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### Persistence of benthic invertebrates in polluted sediments.

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# CHAPTER 7

## CONCLUDING REMARKS

The aim of the present thesis was to explain the persistence of benthic invertebrate species in polluted sediments. In the previous chapters, it has been demonstrated that the responses of the two model species, the mayfly *Ephoron virgo* and the midge *Chironomus riparius*, to polluted sediments were species-specific, and that also their responses to natural environmental variables were very different.

*E. virgo* and *C. riparius* do not occur in the same habitat, and the application of these model species to floodplain lakes is not supported by a proven natural distribution in these habitats. Nevertheless, these model species may serve as good representatives for sediment inhabiting invertebrates. It is attempted here to compare the species-specific responses of the model species to several environmental variables and biotic interactions. Subsequently, it will be discussed how these species-specific responses determine the species' persistence in polluted sediments.

### RESPONSES TO ENVIRONMENTAL GRADIENTS

Both model species are typical inhabitants of the slow-flowing lower reaches of rivers. The environmental conditions in these river reaches are often similar to conditions in waters that are more or less stagnant (WARD 1992). Therefore, many benthic invertebrates inhabiting slow-flowing river habitats are facultative riverine lake and pond species, which also holds for *E. virgo* and *C. riparius* (ILLIES & BOTOSANEANU 1963).

During their development both species live in the sediment where they build tubes. These tubes have several important functions, e.g. protection against predation and disturbance, and acquisition of food and oxygen (e.g. ERIKSEN 1968, RASMUSSEN 1984, HERSHEY 1987, KURECK & FONTES 1996). It is therefore of critical importance for both species to find a substrate in which they can construct a stable tube. Although few benthic invertebrate species are restricted to a specific substrate, most species exhibit distinct preferences for a general bottom type (e.g. mud, sand, gravel). The close association of a particular species with a given substrate may reflect current preferences, requirements for shelter, respiratory needs, or food habits, rather than directly indicating an affinity for this specific type of substrate.

*E. virgo* nymphs preferably burrow their U-shaped tubes in loamy to sandy riverine substrates (SCHLEUTER 1989, GYSELS 1991, IBÁÑEZ et al. 1991, BIJ DE VAATE 1992) that contain low amounts of organic material (SCHLEUTER 1989), although their substrate preference is not very strict (SCHOENEMUND 1930, TOBIAS 1996). This might especially apply to the early instar nymphs that live freely on the sediment because in two 'clean' sediments substrate grain size (silt-loam vs. sandy-loam) had no effect on survival of the nymphs (CHAPTER 2).

*C. riparius* larvae preferably build their tubes in organic rich sediments in which clay- and silt-sized particles predominate (OLIVER 1971, PINDER 1986). However, they may also build their tubes in coarse sand (VOS et al. 2002), although they are not able to efficiently construct tubes in the latter substrate (ÅKERBLOM & GOEDKOOP 2003).

Thus, both species prefer different types of substrates (Table 7.1). *E. virgo*, which prefers loamy to sandy substrate with low organic enrichment, might not be present in the highly organic silt sediments that are preferred by *C. riparius*, and vice versa.

In highly organic sediment, oxygen deficits commonly occur. Even when the overlying water is well oxygenated, the oxygen levels may rapidly decline with sediment depth, which is mainly caused by bacterial respiration associated with decomposition of organic matter (CHAPTER 5).

The ventilatory movements of many benthic invertebrates enhance oxygen uptake by creating currents across boundary layers (JAAG & AMBÜHL 1964). Burrowing mayflies beat their gills with a frequency inversely related to the oxygen concentration in their burrow, to maintain a current through their burrow which directly functions in oxygen uptake (ERIKSEN 1968). VAN DER GEEST et al. (2002) demonstrated that anoxic and hypoxic conditions had detrimental effect on the survival of early *E. virgo* nymphs, but that at oxygen concentrations of 50% or higher, no effects on survival were observed after four days of exposure.

