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The use of fish parasites as bioindicators of heavy metals in aquatic ecosystems: a review

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Abstract

Parasites are attracting increasing interest from parasite ecologists as potential indicators of environmental quality due to the variety of ways in which they respond to anthropogenic pollution. In environmental impact studies certain organisms provide valuable information about the chemical state of their environment not through their presence or absence but instead through their ability to concentrate environmental toxins within their tissues. Free living invertebrates, notably bivalve molluscs, are commonly employed in this role as 'sentinel organisms' to monitor the concentrations of bioavailable metals in aquatic ecosystems. Also certain parasites, particularly intestinal acanthocephalans of fish, can accumulate heavy metals to concentrations orders of magnitude higher than those in the host tissues or the environment. The comparison of metal accumulation capacities between acanthocephalans and established free living sentinel organisms revealed significantly higher concentrations of several elements in *Acanthocephalus lucii* (Müller) than in the Zebra mussel *Dreissena polymorpha* (Pallas) which is a commonly used bioindicating organism in Europe. In contrast to the high heavy metal concentrations recorded in adult acanthocephalans, the larval stages in their respective crustacean intermediate hosts show little tendency to accumulate metals. A number of experimental studies demonstrate a clear time dependent accumulation of lead for acanthocephalans in their final hosts. These investigations provide evidence that the extremely high metal concentrations in intestinal acanthocephalans of fish are not the result of a slow process of accumulation but instead a relatively rapid uptake to a steady-state level. Thus, metal concentrations in adult acanthocephalans respond rapidly to changes in environmental exposure of their hosts. The value of parasites for environmental monitoring will be discussed in detail in the present article.

Introduction

Knowledge of fish parasites is of particular interest in relation not only to fish health but also to understand ecological problems. Although the majority of parasitological papers have dealt with parasites as a threat for the health of fish (e.g., Roberts, 1989; Schäperclaus, 1990), several hundred papers have been published since 1980 that are directly concerned with the relationship between pollution and parasitism in the aquatic environment (reviewed by Khan & Thulin, 1991; Poulin, 1992; Vethaak & ap Rheinallt, 1992; Overstreet, 1993; MacKenzie et al., 1995; Lafferty, 1997; Kennedy, 1997a; Sures et al., 1997a; Valtonen et al., 1997; Sures et al., 1999a). This increasing interest especially in fish parasites is related with the high number of parasite species commonly found in or on freshwater and marine fish. As much as 30,000 helminth species were assumed to be parasites of fish (Williams & Jones, 1994). Looking for example for the parasites of eels, *Anguilla anguilla* (L.) and *A. rostrata* (Lesueur), up to 27 different parasite species have been determined from the respective collectives being investigated around the world (e.g., Køie, 1988; Kennedy, 1995; Marcogliese & Cone, 1996; Kennedy et al., 1996; Kennedy, 1997b; Schabuss et al., 1997; Kennedy et al., 1998; Sures et al., 1999b). Eels sampled in the river Rhine in 1995 showed a total of

9 different species of helminths on and in the organs of the fish (see Table 1). The gills of nearly 50% of the eels were infected with the monogeneans *Pseudodactylogyrus bini* (Kikuchi) and/or *P. anguillae* (Yin & Sproston). The intestine of the eels was found to contain the richest parasite community as six different helminth species were recorded for this microhabitat. Among these parasites, acanthocephalans were the most prevalent worms with *Paratenuisentis ambiguus* (Van Cleave) as the dominant species of the intestinal community. But even the swimbladder was heavily parasitised. 84% of the investigated eels were infected with either larvae and/or adults of the nematode *Anguillicola crassus* (Kuwahara, Niimi & Itagaki).

Another interesting ecological aspect of this eel parasite community is that among the 9 metazoan parasites 4 of them were introduced into German eel populations within recent years from different regions of the world. Among these introduced parasite species the swimbladder nematode *A. crassus* has attained increasing interest due to its harmfulness. A number of studies was initiated to investigate the life cycle (e.g., Moravec & Konecny, 1994, Knopf et al., 1998; Sures et al., 1999c), the distribution (e.g., Würtz et al., 1998; Barse & Secor, 1999) and the severe pathological effects (e.g, Würtz et al., 1996; Würtz & Tarschewski, 2000) caused by this parasite.

