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Impact of a flood disaster on sediment toxicity in a major river system – the Elbe flood 2002 as a case study

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The extraordinary Elbe flood in August 2002 did not result in an overall increase of environmental contamination.

Abstract

The ecotoxicological implications of a flooding disaster were investigated with the exceptional Elbe flood in August 2002 as an example. Sediment samples were taken shortly after the flood at 37 sites. For toxicity assessment the midge *Chironomus riparius* (Insecta) and the mudsnail *Potamopyrgus antipodarum* (Gastropoda) were exposed to the sediment samples for 28 days. For a subset of 19 sampling sites, the contamination level and the biological response of both species were also recorded before the flood in 2000. The direct comparison of biological responses at identical sites revealed significant differences for samples taken before and immediately after the flood. After flood sediments of the river Elbe caused both higher emergence rates in the midge and higher numbers of embryos in the mudsnail. Contrary to expectations the toxicity of the sediments decreased after the flood, probably because of a dilution of toxic substances along the river Elbe and a reduction in bioavailability of pollutants as a result of increasing TOC values after the flood.

Keywords: Flood disaster; Sediment toxicity; Effect monitoring; *Chironomus riparius*; *Potamopyrgus antipodarum*

1. Introduction

Despite their ecotoxicological potential, extraordinary flood events in major river systems have not yet been explored in this regard. However, such events may cause considerable ecological damage by a flooding of industrial facilities, urban areas and smaller settlements with a discharge of toxic substances into the surface water body and into the sediment resulting in an increase in environmental pollution and a disturbance of formerly not polluted sites. As another conceivable consequence, such flooding events could lead to a dilution of toxic chemicals (Müller et al., 2003) at highly contaminated sites that will cause a decrease of pollution.

In August 2002, exceptional meteorological conditions provoked extreme rainfall in many regions of eastern Germany, Austria and the western Czech Republic. Statistical expectations for such events are one in a century. Precipitations of up to 70% of the monthly average were recorded at a single day. The precipitations were discharged mainly through the river Elbe catchment area. This resulted in a record water level of 9.40 m for the Elbe at the city of Dresden, while the normal level is 2.05 m. As the flood wave propagated

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down the Elbe, widespread damage was caused in cities such as Wittenberg, Dessau and Magdeburg. The Elbe bursted its banks more than 12 km in total at several points, flooding farmland, settlements, urban areas, sewage treatment works and industrial facilities.

The Elbe is one of the major streams in Central Europe flowing over a distance of 1091 km from its source in the Riesengebirge (Czech Republic) to its mouth at the North Sea near Cuxhaven (Germany). Numerous chemical plants are located along its banks as well as along the tributaries of the upper Elbe course in the Czech Republic. In the middle course a number of chemical plants and large industrial areas are situated with the area of Bitterfeld on the river Mulde achieving the most notoriety for excessive contaminant emission until 1990. The lower course of the Elbe is running through Hamburg, receiving a significant load of contaminants and nutrients originating from municipal and industrial sewage, especially in the international harbour. Previously, the river Elbe was one of the most polluted rivers in Central Europe as many industrial plants and cities in the Czech Republic and the former German Democratic Republic discharged their untreated wastewaters into the river. Due to technical improvements of production processes or sewage treatment the contaminant input was reduced considerably in recent years but sediment toxicities remained at a high level (Traunspurger et al., 1997; Ahlf et al., 1999; Duft et al., 2003c).

It was the main objective of the study to compare the biological effect monitoring results and chemical residue analyses before and after the flood. To this end the data of the current investigation are compared with results of two Elbe sediment toxicity investigations performed in 2000 before the flood (Schulte-Oehlmann et al., 2001b; Duft et al., 2003c). Furthermore, the present study provides an opportunity to generalise the effects of high flood events in major river systems.

2. Material and methods

2.1. Sediment sampling

Samples were obtained between 09/08/02 and 09/16/02, immediately after the flood had subsided in parts of the river where there is little flow activity such as between breakwaters, in abandoned channels or in harbours. In some cases the sampling was performed from boats. Aerobic sediment samples were collected at 37 sampling locations along the river Elbe using a Van Veen bottom grab or a spoon spatula. Fig. 1 shows the sampling sites with their respective location along the river course. The top aerobe 2 cm was removed, placed in polycarbonate containers and cooled at 4 °C. Samples were homogenized before aliquoting for toxicity testing.

