The biotransformation process is carried out through a series of enzymes like cytochrome P-450, which catalyses mainly oxidation, reduction and hydrolysis reactions. In vertebrates, the liver is the major contributor in the biotransformation process. However, in other organs there are cytochromes, which are able to perform this function according to the entryway (i.e. lungs, intestine or skin) (Ramesh et al. 2004). In addition, conjugation enzymes such as sulphotransferases, epoxide hydrolase, glutathione transferase and UDP-glycosyltransferase can metabolise PAHs, producing a variety of phenols, catechols, quinones and radical cations. Once they are formed, these compounds may produce adverse effects by means of various mechanisms, such as DNA damage diol-epoxides (that give place to formation of adducts), interaction with membranes and oxidative stress (Li et al. 2015; Sikkema et al. 1994; Zhang et al. 2016). Given that, some PAHs are described as carcinogenic (c-PAHs), namely, human carcinogen BaP (Group 1), whilst others are considered as probably (Group 2A) or

possibly carcinogenic (Group 2B) by the International Agency for Research on Cancer (IARC 2005). All these compounds have been described as contributing causes of breast, bladder, lung, skin or gastrointestinal cancers (Alicandro et al. 2016; Boada et al. 2015, 2016; Flesher and Lehner 2016; Korsh et al. 2015).

Often mixtures of hydrocarbons and/or their derivatives (such as nitro-PAH) are more harmful, due in part to synergistic properties. In general terms, the lower the molecular weight, the lower the carcinogenicity potential, but they are more prone to cause acute health effects, such as cardiovascular diseases (thrombosis, haematopoietic effects), dyspnoea, asthma (Al-Daghri et al. 2014), diarrhoea, vomiting, nausea and eye, dermal or bronchial irritation or inflammation (Ramesh et al. 2004). Moreover, it is well known that some PAHs are endocrine disruptors in animals and humans. Neurological, congenital and development problems in the offspring and mothers (Jedrychowski et al. 2013; Neal et al. 2008; Oliveira et al. 2017) or immunosuppressant effects (Bolden et al. 2017; Ramesh et al. 2004) have been reported.

4.3 Biomonitoring of Polycyclic Aromatic Hydrocarbons

Given the toxicity and environmental prevalence of these compounds, the monitoring of PAHs is a relevant issue, and there is plenty of interest in control and assessment of these substances in food, environmental compartments, living beings and of course humans. Environmental monitoring of these substances is achieved by sampling and analysing samples such as air, water, food or soil (Bucchia et al. 2015; de la Gala Morales et al. 2015; García-Álvarez et al. 2014b; Hernandez et al. 2015; Kakuschke et al. 2010). Specifically, biomonitoring – the monitoring of these compounds in living beings – is usually considered the best approach as it provides a real picture of the exposure of living beings, meaning that it provides an assessment of the whole uptake through all exposure routes (Srogi 2007).

The biomonitoring of human populations may be done either by direct measurement in samples taken from study populations or extrapolating the data from the environmental exposure of other organisms (bioindicators or sentinels). This biomonitoring can be done by directly determining the individual PAHs and/or their metabolites, as well as by determining biomarkers of the effect they produce. In the case of PAHs, it is common to determine the presence of adducts of PAHs with DNA, or the detection of tetrahydroxy-PAHs that can also be measured as an indicator of tissue damage.

For reasons of practicality and ease of collection of samples, it is often considered that urinary metabolites of PAHs are better bioindicators of exposure, being considered the gold standard to determine recent exposure to a single PAH, in particular when multiple routes of exposure have to be taken into account (Jacob and Seidel 2002) or in occupational meaning (Unwin et al. 2006). The main m-PAHs that should be included in biomonitoring studies are 1-hydroxynaphthalene (1-napthol), 2-hydroxynaphthalene (2-naphthol), 1,2-dihydroxynaphthalene, 2-hydroxyfluorene (2-FLUO), 3-hydroxyfluorene (3-FLUO), 9-hydroxyfluorene

