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Reproduction of European eel jeopardised by high levels of dioxins and dioxin-like PCBs?

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ABSTRACT

Dioxins, furans and dioxin-like polychlorinated biphenyls (PCBs) were analysed in muscle tissue from yellow phased European eel (*Anguilla anguilla*) from 38 sites in Belgium. Dioxin concentrations in eel vary considerably between sampling locations, indicating that yellow eel is a good indicator of local pollution levels. Measured levels of dioxin-like PCBs are much higher than those of the dioxins and furans. In the majority of the sites, eel has levels considered to be detrimental for their reproduction. Field levels of dioxin and dioxin-like PCBs are therefore suggested as an additional causal factor contributing to the decline of the European eel. 42% of the sampling sites show especially dioxin-like PCB levels exceeding the European consumption level (with a factor 3 on average). Human consumption of eel, especially in these highly contaminated sites, seems unjustified.

1. Introduction

Polychlorinated dibenzo-*p*-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs) are highly toxic persistent environmental contaminants, which influence the health and reproductive success of many species, including freshwater fish and eel (Palstra et al., 2006). PCDDs, PCDFs, and dioxin-like PCBs (DL-PCBs) have comparable chemical structures and toxicological characteristics. PCB mixtures have been used for a variety of applications largely based on their chemical stability and physical properties (Bhavsar et al., 2007). De Boer et al. (1994, 2010) demonstrated that elimination half-lives of PCBs are in the order of years. They found half-lives of some organochlorine pesticides and tri-pentachlorobiphenyls to be ca. 300–1500 days, whereas for hexa-octachlorinated biphenyls no apparent elimination was found. Their stability is also responsible for their continued presence in the environment, even decades after extensive regulatory actions and an effective ban from the EU in 1985. Although, long time series of PCB monitoring in eel illustrate a decreasing trend, these series clearly show that PCBs are still ubiquitous in our aquatic ecosystems (De Boer et al., 1994; Maes et al., 2008). The situation is similar for dioxins, furans and DL-PCBs. The half-lives of 2,3,7,8-substituted tetrachlorinated dibenzo-*p*-dioxin (2,3,7,8-TetraCDD) in rodents, for example, is usually 2–4 weeks but in humans it has been estimated to be 7–11 years, although, with wide individual variation. The half-lives of other PCDD/Fs may vary between six months and 20 years (Srogi, 2008). DL-PCBs are a group of 12 PCBs that share a common toxic mechanism, with the most toxic dioxin compound 2,3,7,8-TetraCDD and, generally are among the most toxic PCB congeners as they incur toxic effects at relatively lower concentrations than those of non-DL-PCBs (NDL-PCBs) (Giesy and Kannan, 1998; McFarland and Clarke, 1989). Environmental hazards associated with the DL-PCBs are generally assessed separately from the NDL-PCBs.

Recent studies pointed out that PCBs and other dioxin-like chemicals play an important role in the actual decline of the European eel (Belpaire et al., 2009, 2011; Geeraerts and Belpaire, 2010; Palstra et al., 2006; Van Ginneken et al., 2009), although causative relationships between PCB exposure and effects on population level are difficult to demonstrate, considering the complex life cycle of this panmictic catadromous species. Palstra et al. (2006) artificially stimulated female and male silver eel to maturation and reproduction and studied the effects of dioxin-like compounds in muscle and gonad tissues on embryonic development. They reported large differences in embryonic development of eel eggs. The observed correlation between embryo survival time and 2,3,7,8-TetraCDD toxic equivalent (TEQ) levels (EC, 2006) in the gonads implied TEQ-induced teratogenic effects. The disrupting effects occurred at levels below 4 ng TEQ kg^{-1} gonad, which is below the EU eel consumption standard.

Eel is particularly prone to bioaccumulating hazardous substances (such as polychlorinated biphenyls (PCBs; Belpaire et al., 2011; Maes

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et al., 2008), organochlorine pesticides (OCPs; Maes et al., 2008), heavy metals (Maes et al., 2008) and selected brominated flame retardants (BFRs; Roosens et al., 2010)). The European eel is well suited for use in chemical monitoring as widespread, semelparous, benthic, carnivorous and lipid rich species, highly sedentary in its yellow eel phase (Belpaire and Goemans, 2007; De Boer and Hagel, 1994). Moreover, the eel is a popular human food in many European countries, and therefore represents a potential health hazard to consumers.