2.2. Bulk-phase sediment chemical analyses

Bulk-phase sediment chemical analyses were conducted at GALAB Laboratories (Geesthacht, Germany). Polycyclic aromatic hydrocarbons ((PAH): naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysen/triphenylene, benzo[*b*]fluoranthene, benzo[*k*]-fluoranthene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]anthracene, dibenz[*a*,*h*]anthracene, benzo[*g*,*h*,*i*]-perylene) were analysed according to DIN 38414 S23 (2002). PCBs (congeners No. 28, 52, 101, 138, 153 and 180), dichloro-diphenyl-trichloroethane (DDT) and metabolites were analysed using DIN 38414 S20 (1996). To determine the concentrations of polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans (PCDD/PCDF) DIN 38414 S24 (2000) was applied. Analytical methods for quantification of xenoestrogens (bisphenol A (BPA) and metabolites, alkylphenols, alkylphenol ethoxylates, alkylphenol carboxylates) and organotin compounds (monobutyltin (MBT), dibutyltin (DBT), tributyltin (TBT), tetrabutyltin (TeBT), monooctyltin (MOT), dioctyltin (DOT) and triphenyltin (TPT)) are described by Heemken et al. (2001) and Kuballa et al. (1995), respectively.

2.3. The 28 d-sediment toxicity tests

The non-biting midge *Chironomus riparius* (Arthropoda, Diptera) was used as one of the test organisms. This genus is widely applied for sediment characterisations (Ingersoll and Nelson, 1990; Bleeker et al., 1999; Ristola et al., 1999). The second test organism was the parthenogenetic and ovoviviparous freshwater snail *Potamopyrgus antipodarum* (Gastropoda). Duft et al. (2002, 2003a,b) established a sediment toxicity test with this species. Both species inhabit the upper layers of aquatic sediments, feeding on plants and detritus.

The brood stock of *C. riparius* was originally received from Bayer AG Leverkusen. The 28 d-sediment toxicity tests with this species were carried out according to OECD Guideline 218 (OECD, 2002) with the following modifications. The control sediment was made with quartz sand with a predominate grain size between 250 and 355 μm (Quarzwerke Frechen, Germany). For the control sediment 1.6% pulverized leaves of alder (*Alnus glutinosa*) and 0.5% pulverized leaves of stinging-nettle (*Urtica dioica*) were added as carbon sources. The content of total organic carbon (TOC) in the control sediment was 1.48%. In addition to these carbon sources, the test organisms were fed daily during exposure with TetraMin® (1 mg/individual/d). Every test vessel contained 80 g ww of field sediment and was filled up with 400 mL of reconstituted water (530 μS/cm). To ensure sufficient aerobe conditions in the sediment all samples were aerated for 21 days before

