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The zooplankton of the lower river Meuse, Belgium: seasonal changes and impact of industrial and municipal discharges

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Abstract

Between 1991 and 1993, samples were collected upstream and downstream of the industrial basin and urban centre of Liège. Rotifers and crustaceans (cladocerans and copepods) were identified and counted. Their population dynamics were related to physical and chemical factors (temperature, oxygen, ammonium, nitrates, nitrites, phosphates) and to phytoplankton biomass. The zooplankton was dominated by rotifers; crustaceans (cladocerans and copepods) were less abundant. There was a succession of groups and species, some thriving in the spring and others in summer or autumn. The dominant rotifer species were *Brachionus calyciflorus* Pallas, *Brachionus angularis* Gosse, *Keratella cochlearis* (Gosse) and *Synchaeta* spp.; *B. calyciflorus* and *B. angularis* are spring species. *K. cochlearis* was present between May and November. Crustacean biomass was important in summer and autumn, but the faunal spectrum and biomass also varied with sampling location. Low spring and summer discharges allowed the phytoplankton to develop significantly. The zooplankton development followed a similar pattern. During low flow, when plankton populations become established, some declines in phytoplankton could only be explained by sedimentation and grazing pressure by zooplankton. Although these factors provided a good explanation of the longitudinal variation, some local conditions (e.g. oxygen deficit, high level of phosphate) also induced changes (e.g. industrial and municipal waste water discharge).

Introduction

The use of bioindicators in water quality assessment started more than a century ago (Liebmann, 1962). The organisms used are diverse (e.g. bacteria, fungi, microalgae, protozoans, rotifers, cladocerans, copepods, macroinvertebrates – see e.g. Woodiwiss, 1964; Tuffery & Vernaux, 1968; Sládecek, 1973; Persoone & De Pauw, 1979; Herricks & Cairns, 1982; Descy & Coste, 1991). Zooplankton is a good indicator of changes in water quality because it is strongly affected by environmental conditions and responds quickly to changes in environmental quality (Gannon & Stemberger, 1978).

Among zooplankton, rotifers, with their high population turnover rates, are particularly sensitive to water quality changes. They are good indicators of saprobity (Sládecek, 1983) and form the basis of a system for classifying ponds in England (Pontin & Langley, 1993). Due to bioaccumulation, moreover, zooplankton organisms may become highly contaminated by persistent organic and inorganic micropollutants (e.g. 1.1 μg PCBs g^{-1} D.W.; Thomé *et al.*, 1993). Levels of such contaminants can provide precious indications as to the extent of industrial pollution. Other studies have shown that eutrophication affects zooplankton composition, shifting the dominance from larger species (e.g. calanoid copepods) to smaller species (especially rotifers) (Premazzi & Chiaudani, 1992).

Besides by effects of pollution, riverine plankton communities are likely to exhibit changes depending on physical flow, particularly longitudinal changes during downstream transport. Owing to the characteristics of the lotic environment, i.e. short water residence time relative to the generation time of the plankton animals, a dominance of fast growing rotifers over slower growing crustaceans is not surprising (Hynes, 1970). Furthermore, a close dependency upon the availability of food is expected, which entails that factors which control the development and composition of the phytoplankton (see e.g. Reynolds *et al.*, 1994; Descy & Gosselain, 1994) also influence zooplankton.

As a consequence, zooplankton dynamics in a polluted river stretch may be governed by a complex combination of natural variables (light, temperature, flow rate, ...), of disturbances by various pollutants and by the life histories of the animals. In an attempt to identify such factors, we conducted a study aimed at evaluating the impact of industrial and municipal waste discharges on the plankton of the river Meuse, Belgium. Here, we focus on zooplankton and on downstream qualitative and quantitative changes which it undergoes along a gradient of water quality deterioration.

Description of the sites studied

The river Meuse (total length: 885 km; watershed area: 36 000 km^2) rises in France and flows through Belgium and the Netherlands, before reaching the North Sea in a delta common with the river Rhine. The Belgian section (*c* 200 km) has been heavily regulated for navigation. The depth of its channel (3 to 6 m) and its low water transparency (range of vertical extinction coefficient: 1.5–3.5 m^{-1}) do not allow significant benthic primary production, and the barely developed benthic vegetation is restricted to the banks. On the other hand, a large phytoplankton biomass develops at favourable weather and flow rate conditions (Descy, 1987; Gosselain *et al.*, 1994), in a successional pattern that is driven mostly by external physical factors. Among those, discharge plays a prominent role; the hydrological regime is characterized by large variations between winter (200–800 $m^3 s^{-1}$ at Ampsin-Neuville, up to 2000 $m^3 s^{-1}$ in major floods) and summer (often less than 30 $m^3 s^{-1}$). Typically, phytoplankton development begins in spring when discharge falls below 200 $m^3 s^{-1}$ and is maintained throughout the growing season, showing distinct community changes which have been discussed elsewhere (Descy, 1987, 1992; Gosselain *et al.*, 1994). It is, however, worth noting here that phytoplankton declines in summer may be attributed, at least in part, to grazing by herbivorous zooplankton. Phytoplankton also shows a distinct longitudinal pattern of development, related to the morpho-dynamic variations from source to mouth; depending on discharge, temperature and light conditions, their maximal biomass is reached in different river reaches of the Belgian part (Descy, 1992). As the river runs through industrial basins and cities, it receives sewage waters carrying pollutants such as heavy metals (As: 4–6 $\mu g\ l^{-1}$; Cd: 2–3 $\mu g\ l^{-1}$; Cr: 10–20 $\mu g\ l^{-1}$; Cu: 10–25 $\mu g\ l^{-1}$; F: 0,5–1 $\mu g\ l^{-1}$; Hg: 0,1 $\mu g\ l^{-1}$; Pb: 10 $\mu g\ l^{-1}$ and Zn: 100 $\mu g\ l^{-1}$) (RIWA, 1993) and many other inorganic and organic compounds (e.g. organohalogens such as PCBs) (Jeuniaux *et al.*, 1984; Henry *et al.*, 1989; Admiraal *et al.*, 1993; Thomé *et al.*, 1993).This pollution by xenobiotics on the one hand, and municipal waste water carrying abundant faecal matter which produce heavy organic pollution (Descy *et al.*, 1981) on the other hand, may induce zooplankton community disturbances.

