Dioxin and Dioxin-like PCB Exposure and Their Risk Estimation: Survival Rate of Common Cormorant in Japan

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Abstract

The liver and eggs of common cormorants reside in and around Tokyo Bay have been analyzed for polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and dioxin-like PCBs and their exposure levels, distribution in tissues/organs and contribution to toxic equivalency (TEQ) are discussed. The concentrations of contaminants in the liver of common cormorant are significantly higher than those in ten other species of birds that belong to different ecological systems and food chains. 2,3,7,8-Substituted PCDD/Fs comprised 97% and 90%, respectively, of the total PCDD/Fs in the liver and eggs. Particularly, 1,2,3,7,8-PeCDD and 2,3,4,7,8-PeCDF have shown the greatest contribution to TEQ because of the high load of these congeners in avian liver. The high biomagnification factor (BMF) and the maximum TEF value of 1 for these congeners are also taken into further consideration. The half-lives of 2,3,7,8-TCDD in common cormorant are conjectured by using an analysis value of fish and liver of cormorant for 43 days. The half-lives ranges from three days for 2,3,7,8-TCDF to 122 days for 1,2,3,4,7,8,9-HpCDF. In addition, the half-lives of mono-*ortho* PCBs were found to range from several hundred days to several years. Using tissue distribution data and half-lives, risk transition of the exposure of PCDD/Fs and dioxin-like PCBs and egg mortality of common cormorant in the past were predicted. Consequently, using only sediment data, I further estimated biota-sediment accumulation factors (BSAFs) and their

transport to the cormorants that feed on fish in the same environment. Therefore, the risk transition was estimated to be two times higher in the 1970s because of elevated dioxin-like PCBs during that period compared to the present time. Eventually, risk transition seems to have declined after the 1970s due to emergence of regulatory policies.

1. Introduction

Dioxin and its related compounds include the so-called polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and dioxin-like PCBs. These are widely studied chemicals even in Japan because of their adverse effects on humans. Although dioxin and its related compounds are present in various environmental mediums, such as atmosphere, soil and water, they are again bioaccumulated in the biota through the food chain because of their highly lipophilric characteristics (Fletcher *et al.*, 1993). Thus, high concentrations are found in marine mammals and birds belonging to the top of the food chain (Norstrom *et al.*, 1990; Braune *et al.*, 1989). Due to their high accumulation in animals, the toxic implication in these animals is rather high. There are several studies that have related death and deformity of birds to elevated concentrations of PCBs and DDTs in and around the Great Lakes of USA (Postupalsky *et al.*, 1978; Gilbertson *et al.*,1988; Fox *et al.*,1991).

The Environment Agency Government Japan has been surveying levels of PCDD/F and dioxin-like PCB contamination since 1998 the relative consequences of such contamination in wildlife. However, no systematic evaluation has been carried out by them thus far. In other words, the study of exposure, distribution, action, and half-lives in the body is necessary to further reduce the risk of avians to fatal chemicals. In this study, residual levels, i.e. the characteristic accumulation features in the livers of cormorants, were compared to those of 10 species of wild birds from various habitats. Further more, the distribution of contaminants in tissues/organs was determined. Under the assumption of a static state in a one-compartment model, half-lives were estimated using analyzed data and distribution in tissues. Body concentration and changes of environmental levels were related to half-lives. Based on the analytical values derived from eggs, a formula was evaluated to determine levels of concentrations in eggs from those in fish. To estimate risk transition to the eggs of common cormorants, previous levels of contamination in fish were calculated using BSAF and reported data of sediment core collected from Tokyo Bay (Yao *et al.*, 2000). Therefore, under the assumption that common cormorants ate fish, levels of contamination in eggs were estimated each year. Egg mortality was estimated and risk assessment was performed on an individual basis to determine the endpoint because they greatly affect population size. Risk was expressed as an estimation of the lethal dose (LD) for embryos of double-crested cormorants in the Great Lakes, USA (Giesy *et al.*, 1994). Considering those facts, in this study, I report certain schemes of evaluation using common cormorant, the colonial fish-eating bird that reside in and around Tokyo Bay of Japan.

2. Materials and Methods

2.1 *Biological samples*

Stranded common cormorants were collected from 1997-1999 at Shinobazu Pond (*n*=10), and one captive dead cormorant was obtained from Sayama Lake. Four dead cormorants were provided by Gyotoku Wild Bird Observatory and only one bird was shot to survey its diet in Sagami River by dissecting and separating the liver that was stored at –30^{o}C until analysis. In the case of egg samples (*n*=9), they were collected by our team in 1998 from 6th Odaiba Island, Tokyo Bay , with permission from the Environment Agency of Japan. Three species of fish [viz., sea bass (*n*=1, *Lateolablax japonicus*), gizzard shad (*n*=3, *Konosirus punctatus*) and conger eel (*n*=1, *Conger myriaster*)] were collected from Tokyo Bay in 1998. The analyzed samples and some ecological notes are shown in Table 1. In addition, common cormorant (*n*=1) for tissues/organs analysis was also collected from Sagami River for survey of their diet and 10 species of other birds were not included in Table 1.