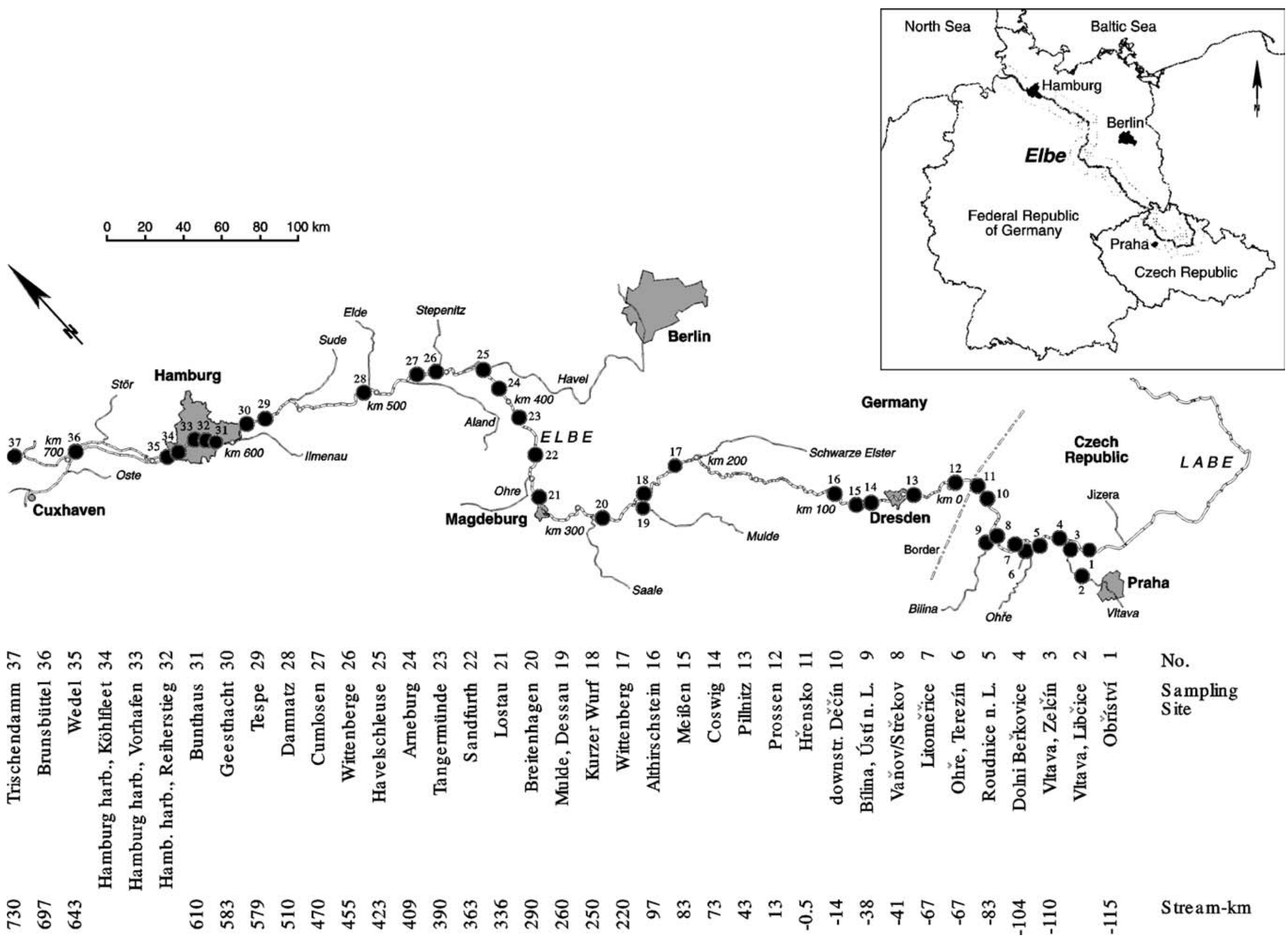

Fig. 1. Map of the river Elbe from the source in the Czech Republic to the estuary showing the 37 sampling sites.

inserting the test organisms. Each sediment sample and the control were replicated twice. The total emergence per test vessel was recorded daily.

P. antipodarum used in the experiments came from our own lab culture, which was built up with specimens collected from a small creek near Ibbenbüren, Germany. The breeding stock was kept at 16 ± 1 °C and a 16:8 h light cycle in 10 L-tanks with reconstituted water. The organisms were fed daily with pulverized TetraPhyll® (1 mg/individual/d). Groups of prosobranch snails (50 sexually mature snails per sample) with a shell height ≥ 3.7 mm were exposed to the test sediments and the control sediment. For the control sediment the same quartz sand was used as for the test with *C. riparius* containing 5% pulverized leaves of beech (*Fagus sylvatica*) as carbon source (equivalent to 1.95% TOC). The experiment was conducted in a static system in 1 L-Erlenmeyer flasks under the same conditions as summarised for the breeding stock. All flasks were filled with 50 g of wet sediment and 1 L reconstituted water. To ensure sufficient aerobe conditions in the sediment all samples were aerated using a glass Pasteur pipette for 21 days before inserting the test organisms. On day 28, the mortality of adults and the reproduction success were assessed by counting the embryos in the brood pouch of 20 randomly selected maternal snails per treatment (for details c.f. Schulte-Oehlmann et al., 2001b; Duft et al., 2003a,b).

2.4. Data analysis

The biological variables were tested for normality with the Kolmogorov–Smirnov-test and for homogeneity of variance with the Cochran-test. Differences between the controls and the field samples were calculated by using an Analysis of Variance (one-way ANOVA) followed by Tukey HSD test. This statistical analysis was performed using Statistica 5.0® and Prism 4.0® software. Chemical variables were subjected to a Principal Component Analysis (PCA) in order to find the main trends in the data. An initial PCA revealed that most of the variance in the data is due to three extreme outlier sampling sites. Consequently, these sampling sites (11, 19, 21) were excluded from further analyses.

