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PACHECO FERREIRA, ALDO

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Polychlorinated Biphenyl (PCB) congener concentrations in aquatic birds. Case study: Ilha Grande Bay, Rio de Janeiro, Brazil

ALDO PACHECO FERREIRA

Escola Nacional de Saúde Pública Sérgio Arouca, FIOCRUZ,
Rua Leopoldo Bulhões, 1480, 21041-210 Rio de Janeiro, RJ, Brasil

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ABSTRACT

Livers from 108 birds found prostrate or dead in Ilha Grande Bay between 2005 and 2010 were analyzed for 16 PCB congeners (IUPAC numbers 8, 18, 28, 31, 52, 77, 101, 118, 126, 128, 138, 149, 153, 169, 170, and 180). The species analyzed were *Egretta caerulea* (Linnaeus 1758), *Nycticorax nycticorax* (Linnaeus 1758), *Egretta thula* (Molina 1782), and *Ardea cocoi* (Linnaeus 1766). The analysis were performed using Origin software (7.5, 2004) with a significant level of $p<0.05$. Data were checked for adherence to the standard assumptions of parametric tests using the Kolmogorov-Smirnov test for normality and the Levene's test for homogeneity of variances. This has revealed differences in concentration for some congeners. Results indicate relatively low PCBs contamination in aquatic birds, but it is implied the close relationship of environmental contamination, showing potential power of widespread biological and mutagenic adverse effects in trophic levels, and therefore, signalling risk to human health.

Key words: contamination, Ilha Grande Bay, polychlorinated biphenyl, risk to environment and health, aquatic birds.

INTRODUCTION

The pollution of aquatic systems is due not only to natural causes but above all to anthropogenic activity such as discharges of domestic or industrial effluents, leaching and runoff of pesticides in agricultural lands, among others (Morley 2010). Planar halogenated aromatic hydrocarbons such as non- and mono-ortho polychlorinated biphenyls (PCBs) have become known as biologically persistent, and extremely lipophilic environmental contaminants (Giesy et al. 1994, Breivik et al. 2002). These characteristics together with high lipophilicity (log K_{ow} for PCBs from 4.9 to 8.2 (Mackay et al. 1991) result in accumulation of PCBs in food web, as well as causing an ample assortment of toxic and biological effects such as reproductive failure, immune deficiency, teratogenesis, and irregular performance in animals and humans (Shaw et al. 2006).

Pollution in the marine environment has become an issue of great concern, especially to coastal states (Schmitt-Jansen et al. 2008). The oceans cannot provide an infinite sink for anthropogenic wastes but little attention has been given to evaluating the limits of capacity of coastal areas for waste assimilation (Carpenter 1998). Consequently, instances of fisheries closures, spoiled beaches, destroyed coral reefs and wildlife

Correspondence to: Aldo Pacheco Ferreira
E-mail: aldopachecoferreira@gmail.com

habitat, toxic blooms and lost coastal ecological communities are widespread; with a corresponding determination of cost benefit (Scheren et al. 2004).

Recent concerns about connectivity of ocean health issues and the relationship to human disease highlight an important area for study. Knowledge of the ocean and the impact of human activities on it can reveal the complexity and interdependence of all aspects of the system (Costanza and Farley 2007). Improved acquaintance and predictive capabilities are required for more effective and sustained development of the marine environment to obtain associated economic benefits and to preserve marine resources.

The aquatic environment with its water quality is considered the main factor controlling the state of health and disease in both man and animal. Nowadays, the increasing use of the waste chemical and agricultural drainage systems represents the most dangerous chemical pollution (Lacerda and Molisani 2006); and, their levels may be elevated as a result of increased input into the oceans resulting from industrial activities. In some cases the concentrations of certain chemicals in marine waters have reached levels which cause damage to wildlife populations and created serious human health problems. Identifying levels in wildlife which are elevated as a result of pollution is difficult, since very few data have been reported concerning the natural levels of PCBs in any species of marine vertebrates (Storelli et al. 2007).