The present study was carried out between 1991 and 1993 in the lowest section of the Belgian Meuse, in the vicinity of Liège, from km 580 to km 620 (Fig. 1). Already eutrophic upstream, the river water is here submitted to heavy industrial and urban pollution, extending from Ampsin (km 580) to Hermalle-sous-Argenteau (km 620). From March to November 1991, samples were taken every two weeks at four stations (Fig. 1): Ampsin (reference sampling station), Fragnée (just upstream of the city of Liège and downstream of the main industrial basin), Monsin, and Hermalle-sous-Argenteau (downstream of the city waste water discharge). At Monsin, two additional sampling series were carried out at several depths over a cross-section of the river in order to test the supposed evenness of zooplankton distribution. Samples were taken at five points across the river and three depths (0.5 m, 1.5 m and 3 m). The first of these sampling series took place in June (river discharge: 50 $m^3 s^{-1}$, June 6, 1991) The second was done at low-flow (20 $m^3 s^{-1}$, September 19, 1991). It is worth to mention that the Monsin site is located downstream of a major water abstraction (Albert Canal) and that the flow is slowed by a large weir; so, at this location, the hydrodynamic characteristics should favour heterogeneity of zooplankton distribution.

In 1993, weekly samples were taken at the two downstream stations (Monsin and Hermalle-sous-Argenteau) between April and November.

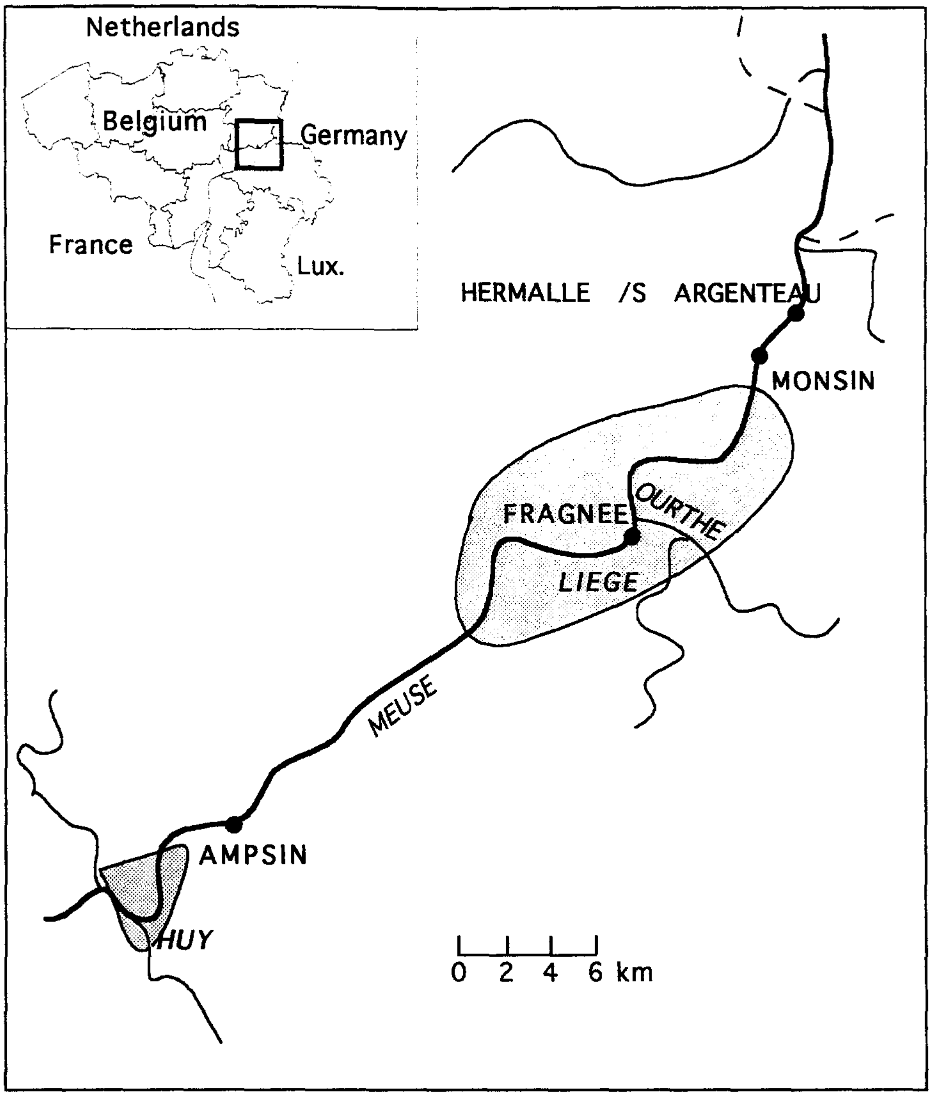

Fig. 1. Map of the basin of the river Meuse, locations of the sampling stations.

Materials and methods

Zooplankton analysis

The standard sampling procedure consisted of collecting 50 litres of river water which were filtered through a 50 μm mesh net fitted in a special system which keeps the plankton under two centimetres of water to prevent damage to the fragile specimens. This system was tested in order to make sure that all the organisms were caught on the net. Rotifers, cladocerans, and copepods were identified to various taxonomic levels (genera or species) and counted. Several identification keys were used for these determinations (Pontin, 1978; Pourriot & Francez, 1986; Amoros, 1983; Harding & Smith, 1974; Dussart, 1969). The biomass was estimated from regression equations relating dry weight to body length and from dry weight data available in the literature (Dumont *et al.*, 1975; Pauli, 1989; Andrew & Fitzsimons, 1992).

Physical and chemical variables, phytoplankton biomass

The main inorganic nutrients (ammonium, nitrites, nitrates, phosphates) were analysed by standard photometric methods with an SQ 118 photometer and Spectroquant® kits (Merck, Darmstadt, Germany). Ammonia-N was assayed as indophenol blue at 690 nm (Bolleter *et al.*, 1961). Nitrate-N (after reduction) and nitrite-N were assayed with sulfanilic acid and l-naphtylamine, the absorbance being measured at 526 nm (Bratton *et al.*, 1939). Orthophosphate ions were reduced to phosphomolybdenum blue (PMB) whose absorbance was measured at 712 nm.