For a subset of 19 sampling sites (12, 15, 19, 21–36), organotin compounds and two biological response variables (total emergence of *C. riparius* and number of embryos of *P. antipodarum*) were recorded also in 2000, before the 2002 flood. This allowed before/after comparisons that were carried out with one-way ANOVAs and Bartlett's test of variance homogeneity. The effect of the flood on spatial distribution of organotin contamination was assessed by spatial autocorrelation analysis. The statistical significance of Geary's C was estimated from 1000 randomisations (Sawada, 1999).

3. Results and discussion

3.1. Chemical contamination

The three outlier samples 11, 19 and 21 in the initial PCA (see chapter 2.4) were excluded from further statistical analyses due to their exceptionally high content of PAHs and organotin compounds. PCA extracted two axes with eigenvalues greater than 7, representing 21.5% and 11.3% of total variance in the data. Inspection of the factor-loading matrix revealed that axis 1 is essentially a gradient of PAHs, whereas axis 2 opposes sampling sites with high and low concentration of organotin compounds. While the position of sampling sites on axis 1 follows no obvious pattern, axis 2 associates the sampling sites from the source and the upper Elbe with high organotin concentrations (Fig. 2).

3.2. The 28 d-sediment tests

Emergence as an endpoint has been used in numerous studies with Chironomidae for sublethal toxicity (Pascoe et al., 1989; Groenendijk et al., 1998, 1999; Oetken, 1999; Oetken et al., 2001). As shown in Fig. 3 the total emergence of *C. riparius* ranged from 50% (site 19) to 100% (sites 2 and 36). Compared to the control (95% emergence) there are no significant differences in mean emergence for any of the field sediments (one-way-ANOVA, Tukey HSD test) even at those few sampling sites (19, 26, 30 and 31) with a comparably low emergence, due to high standard deviations. All control emergence rates exceeded the minimum of 70% and hatching occurred between days 12 and 23 after insertion of the larvae. These values are recommended for test acceptability of the OECD guideline 218 (OECD, 2002).

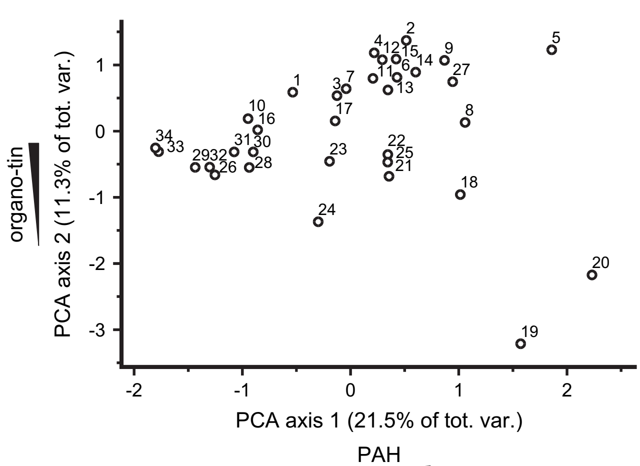

Fig. 2. Scatterplot of factor scores of sampling sites on Principal Component axes 1 and 2. Axis 1 is a gradient of polyaromatic hydrocarbons (PAH) and axis 2 of organotin compounds. Sampling sites are labelled by their number (see Fig. 1).

P. antipodarum has successfully been used in studies assessing the reproductive toxicity of freshwater sediments (Schulte-Oehlmann et al., 2001b; Duft et al., 2002) and investigating the effects of endocrine disrupting chemicals with either androgenic (Schulte-Oehlmann, 1997; Duft et al., 2003a) or estrogenic activities (Schulte-Oehlmann et al., 2001a; Duft et al., 2003b).

On day 28, the mean number of embryos in the brood pouch of control snails was 14.0 (Fig. 4). The average number of embryos in snails exposed to native sediments ranged from 11.6 (site 11) to 27.6 (site 37). Fifty-one percent of the field sediments turned out to have a promoting effect on the embryo production per female compared with the control sediment. The differences to the control snails were significant for 19 sampling sites (4, 5, 9, 13–15, 17, 21, 23, 24, 26–29, 32, 34–37; one-way ANOVA with Tukey HSD test, $p < 0.001$).

