



National Institute for Public Health  
and the Environment  
*Ministry of Health, Welfare and Sport*

**Risk-based standards for PCBs in soil**  
Proposals for environmental risk limits  
and maximum values

RIVM report 2014-0119

E.M.J. Verbruggen | E. Brand



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and the Environment  
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RIVM Report 2014-0119

## Colophon

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## Abstract

### **Risk-based standards for PCBs in soil**

Proposals for environmental risk limits and maximum values

The RIVM proposes new science-based quality standards for PCB contaminants (polychlorinated biphenyls) in the soil. This is necessary to assess the possibilities for the reuse of excavated soils. The quality of the soil determines the type of land use for which the soil can be reused. If the quality is high enough, this can be the land use 'residential with garden', if not it can be reused for the land use 'industrial'.

For the reusing of soil it is determined which contaminants are in the soil by use of a standard list. In 2008, PCBs were added to this list of substances. The quality of the soil is determined based on the proposed quality standards (so-called maximum values). To date, however, there have been no scientifically based maximum values for PCBs.

The proposed science-based quality standards for PCBs in soil are based on the exposure of mustelids to PCBs. It appears that mustelids are designated as the family of animals that are most sensitive to PCBs.

Up to now, the risk limits of PCBs in the soil were based only on the direct toxicity of these substances for humans and ecosystems. However, the accumulation of PCBs in the food chain is a greater risk than direct toxicity. If accumulation in the food chain is not included in risk assessment, the risk is underestimated.

Keywords:

Polychlorinated biphenyls (PCBs), maximum values, risk assessment, ecosystems, secondary poisoning, human



## Publiekssamenvatting

### **Risico onderbouwde normen voor PCB's in de bodem**

Voorstellen voor risicogrenswaarden en maximale waarden

Het RIVM doet een voorstel voor nieuwe wetenschappelijk onderbouwde normen voor PCB-verontreinigingen (polychloorbifenylen) in de bodem. Dit is nodig om te beoordelen voor welke doeleinden afgegraven grond kan worden hergebruikt. De kwaliteit van de grond bepaalt óf dat kan en voor welke bestemming het mag worden hergebruikt: bij voldoende kwaliteit voor 'wonen', anders voor 'industrie'.

Voor de beoordeling van hergebruik van grond wordt aan de hand van een standaardlijst gemeten welke vervuilende stoffen in de bodem zitten. In 2008 zijn PCB's aan deze stoffenlijst toegevoegd. De kwaliteit van de grond wordt beoordeeld op basis van de grenswaarden voor deze stoffen (de zogenoemde Maximale Waarden). Tot op heden bestonden er geen wetenschappelijk onderbouwde maximale waarden voor PCB's.

De voorgestelde wetenschappelijk onderbouwde normen voor PCB-verontreinigingen zijn gebaseerd op de blootstelling van marterachtigen aan PCB's. Het blijkt namelijk dat marterachtigen als meest gevoelige soort kunnen worden aangeduid.

Tot nu toe waren de risicogrenswaarden van PCB's in de bodem alleen gebaseerd op de directe giftigheid van deze stoffen voor mens of ecosystemen. Het is echter gebleken dat ophoping van PCB's in de voedselketen een groter risico vormt dan directe giftigheid. Als ophoping in de voedselketen niet wordt meegenomen, wordt het risico onderschat.

Kernwoorden:

Polychloorbifenylen (PCB's), maximale waarden, risicobeoordeling, ecosystemen, doorvergiftiging humaan



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## Summary

In 2008, polychlorinated biphenyls (PCBs) were added to the list of relevant compounds when preparing soil quality maps. Soil quality maps are used to classify batches of soil for reuse elsewhere. The reuse of soil is allowed if the quality of the soil complies with the Maximum Value belonging to the soil function and if the receiving soil has a comparable quality. For the land use category 'residential' and the land use category 'industrial', scientifically risk-based values were lacking.

In 2010 the soil quality decree was adjusted in such a way that batches of soil or sediments which were contaminated with PCBs up to a maximum of two times the generic background value (equal to 0.040 mg/kg d.w.) could be reused as clean soils or sediments. Still, the need for a more solid foundation remained.

### Ecotoxicological and human risk assessment

Both ecotoxicological and human risk are part of the method used to derive quality standards. Secondary poisoning in the food chain turned out to be the most critical parameter for exposure to PCBs. In previous evaluations, secondary poisoning by PCBs was not included in the quality standards and the risk limits were based on direct toxicity only. As a result, risk values for PCBs determined without the inclusion of secondary poisoning are an underestimation.

In the derivation of risk limits, usually a Species Sensitivity Distribution (SSD) is prepared to derive an HC5 (Hazardous Concentration of 5%), HC20 (maximum value for 'residential') and HC50 (maximum value for 'Industrial') if sufficient data are available. In this study, for pragmatic reasons (the data for PCBs are too numerous), it was decided to use the no observed effect concentration (NOEC) for the mink as a starting point for deriving the reference values. With the chronic reproduction toxicity studies for the mink, a very sensitive species and endpoint have been covered. Because no SSD was constructed as is usually done, an assessment factor (AF = 10) was applied to the lowest NOEC value to derive a value for the predicted no effect concentration (PNEC). The resulting PNEC is **0.046 µg Σ7 PCBs /kg<sub>Dutch standard soil</sub>**. Furthermore, the NOEC for mink with an assessment factor of 10 (= PNEC) acts as maximal permissible concentration (MPC<sub>eco</sub>), the NOEC without an assessment factor acts as a SRC<sub>eco</sub> (**2.8 x 10<sup>-3</sup> mg/kg d.w.**) value (also equal to maximum value for 'industrial') and the geometric average of the MPC<sub>eco</sub> and SRC<sub>eco</sub> is taken to represent the HC20 (**3.6 x 10<sup>-4</sup> mg/kg d.w.**) (which is also equal to maximum value for 'residential').

Based on the available information, it can be concluded that a PNEC based on Mink (a very sensitive species) gives sufficient protection for the rest of the ecosystem and to humans. The derived risk limit in soil based on the Maximal permissible risk (MPR) for human exposure derived by Baars et al. in 2001 is higher than the ecotoxicological risk limits in this report.

For humans, the reference values (basis for maximal values) for PCBs (sum7) were based on two types of mixtures, Arochlor 1254 and the distribution of the individual PCB congeners in the background concentration of Dutch soils. The derived MPR by Baars et al. in 2001 has not been revised since then.

The derived maximal values for ecotoxicology and humans are presented in Table S1.1.

*Table S1.1 Background values and the reference values for 'residential with garden', 'residential with vegetable garden', 'green with nature values' and 'industry' for PCBs (sum7) based on mixtures of Arochlor1254 and a mixture based on distribution as background concentrations in the Netherlands (AW2000). The values given in bold represent the Proposed Maximal Values for 'residential' and 'industrial'.*

Mixture	Back-ground value (mg/kg. d.w.)	Reference value				Reference value based on
		Residential with garden (mg/kg. d.w.)	Residential with vegetable garden (mg/kg b.w./day)	Green with nature values (mg/kg b.w./day)	Other greens, infrastructure, buildings and industry (mg/kg d.w.)	
PCBs (sum 7) based on Arochlor 1254	0.02*	<b>3.6 x 10<sup>-4</sup></b>	3.6 x 10 <sup>-4</sup>	3.6 x 10 <sup>-4</sup>	<b>2.8 x 10<sup>-3</sup></b>	Ecotoxicology
PCBs (sum 7) based on Arochlor 1254	0.02*	0.39	0.08	15.0	15.0	Human
PCBs (sum 7) based on BC AW2000	0.02*	0.36	0.07	15.0	15.0	Human

\* Generic background concentration based on the 95<sup>th</sup> percentile of measurements in the research of Lame et al. (2008).

### Consequences for policy

The current maximal value for the land-use category 'residential' is 0.04 mg/kg d.w. The proposed maximal value (3.6 x10<sup>-4</sup> mg/kg d.w., based on mink) in this report is well below this value. Based on the methodology of using the 95-percentile of measured concentrations as the lower boundary for quality standards for reuse, the value of 0.02 mg/kg applies for the Maximal Value for 'residential'.

For the land-use category 'industrial', the proposed maximal value is considerably lower than the current maximal value of 0.5 mg/kg d.w. The derived PNEC for mustelids is the reason to lower the maximal value for 'industrial', especially when it comes to bare-green areas. Based on the methodology of using the 95-percentile of measured concentrations as the lower boundary for quality standards for reuse, the value of 0.02 mg/kg should also apply for the Maximal Value for 'industrial'. This derived value for 'industrial' would have a great impact on the reuse of soil and sediment. We advise evaluating the Maximal Value for industrial, taking into account the derived risk limits in this report.

# 1 Introduction

Within the framework of the Soil Quality Decree, municipalities must prepare soil quality maps for the reuse of soil and sediments in or on the soil. The soil quality maps form the basis for defining the quality requirements that apply to a location or the reuse of soil. In the 'Guideline on Soil Quality Maps', it is described how a soil quality map can be made and which substances are relevant in general. The guideline uses the default compounds listed in NEN 5740 as a starting point to determine which compounds are relevant. This list of compounds, in effect since July 1<sup>st</sup> 2008, includes polychlorinated biphenyls (PCBs) (sum 7).

## 1.1 Problem definition

The soil quality decree allows the reuse of soil if the quality of the soil complies with the maximum value belonging to the soil function and if the receiving soil has a comparable quality. The maximum values are divided in the soil functions 'residential with garden' (hereinafter 'residential') and 'Other greens, infrastructure, buildings and industry' (hereinafter 'Industrial'). Furthermore, the classification background value is used for undisturbed areas (e.g. nature areas).

To date, no scientifically based maximum value for the soil function 'residential with garden' exists for PCBs (sum 7). The maximum value for 'residential', therefore, was set equal to the background value AW2000<sup>1</sup> (0.020 mg /kg d.w.). By a policy decision, it was changed to 0.04 mg/kg in 2013 for practical reasons. The maximum value for the soil function 'industrial' is 0.5 mg/kg d.w., which is based on the intervention value for contaminated soils. The derivation of these values is not completely clear, but contains a high level of expert judgment.

For many municipalities, the absence of a risk-based maximum value for 'residential' posed problems, because batches of soil were classified as quality 'industrial' when the concentrations in the soil were only slightly higher than the generic background value AW2000 (0.020 mg /kg d.w.).

In 2010, through a policy decision, the soil quality decree was temporarily adjusted in such a way that batches of soil or sediments which were contaminated with PCBs up to a maximum of two times the background concentration (equal to 0.040 mg/kg d.w.) and which, for other compounds, were classified as clean could be reused in all land use categories. The choice to select two times the background concentration as a limit was based on a small-scale study conducted by the RIVM. In this study, it was concluded that there was no risk to humans and probably also not for ecology if two times the background value were used (Lijzen & Verbruggen, 2011). A more thorough evaluation conducted in 2013 learned that this conclusion was incorrect and the need for a more solid foundation remained, which led to this report.

This report provides a scientific underpinning for new reference values that form the basis for maximum values.

<sup>1</sup> AW2000 stands for background concentrations 2000 and refers to generic background concentrations in soil for different compounds in the Netherlands, based on the 95<sup>th</sup> percentile of measurements in relatively undisturbed areas. The AW 2000 values are described in Lame et al. (2008).

## **1.2 Reading guide**

In Chapters 2 and 3, the derivation of the ecotoxicological risk values and human risk values, respectively, is described. In Chapter 4, the human and ecotoxicological risk values are integrated and reference values for different soil functions are proposed according to the standard procedure to derive quality standards for the reuse of soil (Dirven et al. 2007). In Chapter 5, conclusions and consequences for policy are described.

## 2 Ecotoxicological risks

### 2.1 Introduction

When deriving reference values for ecotoxicology, as was done in 2007 by Dirven-van Breemen et al., a complete literature search is performed. For PCB, however, the available literature was too extensive. In a literature search, approximately 300 references were found with relevant toxicological data for mammals and birds. Because the assessment of all these data for many species would be very time-consuming, it was decided to limit the assessment to the species that is known to be one of the most sensitive organisms (Basu et al. 2007). This selected species is the mink (*Mustela vison*). Furthermore, secondary poisoning in the food chain is expected to be the most critical parameter for exposure to PCBs. In previous evaluations, the secondary poisoning by PCBs was not included and the risk limits were based on direct toxicity only, as was the procedure at that time (Verbruggen et al. 2001). As a result, the risks presented by PCBs without the inclusion of secondary poisoning are an underestimation. In this study, secondary poisoning is included in the derivation of the risk limits.

The toxic effects of PCBs on the mink have been studied extensively. It appears that species that are classified as mustelids are the ones that are most sensitive to the toxicity of PCBs. The sensitivity of the European otter (*Lutra lutra*) is comparable to that of the mink. The toxicity of PCBs is usually described by its dioxin-like toxicity, in which the potency of individual PCB congeners is expressed in toxic equivalency factors (TEF), which is the relative potency compared to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). The toxic equivalent (TEQ) is the summation of the product of toxic equivalency factors and concentrations. Concentration addition is thus used to estimate the overall toxicity of the dioxin-like substances (Van den Berg et al. 2006). Sediment quality criteria, reflecting the range of 1 to 50% effect on relative litter size for the otter, were 3 to 7 ng TEQ/kg organic carbon, based on bioaccumulation in otters and reproduction in mink (Traas et al 2001, Smit et al. 1996c), which would be in the order of 17 to 39 µg indicator PCBs/kg organic carbon.

### 2.2 Toxicity to mustelids

The literature on the toxicity of PCBs<sup>2</sup> contains hundreds of toxicity studies on dozens of different mammal and bird species at different stages of the life cycle when exposed to PCB mixtures in many ways. In the field, the animals that are higher in the food chain are more susceptible because of the biomagnification of PCBs in the food chain. Mink is considered a viable sentinel species for the environmental effects of PCBs since it is a high-trophic-level, fish-eating mammal which bioaccumulates PCBs and is sensitive to their toxic effects (Basu et al. 2007). Female mink and ferrets can pass on PCBs to their sucklings (Bleavins et al. 1982, Bleavins et al. 1984) and therefore male mink have higher PCB concentrations and male mink are used in environmental monitoring (Persson et al. 2013). The increased concentrations in males have also been observed in the European otter (*Lutra lutra*). It appears that female otters lose a

<sup>2</sup> It is noteworthy that not all studies referred to in this chapter include the 7 PCB congeners as used in the Dutch policy framework. Therefore, in this chapter the term PCBs includes various mixtures of PCB, whereas the rest of the report uses PCB (sum 7)

considerable amount of PCBs, especially the toxic planar PCBs, through lactation (Leonards et al. 1996, Smit et al. 1996a, Smit et al. 1996b).

A study was performed in which mink were fed with diets containing increasing amounts of contaminated carp from Saginaw Bay instead of uncontaminated marine fish (Heaton et al. 1995, Tillitt et al. 1996). Details for this study and the following studies are summarised in Table 2.1. The study lasted for 182 days, during which period the mink reproduced and nursed the kits. Adults were necropsied after 182 days for organ analysis, including analysis of PCBs in the liver. The survival and body weight of the kits were monitored until six weeks of age (weaning). The survival of the kits at the study's termination (6 weeks old) was the most sensitive endpoint in the study. In the control group, 85% of the kits survived. In the 10%, 20% and 40% carp diets, 28, 11, and 0% of the kits survived. Thus, a very steep dose-response curve was observed, from which an EC10 (effect concentration) can be obtained with reasonable certainty.

*Table 2.1 Overview of four toxicity studies involving mink (Mustela vison) given different diets containing fish contaminated with PCBs.*

<b>Reference</b>	Heaton et al 1995; Tillitt et al 1996	Bursian et al 2006a	Bursian et al 2006b,c	Bursian et al 2013 a,b
<b>Contaminated fish</b>	Common carp	Common carp	Goldfish and common carp	Common carp
<b>Source of fish</b>	Saginaw Bay, Michigan, USA	Saginaw River, Michigan, USA	Housatonic River, Massachusetts, USA	upper Hudson River, New York, USA
<b>Dose of contaminated fish</b>	0 (control), 10, 20, and 40%	0 (control), 10, 20, and 30%	0 (control), 0.22, 0.44, 0.89, 1.77, and 3.54%	0 (control), 2.5, 5, 10, 15, and 20%
<b>Uncontaminated fish</b>	Ocean fish scrap	Atlantic herring	Atlantic herring	Atlantic herring
<b>Exposed females per group</b>	12	10	12	15 or 10
<b>Exposed males per group</b>	3	0 (males not exposed)	0 (males not exposed)	5
<b>Start of exposure</b>	December 29, 1988	February 12, 2002	January 4, 2000	January 3, 2007
<b>Mating</b>	from February 13 to March 13, 1989	from March 1 to March 18, 2002	from March 1 to March 22, 2000	from March 1 to March 21, 2007
<b>Whelping</b>	from April 14 to May 11, 1989	from April 26 to May 11, 2002	from April 17 to May 5, 2000	from April 22 to May 9, 2007
<b>Weaning</b>	6 weeks	6 weeks	6 weeks	6 weeks
<b>End of exposure</b>	June 28, 1989	November 5, 2002	May 30–June 16, 2000	December 4 and 5, 2007

The study results can be expressed in several metrics: as a dose or a diet concentration (diet as such or caloric content) or as internal liver concentration, and as TEQ or as  $\Sigma$ PCB. This will facilitate the calculation of the reference values for PCBs according to the newly developed method for secondary poisoning based on the energy content of the diet (Verbruggen, 2014). However, it will

also facilitate the comparison of the toxicity observed for the mink with that for the otter, as will be shown below.