The above mentioned examples demonstrate the great importance and overall presence of parasites in aquatic ecosystems. Due to their wide abundance and distribution several researchers started to focus on the use of parasites as indicators of environmental quality. Parasites are also attracting increasing interest as environmental indicators due to the variety of ways in which they respond to anthropogenic pollution. The majority of recent investigations have examined the effects of various forms of pollution on the abundance and distribution of parasites. But nowadays there is also an increasing number of papers dealing with the accumulation of toxins within parasites. Thus, the possible use of parasites as bioindicators can be subdivided into the two groups ‘effect indicators’ and ‘accumulation indicators’ as it is commonly known for conventional free living bioindicating organisms (Gunkel, 1994).

Effect indication with parasite populations

In environmental impact studies certain organisms provide valuable information e.g. about the chemical, physical, biological and ecological state of their environment through their presence or absence. Changes in the diversity and structure of parasite communities of different fish hosts have therefore received increasing attention due to the possible application of parasites as indicators of ecosystem integrity and health (MacKenzie et al., 1995; Kennedy, 1997a, b; Valtonen et al., 1997; Dušek et al., 1998). An interesting way to try to perform effect indication with parasites may be the use of monogenean trematode populations on the gills of fish (Koskivaara, 1992; Bagge & Valtonen, 1996; Dušek et al., 1998). Monogenean trematodes are ectoparasitic and therefore in direct contact with both the surrounding environment and the fish host. They are common worms on the gills of fish with short life cycles and are thus capable to react immediately on changes in environmental factors. There are several studies in which the composition of gill monogenean communities of fish are associated with different forms of pollution like paper mill effluent (e.g., Siddall et al., 1997) or other environmental factors (e.g., Koskivaara, 1992). *Dactylogyrus* species were found to show higher abundance and species diversity on the gills of roach, *Rutilus rutilus* (L.), from a lake in central Finland receiving effluent from a pulp and paper mill when compared to an uncontaminated reference lake (Koskivaara et al., 1991; Koskivaara & Valtonen, 1992). However, contradictory evidence of a decrease in ectoparasite infections associated with pulp and paper mill pollution was also provided (reviewed in Khan & Thulin, 1991). An influence of water pollution was recently demonstrated by the distribution of species abundances within communities of *Dactylogyrus* and *Paradiplozoon* (Dušek et al., 1998). Assemblages of host specific monogeneans in a polluted site exhibited a significantly reduced species richness and unequal distribution of abundances. The opposite pattern was observed in the case of generalists, parasitizing a broad range of possible hosts (Dušek et al., 1998). Therefore, monogenean parasites and their diversity appear to be a sensitive and meaningful model for environmental studies. By using monogenean populations one do not analyse only slight changes in the physiology or the behaviour of a single test organism but instead changes of the abundance of a particular parasite species were recorded. Thus, changes of the whole population structure de-

Table 1. Prevalence (P) in percent, mean intensity (MI) (± SD), and abundance (A) (±SD) of the parasites of eels ($n = 61$) from the river Rhine[1]

Parasite species	Parasite group	Site of infection	P	MI	A
Anguillicola crassus	Nematoda	Swimbladder	83.6	5.3 (4.9)	4.4 (4.9)
Paratenuisentis ambiguus	Acanthocephala	Intestine	39.3	41.6 (43.9)	16.4 (34.0)
Acanthocephalus lucii	Acanthocephala	Intestine	1.6	5.0[2]	0.1 (0.6)
Acanthocephalus anguillae	Acanthocephala	Intestine	1.6	1.0[2]	0.0 (0.1)
Pomphorhynchus laevis	Acanthocephala	Intestine	6.6	14.0 (14.4)	0.9 (4.7)
Raphidascaris acus	Nematoda	Intestine	6.6	3.3 (1.3)	0.2 (0.9)
Bothriocephalus claviceps	Cestoda	Intestine	4.9	3.7 (0.6)	0.2 (0.8)
Pseudodactylogyrus sp.	Monogenea	Gills	45.9	n.d.[3]	n.d.

[1]: data from Sures et al., 1999b.
[2]: only one eel infected.
[3]: n.d.: number of parasites not determine.

pending on the pollution of the environment could be monitored. This is in contrast to conventional free living organisms like e.g. the Zebra mussel *Dreissena* polymorpha or Rainbow trout, *Salmo gairdneri* (Rich.), which are used as effect indicators for water treatment in sewage plants (Gunkel, 1994). However, interactions between the environment and host-parasite systems are complex under natural conditions and not easily interpreted as they are dependent on a wide variety of factors (Kennedy & Guégan, 1996; Kennedy, 1997a). Lafferty (1997) drew attention to the conflicting evidence on and inconsistent associations between environmental impacts and parasites, concluding that few parasite-pollution combinations show predictable changes despite the considerable effort that has been put into linking levels of parasitic infection with pollution. It remains difficult to judge which single factor or combination of factors out of a set of 'anthropogenic factors' affects or determines the diversity of the parasite community in fish. Thus, although studies on diversity of fish parasites in different biotopes are important and extremely interesting, they do not allow any conclusions to be drawn concerning the concentration of specific toxins in the environment. But by using parasites as accumulation indicators they have also alternative applications for environmental impact studies.