A possible explanation for the increase of embryo numbers in more than 50% of the Elbe sediments is the presence of high concentrations of estrogenic compounds in the field sediments. The Elbe is known to be contaminated with a range of estrogenic compounds including alkylphenols, alkylphenol ethoxylates, alkylphenol carboxylates and bisphenol A (Stachel et al., 2003). The analysed sediment samples taken after the flood in September 2002 showed the following median values (in μg/kg dw): BPA = 30, branched nonylphenols (NP) = 124 and nonylphenol ethoxylates (NPnEO, $n = 1$ and 2) = 75. Duft et al. (2003b) demonstrated a stimulation of embryo production in *P. antipodarum* within a 28 d-sediment test, when the test species was exposed to the two xenoestrogens BPA and 4-*tert*-octylphenol (OP). The authors calculated EC_{50} values of 5.67 μg BPA/kg dw and 108 μg OP/kg dw, respectively. While in all field sediments the concentration of the latter was below the EC_{50}, the concentrations of BPA were above 5.67 μg/kg dw (median = 30 μg/kg dw) at most of the sampling sites. However, no significant correlation between the measured BPA values and the embryo numbers was found ($r = 0.033$). There was also no significant correlation between the biological effects (total emergence in *C. riparius* and embryos in

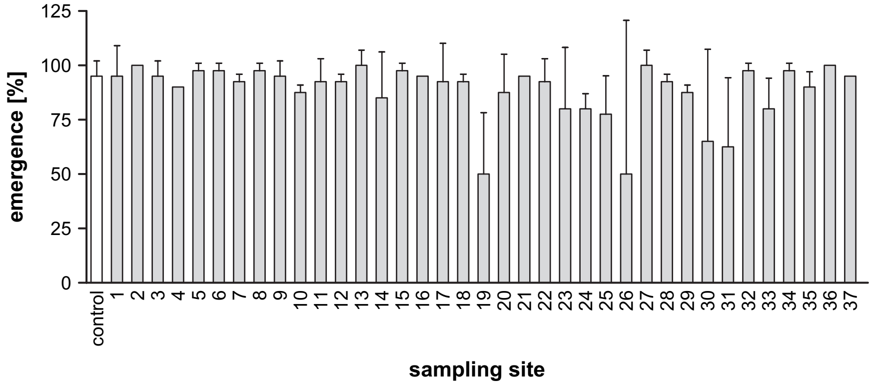

Fig. 3. The 28 d-sediment toxicity test with *C. riparius*. Mean emergence (%; ± SD) after exposure to field sediments taken from 37 sampling sites along the river Elbe shortly after the Elbe flood in August 2002. Data of the 2 replicates per sampling site were pooled, 20 first instar larvae per replicate were used.

P. antipodarum) and sediment contamination of the two most prominent factors of contamination, PAHs and organotin compounds, as revealed by the outcome of the PCA analysis (Table 1). This indicates that other factors such as mixture effects of contaminants, not measured compounds or abiotic sediment properties like organic carbon content may have contributed to the observed variation in biological effect data after the flood in addition to the exposure to estrogenic compounds.

3.3. Comparison of organotin compounds and biological response before and after the flood

A comparison of average concentrations of organotin compounds (sum of MBT, DBT, TBT and TeBT concentrations in sediments) taken in 2000 and 2002 revealed no significant overall differences between the years (2000: 1093 ± 3476 μg Sn/kg dw; 2002: 737 ± 762 μg Sn/kg dw). However, the variances of the total organotin concentrations were significantly different among the years (Table 2a). This indicated that these contaminants were more homogeneously distributed among sampling sites in 2002 than in the previous year. In contrast to chemical variables, biological responses of both test species differed significantly between the years. The average emergence of *C. riparius* (2000: 67.8% ± 26.6% referred to control; 2002: 87.8% ± 16.8% referred to control) and also the average numbers of embryos in the brood pouch of *P. antipodarum* (2000: 38.3% ± 28.5% referred to control; 2002: 143% ± 23.2% referred to control) were significantly higher after the flood (Table 2b, c; Fig. 5).