Some benthic invertebrate species possess haemoglobin, which functions in oxygen transport during hypoxic conditions and a short-term oxygen store (WEBER 1980, ZEBE 1991). *Chironomus* larvae have the ability to synthesise haemoglobin in response to low levels of oxygen (HEINIS 1993, FOX 1955). The short-term oxygen storage properties of haemoglobin allow *Chironomus* larvae to alternate periods of feeding with undulatory movements under low-oxygen conditions (WALHSE 1951). Once haemoglobin oxygen stores are depleted, *Chironomus* larvae can rely on alcoholic fermentation for anaerobic energy production (REDECKER & ZEBE 1988). Yet, the increased energy spent on

respiration under these low-oxygen conditions may reduce the amount of energy left for growth (BECKER 1987) as also was observed in the field enclosures in CHAPTER 6.

Thus, the two model species occupy very different positions in natural oxygen gradients (Table 7.1). *E. virgo*, which is sensitive to low-oxygen conditions, might not be present in highly organic sediments, in which low oxygen levels are commonly encountered. In contrast, *C. riparius* is particularly fit to cope with such oxygen-depleted sediments.

Food quantity has been presumed to be the principal environmental factor influencing growth rates of benthic invertebrate species (ANDERSON & CUMMINS 1979), with consequent effects on duration of their life cycle (PINDER 1986). *C. riparius* is frequently encountered in high densities in organically enriched sediments in which food is rarely limiting (LEARNER & EDWARDS 1966, GOWER & BUCKLAND 1978, FERRINGTON & CRISP 1989). The most commonly reported food ingested by *C. riparius* is detritus (RASMUSSEN 1984, 1985, PEREIRA et al. 1982, PINDER 1986), consequently, the growth rate of *C. riparius* larvae is often related to the amount of detritus available (RASMUSSEN 1985).

Recent studies showed, however, that it is not the amount of food, but the nutritional value of the food that determines the growth rates of benthic invertebrates (GOEDKOOP et al. 1998, VOS 2001). In agreement, the present observations showed that the growth rate of *C. riparius* larvae were higher at sediments with higher nutritional value, with (polyunsaturated fatty acids and chlorophyll *a* as food quality indicators, (CHAPTER 2 & 6) leading to shortening of the time to emerge (CHAPTER 2). The addition of a highly nutritive food source stimulated growth rates (CHAPTER 2 & 3) and reduced the time to emergence (CHAPTER 2), especially in sediments that were low in food quality (CHAPTER 2).

Thus when high quality food is highly available, larvae may spend less energy on foraging, enabling them to allocate more energy for growth than in environments where food is scarce (RISTOLA et al. 1999). Because *C. riparius* larvae feed by extending their head and anterior part of the body outside the tube (EDGAR & MEADOWS 1969, RASMUSSEN 1984), their foraging areas are restricted to a region immediately surrounding their tube (RASMUSSEN 1984). It is therefore of crucial importance for the larvae to settle in an environment with a sufficient amount of high quality food. In CHAPTER 4 it was demonstrated that a higher number of larvae settled on substrates with higher food quality compared to substrates with lower food quality. When food availability is low

the larvae may leave their tubes in search for a habitat with higher food availability (CHAPTER 4 & 5).

The food quality in the sediments tested in the present thesis did not lead to detectable differences in survival and growth of *E. virgo* nymphs. Also additional feeding with highly nutritious diatoms did not enhance survival or growth (CHAPTER 2). Thus the amount of food in the tested sediments was probably saturating for *E. virgo*, in contrast to *C. riparius*. Thus *E. virgo* has a much lower food demand than *C. riparius*. *E. virgo* nymphs may consequently be food limited only under oligotrophic conditions. Rhine water flowing through culture systems increased the growth rate of early instar *E. virgo* nymphs, indicating their dependence on nutritional supplements as was demonstrated by KURECK et al. (2001).