Table 1. Analyzed biological samples in this study.

Organ	Age	Sex	*n*	Body weight (g)	Fat (%)	Moisture content (%)
Common cormorant						
Liver	Chick	F	*1*	940	2.7	73
	Juvenil	F	*3*	1300±520	3.6±1.4	74±4.2
		M	*6*	1500±480	3.5±1.3	74±1.7
	Adult	F	*2*	1500±720	7.0±4.1	73±2.4
		M	*4*	2100±350	3.1±1.6	74±1.3
Egg			*9*	45±6.3	6.0±0.38	81±1.3
Fish						
Sea bass (*Lateolablax japonicus*)			*1*	270	4.3	73
*Gizzard shad (*Konosirus punctatus*)			*3*	**393	**8.2	**73
Conger eel (*Conger myriaster*)			*1*	130	8.6	71

*Denotes 3 number samples were pooled, **Standard deviation data are not available.

2.2 *Chemical analysis*

Clean-up conditions for chemical analyses were given elsewhere Nakamura *et al.* (1996) and Sakurai *et al.* (1997). Breifly, three different species of fish were homogenized, freeze-dried and used for analysis. Homogenized cormorant whole liver tissue and egg samples were freeze-dried, and the moisture content was measured. Then, the samples were extracted using a Soxhlet apparatus a dichloromethane as solvent. After fat content determination, the extract was treated with sulfuric acid and purified by fractionation. Silica gel and alumina were used for fractionation. Silica gel removed most pesticides such as DDT (if any) and its metabolites. Alumina removed most of the existing mono-*ortho* and other PCBs. Further more, in a charcoal-impregnated silica-gel column, PCDD/Fs and dioxin-like PCBs were separated into two fractions. The first fraction, eluted with 25% dichloromethane in hexane, consisted of mono-*ortho*-substituted PCBs. The second fraction, eluted with toluene, comprised of PCDD/Fs homologues and non-*ortho*-substituted PCBs. Quantification and identification techniques were adopted for non-*ortho*- and mono-*ortho*-substituted PCBs, followed by HRGC-HRMS [AutoSpec Ultima (micromass)]. The separation of PCDD/Fs was achieved using a HP 6890 (Hewlett-Packard) machine equipped with DB-5 and DB-17 columns (J&W Scientific) in the splitless and solvent cut mode. Gas chromatographic separation of non-*ortho* and mono-*ortho*-substituted PCBs was performed in a DB-5 capillary column.

3. Result and Discussion

3.1. *PCDD/F and dioxin-like PCB levels in liver and egg of common cormorant*

The levels of dioxin and relative compounds in the analyzed liver of common cormorant (*n*=16) were significantly higher than those in the other ten bird species (Fig.1). PCDD/F and dioxin-like PCB levels were, on average, 33,000pg/g fat and 33,000ng/g fat, respectively (Table 2). Particularly, 2,3,7,8-substituted PCDD/F congeners comprised 90 and 97%, respectively, of total PCDD/Fs in the liver and eggs of common cormorants. Notably, 1,2,3,7,8-PeCDD, 1,2,3,6,7,8-HxCDD and 2,3,4,7,8-PeCDF congeners were found to accumulate higher levels than other PCDD/Fs. These congeners were also found in high levels in fish species collected from same location. Fish seems to the major source of dioxin accumulation in birds. The levels of PCDD/Fs and dioxin-like PCBs in the three fish samples analyzed were 210-710pg/g fat and 54-470ng/g fat, respectively. Among the fish species analyzed, congener-specific accumulation was also observed to vary. Due to the fact that dioxin accumulation in birds is due mainly to fish diet, the relative biomagnification factor (BMF) of each isomer was calculated by using the mean levels of what in fish and birds. The results showed that BMF of the liver was 10 times greater than that of eggs (Figs. 2 and 3). Some congeners showed high BMF of the following order: 1,2,3,6,7,8-HxCDD (300) > 1,2,3,4,7,8-HxCDD (200) > 1,2,3,7,8-PeCDD (140)