(9-FLUO), 1-hydroxyphenanthrene, 2-hydroxyphenanthrene, 3-hydroxyphenanthrene, 4-hydroxyphenanthrene, 9-hydroxyphenanthrene, 1-hydroxypyrene (1-PYR) and 3-hydroxybenzo(a)pyrene (3-OHBaP) (Wang et al. 2014a, b). 1-PYR has been linked to dietary exposures, whilst both 1-PYR and 2-naphthol are well correlated with smoking in a non-occupational population (Nethery et al. 2012; Srogi 2007). Urinary 3-OHBaP may be a suitable biomarker to assess BaP genotoxic exposure in humans (Marie-Desvergne et al. 2010; Oliveira et al. 2017). One decisive factor to take into account when determining urinary metabolites is sampling time, due to the high rate of biotransformation of these compounds (Cathey et al. 2018; Grova et al. 2017a, b). Taken together, those results suggest that it is better to use a combination of metabolites, since each metabolite gives an information about a single or few parental PAHs (Castano-Vinyals et al. 2004; Grova et al. 2017b; Hilton et al. 2017; Singh et al. 2008).

Other excretion routes, such as nails, hair, sweat or feathers, amongst others, have been also investigated regarding their content in m-PAHs, as a means of determining long-term exposure to these substances. In fact, some authors have pointed out that these matrices are more appropriate for the determination of HMW-PAHs (Grova et al. 2017b; Marie-Desvergne et al. 2010).

On the other hand, not only for assessing exposure but also the toxicological effect of PAHs, some other authors prefer to determine the amount of PAHs-DNA adducts in peripheral white blood cells, or their binding to plasmatic proteins, especially in occupational studies (Oliveira et al. 2017; Pleil et al. 2010). Other authors correlate the level of oxidative stress induced by PAHs as an indirect indicator of the carcinogenicity of these compounds (Singh et al. 2008). However, these studies of biomarkers have the disadvantage in that the analytical techniques are complex, have low sensitivity and do not allow deriving the global exposure to these compounds.

Finally, some authors consider that the direct measurement of PAHs in blood is the best way to estimate the total body burden and also the most realistic way to estimate exposure (Boada et al. 2015; Pleil et al. 2010). It has the disadvantage in that sampling is invasive, especially taking into account that WHO recommends that biomonitoring studies include mainly children, because it has been estimated that children aged 6–11 are the sector of the population most exposed to these compounds (Singh et al. 2013). In addition, and as we said before, it is possible to evaluate human exposure to PAHs indirectly, using bioindicator species. In these cases also blood is often the easiest sample to take, so comparison with human levels is simpler (Boada et al. 2015; Bucchia et al. 2015; Camacho et al. 2012b, 2014; Camacho et al. 2013b; García-Álvarez et al. 2014a, b; Luzardo et al. 2014).

In this sense, studies of the effects of environmental exposures on vegetables or animals can corroborate or support epidemiological studies in humans or in the environment. In these cases, the levels determined in these easy-to-sample species may reflect the exposure of a group of environmentally related species, rather than the individual exposure. Thus, the use of microbial bioindicators in order to evaluate contamination of some PAHs in agricultural soils (Niepceron et al. 2013) and in the gas and aqueous phases (Cho et al. 2014) has been reported. In the same way, moss,