The influence of food on the performance of both model species is markedly different and may be explained by their differences in growth rate (~ 0.2 mm for *E. virgo* vs. ~ 8 mm for *C. riparius* after 10 days) and life cycle duration. *E. virgo* is restricted to a univoltine life cycle, because of the required temperatures for egg diapause and hatching of the nymphs (LEHMKUHL 1974), whereas, in temperate regions, *C. riparius* has a multivoltine life cycle with 2 to 3 generations per year (TOKESHI 1995), and even 7 generations per year were observed for *C. riparius* in an organically polluted English river (LEARNER & EDWARDS 1969). In accordance, nymphs of *E. virgo* take much more time to develop, 3 to 4 months (KURECK & FONTES 1996), compared to *C. riparius*, which is able to complete their life cycle in 3 to 4 weeks (CHAPTER 2). Thus, the amount of food needed per unit of time is much higher for *C. riparius* than for *E. virgo* (Table 7.1). When the food demands of both species are saturated, *C. riparius* grows much faster and develops far more rapid. This high growth is reached on sediments with a sufficient amount of high quality food and not on sediments with low amounts of high quality food that may sustain the low developmental rate of *E. virgo*.

Contaminated food might decrease the organisms' health by reducing food ingestion rates (HARKEY et al. 1994a, LEPÄNEN et al. 1998) or other metabolic processes (DEPLEDGE 1998). Some studies observed that *E. virgo* has a high intrinsic sensitivity to contaminants (VAN DER GEEST et al. 2000a, 2000b), although it showed to be resistant to zinc (BOIVIN et al. 2001, VAN DER GEEST et al. 2001). The present thesis demonstrated that sediments with high levels of a mixture of contaminants (metals, PAHs, PCBs, and numerous unidentified compounds) had detrimental effects on survival and growth of *E. virgo* (CHAPTER 2 & 5).

In contrast to *E. virgo*, the amount of toxicants in natural sediments had little effect on the performance of *C. riparius* (CHAPTER 2 & 6). *C. riparius* is, however, not necessarily tolerant to contaminants. Copper spiked 'clean' sediment caused negative effects on survival, dry weight, and length of the larvae (CHAPTER 3), though the median effect concentrations were much higher than the highest copper concentration measured in the natural sediments tested. Addition of highly nutritive food to the sediment probably improved the physiological state of the larvae, and therewith enabling them to put more energy in detoxication or repair mechanisms (HEUGENS et al. 2001), which was, at least partly, observed as a decrease in copper accumulation with increasing amounts of highly nutritive food (CHAPTER 3). Moreover, when the food quality in two sediments was equal a higher proportion of the larvae choose the sediment with the lower contaminant load (CHAPTER 4). The ability of *C. riparius* to maintain populations in sediments that are contaminated does, however, not necessarily imply that they do not suffer from the high contaminant levels, because at higher contaminant concentrations in the sediment a higher incidence of mentum deformities was observed (CHAPTER 6). Under these specific circumstances even genetic adaptation is realized by *C. riparius* (POSTMA & GROENENDIJK 1999)

Thus, both species differed widely in their sensitivity to historically contaminated sediments (Table 7.1). The amount of contaminants in most tested sediments exerted toxic effects on *E. virgo* and although high incidences of mentum deformities were induced in *C. riparius*, their development was not impaired.

## RESPONSES TO BIOTIC INTERACTIONS

Competition may affect growth and development of benthic invertebrates, mainly, but not always, through the exploitation of food resources (RASMUSSEN 1985, VAN DE BUND 1994, RISTOLA et al. 1999). Although the mechanisms of competition are not always clear, it appears that intensive disturbance leads to lower food uptake, and, consequently, reduces growth and survival rates (e.g. KAJAK 1988, REYNOLDSON et al. 1994, KURECK 2001). Nevertheless, several detritivorous chironomid taxa showed to have similar distribution patterns (CHAPTER 6). It is plausible that resource partitioning or character displacement between those species occur (RASMUSSEN 1985, JOHNSON 1986), which may be important in reducing competition. RASMUSSEN (1984, 1985) hypothesised that different feeding modes allowed the collector-filterer *Glyptotendipes paripes* and the collector-gatherer *C. riparius* to co-exist at high densities in a Canadian prairie pond. Character displacement has been found in two deposit feeding mud

snails co-existing in brackish waters (FENCHEL 1975) by differentiating food sources and adapting the ingestion apparatus to the different particle sizes they feed on. Likewise, the difference in mentum sizes of different chironomid species, and larval instars, enable them to co-exist (MCLACHLAN 1977, HODKINSON & WILLIAMS 1980).