Temperature, pH, conductivity, and oxygen concentration were measured with a digital thermometer associated with a pH meter (pH 96 WTW, Weilheim, Germany), a conductivity meter (LF 92 WTW), and an oxymeter (OXI 96 WTW).

For assessing phytoplankton biomass, water samples were filtered through 1.2 μm Whatman GF/C filters and chlorophyll *a* was extracted as described by Pechar (1987) with a mixture of acetone (90%) and methanol (5:1; V:V) (2 minutes at 65 °C). The absorbance was measured at 664 nm before and after acidification with HCl 0.1 N.

Other data on various variables (dissolved and particulate organic carbon, dissolved inorganic nitrogen, particulate organic nitrogen, soluble reactive phosphorus, particulate phosphorus, algal carbon derived from chlorophyll *a*) have been given in Descy & Gosselain (1994).

Statistical analysis

Statistics on the data were performed using a two-way ANOVA (SigmaStat^{TM} for Windows^{TM} Software, Jandel Scientific, Erkrath, Germany).

Results

The hydrological regime of the river Meuse during the period studied is characterized by large variations (Fig. 2): the discharge is high from November to March (up to 200 $m^3 s^{-1}$ at Ampsin-Neuville) and low during summer (often less than 30 $m^3 s^{-1}$).

Every year, the temporal variation of temperature follows a quite similar evolution. In 1991 (Fig. 2), between March and November, the temperature of the river Meuse in the section studied ranged from 10 °C to 25 °C. Along the stretch, water temperature was influenced by various thermal waste water discharges (e.g. Nuclear Power Plant of Tihange, steel works, gas/steam turbine power plant of Seraing). It increases significantly downstream from Ampsin and reaches maximum values at the Fragnée sampling station. It then decreases downstream from Liège, a temperature drop probably caused by the inflow of colder water from the river Ourthe.

Dissolved oxygen was significantly different between stations (ANOVA; $P \leq 0.01$), except between Fragnée and Monsin. The concentrations tend to decrease as the water flows downstream, especially between Monsin and Hermalle-sous-Argenteau. Indeed, during 1991 and 1993, dissolved oxygen measured at the last sampling station was regularly below 2 mg l^{-1} in summer. Figure 3 illustrates this for 1993 and shows that oxygen deficiency is severe in Hermalle-sous-Argenteau when water level is lowest (minimum value: 0.7 mg l^{-1} in 1993). Many variables, including temperature, river discharge, reaeration, sewage inflow, phytoplankton photosynthesis and heterotrophic activities explain the fluctuations observed between April and October at all sampling stations.

As far as nitrogen compounds are concerned (ammonium, nitrates and nitrites), there was no statistically significant difference between the concentrations measured at different sampling locations (ANOVA; $P \leq 0.05$), except for ammonium, which increased significantly between Ampsin and the three downstream stations (ANOVA; $P \leq 0.01$). Figure 4 presents the observed relation between oxygen and nitrogen compound concentrations (ammonium and nitrate) at the station of Hermalle-sous-Argenteau in 1991 and 1993. The oxygen content of the water varied considerably, as did concentrations of the main forms of inorganic nitrogen. These concentrations were related to river discharge, which varied with the inputs from the watershed, i.e. with precipitations which leach the soils and entrain nitrate. Although this well-known process explains most of the variations of nitrate in the river Meuse water, denitrification took place at low O_2 concentrations during the low flow period. But the ammonium content of the river water also depends on waste water inputs and biodegradation of organic matter, influenced by bacterial activity and temperature. In these particular conditions, oxygen concentrations are also usually low (due to the bacterial activities). This is well shown in Fig. 4: when oxygen concentration is low (0.13 mg l^{-1}), ammonium is highest (2.5 mg N l^{-1}) and nitrate is lowest (0.2 mg N l^{-1}).These observations are in good agreement with those of several authors who have shown that municipal waste water carrying abundant faecal matter produces heavy organic pollution (Descy *et al.*, 1982) and causes eutrophication, which results in overabundance of phytoplankton (Descy, 1987), excessive growth of bacteria (Servais, 1989) and fungi, oxygen deficiency and increased ammonia levels (Admiraal *et al.*, 1993).

Phosphate concentrations were significantly lower in Ampsin (upstream reference station) than in the three downstream stations (ANOVA; $P \leq 0.01$).

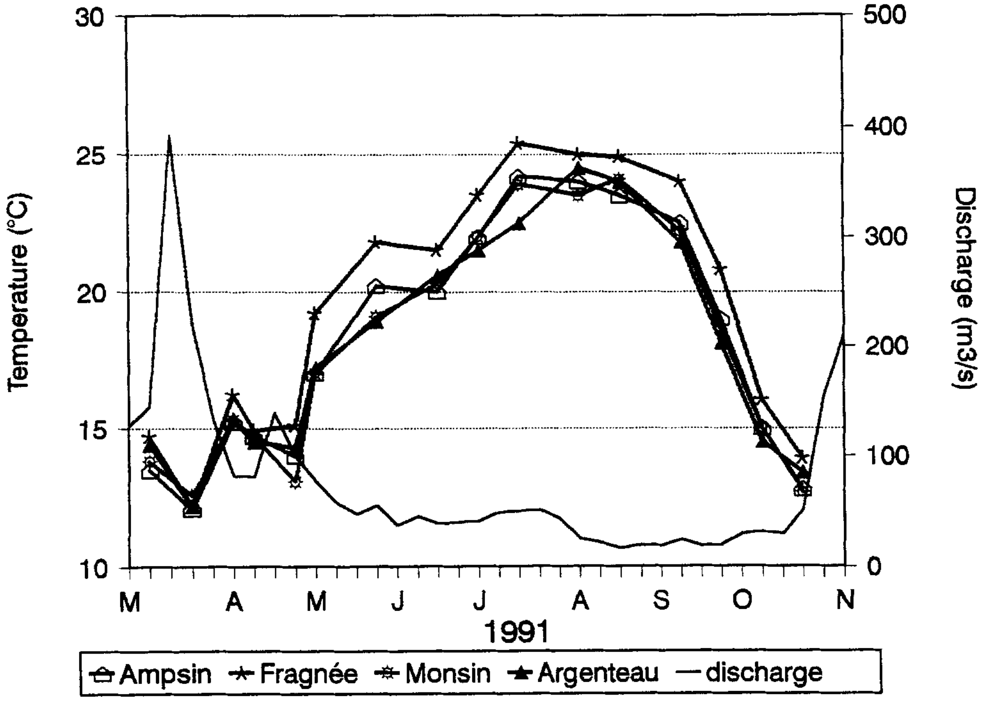

Fig. 2. Temporal variation in temperature and discharge at the four sampling stations in the river Meuse, 1991.