For the interpretation of these results, it is important that the content of TOC in the sediments increased significantly after the flood (6.11% ± 3.43%) compared to values measured before the flood by Duft et al. (2003c) (2.57% ± 1.94%) at the same stations. An increase of the TOC was also observed shortly after an extreme flood event in the river Odra in August 1997. The average TOC doubled with regard to the normal

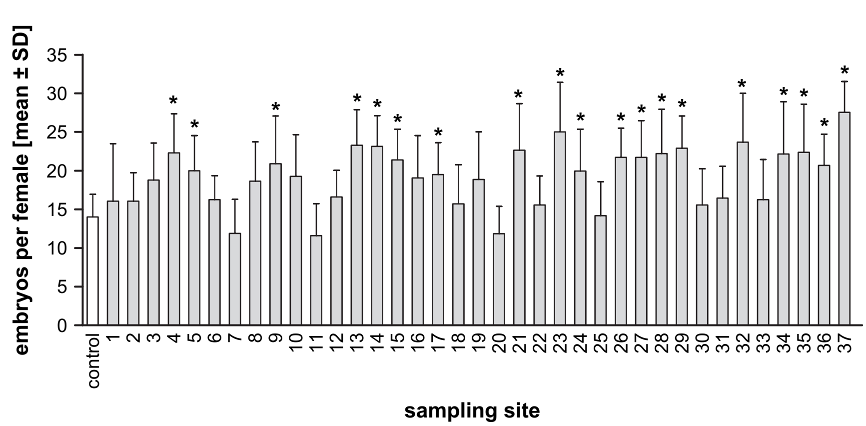

Fig. 4. Mean number of embryos (%; ± SD) in the brood pouch of *P. antipodarum*. The snails were exposed to sediments of the river Elbe, taken from 37 sampling sites shortly after the Elbe flood in August 2002. The embryo number of 20 snails per treatment was determined after 28 days of exposure (one-way ANOVA; Tukey HSD test; * = significantly higher number compared to control; $p < 0.001$).

Table 1
Correlation coefficients and their statistical significance of chemical variables and biological response variables

	Chironomus riparius		*Potamopyrgus antipodarum*	
	Emergence		Embryos per female	
	r	*p*	*r*	*p*
PCA axis 1	−0.032	0.430	−0.250	0.077
PCA axis 2	0.226	0.099	−0.082	0.323

conditions (Müller and Wessels, 1999). Müller et al. (2003) investigated contaminant levels and ecotoxicological effects in sediments from the flood event in the river Odra. The authors also found a dilution of contaminants, particularly for the PAHs, as a result of the flood. On the other hand, the assessment of sediment toxicity, based on the testing of elutriates, did not reveal any changes of the ecotoxicological potential in the Odra sediments.

In case of the Elbe sediments food supply may have been enhanced for herbivorous organisms such as *C. riparius* and *P. antipodarum* after the flood, thereby masking the toxicity of contaminants (Besser et al., 2003; de Haas et al., 2004). Furthermore, the bioavailability of pollutants may have been reduced by adhesion on the organic fraction (Nikkila et al., 2003; Hecht et al., 2004). Additionally, the after flood sediments mainly consisted of a major fine grit fraction what turned out to be a major difference to sediments taken before the flood. The median grain size of 19 sampling sites was 115 μm before the flood (Duft et al., 2003c) instead of 22 μm after the flood, resulting in a decrease of bioavailability of toxic substances (Lin et al., 2003; Maenpaa et al., 2003). These findings are supported by Szyposzynska (2002) who identified a negative correlation of the emergence of *C. riparius* and the sediment toxicity of triphenyltin (TPT) in fine-grained sediment.

Table 2
ANOVA tables with Bartlett's test for variance heterogeneity for (a) total organotin concentrations, (b) emergence of *C. riparius* and (c) number of embryos of *P. antipodarum* on 19 sampling sites measured in 2000 and 2002

Source of variation	d.f.	SS	MS	*F*	*p*
(a) Bartlett's test on variance homogeneity: $\chi^2 = 46.70$; d.f. = 1; $p = 1 \times 10^{-10}$*					
Among 2000 and 2002	1	5.1×10^6	5.1×10^6	0.42	0.52
Within years	36	4.4×10^8	1.2×10^7		
Total	37	4.5×10^8			
(b) Bartlett's test on variance homogeneity: $\chi^2 = 3.35$; d.f. = 1; $p = 0.06$					
Among 2000 and 2002	1	3263	3262	7.45	0.0097*
Within years	36	15 759	438		
Total	37	19 021			
(c) Bartlett's test on variance homogeneity: $\chi^2 = 0.74$; d.f. = 1; $p = 0.38$					
Among 2000 and 2002	1	62 449	62 449	92.1	2×10^{-11}*
Within years	36	24 384	677		
Total	37	86 832			

*Significant differences are marked with asterisks. d.f. = degrees of freedom, SS = sums of squares, MS = mean squares.

