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MOLLUSC COMMUNITIES OF LOWLAND RIVERS AND OXBOW LAKES IN AGRICULTURAL AREAS WITH ANTHROPOGENICALLY ELEVATED NUTRIENT CONCENTRATION

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ABSTRACT: Anthropogenically elevated nutrient concentration occurs in both ground and surface waters and cause grave environmental problems which are responsible for the degradation of water quality on a worldwide scale. The progressive anthropogenic pressure caused by prolonged agricultural activity has a negative effect on most water ecosystems including rivers and reservoirs. Freshwater molluscs, which are hololimnic organisms, have a limited mobility; therefore, they are good bioindicators of changes in their habitats. They reflect the abiotic or biotic state of water habitats, which represents the impact of environmental change on the habitat, community as well as on the ecosystem. A long-term survey of mollusc communities revealed the occurrence of 44 species including some rare, vulnerable, threatened or legally protected species, e.g. *Borysthenia naticina*, *Unio crassus*, *Anodonta cygnea*, *Pseudanodonta complanata* or *Sphaerium rivicola*, in the Wkra River, its tributaries and selected oxbow lakes. In the Wkra River catchment area, the mollusc communities, including *U. crassus* (Habitats Directive Natura 2000), are under a combined effect of several environmental factors. Conductivity, hardness, current velocity, river width, temperature, nutrient concentration and the size of sediment particles were the parameters most associated (statistically significant according to the forward selection results) with the distribution of mollusc species. The forward selection results showed that the concentration of dissolved oxygen and the occurrence of macrophytes also exerted a significant influence on the distribution of Mollusca. Among environmental factors, anthropogenically elevated concentrations of nitrites and nitrates in the water were the most predictive parameters that negatively influenced (stressors) the structure of mollusc communities. The novel findings of this survey showed that some mollusc species could tolerate wider ranges of some environmental factors and higher concentrations of nutrients than had been expected. *U. crassus* is more tolerant of a relatively high concentration of nitrates in the water than was previously expected, as well as of nitrites and phosphates, and it can survive in rivers with relatively high nutrient concentration. The present results confirm both the field and the toxicological research on the negative effect of nitrites and nitrates, which is caused by their toxicity to freshwater molluscs. The number of individuals, number of mollusc species and density decreased dramatically within the last year of the survey in the Wkra River and its tributaries. This phenomenon seems to be a direct result of the nutrient enrichment in the Wkra River catchment area, the regulation of riverbeds or an indirect result, e.g. lack of primary or appropriate host fishes for unionid species. The declining trend in the global mollusc population has probably begun to occur in the Wkra River catchment area.

KEY WORDS: Mollusca, environmental factors, nutrient enrichment, lowland river, nitrates, nitrites, CCA analysis, agricultural areas

CONTENTS

INTRODUCTION

1.1. ANTHROPOGENIC SOURCES OF NUTRIENT ENRICHMENT IN SURFACE AND GROUNDWATERS OF AGRICULTURAL AREAS

In lotic ecosystems, eutrophication is defined as an increase in a nutritive factor or factors that leads to greater whole-system heterotrophic or autotrophic metabolism. When defining the trophic status of these ecosystems, allochthonous and autochthonous sources of carbon, inorganic and organic forms of nutrients, such as nitrogen and phosphorus, should be taken into consideration (Dodds 2006).

Anthropogenic supply of phosphorus into a river includes both direct and indirect wastewater discharges from sewage treatment works and more local input such as leakage from septic tanks located near river channels, runoff from the use of fertilisers in agricultural areas or runoff from farmyards (Neal et al. 2010). Inorganic nitrogen enters freshwater ecosystems via anthropogenic points (livestock, agricultural operations, municipal and industrial sewage effluents) and diffuse sources (fertilisation, manure) of pollution (Camargo et al. 2005, Soucek & Dickinson 2012). The atmospheric deposition of inorganic nitrogen has dramatically increased due to the extensive use of nitrogen fertilisers and the combustion of fossil fuels. As a result, anthropogenically elevated concentrations of nitrates occur in both ground and surface waters and cause the most prevalent environmental problems that are responsible for the degradation of water quality on a worldwide scale (Camargo et al. 2005). Nutrient input from livestock farming includes manure, fertilisers and feed concentrates for animals. Nutrients are exported from grasslands via runoff, in solutions and in association with particles and colloids. Livestock farming is an important source of phosphorus and nitrogen in lowland rivers which contributes to the eutrophication of surface water resources (Jarvie et al. 2008). Depending on the type of soils, intensive livestock farming (especially on poorly drained heavy clay soils with direct access to the stream channel) constitutes the greatest risk of diffuse-source phosphorus loss in all agricultural areas. Compared with arable farming, livestock farming produces a substantially higher diffuse source of all phosphorus fractions (Jarvie et al. 2008, 2010). The loss of nitrates depends mainly on the texture of the soil (Pastuszak et al. 2012). In Europe, nutrient loads from point sources (industry, wastewater treatment plants) and diffuse sources (agricultural areas) are major environmental concerns for the Baltic Sea. Agricultural areas are the main contributors of diffuse nutrient emissions, which are influenced by land abandonment, crops, soil types, artificial drainage, fertilisers and tillage, as well as the intensity of production (fertiliser application rate, choice of crops, livestock density, etc.) (Mewes 2012).

In Europe, the excessive use of fertilisers constitutes an environmental risk, for which a common strategy is needed in order to control the problems that arise from intensive livestock production and that agricultural policy has to take greater account of environmental policy (Council Directive 91/676/EEC). According to the European Union Water Framework Directive (EU WFD) (Directive 2000/60/EC), both nitrates and phosphates, which contribute to eutrophication, are included in the indicative list of the main pollutants. For example, between 2004 and 2007, the yearly total amount of mineral nitrogen fertiliser consumption increased from 11.4 to 12.1 million tons in the 27 Member States of the EU, while the amount of nitrogen from animal husbandry spread on agricultural soil decreased annually from 9.4 to 9.1 million tons. The contribution of nitrogen loads from agriculture into surface waters has decreased in the EU; however, it still remains high. Agriculture in the Member States of the EU is responsible for over 50% of the total nitrogen discharge into surface waters.

The implementation of the regulatory requirements of the EU WFD in order to achieve a good surface water status by 2015 (Directive 2000/60/EC) through a 75% reduction in the total nitrogen and phosphorus loads was proposed in Poland. Achievement of this target will mean the restoration of a good state of surface and groundwaters and the implementation of a programme to combat the eutrophication of the Baltic Sea (The National Environmental Policy 2008). The Council Directive (Council Directive 91/676/EEC) (hereafter referred to as the Nitrates Directive) has the objective of reducing the water pollution that is caused or induced by nitrates from agricultural sources in order to protect health, living resources and aquatic ecosystems and to prevent any further pollution.

In Poland, few areas show elevated concentrations of nitrates, i.e. above 25 mg NO_3^- dm^{-3} in surface waters (Report EU 2011). However, in the largest Polish rivers, i.e. in the Vistula and Odra, nutrient concentrations and loads showed significant declining trends in 1988–2008 (Pastuszak et al. 2012). According to the requirements of the Nitrates Directive, all Member States including Poland should designate areas on their territories as the Nitrate Vulnerable Zones (NVZs) within two years following the notification of this directive. It is also necessary to establish and implement actions which reduce water pollution from nitrogen compounds in the NVZs. The NVZs are designated according to the monitoring results, which indicate that both surface freshwaters and groundwaters in these zones are affected by nitrate pollution originating from agricultural activities and contain or may contain more than 50 mg NO_3^- dm^{-3} nitrates (Council Directive 91/676/EEC, Dz.U. 2002). Therefore, 21 areas were designated as the NVZs in the territory of Poland in 2004. The number of the NVZs was reduced between 2008 and 2012, and today 19 areas are left. The total area of the NVZs amounts to 4,623.4 km^2, which comprises about 1.49% of the total territory of the country. In accordance with the Nitrates Directive, the concentrations of nitrates in surface freshwaters and groundwaters should be monitored continuously within these zones. However, the concentrations of nitrates are still higher in surface freshwaters and groundwaters than the trigger threshold value set in the Nitrates Directive, i.e. 50 mg NO_3^- dm^{-3}, both within and outside the zones. Alongside the Nitrates Directive, the Rules of Good Agricultural Practice (Kodeks 2004), which have the objectives of reducing pollution by nitrates with respect to conditions in different regions, should be implemented. The Rules of Good Agricultural Practice contain, e.g. appropriate periods for the use of fertilisers, the effect of using fertilisers near water courses, storage vessels for livestock manures, measures to prevent water pollution by runoff and seepage of liquids that contain manure and effluent from stored plant materials (silage) into the surface waters and groundwaters, and land use management (Council Directive 91/676/EEC, Kodeks 2004). A survey of Kupiec et al. (2008), which was carried out within one of the NVZs in Poland, showed that the majority of farms used intensive agricultural production and that they also specialised in pig farming. Many farms lacked buildings and equipment for the proper storage of manure, which led to the introduction of large quantities of nutrients into agro-ecosystems and caused a surplus of nutrients. Despite the fact that the majority of farmers complied with the recommendations of the Rules of Good Agricultural Practice, the level of fertilisation in the catchment area was high (Kupiec et al. 2008).

In Poland, more than 55% of the households are connected to sewage treatment plants, suggesting that 45% of households discharge sewage directly into surface waters (Alterra 2007). Pollution of surface freshwaters and groundwaters and its assessment require the analysis of the sources of nitrates, including farming systems, the density of the production of livestock, the applying of fertilisers, soil type and hydrology.

Many barns, farmyards and manure heaps are considered to be within the category of "micro hot spots" as sources of nitrate pollution. The losses of nitrates that leach from such farms and manure heaps constitute up to 40% of the total leaching losses. The leakages of nitrogen from livestock farms, manure heaps and farmyards are the major sources of its concentration in surface freshwaters and groundwaters. For example, the concentration of nitrates in the groundwater near the manure heaps may range from 0 to 1,400 mg NO_3^- dm^{-3} (Alterra 2007).