The different ways to express the 10% effect level are shown in Table 2.2. Because the composition of the PCB mixtures is given, the data on TEQ can be recalculated using the most recent TEF values (Van den Berg et al. 2006). It should be noted that compounds other than PCBs (polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDF)) do contribute to the overall TEQ as well. PCDDs and PCDF made up 9%, 37%, 26% and 28% of the recalculated TEQ in the control, with 10%, 20% and 40% carp diets, respectively.

*Table 2.2 10% and 50% effect levels for the survival of mink kits up to 6 weeks of age in the study using contaminated fish from Saginaw Bay (Heaton et al. 1995). Values are expressed in different metrics that are relevant for the assessment of secondary poisoning.*

	<b>10% effect</b>	<b>50% effect</b>	<b>Unit</b>
Dose	0.040	0.098	mg $\Sigma$ PCBs/kg <sub>bw</sub> /d
Dose	$9.6 \cdot 10^{-7}$	$2.3 \cdot 10^{-6}$	mg TEQ <sub>2005</sub> /kg <sub>bw</sub> /d
Dietary concentration	0.17	0.51	mg $\Sigma$ PCBs/kg <sub>diet</sub>
Dietary concentration	$4.1 \cdot 10^{-6}$	$1.2 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>diet</sub>
Dietary concentration	$3.0 \cdot 10^{-5}$	$8.2 \cdot 10^{-5}$	mg $\Sigma$ PCBs/kJ <sub>diet</sub>
Dietary concentration	$7.4 \cdot 10^{-10}$	$1.9 \cdot 10^{-9}$	mg TEQ <sub>2005</sub> /kJ <sub>diet</sub>
Liver concentration adult	1.0	1.8	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration adult	$8.3 \cdot 10^{-5}$	$1.8 \cdot 10^{-4}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>

In a second study, carp from the Saginaw River at Bay City, Michigan was added to the diet (Bursian et al, 2006a). The fraction was similar to the first study (10-30% contaminated fish). Males were not exposed. The time between the start of the exposure and the beginning of mating was shorter than in the other studies. In this study too, the composition of the PCB mixtures was given and the TEQ was recalculated using the most recent TEF values (Van den Berg et al., 2006). PCDDs and PCDF made up 94%, 62%, 63% and 60% of the recalculated TEQ in the control, with 10, 20, and 30% carp diets, respectively. The contribution of the non-ortho PCB 126 to the TEQ was around 32% in the contaminated fish, and of the mono-ortho PCB 118 3%. Body weight and food consumption were not given in this study. Consequently, no dose could be derived from the data. The study presented the nutritional information and, therefore, diet concentrations can be expressed on an energy basis. In this study, no effects on survival were found up to the highest concentration. The results are presented in Table 2.3. The lack of effects on survival starkly contrasts with the study from Saginaw Bay (Heaton et al. 1995; Tillit et al. 1996), in which strong effects were observed at equivalent TEQ. The authors indicate that the fish in the present study contain relatively high amounts of PCDDs and PCDFs. They point to the limitations of the TEQ approach, which assumes additivity, while antagonistic and synergistic effects may occur. The fact that the males were not exposed in this study is also considered to be an explanatory factor by the authors. Based on their subsequent studies (Bursian et al. 2006b,c; Bursian et al. 2013a,b), this may indeed be a relevant factor. In the study, in which male mink were not exposed (Bursian et al. 2006b,c), effect levels were higher than in the other two studies. Furthermore, the exposure time before mating is not mentioned by the authors, but this appeared to have been significantly shorter than in the other studies. In Table 2.1 the study details for all studies are compared.



Table 2.3 No observed effect and lowest observed effect levels for the survival of mink kits up to 6 weeks of age in the study using contaminated fish from the Saginaw River (Bursian et al. 2006a). Values are expressed in different metrics that are relevant for the assessment of secondary poisoning.

	No effect	Lowest observed effect	Unit
Dietary concentration	1.7	>1,7	mg $\Sigma$ PCBs/kg <sub>diet</sub>
Dietary concentration	$5.7 \cdot 10^{-5}$	> $5.7 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>diet</sub>
Dietary concentration	$2.1 \cdot 10^{-4}$	> $2.1 \cdot 10^{-4}$	mg $\Sigma$ PCBs/kJ <sub>diet</sub>
Dietary concentration	$7.1 \cdot 10^{-9}$	> $7.1 \cdot 10^{-9}$	mg TEQ <sub>2005</sub> /kJ <sub>diet</sub>
Liver concentration 6-w kit	19	>19	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration 6-w kit	$2.4 \cdot 10^{-4}$	> $2.4 \cdot 10^{-4}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>
Liver concentration 27-w kit	18	>18	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration 27-w kit	$7.8 \cdot 10^{-5}$	> $7.8 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>

A third study examined the toxicity of fish from the Housatonic River in Massachusetts (Bursian et al. 2006b,c). The study was very similar to the former one. In this study, as well, males were not exposed. The fraction of contaminated fish was much lower in this study due to the higher contamination. Again, kit mortality after 6 weeks was the most sensitive endpoint relevant for reproduction. Survival was only affected in the highest dose. In this study, too, the composition of the mixtures was given and the TEQ was recalculated using the most recent TEF values (Van den Berg et al. 2006). PCDDs and PCDF made up 39%, 19%, 13%, 9%, 6% and 6% of the recalculated TEQ in the control, with 0.22, 0.44, 0.89, 1.77 and 3.54% carp/goldfish diets, respectively. The contribution of the non-ortho PCBs to the TEQ was around 88% in the contaminated fish, with PCB126 alone accounting for around 81%.

Table 2.4 No observed effect and lowest observed effect levels for the survival of mink kits up to 6 weeks of age in the study using contaminated fish from the Housatonic River (Bursian et al. 2006b,c). Values are expressed in different metrics that are relevant for the assessment of secondary poisoning.

	No effect	Lowest observed effect	Unit
Dose	0.169	0.414	mg $\Sigma$ PCBs/kg <sub>bw</sub> /d
Dose	$9.6 \cdot 10^{-7}$	$2.3 \cdot 10^{-6}$	mg TEQ <sub>2005</sub> /kg <sub>bw</sub> /d
Dietary concentration	1.6	3.7	mg $\Sigma$ PCBs/kg <sub>diet</sub>
Dietary concentration	$1.2 \cdot 10^{-5}$	$5.0 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>diet</sub>
Liver concentration adult	3.08	3.13	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration adult	$5.0 \cdot 10^{-5}$	$1.9 \cdot 10^{-4}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>
Liver concentration 6-w kit	1.9	3.7	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration 6-w kit	$7.7 \cdot 10^{-5}$	$1.7 \cdot 10^{-4}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>
Liver concentration 31-w kit	3.5	8.6	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration 31-w kit	$7.7 \cdot 10^{-5}$	$1.5 \cdot 10^{-4}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>

A similar, but more recent study was performed with mink fed with contaminated food from the upper Hudson River (Bursian et al. 2013). In this study, the contribution of PCBs to the total TEQ in the diet was 97%, with non-ortho-PCBs accounting for 75% and PCB126 for 74%. Thus, PCB126 was by far the most important congener for the total TEQ. In the livers of the female mink, these percentages increased to 98%, 82% and 82%, respectively. The lowest

diet concentration is considered as NOEC (No Observed Effect Concentration). It should be noted that, if the number of kits whelped alive per dam is multiplied by the chance of surviving up to six weeks of age (point of weaning) and then by the chance of surviving from the point of weaning to 31 weeks of age, there is still an effect of 23% compared with the control. At all higher concentrations, none of the kits survived up to 31 weeks of age, pointing to a very steep dose-response relationship. Besides the steep dose-response relationship, the number of kits alive per dam was probably not impacted at the lowest exposure concentration, because it was higher than the control in the next exposure concentration. This leaves an increased mortality from birth until the end of the study of around 15% compared with the control group (there appears to be some inconsistencies between text, tables and figures). With this value being close to 10% and in view of the steep dose-response, it is further considered as a NOEC.

It should be noted that at six weeks of age the effect on mortality was not significant in the next higher concentration either. However, the body weight of the kits was significantly reduced at this concentration and higher concentrations at both three and six weeks of age. For a comparison with the other studies, the data based on kit mortality at six weeks of age are presented in Table 2.5.

*Table 2.5 No observed effect and lowest observed effect levels for the survival of mink kits up to 6 weeks of age in the study using contaminated fish from the upper Hudson River (Bursian et al. 2013a,b). Values are expressed in different metrics that are relevant for the assessment of secondary poisoning.*

	<b>No effect</b>	<b>Lowest observed effect</b>	<b>Unit</b>
Dose	0.15	0.27	mg $\Sigma$ PCBs/kg <sub>bw</sub> /d
Dose	$9.7 \cdot 10^{-7}$	$1.7 \cdot 10^{-6}$	mg TEQ <sub>2005</sub> /kg <sub>bw</sub> /d
Dietary concentration	1.5	2.8	mg $\Sigma$ PCBs/kg <sub>diet</sub>
Dietary concentration	$1.0 \cdot 10^{-5}$	$1.8 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>diet</sub>
Liver concentration adult	2.9	3.4	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration adult	$6.1 \cdot 10^{-5}$	$1.0 \cdot 10^{-4}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>

The more sensitive NOEC values for juvenile mortality (kit at 31 weeks of age) will be used in the derivation of the risk limits. These no observed effect levels are presented in Table 2.6.

*Table 2.6 No observed effect levels for the survival of mink kits up to 31 weeks of age in the study using contaminated fish from the upper Hudson River (Bursian et al. 2013a,b). Values are expressed in different metrics that are relevant for the assessment of secondary poisoning.*

	<b>NOEC</b>	<b>Unit</b>
Dose	0.070	mg $\Sigma$ PCBs/kg <sub>bw</sub> /d
Dose	$4.7 \cdot 10^{-7}$	mg TEQ <sub>2005</sub> /kg <sub>bw</sub> /d
Dietary concentration	0.72	mg $\Sigma$ PCBs/kg <sub>diet</sub>
Dietary concentration	$4.8 \cdot 10^{-6}$	mg TEQ <sub>2005</sub> /kg <sub>diet</sub>
Liver concentration adult	1.5	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration adult	$3.3 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>
Liver concentration 6-w kit	1.2	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration 6-w kit	$3.9 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>
Liver concentration 31-w kit	0.89	mg $\Sigma$ PCBs/kg <sub>liver</sub>
Liver concentration 31-w kit	$2.3 \cdot 10^{-5}$	mg TEQ <sub>2005</sub> /kg <sub>liver</sub>

A dose-response relationship was developed between liver vitamin A content and liver concentrations of PCBs in otters (*Lutra lutra*) that were found dead in the Netherlands (Murk et al. 1998). It appeared that there was a very sharp dose-response relationship, especially with the TEQ as determined by the CALUX assay. The data are based on measured concentrations in the liver. Data based on the sum of seven indicator PCBs were estimated on the basis of relationships between seven indicator PCBs and TEQ (Smit et al. 1996a). On the basis of the same study, a translation to a concentration in fish was made (Murk et al. 1998), which is also shown in Table 2.7.

Table 2.7 Levels corresponding to 10% and 90% reduction in vitamin A levels in livers from otters.

	10% effect	90% effect	Unit
Liver concentration	4	11	mg 7PCBs/kg <sub>liver, lw</sub>
Liver concentration	$2 \cdot 10^{-3}$	$5 \cdot 10^{-3}$	mg TEQ <sub>CALUX</sub> /kg <sub>liver, lw</sub>
Liver concentration	0.084	0.21	mg 7PCBs/kg <sub>liver, ww</sub>
Liver concentration	$8.4 \cdot 10^{-5}$	$2.1 \cdot 10^{-4}$	mg TEQ <sub>CALUX</sub> /kg <sub>liver, ww</sub>
Dietary concentration	$1.1 \cdot 10^{-6}$	$2.9 \cdot 10^{-5}$	mg TEQ <sub>CALUX</sub> /kg <sub>diet, lw</sub>
Dietary concentration	$6.8 \cdot 10^{-7}$	$1.8 \cdot 10^{-6}$	mg TEQ <sub>CALUX</sub> /kg <sub>diet, ww</sub>

It can be concluded based on a comparison of Tables 2.1 - 2.7 that, especially for the liver concentrations based TEQ concentration (on wet weight), there is a very good correspondence between the effect range in mink and otter:

- $8.3 \cdot 10^{-5}$  and  $1.8 \cdot 10^{-4}$  mg TEQ<sub>2005</sub>/kg<sub>liver</sub> in the livers of adult females correspond to 10 and 50% effect in kit mortality at six weeks of age in the first study using fish from Saginaw Bay, and
- $2.4 \cdot 10^{-4}$  and  $7.8 \cdot 10^{-5}$  mg TEQ<sub>2005</sub>/kg<sub>liver</sub> in liver of 6-week and 27-week old kits in the highest tested concentration correspond to no effect in kit mortality at six weeks of age in the second study using fish from the Saginaw River, and
- $5.0 \cdot 10^{-5}$ ,  $7.7 \cdot 10^{-5}$  and  $7.7 \cdot 10^{-5}$  mg TEQ<sub>2005</sub>/kg<sub>liver</sub> in the liver of an adult female, 6-week and 27-week old kits correspond to no effect in kit mortality at six weeks of age in the third study using fish from the Housatonic River, while  $1.9 \cdot 10^{-4}$ ,  $1.7 \cdot 10^{-4}$  and  $1.5 \cdot 10^{-4}$  mg TEQ<sub>2005</sub>/kg<sub>liver</sub> corresponded to increased mortality for six-week old kits, and
- $6.1 \cdot 10^{-5}$  mg TEQ<sub>2005</sub>/kg<sub>liver</sub> in the liver of an adult female corresponds to the NOEC for kit mortality at six weeks of age in the fourth study for the mink using fish from the upper Hudson River, while  $1.0 \cdot 10^{-4}$  mg TEQ<sub>2005</sub>/kg<sub>liver</sub> corresponded to an increased mortality at six weeks of age; for the mortality of kits at 31 weeks of age, liver concentration that resulted in no effects were as low as  $3.3 \cdot 10^{-5}$ ,  $3.9 \cdot 10^{-5}$ ,  $2.3 \cdot 10^{-5}$  mg TEQ<sub>2005</sub>/kg<sub>liver</sub> in the liver of an adult female, 6-week and 31-week old kits, respectively, and
- $8.4 \cdot 10^{-5}$  to  $2.1 \cdot 10^{-4}$  mg TEQ<sub>CALUX</sub>/kg<sub>liver</sub> marked the range of liver concentrations corresponding to a 10 to 90% decrease in hepatic vitamin A concentrations in the otter.

On the basis of these results, it can be concluded that mink and otter have a similar sensitivity to the toxicity of dioxin-like compounds, such as PCBs.

### 2.3 Biomagnification

Biomagnification expresses the accumulation of substances from lower trophic levels in the food chain to the predators. For an assessment of the exposure of terrestrial predators to these substances, the magnification in the terrestrial

food chain should be known. Biomagnification in mustelids, including the otter, and the rest of the terrestrial predators is described in this section.

### 2.3.1 *Biomagnification in otters*

In the 1990s, quality objectives for PCBs were developed based on the toxicity for the otter (*Lutra lutra*), which is considered as a sentinel species, and its closely related relative, the mink (Traas et al. 2001; Smit et al 1996c). Because the quality objectives for otters are based on internal effect concentrations in otter livers, the biomagnification from fish to otters is a key parameter in the derivation of these objectives. The biomagnification factors used are remarkably high, especially for the planar PCBs and consequently also for the TEQ (Smit et al. 1996a, Leonards et al. 1997).

The biomagnification studies used for this purpose, however, have some shortcomings. In both studies, the otter samples pre-dated the fish samples by several years. It is necessary, therefore, to verify whether the quality objectives for PCBs derived for the otter (Traas et al. 2001) are not too stringent because of the use of excessively high BMF values, especially for the planar non-ortho substituted PCBs.