Heavy metal accumulation in intestinal helminths

In addition to different ways of effect indication the parasite's ability to concentrate environmental toxins within their tissues may also be advantageous for environmental monitoring proposes. Until now, different helminth species have been investigated in respect of their heavy metal accumulation capacity (see Sures & Taraschewski, 1999; Sures et al., 1999a). The most promising parasites are acanthocephalans, a group of intestinal worms, commonly found in fish. For example the parasite community of eels from the river Rhine revealed up to four different acanthocephalan species occurring simultaneously in the host (Table 1). Although these parasites are very abundant in aquatic biotopes, they were seldom considered by aquatic ecologists in environmental impact studies.

Biology of acanthocephalans

Acanthocephalans are widely distributed intestinal worms often parasitizing fish (Figure 1) but also other vertebrates including mammals were used as definitive hosts. Adult worms live inside the intestine of the final host and absorb their nutrients across their tegument as the worms are lacking an own mouth and intestine. The embryonated eggs of the female acanthocephalans reach the water with the gut content of the host. The eggs which already contain the first larvae, called acanthors, were eaten by a crustacean intermediate host. The acanthor will hatch inside the intestine of the crustacean and develop in the hemocoel of the host into the cystacanth. This larvae will be infective for the final fish host and is taken up orally by a fish while feeding on infected crustaceans (Figure 1).

Heavy metal accumulation has been investigated so far using three different acanthocephalan species: *Pomphorhynchus laevis* (Müller), *Acanthocephalus lucii* and *Paratenuisentis ambiguus*. Different species of the amphipod genus *Gammarus* were found to be intermediate hosts for *P. laevis* (Rumpus & Kennedy,

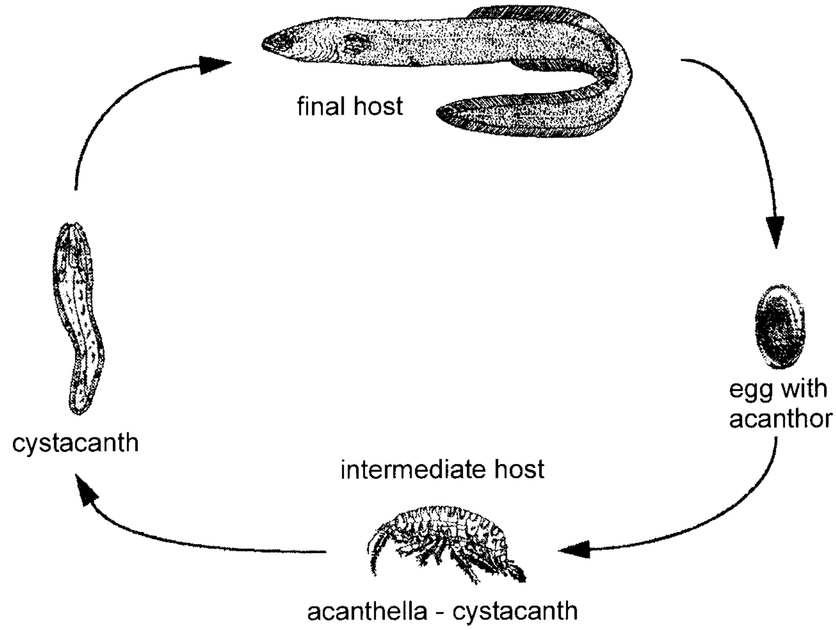

Figure 1. Life cycle of acanthocephalans.

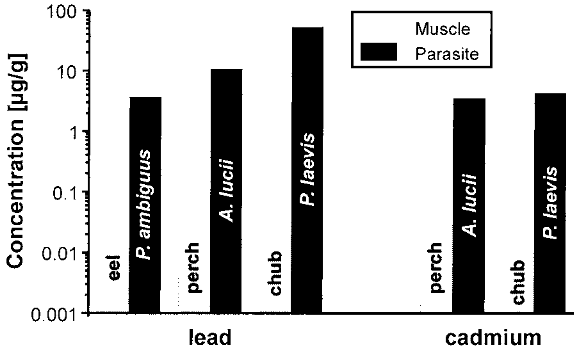

Figure 2. Metal concentrations in acanthocephalans compared to the muscle of the final fish host (data from Sures et al., 1994a, b, c; Sures & Taraschewski, 1995) eel (*Anguilla anguilla*, perch (*Perca fluciatilis*) and chub (*Leuciscus cephalus*).