In Poland, most soils are light-textured sand soils, which are vulnerable to nitrates leaching losses. Many streams and rivers flow through agricultural areas, surrounding lakes and ponds and are additionally intersected by drainage ditches. Due to the seasonal variations in rainfall and evapotranspiration, these surface freshwaters have variable water levels. Thus, temporary flooding and close contact between land and surface waters give way to the transfer of nitrates from agricultural areas into surface waters (Alterra 2007). In 2009, agricultural land including arable land (38.7%), meadows (7.9%), pastures (2.3%), orchards (1.1%) and others (1.6%) constituted 51.6% (18.98 million ha) of the total Polish territory. From 2004 to 2009 the use of nitrogen and phosphate fertilisers increased from 56.3 to 66.3 and from 20.4 to 22.8 kg per 1 ha of agricultural land, respectively (Statistical Yearbook of Agriculture 2011).

The degradation of lowland rivers, caused by prolonged agricultural activity, is a problem worldwide (Harding et al. 1999, Ulén et al. 2007, Jarvie et al. 2008, 2010, Maret et al. 2010, Virbickas et al. 2011, Mewes 2012, Woodward et al. 2012). Nutrient enrichment alters the structure of macroinvertebrate communities, increases macroinvertebrate abundance, while a greater organic carbon load causes dominance by pollution-tolerant macroinvertebrates. When the load of nitrogen, phosphorus and carbon is simultaneously high, the highest rates of carbon consumption and the greatest biomass of heterotrophic organisms occur (Dodds 2006). Rivers in agricultural areas contain the highest concentrations of nutrients from the application of fertilisers and manure and have commonly altered riparian and instream habitats which can influence nutrient uptake and algal growth (Maret et al. 2010). Eutrophication alters the structure of aquatic plant communities and standing crop in rivers, and may also enhance flood risk by increasing the macrophyte biomass, which in turn physically impedes the water flow (O'Hare et al. 2010). River eutrophication resulting from phosphorus and nitrogen enrichment leads to the deterioration of water quality for potable supply and amenities and causes a reduction in aquatic biodiversity. In rivers in agricultural areas, anthropogenic sources of nutrient enrichment increase primary productivity, which may result in excessive plant growth followed by severe diurnal variations in dissolved oxygen and a reduction in the available instream habitat.

These eutrophication factors can lead to reduction in aquatic biodiversity that favours invasive species over native species (Maret et al. 2010). In streams with a predominance of agricultural land (pasture and arable) in their catchment areas, pH is more alkaline (Townsend et al. 1983).

Many small water bodies were lost in the last century. Those that remain, are faced with increasing anthropopressure by many factors including area drainage, fertilisation, water pollution, urban development, increased transport infrastructure or simply with natural succession. Small water bodies support a greater number of aquatic macroinvertebrate species than river systems, including particularly rare species which are of specific conservation interest (Wood et al. 2003). For example, both the Bug River and its oxbow lakes support a habitat for aquatic insects; however, Lechowski & Buczyński (2006) recorded a higher number of species in the oxbow lakes (18 species) than in the Bug River. Oxbow lakes, which constitute an essential part of the river ecosystem, increase the biodiversity by providing important habitats for diverse macrophytes and macroinvertebrates. Therefore, under Annex 1 to Council Directive 92/43/EEC (1992) of the EU and its consolidation version of 01.01.2007, oxbow lakes are protected natural habitats (code 3150) and constitute a refuge for many species that are of interest in the European Union.

1.2. SOME ASPECTS OF FUNCTIONAL ROLE OF MOLLUSCA IN FRESHWATER ECOSYSTEMS

An increase in agricultural intensity negatively affects the structure of macroinvertebrate communities including Mollusca. The values of some metrics, e.g. the total taxa richness, number of sensitive taxa, % collector filterers or % Coleoptera respond negatively to increase in the cattle grazing (Braccia & Voshell 2007). Among macroinvertebrates, freshwater molluscs, which are hololimnic organisms that are present in the waters throughout their whole life cycle, are especially vulnerable to water pollution. In freshwater ecosystems, molluscs constitute an important link in the circulation of organic matter and nutrients; participate in the self-purification processes of the water, nutrient cycles (accumulate and release nutrients) or accumulate heavy metals. Freshwater molluscs are an important link in the circulation of organic matter because they are the first order consumers that mainly provide a food basis for consumers of the second order. Gastropods are primary consumers or detritus eaters (Økland 1990). Pulmonate gastropods can achieve a net uptake of the dissolved organic carbon (DOC) produced by macrophytes and metabolise it (Thomas & Kowalczyk 1997). Bivalves filter phytoplankton, bacteria and particulate organic matter from the water column, and therefore increase the water clarity. Some species supplement suspension feeding from the water column by feeding on organic matter and bacteria, filtering the interstitial water in the sediments, and by deposit feeding (pedal feeding utilising cilia on the foot). The volume of water filtered by unionids within dense beds can equal or exceed the daily stream discharge. For example, *Anodonta anatina* is able to filter up to 2.9 dm^3 of water during 1 hour, while *Unio pictorum* can filter up to 4.6 dm^3 $1h^{-1}$ and *Unio crassus* up to 4.1 dm^3 $1h^{-1}$ (Kryger & Riisgård 1988). It was shown that filtration by unionids with a density of up to 350 indiv. m^{-2} led to "biological oligotrophication" by decreasing the biomass of phytoplankton and the concentration of total phosphorus in a river (the River Spree, Germany). Fluviatile unionid species filter more bacteria than pond species (the gills of fluviatile species are characterised by more complex and larger cirri compared to the gills of pond species) (Vaughn & Hakenkamp 2001). On the other hand, bivalves affect the nutrient dynamics through deposition of faeces and pseudofaeces, excretion of remineralised nutrients in forms that are readily available to phytoplankton. Burrowing bivalves increase the oxygen content in sediments and release nutrients from the sediments into the water column. Mussel beds provide habitats for epiphytic and epizoic organisms. Both gastropods and bivalves are hosts of both adult forms and larval stages of parasites. Molluscs constitute a food resource for macroinvertebrates, fish, waterfowl and mammals (Włosik-Bieńczak 1997, Lewandowski & Stańczykowska 2000, Vaughn & Hakenkamp 2001, Strayer 2013).

Some studies have suggested that an increase in nutrient loads is associated with an increase in the production and biomass of algae, bacteria and fungi, which constitute the food resources for bivalves, and cause an increase in their growth, fecundity and population abundance. However, analyses of the productivity of the ecosystem and nutrient concentrations in the water were not carried out in these studies (Strayer 2013). The load of nitrogen and phosphorus in the water, which support algal productivity, have probably increased 2–20 fold, especially in Europe and North America. Therefore, this phenomenon may have affected bivalve populations around the world and may suggest that increases in nutrient load in the water should lead to the development of larger mussel populations. The increase in nutrient load in freshwater ecosystems seems likely to have both positive and negative effects on the bivalve populations, but only up to the threshold value of the nutrient concentration. Extreme eutrophication radically changes the species composition of the algal communities towards the species which may diminish the nutritional quality of the algae (the content of fatty acid decreases and leads to strong, negative consequences for consumers). Thus, the insufficient food quality as well as the occurrence of toxic algae leads to diminishing mussel populations that are exposed to high nutrient concentration. Furthermore, high nutrient concentrations cause a higher rate of photosynthesis which in turn drives up pH and shifts the balance away from NH_4^+ towards the more toxic NH_3. Strayer (2013) hypothesised that at first, the size of mussel populations increased together with the nutrient concentration in the water and the availability of food resources (phytoplankton) and then decreased as the nutrient load increased. The threshold values of acceptable nutrient concentration as well as an early warning that the "death threshold" values are about to be exceeded, in relation to mussels, have not yet been specified (Strayer 2013).

Progressive anthropogenic pressure degrades the integrity of the majority of water ecosystems including rivers and reservoirs. Freshwater molluscs have a limited capability of movement; therefore, they are good bioindicators of changes in these water environments. They readily reflect the abiotic or biotic state of water environments, which represents the impact of environmental change on the habitat or community as well as on the ecosystem. Bivalves are especially useful for monitoring freshwater ecosystems because of their sedentary behaviour and filtration activity, through which great quantities of pollutants are taken up and accumulated in both their soft tissues and shells. Therefore, many authors consider Mollusca to be very good bioindicators of, for example, heavy metal pollution (Jurkiewicz-Karnkowska 1994, Naimo 1995, Tessier et al. 1996, Labrot et al. 1999, Flessas et al. 2000, Callil & Junk 2001, Beone et al. 2011, Waykar & Deshmukh 2012, Bodin et al. 2013), radionuclide pollution (Frantsevich et al. 1995, Ravera & Riccardi 1997), and indicators of biodegradable pollution (Mouthon & Charvet 1999). Some bivalve species can be effective bioindicators of the contamination of freshwater habitats with the oocysts of *Cryptosporidium* and the cysts of *Giardia* (Graczyk et al. 2003, Domingo et al. 2012) as well as bioindicators of the influx of non-purified sewage into river ecosystems (Lukashev 2010). Freshwater molluscs are sensitive to low pH, humic substances, sulphur hydrogen, heavy metals, ammonium and biodegradable pollution. Gastropods are more sensitive to low pH than fish. Therefore, gastropods can give an early warning of acidification because in the case of a pH decrease they will disappear from freshwater ecosystems before fish (Økland 1983). Freshwater molluscs, both gastropods and bivalves, are used in the assessment of water quality in saprobic system and biotic indices including multimetric indices (Aqem Consortium 2002, Rolauffs et al. 2004, Gabriels et al. 2010, Bis & Mikulec 2013) in all of the Member States of the EU in accordance with the requirements of the EU Water Framework Directive (EU WFD) (Directive 2000/60/EC).

