**FACTORS AFFECTING THE ORGANOCHLORINE POLLUTANT LOAD IN
BIOTA OF A RICE FIELD ECOSYSTEM
(EBRO DELTA, NE SPAIN).**

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Abstract

The concentrations of PCBs, DDTs, HCHs, HCB and OCS were determined in sediments and associated biota, both invertebrates (*Physella acuta*, *Hirudo medicinalis*, chironomid larvae, *Hydrous pistaceus*, *Helochaeres lividus*) and vertebrates (*Rana perezi*), in a temporary aquatic system, a rice field in the Ebro Delta (NE Spain). The qualitative and quantitative distribution of organochlorine compounds in sediments and aquatic biota has been explained by two mechanisms: equilibrium partitioning and/or biomagnification through the trophic web. Nevertheless, bioaccumulation processes are by far more complex, since several biotic and abiotic factors contribute to the observed pollutant loads in the organisms. In this respect, the biological characteristics of the organisms considered (e.g. species, age, lipid contents, feeding habits, etc.), as well as ecological factors (e.g. the habitat of the species and vertical distribution), have been shown to account for the organochlorine levels observed.

Key words: POPs, aquatic ecosystem, rice field, bioaccumulation, sediments, invertebrates, vertebrates.

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1. Introduction

In the aquatic environment hydrophobic compounds, such as PCBs and DDTs, tend to be adsorbed onto particulate matter according to a simple partitioning process (Bierman, 1990). On the other hand, the main transfer pathways of these contaminants to aquatic organisms are controlled by an equilibrium partitioning between the external media (water or sediment) and the organism lipid fraction and/or by the food-web constraints (Farrington, 1991, MacDonald *et al.*, 1993, Nowell *et al.*, 1999). However, other factors such as the intrinsic characteristics of the organism biology, ecology or life-history traits as well as the environmental conditions may also affect the accumulation of organic contaminants by aquatic biota. Feeding behaviour, migration habits, breeding season, larval or post-metamorphic conditions, in addition to trophic level, age, or any other circumstances affecting exposure, mobility or changes in the lipid fraction of their bodies, can alter the final concentration of pollutants found in their tissues (Wiederholm, 1984, Barron, 1990, Clark *et al.*, 1990, Cortet *et al.*, 1999, Nowell *et al.*, 1999).

Most of the field studies done until present regarding the compartmental distribution of pollutants in aquatic organisms have been largely focused, on the two aforementioned mechanisms, but the influence of many of such organism characteristics has not been yet properly assessed (see Nowell *et al.*, 1999 for a review). Bierman (1990) has already suggested that ecological variables other than trophic level (biomagnification) should be considered to account for the accumulation of organic pollutants observed in sediment-dwelling organisms, which are actually much higher than those expected by equilibrium partitioning only.

The aim of this paper has been to examine different abiotic and biotic factors influencing the organochlorine load of sediments and organisms, in a temporary aquatic ecosystem, the Ebro Delta rice fields (Northeast Spain).

Rice fields are dynamic systems of particular environmental concern because they exhibit a high capacity to concentrate and deliver pollutants. In fact, they receive large pesticide applications, they are also extensively irrigated with waters which may contain significant amounts of pollutants, and their sediments are periodically re-suspended, enhancing water-biota exchanges (Forés and Comin, 1991; Mañosa, 1997). In this respect, the bioaccumulation processes in shallow-water ecosystems have been found to be one order of magnitude higher than those occurring in deeper waters (MacDonald *et al.*, 1993). On the other hand, rice fields are usually keystone environments for waterbirds; they host large amounts of prey which constitutes the bird's diet, and the pollutant levels could pose a risk for them directly, through poisoning, and indirectly through decline in preys availability (Fasola and Ruiz, 1995, Tourenq *et al.*, 2001).

Previous studies carried out in the Ebro Delta rice fields (Figure 1) have already reported the accumulation of noticeable amounts of organic and inorganic pollutants, namely organochlorine compounds and heavy metals, in birds and other biota feeding-on and/or living in the area (Llorente *et al.*, 1987, Ruiz and Llorente, 1991, Porte *et al.*, 1992, Ruiz *et al.*, 1992, Morera *et al.*, 1997, Ramos *et al.*, 1999). Changes in the biodiversity of the rice field macrofauna attributed to the use of pesticides have also been recorded (Mañosa, 1997). Therefore, a complete understanding of the processes and mechanisms governing bioaccumulation in this singular temporary aquatic

ecosystem is needed in order to prevent losses in wetland biodiversity and other threats concerning waterbird food-shortage and poisoning risks.