Aquatic birds are conspicuous animals they are a suitable choice to play a role as sentinel organisms; unexpected changes in their numbers, health or breeding success provide an alarm that may indicate an unknown pollution or food supply problem (Lauwerys and Hoet 1993). Worldwide, seabird research has undergone a major evolution in terms of data collection, interpretation of the information and application in the field of management and policy (Tasker and Reid 1997). Aquatic birds are top consumers in marine food chains which offer opportunities to detect and assess the toxicological effects of different inorganic elements on the marine ecosystem (Walker et al. 2006). Consequently, studies assessing avian population status, reproductive success, and toxicological importance PCBs exposures can be extrapolated to other wildlife and probably humans (Ferreira 2008, Mallory et al. 2010).

Fish-eating birds may be well suited for the assessment of effects of PCBs due to their wide distribution (Basler 1994). They also bioaccumulate relatively high levels of PCBs due to their higher trophic levels and due to their limited abilities to metabolize anthropogenic compounds (Becher et al. 1995). To evaluate the ecotoxicological risks, toxic equivalency quantifications (TEQs) are constantly calculated with the World Health Organization (WHO) toxic equivalency factors (TEFs). Each congener of dioxins or dioxin-like PCBs exhibits a different level of toxicity. In order to be able to sum up the toxicity of these different congeners, the concept of TEFs has been introduced to facilitate risk assessment and regulatory control. This means that the analytical results relating to all the individual congeners or compounds of toxicological relevance are summed and expressed as TCDD toxic equivalent concentration or TEQ (Wania et al. 1998, Moriarty 1999).

The goal of this work was to evaluate concentrations of PCBs in livers of *Egretta caerulea* (Linnaeus 1758), Little Blue Heron, *Nycticorax nycticorax* (Linnaeus 1758), Black-crowned Night-heron, *Egretta thula* (Molina 1782), Snowy Egret, and *Ardea cocoi* (Linnaeus 1766) Coccoi Heron, collected from Ilha Grande Bay, which is situated in the southern Atlantic Coast of Rio de Janeiro State, Brazil.

MATERIALS AND METHODS

Study Site

Ilha Grande Bay is located in the southern state of Rio de Janeiro (22° 50'- 23° 20'S, 44° 00' - 44°

45'W), and has an area of about 65.258 ha and 350 km perimeter on the waterline (Figure 1). The region has great scenic beauty a rich fauna and flora, and therefore a natural sanctuary for biodiversity (hot-spot), which lies between the two largest cities in South America - the cities of Rio de Janeiro and São Paulo. This richness and diversity of species, still little known, are due to geographic peculiarities, and hydrographic oceanographic region, coupled with factors such as diversity and connectivity of coastal systems, input of organic matter from rivers, physical variation and chemical oceanographic factors (Lailson-Brito et al. 2010).

The region of Ilha Grande Bay is home to the territories of the cities of Parati and Angra dos Reis, who had 145,000 inhabitants in 2010. In view of the beautiful landscape of the region, its main vocation naturally focuses on tourism and nautical leisure. Consequently, along the coast there is a green series of developments that, through the occupation of hillsides, riverbanks or islands and the landfill of mangrove areas, cause deforestation and polluted coastal waters. This growth as tourist hub promoted a disorderly development and causes severe damage to coastal systems. In the region there are still other large projects, such as a commercial port, a petroleum terminal, an ore terminal, two nuclear power plants and a shipyard (Ferreira 2010).

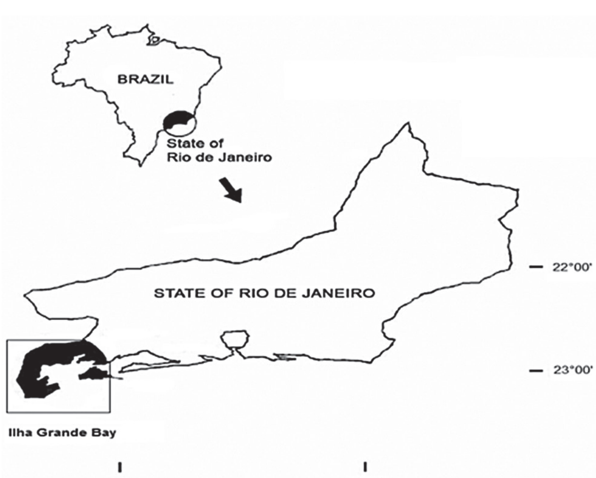

Figure 1 - Study area: Ilha Grande Bay, Rio de Janeiro, Brazil.