In order to gain insight in the current status of pollution by dioxins and related compounds in Flanders – the northern part of Belgium – a baseline spatial analysis was conducted in yellow eel from different Flemish locations. Spatial variation in the level of dioxin pollution might indicate areas of concern for these substances. We anticipate that the levels in eel might be of toxicological relevance, potentially causing effects on the eel and putting additional pressure on the state of the already imperilled eel stock. Results also may indicate if the current dioxin concentrations in Belgian wild eel exceed the international food safety standards.

2. Materials and methods

Between 2000 and 2007, yellow eel were caught at 38 locations spread over Flanders (Table 1 and Fig. 1). Sampling sites included canals ($n = 16$), polder water courses ($n = 3$), rivers ($n = 13$) and closed water bodies ($n = 6$). To avoid effects of possible variation in body burden of individual eel from a particular site due to variation in size, sex or age, we aimed to analyse pooled samples from 10 individuals per site. But, this objective could not be met at all sites due to low abundances. On each locality 4–10 yellow eel were captured and live transported to the laboratory. Again low abundances did not allow sampling for a standardised eel length, so eel were of variable length (range 33.9–64.1 cm) and weight range (59.7–566.4 g). At the lab, fish were measured, weighed, skinned and samples of muscle tissue (10 g fresh weight each) were removed, labelled and stored at −20 °C. From each sampling location, tissues from individual eel were pooled prior to homogenisation and analysis (5.0 g).

All samples were processed in our ISO17025 BELAC accredited laboratory successfully participating in relevant inter-calibration

Table 1
Overview of the pool samples with the description of the sampling sites and year, number of eel in the sample (N) and mean parameters of eel (mean length (LM), mean weight (WM) and the muscle lipid content of the pooled sample (Fat%)). For each sample the absolute dioxin concentrations (ΣPCDD/Fs), the sum of dioxin-like PCB concentration (ΣDL-PCBs), the sum of both (ΣPCDD/F + DL-PCBs) expressed as pg g^{-1} fresh weight, and the body burden (ng g^{-1} fresh weight) are given. Minimum, maximum, mean, standard deviation (SD) and median are also given. NA = not available.