2. Experimental

2.1. Organisms studied

The organisms considered in the study and their ecological characteristics are summarized in Table 1.

Invertebrates .- Freshwater gastropods make up a large proportion of benthic animal biomass in variable aquatic habitats and may play a keystone role in the transfer of matter and energy in rice field ecosystems. In the Ebro Delta, *Physella acuta* is one of the most abundant bassommatophoran species. It seems to forage mainly on epiphytic vegetation and adventitious plants (González- Solis and Ruiz, 1996).

Among the Annelids, bloodsuckers (Cl. Hirudinea, *Hirudo medicinalis*) attain a high density in the Ebro delta rice fields and constitute a prey for Cattle Egrets (Ruiz, 1985). Chironomid (diptera) larvae and aquatic coleopterans (Fam. Hydrophilidae: *Hydrous pistaceus*, *Helochaeres lividus*) are among the main insect fauna in the rice field ecosystem. Chironomid larvae can be very abundant and constitute an essential link in the rice field trophic web since they are directly preyed by ducks and waders, and constitute also the main food supply for larger insects and crustaceans later taken by herons and egrets. Gonzalez-Solís *et al.* (1996) reported that ducklings consume chironomids as the main source of protein during growth. In the Ebro Delta, coleopterans (adult and larvae) are the main components of the insect community in terms of biomass . Arthropod preys have been registered in the diet of all the colonial

herons breeding in this area and are the bulk of the biomass consumed by the Cattle egret (Gonzalez-Solis *et al.*, 1996).

Vertebrates .- The green frog (*Rana perezii*) is the commonest amphibian species in the area. Their diet accommodate to changes produced by seasonal dynamics of the rice crop: during the dry period (January-April) terrestrial prey dominate the diet while flying (dipterans) prey are common during the flooding period (May-November) (Jover, 1989). Frogs are the staple food for Cattle egrets but they are also actively predated by other waterbirds (Mañosa, 1997).

2.2. Sampling

Surface sediments (5cm depth) were collected with a grab sampler, placed in glass jars and kept frozen (-20°C) until analysis. Organisms were collected along a rice cultivation cycle, using a biocenometer device (a metallic square of 60 cm by side and 40 cm high) sunk at 5 cm depth in the paddy field soil. All the volume was filtered through a mesh sieve of 1.2 mm, and the organisms were washed, classified and frozen until their analysis. Biocenometer sampling was conducted at five different locations within the area.

Because the size of most of the species collected was too small to provide a sample volume enough for the analysis at the individual level, composite samples including all the individuals from the same species were pooled for each sampling location (n= 5).

2.3. Analysis

Sediments were freeze-dried, sieved at 1mm to remove coarse sands and organic debris and Soxhlet extracted for 12 h with a mixture of *n*-hexane-dichloromethane (4:1). Biota tissues were dried at 50°C to constant weight to determine their water contents and extracted with *n*-hexane in a Soxhlet apparatus. The solvent extracts were fortified with octachloronaphthalene as internal standard, evaporated to near dryness, the residue dissolved in 3 ml of *n*-hexane and cleaned-up by vigorous shaking with 1-2 ml of conc. H₂SO₄. The resultant extract was washed with water, dried over Na₂SO₄ and conveniently diluted with *n*-hexane for GC-ECD analysis. An aliquot of the solvent extract was evaporated for the gravimetric determination of the sample fat contents.

The gas chromatograph (Hewlett-Packard Model 5890, Palo Alto, CA, USA) was equipped with an automatic injector (Hewlett-Packard Model 7673A) and a ⁶³Ni electron capture detector. The column, a 30 m DB-5 fused silica (J&W Scientific, Folsom, CA, USA), was programmed from 60 to 300° C at 6° C·min⁻¹, the final temperature being maintained for 10 min. Helium was used as carrier gas. The injector and detector were set at 270 °C and 300 °C, respectively.

Seven PCB congeners (IUPAC no. 28, 52, 101, 118, 138, 153 and 180, purchased from PROMOCHEM, Wesel, Germany) were selected for quantitative analysis of PCBs, following the criteria of International Monitoring Programs (Tuinstra *et al.*, 1985). The analysis of DDT encompassed the determination of pp'-DDT, pp'-DDD and pp'-DDE by comparison with calibrated solutions of authentic standards as well as for the

hexachlorocyclohexanes (α -HCH and γ -HCH), hexachlorobenzene (HCB) and octachlorostyrene (OCS), all them purchased from PROMOCHEM, Wesel, Germany.