1974), whereas *P. ambiguus* uses only *Gammarus tigrinus* (Sexton) (Samuel & Bullock, 1981) and *A. lucii* needs *Asellus aquaticus* (L.) (Lee, 1981) as an appropriate intermediate host. The final hosts of *P. ambiguus* are American and European eels (Samuel & Bullock, 1981; Taraschewski et al., 1987). The range of possible final hosts for *A. lucii* and *P. laevis* is much broader comprising different perciform and anguilliform species for *A. lucii* (Lee, 1981) and cypriniform, salmoniform and anguilliform species for *P. laevis* (Kennedy et al., 1978, 1989).

Field studies on metal accumulation by acanthocephalans

Different naturally infected fish species sampled from the field were analysed for their metal concentrations in comparison to the respective levels of their acanthocephalans (Figure 2). The highest metal accumulation rates described so far were found for chub, *Leuciscus cephalus* (L.) infected with *Pomphorhynchus laevis*. The mean concentrations of lead and cadmium in *P. laevis* were, respectively, 2700 and 400 times higher than in the muscle of the host and 11,000 and 27,000 times higher than in the water (Sures et al., 1994a; Sures & Taraschewski, 1995). The lowest accumulation capacity was found for *Paratenuisentis ambiguus* (Sures et al., 1994b) which resembled that of cestodes dwelling in the intestine of freshwater fish (Riggs et al., 1987; Sures et al., 1997b; Tenora et al., 1997). Anyway, all acanthocephalans analysed so far were found to contain significantly higher amounts of metals than the host tissues (see Sures et al., 1999a). This is opposed to the results of metal accumulation in the swimbladder nematode *Anguillicola crassus* that showed no (Sures et al., 1994b; Zimmermann et al., 1999a) or a rather poor metal bioconcentration (Tenora et al., 1999). In contrast to the high heavy metal concentrations recorded in adult acanthocephalans, the larval stages in their respective crustacean intermediate hosts show little tendency to accumulate metals

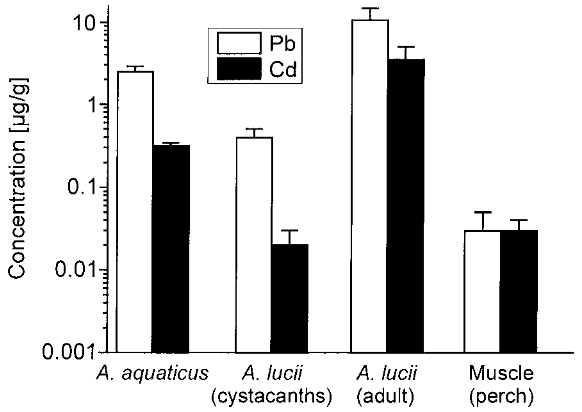

Figure 3. Metal concentrations in larval and adult *Acanthocephalus lucii* as compared to their intermediate and final host *Asellus aquaticus* and *Perca fluciatilis*, respectively (data from Sures et al., 1994b; Sures & Taraschewski, 1995).

(Brown & Pascoe, 1989; Sures & Taraschewski, 1995; Siddall & Sures, 1998). A comparative study was carried out on larval *Acanthocephalus lucii* from naturally infected *Asellus aquaticus* and adult *A. lucii* in perch, *Perca fluviatilis* (L.) from the same biotope (Figure 3). Compared to the larval worms the adults of *A. lucii* contained approximately 30 times more lead and 180 times more cadmium although metal levels were higher in the intermediate host than in the final host. In another field study cadmium has been detected at approximately the same concentration in cystacanths of *P. laevis* as in its naturally-infected crustacean host *Gammarus pulex* (L.) (Brown & Pascoe, 1989). Therefore the main uptake and accumulation of metals occur in adult worms inside the gut of the final host and not in the larvae located in the hemocoel of the crustacean intermediate host.