The main threats to freshwater molluscs are the modification and destruction of habitats, including water eutrophication and pollution, regulation of rivers, habitat loss resulting from drainage, loss of marshy habitats, drying up of bogs and eutrophication of reservoirs including oxbow lakes (Mouthon 1996a, Piechocki 1996, 2002, 2008, Dyduch-Falniowska & Zając 2002, Bogan 2008, Shuhaimi-Othman et al. 2012). Freshwater Bivalvia are threatened, and their populations are declining at a global scale, and among them freshwater mussels (Bivalvia: Unionacea) are one of the most endangered groups in the world (Bogan 2008, Vaughn 2012). This phenomenon has been best documented for rivers (Aldridge 2000, Grabow et al. 2000, Cosgrove & Hastie 2001, Weber 2005, Gangloff & Feminella 2007, Mouthon & Daufresne 2008, Sousa et al. 2008, 2011). Therefore, the most endangered species of molluscs are protected under European legislation, i.e. the Council Directive 92/43/EEC (1992) of the EU or more local laws, i.e. the Polish legislation (Dz.U. 2011a), as well as being included into the European Red List of Nonmarine Molluscs (Cuttelod et al. 2011), the IUCN Red List of Threatened Species (Iucn 2012) and the Red List of Threatened Animals in Poland (Dyduch-Falniowska & Zając 2002, Piechocki 2002).

1.3. OBJECTIVES OF THE SURVEY

To date, no long-term survey on the environmental factors that determine the occurrence of molluscs in lowland rivers and oxbow lakes with a high nutrient enrichment, which includes the physical and chemical parameters of the water, bottom sediments and macrophytes, has been carried out. Therefore, due to the problems that are highlighted above and the facts related to the degradation of lowland rivers or reservoirs that are caused by prolonged agricultural activity and the unqestionable, functional role of Mollusca in freshwater ecosystems as well as the initial field recognition and choice of sampling sites or initial laboratory analysis before the fundamental field survey, the formulation the following objectives of the survey is justified.

The objectives of the survey were:

– to analyse the structure of mollusc communities and their changes in terms of the number of species, their density, dominance and constancy pattern as well as biodiversity in lowland rivers and selected oxbow lakes with anthropogenically elevated nutrient concentration; special attention

was paid to *Unio crassus*, a species which is of interest in the European Union,
- to determine the environmental factors which are the most predictive parameters that negatively (stressors) or positively (drivers) influence the structure of mollusc communities,
- to analyse any long-term, time-dependent drift and transient responses of the structure of mollusc communities to changes in environmental factors including nutrient enrichment in the Wkra River, its tributaries as well as oxbow lakes,
- to determine the ranges of the environmental variables which influence the structure of mollusc communities both in the rivers and in selected oxbow lakes in the Wkra River catchment area to the greatest extent,
- to check if the population of *U. crassus* can reproduce in the Wkra River and if there is recruitment of juveniles,
- to ascertain whether molluscs can be bioindicators of the water quality in lowland rivers in agricultural areas.

The hypotheses of the investigation were the following:
- among the environmental factors, the nutrient concentrations in the water (nitrites, nitrates, phosphates) are the most predictive parameters in the distribution of molluscs,
- the values of the environmental variables as well as the values of the indices calculated for the mollusc communities at sampling sites in rivers and selected oxbow lakes vary in space and time,
- in terms of the heterogeneity of microhabitats and environmental variables, the number of mollusc species, their density and the values of the diversity indices calculated for mollusc communities vary among the sampling sites in the upper, middle and lower courses of the Wkra River.

STUDY AREA

The Wkra River, which is a right-bank tributary of the Narew River, has a total length of 249.1 km and a catchment area of 5,322 km². The investigated catchment area consists of the Wkra River, seven oxbow lakes that are located in the lower course of the Wkra River and its six main tributaries: the Sona River, the Łydynia River, the Raciążnica River, the Mławka River, the Szkotówka River and the Naruszewka River. The rivers flow through agricultural areas of the Polish Lowlands within a natural physico-geographical region, i.e. the North Mazovian Lowland, Central Poland (Kondracki 2002), which comprises part of the Eastern Plains (Ecoregion 16) according to the division of the EU Water Framework Directive (EU FWD) (Directive 2000/60/EC) (Fig. 1). The Wkra River and its tributaries are typical

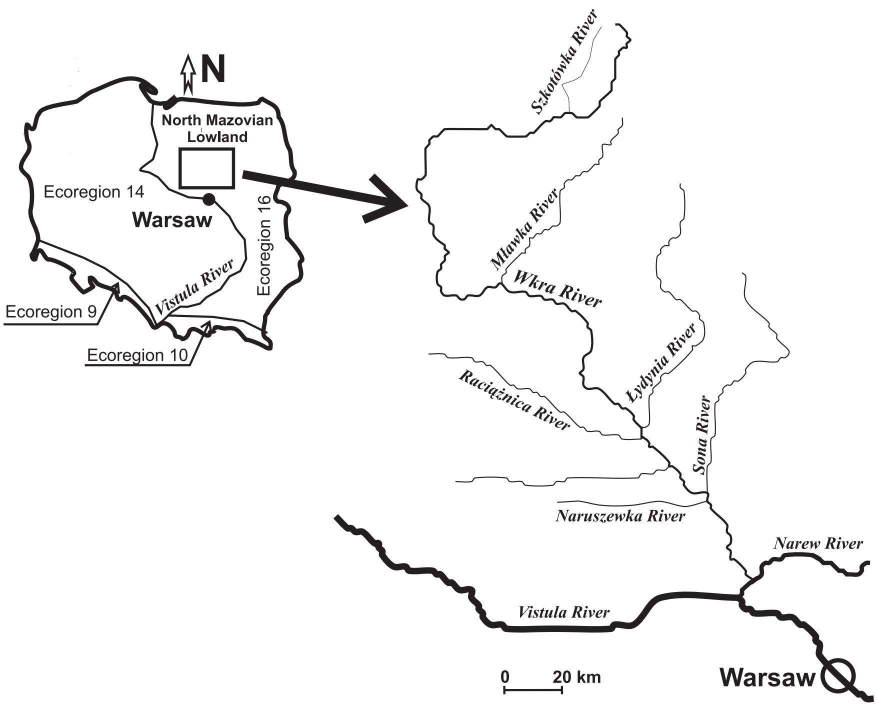


Fig. 1. Location of the study area (The North Mazovian Lowland, Central Poland) (Ecoregions for rivers and lakes: 9 – Central Highlands, 10 – the Carpathians, 14 – Central Plains, 16 – Eastern Plains)

lowland rivers (<200 m a.s.l.). The Sona River, the Łydynia River and the Mławka River are the left-bank tributaries, whereas the Szkotówka, the Raciążnica and the Naruszewka rivers are the right-bank tributaries of the Wkra River (Figs 2 and 3). The sources of the Wkra River are located in the area of reclaimed marshes near the Kownatki Lake. The upper course of the river, from the sources to Działdowo, is called the Nida River. The middle course, from Działdowo to the mouth of the Swojęcianka River, is called the Działdówka River. The lower course, which is called the Wkra River, runs from the mouth of the Swojęcianka River to its final outflow into the Narew River near Pomiechówek (Figs 2 and 3).

According to the EU WFD size typology based on the catchment area, the Wkra River is a large river, whereas its tributaries are medium-sized rivers. The characteristic morphometric features of the rivers and their typology based on the EU WFD requirements and adaptation to the Polish conditions (Błachuta et al. 2005, Report 2005) are shown in Table 1.

Arable land comprises most of the catchment area of the Wkra River, and forests comprise 20%. Cereals, mainly rye, potatoes, sugar beets, spring rape and agrimony are cultivated there.

The valley of the lower course of the Wkra River is included in the European Ecological Natura 2000 Network Programme of protected sites which represent areas of the highest value for natural habitats and rare, endangered or vulnerable plants and animal species in the European Community as "Dolina Wkry" (PLH 140005) (Fig. 3). This area, which comprises 24.0 ha, protects several types of habitats, among others a mixed lime-oak-hornbeam forest *Tilio-Carpinetum* and *Galio-Carpinetum*, an alluvial forest with *Alnus glutinosa* (L.) Gaertn. and *Fraxinus excelsior* L., brush communities (*Salicetum albo-fragilis* R. Tx. 1955, *Populetum albae* Br.-Bl. 1931, *Alnenion glutinoso-incanae* Oberd. 1953) as well as mammals that are of Community Importance: otter *Lutra lutra* (L.) and beaver *Castor fiber* L. The area of Natura 2000 (PLH 140005) shares some coverage with the

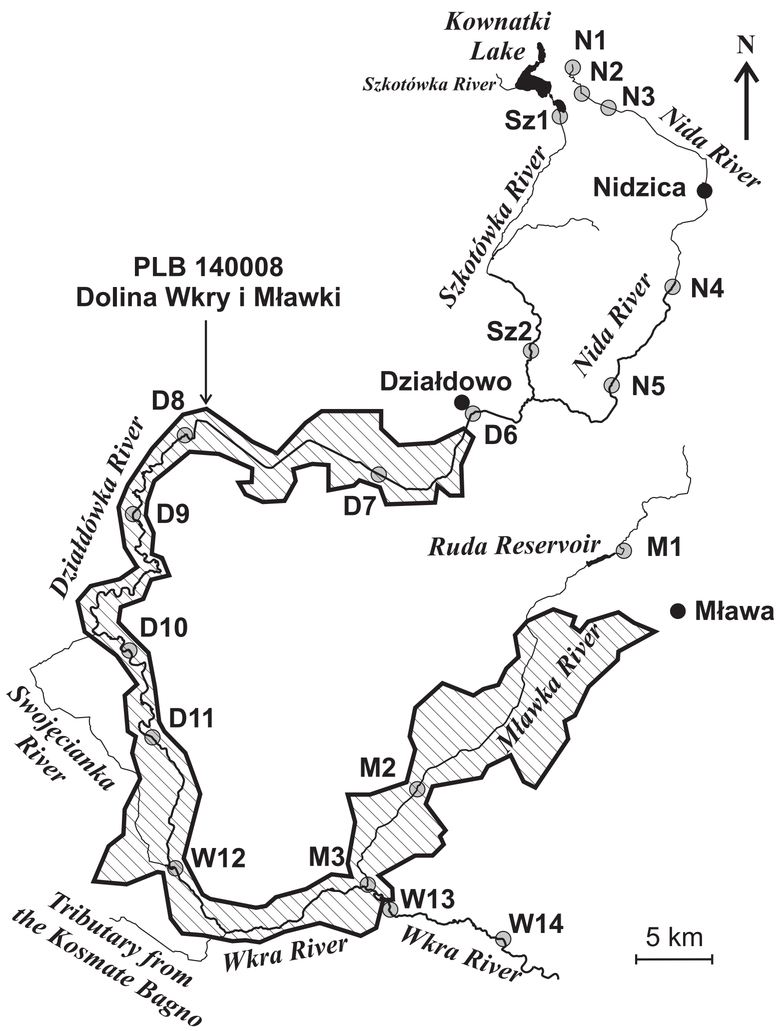


Fig. 2. Location of the sampling sites in the upper and middle course of the Wkra River and its tributaries