Rank order statistics for independent data, Kruskal-Wallis (χ^2_{K-W}) and Mann-Whitney (W_{M-W}) tests, were used to assess differences in total organochlorine concentration among groups of organisms. Sequential Bonferroni correction was applied in pairwise comparisons to ensure that overall alpha error was maintained at desired 0.05.

3. Results

Concentrations of organochlorines in the rice field sediment and biota are given in Table 2, both in dry weight and in fat basis (ng.g^{-1} dw). Significant differences in total organochlorine concentrations were found among organisms ($\chi^2_{K-W}=23.7$; d.f.=8; $p<0.01$). The lower concentrations were generally found in the imago of *Hydrous pistaceus*, an aquatic herbivore. For OCS, the levels in sediments were relatively higher than those shown by some organisms, as it is the case for *H. pistaceus* and *H. lividus*. The imago of *H. lividus* showed the highest organochlorine levels, excluding OCS.

Filter-feeders organisms, midges (chironomidae larvae) and tadpoles of *Rana perezi*, did not show statistically significant differences in total organochlorine concentrations (fat and d.w.: $W_{M-W}=18$; n.s.). Amongst the herbivores, *P. acuta* showed levels higher than those of *H. pistaceus*, both in fat and dry weight basis (fat: $W_{M-W}=15$; $p<0.05$; d.w.: $W_{M-W}=15$; $p<0.05$). Carnivore and detritivore organisms usually reached to higher accumulation levels (Figure 2), although differences can be observed according to the family of compounds considered. DDTs and PCBs predominate in all species and they represent more than 75% of the total compounds found (Figure 3). In general, DDTs are

by large the predominant components, although insect species (chironomid larvae and *H. pistaceus* and *H. lividus*) exhibit a relatively higher proportion of PCBs.

The distribution of the different DDT metabolites (pp'-DDT, pp'-DDD and pp'-DDE) is shown in Figure 4. pp'-DDT represented only about 7% of DDTs in sediments, which also contained the higher proportion of pp'-DDD. These percentages were lower in biota, except in the larvae of *H. pistaceus*. pp'-DDT was even not detected in many of them, but the relative amount of pp'-DDD was higher in snails and tadpoles.

The distributions of PCBs were also assessed according to the level of chlorination (Figure 5). The different congeners were assembled in two groups, namely with low (3, 4 and 5 chlorines, hereafter LC PCBs) and high degree of chlorination (6 and 7 chlorines, hereafter HC PCBs). Seventy-five percent of total PCBs in sediments corresponded to hexa- and hepta-chlorinated congeners. The organisms did not show a clear distribution pattern. While HC PCBs predominate in *H. lividus* and *P. acuta*, a greater proportion of lower chlorine congeners were found in the larva of *H. pistaceus* and in the tadpoles of *Rana perezii*.

When comparing adults and larvae of the same species the following differences were found. In general, a relative and concomitant increase of pp'-DDE at the expenses of pp'-DDT was observed in passing from larval to adult forms. The percentage of DDE was higher in the adults of the green frog than in the tadpoles ($W_{M-W}=16$; $p<0.05$). On the other hand, the ratio LC/HC PCBs was significantly higher in larvae than in adults ($W_{M-W}=15$; $p<0.05$).

The larva of *H. pistaceus* showed higher total organochlorine levels than the imago of the same species ($W_{M-W}=15$; $p<0.05$; d.w. and fat basis). However, no significant differences were found between the larval form and the imago of *H. lividus* ($W_{M-W}=24$; n.s; d.w. and fat basis), which showed a very high predominance of the HC PCBs.

4. Discussion

The chlorinated compounds found in the rice fields of the Ebro Delta are the result of the agricultural activities as well as of their past industrial production upstream of the river (Cid *et al.*, 1990). At present, all these compounds are banned, so that their presence reflects their long persistence in the aquatic environment. Particularly noticeable is the occurrence of OCS, rarely described in the literature and originated as a sub-product in the chloro-alkali processes (Ernst *et al.*, 1984; Kaminsky and Hites, 1984). Such a factory is located 200 km upstream the Ebro River.