In the Danish habitat (only freshwater lakes were used by Traas et al. 2001), fish were sampled in the period April-June 1995. Sediment was sampled concurrently. The nine otters (five males, four females) were retrieved from a database and were from 1988 or later (Smit et al. 1996a).

In the Dutch habitat, (freshwater lakes) sediment and invertebrates were sampled in June 1993. Fish samples of several species were collected from June 1990 to January 1991. The otters were found dead between 1982 and 1988 and were the last five dead otters found in the Netherlands. Apparently, they are not from the same location as the fish, although all were from the northern part of the Netherlands. Besides that, arithmetic means of concentrations have been used for the prey items, while the geometric mean was calculated for the otters. It is suggested that PCB77 is metabolized by otters because its relative contribution to the TEQ decreases from fish to otter and that PCB126 and PCB169 are increased due to selective retention. The authors conclude from the literature that this specific retention is not observed for other mammals and birds besides the otter (Leonards et al. 1997). Geometric means of lipid normalized concentrations do not show a decrease from fish to otter for PCB77, but there is very strong increase for PCB126 and PCB169.

As can be concluded, the temporal scale between the sampling of fish and the otters is large, which makes the resulting biomagnification factors less reliable. In the Dutch study as well, the spatial scale for the sampling is substantial, which makes the biomagnification parameters even less certain. Besides that, the otter itself is a migratory species, which makes it more complicated to link it to a diet in a restricted area. Especially the temporal aspect may lead to erroneously high BMF values if the prey species are sampled several years after the predator species in cases of declining environmental concentrations. With regard to this aspect, it is noteworthy that reductive dechlorination, especially, removes *meta* and *para* chlorine atoms which strongly reduce the non-*ortho* PCBs (i.e. PCB126 and PCB 169) with relatively short half-lives of 6 years, as assumed in the modelling exercises (Traas et al., 2001). On the contrary, it is stated that the concentration and profiles of PCBs have not markedly changed in the Netherlands (Leonards et al. 1998). Besides that, the BMF values obtained

from the Danish otters (Smit et al. 1996a) and from the Dutch otters (Leonards et al. 1997) show the same pattern, with the highest BMFs observed for the planar PCBs.

### 2.3.2 Biomagnification in other mustelids in comparison with otters

A similar biomagnification study was performed for other mustelids. Weasels (*Mustela nivalis*; 6 animals, 4 males; 2 females), stoats (*Mustela erminea*; 7 animals, 4 males; 3 females), and polecats (*Mustela putorius*; 4 animals, 1 male; 3 females) were collected in the period between 1985 and 1993 from the Oude Venen area or within a radius of 7 km (Leonards et al. 1998). The prey species of these terrestrial mustelids were sampled in 1993 in the Oude Venen or in the surrounding area. These prey species consisted of red vole (*Clethrionomys glareolus*), common vole (*Microtus arvalis*), wood mouse (*Apodemus sylvaticus*), common shrew (*Sorex araneus*), common hare (*Lepus europaeus*), common frog (*Rana temporaria*), lake frog (*Rana esculenta*), common toad (*Bufo bufo*), and natterjack (*Bufo calamita*).

All mustelids are able to metabolize PCBs with vicinal hydrogen atoms in the *meta* and *para* position. Evidence for this comes also from the fact that methylsulphonyl metabolites of these congeners were detected in the liver of various mustelids. Only metabolites of PCB149 were detected in all mustelid species. Metabolites of other congeners were mainly restricted to the otters. Besides that, concentration ratios of the methylsulphonyl metabolites to the parent compound PCB149 were far higher for otters than for the other mustelid species. This is also confirmed by the fact that biomagnification factors for these congeners are much lower in otters than in the other mustelids.

It is suggested that only polecats are able to metabolize PCB congeners with five or more chlorine atoms and vicinal hydrogen atoms on the *ortho* and *meta* position, such as PCB126. This is deduced from the fact that for polecats the ratio of PCB126 to PCB153 is much lower than it is for the other mustelids. For these other mustelid species, it is suggested that these PCB congeners, such as 126 and 169, are selectively retained in the liver. However, the biomagnification factor for this PCB congener 126 is among the highest for polecats in comparison with the other mustelids. The biomagnification factor for PCB153 and other non-metabolisable congeners is moreover much higher for polecats than it is for the other mustelids.

A closer look at the data shows that the relatively low amount of PCB126 in liver of polecats might be partly related to the food items as well. It appears that amphibians contain a much lower amount of PCB126 and the other co-planar PCBs in comparison with other prey organisms. Polecats are the only mustelids that eat substantial amounts of amphibians. In the diet used to calculate the biomagnification factors, this amount was 20%. Yet this was not determined in the study itself, but rather from older data taken from another location and the Oude Venen location from this study is a marsh type of landscape. Data from this study are analysed here for their relative amount of TEQ. The low amount of PCB126 and other coplanar PCBs in amphibians is reflected in the low TEQ value of the PCBs.

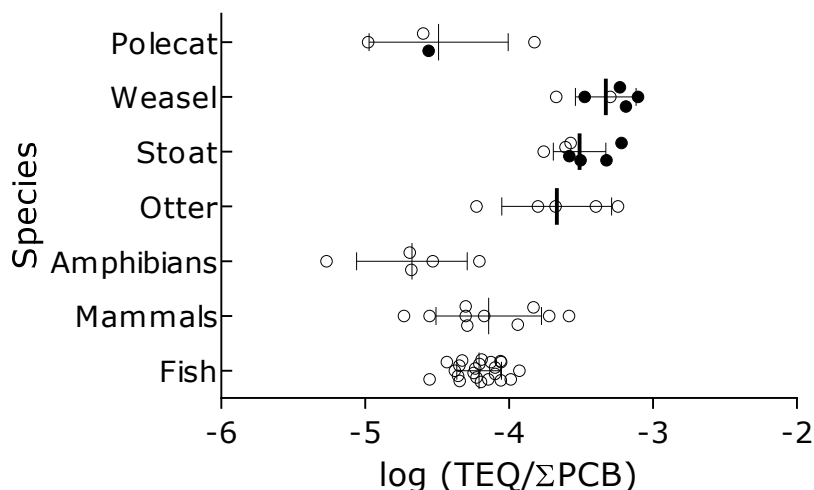


Figure 2.1 Ratio of TEQ values to total PCBs for various species of mustelids and some groups of prey items from the Oude Venen area (data from Leonards et al. 1997, 1998). Filled symbols refer to males for the terrestrial mustelids.

Another interesting aspect is the fact that, in the case of stoat and weasel, more than half of the individuals were males, while in the group of polecats there was only one male. For both weasel and stoat, it appeared that the males had relative TEQ concentrations compared with the sum of PCBs that were about twice those of females. For polecats, there is only one male and therefore a meaningful comparison is impossible, especially due to the large variation in the ratio between TEQ and the sum of PCBs. A possible explanation for this large variation up to a factor of 14 could be a difference in diet of the polecat as discussed above. The study on otters did not address the difference between male and female otters explicitly, although the differences caused by lactation were recognized (Leonards et al. 1996; Smit et al. 1996a; Smit et al. 1996b). For the Dutch habitat, the gender of the otters was not given at all (Leonards et al. 1997). For the Danish habitat, the gender and life stage was reported and in the appendix BMF values were given separately for male and female otters. On a TEQ basis, the BMF was 174 for males and 44 for females, while the geometric mean was 95 (Smit et al. 1996a). This means that the data for the stoat and weasel are in line with the data for the otter.

In a laboratory toxicity study with mink (*Mustela vison*), the accumulation of PCBs was determined as well (Tillitt et al. 1996). In such a study, diet concentrations are controlled and the concentration in mink can be directly coupled to the concentrations in the diet. In this way, temporal and spatial variation, as encountered in the field studies, is eliminated. In line with the studies for other mustelids, biomagnification factors were highest for the planar PCB congeners 126 and 169 in comparison with total PCBs (BMF = 8-15 for PCB126 and 12-21 for PCB169 in comparison with 2.4-3.9 for  $\Sigma$ PCB, based on Aroclor standards). Leonards et al. (1998) stated that there was no explanation for the fact that the BMFs for PCB126 were much higher in stoat and otter than they were in mink. However, it should be noted that not only PCB126 but all PCBs had lower biomagnification factors in the mink. There are two plausible explanations for this. First, the mink had been exposed in captivity for a limited exposure time of 182 days, while feral mustelids were exposed their whole life long. Perhaps even more important, however, is the fact that the mink that were

analysed were lactating females. Nursing is known to have a dramatic decreasing effect on the PCB concentration in the parent female (Leonards et al. 1996; Smit et al. 1996a; Smit et al. 1996b).

### 2.3.3 Biomagnification from earthworms to shrews

In a field study, earthworms and shrews were sampled from two different locations in the Kalamazoo River floodplain (Blankenship et al., 2005). One area, Fort Custer State Recreation Area (FC), was a reference site and the other area, Former Trowbridge Impoundment (TB), was a contaminated site. Not all congeners were separated from each other, but the indicator PCBs mostly dominated the non-separated peaks. The BMFs were calculated for the different congeners, including the indicator PCBs, the sum of the seven indicator-PCBs ( $\Sigma 7$ PCB), the planar PCBs and the sum of the TEQ. For both sites, the worms were rinsed and directly determined (fresh) or allowed to clear their gut for 24 to 48 hours (depurated). In Table 2.8 only the values for the indicator PCBs and the planar PCBs are given.

*Table 2.8 Lipid-normalized biomagnification factors from earthworms to shrews taken from two sites; Fort Custer reference site, Trowbridge contaminated site, in the Kalamazoo River floodplain (Blankenship et al. 2005). Indicator PCBs are given in bold, (co)planar PCBs are given in italics.*

PCB congener	Fort Custer		Trowbridge	
	Fresh	Depurated	Fresh	Depurated
<b>28,31</b>	0.45	0.02	0.15	0.20
<b>52</b>	5.68	0.74	0.04	0.06
56,92,84,90, <b>101</b> ,113	3.39	0.80	0.06	0.10
<b>118</b>	1.62	0.45	2.40	2.72
<b>138</b> , 158	1.09	1.29	2.62	3.54
<i>105,132</i> , <b>153</b>	2.61	1.90	3.07	4.14
<b>180</b>	4.53	7.84	12.20	9.45
$\Sigma 7$ PCB	2.28	0.96	0.85	1.23
<i>156,171,202,157,201</i>	42.02	2.02	4.35	6.73
<i>128, 167,185</i>	2.27	5.16	3.45	4.04
<i>77</i>	0.79	0.25	0.07	0.13
<i>81</i>	1.40	1.51	0.50	0.84
<i>126</i>	2.94	1.38	22.22	31.85
<i>169</i>	1.44	1.73	0.68	2.13
TEQ	2.63	1.41	12.31	17.53

In a similar study in the Netherlands, soil, earthworm and shrew samples were collected from two flood plain sites of the Rhine delta (Hendriks et al., 1995). Worms were allowed to clear their gut for 24 hours to remove the soil present in the digestive tract. From these data, BMF values can be calculated. The results for the seven indicator PCBs and, if available, the planar PCBs are tabulated in Table 2.9.

Table 2.9 Lipid-normalized biomagnification factors from earthworms to shrews at two flood plain sites in the Rhine delta (Hendriks et al. 1995). Indicator PCBs are given in bold, (co)planar PCBs are given in italics.

PCB congener	Ochten	Gelderse Poort
<b>28</b>	A	6.32
<b>52</b>	2.20	3.00
<b>101/090</b>	2.95	2.30
<i>118</i>	A	ab
<b>138/163/164</b>	4.29	2.78
<b>153</b>	6.67	6.25
<b>180</b>	B	ab
$\Sigma$ 7PCB	3.75	3.50
<i>105/(132)</i>	Ab	ab
<i>123/147</i>	6.00	14.08
<i>132/(105)</i>	2.46	a
<i>156</i>	Ab	ab
<i>167</i>	Ab	ab
<i>189</i>	Ab	ab

a No value for shrews reported

b No value for earthworms reported

In another study from the Netherlands (Van den Brink, 2000), the food chain of the little owl was studied in two areas: the Gelderse Poort and the Achterhoek. The data for the Gelderse Poort contained both earthworm and shrew concentrations. BMF values could be derived from these data (Table 2.10).

Table 2.10 Lipid-normalized biomagnification factors from earthworms to shrews at two locations in the Gelderse Poort (GP) (Van den Brink, 2000). All PCBs are indicator PCBs, (co)planar PCB 118 is given in italics.

PCB congener	GP9	GP11
28	0.15	0.18
52	0.05	0.08
101	0.04	0.07
<i>118</i>	0.76	0.08
138	0.80	0.91
153	0.41	0.89
180	1.16	1.67
$\Sigma$ 7PCB	0.55	0.78

#### 2.3.4 Biomagnification in other birds and mammals

A food web study was performed with tree swallows (*Tachycineta bicolor*) in the Hudson River area, in New York state, US (Echols et al. 2004). Two of the sites, Remnant Deposit 4 and Special Area 13 are known areas of high PCB contamination. It appeared that congener pattern in food (diptera, collected as food bolus from the swallows), as well as eggs, nestlings, and adult swallows, were different from individual technical Aroclor mixtures, but resembled a combination of Aroclor 1248 and Aroclor 1254 (determined with principle component analysis, PCA). The congener pattern in the eggs was closely related to that in the adults, while that in nestlings was closely related to the food (clusters in the PCA).



The reported uptake rates from these areas were not equal to the daily dose the nestlings received with the food, because uptake efficiency, metabolism, and elimination also played a role. Metabolism was relatively unimportant, given the fact that the congener patterns in food and nestlings were comparable. However, elimination over 10 days may account for a significant difference between these values and the daily dose received.

It is stated that the use of a biomagnification factor in this case was not appropriate, because the nestlings were far from a steady state. Indeed, from the supporting information, it is evident that the total lipid normalized PCB concentrations in 5-d, 10-d and 15-d old nestlings was only 7, 12 and 15% of that of adult birds, who might have been exposed elsewhere as well, yet 0.9, 1.5 and 1.9 times higher than the lipid normalized concentration in food (diptera), respectively. This shows that, indeed, the biomagnification process was not in a steady-state after 15 days.

The ratio of the seven indicator PCBs as a fraction of total PCBs (103 peaks, 121 congeners) appeared to be rather constant over all four sampling sites and over all biota, varying between 20 and 34%. However, eggs and adults seemed to have a somewhat higher content of indicator PCBs. This was also observed for all samples from the site at Champlain Canal, which is located upstream from the contaminated sites. These eight samples had an average percentage of 31% indicator PCBs. The lower percentages of indicator PCBs might, therefore, be representative of this particular contamination.

In another food web study involving tree swallows (Papp et al. 2007), the samples were analysed for 85 congeners. Sixty-six congeners were detected, 59 of which were detected in all nestling samples. In the two groups of insects, Hexagenia and Chironomidae, only 53 and 34 congeners were detected, respectively. Of the seven indicator-PCBs, PCB 28 was not reported, so the sum derived from this study was based on the remaining six congeners. The ratio between the 13-d old nestlings and insects (hexagenia and chironimidae) varied between 0.55 and 1.61 for the six congeners, with a value of 1.05 for the sum of the six congeners. These values were slightly lower than in the other study involving tree swallows. No time trend was available from this study.

In a study that examined the magnification from food to great tits (*Parus major*), it appeared that the main food item for the nestlings (caterpillars) was not the most important source for PCBs, but rather maternal transfer to the eggs. As concentrations of nestlings decreased instead of increased with age, the ratio between PCB concentration in nestlings and caterpillars was not considered to be a good metric for biomagnification (Dauwe et al. 2006). In this study, a total of 22 PCBs were analysed. Of the seven indicator PCBs, PCB 28, PCB 52 and PCB 118 were not reported, but probably belonged to the set of 22 congeners.

Based on the studies involving the nestlings of tree swallows and great tits, it appears that the biomagnification was not necessarily representative of the exposure from local food sources, as is reflected by the important role played by the transfer from adult to eggs, the build-up of or the decrease in nestling concentrations dependent on the situation, and the observed variability in congener pattern between adults, eggs and nestlings. These biomagnification studies, therefore, appear to be rather unsuitable for assessing the transfer of PCBs from local soil sources to birds.

