



## Original Articles

# The food resources exploitation by small-sized fish in a riverine macrophyte habitat



Maria Grzybkowska<sup>a,\*</sup>, Małgorzata Dukowska<sup>a</sup>, Joanna Leszczyńska<sup>a</sup>, Joanna Lik<sup>b</sup>,  
Eliza Szczerkowska-Majchrzak<sup>a</sup>, Mirosław Przybylski<sup>a</sup>

<sup>a</sup> Department of Ecology & Vertebrate Zoology, Faculty of Biology and Environmental Protection, University of Łódź, 12/16 Banacha Str., Łódź 90-237, Poland

<sup>b</sup> Provincial Board of Land Reclamation and Water Facilities in Łódź, 14 Solna Str., Łódź 91-423, Poland

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## ABSTRACT

In an altered stretch of the lowland River Warta, central Poland, downstream of the Jezioro Reservoir, patches of submersed aquatic macrophytes (SAM) are dependent on water discharge regulations. Such unstable habitat supports abundant epiphytic and benthic macroinvertebrates, and zooplankton flushed out from the reservoir. Patches of vegetation are important for small-sized eurytopic fish as abundant food resources and shelters. We investigated how the abundance and size of prey types constituting the SAM-related food base was partitioned between three fish species, perch *Perca fluviatilis*, ruffe *Gymnocephalus cernua* and roach *Rutilus rutilus* in two vegetation periods of 2004 and 2011, between which SAM differed in area of bed that was overgrown and in biomass. One-way ANOSIM showed that in the larger SAM (in 2004) the diet of roach differed from those of ruffe and perch. SIMPER analysis indicated that average dissimilarity between species was  $70.94 \pm 9.62\%$ , mainly due to differences in the importance of five prey groups, which contributed nearly 70% to cumulative dissimilarity. The two percid species consumed mainly mid-sized chironomids, while roach fed on plant material, detritus with algae and small-sized chironomids. A different pattern of fish feeding was observed in 2011 when SAM was significantly smaller and food resources were scarcer. Ruffe diet was distinct from both roach and perch diets, though average dissimilarity was smaller, at  $69.64 \pm 0.84\%$ , and eight food categories contributed over 96% of cumulative dissimilarity, 40% of which by Cladocera alone, which were consumed mainly by perch and roach.

## 1. Introduction

The damming of riverine ecosystems is one of the major anthropogenic impacts on the natural functioning of freshwater ecosystems. Responses of streams to impoundments can be complex and varied (Ward and Stanford, 1980; Power et al., 1996; Poff, 2014; Van Cappellen and Maavara, 2015). Dams can affect the physical structure of the natural watercourse, with the loss of heterogeneous habitats, such as pools and riffles, and exacerbate environmental variations, including temperature fluctuation and flow modification (Baxter, 1977; Ward and Stanford, 1983; Petts, 1984). During low and stabilized water levels, resulting from flow impedance, patches of submersed aquatic macrophytes (SAM) may appear in the tailwaters of reservoirs (Grzybkowska et al., 2003, 2017; Moore et al., 2010; Głowacki et al., 2011). Dense macrophyte stands can create microhabitats with a high abundance of food resources for invertebrates and vertebrates (Tolonen et al., 2003; Figueiredo et al., 2015; Bakker et al.,

2016; Grzybkowska et al., 2017). In such habitat, macrophytes provide a surface for epiphytic algae, which serves as a food resource for invertebrates. Greater macrophyte abundance can also result in greater numbers and biomass of benthic macroinvertebrates through an increased supply of organic matter to the substrate, forming habitat for sediment-living taxa (Tokeshi and Pinder, 1985; Franklin et al., 2008; Kleeberg et al., 2010; Tóth et al., 2012). Moreover, tailwater SAM can also provide shelter for organisms flushed-out from reservoirs, such as zooplankton and young fish (Lik et al., 2017). SAM presence, by increasing habitat complexity, is potentially important for the survival of small-sized ubiquitous fish species (Penczak et al., 2012). Adults may occupy different functional feeding guilds, but the young stages frequently feed on benthic invertebrates, zooplankton and periphyton (Lik et al., 2017).

Predation is one of the most important type of interactions in ecosystems. The main factor affecting prey-predator interaction is the body size, which can affect all components of the predation cycle, including

\* Corresponding author.