This enormous heavy metal bioconcentration capacity provides evidence that adult acanthocephalans may in fact be valuable as accumulation indicators for heavy metals. Interestingly, recent field studies have now demonstrated that acanthocephalans can accumulate toxic metals from the aquatic environment to concentrations even surpassing those in *Dreissena polymorpha* (Sures et al., 1997c, 1999d). This free-living animal generally recognised as a passive (i.e. naturally present) or active (i.e. transplanted) biomonitor is one of the best established accumulation indicators in fresh and brackish waters of Europe (Reeders et al., 1993; Stäb et al., 1995) and the USA (Doherty et al., 1993). A comparison of several different elements between organs of perch, its acanthocephalan *A. lucii* and *D. polymorpha* from Lake Mondsee in Austria revealed for most element concentrations significantly lower values in *D. polymorpha* than in *A. lucii* (Sures et al., 1999d; Figure 4). Only barium was found in higher levels in the mussel than in *A. lucii* which may be related with physiological properties of the mussel. Some elements were higher in the parasite but not significantly different from the mussel (Al, Co and Ni). The levels of all other elements detected within the soft tissue of *D. polymorpha* were significantly lower than in the parasite (Figure 4). The higher accumulation rates of nearly all elements in *A. lucii* resembled results of an earlier investigation where significantly higher lead and cadmium concentrations were described in the acanthocephalan than in the mussel at two different sampling sites from lake Mondsee, Austria (Sures et al., 1997c). However, there is a higher degree of variability among the metal burden of the parasites than among individual mussels (Sures et al., 1997c). This variability, which may reflect the mobility of the fish host, can obscure the differences that might otherwise be detected between sites. For this reason it has been suggested that Zebra mussels are more suitable for detecting localized differences in contamination than the parasites (Sures et al., 1997c). Despite this, acanthocephalans can also provide ecologically valuable information on the average exposure of a mobile fish host within its natural range. Especially due to the great impact of *D. polymorpha* on biotopes in having free-swimming veliger larvae which numerically dominate the benthic community (Garton & Haag, 1993) it seems reasonable to investigate other common aquatic organisms in respect of their bioindicating properties like e.g. acanthocephalans which appear to be even more suitable to accumulate metals than the Zebra mussels (Sures et al., 1997c, 1999d).

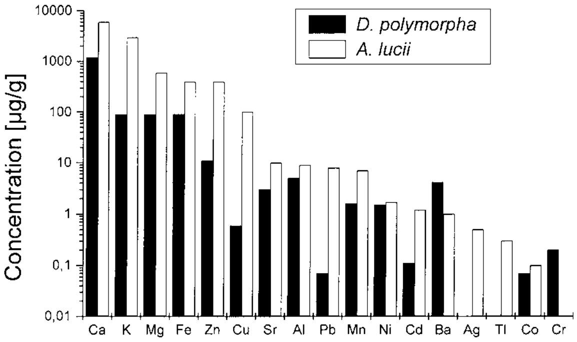

Figure 4. Element concentrations in *Dreissena polymorpha* and *Acanthocephalus lucii* sampled at the same station in lake Mondsee, Austria (data from Sures et al., 1999d).

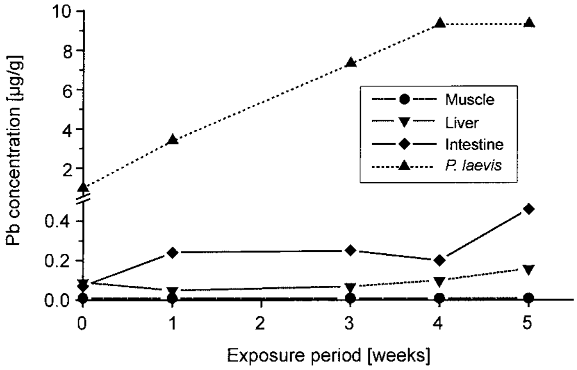

Figure 5. Uptake of lead by chub (*Leuciscus cephalus*) experimentally infected with *Pomphorhynchus laevis* (data from Sures, 1996).

But also considering the tissues of the final host perch, most elements were significantly concentrated in the parasites (Sures et al., 1999d). Ni was present in approximately the same concentrations in all tissues of perch and in its intestinal parasite. Cr was the only element which could not be determined in *A. lucii* because the values were below the detection limit. The acanthocephalan showed significantly higher levels for all elements except for Fe and Co when compared to the liver of its host, the latter even being significantly higher in perch liver than in the parasite. Although the values for Co and Mn in the worm exceeded those in the intestine of the fish, these were the only elements being not significantly higher in *A. lucii* than in the intestine. The bioconcentration of elements in *A. lucii* compared to the host intestinal wall listed in order of decreasing values can thus be given as follows: Cu = Ag > Pb > Tl > Cd > Sr > Ca > Ba > Zn > Fe > Ni > Al = Mg > Co = Ga = Mn. Spearman correlation analysis revealed that the concentrations of several elements within the parasites decreased with an increasing infrapopulation. Furthermore, the levels of some elements in the perch liver were negatively correlated with the weight of *A. lucii* in the intestine. Thus, it emerged that there is competition for elements not only among the acanthocephalans inside the gut but also between the host and the parasites. This competition for elements between the host and the parasites might have an important impact on the amount of elements accumulated in the host's tissues (see also next chapter).