In some cases, however, the presence of one species benefits another by conditioning the environment. Benthic chydorids (Crustacea: Cladocera) were able to take advantage of the presence of chironomids by feeding on their faeces (VAN DE BUND & DAVIDS 1993). Another example of commensalism is the re-fractionating of large organic matter particles by shredders, thereby increasing the amount of food available for collector-gatherers (CUMMINS 1973).

In contrast, behavioral interference may reduce feeding rates, negatively affecting growth and development (TOKESHI 1995) or even reduce survival. An increase in the density of *C. riparius* resulted in a decrease in survival of *E. virgo* nymphs, due to the intensified competition for both space and food (CHAPTER 5). Similar observations were made by KURECK et al. (2001): in the presence *Dikerogammarus villosus*, a highly mobile neozoan gammarid, a smaller proportion of early instar nymphs survived after 2 months of exposure compared to non-exposed nymphs. Older nymphs were more able to survive the exposure to *D. villosus*, because late instar nymphs live in burrows, while the early instar nymphs live freely on the sediment.

At high densities of larvae in sediments disturbance can even be so strong that the abundance of newly hatched chironomid larvae can be greatly reduced or even be eliminated (KAJAK et al. 1980). It is therefore of crucial importance for benthic invertebrates to select a suitable habitat, especially when they are sedentary during at least a part of their lifespan. The first instar larvae of many chironomid species are pelagic until a suitable habitat has been found (OLIVER 1971) and competition for resources is limited (CHAPTER 4). Competition among prey species increases their availability to predators owing to increased mobility (e.g. in search for food) (CHAPTER 5), while the poorer physical conditions of the competing species increases their vulnerability (KAJAK et al. 1980, MACCHIUSI & BAKER 1992). Similar observations were made in CHAPTER 6: in field enclosures stocked with young *C. riparius* larvae their survival was inversely related to the density of Ceratopogonidae, although other prey species were much more numerous. The high abundance of resident benthic invertebrates could have prevented the small introduced larvae to settle.

Benthic invertebrate predators are strongly dependent on the abundance of available prey that also hides in sediment. Predators may exploit the prey especially when these abound (KAJAK 1980). TEN WINKEL (1987) found that this was the case with Hydracarina (Arachnida) feeding on dense chironomid

populations. Some larval Tanypodinae (Diptera: Chironomidae), although not feeding on chironomids in the field, do select them in laboratory experiments, obviously due to the higher prey availability (BAKER & MCLACHLAN 1979). Fish can also increase prey availability to invertebrate predators; by stirring up the bottom the tubes of chironomids were damaged making them more vulnerable to predatory Hydracarina (TEN WINKEL 1987). Such situations appear to be exploited by 'opportunistic' predators, which become predatory only when prey is particularly easily available and vulnerable (KAJAK 1988).

Summarizing, the early instars of both *E. virgo* and *C. riparius* are vulnerable to disturbance, but later stages are better able to cope with disturbance, because they live in protective tubes (Table 7.1).

**Table 7.1.** Traits of the two model species determining their species-specific response to some environmental variables.

	<i>Ephoron virgo</i>	<i>Chironomus riparius</i>
habitat	lotic (lentic) sediment	lotic (lentic) sediment
habit	burrower	burrower
generations per year	univoltine (1)	multivoltine (2-7)
metamorphosis	hemimetabolous	holometabolous
growth rate	low	high
functional feeding	collector	collector
trophic level	detritivore	detritivore
food demand	low	high
substrate preference	loamy to sandy, low organic	silty, organically enriched
oxygen requirement	high	low
resistance to toxicants	low, resistant to some (e.g. Zn)	intermediate, genetic adaptation to metals
response to disturbance	reduced survival and growth rates	reduced survival and growth rates during crowding
response to predation	hiding in tube, sensitive?	hiding in tube, reduced survival when disturbed