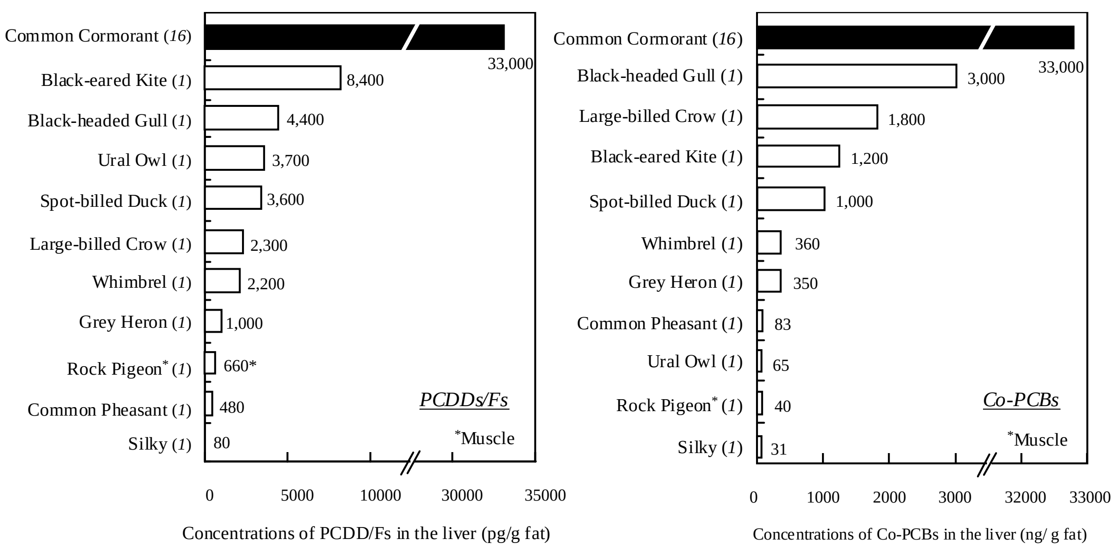

Fig. 1. Comparison of concentrations of PCDD/Fs and dioxin-like PCBs in the liver among bird samples.

among PCDDs; 2,3,4,7,8-PeCDF (340) > 1,2,3,6,7,8-HxCDF (290) among PCDFs; and CB189 (3100) > CB169 (420) among dioxin-like PCBs. On the other hand, the most toxic congeners, namely, 2,3,7,8-TCDF and CB77, showed small BMF values of 1.9 and 6.9, respectively. The accumulation pattern and the elimination capacity of various organs of cormorants also differed.

Table 2. Concentrations (pg/g, fat) of PCDD/Fs in the liver of common cormorants and fish.