Experimental studies on metal accumulation by acanthocephalans

The phenomenon of heavy metal bioconcentration in acanthocephalans derived from field studies was supported by experimental data obtained from the two different acanthocephalans: *Pomphorhynchus laevis* and *Paratenuisentis ambiguus*.

In a series of lead exposure studies chub were experimentally infected with cystacanths of *P. Laevis* and afterwards exposed to lead in the water at different concentrations and for different periods of time (Sures, 1996; Siddall & Sures, 1998; Sures & Siddall, 1999). Results of a five weeks exposure of infected fish to an aqueous lead concentration of 10 μg/l are summarised in Figure 5. It is obvious that the accumulation kinetics differed markedly between the parasite and the tissues of chub. After an exposure period of about 4–5 weeks there seems to be a steady state in *P. laevis* while there are still increasing levels of lead in liver and intestine of chub. Concerning the lead level of chub muscle no lead uptake occurs due to the low exposure concentration. Comparing the lead burdens in the parasite with that of host's muscle about 1000 times higher concentrations were found for the worms. Thus, it emerges that the metal concentration in the parasite is likely to respond rapidly to changes in environmental exposure. Furthermore, as the acanthocephalans are able to accumulate such high amounts of lead in a very short time it seems likely that the lead uptake by *P. laevis* affects the lead burden in the organs of chub. To investigate such an influence of *P. laevis* on the host tissues, infected versus uninfected chub were exposed to lead for a period of five weeks (Sures & Siddall, 1999). Regarding the accumulation kinetics this exposure period should be long enough for the worms to reach the steady state. Although infected and uninfected fish exposed to lead accumulated approximately the same amount of the metal in the liver, the presence of acanthocephalans had a significant impact on lead accumulation in the intestinal wall, which was for the infected fish only half that of uninfected chub (Figure 6). Thus, the presence of acanthocephalans in the intestine of the final host reduces the lead concentration in the intestinal wall.

Sures & Siddall (1999) presented a hypothesis how *P. laevis* may get access to lead in the intestine of chub (Figure 7). With the permanent osmotic inflow of water across the gills of freshwater fish lead ions are able to pass across the epithelial membrane by paracellular diffusion and enter the bloodstream (Hofer &

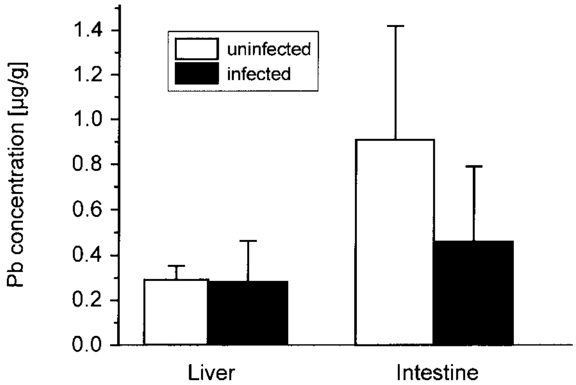

Figure 6. Lead concentrations in liver and intestine of infected and uninfected chub (*Leuciscus cephalus*) (data from Sures & Siddall, 1999).

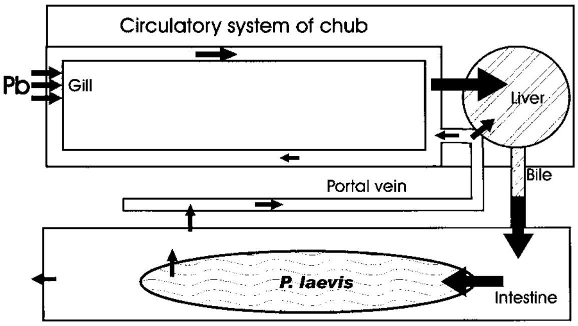

Figure 7. Schematic diagram showing the uptake, transport, excretion and entero-hepatic cycling of lead in chub (*Leuciscus cephalus*) and the route of uptake of the metal by intestinal acanthocephalans, *Pomphorhynchus laevis* (Sures & Siddall, 1999).