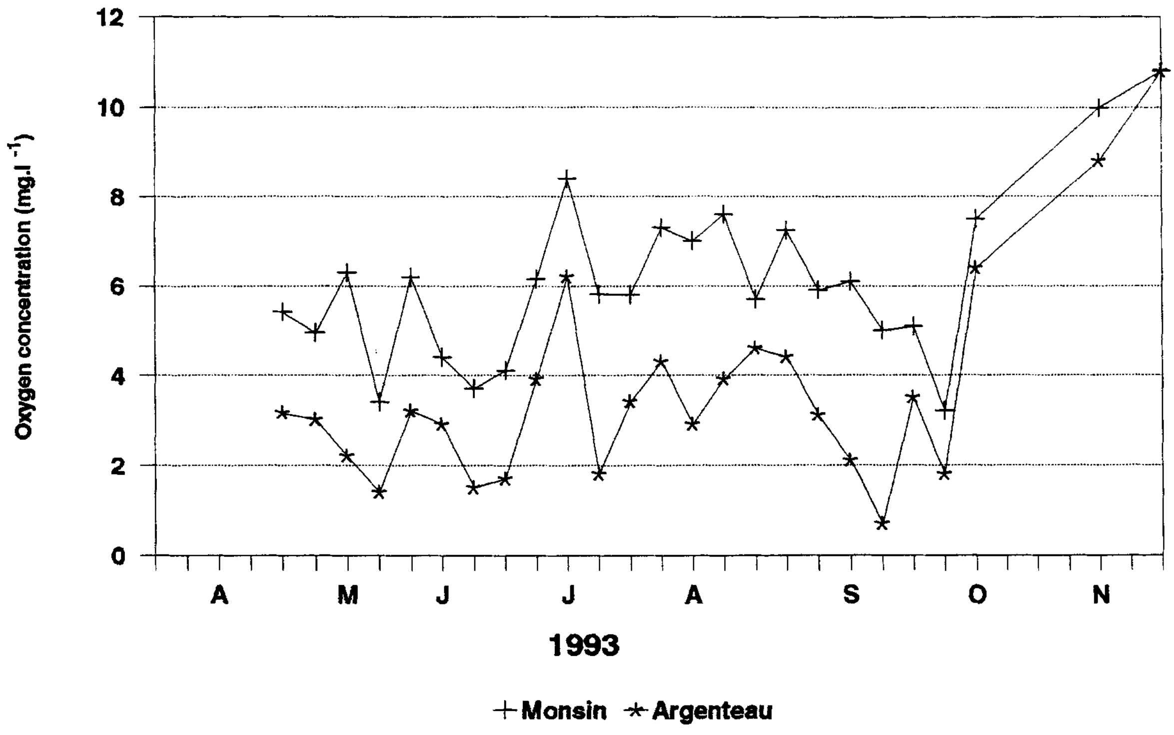

Fig. 3. Temporal variation in oxygen concentration at the sampling stations of Monsin and Hermalle-sous-Argenteau in the river Meuse, 1993

Table 1. List of the species or genera found in the river Meuse with the maximum densities recorded during the study in Ampsin (upstream from Liège) and Hermalle-sous-Argenteau (downstream from Liège).

	Maximum densities of the different species at the Ampsin sampling station. ind l^{-1}	Maximum densities of the different species at the Hermalle-sous-Argenteau sampling station. ind l^{-1}
ROTIFERA		
Asplanchna sp	6	14
Asplanchna brightwelli		
Asplanchna priodonta		
Brachionus angularis	496	599
Brachionus calyciflorus	352	1027
Brachionus leydigi	48	47
Brachionus quadridentatus	1	1
Brachionus urceolaris	5	64
Cephalodella sp.	2	30
Epiphanes senta	1	1
Euchlanis dilatata	27	9
Filinia longiseta	6	55
Keratella cochlearis	744	987
Keratella quadrata	40	70
Kellicottia longispina	<1	<1
Notholca acuminata	5	0
Notholca squamula	5	38
Polyarthra spp.	73	192
Rhinoglena frontalis	5	0
Rotaria spp.	20	24
Synchaeta spp.	510	188
CRUSTACEANS		
CLADOCERANS	27	544
Daphnia spp.		
Moina macropa		
Bosmina longirostris		
COPEPODS	253	624
Cyclops sp.		
Acanthocyclops sp.		
Diaptomus sp.		

Indeed, Fig. 5 shows that phosphate concentrations, already rather high at the downstream site (i.e. large enough to saturate phytoplankton growth) increase sharply between Ampsin and Fragnée and remain high downstream. Such profiles are clearly influenced by industrial discharges (Van Craenenbroeck & Van den Bos, 1983), while decreases may be explained by phytoplankton uptake during bloom periods (Marneffe & Thomé, 1991).

Composition of the zooplankton

Zooplankton in the Belgian Meuse is clearly dominated by rotifers; crustaceans (cladocerans and copepods) are less abundant. However, crustaceans can represent an important part of total zooplankton biomass due to their high individual weight. The faunistic list of taxa found in the studied stretch is presented in Table 1, along with the maximum densities observed during the survey period in Ampsin (upstream) and in Hermalle-sous-Argenteau (downstream). Generally, the maximum densities of the different species were higher in Hermalle-sous-Argenteau than in Ampsin. Figures 6 and 7 show the changes in zooplankton biomass (rotifers plus crustaceans) in 1991. In the course of the year, there was a succession of species, some thriving in spring, others in summer or autumn. The faunal spectrum and species biomass varied with sampling location. It appears from these figures that copepod biomass was higher in Monsin than in Ampsin, i. e. the reference upstream station. It also appears that maximum cladoceran biomass was reached in the last station (Hermalle-sous-Argenteau).