Isomers	WHO TEF (birds)	Cormorant: Chick (*n* =1)	Juvenile (*n* =9) Mean (min.-max.)	Adult (*n* =6) Mean (min.-max.)	Mean (*n* =16)	Fish: Sea bass	Gizzard shad*	Conger eel
PCDDs								
1,3,6,8-TCDD		3.0	2.5 (n.d.-4.7)	1.3 (n.d.-4.2)	2.1	43	23	23
1,3,7.9-TCDD		0.96	4.5 (n.d.-32)	1.0 (n.d.-3.7)	3.0	1.9	3.8	1.4
2,3,7,8-TCDD	*1*	740	400 (43-1500)	700 (98-1800)	530	23	2.4	2.8
Total TCDD		*750*	*420 (52-1600)*	*720 (98-1800)*	*560*	*69*	*30*	*27*
1,2,3,7,8-PeCDD	*1*	1400	3400 (340-19000)	4500 (490-11000)	3700	68	7.1	2.9
Total PeCDD		*1400*	*3400 (350-19000)*	*4500 (490-11000)*	*3700*	*72*	*11*	*4.9*
1,2,3,4,7,8-HxCDD	*0.05*	250	1100 (91-7400)	1300 (114-2400)	1100	12	2.5	2.0
1,2,3,6,7,8-HxCDD	*0.01*	920	4900 (340-33000)	5600 (1000-9500)	4900	34	6.9	8.6
1,2,3,7,8,9-HxCDD	*0.1*	30	71 (6.1-420)	57 (11-83)	63	3.3	0.63	0.58
Total HxCDD		*1200*	*6100 (440-41000)*	*6900 (1100-12000)*	*6100*	*59*	*10*	*10*
1,2,3,4,6,7,8-HpCDD	*0.001*	750	2600 (200-17000)	1900 (340-3900)	2300	40	14	6.6
Total HpCDD		*770*	*2600 (200-17000)*	*2000 (360-3900)*	*2300*	*55*	*16*	*25*
OCDD	-	1900	3400 (170-18000)	4500 (320-12000)	3700	136	97	33
PCDFs								
1,3,6,8-TCDF		0.39	7.4 (n.d.-29)	2.2 (n.d.-7.8)	5.0	0.75	1.4	0.36
2,3,7,8-TCDF	*1*	16	30 (2.9-160)	66 (5.5-130)	40	38	11	18
Total TCDF		*26*	*270 (18-1100)*	*300 (37-820)*	*270*	*43*	*18*	*18*
1,2,3,7,8-PeCDF	*0.1*	5.0	7.2 (2.8-15)	13 (n.d.-36)	8.7	9.9	2.5	4.4
2,3,4,7,8-PeCDF	*1*	2300	8900 (1000-47000)	15000 (2300-25000)	10000	69	16	11
Total PeCDF		*2300*	*9000 (1000-48000)*	*15000 (2300-25000)*	*11000*	*71*	*13*	*11*
1,2,3,4,7,8-HxCDF	*0.1*	480	1500 (140-8900)	1900 (190-3300)	1500	18	3.1	3.4
1,2,3,6,7,8-HxCDF	*0.1*	380	1300 (110-7500)	1500 (200-2900)	1300	10	2.2	1.9
2,3,4,6,7,8-HxCDF	*0.1*	400	1600 (170-7800)	2600 (500-5600)	1800	19	5.3	5.3
1,2,3,7,8,9-HxCDF	*0.1*	22	35 (4.9-110)	46 (14-75)	36	5.9	0.61	1.3
Total HxCDF		*1300*	*4700 (450-24000)*	*6100 (920-12000)*	*5000*	*104*	*26*	*31*
1,2,3,4,6,7,8-HpCDF	*0.01*	160	360 (58-1300)	320 (45-550)	310	28	5.1	5.1
1,2,3,4,7,8,9-HpCDF	*0.01*	70	160 (17-780)	220 (40-290)	170	4.5	0.78	1.3
Total HpCDF		*290*	*580 (100-2200)*	*800 (160-1200)*	*640*	*55*	*16*	*25*
OCDF	*0.0001*	98	110 (24-400)	300 (33-580)	170	45	10	27
Total PCDDs**		6000	16000 (1800-95700)	19000 (2400-32000)	16000	390	160	100
Total PCDFs**		4000	15000 (1600-75000)	22000 (3400-39000)	17000	320	83	110
Total PCDDs/Fs**		10000	31000 (4000-170000)	41000 (5800-70000)	33000	710	240	210
WHO(Birds)-TEQ		4600	13000 (1500-71000)	21000 (3000-39000)	16000	210	38	37
(2,3,7,8-PCDDs/Fs) /(total PCDDs/Fs) (%)		99	97 (93-99)	98 (94-99)	97	79	75	58

*Pooled sample, **Values rounded

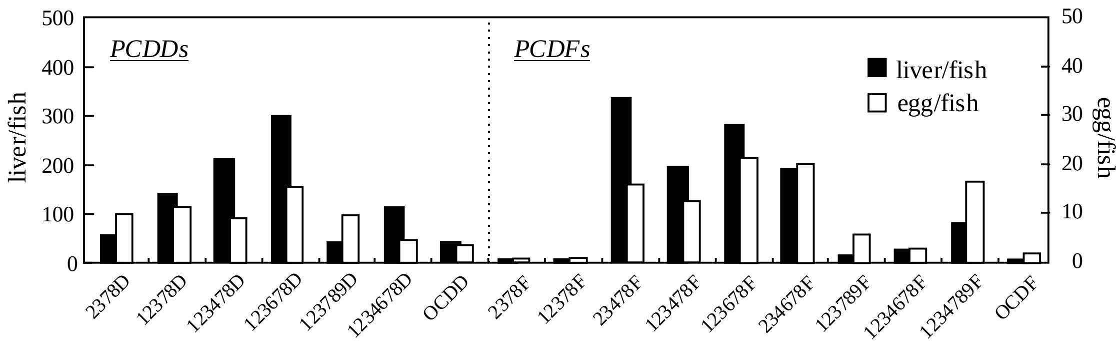

Fig. 2. Biomagnification factor of PCDD/Fs in the liver and egg of common cormorants collected from Tokyo Bay, Japan.

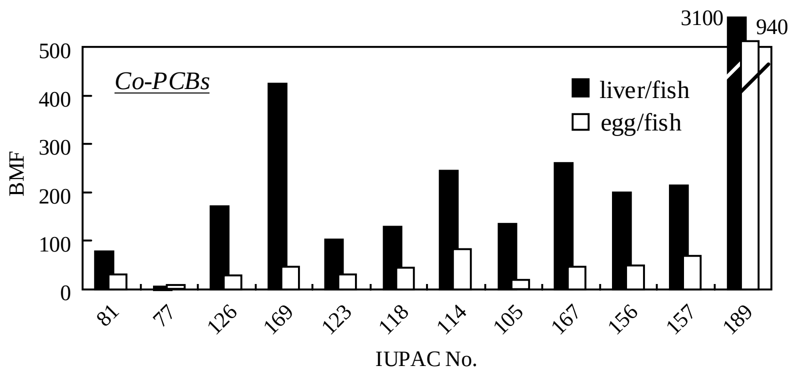

Fig. 3. Biomagnification factor of dioxin-like PCBs in the liver and egg of common cormorants collected from Tokyo Bay, Japan.