lichens and plants have been used as passive phytomonitors instead of the active samplers and several studies have found promising results. In wildlife, some authors have proposed different species as possible sentinels of exposure. Some invertebrates have been studied. Amphipod (*Talitrus saltator*) appears to be a good bioindicator of this class of organic compounds in supralittoral zone (Ugolini et al. 2012). The possibility that molluscs are good bioindicators of the contamination of PAHs from the waters or sediments in mudflats of Malaysia (Tavakoly Sany et al. 2014) and mangrove oysters (*Crassostrea rhizophorae*) (Ramdine et al. 2012) has also been reported. Studies on oil spills such as those occurring on the northern Cantabrian sea and in Guanabara Bay, Brazil, respectively, concluded that barnacles are good indicators for oil spill evolution (Soares-Gomes et al. 2010; Vinas et al. 2009). Other species in the highest levels of the food chain have also been described as efficient indicators of recent pollution. Fuentes-Rios et al. (2005) determined that the cat shark is a good bioindicator for exposure to PAHs on the Chilean Pacific coast, showing good correlation with the concentration of pyrene in water and urinary 1-PYR. On Atlantic eastern coast and Mediterranean sea, several authors (Bucchia et al. 2015; Camacho et al. 2012a, 2013a, 2014, García-Álvarez et al. 2014a) investigated serum levels of PAHs in different populations of sea turtles (*Caretta caretta*) and bottlenose dolphins (*Tursiops truncata*) indicating that both species could be good indicators of local and recent pollution in the marine environment.

Since the iconic 'canary in the cage' began to be used to detect the presence of toxic gases in the coal mines, pets and other animals in the human immediate environment have been used as sentinels of human exposure to many other chemical classes. In this case, they were used as an early warning system, since the canary is more sensitive to carbon monoxide poisoning than humans and other domestic animals like cats, dogs, pigeons or rabbits. Livestock, including bees, cattle, horses, sheep and goats, can be good bioindicators for outdoor air, whilst pet cats and dogs can share the indoor air, water, food or even household dust. However, daily routine and diet, especially in people who are occupationally exposed, smokers or on some kind of diets, are confounding factors. The different metabolism and elimination capacity amongst the species should be also taken into account as confounding factors.

4.4 Pet Dogs as Sentinels forHuman Exposure to PAHs

Pet dogs are particularly interesting as sentinels for human exposure to PAH, given that they share the habitat with humans and they respond to toxic assaults similarly than their owners (Backer et al. 2001). As far as we know, there is only one research article that has assessed exposure to PAHs in dogs and humans to date (Ruiz-Suárez et al. 2016). In this study, the authors included blood samples from 87 pet dogs (46 males and 41 females, 0.5–13 years old) visiting the veterinary hospital of the Faculty of Veterinary Medicine of the University of Las Palmas de Gran Canaria (Canary Islands, Spain) for routine care. Only clinically normal animals (negative stool sample, negative result on a heartworm test and no overt disease) were included

in the study, after owners' consent. In parallel, human blood samples from 60 males and 40 females (19–34 years old) were collected from a blood bank during the same period that dogs' samples were drawn. For logistical reasons the researchers could not get blood from the owners of the same dogs included in the study. Even so it has been estimated that there are about six million domestic dogs in Spain and that more than 40% of Spanish homes have at least one dog, so the authors assumed that a high percentage of these blood donors share habitat with some dog.

In this research work the authors determined 21 PAHs, including the 13 c-PAHs and also 6 common m-PAHs (Table 4.1), by means of solid phase extraction and gas chromatography coupled to tandem triple-quadrupole mass spectrometry. In this research the authors detected the totality of the PAHs and m-PAHs in any of the samples, both in humans and dogs, with the only exception of benzo(a)pyrene, which was not detected in none of the dog plasma samples.

The compounds most frequently detected in both species were phenanthrene, fluorene and fluoranthene and 2-naphthol, which were present in nearly 100% of the samples. The frequencies of detection of the rest of the compounds of this chemical group were highly variable and different between the two species (Table 4.1). The mean values of $\sum$PAH21 were much lower in dogs than in humans (782.2 vs. 1623.3 ng/g lw, respectively). Regarding the c-PAHs, the authors considered only seven compounds (PAH7, benzo(a)anthracene, chrysene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenzo(a,h)anthracene and indeno(1,2,3-c,d)pyrene), and the mean values were also much lower in dogs than in humans (6.8 vs. 21.9 ng/g lw, respectively). On the opposite, according to the authors' results, it seems that dogs may have a higher capacity of biotransformation of these compounds, because in parallel to the lower levels of untransformed PAHs, dogs also had higher levels of PAHs metabolites than humans, in whom the relationship was inverse ($\sum$m-PAH = 198.1 and 131.6 ng/g lw in dogs and humans, respectively; $p < 0.0001$).