E-mail addresses: [maria.grzybkowska@biol.uni.lodz.pl](mailto:maria.grzybkowska@biol.uni.lodz.pl) (M. Grzybkowska), [malgorzata.dukowska@biol.uni.lodz.pl](mailto:malgorzata.dukowska@biol.uni.lodz.pl) (M. Dukowska), [joanna.leszczyńska@biol.uni.lodz.pl](mailto:joanna.leszczyńska@biol.uni.lodz.pl) (J. Leszczyńska), [jolik86@ten.pl](mailto:jolik86@ten.pl) (J. Lik), [eliza.szczerkowska@biol.uni.lodz.pl](mailto:eliza.szczerkowska@biol.uni.lodz.pl) (E. Szczerkowska-Majchrzak), [mirosław.przybylski@biol.uni.lodz.pl](mailto:mirosław.przybylski@biol.uni.lodz.pl) (M. Przybylski).

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encounter rate, prey avoidance capacity, capture success and handling time (Stephens and Krebs, 1986; Wahlström et al., 2000). Consumption of a wide range of macroinvertebrate prey by lacustrine fish species may be a key mechanism facilitating the coexistence of these fishes (Strayer, 1991; Vlach et al., 2013). While the influence of fish on the zooplankton community is well recognized (e.g. Brooks and Dodson, 1965; Gliwicz, 1986; Jeppesen et al., 1997; Piovio-Scott et al., 2017), the effects of fish on the zoobenthic community is less clear (Strayer, 1991). In general planktonic ecologists observed that fish in lakes tend to select the largest zooplankton as prey. However, in lentic habitats zooplankton is not the primary food resource for fish foraging on zoobenthos, especially chironomid larvae (e.g. Przybylski and Bańbura, 1989; Kornijów, 1997; Lik et al., 2017). The availability of physical refuges, as well as an association with substrate resources, could be responsible for weak effects of vertebrate predation on zoobenthic community structure (Tolonen et al., 2003). In addition, the widespread occurrence of morphological, physiological and behavioral defences against vertebrate predation among benthic invertebrates might also mask the effects of predation (Strayer, 1991). Among size-structured invertebrate assemblages, small but exposed chironomid larvae may be more available to fish than larger individuals, possibly because they are the least proficient tube builders, and occur closer to the sediment surface; thus fish may have a greater effect on densities of small chironomids than on large, more deeply buried larvae (Hershey, 1985).

The aim of the study was to investigate the food resources partitioning (prey size and abundance) among small-sized fish i.e. perch *Perca fluviatilis*, ruffe *Gymnocephalus cernua*, and roach *Rutilus rutilus* coexisting in ephemeral habitat type of macrophyte patches in the riverine bed in the large lowland river under influence of reservoir dam operation. We focus on two periods (2004, 2011) which are distinct as regards percentage of SAM coverage and biomass of associated fauna in the tailwater. We tested the following hypothesis: fish species foraging on three ecological groups, i.e. zooplankton, benthos and epiphyton, selected food items in respect of prey abundance and size.

In this paper, we presented specific indicators determining fish feeding patterns that can be useful in the assessment of exploitation of food resources by fish in freshwater ecosystems.

## 2. Study area

The River Warta rises 380 m above sea level, is 808 km long and debouches into the River Oder 13 m above sea level. Its catchment area is ca. 53,710 km<sup>2</sup> and its slope ranges from 2.0 to 1.0‰ in the upper course, and from 0.3 to 0.1‰ in the middle and lower courses (IMGW, 2007). The study site was located in a seventh stream order section of this lowland alluvial river (Strahler, 1957, Fig. 1), about 1.5 km downstream of the Jeziorsko Reservoir, which started functioning in 1986. In the reservoir's tailwater, the Warta River was approximately 70 m wide, with a maximum depth at 1.2 m. Similarly to previous studies that focused on zoobenthos (Grzybkowska et al., 1990, 2003) we examined macrophyte habitat located between the marginal zone and the mid-river channel. Such habitat was structured mainly by two aquatic plants species, i.e. sago *Stuckenia pectinata* and shining pondweed *Potamogeton lucens*, the abundance of which strictly depends on discharge regulations (Grzybkowska et al., 2003). The largest patch of vegetation, covering nearly 56% of river bed was observed in 2004, when low summer discharge was noted. Contrarily, in late spring 2011 lower, 30% coverage was recorded as a result of strong discharge fluctuations. In consequence, a different pondweeds biomass, i.e. 149.4 g dry weight m<sup>-2</sup> in 2004 and 91.2 g d w m<sup>-2</sup> in 2011, was observed (Grzybkowska et al., 2017).