The dominant rotifer species in the Meuse were *Brachionus calyciflorus* Pallas, *Brachionus angularis* Gosse, *Keratella cochlearis* (Gosse) and *Synchaeta* spp.; *B. calyciflorus* and *B. angularis* are spring species. *K. cochlearis* is present between May and November. Crustacean biomass became important in summer and autumn as a consequence of large numbers and high individual weights.

The first important zooplankton bloom occurred between June and July at all four stations. Its duration was variable, increasing sharply from upstream (15 days in Ampsin) to downstream (one month in Hermalle-sous-Argenteau); the same was true for its magnitude (300 $\mu g\ l^{-1}$ in Ampsin to 1506 $\mu g\ l^{-1}$ in Monsin). The second zooplankton bloom occurred in September. Its magnitude increased from Ampsin (237 $\mu g\ l^{-1}$) to Monsin (1866 $\mu g\ l^{-1}$) but biomass increase was low at Hermalle-sous-Argenteau (92 $\mu g\ l^{-1}$). The general tendency was towards an increase in biomass and abundance from upstream to downstream, especially at Monsin where biomass was significantly higher than elsewhere (ANOVA; $P \leq 0.01$). This

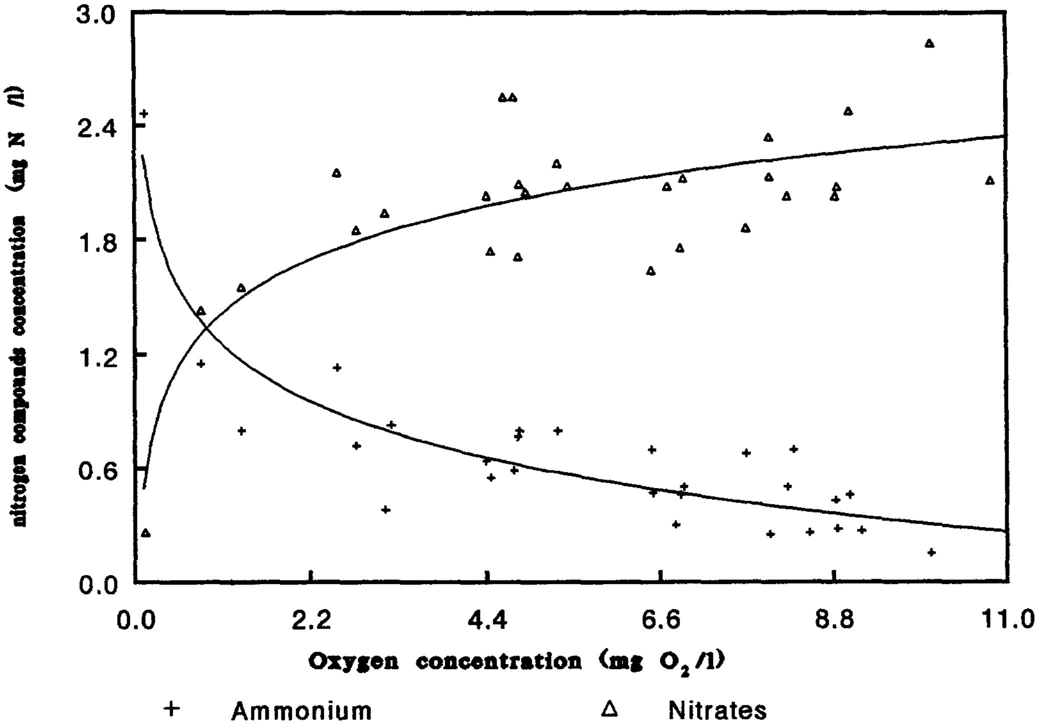

Fig. 4. Relation between ammonia, nitrate, and oxygen concentrations in the river Meuse at the sampling station of Hermalle-sous-Argenteau in the river Meuse, 1991.

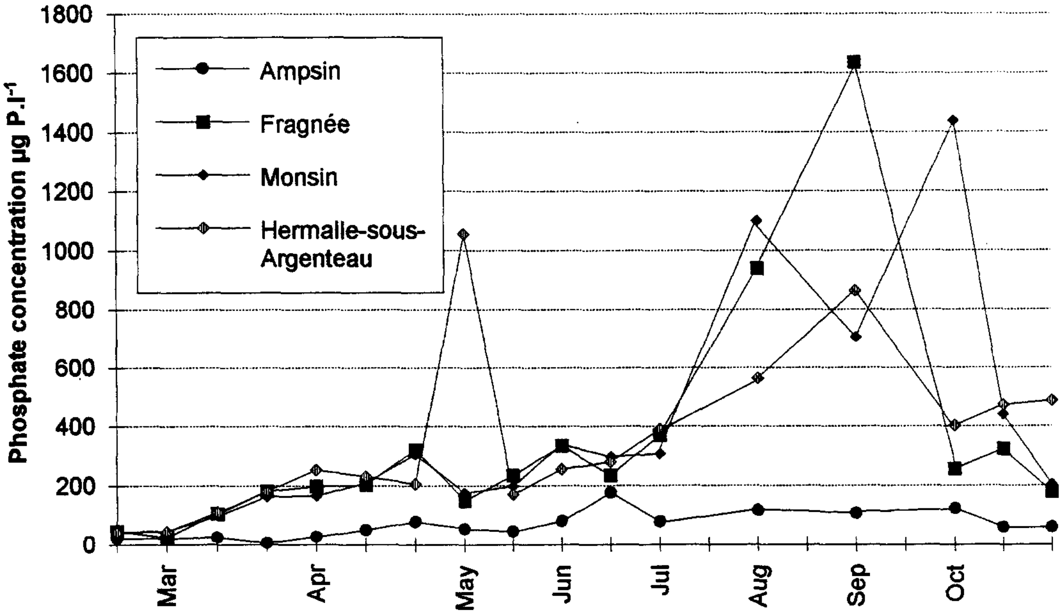

Fig. 5. Temporal variation in phosphate concentrations at the four sampling stations in 1991.