The importance of the employment of sentinel species for the assessment of human exposure to chemicals has been widely demonstrated for many chemical classes, since the sentinel species may reflect the actual human exposure of a given population, much more accurately than the comparison to other remote populations. However, it does not seem to be the case of pet dogs as sentinels of human exposure to PAHs, because the authors of the only study available in this regard found that there were many significant differences between these two species (Fig. 4.2), both in the levels of many parental compounds and in their metabolites. These results suggest that exposure of both species to this contaminant group could be different, but also may be indicating that dogs have a higher capacity to metabolise these compounds than humans. Obviously, to confirm this point, additional research is needed, but these results allowed the authors to hypothesise that the lower levels of PAHs detected in the plasma of dogs could be due to a higher rate of biotransformation and elimination thereof. Furthermore, as shown in Fig. 4.2c, neither the profiles of PAHs contamination were similar between dogs and humans, with a clear predominance of the four-ring compounds in humans and three-ring compounds in dogs. In fact, it is noteworthy that some compounds such as pyrene, which was

Table 4.1 Concentrations of individual PAHs and PAHs metabolites concentrations (ng/g lw) in dog (n = 87) and human (n = 100) serum samples from the Canary Islands, Spain

	Dog serum		Human serum		
	Mean ± SD	Freq. (%)	Mean ± SD	Freq. (%)	P^*
Benzo(a)anthracene	4.2 ± 5.1	12.6	6.0 ± 17.9	12.0	
Benzo(a)phenanthrene (chrysene)	5.4 ± 7.3	10.3	4.4 ± 17.1	13.0	
Benzo(a)pyrene	n.d.	0.0	4.4 ± 8.5	8.0	
Benzo(b)fluoranthene	4.6 ± 5.4	6.9	4.7 ± 9.2	10.0	
Benzo(k)fluoranthene	4.3 ± 6.9	11.9	17.9 ± 34.1	36.0	0.0015**
Dibenzo(a,h)anthracene	4.9 ± 6.1	2.3	4.43 ± 7.8	6.0	
Indeno(1,2,3-cd)pyrene	5.2 ± 7.4	2.3	5.2 ± 4.4	3.0	
Benzo(j)fluoranthene	4.5 ± 5.6	7.9	11.7 ± 3.7	14.0	0.0056**
Benzo(j,k)fluorene (fluoranthene)	6.6 ± 4.3	97.7	77.5 ± 26.4	99.0	<0.0001***
Dibenzo(a,e)pyrene	5.2 ± 5.6	1.2	4.3 ± 5.3	5.0	
Dibenzo(a,h)pyrene	6.3 ± 7.4	2.3	4.5 ± 5.8	6.0	
Dibenzo(a,l)pyrene	4.4 ± 6.1	1.2	6.2 ± 5.9	5.0	
5-Methylchrysene	6.3 ± 6.1	12.6	7.6 ± 17.9	14.0	
Acenaphthene	7.6 ± 25.5	13.8	8.7 ± 19. 2	16.0	
Acenaphtylene	51.2 ± 34.4	75.8	12.5 ± 6.3	6.0	<0.0001***
Anthracene	4.7 ± 26.2	4.6	6.8 ± 34.1	10.0	
Benzo(ghi)perylene	n.d.	0.0	4.5 ± 5.2	3.0	
Fluorene	76.9 ± 42.5	98.8	42.5 ± 17.2	98.0	<0.0001***
Phenanthrene	382.5 ± 0.21	100.0	313.3 ± 137.5	100.0	<0.0001***
Pyrene	7.6 ± 17.2	17.2	43.7 ± 25.9	94.0	<0.0001***
Naphtalene	34.1 ± 51.6	28.7	34.1 ± 59.6	41.0	
1-Naphthol	76.9 ± 69.8	79.3	5.2 ± 17.2	8.0	<0.0001***
2-Naphthol	95.1 ± 52.6	96.6	67.2 ± 23.5	98.0	<0.0001***
2-OH-Fluorene	5.8 ± 8.5	11.5	7.9 ± 12.3	21.0	0.1623
1-OH-Phenanthrene	17.4 ± 35.6	23.5	4.2 ± 6.6	10.0	0.0459*
7-OH-Benzo(c)fluorene	51.4 ± 8.5	36.8	4.5 ± 8.5	12.0	0.0089**
1-OH-Pyrene	4.5 ± 17.2	14.9	17.0 ± 43.5	6.0	0.4185
$\sum$PAH7[a]	6.8 ± 17.2	53.2	21.9 ± 43.5	78.0	0.01**
$\sum$PAH21[b]	782.2 ± 323.8	100.0	1623.5 ± 799.2	100.0	<0.0001***
$\sum$m-PAH[c]	198.1 ± 110.5	100.0	131.6 ± 148.5	100.0	<0.0001***