Lackner, 1995; Hodson et al., 1978). After binding to the membrane of erythrocytes they are transported by the circulatory system in the liver where the majority of lead is removed from the blood and excreted into the intestine via the bile (Hofer & Lackner, 1995; Grahl, 1990). The bile contains steroids with which the heavy metal ions form organometallic complexes that then pass down the bile duct into the small intestine (Hofer & Lackner, 1995; Grahl, 1990). In the small intestine these organometallic complexes can either be reabsorbed by the intestinal wall and run through the hepatic-intestinal cycle or they can be excreted with the faeces of the fish. But the production of bile by the host is also extremely important for acanthocephalans due to their inability to synthesise their own cholesterol and fatty acids (Barrett et al., 1970). Combining these aspects Sures & Siddall (1999) concluded that organometallic complexes were taken up by the acanthocephalans in the small intestine concurrently with bile salts. Due to the efficiency of acanthocephalans in taking up bile salts, the concentration of bile-bound lead in the intestinal lumen of infected fish, and hence the amount which could be reabsorbed by the intestinal wall, will be markedly reduced compared to uninfected conspecifics. Thus, the parasites are able to reduce or even interrupt the hepatic-intestinal cycling of lead as the worms and the intestinal wall of the fish compete for the bioavailable heavy metals. The reduced lead levels in the intestinal wall of infected fish compared to uninfected clearly support the indication of competition between hosts and parasites derived from correlation analysis of data from naturally infected, unexposed fish (Sures et al., 1999d; see also previous chapter in this issue).

The essential role of bile in the uptake of lead by parasites proposed here is confirmed by in vitro studies on the lead uptake of *P. laevis* larvae (Sures & Siddall, 1999) and also by a study on lead accumulation in the liver fluke *Fasciola hepatica* (L.) (Sures et al., 1998) inhabiting the bile ducts of cattle. This trematode accumulated lead to concentrations 172 and 115 times higher than in muscle and liver respectively, of its bovine definitive host. Except for the liver fluke being surrounded by bile liquid, the essential role of bile salts for metal uptake by intestinal parasites could also help to explain that the ability to bioconcentrate metals appears to be closely linked to the intestinal location of the parasites. Metal concentrations lower than in the host have been detected in intraperitoneal cestode plerocercoids (Pascoe & Mattey, 1977) and also in the acanthocephalan *P. laevis* from the body cavity of experimentally infected goldfish (Sures, 1996).

Although there are comparatively few data for acanthocephalans of estuarine or marine fish compared to their freshwater counterparts preliminary studies on the acanthocephalan *Echinorhynchus gadi* (Müller) from North Sea cod, *Gadus morrhua* (L.), have recorded lead concentrations which suggest also a remarkable metal accumulation within parasites from marine and brackish environments (Sures et al., 1999a). Nevertheless, little is known about the influence of environmental factors such as water salinity on the heavy metal accumulation capacity of host-parasite-systems. In a recent study on the lead uptake by European eels (*Anguilla anguilla*) infected with *Paratenuisentis ambiguus* (Zimmermann et al., 1999a, b) the influence of water salinity on the bioavailability and the bioaccumulation of lead was investigated depending on the lead application mode (diet versus

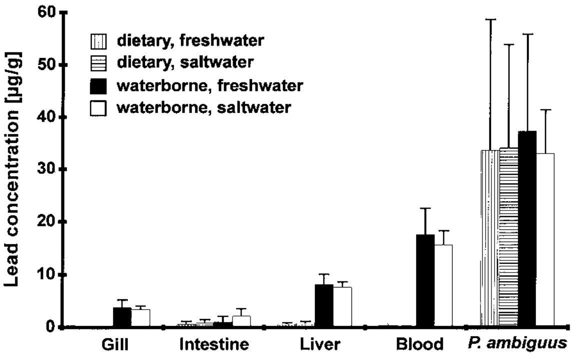

Figure 8. Lead concentrations (µg/g wet weight) in different tissues of eel (*Anguilla anguilla*) and in *Paratenuisentis ambiguus* after experimental exposure (Zimmerman et al., 1999a).

water). *P. ambiguus* is known as an endemic eel parasite of brackish water along the east coast of the USA (Samuel & Bullock, 1981). After its introduction to Europe the parasite dwell the intestine of European eels. Due to the worms origin this host-parasite-system is assumed to be well adapted to a wide range of water salinity.