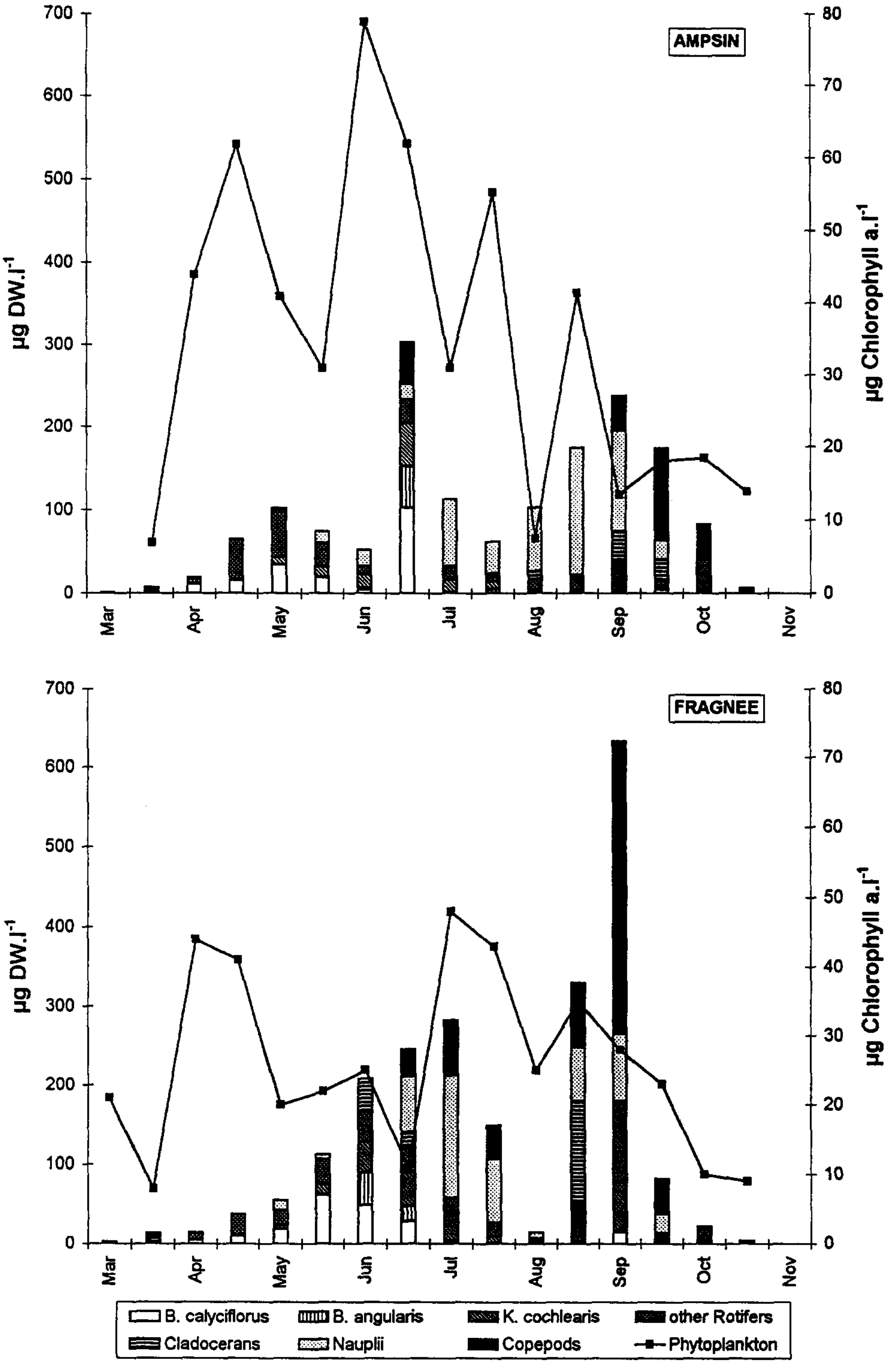

Fig. 6. Temporal variation in zooplankton biomass and chlorophyll *a* concentrations at the sampling station of Ampsin (A) and Fragnée (B) in the river Meuse, 1991.