#### PERSISTENCE OF BENTHIC INVERTEBRATES IN POLLUTED SEDIMENTS

A direct comparison of *C. riparius* and *E. virgo* tested on the same historically polluted floodplain lake sediments clearly showed a high sensitivity of the mayfly and a certain indifference of the chironomid. This observation accords with the general ranking of these species as 'pollution-sensitive' and 'pollution-tolerant', respectively (DE PAUW & VANNEVEL 1991, BODE 1996). The persistence of *E. virgo* in recently deposited and, therewith, relatively clean

sediments in the main bed of the River Rhine combined with the observations from this thesis, confirm that *E. virgo* is able to persist only in sediments with low levels of sediment-bound toxicants, low food levels, and high availability of oxygen. Their low growth rates with resultant low food demands enable them to complete their univoltine life cycle on such sediments.

Pollution-tolerant chironomid taxa related to *C. riparius* were observed to persist in the historically polluted floodplain deposits (CHAPTER 6). *C. riparius* itself is able to persist in even more extreme environments, such as the River Dommel, which is both organically enriched and heavily polluted with metals (GROENENDIJK et al. 1998). Due to their high growth rates with resultant high food demand they require food rich sediments to complete several life cycles; combined with their tolerance to low oxygen levels they can persist in such sediments where oxygen depletion frequently occurs. Absence of more sensitive competing species and predators may lead to continued mass occurrence of *C. riparius* on such sediments.

The different responses of the two model species to several environmental variables and biotic interactions (Table 7.1) indicate that the long-term persistence of these species under specific conditions is bound to a set of characteristics and is unlikely determined by single traits. Therefore, sediment pollution is likely to act only occasionally as a single selective force reducing the persistence of sensitive species. Yet, this was the case in the present thesis: it was observed that the pollution level in some floodplain lake sediments of the River Rhine is prohibitive for insects with the sensitivity of *E. virgo*. In other cases, however, the combination of conditions is likely to determine the persistence of species in polluted sediments. These combined effects may cause limiting conditions to indirectly promote certain species, as shown here for *C. riparius*. Although sediment pollution drives this species close to intoxication, the high availability of food, caused by the extinction of less tolerant species, enables them to persist very well.

#### **IMPLICATIONS FOR ENVIRONMENTAL MANAGEMENT**

Sediment pollution is currently regulated through simplified sets of chemical standards, not really reflecting the actual risks for biota. The Stimulation Programme on System-oriented Ecotoxicological Research (SSEO) was set up to promote scientific analysis of ecosystems actually responding to complex pollution. The current chemical safety standards are applied here to floodplain lake sediments and compared with observations on benthic invertebrates exposed to these sediments in the laboratory.

Floodplains are currently being reshaped to create landscape conditions that favour biodiversity, river management, and recreation. This is done by lowering large parts of the floodplain and by dredging and enlarging lakes and side channels that are already present in the area (NCR 2000). This enlargement of the floodplain lakes and side channels involves re-exposure of historically polluted sediment layers. So far reconstruction of the floodplains has been accompanied by measures containing the most polluted deposits in certain areas covered with layers of clay to prevent the distribution of this material (NCR 2000). The present study confirms that such protective measures are warranted, since re-exposure of polluted sediments may preclude the persistence and colonization of sensitive benthic species, such as mayflies, and pose a continuous selective force on benthic communities.

In the Netherlands, maximum permissible concentrations (MPC) and negligible concentrations (NC) are used to assess the general quality of sediments in surface waters (CROMMENTUIJN 2000a, 2000b). When concentrations in sediments fall below the MPC the risk of the substance is considered to be tolerable. Concentrations in the environment below which the occurrence of adverse effects is considered to be negligible are called NCs.

MPCs are based on the log-logistic distribution of the median effect concentrations and no observed effect concentrations (for acute and chronic toxicity, respectively) of at least four species from different taxonomic groups. MPCs are assumed to be protective for 95% of the aquatic species but their reliability depends on the number and quality of ecotoxicological data (CROMMENTUIJN 2000a, 2000b). Due to the lack of ecotoxicological data for a large part of sediment-bound toxicants, MPCs for sediment are often derived from MPCs in water by applying a modified equilibrium partitioning method (EqP; DiTORO et al. 1991) standardized for sediments with 10% organic matter and 25% lutum (CIW 2000).