3.2. *Distribution level in tissues and organs*

Tissues and organs such as muscle, blood, liver and kidney collected from one individual cormorant that also had egg before laying were analyzed and compared to understand specific tissue/organ accumulation of PCDD/Fs and dioxin-like PCBs. The overall result shows that all 2,3,7,8-substituted PCDD/Fs accumulated markedly in the liver (Fig. 4). Blood had considerably high levels of OCDDs compared to other tissues and egg. It was also found that 1,2,3,7,8-PeCDD, 1,2,3,6,7,8-HxCDD and 2,3,4,7,8-PeCDF accumulated specifically in the liver compared to the other congeners. Similarly, non-*ortho* PCBs such as CB126 and CB169 also showed liver-specific accumulation. On the other hand, mono-*ortho* PCBs were found to have the same levels in tissues and organs on a lipid-normalized basis.

3.3.*Toxic equivalency and their contribution*

Toxic equivalency (TEQ) in liver, egg and fish was estimated based on WHO birds TEF (Van den Berg *et al.*,1998). Consistent with the accumulation pattern, liver showed the highest TEQ

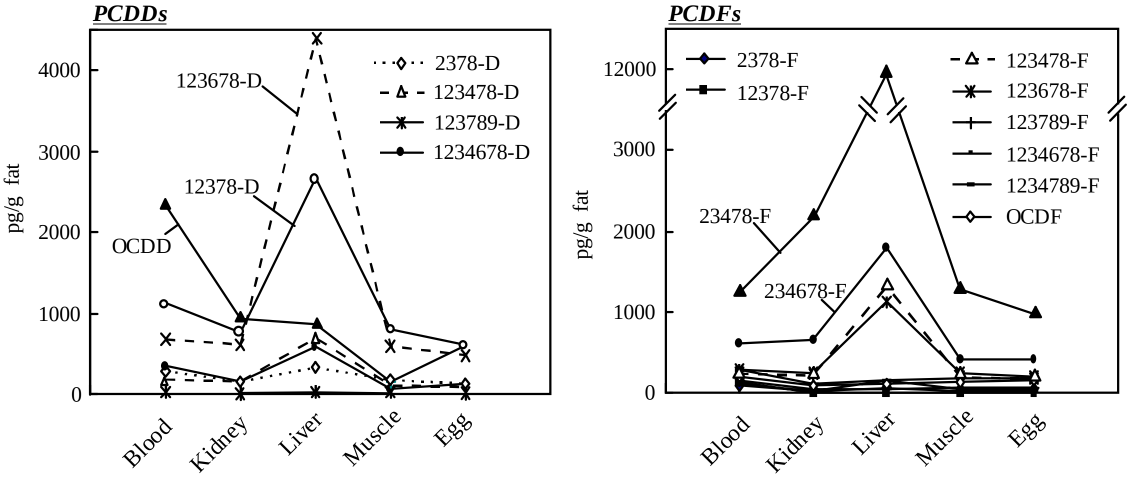

Fig. 4. Concentration and distribution of PCDDs/Fs in different tissues of common cormorant.

(27,000) followed by egg (3,600) and fish (330) on pgTEQ/g fat basis. Especially, 1,2,3,7,8-PeCDD and 2,3,4,7,8-PeCDF contributed most to the toxicity in cormorants. Altogether, PCDD/F contribution to TEQ in livers and egg is 50 and 30%, respectively. Among PCBs, the non-*ortho* dioxin-like PCB isomer, CB126, contributed predominantly to the toxicity in cormorants. In contrast, the non-*ortho* dioxin-like PCB isomer, CB77, was the most prevalent contributor in fish (Fig. 5). The relationship between egg mortality and the concentration of TCDD-EQ, determined using H4IIE assay from a survey of the double-crested cormorants from the Great Lakes, was reported. The reported threshold concentration that affects hatching success of the cormorant is expressed as the lethal dose (LD) (Giesy et al., 1994). Therefore, the regression coefficient was estimated from a plot of LD values and the concentrations of TCDD-EQ. Then, the regression coefficient is established from these relationships, the LD value of egg of common cormorant from Tokyo Bay was estimated using the regression coefficient and average TEQ concentration in egg from Tokyo Bay (226pgTEQ/g wet wt.).