ANALYTICAL PROCEDURES

A total of 108 male aquatic birds (*Egretta caerulea*, n=21; *Nycticorax nycticorax*, n=25; *Egretta thula*, n=35; *Ardea cocoi*, n=27) were collected as dead stranded animals from 2005 to 2010. All fresh carcasses were necropsied following a standardised protocol (Jauniaux et al. 1998). Livers were collected, weighed and kept frozen (-18°C) prior to chemical analyses.

Chemical analysis of coplanar PCBs followed the method described in a previous report: USEPA Method 1668 (2003) and USEPA Method 8290 A (2007). Five grams of liver samples were weighed and lyophilised. Dry tissues were inserted in a steel extraction cell and placed in the Accelerated Solvent Extractor (ASE 200, Dionex). This machine using organic solvents operates under high pressure and temperature conditions (10 minutes at 125°C and 1,500 psi) and allows the extraction of the different organic compounds present from the biological matrix. After being extracted, the samples were concentrated using Kuderna-Danish, the extract evaporated down to 1 mL, and the solvent was transferred to 10 mL of n-hexane. Fat content was determined gravimetrically from an aliquot of the extract (Kiviranta et al. 1999).

12 dioxin-like PCBs (IUPAC Nos. 81, 77, 126, 169, 105, 114, 118, 123, 156, 157, 167, and 189) were spiked. Furthermore, aliquots were treated with sulphuric acid (approximately 7-10 times) in a separation funnel. Then the hexane layer with PCBs was rinsed with hexane-washed water and dried by passing through anhydrous sodium sulphate in a glass funnel. The solution was concentrated to 2 mL and sequentially subjected to silica gel, alumina, and silica gel-impregnated activated carbon column chromatography. Extracts were passed through a silica gel-packed glass column (Wakogel, silica gel 60; 2 g) and eluted with 130 mL of hexane. The hexane extract was Kuderna-Danish concentrated and passed through alumina

column (Merck-Alumina oxide, activity grade 1; 5 g) and eluted with 30 mL of 2% dichloromethane in hexane as a first fraction, which contained multi-ortho-substituted PCBs. The second fraction eluted with 30 mL of 50% dichloromethane in hexane, containing non- and mono-ortho-PCBs, was Kuderna-Danish concentrated and passed through silica gel-impregnated activated carbon column (0.5 g). The first fraction eluted with 25% dichloromethane in hexane contained mono- and di-ortho-PCBs.

Identification and quantification of dioxin-like PCBs (non- and mono-ortho-substituted congeners) was performed by use of a (i) Shimadzu GC-14B gas chromatograph with AOC-1400 auto-sampler. Columns: CBP-1 (SE-30) and CBP-5 (SE-52/54 confirmatory column). Injection: Splitless (30 s) 300°C. Temperature program of the oven: 110oC (1 min.); 15°C/min up 170°C; 7.5°C/min up to 290°C, hold for 10 minutes. Total run time: 25 minutes. Electron Capture Detector (^{63}Ni) temperature: 310°C; (ii) HPLC: Shimadzu LC-10AS; Mobile phase: acetonitrile: water 80%, isocratic run. Column: Shimadzu STR-ODS-II (C-18 reverse phase) 25 cm, L: 4mm ID. UV/VIS detector model: Shimadzu SPD-10A.

A procedural blank including extraction of blank Kimwipe and whole purification procedure was run with every batch (normally seven samples). The limit of quantification (LOQ) was set at 2 times the detected amount in the procedural blank. Reproducibility and recovery were confirmed through four replicate analyses of an abdominal adipose tissue sample with and without standard spiking. The relative standard deviations of concentrations of individual PCB-congeners were less than 5.8%, and the recoveries were more than 96%. The lipid contents were determined gravimetrically after aliquots of the sample extracts were evaporated to complete dryness.

The different congeners present in the sample were then analysed using a Gas Chromatography equipped with a capillary column of 40 µm coupled to a High Resolution Mass Spectrometer (GCHRMS). They can be quantified and their concentration calculated when compared to the added internal 13^{C} standard (Windal 2001). Results are expressed either as pg/g of lipid mass or in terms of toxicity, using WHO TEF for birds (Van den Berg et al. 2006) as pg TEQ/g, lipid weight.