The lower variance of organotin residues between sampling sites indicates a more uniform distribution of organotin compounds in the Elbe sediments after the flood in 2002. This applies also for the lower variance of the TOC and grain size data after the flood. Obviously, the flood had an equalizing effect on organotin concentrations and therefore also on the overall contamination level. This conclusion is supported by spatial analysis. In the year 2000 samples, the content of organotin compounds is significantly autocorrelated between sampling sites not further apart than 30 km. After the flood, no such interdependence was found.

These findings pose a question on generalization of the consequences of flood disasters in major river systems. Although there seem to be major discrepancies on first glance, even if our own results are compared with other investigations at the Elbe after the flood, the time of sampling relative to the flood event and the regional extension of the investigation area have to be considered. Grote et al. (2004) analysed the toxicity of 11 sediment samples from the Elbe which were taken seven to eight months after the flood in March and April 2003. Native sediments were tested using the 24 h *Danio rerio* egg development assay and the 7 days *Lemna minor* growth test. Furthermore the toxicity of organic sediment extracts was assessed with *Vibrio fischeri* (bioluminescence assay), *Scenedesmus vacuolatus* (24 h reproduction test), *Daphnia magna* (24 h acute test), the umu test for genotoxicity and fish RTL-W1 cell lines for cytotoxicity. Most sediment samples exhibited marked responses in at least one of the applied assays but the authors found no evidence for higher toxicities in the tested samples compared to the period before the flood. However, the tested samples do not necessarily represent flood sediments due to sedimentation processes in the months after the flood. It has been shown by Stachel et al. (in press) that concentrations of many contaminants in sediments increased in the months after the Elbe flood from August 2002 and attained again pre-flood levels.

While the findings of the present study are consistent with a toxicity assessment in Elbe sediments from the Hamburg harbour area (Maaß et al., 2003), the results of an investigation performed by Heise et al. (2003) showed an increase of sediment toxicity of after flood sediments taken from the estuary. The authors determined increasing toxic effects on algae (*Pseudokicheriella subcapitata*) and nematodes (*Caenorhabditis elegans*). It can be assumed that contaminated sediments transported with the flood wave deposited in the estuary providing an explanation