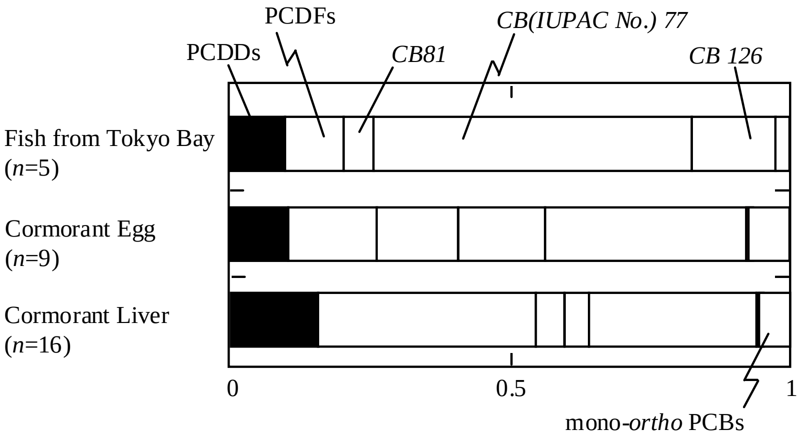

Fig. 5. Toxic Equivalent Contribution (Birds-TEF) of PCDDs, PCDFs and dioxin-like PCBs in cormorant and fish from Tokyo Bay, Japan.

The overall LD estimated for egg of common cormorant from Tokyo is 27, and this value indicates that 27% of egg in the colony of common cormorant would die due to toxicity. Despite the observed differences between the results from Great Lakes and those of the present study, it is plausible that different contribution by TEQ and distribution of dioxin in the fish as diet. Nevertheless, it is worth discussing LD calculated by similar study on ecologically similar bird species.

3.4. *Estimation of half-lives of 2,3,7,8-substituted PCDDs, PCDFs and dioxin-like PCBs*

Knowing the levels of contaminant in certain organs, would be a very useful tool to predict exposure levels in wildlife. For example, if there is a relation in the levels of contamination in the muscle, liver, and egg, it will enable the estimation of contamination in levels in the non-analyzed tissues. Further more, this value for any specific tissue on a fat weight basis may not be useful because the fat weight basis for all tissues always gives same results. Therefore, the accumulation of each congener in the liver was considered and it was found that the accumulated levels arose from the diet. In this study, the burden of each congener of dioxin and its related compounds was estimated using the ratio of tissue and organ weight to whole body weight of four female birds. The total of concentration of in six body parts such as muscle, blood, liver, kidney, egg and others (the internal organs and skin) was treated as the whole body concentration, and bone and feather were ignored. Again, fat tissue (subcutaneous) was also ignored because it was seldom found during the dissection process. Considering all these parameters, the ratio of weight and fat and $Ratio_{liver}$ are shown in Table 3, where $Ratio_{liver}$ indicates burden in liver against that in the whole body (six body parts) calculated from tissue weight and concentration on a wet weight basis. In contrast, the concentrations in other body parts were treated as concentration in others and muscle were same values. Considering those facts, the sum total of the six body parts was treated as 1.

Table 3. Parameters used to calculate half-lives in common cormorant.

	Symbol	
[1] *Weight ratio in whole body*		
Muscle	W_M	*0.41*
Liver	W_L	*0.029*
Kidney	W_K	*0.0090*
[2]Egg	W_E	*0.021*
[3]Blood	W_B	*0.074*
Others	W_O	*0.23*
[4] *Fat ratio in the tissues/organs*		
Muscle	F_M	*0.036*
Liver	F_L	*0.037*
Kidney	F_K	*0.028*
Egg	F_E	*0.060*
Blood	F_B	*0.0034*
Ratio $_{liver}$		
2,3,7,8-TCDD		*0.075*
1,2,3,7,8-PeCDD		*0.13*
1,2,3,4,7,8-HxCDD		*0.21*
1,2,3,6,7,8-HxCDD		*0.24*
1,2,3,7,8,9-HxCDD		*0.088*
1,2,3,4,6,7,8-HpCDD		*0.24*
OCDD		*0.15*
2,3,7,8-TCDF		*0.048*
1,2,3,7,8-PeCDF		*0.026*
2,3,4,7,8-PeCDF		*0.28*
1,2,3,4,7,8-HxCDF		*0.22*
1,2,3,6,7,8-HxCDF		*0.17*
2,3,4,6,7,8-HxCDF		*0.15*
1,2,3,7,8,9-HxCDF		*0.031*
1,2,3,4,6,7,8-HpCDF		*0.14*
1,2,3,4,7,8,9-HpCDF		*0.036*
OCDF		*0.033*
Parameters		
Administrative dose	*AD*	*BW* × *0.26*
[5]Absorption ratio	*f*	*0.94*
Body weight	*BW*	*1885*
$Log_e 2$		*0.693*
Fish fat%		*0.07*

[1]Assumption from real value of female (*n* =4)
[2]Egg data measured in this study.
[3]Blood data is value from other study, unpublished.
[4]Fat ratio is value of analyzed sample in this study.
[5]Absorption ratio is using fat ratio of fecal matter/ fish.