Site code	Water body	Water course type	Sampling year	N	LM (cm)	WM (g)	Fat%	ΣPCDD/Fs	ΣDL-PCBs	ΣDL-PCBs + PCDD/Fs	Body burden ΣPCDD/F + DL PCBs
KND	Canal Nieuwpoort-Duinkerke	Canal	2005	10	39·4	133·0	11·6	1·7	9879·5	9881·2	1314·2
KNN	Creek of Nieuwendamme	Polder	2002	5	35.3	77.8	10.	2.4	5384.6	5387.0	419.1
YZ	Yser	River	2000	5	43.2	201.7	15.6	5.2	29,960.3	29,965.5	6044.0
IK	Iepercanal	Canal	2002	10	37.4	93.7	10.6	2.0	11,426.3	11,428.4	1070.8
HV	Handzamevaart	Polder	2002	10	33.9	59.7	0.7	110.5	18,089.9	18,200.4	1086.6
LE	Leie	River	2001	6	56.5	466.1	3.2	10.4	45,917.4	45,927.7	21,406.9
BBV	Blankenbergse Vaart	Polder	2003	10	37.2	99.7	9.2	2.1	2418.9	2421.0	241.4
BK	Boudewijncanal	Canal	2006	4	64.1	566.4	7.8	2.4	25,027.0	25,029.3	14,176.6
DAV	Damse vaart	Canal	2006	9	39.9	109.0	17.6	14.0	53,743.2	53,757.2	5859.5
DGH	Gavers	Closed waterbody	2000	5	60.5	388.3	16.4	7.6	40,387.9	40,395.5	15,685.6
LEO	Leopoldcanal	Canal	2003	9	36.3	78.0	7.9	2.2	5101.2	5103.4	398.1
KGO	Canal Ghent-Oostend	Canal	2004	6	40.6	126.0	6.9	8.0	41,696.1	41,704.1	5254.7
BGG	Oude Leie Bourgoyen	Closed waterbody	2000	10	39.0	96.0	14.3	5.1	38,051.7	38,056.8	3653.5
DE1	Dender	River	2006	7	53.5	253.5	4.8	1.7	8600.4	8602.1	2180.6
DE2	Dender	River	2006	9	49.6	202.8	3.6	1.5	20,098.5	20,100.0	4076.3
DE3	Dender	River	2002	8	42.1	141.3	10.6	4.3	48,447.0	48,451.3	6846.2
ODU	Oude Durme	Closed waterbody	2002	5	38.6	99.6	8.9	3.6	17,643.3	17,646.9	1757.6
KZ	Klein Zuunbekken	Closed waterbody	2002	5	39.6	107.0	15.0	7.1	87,641.2	87,648.3	9378.4
WBV1	Willebroekse vaart	Canal	2002	10	36.3	84.8	12.3	5.2	107,734.9	107,740.1	9136.4
WBV2	Willebroekse vaart	Canal	2002	5	39.7	103.1	10.1	3.7	113,685.9	113,689.6	11,721.4
KDS1	Canal Dessel-Schoten	Canal	2003	10	48.1	181.9	10.6	6.5	124,752.3	124,758.8	22,693.6
KDS2	Canal Dessel-Schoten	Canal	2003	10	40.4	126.0	2.3	1.8	23,399.3	23,401.1	2948.5
DIJ	Dijle	River	2006	3	52.1	296.6	27.4	6.1	19,682.0	196,88.2	5839.5
KN	Kleine Nete	River	2003	10	40.4	110.1	11.7	3.6	13,157.8	1,3161.4	1449.1
RM	Rotselaar lake	Closed waterbody	2007	10	37.9	94.9	3.7	1.1	3036.1	3037.1	288.2
KBH1	Canal Bocholt-Herentals	Canal	2002	5	41.3	115.1	10.2	7.9	264,843.7	264,851.5	30,484.4
DEM	Demer	River	2003	3	50.6	308.0	4.8	1.6	6745.2	6746.8	2078.0
COM	Congovaart	Canal	2001	5	43.2	162.3	10.6	10.3	409,151.8	409,162.0	66,407.0
GN	Grote Nete	River	2000	7	39.4	101.2	13.4	4.6	9097.4	9101.9	921.1
KB1	Canal of Beverlo	Canal	2005	5	41.2	110.1	3.6	1.6	5428.1	5429.7	597.8
KB2	Canal of Beverlo	Canal	2005	10	49.9	244.3	7.9	5.6	33,628.3	33,633.9	8216.8
DO	Dommel	River	2006	9	43.1	154.2	7.1	2.1	4940.6	4942.8	762.2
AK	Albertcanal	Canal	2000	8	45.2	157.8	12.7	10.0	110,113.9	110,123.9	17,377.6
KBH2	Canal Bocholt-Herentals	Canal	2002	10	40.4	110.5	3.1	2.1	35,390.6	35,392.7	3910.9
IB	Itterbeek	River	2005	9	38.3	109.3	5.5	1.8	3031.2	3033.0	331.5
AB	Abeek	River	2004	6	42.1	116.5	4.4	1.5	13,689.1	13,690.6	1595.0
OMS	Old Meuse	Closed waterbody	2002	10	40.9	109.5	1.7	31.7	NA	NA	NA
MA	Grensmaas	River	2002	10	44.6	159.4	6.5	2.7	45,061.3	45,064.0	7183.2
Minimum			2000	3	33.9	59.7	0.7	1.1	2,418.9	2,421.0	241.4
Maximum			2007	10	64.1	566.4	27.4	110.5	409,151.8	409,162.0	66,407.0
Mean					43.2	164.6	9.1	8.0	48,859.0	48,867.0	7,759.4
SD					6.9	110.5	5.3	17.9	78,453.9	78,454.0	12,145.6
Median					40.8	115.8	9.1	3.7	21,748.9	21,750.6	3,782.2