nature reserve "Dolina Wkry" which was established in 1991 in order to protect a part of the Wkra River valley (Fig. 3) (REPORT 2011). Under the Birds Directive (DIRECTIVE 2009/147/EC), according to the regulation of the Minister of the Environment (DZ.U. 2011b), 28,751.5 ha were designated as the Special Protected Area "Dolina Wkry i Mławki" (PLB 140008) (Fig. 2). Twenty eight bird species, which are listed in Annex I to the DIRECTIVE 2009/147/EC, were recorded in the "Dolina Wkry i Mławki", including *Botaurus stellaris* (L.), *Ardea purpurea* L., *Circus pygargus* (L.), *Crex crex* (L.) and *Philomachus pugnax* (L.). The species mentioned in Annex I should be subject to special conservation measures concerning their habitats. This area also constitutes the roost for 2,000 geese during their autumn and winter migration (ŁAWICKI et al. 2012).

The riverbeds of the Wkra River catchment area are regulated. For example, heavy morphological transformations have been done in the upper and middle courses of the Wkra River (from the sources to the mouth of the Mławka River), in the rivers Mławka (upper and lower courses), Łydynia (from Pławnica to the mouth as well as in the communes of Stupsk, Szydłowo and Grudusk), Szkotówka (from the Lipowska Struga to the mouth), and Raciążnica (from the sources to the Rokitnica) (REPORT 2005, PIELECH & KISIEL 2010). Some parts of the rivers

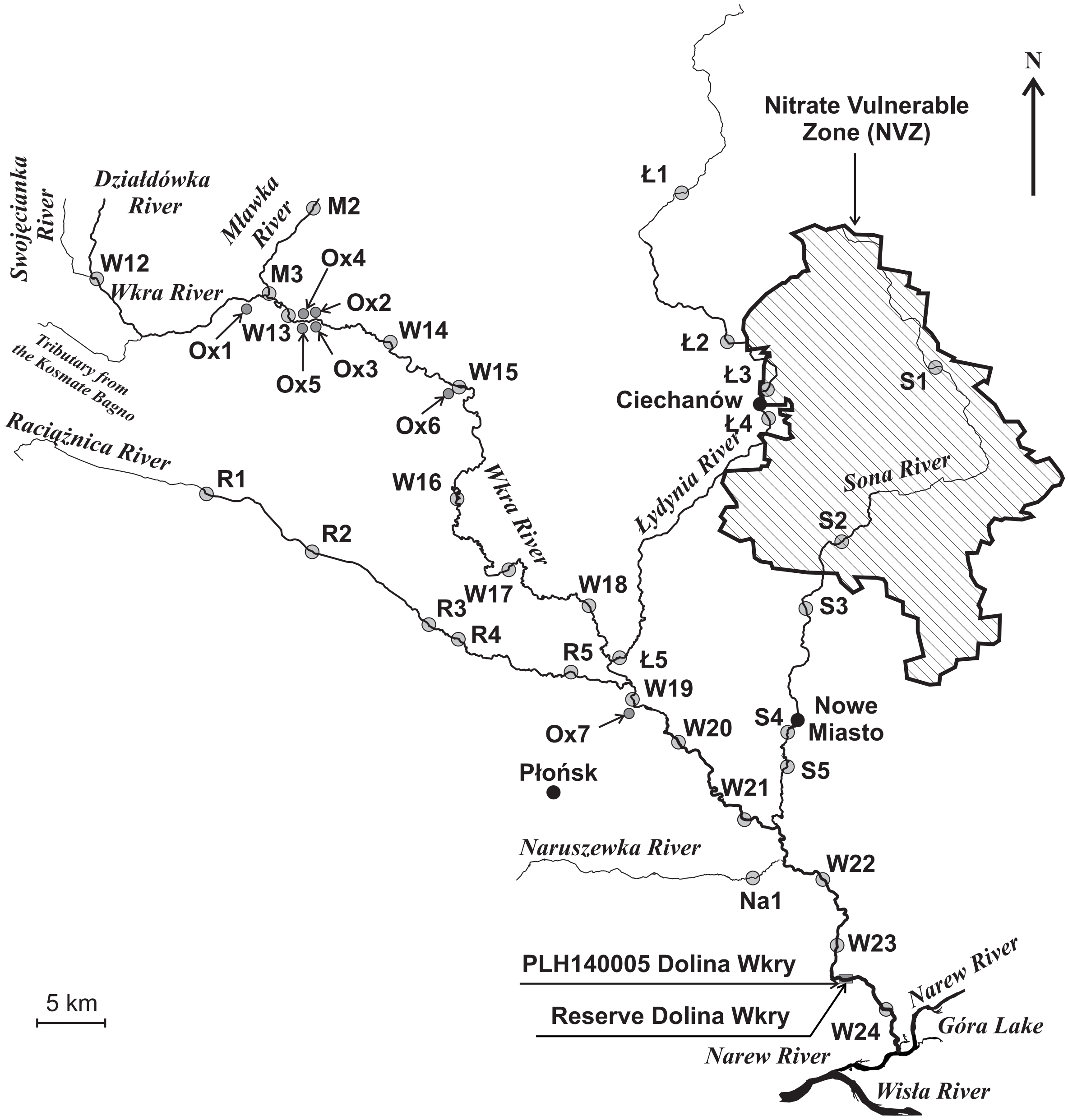


Fig. 3. Location of the sampling sites in the lower course of the Wkra River, its oxbow lakes and tributaries

Table 1. Characteristic morphometric features of the Wkra River and its tributaries

Parameter	Wkra River	Sona River	Łydynia River	Raciążnica River	Mławka River	Szkotówka River	Naruszewka River
River length (km)	249.1	63.7	72.0	56.9	43.4	25.3	23.4
Catchment area (km^2)	5,322.1	536.5	697.9	616.7	676.0	241.5	133.9
Size typology (catchment area)	large	medium	medium	medium	medium	medium	medium
Code of stream type	19, 24	17, 24	17, 19	17, 19, 24	24	24	17
River gradient (‰)	0.52	1.41	0.47	0.44	1.20	2.13	1.32
Elevation (m a.s.l.)	58–188	82–118	92–126	93–118	108–139	131–165	109
Sources	marshes near Lake Kownatki	near Koziczyn	near Choszczewka	near Kusek	near Mława	from Szkotowskie Lake	Radzyminek
Mouth	to the Narew River near Pomiechówek	near Popielżyn–Zawady	near Sochocin	near Sochocin	near Ratowo	near Działdowo	Nowa Wrona
River width (m)	0.5–60.0	0.8–15.0	3.0–15.0	1.5–8.0	2.0–12.0	2.0–6.0	4.0–6.0
Depth of collecting samples (m)	0.25–0.80	0.10–0.70	0.20–0.70	0.30–0.70	0.30–0.80	0.25–0.60	0.30–0.50

do not fulfil the EU WFD requirements of the good water quality because they are degraded by diffuse sources of pollution, especially by nitrates, i.e. the upper course of the Wkra River (from the sources to the tributary near Zagrzew) or some parts of the Raciążnica River.

The anthropogenic pressure in the Wkra River catchment area and adjacent areas is associated with the excessive application of fertilisers (fields, meadows, pastures), chemical pesticides, improper management of manure, melioration, wastewater discharge from sewage treatment works and food processing

Table 2. Characteristic morphometric features, location, geographic coordinates and codes of the sampling sites in the oxbow lakes of the Wkra River

Parameter	Oxbow lakes						
Location of the sampling site	Witowy	Radzanów	Radzanów	Radzanów	Radzanów	Strzegowo	Sochocin
Geographic coordinates	N 52°57.024' E 20°04.276'	N 52°56.519' E 20°06.021'	N 52°56.503' E 20°06.011'	N 52°56.511' E 20°05.855'	N 52°56.507' E 20°05.913'	N 52°53.396' E 20°16.905'	N 52°41.154' E 20°28.063'
Code	Ox1	Ox2	Ox3	Ox4	Ox5	Ox6	Ox7
Surface area (m^2)	1,100	1,000	1,200	1,350	1,100	1,000	300
Elevation (m a.s.l.)	108	97	113	98	108	104	84
Distance from the Wkra River (m)	102	60	46	65	153	25	200
Depth of samples collecting (m)	0.30–0.70	0.40–0.70	0.50–0.70	0.40–0.60	0.50–0.80	0.40–0.70	0.20–0.70
Surrounding area	meadows, pastures	meadows, pastures	meadows, pastures	meadows, pastures	meadows, pastures	wasteland, riverine bush	wasteland, scrub, detached houses

plants, leakage from septic tanks near river channels, lack of rural sewage treatment, discharge of domestic sewage directly into the rivers, improper storage of domestic sewage, etc. In some cases, sewage from rural households is discharged into the reservoirs and then into the rivers or is stocked in the fields or forests. In addition, illegal waste dumps in forests or near rivers and reservoirs increase the nutrient load in the surface waters (Report 2009, 2011).