P^*: Mann-Whitney U test
[a] $\sum$PAH7, sum of carcinogenic PAHs
[b] $\sum$PAH21, sum of 21 priority PAHs
[c] $\sum$m-PAH, sum of 6 PAH metabolites

detected in almost 100% of the human samples, were barely detectable in 17% of samples from dogs, and yet, others such as acenaphthylene or 1-naphthol were much more frequently detected in dog plasma than in human plasma. Thus, in the light of the above, the authors concluded that the pet dogs do not seem to be good sentinels for human exposure to PAHs.

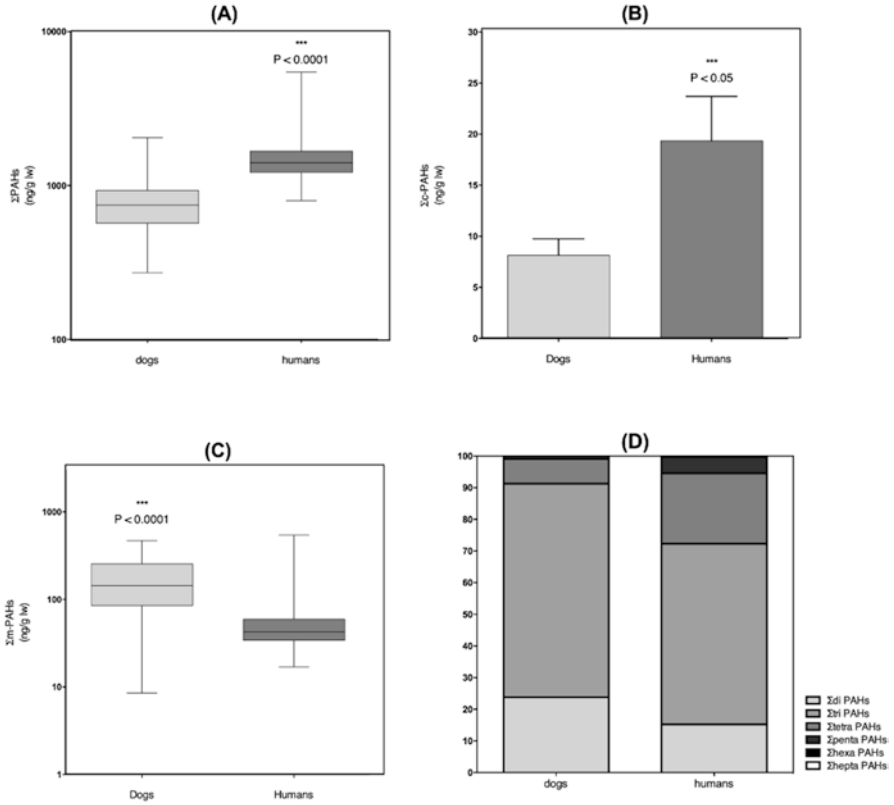

Fig. 4.2 Levels of PAHs in plasma samples. (**a**) (main body). Box plots of ∑PAH21 in dogs and humans. (**a**) (inset). Bar graph of ∑PAH7 (carcinogenic PAHs, median and interquartile range) in dogs and humans. (**b**) Box plots of ∑PAH metabolites in dog and humans. (**c**) Profile of distribution of PAHs in dogs and humans. The line inside the boxes represents the median, the bottom and top of the boxes are the first and third quartiles of the distribution, and the lines extending vertically from the boxes indicate the variability outside the upper and lower quartiles. (**d**) distribution profile of PAH in dogs and humans

4.5 Pet Cats as Sentinels for Human Exposure to PAHs

As far as we know, there is no published study that explored the role of domestic cats as sentinels of human exposure to PAHs. However, the cat that lives inside the house is usually considered a good bioindicator, even better than the dog, to assess the exposure of man to the contaminants present in the domestic environment. This is mainly due to their grooming habits, which cause cats to ingest high amounts of household dust, with all the load of contaminants associated with it. Thus, in different publications, it has been indicated that this pet is ideal for evaluating human exposure to different kinds of contaminants. (Bost et al. 2016; Chow et al. 2015; Dirtu et al. 2013; Henriquez-Hernandez et al. 2017a). In addition, other studies have

shown that dietary exposure to different contaminants (including PAHs) is different between dogs and cats (Ruiz-Suarez et al. 2015), so although, as we said earlier, dogs do not seem to be good sentinels of human exposure to PAHs, cats present differential facts that could make them suitable for this purpose, so this possibility is worth investigating.