Calculations

The concentrations are expressed as geometric means (mg/kg wet weight) as suggested by Newton (1988) and Newton et al. (1993). The geometric mean removes the disportionate effects of outlying values and gives a value similar to the median but with greater potential for statistical evaluation (Kruuk and Conroy 1996). To calculate the geometric mean of each PCB congener to the total PCB concentration, all no detected values were treated as the value equivalent to half of the limit of detection. The total 2,3,7,8-TCDD toxic equivalent (TEQ) value for each sample was calculated using the toxic equivalency factors (TEFs) for birds recommended by the World Health Organization (Van den Berg et al. 2006). All statistical tests were performed using Origin software (7.5, 2004) with a significant level of $p<0.05$. Data were checked for adherence to the standard assumptions of parametric tests using the Kolmogorov-Smirnov test for normality and the Levene's test for homogeneity of variances.

RESULTS

The Table I presents the descriptive statistics of PCBs concentrations (pg/g lipid weight) in *Egretta caerulea*, *Nycticorax nycticorax*, *Egretta thula* and *Ardea cocoi*.

Concentrations of PCB-congeners with fat percentages are presented in Table II. Fat-based log-transformed concentrations were used to determine whether there were significant differences between group geometric means (Tukey test). Null hypothesis (equality of means) was rejected at the

TABLE I
Descriptive statistics of PCBs concentrations (pg/g lipid weight) in aquatic birds.

Aquatic bird	PCBs (pg/g lipid weight)											
	Mean	sd(yEr±)	se(yEr±)	P25	P75	P95	Min	Max	Range	Median	Var	Coef Var
Egretta caerulea	85.08333	73.11567	21.10668	17	132	228	12	228	216	68.5	5,345.90152	0.85934
Nycticorax nycticorax	63.75	53.30892	15.38896	19	98	166	12	166	154	46	2,841.84091	0.83622
Egretta thula	129.66667	97.75975	28.22081	43	211	298	37	298	261	88.5	9,556.9697	0.75393
Ardea cocoi	54.5	46.51197	13.42685	16	65	147	9	147	138	39.5	2,163.36364	0.85343

Mean. Sd = Standard deviation of data. Se = Standard error of the mean. P25 = 25 percentile. P75 = 75 percentile. P95 = 95 percentile. Min= Minimum. Max= Maximun. Range. Median. Var =Variance. Coef Var = Coefficient of variation.

95% significance level ($p<0.05$). The medians of concentrations of PCBs (pg/g lipid weight) and toxic equivalents of PCBs (pg TEQ/g lipid weight) presented expressed no significant species-related differences in PCB. There were no statistically significant differences between mean PCB-congeners concentrations between the species.

Data in figure 2 shows the distribution of PCB congeners. PCB 105 congener accounted for 22.33% of ΣPCB in *Egretta caerulea*, 22.05 % of ΣPCB in *Egretta thula*, and 22.47% of ΣPCB in *Ardea coccoi*. PCB 114 congener accounted for 21.70% of ΣPCB in *Nycticorax nycticorax*.

TEQs of PCBs were calculated using TEFs for birds proposed byWHO (Van der Berg et al. 2006), and compositions are shown in figure 3.

DISCUSSION

Increased human activities such as industrialization, coupled with over-population and increased ambient temperature amongst other factors, have become major environmental issues in recent years. As a result of such actions, additional studies which include the environment and their indicators are important because they can show potential impacts that are being reflected, and extending to public health. Thus, the study of ecotoxicology is a very broad field of science where issues such as uptake and effects in organisms, as well as distribution and residence time of the pollutants in the trophic level are studied in many different ways.

The fundamental question to answer is whether the trophic level is harmfully disturbed when polluted by toxicants. To answer this important question, quantitative understanding of the pollutants behaviour within ecosystems is essential, and therefore researchers develop methods to manage this. The presence of anthropogenic pollutants, such as PCB-congeners, throughout all compartments of the marine environment has been of international concern for a number of decades (Kumar et al. 2001). While a great number of datasets documenting absolute concentrations of persistent organic pollutants in a variety of marine biota are available, the bioaccumulative nature, toxicity, biomagnification, and the fate of these compounds in the marine ecosystem is still poorly understood (Pereira 2004). Data on contaminant levels in Brazilian aquatic birds are limited, and no information exists regarding levels of new or emerging contaminants.