## 3. Material and methods

Sampling was conducted from May to August in 2004 (9 occasions) and from June to August in 2011 (7 occasions). Samples of

zooplankton, epiphyton, zoobenthos and fish from the tailwater site were collected along the macrophyte zone within an area measuring 40 × 2.5 m. To estimate the amount of dry weight of macrophytes growing in the study site, a special frame (0.5 × 0.7 m) was placed randomly on the riverine bottom and all the *S. pectinata* within the frame was collected. This procedure was repeated three times on each sampling occasion. In the laboratory, the pondweeds were dried for 24 h at 65 °C to estimate their dry weight per 1 m<sup>2</sup> (d w m<sup>-2</sup>).

To evaluate the biomass of zooplankton (mainly Cladocera), 0.03 m<sup>3</sup> samples of river water were filtered through a plankton net of 50 µm mesh size, and preserved in 4% formalin. In the laboratory, zooplankton were identified to genus or species, counted, and their biomass estimated on the basis of body length. Zooplankton (Cladocera and Copepoda) were classified into three size classes of body length and weight:

1. large-sized: *Leptodora kindtii* (Focke) (mean length of 2.10 mm, wet weight of 0.032 mg; Rosen, 1981) and Copepoda (mean length of 1.92 mm, wet weight of 0.338 mg; Bottrell et al., 1976)
2. mid-sized: *Daphnia galeata* Sars (mean length of 1.06 mm, wet weight of 0.053 mg; Bottrell et al., 1976)
3. small-sized: *Bosmina coregoni* Baird (mean length of 0.43 mm, wet weight of 0.019 mg; Persson and Ekbohm, 1980) and *Chydorus sphaericus* (O. F. Müller) (mean length of 0.38 mm, wet weight of 0.027 mg; Rosen, 1981).

Five subsamples of the epiphytic fauna settled on *S. pectinata* were collected on each sampling occasion. Each of the subsamples consisted of three fragments of stems (about 200 mm long) cut off below the water surface, stored in plastic containers, and preserved in 4% formalin in the field. In the laboratory, the plant material was removed from the containers and the invertebrates were washed off the plants, sorted by hand, identified to species level when possible, counted, and their wet mass assessed. The data were recalculated to estimate the biomass of epiphytic invertebrates per 1 m<sup>2</sup> of *S. pectinata* covering the river substrate on each sampling occasion.

Each benthic sample consisted of five subsamples uniformly distributed within the habitat. A subsample covered 100 cm<sup>2</sup> of streambed area, and was collected with a tubular sampler with a catching area of 10 cm<sup>2</sup>. Macroinvertebrates in the samples were sorted from detritus and benthic sediments by hand and preserved in 4% formalin. All invertebrates from these quantitative samples were counted and their wet mass assessed; these data were used to estimate the biomass of zoobenthos (g m<sup>-2</sup>). Where possible, groups of macroinvertebrates were classified to the lowest taxonomic level, while chironomids were identified to species level. As the identification of chironomid larvae to species was often impossible, we reared immature stages in the laboratory from additional qualitative samples taken on each sampling occasion to obtain larval and pupal skins and imagines for identification.

Both benthic and epiphytic chironomids were classified into three size classes of body weight:

1. large-sized (6–12 mg): *Macropelopia* sp. (Tanypodinae); *Chironomus riparius* Meigen and *Glyptotendipes cauliginellus* (Kieffer) (Chironominae-Chironomini);
2. mid-sized (1–6 mg): *Ablabesmyia monilis* (Linnaeus) and *Procladius* sp. (Tanypodinae); *Cricotopus* spp. and *Rheocricotopus* sp. (Orthocladinae); *Dicrotendipes nervosus* (Staeger), *Microtendipes chloris* (Meigen), *Cryptochironomus defectus* (Kieffer), *Endochironomus dispar* (Meigen), *Endochironomus albipennis* (Meigen), *Parachironomus gracilior* (Kieffer), *Stictochironomus histrio* Fabricius, *Paracladoplema camptolabis* (Kieffer) and *Polypedilum* spp. (Chironominae-Chironomini);
3. small-sized (≤ 1 mg): *Cladotanytarsus mancus* (Walker), *Paratanytarsus* sp., *Rheotanytarsus* sp. and *Tanytarsus* sp. (Chironominae-Tanytarsini).

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