These pollutants can be distributed among the three main compartments, namely water, sediment and biota, according to partitioning models (Carey *et al.*, 1998). These models have been effective in estimating the distribution of hydrophobic pollutants between water and sediments and the bioaccumulation potential in aquatic biota, especially fish and benthic invertebrates. Although sediments constitute a sink, these pollutants can be made available to biota by desorption processes to water, by contact with the organism body surface(s) and/or by bioturbation, thus enhancing the ingestion of re-suspended particles (Schaw and Connell, 1986, Nowell *et al.*, 1999).

According to the equilibrium partitioning hypothesis, the bioaccumulation depends on the affinity of the chemical for the tissues of the organisms and thus, the levels of

persistent pollutants should not present large differences amongst the organisms analysed if results are referred to fat content (Van der Oost *et al.*, 1988). On the other hand, it has been generally stated that biomagnification plays an important role in the bioaccumulation processes (Bierman, 1990; Lake *et al.*, 1990, MacDonald *et al.*, 1993). However, a detailed assessment of the results shown in Table 2 indicate that bioaccumulation is also greatly influenced by other aspects, such as the biological or ecological characteristics of the organisms examined.

In this regard, the ratio of DDT metabolites over total DDTs (DDT+DDE+DDD) has been used as an estimate of the time elapsed since the use of the pesticide. The proportion of pp'-DDT in the samples analysed (Figure 4) suggests that historical inputs are the main source of DDTs in this ecosystem. Nevertheless, the relatively high proportion of pp'-DDT (ca. 12%) in the larva of *Hydrous pistaceus* is difficult to explain, since it has no parallel in other organisms with analogous biology and ecology. The lack of metabolic systems for this compound or selective processes of uptake, could be among the possible causes. It is worth pointing out that the ability to accumulate pp'-DDT would give this species an excellent role as sentinel in the environmental assessment of this pesticide.

Interestingly, those organisms known to be highly resistant to pp'-DDT, such as *H. medicinalis* (Sawyer, 1974) or *P. acuta* (Harman, 1974), do not show detectable levels of that contaminant. In *Hirudo medicinalis* and in the imago and larval forms of *Helochares lividus*, the high levels of DDTs are accompanied by an increase in pp'-DDE, thus constituting an additional risk for their vertebrate predators.

Most vertebrates can metabolise DDT owing to the presence of hepatic MFO systems (Carey *et al.*, 1998). Their capacity is well in agreement with general metabolic rate (small vertebrates higher than large ones), directly related to variability of ingested food (omnivorous and herbivorous higher than carnivorous) and to some extent, also determined by phylogenetic constraints (higher in mammals than in birds and fish) (Walker, 1980). In *Rana perezi* changes associated to metamorphosis can explain intra-specific differences in DDT metabolites, as it occurs with dieldrin (Cooke, 1979, Schuytema *et al.*, 1991), since microsomal enzymatic systems might be less functional in tadpoles and detoxification capacities seems to develop as the larva grows up thus explaining the higher percentage of DDE found in the adults.

DDD is mainly produced in anoxic sediments (Zoro *et al.*, 1974) and its abundance in the sediments of the rice field reveals the reductive condition of this compartment. Thus, the ratio DDD/DDE can be related to the benthic habits of the organisms studied, especially in those also feeding in the sediment fraction. In the tadpoles, the value DDD/DDE is very close to that of the sediment (0.9 to 1.2). Similarly, organisms such as *P. acuta* which move over the rice field bottom, bioturbate the sediment and this would allow a higher DDD availability. On the other hand, organisms living in the water column (both imago and larva of *Hydrous pistaceus* and *Helochaeres lividus*) show mean ratio values around 0.3.

The case of midges, which are also filter-feeders, do not show detectable amounts of DDD. This is particularly interesting because they feed on particles suspended in the water column, prior to reaching the anoxic environment, i.e. before DDD synthesis occur. The values for metamorphosed *Rana perezi* are very low, perhaps because of the

nectonic habits of the adults, in addition to the increased detoxification capacities mentioned above.

As far as the transfer of PCBs to biota, LC PCB present a higher solubility and are associated mainly to the water dissolved phase, whereas HC PCB tend to be more associated to particulate matter, both in the water fraction and in the sediment (Valls *et al.*, 1990), thus being prone to biomagnification through the food web.