Under the assumption of static state in a one-compartment model, the relation between concentration and half-lives is shown in the formula provided below;

$$\frac{C_{Fish} \times AD \times f}{Fat_{liver}} \times \frac{T_{1/2}}{Log_e 2} \times Ratio_{liver} = C_{liver}$$

where C is concentration, AD is administered dose, f is absorption ratio, $Log_e 2$ is 0.693 and $Ratio_{liver}$ is the burden in liver compared to those in whole body.

The relative half-lives ($T_{1/2}$) could be calculated from the above formula using the arithmetic mean concentration of in the liver of adult common cormorant (*n*=9) and whole fish (*n*=5) (Table 4). On the whole, the half-lives of 2,3,7,8-TCDD in the common cormorant was calculated to be 43 days, although it is influenced largely by the concentrations of dioxin in fish. The standard deviation (S.D.) value was calculated from the reported data of 2,3,7,8-TCDD by Tokyo Metropolitan, further range of 2,3,7,8-TCDD half-lives was estimated using these S.D. The half-lives estimated ranged from 31 to 69 days. The obtained half-lives of 2,3,7,8-TCDD in common cormorant was more or less similar to that in mouse (11-24days) and rat (17-31days) (Streit *et,.al.*, 1992）. Collectively, the half-lives of 1,2,3,7,8-PeCDD and 2,3,4,7,8-PeCDF were respectively 1.67 and 1.37 times longer than that of TCDD.

Table 4. Calculated half-lives (T1/2) of PCDD/Fs in common cormorant.

Isomers	Half-lives
2,3,7,8-TCDD	43
1,2,3,7,8-PeCDD	59
1,2,3,4,7,8-HxCDD	51
1,2,3,6,7,8-HxCDD	61
1,2,3,7,8,9-HxCDD	19
1,2,3,4,6,7,8-HpCDD	17
OCDD	14
2,3,7,8-TCDF	2.6
1,2,3,7,8-PeCDF	3.9
2,3,4,7,8-PeCDF	72
1,2,3,4,7,8-HxCDF	47
1,2,3,6,7,8-HxCDF	79
2,3,4,6,7,8-HxCDF	73
1,2,3,7,8,9-HxCDF	24
1,2,3,4,6,7,8-HpCDF	7.8
1,2,3,4,7,8,9-HpCDF	120
OCDF	14

3.5. *Validation of obtained half-lives*

When five years old was assumed by using the half-lives, the calculated results of the concentrations showed very near value with the actual measurement in the liver of five years old cormorant which is identified by the bird by banding survey. However, the results of a similar calculation for the egg for exceeded the measured value, due to several reasons such as egg distribution ratio. Using 0.2 as the revised value, the egg level was calculated to fit to the value in the egg from Tokyo in this study.

3.6. *Trend of dioxin exposure risk and its effect on mortality of common cormorant in Tokyo*

Changes in the levels of dioxin in the sediment core from Tokyo Bay from 1935 to 1989 were investigated (Yao *et al.*,2000). The exposure of common cormorant egg to contamination in the past was predicted, and the exposure risk was estimated. BSAF was calculated from the sediment core in 1989 and fish collected in 1998. Since the concentrations of contaminants in fish were treated on a wet weight basis, the calculation for risk estimation was done under the assumption that organic matter content (OM) in sediment core and fat content in fish had been standardized. Based on this assumption, a large variation in the concentration of contaminants in fish as investigated by Tokyo Met. was not observed from 1990 to 1999 year. We believe that these assumptions for calculation are adequate due to the reason that the analysis values are the same as the levels reported by Tokyo Met. Concentrations of contaminants in fish in the past were predicted using BSAF from sediment core of the period investigated. When TEQ in fish was calculated from bird TEF, TEQ in fish seemed to increase from 1960, and to reach a maximum in 1967-72. Regarding contribution to TEQ, CB77 accounts for 50-60% of total amount of contaminants in fish, while CB126 account for about 10%. Under such circumstance that the common cormorant feeds on those fish, TEQ in egg was calculated by using half-lives in the same period. Thus, LD for egg for each year was estimated from the derived TEQ and the regression coefficient of embryo mortality, and the relative risk of mortality as LD was shown (Fig.6). The results also suggested that risk increased drastically from the early 1960s and declined from 1967- 72. CB126 and CB77 contributed most to the TEQ in egg. In addition, concentration levels of these congeners in the environment of current level further influence the toxic effects such as mortality of egg. The risk posed by PCDD/Fs became high from the 1970s and has not shown any declining trend to date. The population of common cormorants in Tokyo decreased to about 170 in 1970, and ornithologists assumed that this was due to environmental pollution (Ishida *et al.*, 2000). The estimation of the levels of dioxin and its related compounds in common cormorant over the past few years further supported their assumption. However, the influence of other environmental factors can not be verified completely, and these risk estimation for common cormorant does not exceed the stage of one hypothesis at present.