With the purpose of completing the information in this chapter, we decided to shed light on the question whether or not cats would be good sentinels of human exposure to PAHs. For this, we collected venous blood from a total of 25 cats that were recently admitted for routine health check-ups and vaccination in the clinical hospital of the Faculty of Veterinary Medicine of the University of Las Palmas de Gran Canaria. In parallel, blood was collected from 25 volunteers from the same faculty, from amongst the staff and the students of the same, all of them owners of cats (although not from the same cats participating in the study). The serum was obtained, and the PAHs were extracted by solid phase extraction following the procedure described elsewhere (Camacho et al. 2012a). In this work, we included only the 16 priority PAHs for the USEPA, whose analysis was performed by gas chromatography coupled to tandem triple-quadrupole mass spectrometry (Luzardo et al. 2013b). All human volunteers and cat owners provided their written informed consent to participate in this study.

We found only 8 out of the 16 compounds analysed both in humans and cats. In addition, acenaphthylene was also detected in cats, but not in humans. The summary of the results of this study is shown in Table 4.2. As it can be seen, the most frequently detected compounds were acenaphthene, phenanthrene and fluorene, with frequencies of 90% or more in both species. For the rest of the substances, the detection percentage between both species was highly variable. We want to highlight the differences found between cats and humans for chrysene and fluoranthene (percentages of detection of 18.2 vs. 90% and 31.3 vs. 100%, respectively). The median of the $\sum$16PAHs was similar in both species (1.93 vs. 2.08 ng/mL or 232 vs. 257 ng/g lw, respectively). However, although the total concentrations do not show significant differences between both species, when we focus on carcinogenic compounds for EFSA, the outlook changes radically, since these compounds were practically undetectable in the group of cats, whilst they were present in the group of cat owners (Fig. 4.3). Obviously, this is only a preliminary study, and the conclusions that derive from it should be taken with caution because of the low sample size. However, based on the results obtained, it could not be considered that the cat is the ideal sentinel to assess human exposure to PAHs, although it does seem to be better than dogs in this sense.