The negligible concentrations (NC) of toxicants are based on the MPCs divided by a factor 100 and for metals this value is added up to the background concentration. This safety factor is applied to take into account effects of several substances present in the environment or biological interactions not included in the derivation of the MPC (CROMMENTUIJN 2000a, 2000b).

Deriving sediment quality guidelines (SQGs) solely based on chemical standards is controversial because of the many uncertainties in predicting the degree of adverse biological effects caused by sediment-associated contaminants. For instance, corrections of total concentrations accounting for a standard composition of 10% organic matter and 25% lutum may compromise the bioavailable fractions (VINK et al. 1999). For example, aging processes may

reduce the bioavailable fraction of the organic contaminants (LANDRUM et al. 1992), the speciation of metals to sulfides may reduce their bioavailability (DITORO et al. 1991) and also bioturbating may alter the sorption equilibrium of the toxicants to sediments (GRANELI 1979a, PETERSEN et al. 1995, this thesis). Furthermore, the effect of temporal changes of environmental variables, e.g. oxygen or food quality that modify the sensitivity of organisms for toxicants are not accounted for.

**Table 7.2.** Standardized concentrations (25% lutum, 10% organic matter) of substances in the floodplain lake sediments of the River Waal measured in September 2000 (KOELMANS & MOERMOND 2000) and sediment quality measures for freshwater sediments (CIW 2000). Metals and  $\Sigma$ PAHs in mg/kg dw, PCBs in  $\mu$ g/kg dw. NC= negligible concentration; MPC = maximum permissible concentration. Ant= anthracene; Phe = phenantrene; Flu = fluoranthene; B[a]A = benz[a]anthracene; Chr = chrysene; B[k]F = Benzo[k]fluoranthene; B[a]P = benzo[a]pyrene; B[ghi]P = benzo[ghi]perylene; and Ind = indenopyrene.

	sampling locations							standards	
	D2	G1	G3	O2	3A	3B	D4	NC	MPC
metals									
Cd	0.26	0.52	0.97	1.20	1.50	2.27	1.77	0.80	12
Cu	20	26	31	41	55	67	57	36	73
Zn	71	146	404	259	424	424	542	140	620
PAHs									
Ant	0.01	0.09	0.06	0.05	<b>0.14</b>	<b>0.15</b>	<b>0.36</b>	0.001	0.10
Phe	0.03	0.04	0.12	0.21	0.37	0.45	<b>0.89</b>	0.005	0.50
Flu	0.07	0.08	0.22	0.38	0.82	0.87	1.59	0.03	3.0
B[a]A	0.05	0.04	0.11	0.19	<b>0.45</b>	<b>0.44</b>	<b>0.88</b>	0.004	0.40
Chr	0.04	0.05	0.11	0.20	0.45	0.45	0.82	0.11	11
B[k]F	0.03	0.04	0.07	0.11	0.27	0.27	0.47	0.02	2.0
B[a]P	0.06	0.05	0.14	0.22	0.56	0.53	1.01	0.03	3.0
B[ghi]P	0.03	0.04	0.11	0.19	0.40	0.40	0.61	0.08	8.0
Ind	0.04	0.31	0.14	0.18	0.52	0.48	0.90	0.06	6.0
PCBs <sup>A</sup>									
PCB <sub>28</sub>	b.d.	b.d.	0.89	0.85	1.85	3.42	3.10	1.0	4.0
PCB <sub>52</sub>	b.d.	1.09	1.81	2.01	<b>6.40</b>	<b>11.46</b>	<b>10.25</b>	<b>1.0</b>	4.0
PCB <sub>101</sub>	2.45	1.09	2.79	<b>4.02</b>	<b>12.13</b>	<b>20.62</b>	<b>16.93</b>	4.0	4.0
PCB <sub>118</sub>	1.67	1.33	1.16	1.67	<b>6.11</b>	<b>11.79</b>	<b>9.52</b>	4.0	4.0
PCB <sub>138</sub>	3.27	1.83	3.87	<b>5.96</b>	<b>15.72</b>	<b>24.67</b>	<b>21.28</b>	4.0	4.0
PCB <sub>153</sub>	0.74	b.d.	0.64	1.00	3.11	5.07	4.29	4.0	4.0
PCB <sub>180</sub>	2.90	1.26	2.66	<b>4.19</b>	<b>11.79</b>	<b>18.50</b>	<b>19.04</b>	4.0	4.0