Reported biomagnification factors for PCB congeners between passerine birds and sparrowhawks were around 20, and BMF between rodents and buzzards were around 40, whereas BMF between rodents and fox were somewhat lower (BMF around 3). Woodmouse (*Apodemus sylvaticus*) and bank vole (*Clethrionomys glareolus*) were sampled around Antwerp in February and March 2001. The great tits were sampled around Antwerp between 2001 and 2003 (Voorspoels et al. 2007). The foxes (*Vulpes vulpes*) were from the south of Belgium and were sampled between October 2003 and March 2004 (Voorspoels et al. 2006a), while the buzzards (*Buteo buteo*) and the sparrowhawks (*Accipiter nisus*) were from the eastern part of Flanders and were sampled between November 2001 and March 2003 (Voorspoels et al. 2006b). Therefore, the reliability of these BMF values can be argued because prey and predators are not from the same ecosystem.

## 2.4 Terrestrial bioaccumulation

Bioaccumulation in soil is described by biota-to-soil-accumulation-factors (BSAF), which express the ratio of the concentrations of a substance in terrestrial organisms to its concentration in soil. To make a link between the concentrations in the prey organisms for terrestrial predators and the concentration in the soil, the bioaccumulation in terrestrial species must be known. Bioaccumulation from soil in terrestrial species is described in this section.

### 2.4.1 Bioaccumulation from soil to earthworms

Earthworms (*Lumbricus terrestris* L.) were exposed to 25 different soils for 15 days (Krauss et al. 2000). The soils were from Bavaria, Germany, and covered a wide range of land use and texture. Preliminary experiments with three soils showed that 15 days was sufficient to approach equilibrium. The worms were allowed to purge their gut for 48 hours on wet filter paper. The BSAF were not given for each soil separately, but presented in a figure with mean, minimum and maximum values. The values that were derived are presented in Table 2.11. Only the values for the seven indicator PCBs are presented here.

Table 2.11 Lipid and organic carbon normalized BSAF values for earthworms in 25 soils from Germany (Krauss et al. 2000).

PCB congener	geomean	minimum	maximum
28	71	1.8	378
52	11	0.62	41
101	2.3	0.24	8.0
118	1.3	0.10	3.9
138	1.1	0.20	4.5
153	1.1	0.22	4.5
180	0.70	0.15	3.1

In a field study, soil and earthworms were sampled from two different locations in the Kalamazoo River floodplain (Blankenship et al. 2005). One area, Fort Custer State Recreation Area (FC), was a reference site and the other area, Former Trowbridge Impoundment (TB), was a contaminated site. Not all congeners were separated from each other, but the indicator PCBs mostly dominated the non-separated peaks. The BSAFs were calculated from the presented concentrations for the different congeners, including the indicator PCBs, the sum of the seven indicator-PCBs, the planar PCBs and the sum of the

TEQ. For both sites, the worms were rinsed and directly determined (fresh) or allowed to clear their gut for 24 to 48 hours (depurated). In Table 2.12, only the values for the indicator PCBs and the planar PCBs are given.

*Table 2.12 Lipid and organic carbon normalized BSAF values for earthworms taken from two sites; Fort Custer reference site, Trowbridge contaminated site in the Kalamazoo River floodplain (Blankenship et al. 2005). Indicator PCBs are given in bold, (co)planar PCBs are given in italics.*

PCB congener	Fort Custer		Trowbridge	
	Fresh	Depurated	Fresh	Depurated
<b>28</b> ,31	0.31	5.72	0.61	0.46
<b>52</b>	0.33	2.54	0.98	0.69
56,92,84,90, <b>101</b> ,113	1.42	6.02	0.84	0.49
<b>118</b>	0.83	3.01	0.50	0.44
<b>138</b> , 158	2.05	1.73	0.77	0.57
105,132, <b>153</b>	1.56	2.15	0.58	0.43
<b>180</b>	1.47	0.85	0.25	0.32
Σ7PCB	1.31	3.12	0.75	0.52
156,171,202,157,201	0.10	2.04	0.34	0.22
128, 167,185	0.77	0.34	0.35	0.29
77	1.41	4.44	0.42	0.23
81	2.67	2.47	0.55	0.33
126	2.02	4.31	0.29	0.20
169	3.10	2.59	6.31	2.01
TEQ	2.14	4.01	0.38	0.27

In a similar study in the Netherlands, soil and earthworm samples were collected from two flood plain sites in the Rhine delta. Worms were allowed to clear their gut for 24 hours to remove the soil present in the digestive tract. The results for the seven indicator PCBs and some (co)planar PCBs are tabulated in Table 2.13.

*Table 2.13 Lipid and organic carbon normalized BSAF values for earthworms taken from two flood plain sites in the Rhine delta (Hendriks et al. 1995). Indicator PCBs are given in bold, (co)planar PCBs are given in italics.*

PCB congener	Ochten	Gelderse Poort
<b>28</b>	b	0.16
<b>52</b>	0.69	0.20
<b>101</b> /090	0.61	0.31
<b>118</b>	0.53	A
<b>138</b> /163/164	0.51	0.35
<b>153</b>	0.39	0.20
<b>180</b>	a	A
Σ7PCB	0.45	0.24
105/(132)	a	A
123/147	0.96	0.42
132/(105)	0.55	0.32
156	a	A
167	a	A
189	a	A

*a No value for earthworms reported*

*b No value for soil reported*

In another study from the Netherlands (Van den Brink, 2000), the food chain of the little owl was studied in two areas: the Gelderse Poort and the Achterhoek. The data for the Gelderse Poort contained both soil and earthworm concentrations. BSAF values could be derived from these data (Table 2.14).

*Table 2.14 Lipid and organic carbon normalized BSAF values for earthworms from four sites in the Gelderse Poort (GP) floodplain (Van den Brink, 2000).*

PCB congener	GP1	GP9	GP11	K15
28	0.50	0.29	0.17	0.31
52	2.42	1.45	0.65	0.92
101	3.68	2.54	1.44	1.17
118	0.58	0.37	0.74	0.55
138	3.73	3.43	1.73	1.60
153	3.98	4.22	1.50	1.18
180	2.42	2.30	1.18	0.90
Σ7PCB	2.87	2.53	1.27	1.09

Another field study examined the uptake of worms in open woodland and grassland from two sites near the city of Antwerp in Belgium (Vermeulen et al., 2010). No gut depuration of the earthworms was allowed. The BSAF values calculated from these data are presented in Table 2.15.

*Table 2.15 Lipid and organic carbon normalized BSAF values for earthworms taken from two sites near the city of Antwerp (Vermeulen et al. 2010).*

PCB congener	geomean	minimum	maximum
Brasschaat grassland			
101	1.29	0.79	1.88
118	0.58	0.36	0.85
138	0.61	0.37	0.88
153	0.61	0.37	0.88
180	0.58	0.36	0.85
ΣPCB (10 congeners)	1.05	0.64	1.53
Brasschaat open woodland			
101	2.31	1.60	4.99
118	1.35	0.94	2.93
138	1.64	1.14	3.55
153	1.40	0.97	3.03
180	1.71	1.18	3.70
ΣPCB (10 congeners)	2.55	1.76	5.51
Hoboken grassland			
101	1.22	0.43	2.65
118	0.55	0.19	1.19
138	0.57	0.20	1.25
153	0.57	0.20	1.25
180	0.55	0.19	1.19
ΣPCB (10 congeners)	0.99	0.35	2.15
Hoboken woodland			
101	0.68	0.49	1.00
118	0.40	0.29	0.59
138	0.49	0.35	0.71
153	0.42	0.30	0.61
180	0.51	0.36	0.74

PCB congener	geomean	minimum	maximum
∑PCB (10 congeners)	0.75	0.54	1.10

Another study from China didn't specify the soil type and both earthworm and soil concentrations were given on the basis of dry weight (Zhao et al. 2006). The BSAF values that were derived from these concentrations are given in Table 2.16.

*Table 2.16 Dry-weight-based BSAF values for earthworms taken from a polluted farmland (Zhao et al., 2006). Indicator PCBs are given in bold, (co)planar PCBs are given in italics.*

PCB congener	WenTai area, Zhejiang Province of China
<b>28</b>	5.3
<b>52</b>	8.4
<b>101</b>	12.2
<i>118</i>	7.6
<b>138</b>	21.7
<b>153</b>	16.9
<b>180</b>	12.9
∑7PCB	8.1
<i>77</i>	2.4
<i>81</i>	A
<i>105</i>	6.6
<i>114</i>	16.6
<i>126</i>	4.4
<i>156</i>	13.1
<i>157</i>	10.7
<i>167</i>	10.2
<i>169</i>	B
<i>189</i>	4.9
WHO-TEQ (probably 1998)	4.8

*a Worm concentration below detection limit*

*b Both worm and soil concentrations below detection limit*

With standard values (Jager, 1998) for the lipid content and dry-weight content of worms (1% and 16% respectively) and an organic carbon content of 2%, these values should be roughly divided by 3 to obtain a normalized BSAF, but the organic carbon content of paddy fields can be even lower than that (Zhang et al. 2012), leading to lower BSAF values as well.

#### 2.4.2 Bioaccumulation from soil to shrews

Because in three studies data for shrews were also available in addition to the concentrations in soil and earthworms, the accumulation can also be expressed as the ratio of the concentration in shrews compared with that in the soil directly. The studies are from the Kalamazoo River floodplain (Blankenship et al. 2005), from two flood plain sites in the Rhine delta, Ochten and Gelderse Poort (Hendriks et al. 1995) and from two sites in the Gelderse Poort floodplain (Van den Brink, 2000) (Tables 2.17-2.19).

Table 2.17 Lipid and organic carbon normalized BSAF values for short-tailed shrews (*Blarina brevicauda*) and masked shrews (*Sorex cinereus*) taken from two sites in the Kalamazoo River floodplain (Blankenship et al. 2005); Fort Custer reference site, Trowbridge contaminated site.

PCB congener	Fort Custer	Trowbridge
<b>28,31</b>	0.14	0.09
<b>52</b>	1.88	0.04
56,92,84,90, <b>101</b> ,113	4.81	0.05
<b>118</b>	1.36	1.20
<b>138</b> , 158	2.23	2.01
105,132, <b>153</b>	4.08	1.77
<b>180</b>	6.67	3.03
Σ7PCB	2.98	0.64
156,171,202,157,201	4.13	1.47
128, 167,185	1.74	1.19
77	1.12	0.03
81	3.73	0.28
126	5.93	6.37
169	4.46	4.29
TEQ	5.64	4.66

Table 2.18 Lipid and organic carbon normalized BSAF values for white-toothed shrews (*Crocidura russula*) and common shrews (*Sorex araneus*) taken from two flood plain sites in the Rhine delta (Hendriks et al. 1995).

PCB congener	Ochten	Gelderse Poort
<b>28</b>	ab	1.04
<b>52</b>	1.51	0.60
<b>101/090</b>	1.79	0.71
<b>118</b>	a	A
<b>138/163/164</b>	2.21	0.97
<b>153</b>	2.57	1.26
<b>180</b>	11.03	A
Σ7PCB	1.70	0.82
105/(132)	a	A
123/147	5.77	5.88
132/(105)	1.36	A
156	a	A
167	a	A
189	a	A

a No value for shrews reported

b No value for soil reported

*Table 2.19 Lipid and organic carbon normalized BSAF values for white-toothed shrews (*Crocidura russula*) taken from two sites in the Gelderse Poort (GP) floodplain (Van den Brink, 2000).*

<b>PCB congener</b>	<b>GP9</b>	<b>GP11</b>
28	0.04	0.03
52	0.07	0.05
101	0.10	0.10
118	0.29	0.06
138	2.74	1.57
153	1.73	1.34
180	2.67	1.96
Σ7PCB	1.39	0.99

The six lipid and organic carbon-normalized BSAF values seem to follow a normal distribution (Figure 2.2). At the same time, however, there appears to be a strong correlation between the log BSAF and the logarithm of the soil concentration. For this reason, the linear relationship between the concentration of the 7 indicator-PCBs in the shrew and that in soil is used for the calculation.

Unfortunately, very few data are available for other small prey species, apart from these data for shrews. From the Kalamazoo river floodplain, only additional data for combined small mammals other than shrews and for house wren eggs, nestlings and adults are available (Blankenship et al. 2005). The same pattern as observed in Figure 2.2 was found for these species as well. However, the small mammals have a lower accumulation than the shrews, the house wren nestlings almost had a similar accumulation, but the house wren eggs and adults did have higher accumulation than the shrews. For this reason, the shrews are considered here as representative species for small prey birds and mammals. In the Kalamazoo river project, more species were found, but these data are not available. It is recommended that, in future analysis, accumulation in small terrestrial birds and mammals be more thoroughly investigated.

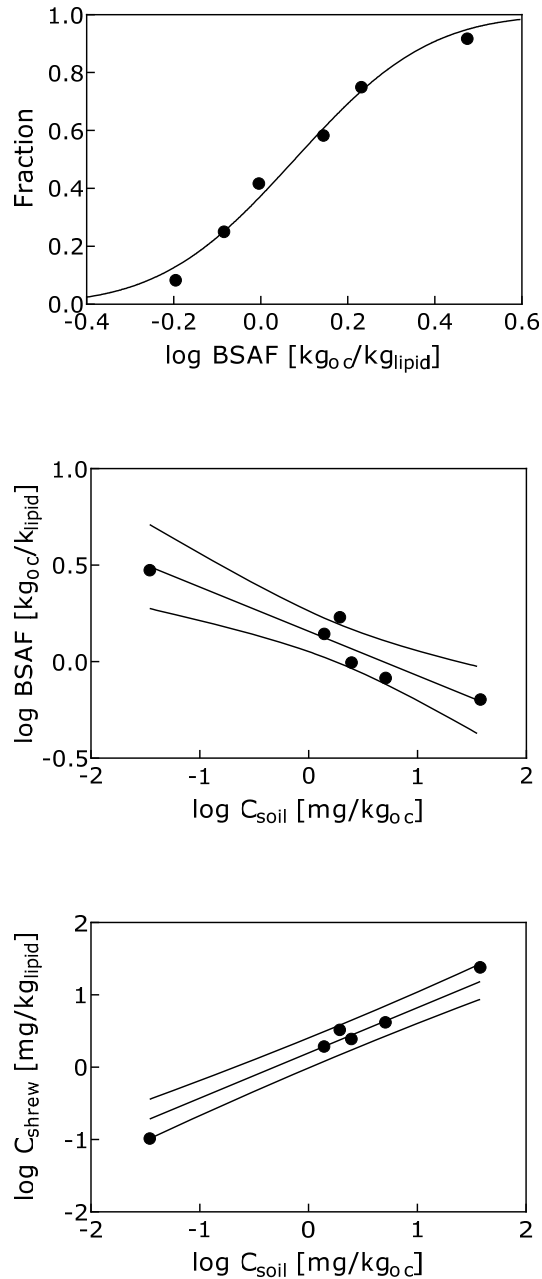


Figure 2.2 Lipid and organic carbon-normalized BSAF values for several shrew species taken from six sites in the Kalamazoo River and river Rhine floodplains (Blankenship et al. 2005, Hendriks et al. 1995, Van den Brink, 2000). In the upper figure, the distribution of the BSAF values is presented. In the middle figure, the BSAF and, in the lower figure, the concentration in shrews is plotted as a function of the soil concentration.

## 2.5 Relationship between indicator PCBs and TEQ

Dioxin-like PCBs are considered to be responsible for the observed toxicity in mustelid species. This is expressed in the TEQ. However, coplanar PCBs are



usually not measured in environmental samples. Very often the seven indicator PCBs are measured. In this section, it is examined whether correlations exist between TEQ and the indicator PCBs in the terrestrial environment. Such correlations will make it possible to predict the dioxin-like toxicity on the basis of the concentration of the non-planar indicator PCBs.

There appears to be a strong correlation between the toxic equivalent (TEQ) and the concentration of the seven indicator-PCBs ( $\Sigma 7\text{PCB}$ ) in fish (Babut et al. 2009). The TEQ in this study was based on the former set of TEF values (Van Den Berg et al. 1998), which was slightly revised in 2005 (Van den Berg et al. 2006). Data were expressed on a fresh weight basis. The data in the data set contain only fish.

In this report, it was further analysed whether such a correlation exists for other species and compartments as well. For some of the data analysed in this study, values for both the seven indicator PCBs and the non-ortho and mono-ortho PCBs are available.

In some studies, it does indeed appear that sometimes there is a very strong correlation between these parameters, e.g. for fish, mustelids (Smit et al. 1996a) and the terrestrial ecosystem (Blankenship et al. 2005). As expected, correlations improve if data are lipid normalized and not expressed on a fresh weight basis, as was observed for all three groups from the two studies mentioned above (data not shown). There is, however, a difference between different groups. This is, for example, shown clearly by the data for fish and otters from the same area (Smit et al. 1996a). The line is higher for otters, indicating a relative increase in the dioxin-like PCB congeners. The data for the terrestrial ecosystem (Blankenship et al. 2005) are somewhere in between the data sets for fish and otters (see Figure 2.3).