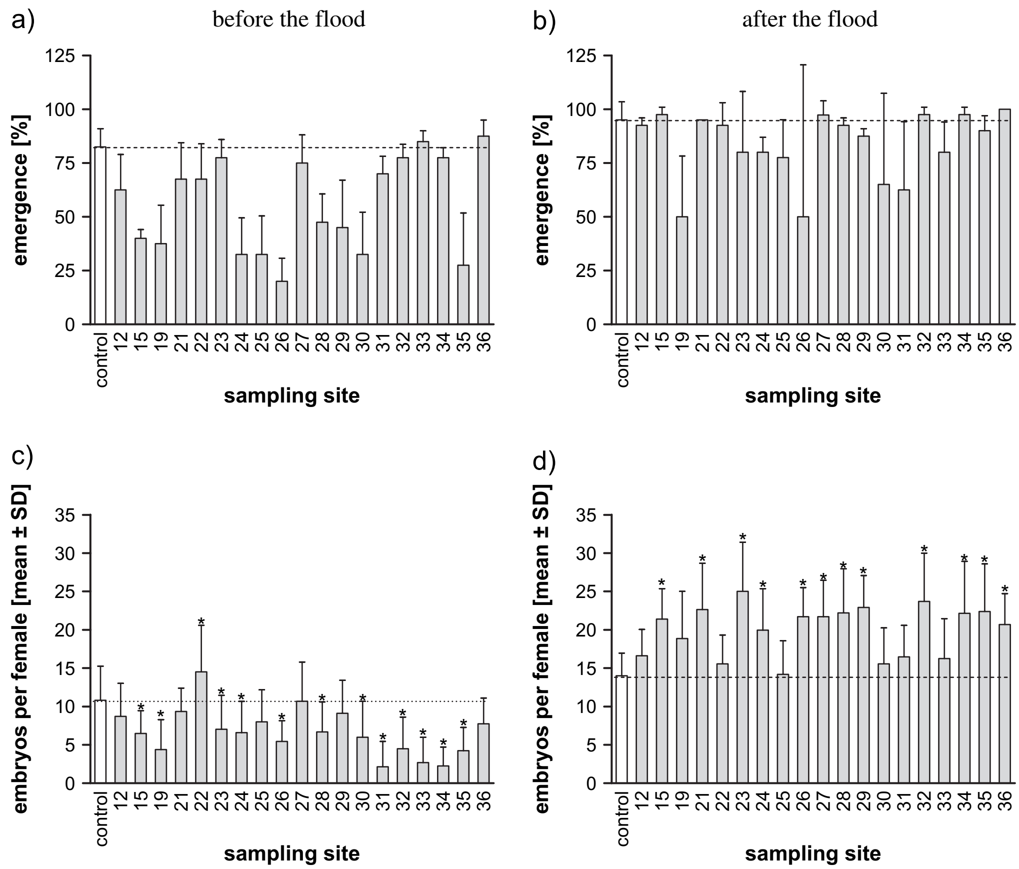

Fig. 5. Comparison of the biological response of *Chironomus riparius* (a, b) and *Potamopyrgus antipodarum* (c, d) exposed to sediments from 19 identical sampling sites of the river Elbe before (left charts) and after (right charts) the Elbe flood. Biological effect data for sediments sampled before the flood are from Duft et al. (2003c) for *C. riparius* and from Schulte-Oehlmann et al. (2001b) for *P. antipodarum*. One-way ANOVA, Dunnett-test; significant differences on the 5% level are indicated by asterisks.

for an increasing sediment toxicity in this section of the Elbe course. Furthermore, the two biotests applied by Heise et al. (2003) may respond to other contaminant classes in sediments as it has been shown by Duft et al. (2003c) in a direct comparison of the *Chironomus riparius* and *Caenorhabditis elegans* sediment bioassay.

Comparable results like in our study were reported by Müller et al. (2003) for the Odra flood in 1997. The flood wave transported contaminated sediments downriver. After the flood, the levels for almost all analysed contaminants dropped below the values measured in the time before this event. In the river reach, the flood had a diluting effect for organic contaminants that continued for PCBs until 1999. The PAH load, however, rose again, because the anthropogenic activities on the upper river course continued.

Hollert et al. (2000) investigated the cytotoxic and genotoxic potentials of settling particulate matter (SPM) carried by the Neckar river during a winter flood event using the neutral red retention assay with fish RTG-2 cells and the Ames test, respectively. Samples taken during the period of hood rise showed the highest cytotoxic activities while there was no indication of any genotoxic activity in native surface waters, pore waters, and SPM.

Kemble et al. (1998) assessed the extent of sediment contamination in the Upper Mississippi River after a flood event in 1993. Effects of 25 sediment samples on survival, growth, and sexual maturation in the amphipod *Hyalella azteca* were tested in a 28-day assay. Amphipod survival and growth was significantly reduced in only one sediment sample. Based on their results the authors conclude that the after flood sediments were relatively uncontaminated compared to other areas of known contamination in the United States.

Our own results and the literature reports indicate that in watercourses exhibiting a comparable pattern of uncontaminated and highly contaminated sites (“hot spots”) as found in the current study, for example also in the river Odra (Wolska et al., 1999), an extreme flood will probably cause both a decrease of the contamination level at “hot spots” and slightly increasing concentration at previously less contaminated sites. Consequently, the contamination will be levelled after a flood. Hence, it can be assumed, that at least for catchment areas with comparable unequal distribution of pollutants like in the river Elbe, the spatial homogenisation of the contamination as well as the ecotoxicological effects found in this case study may represent a general principle.

4. Conclusions

This investigation shows that the extraordinary Elbe flood in August 2002 did not result in an overall increase of environmental contamination compared to levels found before the flood in 2000, in spite of flooding industrial facilities, urban areas and smaller settlements. However, presumably as a result of dilution, the toxic substances are considerably more homogeneously distributed along the river Elbe. As a consequence, sediments sampled shortly after the Elbe flood were less toxic in both test organisms *C. riparius* and *P. antipodarum*. In combination with TOC and grain size analysis, the data suggest additionally, that the bioavailability of pollutants was reduced after the flood. The results of this study may represent a general principle of the impact of flood disasters on contamination levels and resulting sediment toxicity in major river systems with a comparable unequal distribution of pollutants like in the river Elbe.

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