Two groups of organisms can be identified according to the prevalence of one or another PCB congeners pattern, namely those presenting a large predominance (>80%) of HC PCB congeners (*Physella acuta*, Chironomidae, and *Helochaeres lividus*, both larva and imago), and those having noticeable amounts (> 40%) of LC PCB congeners (*Hydrous pistaceus*, larva, *Rana perezi*, larva and *Hirudo medicinalis*) (Figure 5). In *H. lividus*, both larval and imago stages are the organisms showing the higher organochlorine levels and also the high percentages of HC PCBs, thus fitting well the biomagnification model. However, it must be pointed out that the LC PCB group includes a predator form, such as *H. pistaceus* larva, whereas some non-predator animals, like the mollusc *P. acuta*, are placed in the HC PCB group. This would indicate that additional considerations other than trophic level are needed to explain the observed PCB congener bioaccumulation profiles. Moreover, the lower PCB levels in frogs would suggest the action of enzymatic detoxification systems. In midges, which are filter-feeders, most PCBs corresponded to HC PCB (ca. 96%), suggesting that the source of these compounds is the particulate matter they ingest in the water column and that adsorption rates discriminating LC from HC PCBs takes place before these compounds reach the sediment.

The case of *Rana perezi* is especially significant because the transition from the tadpole to the adult is accompanied mainly by a qualitative change in the PCB profile. If a greater proportion of LC PCBs is to be found in solution into the aqueous phase, it is expected that the relative abundance of these congeners will be higher in those organisms showing filter-feeding habits or being gill-breathers which circulates high volumes of water through the body, as is the case with *R. perezi* or *H. pistaceus* larvae. Conversely, the percentage of LC PCBs would be lower in the adult stages of these organisms, which are air-breathers, as it is the case (LC PCBs, ca.25%). The fact that metamorphosis in both species is associated to a change in diet, from predator to herbivore in *H. pistaceus*, and from microphagous to carnivorous in *R. perezi*, put forward that trophic level is not always the main factor governing the pattern of PCB congener distribution.

The pattern observed for PCBs in bloodsuckers (*H. medicinalis*) is in agreement with that of *R. perezi*, since the higher degree of LC PCB must be presumably in correspondence with the levels present in the host's blood, HC PCBs being effectively retained in tissues of higher fat content.

In conclusion, the study clearly indicates that the organochlorine biomagnification according to the trophic level in ecosystems is not a general trend. Actually, it is only observed in particular circumstances. Other biological and ecological factors may play a major part and may lead to higher contamination in invertebrates than in vertebrates. The lack of relationship between the contamination level and the trophic level is emphasized by similar results from other environments and ecosystems (Nowell et al., 1999).

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6. References

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FIGURE CAPTIONS

Figure 1.- Ebro Delta (NE Spain). * Sampling area.

Figure 2.- Boxplot of total organochlorines (log-concentration) in sediments and in different trophic categories (Filter-feeders: chironomids and tadpoles *Rana perezi*; Primary consumers: *Physella acuta*, *Hydrous pistaceus*; Secondary consumers: *Hirudo medicinalis*, *Helochares lividus* adult and larvae, *Hydrous pistaceus* larvae, *Rana perezi*)

Figure 3.- Percentage of different organochlorine levels in sediments and organisms from the rice fields (L= larvae).

Figure 4.- Percentage of different DDT metabolites (pp'-DDT, pp'-DDD and pp'-DDE) in sediments and organisms from the rice fields (L= larvae).

Figure 5.- Percentage of HC PCBs congeners (6 and 7 chlorine atoms) and LC PCBs (3, 4 and 5 chlorine atoms) in sediments and biota from the rice fields (L= larvae).

Figure 1

Figure 2

Figure 3

Figure 4

Figure 5

Table 1. Main ecological characteristics of the rice field organisms analysed.

<u>ORGANISMS</u>	<u>HABITAT</u>	<u>FEEDING</u>	<u>PREYED BY</u>
<u>Mollusca:</u>			
<i>Physella acuta</i>	Sediment-dwelling	Herbivorous	Ducks
<u>Annelida:</u>			
<i>Hirudo medicinalis</i>	Parasitic, mainly on frogs	Vertebrate blood	Waterbirds
<u>Insecta:</u>			
Chironomids	Anoxic waters	Periphyton, phytoplankton filter-feeding	
<i>Hydrous pistaceus</i>	Aquatic	Larva: carnivorous Adult: herbivorous	Duck chicks, mosquito-fish, Ardeidae
<i>Helochaeres lividus</i>	Aquatic	Larva and adult: Detritus-feeder, scavenger	
<u>Amphibia:</u>			
<i>Rana perezi</i>	Aquatic	Larva: sediment filter feeder Adult: Insects and mollusca	Ardeidae

Table 2. Geometric mean of organochlorine compounds analysed in rice field organisms. Results are provided in ng.g⁻¹ dry weight basis (a) and in fat basis (b).