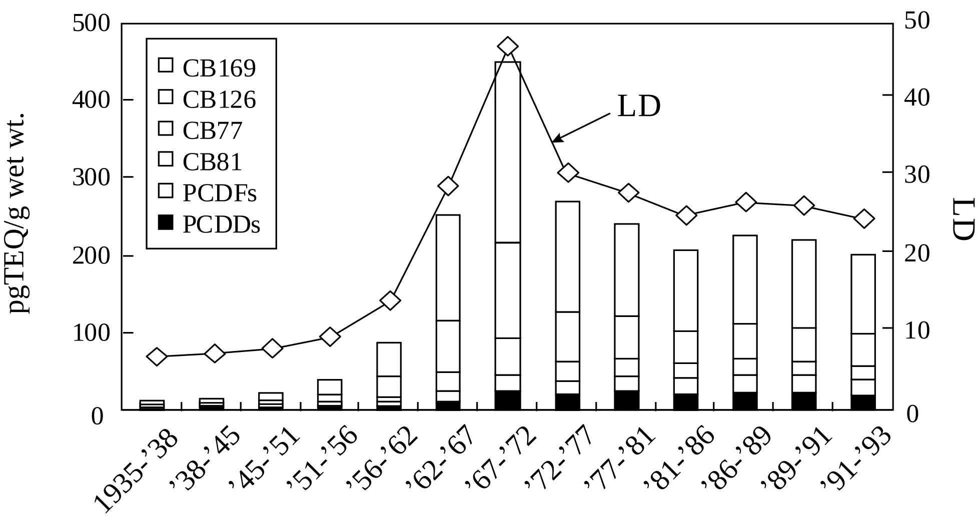

Fig. 6. Trend of TEQ in the egg and its effect on mortality of common cormorant from Tokyo.

4. Conclusions

The influence of dioxins on the mortality of common cormorant was evaluated quantitatively from the point of view of the analyzed residual levels of dioxin in wild birds in Japan. First, the liver and the egg of common cormorants in and around Tokyo Bay were analyzed, and the accumulation tendency of dioxin and its related compounds was discussed. The concentration of dioxin in the liver of common cormorant was much higher than that in ten other species birds. 2,3,7,8-Chlorine substituted PCDD/Fs accounted for more than 97% of the total PCDD/Fs in the liver of common cormorant. The same tendency was also noted in the egg of the same species. When the results were compared with those in fish, the isomer profiles were similar, suggesting that fish is the main source of the contaminants observed in cormorants from Tokyo Bay. The difference in isomer profile of each PCDD/F and dioxin like-PCB was shown in terms of BMF in the liver and congener-specific accumulation in common cormorant. Second, about five body parts were analyzed, and high accumulation of PCDD/Fs in the liver was recognized as a result of comparing the within-body distribution. It was shown that determining the within-body accumulation of each isomer from the result of BMF and distribution was necessary to estimate the exposure risk to dioxin in common cormorants. Third, as a result of the TEQ calculated by using birds TEF, liver and the egg, regarding the toxicity contribution in the liver, PCDD/Fs were found to account for 50%, but in eggs, dioxin-like PCBs accounted for 70% of total TEQ. Likewise, it is clear that there is a difference in toxicity contribution between the liver and the egg. Forth, the half-lives of dioxin and its related compounds in common cormorant were estimated using the ratio of the distribution and the concentrations of dioxin in the liver of adult common cormorant and fish from Tokyo Bay under the assumption of a static state. Finally, based on sediment core data from Tokyo Bay, changes in the concentration of contaminants in the diet were estimated by using BSAF. Concentration of contaminants in the egg of common cormorant was estimated using half-lives of dioxin and the revised formula under the assumption that these fish were eaten by them. Toxic effect of contaminants during the past was shown as LD, which was established for double-crested cormorant in Great Lakes. In the same way, the existence of risk changes in egg of common cormorants was calculated. The results revealed increased risk in the 1960s, which declined in the 1970s. Dioxin-like PCBs, such as CB126 and CB77, contributed most to the toxicity in eggs. The influence of the change in the environment concentration levels on common cormorant egg should be considered with caution.

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