According to the requirements of the Nitrates Directive (Council Directive 91/676/EEC), out of the 21 areas of the Nitrate Vulnerable Zones (NVZs) designated in Poland, only one was established in the Sona River catchment area. The NVZ amounts to 406.64 km^2 in the Sona River catchment area (Kzgw 2008) (Fig. 3). In the Sona River catchment area, the concentration of nitrates of agricultural origin is still higher in surface freshwaters than the trigger threshold value set in the Nitrates Directive, i.e. 50 mg NO_3^- dm^{-3}. Therefore, their concentrations are continuously monitored. The maximum nitrate concentration in the Sona River which was recorded at selected sampling sites by the local voivodeship authority (Report 2007, 2011, Kzgw 2008) was up to 108.83 mg NO_3^- dm^{-3} in 2004, 92.60 mg NO_3^- dm^{-3} in 2006 and 98.00 mg NO_3^- dm^{-3} in 2010. What is more troublesome is that no improvement of the water quality is expected in this area in the following years.

The oxbow lakes of the Wkra River, which have survived, are isolated from the main river channel (lentic type). Therefore, the groundwater supply from the hillslope, geological structure and agricultural land use are among the important factors that influence the quantity and quality composition of the waters that enter the oxbow lakes as well as on their biota including molluscs. The characteristic morphometric features, geographic coordinates of the sampling sites in the oxbow lakes of the Wkra River and their codes are shown in Table 2.

MATERIAL AND METHODS

The study was carried out from 2005 to 2009. The initial field recognition and choice of sampling sites took place in 2004 before the actual field survey. The sampling sites were selected to represent a wide range of environmental conditions and habitats along the mainstream of the Wkra River from the headwaters to the mouth, its main tributaries and selected oxbow lakes of the Wkra River (Figs 4–11).

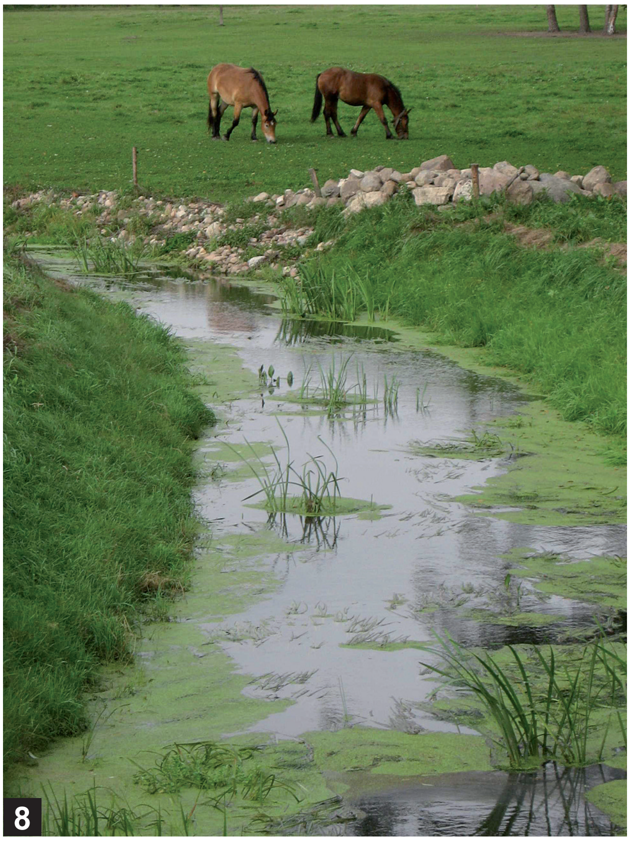
The sampling sites were chosen according to the following criteria:

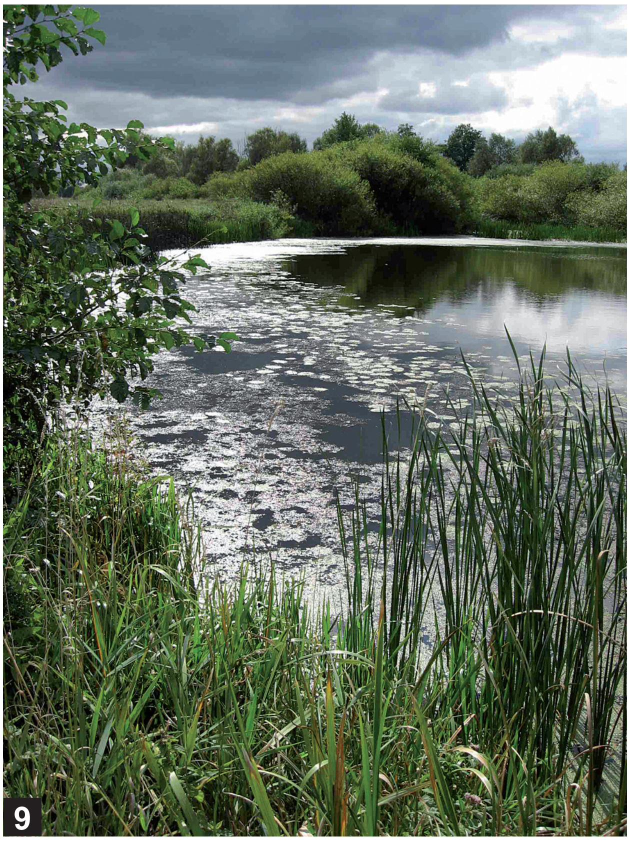
– the catchment area with respect to its physical, chemical, and biological attributes and altitude,
– the nature of catchment area (agricultural),
– the type of substratum including macrophytes and the heterogeneity of microhabitats,
– the water quality as indicated by the physical and chemical analyses performed earlier,
– the absence of any obvious point sources of pollution,
– the degree of regulation of the rivers,
– the previous recognition of the occurrence of mollusc species (mollusc sampling and their laboratory determination to the species level),
– accessibility of the riverbanks and oxbow lakes.

Mollusc samples were collected from the entire length of the rivers from the head to the mouth. In total, 45 sampling sites were chosen within the survey area: the Wkra River (24 sampling sites), the Sona River (5 sites), the Łydynia River (5 sites), the Raciążnica River (5 sites), the Mławka River (3 sites), the Szkotówka River (2 sites) and the Naruszewka River (1 site) (Figs 2 and 3). The number of sampling sites per river depended on the catchment area. Samples were taken once from each of the 45 sampling sites in the rivers in August each year (5 times at one sampling site from 2005 to 2009). In total, 225 samples were taken (from 45 sampling sites) within the survey area. Along the river, starting from the sources, five sampling sites (code N1–N5) were chosen in the upper course of the Wkra River (the Nida River). N1 stands for the first sampling site located in the Nida River, and N2 stands for the second sampling site located in the Nida River etc. The next six sampling sites, which are coded as D6–D11, were chosen in the middle course, i.e. in the Działdówka River. The sampling sites coded as W12–W24 were located in the lower course of the Wkra River. The sampling sites located in the Sona River were coded as S1–S5. The sampling sites located in the Łydynia River, the Raciążnica River, the Mławka River, the Szkotówka River and the Naruszewka River were coded as Ł1–Ł5, R1–R5, M1–M3, Sz1–Sz2, Na1, respectively (Figs 2 and 3). The location of the sampling sites, their codes and the coordinates for the Wkra River and its tributaries are presented in Tables 3 and 4.

Seven oxbow lakes of the Wkra River were selected and were investigated within the same period. These oxbow lakes were isolated from the main river channel (lentic type) (Fig. 3). As in the case of the rivers, samples were taken once from each of the seven oxbow lakes in August each year (5 times per sampling site from 2005 to 2009). In total, 35 samples were collected (from 7 sites) within the survey area. The sampling sites located in the oxbow lakes were coded as Ox1–Ox7 (Fig. 3).

Figs 4–7. Examples of mollusc habitats located in the Wkra River catchment area: 4 – the upper course of the Wkra River; 5 – the middle course of the Wkra River; 6, 7 – the lower course of the Wkra River. Photos: Iga Lewin





Figs 8–9. Examples of mollusc habitats located in the Wkra River catchment area: 8 – the Raciążnica River (tributary of the Wkra River); 9 – an oxbow lake of the Wkra River (Ox3). Photos: IGA LEWIN

Molluscs are more active and they grow and breed more intensively in summer, when the temperature of the water ranges from 20 to 25°C and macrophytes are well developed. Filtration rate of bivalves increases with temperature because their metabolic rate increases. According to DUSSART (1976), the highest seasonal abundance of molluscs was obtained in late summer. He recorded a late summer peak, especially in August, in the density and the number of mollusc species, followed by a decrease in autumn. DUSSART (1979), ØKLAND (1983), BENDELL & MCNICOL (1993), HORNBACH & DENEKA (1996) also carried out surveys on molluscs in relation to environmental factors in summer (June-September). A five-year study is a sufficient period in which to analyse any long-term, time-dependent drift as well as any transient responses of the structure of mollusc communities to any changes in environmental factors including nutrient enrichment in the Wkra River, its tributaries as well as oxbow lakes. A long term-study including repeated collection of samples permits a sufficient number of representative

Figs 10–11. Unionid mussels inhabiting a sandy substratum (sampling site W23, the Wkra River). Photos: IGA LEWIN

Table 3. Location, geographic coordinates and codes for the sampling sites in the Wkra River