2a.

		PCBs	DDTs	HCHs	HCB	OCS	DDTs/PCBs
Sediment	G.M.	5.9	15.7	0.2	1.8	0.6	2.7
	Min	1.5	5.8	< 0.1	0.2	0.1	0.8
	Max	10.6	55.0	2.8	11.9	2.8	10.3
<i>P.acuta</i>	G.M.	43.1	164	5.9	3.9	1.5	3.8
	Min	19.5	119	3.2	1.8	0.0	1.7
	Max	127	213	9.4	17.9	38.3	9.2
<i>H.medicinalis</i>	G.M.	29.6	114	2.7	4.2	2.1	3.8
	Min	3.4	8.6	1.3	0.6	0.1	1.4
	Max	77.4	440	28.2	30.4	16.5	7.1
Chironomid (<i>L</i>)	G.M.	53.8	67.8	8.8	11.3	2.2	1.3
	Min	26.2	35.6	6.5	3.0	0.0	0.7
	Max	107	138	10.9	46.1	30.1	2.4
<i>H.pistaceus</i> (<i>L</i>)	G.M.	55.7	87.6	19.8	11.0	0.1	1.6
	Min	25.2	40.5	2.3	7.3	0.0	0.6
	Max	81.4	115	45.9	17.4	2.8	4.2
<i>H.pistaceus</i>	G.M.	16.0	14.6	2.1	2.6	0.2	0.9
	Min	4.5	3.1	0.7	0.5	0.0	0.6
	Max	54.7	46.0	5.2	14.3	1.1	3.5
<i>H.lividus</i> (<i>L</i>)	G.M.	43.0	206	6.5	4.7	3.1	4.8
	Min	23.0	136	3.7	2.2	0.8	1.2
	Max	111	286	10.2	12.0	9.9	12.4
<i>H.lividus</i>	G.M.	170	265	36.0	12.9	0.2	1.5
	Min	15.6	41.9	6.9	1.9	< 0.1	0.1
	Max	945	10750	221	452	97.7	11.4
<i>R.perezi</i> (<i>L</i>)	G.M.	11.2	51.1	2.6	2.9	1.6	4.5
	Min	5.2	26.2	1.0	0.6	0.3	2.0
	Max	46.0	91.4	4.9	58.1	26.7	15.6
<i>R. perezi</i>	G.M.	9.3	35.4	0.5	2.7	2.4	3.8
	Min	3.0	14.6	0.1	1.1	1.0	1.6
	Max	45.6	76.4	1.8	13.9	5.5	13.1

2b.

		PCBs	DDTs	HCHs	HCB	OCS
<i>P.acuta</i>	G.M.	1215	4619	167	109	43.0
	Min	616	3735	70.7	56.8	0.4
	Max	2817	5646	295	397	849
<i>H.medicinalis</i>	G.M.	303	1165	27.7	43.0	21.6
	Min	34.8	86.2	10.9	6.1	1.3
	Max	711.0	3768	259	257	144
Chironomid (<i>L</i>)	G.M.	805	1014	131	169	32.9
	Min	349	285	86.8	58.2	0.2
	Max	1444	2400	176	718	468
<i>H.pistaceus (L)</i>	G.M.	1358	2137	483	268	3.3
	Min	454	1344	76.4	124	0.2
	Max	3003	4260	1387	642	57.1
<i>H.pistaceus</i>	G.M.	69.0	63.2	9.1	11.4	0.9
	Min	14.5	9.9	2.4	1.5	0.1
	Max	300	206	25.4	71.0	6.1
<i>H.lividus (L)</i>	G.M.	264	1266	39.9	28.7	19.2
	Min	110	1123	23.1	13.1	6.3
	Max	911	1398	84.8	57.7	79.1
<i>H.lividus</i>	G.M.	884	1373	186	67.1	1.1
	Min	90.6	234	40.1	11.11	0.1
	Max	5970	67871	1395	2854	617
<i>R.perezi (L)</i>	G.M.	101	462	23.3	26.4	14.1
	Min	41.2	200	14.3	4.7	1.4
	Max	195	1613	40.4	246	113
<i>R. perezi</i>	G.M.	105	401	5.8	30.7	27.4
	Min	38.6	135	0.9	10.0	13.3
	Max	484	1251	17.1	147	58.6