4.6 Conclusions

Based on the scarce existing bibliography and limitations of the study, it can be concluded that pet dogs and cats are not good sentinels of human exposure to PAHs. The analyses of parental compounds and metabolites in serum and their concentra-

Table 4.2 Concentrations of polycyclic aromatic hydrocarbons (ng/mL) in the whole series of cats ($n = 22$) and humans ($n = 20$)

	Pet cats			Humans			
Congener	% detection	Median	p25th–p75th	% detection	Median	p25th–p75th	P^a
Acenaphthylene	63.6	0.02	0.00–0.04	0	–	–	0.003
Acenaphthene	100.0	0.56	0.42–0.99	100.0	0.52	0.29–0.79	ns
Anthracene	31.0	0.00	0.00–0.07	80.0	0.13	0.05–0.20	0.008
Benzo(a)anthracene	0	–	–	0	–	–	na
Benzo(a)pyrene	0	–	–	0	–	–	na
Benzo(b) fluoranthene	0	–	–	0	–	–	na
Benzo(ghi)perylene	0	–	–	0	–	–	na
Benzo(k) fluoranthene	0	–	–	0	–	–	na
Chrysene	18.2	0.00	0.00–0.00	90.0	0.03	0.01–0.04	0.003
Dibenzo(ah) anthracene	0	–	–	0	–	–	na
Fluoranthene	31.8	0.00	0.00–0.03	100.0	0.05	0.03–0.07	0.001
Fluorene	100.0	0.16	0.09–0.41	90.0	0.12	0.03–0.21	ns
Indeno(123,cd) pyrene	0	–	–	0	–	–	na
Naphthalene	18.2	0.00	0.00–0.00	30.0	0.00	0.00–0.07	ns
Phenanthrene	100.0	1.16	0.57–2.30	100.0	1.22	0.68–1.44	ns
Pyrene	13.2	0.00	0.00–0.00	60.0	0.01	0.00–0.04	0.047

Abbreviations: *p25th–p75th* percentiles 25 and 75 of the distribution, *ns* non-significant, *na* not applicable

[a]Mann-Whitney U test

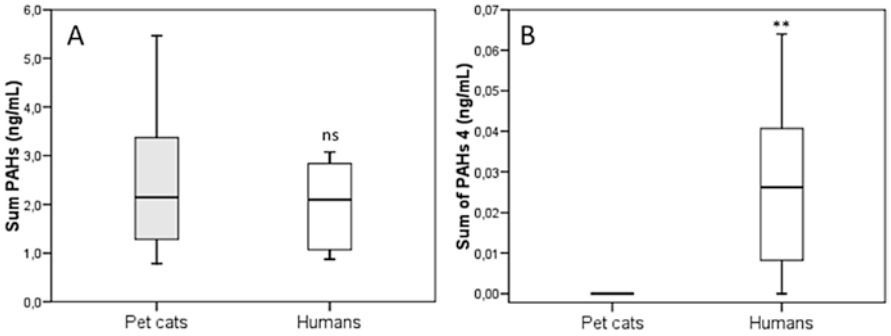

Fig. 4.3 Box plot showing the serum levels of sum of all PAHs (panel A) and sum of PAHs 4 (panel B), amongst cats ($n = 22$) and humans ($n = 20$). Sum PAHs included all the 16 congeners analysed. Sum of PAHs 4 included only benzo(a)pyrene, benz(a)anthracene, benzo(b)fluoranthene and chrysene. *P* values were calculated with Mann-Whitney U test. The lines connect the medians, the boxes cover the 25th to 75th percentiles, and the minimal and maximal values are shown by the ends of the bars. Abbreviations: ns, non-significant. **, $p = 0.003$

tions and contamination profiles are not comparable between species. These results could indicate that different sources of exposure, such as smoking, occupational setting or food intake, in humans exist. In the analysis of PAH metabolites, higher levels in dogs suggest that they metabolise them more effectively than humans. Despite sharing a home and in some cases diet with humans, pets differ greatly from humans to consider them good sentinels for PAHs exposure.

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