No.	River	Location of the sampling site	Geographic coordinates	Code	Elevation (m)
1.	Nida (Wkra)	Januszkowo	N 53°25.825'; E 20°17.503'	N1	188
2.	Nida (Wkra)	Michałki	N 53°25.047'; E 20°18.131'	N2	184
3.	Nida (Wkra)	Rączki	N 53°24.510'; E 20°19.793'	N3	180
4.	Nida (Wkra)	Szymany	N 53°17.999'; E 20°23.075'	N4	158
5.	Nida (Wkra)	Kadyki	N 53°14.584'; E 20°19.419'	N5	153
6.	Działdówka (Wkra)	Kisiny	N 53°13.826'; E 20°11.451'	D6	147
7.	Działdówka (Wkra)	Gnojno	N 53°11.728'; E 20°05.654'	D7	140
8.	Działdówka (Wkra)	Nick	N 53°13.174'; E 19°54.350'	D8	136
9.	Działdówka (Wkra)	Zieluń	N 53°10.449'; E 19°51.290'	D9	126
10.	Działdówka (Wkra)	Bądzyń	N 53°05.521'; E 19°50.604'	D10	123
11.	Działdówka (Wkra)	Poniatowo	N 53°02.517'; E 19°52.090'	D11	118
12.	Wkra	Bieżuń	N 52°57.906'; E 19°53.243'	W12	116
13.	Wkra	Radzanów	N 52°56.517'; E 20°05.969'	W13	113
14.	Wkra	Radzimowice	N 52°55.203'; E 20°12.239'	W14	108
15.	Wkra	Strzegowo	N 52°53.394'; E 20°16.941'	W15	104
16.	Wkra	Glinojeck	N 52°48.986'; E 20°16.756'	W16	101
17.	Wkra	Dziektarzewo	N 52°46.097'; E 20°20.017'	W17	98
18.	Wkra	Malużyn	N 52°44.761'; E 20°25.124'	W18	88
19.	Wkra	Sochocin	N 52°40.986'; E 20°27.868'	W19	85
20.	Wkra	Bolęcin	N 52°39.324'; E 20°30.728'	W20	83
21.	Wkra	Joniec	N 52°36.139'; E 20°34.923'	W21	81
22.	Wkra	Cieksyn	N 52°33.715'; E 20°39.908'	W22	76
23.	Wkra	Śniadówko	N 52°31.145'; E 20°40.906'	W23	74
24.	Wkra	Pomiechówek	N 52°28.397'; E 20°44.303'	W24	58

samples to be obtained in order to perform a statistical estimation. Therefore, the choice of the late summer for collecting samples and five-year period of the study in the Wkra River catchment area are justified.

The mollusc samples were taken according to quantitative methods by placing a quadrat frame (25×25 cm) randomly on the substratum of the rivers and the oxbow lakes along the bank. Sixteen subsamples (one frame each) were taken at each sampling site, which in total constituted one sample (1 m^2), and included bottom sediments up to 5 cm, macrophytes and surface water. A D-frame net, with a sharp cutting edge and mesh size of 465 μm, was used for the sampling area enclosed within each quadrat frame.

The collected material was transported to the laboratory in plastic bags and plastic containers. The samples were washed using a 0.5 mm mesh sieve and then preserved in 75% ethanol. Molluscs were identified to the species level based on their morphological and anatomical features according to Piechocki (1979), Glöer & Meier-Brook (1998), Piechocki & Dyduch-Falniowska (1993), Jackiewicz (1998), Glöer (2002) and Killeen et al. (2004). The species nomenclature follows Glöer & Meier-Brook (1998) and Glöer (2002). The age of *U. crassus* was estimated according to Piechocki & Dyduch-Falniowska (1993) and Abraszewska-Kowalczyk (2002). Empty shells were excluded from the analyses. The density of molluscs was estimated as the number of individuals per square metre.

Immediately prior to mollusc sampling, water samples and bottom sediments were collected from each sampling site. Bottom sediments were taken using a core-type sampler.

Physical and chemical parameters of the water, i.e. conductivity (EC), total dissolved solids (TDS), temperature and pH, were measured in the field using a portable HI 9811-5 pH/EC/TDS/°C meter (Hanna Instruments, USA) and dissolved oxygen using CO-401 oxygen meter (Elmetron, Poland). Analysis of nitrites (adaptation of the EPA method 354.1), nitrates (adaptation of the cadmium

Table 4. Location, geographic coordinates and codes for the sampling sites in the tributaries of the Wkra River

No.	River	Location of the sampling sites	Geographic coordinates	Code	Elevation (m)
1.	Sona	Kołaczków	N 52°53.744'; E 20°48.159'	S1	118
2.	Sona	Sońsk	N 52°46.953'; E 20°41.878'	S2	101
3.	Sona	Łopacin	N 52°44.467'; E 20°39.283'	S3	99
4.	Sona	Nowe Miasto	N 52°39.358'; E 20°37.981'	S4	86
5.	Sona	Miszewo	N 52°38.194'; E 20°37.908'	S5	82
6.	Łydynia	Borzymy	N 53°00.872'; E 20°31.838'	Ł1	126
7.	Łydynia	Grzybowo	N 52°55.062'; E 20°34.544'	Ł2	117
8.	Łydynia	Ciechanów I	N 52°53.007'; E 20°37.095'	Ł3	113
9.	Łydynia	Ciechanów II	N 52°51.984'; E 20°37.168'	Ł4	108
10.	Łydynia	Gutarzewo	N 52°42.521'; E 20°27.369'	Ł5	92
11.	Raciążnica	Bielany	N 52°49.425'; E 20°00.336'	R1	118
12.	Raciążnica	Raciąż	N 52°47.009'; E 20°07.119'	R2	113
13.	Raciążnica	Kiełki	N 52°44.128'; E 20°14.617'	R3	100
14.	Raciążnica	Rzewin	N 52°43.432'; E 20°16.627'	R4	98
15.	Raciążnica	Smardzewo	N 52°41.996'; E 20°23.880'	R5	93
16.	Mławka	Mławka	N 53°08.927'; E 20°19.975'	M1	139
17.	Mławka	Szreńsk	N 53°00.561'; E 20°07.710'	M2	113
18.	Mławka	Ratowo	N 52°57.236'; E 20°04.741'	M3	108
19.	Szkotówka	Szkotowo	N 53°24.295'; E 20°16.792'	Sz1	165
20.	Szkotówka	Sarnowo	N 53°16.132'; E 20°14.768'	Sz2	131
21.	Naruszewka	Józefowo	N 52°33.756'; E 20°35.440'	Na1	109

reduction method), phosphates (adaptation of the ascorbic acid method) and chlorides concentration in the water (a titrimetric determination using a mercury nitrate solution) were carried out in the laboratory using photometers and reagents by Hanna Instruments or Merck. Alkalinity was determined using the acidimetric titration method against phenolphthalein or a mixed indicator, whereas hardness was determined according to an adaptation of the EPA method 130.1 (Hermanowicz et al. 1999, Eaton & Franson 2005). The total hardness was considered within the following ranges: soft water (from 90 to 180 mg $CaCO_3$ dm^{-3}), medium hard water (above 180 to 270 mg $CaCO_3$ dm^{-3}), significantly hard (above 270 to 360 mg $CaCO_3$ dm^{-3}), hard water (above 360 to 450 mg $CaCO_3$ dm^{-3}) and very hard water (above 450 mg $CaCO_3$ dm^{-3}) (Hermanowicz et al. 1999).

The width of the river and the surface velocity of the water at each sampling site were measured according to the methods of Hauer & Lamberti (2007). Organic matter content in the bottom sediments was determined by the loss on ignition (LOI) method, which measures weight loss in the bottom sediment samples after burning at 550°C according to PN-88/B-04481 (Myślińska 2001).

Bottom sediment organic matter content was characterised according to the classification of Verdonschot (2001):

- very high organic matter content (> 10.0%),
- high organic matter content (4.1–10.0%),
- medium organic matter content (1.0–4.0%),
- ow organic matter content (< 1.0%).

The grain size composition of the bottom sediments was determined by using both the sieve and aerometric methods. The particle size classification was characterised according to the Wentworth scale (Allan & Castillo 2007):

- boulder > 256.0 mm,
- cobble: large 128.0–256.0 mm, small 64.0–128.0 mm,
- gravel: very coarse 32.0–64.0 mm, coarse 16.0–32.0 mm, medium 8.0–16.0 mm, fine 4.0–8.0 mm, very fine 2.0–4.0 mm,
- sand: very coarse 1.0–2.0 mm, coarse 0.50–1.0 mm, medium 0.25–0.50 mm, fine 0.125–0.25 mm, very fine 0.0625–0.125 mm,
- silt < 0.0625 mm.

Macrophytes were recorded at the sampling sites and identified to species according to Szafer et al. (1986). The nomenclature of macrophyte species was updated according to Mirek et al. (2002).

The structure of the mollusc communities was analysed using the following indices (GÓRNY & GRÜM 1981):

1. Dominance (D%)

$$D = \frac{n_a}{n} \cdot 100$$

where
n_a – the number of individuals of species a,
n – the total number of individuals in the sample.

The value of the dominance index D was divided into 5 classes:

eudominants > 10.0% of sample, dominants 5.1–10.0% of sample, subdominants 2.1–5.0% of sample, recedents 1.1–2.0% of sample and subrecedents ≤ 1.0% of sample.

2. Constancy (C%)

$$C = \frac{N_a}{N} \cdot 100$$

where
N_a – the number of samples which contain species a,
N – the total number of samples.

The value of the constancy index C was divided into 4 classes:

euconstants 75.1–100.0% of samples, constants 50.1–75.0% of samples, accessory species 25.1–50.0% of samples, and accedents ≤ 25.0% of samples.

3. The total number of species (S)

4. Density

The density of Mollusca was estimated as the number of individuals per square metre.

5. The Shannon-Wiener index (H') (MCCUNE & GRACE 2002, HAUER & LAMBERTI 2007):

$$H' = -\sum_{i=1}^{S} P_i \log_2 P_i$$

where

$P_i = \frac{N_i}{N}$ – the proportion of individuals of species i.

6. The maximum Shannon-Wiener index (H_{max})

$$H_{max} = \log_2 S$$

where
S – the total number of species

7. The Evenness index (J'):

$$J' = \frac{H'}{\log_2 S}$$

where
H' – the value of the Shannon-Wiener index, S – the total number of species.

Canonical ordination analyses for relating the species composition of the molluscs or indices to the environmental variables were carried out using CANOCO for Windows version 4.5 (TER BRAAK & ŠMILAUER 2002). The appropriate type of analysis was chosen to analyse the species data using DCA (Detrended Correspondence Analysis) and the length of the gradient. Preliminary DCA on the biological data revealed that the gradient length exceeded 4 SD (the standard deviation), thus indicating that the biological data exhibited a strong unimodal response to underlying environmental variables. Therefore, a unimodal direct ordination CCA (Canonical Correspondence Analysis) with a forward selection was used for the reduction of the large set of environmental variables. Species that occurred at less than 10% of the sampling sites were excluded from the statistical analyses following a preliminary exploration of their influence in the initial DCA analysis. The bottom sediments were categorised as the weight ratios of grain size fractions (the Wentworth scale: see above) expressed as percentage. The data for the environmental variables were log-transformed.