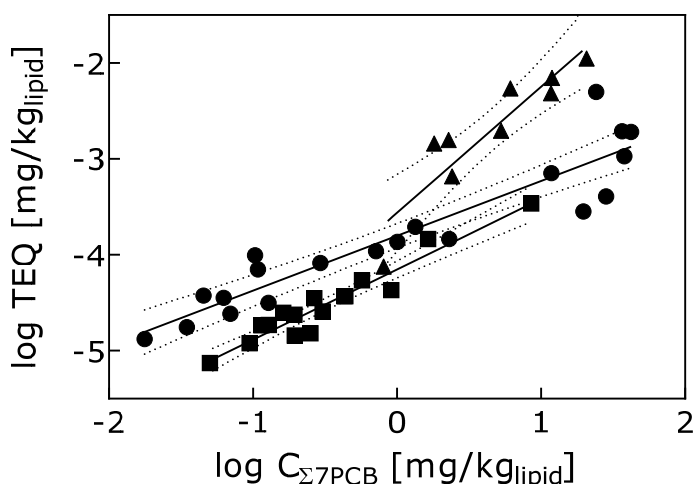


Figure 2.3 Correlation between the concentration of the seven indicator-PCBs and the TEQ. Circles are data taken from the terrestrial ecosystem (Blankenship et al., 2005), squares are data for fish and triangles data for otters (Smit et al. 1996a).

In general, it seems that predatory birds and mammals, especially, accumulate the planar PCBs to a relatively high content in comparison with the seven indicator PCBs. This was observed not only for the Eurasian otter, but also for

the Eurasian eagle owl (*Bubo bubo*) (Gómez-Ramírez et al., 2012). This observation seems to be indicative of the metabolism of PCB congeners with vicinal hydrogen atoms and the selective retention of the planar congeners (Gómez-Ramírez et al. 2012, Leonards et al. 1998, Leonards et al. 1997). The TEQ concentration in eggs of the American herring gull (*Larus smithsonianus*) also appears to be rather high (Metcalf et al. 1997) and this species cannot be considered as a prey species.

After removing these values, a huge data set remains for which data on both the planar PCBs and the indicator PCBs are available and for which concentrations can be expressed on a lipid weight basis. The data for which reasonable estimations can be made comprise several small mammals, several small birds including eggs, many fish, terrestrial and aquatic invertebrates, earthworms, terrestrial plants, and soil (Smit et al. 1996a, Blankenship et al. 2005, Zhao et al. 2006, Çakroğullar et al. 2010, Echols et al. 2004, Schmid et al. 2007, Metcalfe et al. 1997; Kay et al. 2006). After removing the statistically significant outlier for plankton (Metcalf et al. 1997), a strong correlation between TEQ and  $\Sigma 7\text{PCB}$  is found ( $r^2=0.86$ ,  $Sy.x=0.32$ ,  $n=97$ ).

Although the correlation between TEQ and  $\Sigma 7\text{PCB}$  for all data is good, the correlation strongly improves by taking the data for birds and mammals only, i.e. the target species for this terrestrial secondary poisoning assessment ( $r^2=0.95$ ,  $Sy.x=0.20$ ) (see Figure 2.4).

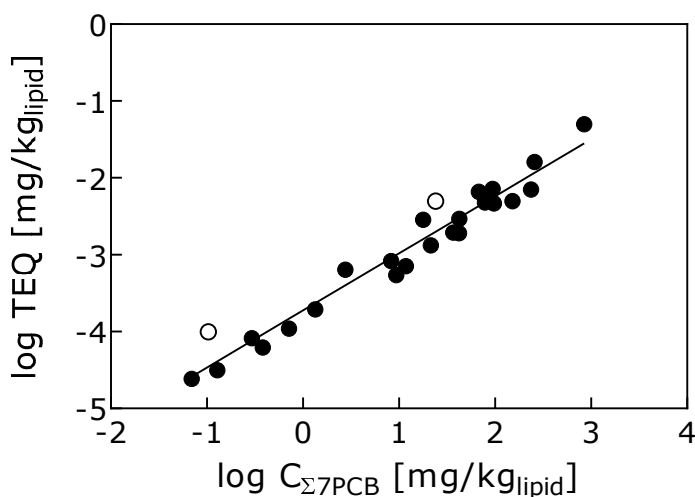


Figure 2.4 Correlation between TEQ and  $\Sigma 7\text{PCB}$  for bird and mammals species. Open symbols denote the shrews.

The few data there are for the shrews seem to be above the line (see Figure 2.4). This would accord with the higher TEQ values for the otter and other predatory birds and mammals. The shrew can be considered as a carnivorous mammal as well. However, the data for the shrew are not outliers in the correlation between TEQ and  $\Sigma 7\text{PCB}$ , as shown in Figure 2.4. The data from the bioaccumulation studies using mustelids (Leonards et al. 1998), for which the sum of the indicator PCBs could not be calculated, show that the TEQ in relation to the total PCB concentration is relatively low in the only shrew that was measured compared with that of the rest of the terrestrial organisms (

Figure 2.5). However, the TEQ shown was recalculated using presented data by assuming that the concentrations of all coplanar PCBs below the detection limit were zero. Although this TEQ might have been higher if half of the detection limit had been taken for the concentrations below the detection limit, the value for the shrew is still not above the line for all data.

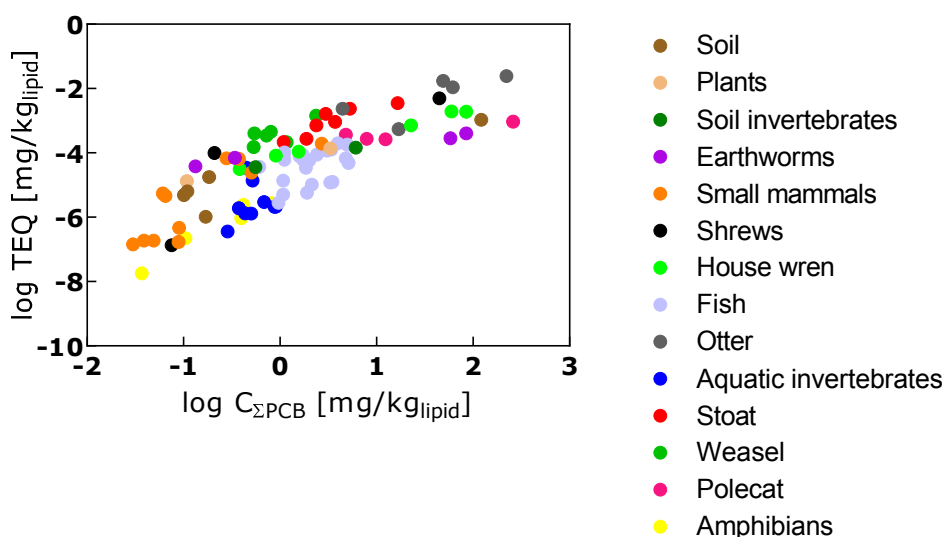


Figure 2.5 Correlation between concentrations of the sum of all PCBs and the TEQ, data taken from the terrestrial ecosystem (Blankenship et al. 2005), and data taken from mustelid bioaccumulation studies (Leonards et al. 1998, Leonards et al. 1997).

It can be concluded that relationships exist between TEQ and the concentration of the 7 indicator PCBs. These correlations may be variable for different groups of species and the relative TEQ is often higher in predators and lower in aquatic species. There appears to be a good correlation between TEQ and  $\Sigma 7\text{PCB}$  for small terrestrial birds and mammals. Therefore, this correlation between TEQ and  $\Sigma 7\text{PCB}$  for all data taken from birds and mammals except predators is used to quantify the dioxin-like toxicity for mustelids based on the  $\Sigma 7\text{PCB}$  concentration in their prey.

## 2.6 Calculation of the final, predicted no effect concentration

The final reference values for PCBs in soil were calculated from the information evaluated above in several steps. First of all, the no observed effect level for mink was recalculated to indicate a concentration in its prey, e.g. small birds and mammals. For this purpose, the newly developed method to normalize diet concentration to its energy content (Verbruggen, 2014) was applied. Next, the PNEC was calculated from this NOEC value for the mink.

Because the PNEC and NOEC are concentrations that are based on TEQ, a recalculation of this TEQ to a  $\Sigma 7\text{PCB}$  concentration in prey is the next step. In the next step, this  $\Sigma 7\text{PCB}$  concentration in prey is recalculated to a  $\Sigma 7\text{PCB}$  concentration in organic carbon of the soil by means of the bioaccumulation metrics that were based on  $\Sigma 7\text{PCB}$  concentrations. In the final step, these concentrations are normalized to a Dutch standard soil containing 10% organic matter.

1. The No Observed Adverse Effect Level (NOAEL) of  $4.7 \cdot 10^{-7}$  mg TEQ<sub>2005</sub>/kg<sub>bw</sub>/d (obtained from the lowest test concentration of  $4.8 \cdot 10^{-9}$  mg TEQ<sub>2005</sub>/kg<sub>food</sub> and the food intake of 97 g<sub>diet</sub>/kg<sub>bw</sub>/d as given in the study) is taken as a starting point for the calculations (Bursian et al. 2013). With the body weight of 1,186 g reported in the study (Bursian et al. 2013) and the regression between Daily Energy Expenditure (DEE) and body weight (BW) for non-marine, non-desert eutherians (Crocker et al. 2002, DEFRA, 2007), a DEE of 1,026 kJ/d can be calculated. The NOAEL can then be recalculated to a NOEC of  $5.4 \cdot 10^{-10}$  mg TEQ<sub>2005</sub>/kJ (Verbruggen, 2013). With the energy content of 7331 kJ/kg<sub>fw</sub> for terrestrial vertebrates (Smit, 2005, EFSA, 2009), the NOEC can then be expressed as a concentration in terrestrial vertebrates of  $3.9 \cdot 10^{-6}$  mg TEQ<sub>2005</sub>/kg<sub>fw</sub>. If this is compared with the initial NOAEL, it means that the mink would eat an amount of birds and mammals equal to 12% of its body weight each day (i.e. calculated as a dose divided by the diet concentration), a value which is comparable to the 9.7% for the diet presented in the study.

An alternative to calculating the lipid normalized diet concentrations is presented (Verbruggen, 2013): the diet concentration can be directly normalized to the energy content of the diet if presented. In the study conducted by Bursian et al. (2013a,b) no energy content of the food was given. In the study by conducted Heaton et al. (1995) the energy content was given. The energy normalized diet concentrations and doses were already presented in Table 2.1. From the study, an average body weight of 1,154 g can be derived, comparable to that in the study conducted by Bursian et al. (2013a,b), from which the energy expenditure can be calculated as described above. The values for the EC50s based on the energy expenditure are 138% of the values based on the diet concentrations normalized for the reported energy content of the diet; for the EC10 this is 149%. For hexachlorobenzene, both ways of calculating resulted in very comparable values for the mink as well (100-109%) (Verbruggen, 2013). It can thus be concluded that the new method results in robust values, at least for the mink.

2. In this study, it is assumed that mink is one of the most sensitive species for PCBs. With the chronic reproduction toxicity studies for the mink, a very sensitive species and endpoint have been covered (see Section 2.2). However, because no species sensitivity distribution was constructed, an assessment factor should be applied to this lowest NOEC value to derive a value for the predicted no effect concentration (PNEC). The assessment factor to be applied to fully chronic studies is 10. The PNEC value obtained from the NOEC is  $3.9 \cdot 10^{-7}$  mg TEQ<sub>2005</sub>/kg<sub>fw</sub>.
3. The values that are derived for terrestrial vertebrates are based on fresh weight. Because most bioaccumulation parameters are based on lipid weight, this value is lipid normalized with the default lipid content of 10% for terrestrial vertebrates (Hendriks et al. 2001, Hendriks et al. 2005). This corresponds to a concentration of  $3.9 \cdot 10^{-5}$  and  $3.9 \cdot 10^{-6}$  mg TEQ<sub>2005</sub>/kg<sub>lw</sub> for the NOEC and PNEC, respectively. It should be noted that, in the studies from the Kalamazoo river, it is mentioned that a whole body of shrews was measured (Strause et al. 2008). This means that, given the lower experimental lipid content, varying from 2.2 to 4.5%, compared with what is assumed for whole mammals, i.e. 10%, the lipid normalized concentration based on the default lipid content might be underestimated in this case, leading to lower values. In the two other studies using shrews (Hendriks et

al. 1995, Van den Brink, 2000), it is explicitly reported that the accumulation in shrews was measured as a concentration in the liver and normalized to the lipid content of the liver.

4. The concentration  $\Sigma 7\text{PCB}$  is calculated from the correlation between  $\Sigma 7\text{PCB}$  and  $\text{TEQ}_{2005}$  from Section 2.5 for small birds and mammals. This equation is:

$$\log(\text{TEQ}_{2005} [\text{mg}/\text{kg}_{\text{lw}}]) = 0.725 * \log(\Sigma 7\text{PCB} [\text{mg}/\text{kg}_{\text{lw}}]) - 3.782$$

The resulting NOEC and PNEC in small terrestrial birds and mammals are 0.14 and 0.0058 mg  $\Sigma 7\text{PCB}/\text{kg}_{\text{lipid}}$ , respectively.

5. For the calculation of the standards, experimental BSAF were derived from three studies, representing six locations for which both the accumulation in earthworms and in shrews was available. The concentration in organic carbon of the soil was not calculated by means of a fixed biota-to-soil-accumulation-factor. Instead, the correlation between the concentrations in lipids of shrews and organic carbon in soil derived in Section 2.4.2 was used:

$$\log(\Sigma 7\text{PCB}_{\text{shrew}} [\text{mg}/\text{kg}_{\text{lw}}]) = 0.770 * \log(\Sigma 7\text{PCB}_{\text{organic carbon}} [\text{mg}/\text{kg}_{\text{oc}}]) + 0.158$$

With this correlation, the bioaccumulation from soil at the bottom of the food chain and the biomagnification up to higher trophic levels, such as the shrew, have been merged into one bioaccumulation step. This is comparable to the recalculation of higher trophic level fish to water by taking the bioaccumulation factor for these fish. For the aquatic compartment, this is also the preferred route (Moermond et al. 2013). The concentrations of 0.14 and 0.0058 mg  $\Sigma 7\text{PCB}/\text{kg}_{\text{lipid}}$  are recalculated to 0.048 and 0.00078 mg  $\Sigma 7\text{PCB}/\text{kg}_{\text{oc}}$  by means of the derived relationship. At these concentrations, this corresponds to BSAF values of 2.9 and 7.4 for  $\Sigma 7\text{PCB}$  at the NOEC and PNEC levels respectively.

6. The values are expressed in concentrations that have been normalized to the organic carbon content of the soil. To express these values of 0.048 and 0.00078 mg  $\Sigma 7\text{PCB}/\text{kg}_{\text{oc}}$  in a concentration in standard soil, they have to be multiplied by the fraction of organic carbon in Dutch standard soil (Van Vlaardingen et al. 2007). This fraction of organic carbon is 5.88% (10% organic matter), resulting in a NOEC and PNEC values of 2.8 and 0.046  $\mu\text{g} \Sigma 7\text{PCB}/\text{kg}_{\text{Dutch standard soil}}$ .

It should be noted that the European standard soil from the REACH guidance has a lower organic carbon content of 2% (ECHA, 2010). With this organic carbon content a NOEC of 0.96  $\mu\text{g} \Sigma 7\text{PCB}/\text{kg}_{\text{European standard soil}}$  is calculated.

The steps taken, with some remarks, are shown in Table 2.20. It should be noted that the values for the PNEC are outside the applicability of the linear regressions mentioned above.

Table 2.20 Calculation of the final soil concentrations corresponding to the no observed effect concentration (NOEC) for mink and the predicted no effect concentration (PNEC).