The PCB congeners detected at the highest concentrations were 105 and 114. The congener profile determined in these birds probably reflects differences in both exposures to PCB congeners and in ability to metabolize them.

TABLE II
Medians (range) of concentrations as pg/g lipid weight of PCBs and toxic equivalents of PCBs (pg TEQ/g lipid weight) in *Egretta caerulea*, *Nycticorax nycticorax*, *Egretta thula*, and *Ardea cocoi.*

Elements	*Egretta caerulea*		*Nycticorax nycticorax*		*Egretta thula*		*Ardea cocoi*	
	Concentration	WHO TEF (birds)	Concentration	WHO TEF (birds)	Concentration	WHO TEF (birds)	Concentration	WHO TEF (birds)
Non-ortho PCBs								
3,3',4,4'-TCB (77)	132 (67 - 445)	6.6	98 (44 - 222)	4.9	211 (52 - 354)	10.55	65 (30 - 291)	3.25
3,4,4',5-TCB (81)	54 (23 - 466)	5.4	37 (14- 152)	3.78	71 (26 - 455)	7.1	48 (22 - 366)	4.8
3,3',4,4',5-PeCB (126)	79 (44 - 192)	7.9	55 (31 - 168)	5.5	93 (32 - 228)	9.3	33 (16 - 102)	3.3
3,3',4,4',5,5'-HxCB (169)	77 (35 - 221)	0.077	26 (11 - 63)	0.026	84 (31 - 167)	0.084	35 (19 - 123)	0.035
Mono-ortho PCBs								
2,3,3',4,4'-PeCB (105)	228 (62 - 355)	0.0228	149 (34 - 296)	0.0149	298 (81 - 404)	0.0298	147 (26 - 210)	0.0147
2,3,4,4',5-PeCB (114)	198 (52 - 311)	0.0198	166 (64 - 259)	0.0166	278 (44 - 386)	0.0278	133 (35 - 222)	0.0133
2,3',4,4',5-PeCB (118)	132 (45 - 266)	0.00132	102 (32 - 218)	0.00102	235 (40 - 339)	0.00235	92 (31 - 266)	0.00092
2',3,4,4',5-PeCB (123)	60 (28 - 148)	0.0006	62 (19 - 127)	0.00062	112 (56 - 264)	0.00112	44 (20 - 136)	0.00044
2,3,3',4,4',5-HxCB (156)	14 (9 -52)	0.0014	13 (7 - 87)	0.0013	39 (15 -122)	0.0039	11 (7 -56)	0.0011
2,3,3',4,4',5'-HxCB (157)	12 (8 - 43)	0.0012	19 (7 - 77)	0.0019	37 (11 - 89)	0.0037	9 (4 - 32)	0.0009
2,3',4,4',5,5'-HxCB (167)	18 (11 - 59)	0.00018	26 (10 - 72)	0.00026	55 (29 - 111)	0.00055	16 (7 - 68)	0.00016
2,3,3',4,4',5,5'-HeCB (189)	17 (9 - 38)	0.00017	12 (6 - 34)	0.00012	43 (16 - 101)	0.00043	21 (14 - 62)	0.00021
	Σ=1,021	Σ=20.02	Σ=765	Σ=14.24	Σ=1,351	Σ=27.10	Σ=654	Σ=11.41

Lailson-Brito et al. (2010) studying organochlorine accumulation in Guiana dolphin (*Sotalia guianensis*), at the same study site found concentrations levels from 0.765 to 99.175 pg/g lipid for ΣPCB. Some oceanic islands, such as São Pedro e São Paulo Archipelago, is a group of small rocky islands that lies in the central equatorial Atlantic Ocean, lying 627 km from the archipelago of Fernando de Noronha, 986 km from the nearest point on the mainland and 1,010 km from Natal, in Rio Grande do Norte, Brazil; may be considered remote areas and preserved due to its distance from the mainland. However, these areas are not exempt from the influence of anthropogenic agents from coastal regions, such as persistent organic pollutants (POPs). The predominant compounds were PCBs that presented 98.15 ng/g to *Sula leucogaster* (Brown Booby) (Dias 2010). The low levels of contaminants suggest a relative degree of isolation and preservation, but the occurrence and distribution profiles of PCBs supports the hypothesis that the main source of contamination in remote areas is long range atmospheric transport, and demonstrates the ubiquity of those pollutants