The statistical significance of the relationship between the mollusc species and the environmental variables or indices was evaluated using the Monte Carlo permutation test (499 permutations) (TER BRAAK & ŠMILAUER 2002).

The significance of differences in the values of the environmental variables and the values of indices in sampling sites between the rivers or the years of the survey was calculated using the Kruskal-Wallis one-way ANOVA test (ELLIOTT 1993, FOVLER et al. 1998) and multiple comparison post hoc test using Statistica version 9. Analogous calculation was carried out for sampling sites in the oxbow lakes.

The value of the environmental variables did not reveal a normal distribution according to the Lilliefors test of normality (Statistica version 9) and this justified the use of a non-parametric test.

The location of the sampling sites and their codes were plotted based on the Raster Map of the Hydrographical Division of Poland according to the geographic coordinates (Figs 2 and 3).

Figures 1–3 were drawn using CorelDraw version 12. The hydrological data of the area of the survey was obtained from the Raster Map of the Hydrographical Division of Poland, which was prepared by the Institute of Meteorology and Water Management on a contract with the Minister of the Environment and financial support by the National Fund for Environmental Protection and Water Management (Poland).

Nomenclature of the mollusc species and the abbreviations of the species names which were used in both the statistical analysis and the figures are shown in Table 5.

Table 5. List of collected species with abbreviations of their names used both in the statistical analysis and in the figures

Species	Abbreviation	Species	Abbreviation
Theodoxus fluviatilis (Linnaeus, 1758)	*T.fl*	*Anisus spirorbis* (Linnaeus, 1758)	*A.sp*
Viviparus contectus (Millet, 1813)	*V.co*	*Anisus vortex* (Linnaeus, 1758)	*A.vo*
Viviparus viviparus (Linnaeus, 1758)	*V.vi*	*Bathyomphalus contortus* (Linnaeus, 1758)	*B.co*
Bithynia tentaculata (Linnaeus, 1758)	*B.te*	*Gyraulus albus* (O. F. Müller, 1774)	*G.al*
Potamopyrgus antipodarum (J. E. Gray, 1843)	*P.an*	*Gyraulus crista* (Linnaeus, 1758)	*G.cr*
Lithoglyphus naticoides (C. Pfeiffer, 1828)	*L.na*	*Hippeutis complanatus* (Linnaeus, 1758)	*H.co*
Valvata cristata O. F. Müller, 1774	*V.cr*	*Segmentina nitida* (O. F. Müller, 1774)	*S.ni*
Valvata macrostoma Mörch, 1864	*V.ma*	*Ancylus fluviatilis* O. F. Müller, 1774	*A.fl*
Valvata piscinalis (O. F. Müller, 1774)	*V.pi*	*Unio pictorum* (Linnaeus, 1758)	*U.pi*
Borysthenia naticina (Menke, 1845)	*B.na*	*Unio tumidus* Philipsson, 1788	*U.tu*
Acroloxus lacustris (Linnaeus, 1758)	*A.la*	*Unio crassus* Philipsson, 1788	*U.cr*
Galba truncatula (O. F. Müller, 1774)	*G.tr*	*Anodonta cygnea* (Linnaeus, 1758)	*A.cy*
Stagnicola palustris (O. F. Müller, 1774)	*S.pa*	*Anodonta anatina* (Linnaeus, 1758)	*A.an*
Stagnicola corvus (Gmelin, 1791)	*S.cor*	*Pseudanodonta complanata* (Rossmaessler, 1835)	*P.com*
Radix auricularia (Linnaeus, 1758)	*R.au*	*Sphaerium corneum* (Linnaeus, 1758)	*S.co*
Radix balthica (Linnaeus, 1758)	*R.ba*	*Sphaerium rivicola* (Lamarck, 1818)	*S.ri*
Lymnaea stagnalis (Linnaeus, 1758)	*L.st*	*Musculium lacustre* (O. F. Müller, 1774)	*M.la*
Physa fontinalis (Linnaeus, 1758)	*P.fo*	*Pisidium amnicum* (O. F. Müller, 1774)	*P.am*
Aplexa hypnorum (Linnaeus, 1758)	*A.hy*	*Pisidium henslowanum* (Sheppard, 1823)	*P.he*
Planorbarius corneus (Linnaeus, 1758)	*P.co*	*Pisidium supinum* A. Schmidt, 1851	*P.sup*
Planorbis planorbis (Linnaeus, 1758)	*P.pl*	*Pisidium subtruncatum* Malm, 1855	*P.su*
Planorbis carinatus (O. F. Müller, 1774)	*P.car*	*Pisidium casertanum* (Poli, 1791)	*P.ca*

RESULTS

4.1. THE WKRA RIVER AND ITS TRIBUTARIES

4.1.1. PHYSICAL AND CHEMICAL PARAMETERS OF THE WATER, VELOCITY, ORGANIC MATTER CONTENT IN BOTTOM SEDIMENTS AND GRAIN SIZE COMPOSITION

Data summarising the physical and chemical parameters of the water, velocity, the organic matter content in the bottom sediments and the grain size composition in the Wkra River and its tributaries are given in Table 6.

The water temperature ranged from 13.5 to 24.0°C in the Wkra River and from 14.0 to 22.0°C in its tributaries. The lowest conductivity value of 220 μS cm^{-1} was recorded in the Szkotówka River, whereas the maximum value of 1,290 μS cm^{-1} – in the Sona River (Table 6). The water conductivity and the concentration of total dissolved solids were considerably higher in both the Sona and the Łydynia rivers.

The water pH was circum-neutral, slightly alkaline to alkaline and it ranged from 6.7 to 9.1. A wider range and higher pH values were recorded in the Wkra River than in its tributaries. The dissolved oxygen concentration in the water was relatively low at some of the sampling sites in the Wkra River. The nutrient concentration was relatively high or very high: nitrites up to 0.59 mg NO_2^- dm^{-3} in the Wkra River and up to 2.64 mg NO_2^- dm^{-3} in the Raciążnica River, and nitrates up to 142.20 mg NO_3^- dm^{-3} in the Sona River. A higher nitrate concentration was recorded in the tributaries: Sona, Łydynia and Szkotówka, than in the Wkra River. The phosphate concentration was also relatively high in both the Wkra River (up to 6.00 mg PO_4^{3-} dm^{-3}) and its tributaries (up to 11.00 mg PO_4^{3-} dm^{-3} in the Mławka River). Very high concentrations of phosphates were recorded in the tributaries, particularly in the Sona, Raciążnica and Mławka. The hardness ranged from 104 (soft water) to 665 mg $CaCO_3$ dm^{-3} (very hard water) in the Szkotówka and Sona rivers, respectively. The water was soft to hard in the Wkra River. A wider range of hardness was recorded in the Sona River (from medium hard to very hard). The minimum alkalinity of 45 mg $CaCO_3$ dm^{-3} was recorded in the Łydynia River, whereas the maximum value of 500 mg $CaCO_3$

Table 6. Physical and chemical parameters of the water (ranges), velocity, organic matter content (%) in bottom sediments and grain size composition (%) in the Wkra River and its tributaries

Parameter	Wkra River	Sona River	Łydynia River	Raciążnica River	Mławka River	Szkotówka River	Naruszewka River
Temperature (°C)	13.5–24.0	16.0–21.5	14.0–21.0	17.0–22.0	15.0–22.0	16.5–22.0	14.0–16.0
Conductivity (μS cm^{-1})	260–780	460–1,290	510–1,070	550–980	370–550	220–500	560–600
Total dissolved solids (mg dm^{-3})	130–390	230–640	250–530	270–490	180–270	100–250	270–290
pH	6.7–9.1	6.8–8.5	7.2–8.4	7.3–8.3	7.2–7.9	7.3–8.4	7.5–7.7
Dissolved oxygen (mg dm^{-3})	0.5–11.8	2.6–11.2	2.1–11.0	2.5–12.3	4.9–11.5	4.1–11.1	4.9–5.5
Nitrites (mg NO_2^- dm^{-3})	0.02–0.59	0.03–0.56	0.10–0.53	0.03–2.64	0.10–0.53	0.10–0.33	0.30–0.50
Nitrates (mg NO_3^- dm^{-3})	0.44–87.71	3.10–142.20	0.44–136.00	6.20–56.26	5.76–27.02	0.30–94.80	11.08–23.48
Phosphates (mg PO_4^{3-} dm^{-3})	0.05–6.00	0.10–6.40	0.23–2.60	0.20–9.56	0.24–11.00	0.11–1.13	0.62–0.74
Hardness (mg $CaCO_3$ dm^{-3})	145–395	245–665	200–375	225–450	215–405	104–281	318–330
Alkalinity (mg $CaCO_3$ dm^{-3})	150–270	140–500	45–360	175–410	155–240	75–215	260–275
Chlorides (mg Cl^- dm^{-3})	10–38	18–88	8–44	18–102	18–32	13–28	18–21
Velocity (m s^{-1})	0.01–2.00	0.00–0.28	0.05–1.20	0.00–0.40	0.10–0.50	0.08–0.40	0.08–0.60
Organic matter (%)	0.29–26.45	0.42–24.02	0.54–4.22	0.53–12.24	0.64–11.52	0.89–2.71	0.51–1.30
Grain size (diameter):							
above 20.0 mm	0.8–29.4%	1.2–23.6%	0.4–30.4%	3.4–16.6%	1.6–28.8%	1.8–5.5%	0.1–1.2%
10.0–20.0 mm	0.4–38.5%	0.9–15.7%	0.5–16.2%	2.6–15.8%	4.5–9.1%	0.5–2.2%	0.1–21.8%
5.0–10.0 mm	0.5–23.2%	0.2–18.3%	1.9–21.2%	4.3–28.8%	3.2–16.8%	3.0–10.5%	0.2–12.3%
2.0–5.0 mm	0.2–26.6%	1.7–17.5%	2.6–24.7%	0.6–10.3%	3.6–23.0%	3.6–11.5%	1.6–3.2%
1.0–2.0 mm	0.3–25.3%	2.2–24.7%	0.2–13.7%	1.4–14.2%	4.4–18.9%	2.9–22.2%	2.2–6.1%
0.80–1.0 mm	0.2–40.1%	1.5–19.4%	0.4–17.2%	0.6–8.3%	2.6–6.5%	3.4–19.2%	0.6–8.0%
0.50–0.80 mm	0.4–66.4%	2.3–22.9%	0.7–72.5%	0.2–34.4%	4.2–12.4%	6.9–34.1%	3.3–10.2%
0.25–0.50 mm	1.2–77.6%	10.8–53.9%	7.5–89.7%	4.9–76.1%	16.6–39.8%	26.5–52.7%	40.1–69.2%
0.10–0.25 mm	1.3–70.4%	1.0–72.2%	1.4–69.4%	4.4–82.1%	1.2–28.4%	1.0–7.4%	4.0–14.2%
0.05–0.10 mm	0.1–16.3%	0.4–6.6%	0.1–24.4%	0.1–10.6%	0.3–4.5%	0.3–1.7%	1.0–2.3%
below 0.05 mm	0.3–16.1%	0.2–10.2%	1.8–4.3%	0.5–4.9%	0.1–1.2%	0.2–1.4%	0.1–1.5%