Matrix	NOEC	PNEC	Comment
TEQ <sub>2005</sub> in fresh weight of terrestrial vertebrates	3.9·10 <sup>-6</sup> mg TEQ <sub>2005</sub> /kg <sub>fw</sub>	3.9·10 <sup>-7</sup> mg TEQ <sub>2005</sub> /kg <sub>fw</sub>	Standard assessment factor of 10 applied
TEQ <sub>2005</sub> in lipids of terrestrial vertebrates	3.9·10 <sup>-5</sup> mg TEQ <sub>2005</sub> /kg <sub>lw</sub>	3.9·10 <sup>-6</sup> mg TEQ <sub>2005</sub> /kg <sub>lw</sub>	Normalized with default lipid content of 10%
Σ7PCB in lipids of terrestrial vertebrates	0.14 mg Σ7PCB/kg <sub>lw</sub>	5.8·10 <sup>-3</sup> mg Σ7PCB/kg <sub>lw</sub>	Regression between TEQ <sub>2005</sub> and Σ7PCB
Σ7PCB in organic carbon of soil	4.8·10 <sup>-2</sup> mg Σ7PCB/kg <sub>oc</sub>	7.8·10 <sup>-4</sup> mg Σ7PCB/kg <sub>oc</sub>	Regression between TEQ <sub>2005</sub> and Σ7PCB in shrews and organic carbon
Σ7PCB in standard soil	2.8·10 <sup>-3</sup> mg Σ7PCB/kg <sub>dw</sub>	4.6·10 <sup>-5</sup> mg Σ7PCB/kg <sub>dw</sub>	Normalized with default organic carbon content of 5.88%

## 2.7 Direct toxicology

As mentioned before, secondary poisoning in the food chain is the most critical parameter for exposure to PCB. In 2001, Verbruggen et al. evaluated the direct toxicity of PCBs. Secondary poisoning by contaminants was not included in the procedure for deriving risk limits. The SRC<sub>eco</sub> derived in 2001 was 3.4 mg/kg. This value is considerably higher (more than a factor of 1,200!) than the value derived for mink in this study and is therefore insufficiently protective for mink and other mustelids.

## 2.8 Ecotoxicological reference values

Because no species sensitivity distribution could be made, the NOEC for mink with an assessment factor of 10 (=PNEC) acts as MPC<sub>eco</sub> in this study. This value is on the conservative side because only one of the most sensitive species is considered. The NOEC without an assessment factor acts as SRC<sub>eco</sub> (which is equal to a maximum value for soil function 'industrial'). To derive an intermediate value to come to a maximum value for soil function 'residential', the geometric average of the SRC<sub>eco</sub> and MPC<sub>eco</sub> is taken as a substitute/surrogate. Table 2.21 summarizes the ecotoxicological reference values.

Table 2.21 Ecotoxicological reference value for PCB (sum7) based on studies for mink.

Mixture	Scenario	
	Residential (mg/kg d.w.)	Industrial (mg/kg d.w.)
PCB (sum 7)	3.6 · 10 <sup>-4</sup>	2.8 · 10 <sup>-3</sup>



## 3 Human risks

### 3.1 Introduction

Reference values form the basis for the maximum values. In general, when deriving reference values based on human risk limits, exposure to non-carcinogens is compared with the Maximum Permissible Risk (MPR) minus the background exposure (BE) from sources other than soil contamination. The MPR's were determined for most substances by Baars et al. (2001). In the same report the BE was determined. This also applies to PCBs (sum 7).

The background concentrations (BC) for the individual PCB congeners in soil are based on the 95<sup>th</sup> percentile value (P95) of measured background concentrations in soils as described in the report of Lame et al. (2008). In this report, generic background concentrations (AW2000) in soil were derived for different compounds in the Netherlands, based on measurements in relatively undisturbed areas.

### 3.2 Toxicity to humans

Table 3.1 presents an overview of the MPR, TCA (tolerable concentration in air), BE and BC for PCBs. For PCBs, the background exposure is equal to the MPR. In other words, the MPR is completely used by PCBs exposure from sources other than soil contamination. This means that the soil can no longer contribute to the exposure of humans and, as a result, all soils have to be clean. This is considered to be unrealistic. For the purpose of deriving soil quality values, the soil advisory group NOBO advised that the background exposure can be no more than 50% of the MPR. In this way, the remaining 50% of the MPR is available for exposure to soil contamination (Ministry of VROM, 2008). This advice has no risk based on underpinning, but was adopted in the policy framework. For the PCBs, this results in a MPR of **5x10<sup>-6</sup> mg/kg b.w./day**.

Table 3.1 MPR, TCA and BE as derived by Baars et al. (2001) and the background concentrations AW2000 derived by Lame et al. (2008).

Compound	MPR (mg/kg b.w./day)	TCA (µg/m <sup>3</sup> )	Background exposure (mg/kg b.w./day)	Background concentration in soil AW 2000 (mg/kg d.w.)
PCBs (sum 28, 52, 101, 118, 138, 153 en 180)	1x10 <sup>-5*</sup>	5x10 <sup>-4</sup>	1x10 <sup>-5</sup>	-
PCB 28	n.a.	n.a.	n.a.	0,0011
PCB 52				0,0018
PCB 101				0,0014
PCB 118				0,0041
PCB 138				0,0037
PCB 153				0,0035
PCB 180				0,0027

n.a.= not available, equal to sum value.

\* MPR is equal to the BE, therefore 50% of the MPR is used, which is equal to 5x10<sup>-6</sup> mg/kg b.w./day.



### 3.2.1 Human risk value for PCBs (sum 7)

To derive a human risk value for the sum of PCBs, the 7 indicator PCBs (PCB 28, 52, 101, 118, 138, 153 and 180) were used. Because, in practice, PCBs are found in a variety of different mixtures, two of these mixtures were investigated in this study. In practice, other mixtures are also possible.

Firstly, a mixture based on the Arochlor standard 1254 was used to derive a summed value. The 7 indicator PCBs make up for 31% of the Arochlor mixture. In Table 3.2 the contribution of the individual PCB congeners to the total mixture are presented.

Secondly, a mixture was based on the distribution of the individual PCB congeners in the generic background concentration in the Dutch soil (AW2000). This distribution was taken from Lame et al. (2008). See Table 3.2 for the contribution of the individual PCB congeners to the total.

*Table 3.2 Percentage of contribution of the individual PCB congeners to a mixture based on Arochlor 1254 and the presence of PCB in the background concentration AW2000.*

Indicator PCB	Mixture contribution (%)		
	Arochlor 1254 <sup>3</sup>		Background concentration AW2000
28	0.2	0.64	6.0
52	5.4	17.2	10.0
101	8.0	25.6	7.7
118	7.4	23.6	22.4
138	5.8	18.5	20.0
153	3.8	12.1	19.0
180	0.7	2.2	14.8
<b>Indicator PCB</b>	<b>-</b>	<b>100</b>	<b>100</b>
<b>Total PCB</b>	<b>31.3</b>	<b>-</b>	<b>-</b>

To compare both summed values, the percentage of the individual PCB congeners in the Arochlor mixture are normalized to make up for 100% of the mixture. Modelling of the corresponding risk limits for both mixtures was done with the human risk assessment model CSOIL 2000 (Brand et al. 2007).

The resulting human reference values for both mixtures are presented in Table 3.3 for the scenarios 'residential with garden', 'residential with vegetable garden', 'green with nature values' and 'other greens, infrastructure, buildings and industry'.

<sup>3</sup> <http://www.atsdr.cdc.gov/toxprofiles/tp17-c4.pdf>

*Table 3.3 Human reference value for PCBs (sum7) based on a mixture of Arochlor 1254 and a mixture based on generic appearance as background concentrations in soil.*

Mixture	Scenario			
	Residential with garden (mg/kg d.w.)	Residential with vegetable garden (mg/kg d.w.)	Green with nature values (mg/kg d.w.)	Other greens, infrastructure, buildings and industry (mg/kg d.w.)
PCBs (sum 7) based on Arochlor 1254	0.39*	0.08*	15.0*	15.0*
PCBs (sum 7) based on BC AW2000	0.36	0.07	15.0	15.0

\* Normalized to 100% contribution of the 7 indicator PCB.

Considering that there is only a small difference between a summed value based on the Arochlor mixture and a value based on the distribution of the individual PCB congeners in the generic background concentration in Dutch soils, a preference is given to the sum value based on the Arochlor mixture. This is largely because of practical reasons, such as familiarity by users and international acceptance.

### 3.2.2 Human risk value for individual PCB congeners

As different mixtures of PCBs might occur in practice, the human risk values for the individual PCB congeners are presented in Table 3.4 for the soil functions 'residential with garden', 'residential with vegetable garden', 'green with nature values' and 'industry'.

*Table 3.4 The human risk values 'residential with garden', 'residential with vegetable garden', 'green with nature values' and 'industry' for individual PCB congeners.*

PCB congeners	Human risk value (mg/kg d.w.)			
	Residential with garden	Residential with vegetable garden	Green with nature values	Other greens, infrastructure, buildings and industry
28	0.34	0.070	13	13
52	0.14	0.030	14	14
101	0.30	0.060	15	15
118	0.95	0.20	16	15
138	0.16	0.030	15	15
153	0.23	0.040	15	15
180	0.085	0.020	16	15

The human risk values are based on the MPR as described by Baars et al. (2001) and have been derived with the human risk assessment model CSOIL 2000 (Brand et al. (2007)).



## 4 Integration of ecotoxicological and human risk levels

### 4.1 Introduction

Since 2012, progress has been made to derive science-based Maximum Values for soil functions 'residential' and 'industrial'. The same procedure and principles are used as for the derivation of the reference values that were derived by Dirven-van Breemen et al. in 2007. Only for the ecotoxicological risks, a different approach was followed (as described in Chapter 2). For a more detailed description of the procedure used to derive reference values for the reuse of soil, we refer to the report of Dirven- van Breemen et al. (2007).

### 4.2 Reference values leading to Maximal Values

The reference values based on human and ecotoxicological risk assessment are summed in Table 4.1.

*Table 4.1 Background values and reference values for 'residential with garden', 'residential with vegetable garden', 'green with nature values' and 'other greens, infrastructure, buildings and industry' for PCBs (sum7), based on mixtures of Arochlor 1254 and a mixture based on distribution as background concentrations in the Netherlands (AW2000).*

Mixture	Back-ground value (mg/kg d.w.)	Reference value				Reference value based on
		Residential with garden (mg/kg d.w.)	Residential with vegetable garden (mg/kg d.w.)	Green with nature values (mg/kg d.w.)	Other greens, infrastructure, buildings and industry (mg/kg d.w.)	
PCBs (sum 7) based on Arochlor 1254	0.02*	$3.6 \times 10^{-4}$	$3.6 \times 10^{-4}$ *	$3.6 \times 10^{-4}$ *	$2.8 \times 10^{-3}$	Ecotoxicology
PCBs (sum 7) based on Arochlor 1254	0.02*	0.39	0.08	15.0	15.0	Human
PCBs (sum 7) based on BC AW2000	0.02*	0.36	0.07	15.0	15.0	Human

\* according to the procedure described in Dirven-van Breemen et al. (2007), the reference values for 'residential with vegetable garden' and 'green with nature values' are equal to the value for 'residential with garden'.

Based on the reference values presented in Table 4.1, the proposed Maximal Values for 'residential' and 'industrial' can be determined by selecting the lowest value for either human exposure or ecotoxicology. It turns out that the reference values for PCBs (sum 7) based on ecotoxicology lie well below the values based on human exposure and the generic background concentration of 0.02 mg/kg (see Table 4.2), being the 95-percentile of measured concentrations in the Netherlands.

*Table 4.2 Background values and proposed Maximal Values based on the reference values for 'residential' and 'industry' for PCBs (sum7), based on mixtures of Arochlor1254 and a mixture based on distribution as background concentrations in the Netherlands (AW2000).*

Mixture	Background value (mg/kg d.w.)	Proposed Maximal Value		Reference value based on
		Residential (mg/kg d.w.)	Industrial (mg/kg d.w.)	
PCBs (sum 7) based on Arochlor 1254	0.02*	$3.6 \times 10^{-4}$	$2.8 \times 10^{-3}$	Ecotoxicology
PCBs (sum 7) based on Arochlor 1254	0.02*	0.39	15.0	Human
PCBs (sum 7) based on BC AW2000	0.02*	0.36	15.0	Human

\* Generic background concentration based on the 95<sup>th</sup> percentile of measurements in the research of Lame et al. (2008).

In Table 4.3, the proposed reference values for the individual PCB congeners are presented, as well as the background values. These values are all based on human risk assessment only. In Appendix 2, a complete overview of the reference values for individual PCBs is given, including the scenarios 'nature', 'places where children play', 'agriculture' and 'greens with nature values'.

*Table 4.3 The background value in soil and the proposed reference values for 'residential' and 'industry' for individual PCB congeners based on human risk assessment.*

PCB congeners	Reference values individual congeners (mg/kg d.w.)		
	Background value	Residential with garden	Other greens, infrastructure, buildings and industry
28	0.0011*	0.3	13
52	0.0018*	0.1	14
101	0.0014*	0.3	15
118	0.0041*	1.0	15
138	0.0037*	0.2	15
153	0.0035*	0.2	15
180	0.0027*	0.1	15

\* the 95<sup>th</sup> percentile of measurements of individual PCB congeners in the research of Lame et al. (2008).

### 4.3 Intervention values for individual PCB congeners and PCBs (sum 7)

In 2001, RIVM proposed new intervention values for individual PCBs (Lijzen et al. 2001). These PCBs were based on direct toxicity and human exposure. There was no intervention value proposed for the sum of PCBs. It was, however, advised that the toxic unit approach (TU) be applied because the individual PCBs have an added effect (concentration addition). When applying the TU, individual risk indices may not exceed 1 when added together. The proposed values from 2001 were never formalized and the current intervention value for PCBs (sum 7) is 1 mg / kg d.w. and dates back to the '90s (Van den Berg, 1991). Table 4.4

provides an overview of the proposed intervention values of 2001 for soil and groundwater.

*Table 4.4 Overview of the intervention values as proposed by Lijzen et al. in 2001 for individual PCB congeners in soil and groundwater.*

<b>PCB congener</b>	<b>Proposed intervention value soil (mg/kg d.w.)</b>	<b>Proposed intervention value groundwater (µg/l)</b>
28	0.69	0.29
52	0.28	0.097
101	0.61	0.031
118	1.9	0.015
138	0.32	0.011
153	0.46	0.011
180	0.17	0.003

An intervention value is derived from the risks for humans and the risk for ecotoxicology, with the lowest value being the determining value. For PCBs it is known that secondary poisoning in the food chain plays an important role in ecotoxicology (see also Chapter 2). Until 2008, however, it was not common to include secondary poisoning in the risk assessment and it still plays no part in deriving intervention values for soil. In 2001, therefore, Verbruggen et al. determined a sum value for the ecotoxicology on soil of 3.4 mg/kg d.w. based on direct toxicology only. This value is above the current intervention value of 1 mg/kg d.w. For PCBs, not taking into account secondary poisoning leads to an underestimation of the risks. If secondary poisoning were taken into account, the value of 3.4 mg/kg d.w. would be much lower, as demonstrated in Chapter 2.

Although no formal proposal for an Intervention value for sum PCBs will be given in this report, as an indication, the contribution of the individual PCB congeners in the Arochlor mixture (Chapter 3) could be used to derive a value for sum PCBs based on the proposed Intervention values from 2001. This value would be 0.78 mg/kg d.w. for soil.

For detailed information on the derivation of the intervention values, see the report of Lijzen et al. (2001).



## 5 Conclusion and discussion

### 5.1 General conclusions

Secondary poisoning in the food chain is the most critical parameter in contrast with the direct toxicity of PCBs in the ecosystem and the human risks involved. Because of secondary poisoning, the ecotoxicological risk is the determining factor compared with the human risks in deriving Maximum Values. In previous evaluations, the secondary poisoning by PCBs was not included in the derivation of the risk limits. The risk limits were therefore based on direct toxicity only. As a result, the risk limits of PCBs without inclusion of secondary poisoning are an underestimation of ecological risks.

Based on the protection of ecology, the Maximal Values that are proposed in this report take into account secondary poisoning involving PCBs in the food chain.

Mink and otter-like species are classified as highly sensitive target species for PCBs. It is assumed that sufficient protection for the mink gives ample protection for the remaining ecosystem and humans. Usually, an SSD is prepared to derive an HC5 ( $MPC_{eco}$ ), HC20 and HC50 ( $SRC_{eco}$ ), based on all available species. However, given the large amount of literature, this would require a significant amount of time and costs. A complete review of risk limits for PCBs should, therefore, be undertaken within a European context. From a pragmatic point of view, therefore, in this report it was decided that the mink would be used as a starting point for deriving the reference values. The proposed reference values are presented in Table 5.1. The values given in bold represent the Proposed Maximal Values for the land-use categories 'residential' and 'industrial'.

*Table 5.1 Background value and the reference values for soil functions 'residential with garden', 'residential with vegetable garden', 'green with nature values' and 'industry' for PCBs (sum7) based on mixtures of Arochlor1254 and a mixture based on distribution as background concentrations in the Netherlands (AW2000). The Proposed Maximal Values for 'residential' and 'industrial' are given in bold.*

Mixture	Back-ground value (mg/kg d.w.)	Reference value				Reference value based on
		Residential with garden (mg/kg d.w.)	Residential with vegetable garden (mg/kg d.w.)	Green with nature values (mg/kg d.w.)	Other greens, infrastructure, buildings and industry (mg/kg d.w.)	
PCBs (sum 7) based on Arochlor 1254	0.02*	<b><math>3.6 \times 10^{-4}</math></b>	$3.6 \times 10^{-4}$ **	$3.6 \times 10^{-4}$ **	<b><math>2.8 \times 10^{-3}</math></b>	Ecotoxicology
PCBs (sum 7) based on Arochlor 1254	0.02*	0.39	0.08	15.0	15.0	Human
PCBs (sum 7) based on BC AW2000	0.02*	0.36	0.07	15.0	15.0	Human

\* Generic background concentration based on the 95<sup>th</sup> percentile of measurements, research of Lame et al. (2008).