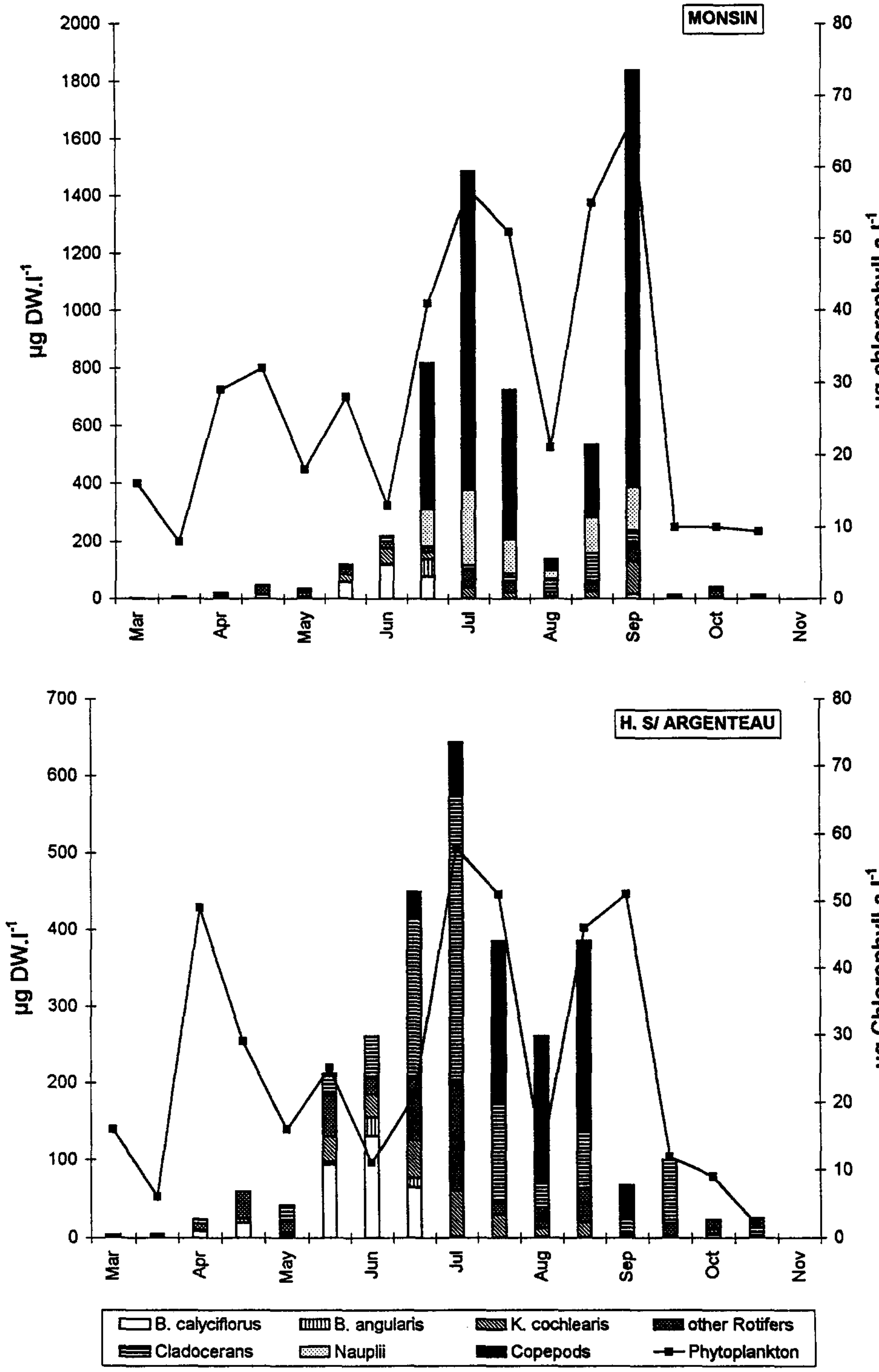

Fig. 7. Temporal variation in zooplankton biomass and chlorophyll *a* concentrations at the sampling station of Monsin (A) and Hermalle-sous-Argenteau (B) in the river Meuse, 1991.

observation is particularly true for adult crustaceans (copepods and cladocerans).

Thus, in the course of a year, as soon as irradiance and water temperature increase, and above all, when discharge decreases, allowing phytoplankton growth, several zooplankton blooms are observed. During low flow, these blooms correspond generally to an increase of phytoplankton biomass. Figures 6 and 7 illustrate this relation. In the same way, a decrease in zooplankton biomass usually corresponds to a decrease in phytoplankton biomass, but this relation is not observed all over the year (e.g. between March and May) and does not appear so clearly at all sampling stations (e. g. Fragnée), suggesting the importance of water quality.

During low-water, the distribution of zooplankton biomass in a cross-section of the Meuse was heterogeneous (Fig. 8A): a maximum was found just below the surface (0.5 m) along the left bank, and a minimum near the bottom of the river (3 m below the surface). At higher flows (6/06/1991; 50 m^3 s^{-1}), zooplankton distribution was homogeneous (Fig. 8B).

Discussion

The recorded zooplankton biomass (mainly rotifers and crustaceans) in the Meuse and the nutrient concentrations measured reflect productive nature of this river. According to Angeli (1976), the simultaneous presence of several species of the genus *Brachionus* is a good indication of the eutrophic nature of an aquatic ecosystem. Descy (1987) showed that discharge, temperature and light are important in controlling phytoplankton biomass in the river Meuse, the limited plankton growth being related to the high flow periods. Discharge acts on algal biomass as a dilution rate, while light and temperature determine growth rates (Descy *et al.*, 1987). Along the course of the river, a low summer discharge allows the phytoplankton to develop a significant biomass at 200 km from the source, and to achieve a full growth within less than 200 km (Descy & Gosselain, 1994). Zooplankton development would be expected to follow a similar pattern. Nevertheless, during the low flow period, when the riverine plankton populations are established, other variables may control these populations. Indeed, during this period, some declines in phytoplankton can only be explained by a decreasing growth rate combined with loss factors such as sedimentation and grazing pressure by zooplankton. In Ampsin, from April to August, the period of phytoplankton decline corresponds to that of zooplankton growth, but this grazing pressure is probably combined with a decreasing growth rate (due to variations in light and water transparency) (Gosselain *et al.*, 1994). Reciprocally, phytoplankton biomass seems to control zooplankton growth since zooplankton blooms or declines usually correspond to phytoplankton biomass changes, even though this relation is not so clear at all stations all the time (for instance from March to May and from September to November). During spring, water temperature is not high enough to allow the development of much zooplankton biomass, while phytoplankton is already well-developed during this period. Indeed, some phytoplankton species (mainly small *Stephanodiscus* spp.) can grow at low temperatures and low irradiances, but they need a low-flow period (Gosselain *et al.*, 1994). In the same way, zooplankton species biology certainly explains part of the observed changes in that community related to a decrease in temperature during September. So, the relationship between the two communities seems to be complex, and interpretation of observed changes is complicated further by water quality changes. On the whole, it seems to emerge from our data that zooplankton is primarily dependant on its food resource (i.e. the more phytoplankton, the more zooplankton), at least when looking at temporal variations at one site.

Zooplankton blooms tend to last longer and to be stronger downstream than upstream. This tendency could be explained by the 'age' of the water, since at low discharge the water needs between 2 and 4 days to flow from the first station to the last. The older the water, the greater its chance of having acquired plankton organisms and the longer they have had to reproduce (Hynes, 1970). It is easy to calculate that this additional travel between the first and the last station allows more generations of fast growing zooplankters such as rotifers. In the same way, the downstream increase in crustacean biomass could be explained by the fact that, having a slower growth, they need more time to build significant populations.