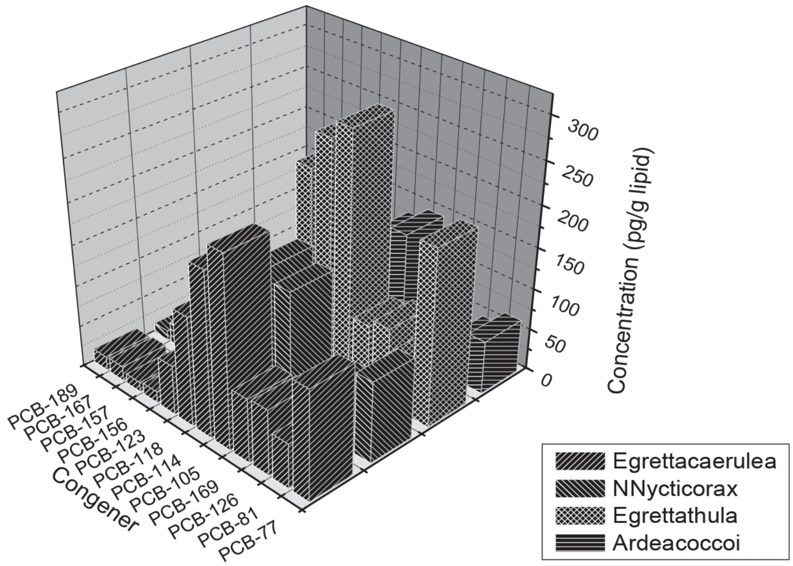

Figure 2 - Concentrations of PCB Congeners in avian species studied (pg/g lipid).

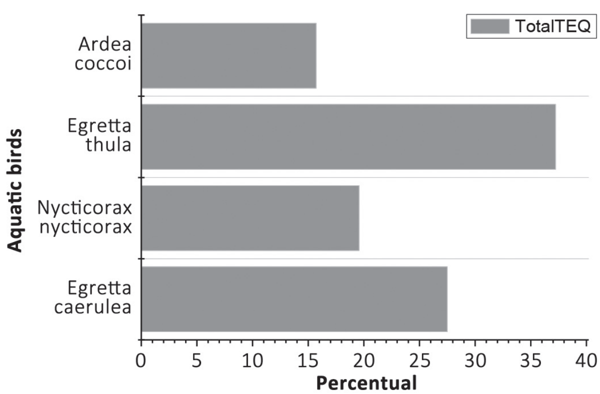

Figure 3 - Contributions of dioxin-like PCBs to Total TEQ.

in the marine environment. Cunha et al. (2012) studying dichlorodiphenyltrichloroethane (DDT) and polychlorinated biphenyls (PCBs), in the eggs of the Brown Booby (*Sula leucogaster*) collected from breeding colonies located on three archipelagos (Saint Peter and Saint Paul, Abrolhos and Cagarras Islands) in the Atlantic Ocean, found concentrations levels of 0.05 $\mu g.g^{-1}$ of $\sum$PCBs and 0.01 $\mu g.g^{-1}$ of $\sum$DDT, 0.19 $\mu g.g^{-1}$ of $\sum$PCBs and 0.03 $\mu g.g^{-1}$ of $\sum$DDT, and 8.4 $\mu g.g^{-1}$ of $\sum$PCBs and 1.8 $\mu g.g^{-1}$ of $\sum$DDT, respectively. The total PCB level is close to the threshold values considered to be harmful to birds. The findings indicate that the brown booby colony closest to the Rio de Janeiro coast has recently been exposed to DDT. Despite the high pollution levels found on the Cagarras Islands, no alterations in the eggshell weight or the thickness of the analyzed eggs were detected.

The toxicity of individual congeners was assessed using the toxic equivalency approach (Van den Berg et al. 2006). There is a paucity of TEF values reported for seabird livers (Guruge et al. 2000, Choi et al. 2001, Kumar et al. 2002). However, care must be taken when comparing total TEQ values as other studies may include TEQ values from additional contaminants including chlorinated dibenzo-p-dioxins, furans, and chloronaphthalenes, giving TEQ values potentially greater than those reported here. However TEQ values based solely on PCBs are still of some use, as PCBs accounted for 60% to 85% of the total TEQ concentrations reported in bird livers analyzed for all contaminant groups (Kannan et al. 2002).