\*\* according to the procedure described in Dirven-van Breemen et al. (2007) reference values for the soil functions 'residential with vegetable garden' and 'green with nature values' are equal to the value 'residential with garden'.



## 5.2 Consequences for policy

### 5.2.1 *Maximum value residential*

The current Maximum Value for soil function 'residential' based on a policy decision is 0.04 mg/kg d.w. in standard soil (10% organic matter), being twice the derived background value. The derived risk limit for secondary poisoning ( $3.6 \times 10^{-4}$  mg/kg d.w.) in this report is well below the level of the current Maximum Values. Because lowering the Maximum Value for soil function 'residential' has a great impact on the reuse of soil and sediment, a policy decision was made to maintain the level of 2 times the BC (0.04 mg/kg d.w.).

Based on the methodology of using the 95-percentile of measured concentrations (background concentration) as the lower boundary for quality standards for reuse of soils (Dirven-van Breemen et al. 2007), this would lead to the value of 0.02 mg/kg for sum PCBs for the Maximal Value for soil function 'residential'. The reason for this is that the derived risk limit is lower than these generic background concentrations. Setting standards below the generic background concentrations is generally not done. Still, it should be noted that at this level sensitive species like the mink and other mustelids will be insufficiently protected for the risks of secondary poisoning.

### 5.2.2 *Maximum value for soil function 'industrial'*

For soil function 'industrial', as well, the proposed reference value ( $2.8 \times 10^{-3}$  mg/kg d.w.) is considerably lower than the current maximum value of 0.5 mg/kg d.w. As with 'residential', the methodology of using the 95-percentile of measured concentrations as the lower boundary for quality standards for reuse of soil (Dirven-van Breemen et al. 2007), leads to Maximal Value industrial of also 0,02 mg/kg.

The study shows that, given the risks of secondary poisoning in sensitive species, such as mink and other mustelids, levels above the background value in the top soil are undesirable. The derived PNEC for mink is a reason to lower the current maximum value for soil function 'industrial', especially when it comes to green areas. Therefore, consideration should be given to lowering the maximum value for 'industrial' (for the part of bare-green areas and only for the top soil) to the same level as for soil function 'residential'. Environmental gains are achieved by the removal of soil with higher levels of PCBs. An alternative would be to conduct a literature review on bioaccumulation studies for a less sensitive target species to derive a risk value or an organism that feeds solely from 'soil organisms', such as small terrestrial birds and mammals. At this moment, it is unclear whether these organisms are protected by the Maximum Value for the soil function 'industrial'.

## References

- Baars A.J., Theelen R.M.C., Janssen P.J.C.M., Hesse J.M., Van Apeldoorn M.E., Meijerink M.C.M., Verdam L., Zeilmaker M.J., (2001). Re-evaluation of human-toxicological maximum permissible risk levels. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 711701 025
- Babut M., Miege C., Villeneuve B., Abarnou A., Duchemin J., Marchand P., Narbonne J.F., (2009). Correlations between dioxin-like and indicators for PCBs: Potential consequences for environmental studies involving fish or sediment. *Environmental Pollution* 157 (12): 3451-3456.
- Basu N., Scheuhammer A.M., Bursian S.J., Elliott J., Rouvinen-Watt K., Chan H.M., (2007). Mink as a sentinel species in environmental health. *Environmental Research* 103 (1): 130-144.
- Brand E., Otte P.F., Lijzen J.P.A., (2007). CSOIL2000: an exposure model for human risk assessment of soil contamination. A model description. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 711701 054.
- Blankenship A.L., Zwiernik M.J., Coady K.K., Kay D.P., Newsted J.L., Strause K., Park C., Bradley P.W., Neigh A.M., Millsap S.D., Jones P.D., Giesy J.P., (2005). Differential accumulation of polychlorinated biphenyl congeners in the terrestrial food web of the Kalamazoo River superfund site, Michigan. *Environmental Science and Technology* 39 (16): 5954-5963.
- Bleavins M.R., Breslin W.J., Aulerich R.J., Ringer R.K., (1982). Excretion and placental and mammary transfer of hexachlorobenzene in the European ferret (*Mustela putorius furo*). *Journal of Toxicology and Environmental Health* 10 (6): 929-940.
- Bleavins M.R., Breslin W.J., Aulerich R.J., Ringer R.K., (1984). Placental and mammary transfer of a polychlorinated biphenyl mixture (Aroclor 1254) in the European ferret (*Mustela putorius furo*). *Environmental Toxicology and Chemistry* 3 (4): 637-644.
- Bursian S.J., Beckett K.J., Yamini B., Martin P.A., Kannan K., Shields K.L., Mohr F.C., (2006a). Assessment of the effects in mink caused by the consumption of carp collected from the Saginaw River, Michigan, USA. *Archives of Environmental Contamination and Toxicology* 50 (4): 614-623.
- Bursian S.J., Sharma C., Aulerich R.J., Yamini B., Mitchell R.R., Orazio C.E., Moore D.R.J., Svirsky S., Tillitt D.E., (2006b). Dietary exposure of mink (*Mustela vison*) to fish from the Housatonic River, Berkshire County, Massachusetts, USA: Effects on reproduction, kit growth and survival. *Environmental Toxicology and Chemistry* 25 (6): 1533-1540.
- Bursian S.J., Sharma C., Aulerich R.J., Yamini B., Mitchell R.R., Beckett K.J., Orazio C.E., Moore D., Svirsky S., Tillitt D.E., (2006c). Dietary exposure of mink (*Mustela vison*) to fish from the Housatonic River, Berkshire County,

- Massachusetts, USA: Effects on organ weights and histology and hepatic concentrations of polychlorinated biphenyls and 2,3,7,8- tetrachlorodibenzo-p-dioxin toxic equivalence. *Environmental Toxicology and Chemistry* 25 (6): 1541-1550.
- Bursian S.J., Kern J., Remington R.E., Link J.E., Fitzgerald S.D., (2013a). Dietary exposure of mink (*Mustela vison*) to fish from the upper Hudson river, New York, USA: Effects on reproduction and offspring growth and mortality. *Environmental Toxicology and Chemistry* 32 (4): 780-793.
- Bursian S.J., Kern J., Remington R.E., Link J.E., Fitzgerald S.D., (2013b). Dietary exposure of mink (*Mustela vison*) to fish from the upper Hudson River, New York, USA: Effects on organ mass and pathology. *Environmental Toxicology and Chemistry* 32 (4): 794-801.
- Çakroğullar G.C., Uçar Y., Oymael B., Bozkurt E.N., Kiliç D., (2010). Pcd/f, dl-PCB and indicator PCBs in whiting, horse mackerel and anchovy in the Black Sea in Turkey. Karadeniz'den (Türkiye) Yakalanan Mezgit, Istavrit ve Hamsi Balıklarında PCDD/F, dl-PCB ve İndikatör PCB'ler 10 (3): 357-362.
- Crocker D., Hart A., Gurney J., McCoy C., (2002). Project PN0908: Methods For Estimating Daily Food Intake Of Wild Birds And Mammals - Final Report. York, United Kingdom: Central Science Laboratory. Laboratory CS.22 pp.
- Dauwe T., Jaspers V.L.B., Covaci A., Eens M., (2006). Accumulation of organochlorines and brominated flame retardants in the eggs and nestlings of great tits, *Parus major*. *Environmental Science and Technology* 40 (17): 5297-5303.
- DEFRA, (2007). Improved estimates of food and water intake for risk assessment. London, UK: Department for Environment, Food and Rural Affairs (DEFRA), Science Directorate, Management Support and Finance Team. 23 pp.
- Dirven- Van Breemen E.M., Lijzen J.P.A., Otte P.F., Van Vlaardingen P.L.A., Spijker J., Verbruggen E.M.J., Swartjes F.A., Groenenberg J.E., Rutgers M., (2007). National land-use-specific reference values: a basis for maximum values in Dutch soil policy. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 711701 053 (in Dutch).
- ECHA, (2010). Guidance on information requirements and chemical safety assessment. Chapter R.16: Environmental Exposure Estimation. Helsinki, Finland: European Chemicals Agency. Report no. ECHA-10-G-06-EN. Agency EC.146 pp.
- Echols K.R., Tillitt D.E., Nichols J.W., Secord A.L., McCarty J.P., (2004). Accumulation of PCB congeners in nestling tree swallows (*Tachycineta bicolor*) on the Hudson River, New York. *Environmental Science and Technology* 38 (23): 6240-6246.
- EFSA, (2009). Guidance Document on Risk Assessment for Birds and Mammals. Parma, Italy: European Food Safety Authority. Authority EFS.358 pp.

- Gómez-Ramírez P., Martínez-López E., García-Fernández A.J., Zweers A.J., van den Brink N.W., (2012). Organohalogen exposure in a Eurasian Eagle owl (*Bubo bubo*) population from South-eastern Spain: Temporal-spatial trends and risk assessment. *Chemosphere* 88 (8): 903-911.
- Heaton S.N., Bursian S.J., Giesy J.P., Tillitt D.E., Render J.A., Jones P.D., Verbrugge D.A., Kubiak T.J., Aulerich R.J., (1995). Dietary exposure of mink to carp from Saginaw Bay, Michigan. 1. Effects on reproduction and survival, and the potential risks to wild mink populations. *Archives of Environmental Contamination and Toxicology* 28 (3): 334-343.
- Hendriks A.J., Ma W.C., Brouns J.J., De Ruiter-Dijkman E.M., Gast R., (1995). Modelling and monitoring organochlorine and heavy metal accumulation in soils, earthworms and shrews in the Rhine-delta floodplains. *Archives of Environmental Contamination and Toxicology* 29 (1): 115-127.
- Hendriks A.J., Traas T.P., Huijbregts M.A.J., (2005). Critical body residues linked to octanol - Water partitioning, organism composition, and LC50 QSARs: Meta-analysis and model. *Environ Sci Technol* 39 (9): 3226-3236.
- Hendriks A.J., Van Der Linde A., Cornelissen G., Sijm D.T.H.M., (2001). The power of size. 1. Rate constants and equilibrium ratios for the accumulation of organic substances related to the octanol-water partition ratio and species weight. *Environ Toxicol Chem* 20 (7): 1399-1420.
- Jager T., (1998). Mechanistic approach for estimating bioconcentration of organic chemicals in earthworms (oligochaeta). *Environmental Toxicology and Chemistry* 17 (10): 2080-2090.
- Kay DP, Blankenship AL, Coady KK, Neigh AM, Zwiernik MJ, Millsap SD, Strause K, Park C, Bradley P, Newsted JL, Jones PD, Giesy JP. 2005. Differential accumulation of polychlorinated biphenyl congeners in the aquatic food web at the Kalamazoo River superfund site, Michigan. *Environmental Science and Technology* 39 (16): 5964-5974.
- Krauss M., Wilcke W., Zech W., (2000). Availability of polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) to earthworms in urban soils. *Environmental Science and Technology* 34 (20): 4335-4340.
- Lamé F.P.J., Brus D.J., Nieuwenhuis R.H., (2008). Background values 2000. Appendix to report AW2000: Datasheets for analysed compounds. TNO, Utrecht, the Netherlands (in Dutch).
- Leonards P.E.G., Broekhuizen S., De Voogt P., Van Straalen N.M., Brinkman U.A.T., Cofino W.P., Van Hattum B., (1998). Studies of bioaccumulation and biotransformation of PCBs in mustelids based on concentration and congener patterns in predators and prey. *Archives of Environmental Contamination and Toxicology* 35 (4): 654-665.
- Leonards P.E.G., Van Hattum B., Madsen A.B., Elmeros M., Cofino W.P., Conroy J.W.H., Gutleb A.C., Mason C.F., Murk A.J., Olsson M., Smit M.D., (1996). Toxic PCBs in European otter populations in relation to biological factors and health status. Development of Otter-Based Quality Objectives for PCBs: 47-62.

Leonards P.E.G., Zierikzee Y., Brinkman U.A.T., Cofino W.P., Van Straalen N.M., Van Hattum B., (1997). The selective dietary accumulation of planar polychlorinated biphenyls in the otter (*Lutra lutra*). *Environmental Toxicology and Chemistry* 16 (9): 1807-1815.

Lijzen J.P.A., Baars A.J., Otte P.F., Rikken M.G.J., Swartjes F.A., Verbruggen E.M.J., Van Wezel A.P., (2001). Technical evaluation of the intervention values for soil/sediment and groundwater. Human and ecotoxicological risk assessment and derivation of risk limits in soil, sediment and groundwater. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 711701 023

Lijzen J.P.A. & Verbruggen E.M.J., (2011). NOBOWA memorandum 'Quickscan Maximum value for 'residential with garden' for PCB-contaminations'. National Institute for Public Health and the Environment (RIVM), Bilthoven, Netherlands. NOBOWA no. 2011-37 (in Dutch)

Metcalfe T.L., Metcalfe C.D., (1997). The trophodynamics of PCBs, including mono and non-ortho congeners, in the food web of North-Central Lake Ontario. *Science of the Total Environment* 201 (3): 245-272.

Ministry of VROM (2008). NOBO: Standards and soil quality assessment, underpinning and policy choices for the soil standards in 2005, 2006 and 2007. Formal Ministry of VROM, The Hague, the Netherlands.  
[http://www.rwsleefomgeving.nl/publish/pages/91751/rapportage\\_nobo\\_nor\\_mstellling\\_en\\_bodemkwaliteitsbeoordeling\\_24\\_263999.pdf](http://www.rwsleefomgeving.nl/publish/pages/91751/rapportage_nobo_nor_mstellling_en_bodemkwaliteitsbeoordeling_24_263999.pdf) (retrieved on 03-07-2014)

Murk A.J., Leonards P.E.G., Van Hattum B., Luit R., Van Der Weiden M.E.J., Smit M., (1998). Application of biomarkers for the exposure and effect of polyhalogenated aromatic hydrocarbons in naturally exposed European otters (*Lutra lutra*). *Environmental Toxicology and Pharmacology* 6 (2): 91-102.

Persson S., Rotander A., van Bavel B., Brunström B., Bäcklin B.M., Magnusson U., (2013). Influence of age, season, body condition and geographical area on concentrations of chlorinated and brominated contaminants in wild mink (*Neovison vison*) in Sweden. *Chemosphere* 90 (5): 1664-1674.

Schmid P., Kohler M., Gujer E., Zennegg M., Lanfranchi M., (2007). Persistent organic pollutants, brominated flame retardants and synthetic musks in fish from remote Alpine lakes in Switzerland. *Chemosphere* 67 (9): S16-S21.

Smit C.E., (2005). Energy and moisture content and assimilation efficiency of bird and mammal food. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 601516013/2005. 57-71 pp.

Smit M.D., Leonards P.E.G., Madsen A., Hatti B.G.M., Murk A.J., Jongh A.W.J.J., (1996a). Bioaccumulation of PCBs in Danish otter habitats. In: *Development of Otter-based Quality Objectives for PCBs*: 7-31.

Smit M.D., Van Hattum B.G.M., Leonards P.E.G., Murk A.J., De Jongh A.W.J.J., (1996b). Modelling Biomagnification of PCBs in Otters. In: *Development of Otter-Based Quality Objectives for PCBs*: 33-45.