Lead analysis after a four weeks exposure revealed that the mode of lead application had a significant influence on the distribution and the concentration of lead in the fish tissues (Figure 8). Only a poor metal accumulation was found in the tissues of eels after oral lead application with levels being lower compared to the respective tissue concentrations following aqueous lead exposure. Influences of water salinity on the lead levels in eel tissues were not found. The acanthocephalans showed significantly higher lead concentrations than the host tissues in all exposed eels. There were no significant differences in the lead levels of the parasites neither with water salinity nor with the mode of lead application. As the mode of lead application, as well as the water salinity, did not affect the lead content in the acanthocephalan, intestinal parasites may be used as sentinel organisms to monitor the concentration of bioavailable metals, not only in freshwater but also in estuarine and marine ecosystems. In contrast to the parasites, the lead concentration of different tissues of the fish host varied depending on the mode of metal application. The distribution of the respective metal within the host may help to identify the main pollution source (dietary or waterborne). Thus, the combination of the results obtained from the host and the parasites would reveal a more reliable and detailed tool to ascertain the source of an environmental contamination than a study based on a single species.

Conclusions and perspectives of parasites as accumulation indicators

What deductions could be drawn from the results summarised in this article? It is already know from literature that helminth infestations may affect the host sensitivity to heavy metals. The vitality of fish (Pascoe & Cram, 1977; Pascoe & Woodworth, 1980) and amphipod crustaceans (McCahon et al., 1988; Brown & Pascoe, 1989) aqueously exposed to cadmium was markedly reduced when they were infected with larval cestodes or acanthocephalans, respectively. As larval parasites always have a more severe impact on their intermediate hosts than adult parasites on their final hosts one could expect an increased sensitivity of parasitised intermediate hosts to heavy metal pollution. Although no mortalities have been observed among adult acanthocephalans containing extremely high concentrations of metals, potential sublethal effects on, for example, egg fertility and larval viability remain to be studied. However, the parasites may have their own detoxification mechanisms or take up metals in a form that has already been detoxified by the host. This is in contrast to the extreme metal sensitivity of certain free-living parasite stages like cercariae of digeneans (Evans, 1982; Siddall et al., 1993; Siddall & des Clers, 1994).

In order to evaluate the relationship between environmental exposure and acanthocephalan metal bioconcentration and to validate the role of parasites in environmental biomonitoring it will be necessary to carry out more laboratory studies on experimentally infected fish to determine the ratio between metal concentrations in the parasites at equilibrium and the exposure concentration (bioconcentration factor). Once this is achieved the same host-parasite system could be used to compare environmental contamination between a variety of sites. This would be possible e.g. throughout Europe as several acanthocephalan species are widespread and common parasites of fish. Although their life-span is generally shorter than that of their host their metal concentrations are likely to respond rapidly to changes in environmental exposure. However, parasites are also subject to limitations when compared to routinely used sentinel organisms such as bivalve molluscs. The fish host must be dissected before the parasites can be isolated. Additionally, the total weight of parasites in an individual fish may be relatively small (depending on the acanthocephalan species). But on the other hand it is generally accepted from a public health viewpoint to monitor the heavy metal contents in fish and other aquatic animals taken for human consumption. Therefore, after dissecting fish, their parasites can be easily removed from the intestine. Instead of considering parasites like acanthocephalans while routinely analysing the metal contents of edible parts of fish, free living invertebrates such as mussels or crustaceans which have even lower metal accumulation rates than adult acanthocephalans are commonly sampled from ecosystems and analysed for their metal burden. By using acanthocephalans in environmental impact studies, exceedingly low concentrations of metals can be detected in the water due to the enormous accumulation capacity of these worms. Additionally, the ratio between metal concentrations of the parasites and host muscle tissue could provide information on the duration of environmental exposure as metal uptake occurs more rapidly in the parasites. Relatively high metal levels in both host muscle and parasites (i.e., low ratio) would indicate a longer exposure time than when the metal burden was high in the parasites but low in the muscle (i.e., high ratio).

The role of endoparasites as a sink for heavy metals within a fish host, as demonstrated recently (Sures & Siddall, 1999), has not previously been recognized. Therefore there is a possibility that fish could tolerate much higher exposure concentrations of certain heavy metals than so far assumed (Hofer & Lackner, 1995; Gunkel, 1994; Köck, 1996) if they are infected with adult acanthocephalan parasites which themselves accumulate huge amounts of these metals. In addition, as fish are often naturally infected by a variety of parasites (e.g., Sures et al., 1999b), parasitism needs to be seriously considered in toxicity studies as a potentially important factor influencing accumulation of toxins by fish and subsequent effects on fish health. Finally, a major task might be to convince ecologists about the environmental value of certain endoparasites. Aquatic ecologists and parasitologists should combine their approaches, techniques and expertises.

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