In previous studies, the monitoring of POPs in aquatic birds has been limited by the availability in organs (Peakall et al. 1990, Shaffer et al. 2006). This approach can easily be combined with ecological investigations of aquatic birds, and so this could dramatically increase the availability of seabird samples, including repeated sampling on identical birds (Holmström and Berger 2008). Recently, electronic tracking tags have revolutionized our understanding of the large-scale movements and habitat use of mobile marine animals (Shaffer et al. 2006).

Reported adverse effects of POPs in wildlife include population declines, increases in cancers, reduced reproductive function, disrupted development of immune and nervous systems, and also elicit toxic responses which could result in the disruption of the endocrine system (Alcock et al. 1998, Rittler and Castilla 2002). The assessment of environmental variables and biological effects in aquatic birds will provide critical insights into the level and extent in public health effects associated with marine areas and resources. Additively, these chemicals may produce a significant effect. In part, the lack of evidence reflects the fact that relatively little research has been done. Also, the direct contaminant loads and exposure will assist regional, and consequently, national decision makers in efforts to ensure the sustained protection to marine ecosystems.

CONCLUSION

The presence of tissue levels of PCBs can be associated with biological and physiological effects in marine organisms, in especially aquatic birds (Montevecchi 1993, Holmström and Berger 2008). The animals sampled in the current study had PCB congeners yet in low values, but in a more expressive than other studies here mentioned. As production of PCBs and their use systematically declined greatly since the late 1960s and early 1970s, it might be expected that current concentrations in comparable biological compartments (e.g., seabird livers) could be lower; but still presents occasionally high. However, this study shows that such a simplistic view is not valid. Indeed, there were large intra- and interspecies differences in total PCB concentrations found in these birds. This suggests that the intra- and interspecies differences were at least partly due to characteristics of the locations in which the birds feed and live. The findings for the individual congeners were similar to the findings for total PCBs, although conclusions for some of the congeners are less clear due to differences in the relative proportion of congeners found in the birds. Factors that might affect the concentration of chemicals in seabird livers residues of any contaminant measured in a bird will vary according to many factors, including exposure to the chemical, metabolic capabilities, and the nutritional state of the bird.

The current study is still the first to report aquatic birds' concentrations of POPs at this study site, and the first for any free-ranging birds from the Ilha Grande Bay. And continued monitoring of POPs is essential in assessing the health and viability of these animals. The present study confirms the ubiquity of POPs in species studied, belonging the marine environment of Ilha Grande Bay, Rio de janeiro, Brazil. Biomagnifications may be the cause of the levels in the species collected and analysed. Further assessments are recommended on organisms at higher trophic levels for ecotoxicological impacts. The ubiquity of these pollutants in Ilha Grande Bay's marine environment supports the need for a greater awareness of bioaccumulation processes, particularly for organisms cultivated (shellfish) or fished locally and destined for human consumption.

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RESUMO

Fígados de 108 pássaros encontrados prostrados ou mortos na Baía da Ilha Grande entre 2005 e 2010 foram analisados para 16 congêneres de PCB (IUPAC números 8, 18, 28, 31, 52, 77, 101, 118, 126, 128, 138, 149, 153, 169, 170 e 180). As espécies analisadas foram *Egretta caerulea* (Linnaeus 1758), *Nycticorax nycticorax* (Linnaeus 1758), *Egretta thula* (Molina 1782) e *Ardea cocoi* (Linnaeus 1766). As análises foram realizadas utilizando o software Origin (7.5, 2004) com um nível significativo de p<0,05. Os dados foram checados para a adesão aos pressupostos padrão dos testes paramétricos, utilizando o teste Kolmogorov-Smirnov para normalidade e teste de Levene para homogeneidade das variâncias. Isto revelou diferenças na concentração de alguns congêneres. Os resultados indicam relativamente baixa contaminação PCBs em aves marinhas, mas está implícita a estreita relação de contaminação ambiental, evidenciando potencial poder de generalização adversa de efeitos biológicos e mutagênicos em níveis tróficos e, por conseguinte, sinalizando risco para a saúde humana.

Palavras-chave: contaminação, Baía da Ilha Grande, bifenilas policloradas, risco ao ambiente e saúde, aves marinhas.

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