<sup>A</sup> Sediment quality measures for PCBs are not based on hazard limits

In the Fourth National Policy Document on Water Management it was proposed that biological effect studies (bioassays) are to be considered for setting environmental sediment quality objectives (CIW 2000, MAAS et al. 2003). Whole-sediment bioassays can be performed relatively rapid and simple and take into account the bioavailable fractions of all present toxicants. However, the organisms used in whole-sediment bioassays must have the appropriate sensitivity in order to assess the ecological risk of sediment pollution on the benthic community. The standard laboratory test organisms used in whole-sediment bioassays: oligochaetes, amphipods, and chironomids (e.g. US.EPA 2000, OECD 2001) are relatively insensitive to polluted sediments. These test organisms, however, can be ranked as 'pollution tolerant' (DE PAUW & VANNEVEL 1991, BODE 1996). Derivation of SQGs from whole-sediment bioassays performed with 'pollution-tolerant' organisms may not be protective for more sensitive species and thus for benthic communities.

**Table 7.3.** Assessment of sediments with results from whole-sediment bioassays using the effect-classification method according to the TRIAD-method (MAAS et al. 1993).

	D2	G1	G3	O2	3A	3B	D4
Exceeded MPCs	no	no	no	yes	yes	yes	yes
Exceeded NCs	yes						
Effect on survival (%) <i>E. virgo</i> <sup>A</sup> (10 d)	-	-	±	±	+	+	+
Effect on survival (%) <i>C. riparius</i> <sup>A</sup> (28 d)	-	-	-	-	-	-	-
Mentum deformities (%) <i>C. riparius</i> <sup>B</sup> (10 d, lab)	-	-	+	+	+	+	+

<sup>A</sup> Effect classification: - = ≤ 10% mortality (no effect); ± = 10% - 50% mortality, when significant difference from control (moderate effect); + = ≥ 50% mortality, when significant difference from control (severe effect) (MAAS et al. 1993).

<sup>B</sup> Effect classification: - =  $X^2 \leq 7.446$  (no effect); + =  $X^2 \geq 11.479$ , when significant difference from control (severe effect). Deformities are compared to a fictitious sample of 100 counts with 9% deformities (MAAS et al. 1993).

Comparing standardized sediment-bound toxicant concentrations with current SQGs indicate that in some sediments tested in the present thesis, the MPCs for some PAHs are exceeded (Table 7.2), whereas in all sediments several NCs are exceeded, especially those for PAHs. This indicates that only in three of the seven sediments the concentrations of substances are in ranges that might lead to biological effects. Minor exceedance of the MPC might lead to an increased probability for biota to respond. In the sediments in which several MPCs are exceeded survival and growth rates of the mayfly *E. virgo* were severely affected (Table 7.3). And although no effects were observed on survival rates of *C. riparius* larvae in chronic whole-sediment bioassays, a high incidence of

mentum deformities was observed in the sediments that exceeded MPCs for some substances (Table 7.3). Thus it has been shown in the present thesis that concentrations of sediment-bound toxicants close to MPC can be responsible for distinct effects on benthic invertebrates. The mortality of the mayflies and the elevated incidence of mentum deformities of the midge were clearly observed also in some sediments with concentrations of toxicants below MPCs. Consistently, sensitive benthic invertebrate species, such as mayflies and caddisflies, were observed only in one of the investigated floodplain lakes. The present study provides evidence that current MPCs may still pose a detectable ecological effect at least for sensitive benthic invertebrates.