Even though increased residence times explain most of the longitudinal variations in the total zooplankton abundance, some local conditions may also induce changes. Indeed, although the blooms are usually more marked downstream than upstream, there are exceptions, especially at Hermalle-sous-Argenteau, where oxygen deficiency can be drastic at the end of summer (e.g. 0.13 mg l^{-1} on September 19, 1991), related to the discharge, between Monsin and

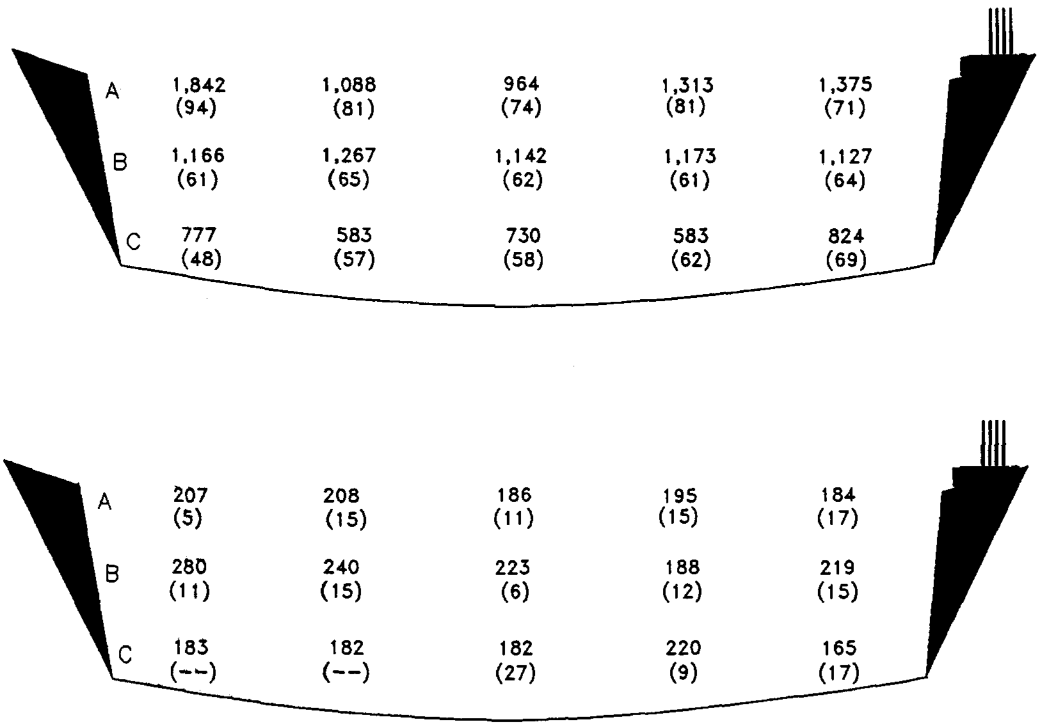

Fig. 8. Depth-wise and transversal distribution of zooplankton (mg m^{-3}) and phytoplankton (mg chlorophyll *a* m^{-3} in brackets) biomasses at the Monsin sampling station (A: September 19, 1991; B: June 6, 1991). Depth: A: 0.5 m B: 1.5 m C: 3 m.

Hermalle-sous-Argenteau, of raw municipal waste water. According to Servais (pers. comm.), bacterial biomass increases sharply between these sites (a mean increase of 120 μgC l^{-1} between the two stations). Maximum bacterial biomass and bacterial production are particularly high at Hermalle-sous-Argenteau (560 μgC l^{-1} and 27 μgC l^{-1} h^{-1} respectively), and consume large amounts of oxygen. The resulting oxygen deficiency may explain the absence, in 1991, of one zooplanktonic bloom which occurred at the other locations in September, since rotifers are unable to live in such nearly anaerobic conditions (Sládecek, 1983).

Another consequence of waste discharges and of decreased flow is the appearance of filamentous cyanobacteria at the end of the summer. This multiplication is typical of eutrophication and organic pollution; it seems to be related to high soluble phosphate concentrations and relatively high water temperatures (Hammer, 1983; Del Giorgio *et al.*, 1991; Angeli, 1976). Responsible for these high phosphate levels are the discharge of phosphate-rich detergents and the presence, upstream from Liège, of chemical plants producing fertiliser and plaster (Van Craenenbroeck & Van den Bos, 1983).

In September, such conditions prevail downstream from Liège, and crustaceans (cladocerans and copepods) reach very high biomasses and probably exert a predatory pressure on rotifer populations, since copepods are major predators of rotifers (Williamson, 1983). The appearance of large filamentous algae not readily used as food by rotifers might favour the development of phytophagous crustaceans. Furthermore, cyanobacteria in excess inhibit the growth of most rotifers because they produce toxins and eliminate the fine micro-algae on which rotifers feed, while some copepods (e.g. *Diaptomus birgei*) exhibit a strong preference for some taxa of cyanobacteria and are able to reject toxic ones (Demott & Moxter, 1991). Moreover, like cladocerans, they can also feed on bacteria, which reach a considerable biomass at the downstream stations. These hypotheses need experimental testing by *in situ* grazing measurements.

Conclusions

In large rivers, true plankters often predominate and fast growing diatoms and rotifers are often dominant. This may be a simple trophic effect or it may be that similar conditions favour both types of organism (Hynes, 1970). In the river Meuse, a few ubiquitous rotifer species dominate the zooplankton: *Brachionus calyciflorus*, *B. angularis*, *Keratella cochlearis*, *Filinia longiseta*, *Polyarthra* spp. and *Synchaeta* spp.. They are eutrophic indicator species, classified by Sládecek (1983) as alpha or beta-mesosaprobic. Changes in plankton biomass and composition appear to be complex, not only depending on discharge, temperature, irradiance, sedimentation, but also on biotic factors such as growth rates and grazing. Although trophic relations between zoo- and phytoplankton in the Meuse now need experimental research, some indications suggest that, during low flow, phytoplankton is heavily grazed by zooplankton. Combined with a decreasing growth rate and sedimentation, this explains the summer phytoplankton decline. As a consequence, the community is inherently unstable and subject to variability. Although some physical and chemical variables (oxygen, ammonium and phosphate) and zooplankton biomass change significantly from upstream to downstream, no simple relation between both can be shown. Plankton biomass is also influenced by local conditions (e.g. oxygen deficit, high level of phosphate, bacteria biomass), brought about by industrial and municipal discharges.

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