- Smit M.D., Van Hattum B.G.M, Leonards P.E.G., Murk A.J., De Jongh, A.W.J.J., Van der Weiden M.E.J., Klein M.H.J., Hendriks A.J., Luttik R., (1996c). Derivation of 'safe' and 'critical' levels of PCBs in otter (*Lutra lutra*), fish and sediment based on observed adverse physiological effects in individual otters; Implications for management of otter areas. In: Development of Otter-Based Quality Objectives for PCBs: 83-102
- Strause K.D., Zwiernik M.J., Newsted J.L., Neigh A.M., Millsap S.D., Park C.S., Moseley P.P., Kay D.P., Bradley P.W., Jones P.D., Blankenship A.L., Sikarskie J.G., Giesy J.P., (2008). Risk assessment methodologies for exposure of great Horned Owls (*Bubo virginianus*) to PCBs on the Kalamazoo River, Michigan. *Integrated Environmental Assessment and Management* 4 (1): 24-40.
- Tillitt D.E., Gale R.W., Meadows J.C., Zajicek J.L., Peterman P.H., Heaton S.N., Jones P.D., Bursian S.J., Kubiak T.J., Giesy J.P., Aulerich R.J., (1996). Dietary exposure of mink to carp from Saginaw Bay. 3. Characterization of dietary exposure to planar halogenated hydrocarbons, dioxin equivalents, and biomagnification. *Environmental Science and Technology* 30 (1): 283-291.
- Traas T.P., Luttik R., Klepper O., Beurskens J.E.M., Smit M.D., Leonards P.E.G., Van Hattum A.G.M., Aldenberg T., (2001). Congener-specific model for polychlorinated biphenyl effects on otter (*Lutra lutra*) and associated sediment quality criteria. *Environmental Toxicology and Chemistry* 20 (1): 205-212.
- Van Den Berg M., Birnbaum L., Bosveld A.T.C., Brunström B., Cook P., Feeley M., Giesy J.P., Hanberg A., Hasegawa R., Kennedy S.W., Kubiak T., Larsen J.C., Van Leeuwen F.X.R., Liem A.K.D., Nolt C., Peterson R.E., Poellinger L., Safe S., Schrenk D., Tillitt D., Tysklind M., Younes M., Wærn F., Zacharewski T., (1998). Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environmental Health Perspectives* 106 (12): 775-792.
- Van den Berg M., Birnbaum L.S., Denison M., De Vito M., Farland W., Feeley M., Fiedler H., Hakansson H., Hanberg A., Haws L., Rose M., Safe S., Schrenk D., Tohyama C., Tritscher A., Tuomisto J., Tysklind M., Walker N., Peterson R.E., (2006). The 2005 World Health Organization re-evaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. *Toxicological Sciences* 93 (2): 223-241.
- Van den Berg R. & Roels J.M., (1991). Exposure of man to contaminated soil. A qualitative and quantitative analysis, resulting in proposals for human toxicologically based C-values. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 725201 006 (in Dutch).
- Van den Brink N.W.G., De jonge J., Bosveld A.T.C., (2000). Verontreinigen in het rivierengebied: een bedreiging voor de steenuil? Wageningen, the Netherlands: Alterra, Research Instituut voor de Groene Ruimte.
- Van Vlaardingen P.L.A., Verbruggen E.M.J., (2007). Guidance for the derivation of environmental risk limits within the framework 'International and national environmental quality standards for substances in the Netherlands' (INS). National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 601782 001.

- Verbruggen E.M.J., (in prep). New method for the derivation of risk limits for secondary poisoning based on metabolic rate and the energy content of food. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands.
- Verbruggen E.M.J., Posthumus R. Van Wezel A.P. (2001). Ecotoxicological Serious Risk Concentrations for soil, sediment and (ground)water: updated proposals for first series of compounds. National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands. RIVM report no. 711701 020
- Vermeulen F., Covaci A., D'Havé H., Van den Brink N.W., Blust R., De Coen W., Bervoets L., (2010). Accumulation of background levels of persistent organochlorine and organobromine pollutants through the soil-earthworm-hedgehog food chain. *Environment International* 36 (7): 721-727.
- Voorspoels S., Covaci A., Jaspers V.L.B., Neels H., Schepens P., (2007). Biomagnification of PBDEs in three small terrestrial food chains. *Environmental Science and Technology* 41 (2): 411-416.
- Voorspoels S., Covaci A., Lepom P., Escutenaire S., Schepens P., (2006a). Remarkable findings concerning PBDEs in the terrestrial top-predator red fox (*Vulpes vulpes*). *Environmental Science and Technology* 40 (9): 2937-2943.
- Voorspoels S., Covaci A., Lepom P., Jaspers V.L.B., Schepens P., (2006b). Levels and distribution of polybrominated diphenyl ethers in various tissues of birds of prey. *Environmental Pollution* 144 (1): 218-227.
- Zhang W., Xu M., Wang X., Huang Q., Nie J., Li Z., Li S., Hwang S.W., Lee K.B., (2012). Effects of organic amendments on soil carbon sequestration in paddy fields of subtropical China. *Journal of Soils and Sediments* 12 (4): 457-470.
- Zhao X., Zheng M., Zhang B., Zhang Q., Liu W., (2006). Evidence for the transfer of polychlorinated biphenyls, polychlorinated dibenzo-p-dioxins, and polychlorinated dibenzofurans from soil into biota. *Science of the Total Environment* 368 (2-3): 744-752.

## Appendix 1: Literature search

A literature search on the toxicity of PCBs yielded a large number of 3,547 hits in the Scopus literature database (www.scopus.com). The following search phrase was used:

```
TITLE-ABS-KEY(PCB) AND TITLE-ABS-KEY(ec50* OR ec20* OR ec10* OR lc50* OR lc20* OR lc10* OR noec* OR loec* OR noael* OR matc OR tlm OR chv OR ecx OR bioassay* OR mortalit* OR phytotox* OR reproduct* OR lethal* OR growth* OR teratogen* OR bioconcentrat* OR bioaccumulat* OR food-web OR trophic* OR biomagnificat* OR bcf* OR baf* OR fwmf* OR tmf* OR bmf* OR bsaf* OR freundlich OR koc* OR kd* OR kp* OR kf* OR partition-coefficient*) AND ( LIMIT-TO(SUBJAREA,"ENVI" ) OR LIMIT-TO(SUBJAREA,"MULT" ) )
```

```
TITLE-ABS-KEY(PCB) AND TITLE-ABS-KEY(mink) AND TITLE-ABS-KEY(ec50* OR ec20* OR ec10* OR lc50* OR lc20* OR lc10* OR noec* OR loec* OR noael* OR matc OR tlm OR chv OR ecx OR bioassay* OR mortalit* OR phytotox* OR reproduct* OR lethal* OR growth* OR teratogen* OR bioconcentrat* OR bioaccumulat* OR food-web OR trophic* OR biomagnificat* OR bcf* OR baf* OR fwmf* OR tmf* OR bmf* OR bsaf* OR freundlich OR koc* OR kd* OR kp* OR kf* OR partition-coefficient*) AND ( LIMIT-TO(SUBJAREA,"ENVI" ) OR LIMIT-TO(SUBJAREA,"MULT" ) )
```

When limited to mink by adding AND TITLE-ABS-KEY(mink), only 60 references were found instead of 3,547.





## Appendix 2: Table with reference values for individual PCB congeners for different scenarios

The tables represent the calculated reference values for the individual PCBs for different soil uses.

Table A2.1 PCB 28

<b>Proposed reference values based on policy decisions in the soil advisory group NOBO</b>							
PCB28							
Criteria	1	2	3	4	5	6	7
	Residential with garden	Playgrounds	Residential with vegetable garden	Agriculture (without residence)	Nature	Green with nature values	Other green, buildings infrastructure and industry
Soil use →							
↓ Soil quality criteria	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
Human risks	0.34	3.5	0.070	0.34	13.00	13.00	13.00
Agricultural risks				0.0011			
Ecological risks							
generic: organisms/processes/plants	n.a.	n.a.	n.a.	n.a.	0.020	n.a.	n.a.
specific: incl. secondary poisoning of key-target species	0.00036	0.00036	0.00036	0.00036	0.020	0.00036	0.0028
Current target value:	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#
Current intervention value:	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*
P95 of AW2000	0.0011	0.0011	0.0011	0.0011	0.0011	0.0011	0.0011
Proposed reference value:	0.0011	0.0011	0.0011	0.0011	0.0011	0.0011	0.0028
* Intervention value for PCB (sum 7)							
# target value for PCB (sum 7)							
Proposed geometric average of the MPCeco and SRCeco in this report (sum 7)							
Proposed SRCeco in this report (sum 7)							
P95 of AW 2000 report Lame et al. (2008)							

Table A2.2 PCB 52

<b>Proposed reference values based on policy decisions in the soil advisory group NOBO</b>							
<b>PCB52</b>							
Criteria	1	2	3	4	5	6	7
	Residential with garden	Playgrounds	Residential with vegetable garden	Agriculture (without residence)	Nature	Green with nature values	Other green, buildings infrastructure and industry
Soil use →							
↓ Soil quality criteria	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
<u>Human risks</u>	0.14	3.57	0.03	0.14	14.10	14.10	13.52
<u>Agricultural risks</u>				0.0018			
<u>Ecological risks</u>							
generic: organisms/processes/plants	n.a.	n.a.	n.a.	n.a.	0.020	n.a.	n.a.
specific: incl. secondary poisoning of key-target species	0.00036	0.00036	0.00036	0.00036	0.020	0.00036	0.0028
Current target value:	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#
Current intervention value:	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*
P95 of AW2000	0.0018	0.0018	0.0018	0.0018	0.0018	0.0018	0.0018
Proposed reference value:	0.0018	0.0018	0.0018	0.0018	0.0018	0.0018	0.0028
* Intervention value for PCB (sum 7)							
# target value for PCB (sum 7)							
<a href="#">Proposed geometric average of the MPCeco and SRCeco in this report (sum 7)</a>							
<a href="#">Proposed SRCeco in this report (sum 7)</a>							
<a href="#">P95 of AW 2000 report Lame et al. (2008)</a>							

Table A2.3 PCB 101

<b>Proposed reference values based on policy decisions in the soil advisory group NOBO</b>							
<b>PCB101</b>							
<b>Criteria</b>	<b>1</b>	<b>2</b>	<b>3</b>	<b>4</b>	<b>5</b>	<b>6</b>	<b>7</b>
	<b>Residential with garden</b>	<b>Playgrounds</b>	<b>Residential with vegetable garden</b>	<b>Agriculture (without residence)</b>	<b>Nature</b>	<b>Green with nature values</b>	<b>Other green, buildings infrastructure and industry</b>
Soil use →							
↓ Soil quality criteria	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
<u>Human risks</u>	0.30	3.7	0.060	0.30	15.47	15.47	15.02
<u>Agricultural risks</u>				0.001			
<u>Ecological risks</u>							
generic: organisms/processes/plants	n.a.	n.a.	n.a.	n.a.	0.020	n.a.	n.a.
specific: incl. secondary poisoning of key-target species	0.00036	0.00036	0.00036	0.00036	0.020	0.00036	0.0028
Current target value:	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#
Current intervention value:	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*
P95 of AW2000	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014
Proposed reference value:	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0028
* Intervention value for PCB (sum 7)							
# target value for PCB (sum 7)							
<a href="#">Proposed geometric average of the MPCeco and SRCeco in this report (sum 7)</a>							
<a href="#">Proposed SRCeco in this report (sum 7)</a>							
<a href="#">P95 of AW 2000 report Lame et al. (2008)</a>							

Table A2.4 PCB 118

<b>Proposed reference values based on policy decisions in the soil advisory group NOBO</b>							
<b>PCB118</b>							
<b>Criteria</b>	<b>1</b>	<b>2</b>	<b>3</b>	<b>4</b>	<b>5</b>	<b>6</b>	<b>7</b>
	<b>Residential with garden</b>	<b>Playgrounds</b>	<b>Residential with vegetable garden</b>	<b>Agriculture (without residence)</b>	<b>Nature</b>	<b>Green with nature values</b>	<b>Other green, buildings infrastructure and industry</b>
Soil use →							
↓ Soil quality criteria	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
Human risks	1.0	3.7	0.20	1.0	15.62	15.62	15.44
Agricultural risks				0.0041			
Ecological risks							
generic: organisms/processes/plants	n.a.	n.a.	n.a.	n.a.	0.020	n.a.	n.a.
specific: incl. secondary poisoning of key-target species	0.00036	0.00036	0.00036	0.00036	0.020	0.00036	0.0028
Current target value:	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#
Current intervention value:	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*
P95 of AW2000	0.0041	0.0041	0.0041	0.0041	0.0041	0.0041	0.0041
Proposed reference value:	0.0041	0.0041	0.0041	0.0041	0.0041	0.0041	0.0041
* Intervention value for PCB (sum 7)							
# target value for PCB (sum 7)							
Proposed geometric average of the MPCeco and SRCeco in this report (sum 7)							
Proposed SRCeco in this report (sum 7)							
P95 of AW 2000 report Lame et al. (2008)							

Table A2.5 PCB 138

<b>Proposed reference values based on policy decisions in the soil advisory group NOBO</b>							
<b>PCB138</b>							
<b>Criteria</b>	<b>1</b>	<b>2</b>	<b>3</b>	<b>4</b>	<b>5</b>	<b>6</b>	<b>7</b>
	<b>Residential with garden</b>	<b>Playgrounds</b>	<b>Residential with vegetable garden</b>	<b>Agriculture (without residence)</b>	<b>Nature</b>	<b>Green with nature values</b>	<b>Other green, buildings infrastructure and industry</b>
Soil use →							
↓ Soil quality criteria	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
<u>Human risks</u>	0.16	3.7	0.030	0.16	15.42	15.42	15.25
<u>Agricultural risks</u>				0.0037			
<u>Ecological risks</u>							
generic: organisms/processes/plants	n.a.	n.a.	n.a.	n.a.	0.020	n.a.	n.a.
specific: incl. secondary poisoning of key-target species	0.00036	0.00036	0.00036	0.00036	0.020	0.00036	0.0028
Current target value:	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#
Current intervention value:	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*
P95 of AW2000	0.0037	0.0037	0.0037	0.0037	0.0037	0.0037	0.0037
Proposed reference value:	0.0037	0.0037	0.0037	0.0037	0.0037	0.0037	0.0037
* Intervention value for PCB (sum 7)							
# target value for PCB (sum 7)							
<a href="#">Proposed geometric average of the MPCeco and SRCeco in this report (sum 7)</a>							
<a href="#">Proposed SRCeco in this report (sum 7)</a>							
<a href="#">P95 of AW 2000 report Lame et al. (2008)</a>							

Table A2.6 PCB 153

<b>Proposed reference values based on policy decisions in the soil advisory group NOBO</b>							
<b>PCB153</b>							
<b>Criteria</b>	<b>1</b>	<b>2</b>	<b>3</b>	<b>4</b>	<b>5</b>	<b>6</b>	<b>7</b>
	<b>Residential with garden</b>	<b>Playgrounds</b>	<b>Residential with vegetable garden</b>	<b>Agriculture (without residence)</b>	<b>Nature</b>	<b>Green with nature values</b>	<b>Other green, buildings infrastructure and industry</b>
Soil use →							
↓ Soil quality criteria	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
<u>Human risks</u>	0.23	3.7	0.043	0.23	15.49	15.49	15.31
<u>Agricultural risks</u>				0.0035			
<u>Ecological risks</u>							
generic: organisms/processes/plants	n.a.	n.a.	n.a.	n.a.	0.020	n.a.	n.a.
specific: incl. secondary poisoning of key-target species	0.00036	0.00036	0.00036	0.00036	0.020	0.00036	0.0028
Current target value:	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#
Current intervention value:	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*
P95 of AW2000	0.0035	0.0035	0.0035	0.0035	0.0035	0.0035	0.0035
Proposed reference value:	0.0035	0.0035	0.0035	0.0035	0.0035	0.0035	0.0035
* Intervention value for PCB (sum 7)							
# target value for PCB (sum 7)							
<a href="#">Proposed geometric average of the MPCeco and SRCeco in this report (sum 7)</a>							
<a href="#">Proposed SRCeco in this report (sum 7)</a>							
<a href="#">P95 of AW 2000 report Lame et al. (2008)</a>							



Table A2.7 PCB 180

<b>Proposed reference values based on policy decisions in the soil advisory group NOBO</b>							
<b>PCB180</b>							
<b>Criteria</b>	<b>1</b>	<b>2</b>	<b>3</b>	<b>4</b>	<b>5</b>	<b>6</b>	<b>7</b>
	<b>Residential with garden</b>	<b>Playgrounds</b>	<b>Residential with vegetable garden</b>	<b>Agriculture (without residence)</b>	<b>Nature</b>	<b>Green with nature values</b>	<b>Other green, buildings infrastructure and industry</b>
Soil use →							
↓ Soil quality criteria	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
Human risks	0.085	3.7	0.016	0.085	15.55	15.55	15.37
Agricultural risks				0.0027			
Ecological risks							
generic: organisms/processes/plants	n.a.	n.a.	n.a.	n.a.	0.020	n.a.	n.a.
specific: incl. secondary poisoning of key-target species	0.00036	0.00036	0.00036	0.00036	0.020	0.00036	0.0028
Current target value:	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#	0.020#
Current intervention value:	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*	1.0*
P95 of AW2000	0.0027	0.0027	0.0027	0.0027	0.0027	0.0027	0.0027
Proposed reference value:	0.0027	0.0027	0.0027	0.0027	0.0027	0.0027	0.0027
* Intervention value for PCB (sum 7)							
# target value for PCB (sum 7)							
Proposed geometric average of the MPCeco and SRCeco in this report (sum 7)							
Proposed SRCeco in this report (sum 7)							
P95 of AW 2000 report Lame et al. (2008)							

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