dm^{-3} was obtained in the Sona River. The maximum values of chloride concentration in the tributaries (up to 88 mg Cl^- dm^{-3} in the Sona and up to 102 mg Cl^- dm^{-3} in the Rciążnica) were higher than in the Wkra River (up to 38 mg Cl^- dm^{-3}). The current velocity ranged from 0.01 to 2.00 m s^{-1} in the Wkra River, and from 0.00 to 1.20 m s^{-1} in its tributaries (Table 6). The Kruskal-Wallis one-way ANOVA test revealed statistically significant differences in the average values of the environmental variables: temperature (H = 18.81, p = 0.005), pH (H = 27.24, p = 0.0001), conductivity (H = 103.05, p = 0.0001), total dissolved solids (H = 98.80, p = 0.0001), dissolved oxygen (H = 15.19, p = 0.02), nitrites (H = 18.48, p = 0.005), phosphates (H = 32.28, p = 0.0001), hardness (H = 67.05, p = 0.0001), alkalinity (H = 52.50, p = 0.0001), chlorides (H = 52.71, p = 0.0001) as well as water velocity (H = 42.37, p = 0.0001), the depth of the samples collected (H = 37.88, p = 0.0001) and river width (H = 62.50, p = 0.0001) at the sampling sites between the rivers.

The organic matter content in the bottom sediments ranged from 0.29% (low organic matter content) to 26.45% (very high organic matter content) (Table 6). The differences in the average organic

Table **7**. Changes in physical and chemical parameters of the water (ranges) in the Wkra River from 2005 to 2009

Parameter	Years of the survey				
	2005	2006	2007	2008	2009
Conductivity (μS cm^{-1})	420–520	430–540	450–530	440–600	260–780
Total dissolved solids (mg dm^{-3})	210–260	210–270	220–260	220–300	130–390
pH	6.7–9.1	7.1–8.8	7.1–7.8	7.0–8.3	7.6–8.5
Dissolved oxygen (mg dm^{-3})	4.2–10.1	4.4–11.8	4.2–8.1	0.5–8.1	1.1–9.6
Nitrites (mg NO_2^- dm^{-3})	0.02–0.16	0.03–0.20	0.03–0.50	0.16–0.59	0.03–0.36
Nitrates (mg NO_3^- dm^{-3})	4.86–67.78	12.85–85.06	3.99–87.71	7.53–51.39	0.44–19.94
Phosphates (mg PO_4^{3-} dm^{-3})	0.68–1.63	0.05–2.74	0.45–3.01	0.37–6.00	0.17–1.75
Hardness (mg $CaCO_3$ dm^{-3})	200–270	204–282	145–365	205–395	179–333
Alkalinity (mg $CaCO_3$ dm^{-3})	170–235	150–215	175–240	150–255	160–270
Chlorides (mg Cl^- dm^{-3})	10–38	11–22	16–31	17–30	16–38

matter content at the sampling sites between the rivers were not statistically significant (the Kruskal-Wallis one-way ANOVA, $H = 5.30$, $p = 0.51$).

Medium-grained sand (0.25–0.50 mm) constituted the highest percentage share (39.8–89.7%) in the grain size composition of the bottom sediments in the Wkra River and its tributaries: Łydynia, Mławka, Szkotówka and Naruszewka. Fine sand (0.10–0.25 mm) constituted the highest percentage (72.2–82.1%) in the grain size composition in the Sona River and in the Raciążnica River. Particles above 20.0 mm constituted from 0.1 to 30.4% in the grain size composition of the bottom sediments (Table 6).

The Kruskal-Wallis one-way ANOVA test revealed statistically significant differences in the average grain size 1.0–2.0 mm ($H = 22.29$, $p = 0.0005$), 0.8–1.0 mm ($H = 16.64$, $p = 0.005$), 0.10–0.25 mm ($H = 12.47$, $p = 0.03$) and 0.05–0.10 mm ($H = 11.71$, $p = 0.04$) at the sampling sites between the rivers.

The conductivity and the concentration of the total dissolved solids increased in the Wkra River between 2005 and 2009. The hardness, the concentrations of nitrites and phosphates showed an increasing tendency up to 2008. In 2009 the nutrient concentration was lower compared to 2008 (Table 7). Statistically significant differences in the average temperature ($H = 17.26$, $p = 0.01$), pH ($H = 78.68$, $p = 0.0001$), conductivity ($H = 56.74$, $p = 0.0001$), total dissolved solids ($H = 51.82$, $p = 0.0001$), dissolved oxygen ($H = 55.89$, $p = 0.0001$), nitrites ($H = 105.99$, $p = 0.0001$), nitrates ($H = 76.04$, $p = 0.0001$), phosphates ($H = 43.59$, $p = 0.0001$), hardness ($H = 45.04$, $p = 0.0001$), alkalinity ($H = 23.80$, $p = 0.002$), chlorides ($H = 50.31$, $p = 0.0001$) and current velocity ($H = 29.54$, $p = 0.0003$), the depth of the samples collected ($H = 40.52$, $p = 0.0001$), river width ($H = 55.23$, $p = 0.0001$) and % of organic matter content in the bottom sediments ($H = 12.76$, $p = 0.047$) were recorded between the sampling sites in the years of the survey. The differences in the average grain size at the sampling sites in the years of the survey were statistically significant: 1.0–2.0 mm ($H = 44.91$, $p = 0.003$), 0.8–1.0 mm ($H = 36.75$, $p = 0.02$).

4.1.2. MACROPHYTES

In total, 53 macrophyte taxa were recorded at the sampling sites in the Wkra River and its tributaries during the survey. The number of taxa was 45 in the Wkra River, whereas it ranged from 11 (Mławka) to 31 (Łydynia) in its tributaries (Table 8). *Hottonia palustris* occurred at only one sampling site (Raciążnica). *Batrachium aquatile* (Łydynia), *B. fluitans* (Wkra, Sona, Łydynia) and *Nuphar lutea* are legally protected in Poland (Dz.U. 2012a). Watercourses harbouring submerged or floating-leaved macrophytes, including *B. fluitans* or *B. aquatile*, constitute natural habitat types that are of interest to the EU Community; their conservation requires designation as special conservation areas within the Habitats Directive Natura 2000 as the type "Lowland and highland rivers with crawfoot communities (*Ranunculion fluitantis*)" (code 3260) (Council Directive 92/43/EEC 1992, Dz.U. 2012b).

Only two species of mosses and two taxa of algae, i.e. *Amblystegium riparium*, *Fontinalis antipyretica*, *Cladophora* sp. and *Enteromorpha* sp., were found. Macrophyte species were absent from the sampling sites in the Naruszewka River.

4.1.3. STRUCTURE OF MOLLUSC COMMUNITIES AND VALUES OF INDICES IN THE WKRA RIVER AND ITS TRIBUTARIES

In total, 44 mollusc species were recorded in the Wkra River and its tributaries: 30 gastropod species and 14 bivalve species. The number of mollusc species was 40 in the Wkra River, and it ranged from 10 (Naruszewka) to 32 (Sona) in the tributaries (Table 9).

Table 8. Macrophyte occurrence at the sampling sites in the Wkra River and its tributaries

Species	Wkra River	Sona River	Łydynia River	Raciążnica River	Mławka River	Szkotówka River
Acorus calamus L.	x	x		x		
Alisma plantago–aquatica L.	x					
Amblystegium riparium (Hedw). Schimp.	x	x	x	x	x	
Batrachium aquatile (L.) Dumort.			x			
Batrachium circinatum (Sibth.) Fr.	x		x			
Batrachium fluitans (Lam.) Wimm.	x	x	x			
Berula erecta (Huds.) Coville	x	x	x	x	x	x
Bidens tripartita L.	x	x	x			
Butomus umbellatus L.	x	x	x	x		
Callitriche cophocarpa Sendtn.	x		x	x		x
Ceratophyllum demersum L. S. Str.	x	x		x		x
Cladophora sp.	x		x			
Eleocharis palustris (L.) Roem. & Schult.				x		
Elodea canadensis Michx.	x	x	x	x	x	x
Enteromorpha sp.	x					
Equisetum sp.	x			x		
Eupatorium cannabinum L.	x	x	x			
Fontinalis antipyretica L. ex Hedw.	x	x	x	x		
Galium palustre L.	x			x		
Glyceria fluitans (L.) R. Br.	x	x	x	x	x	x
Glyceria maxima (Hartm.) Holmb.	x	x	x	x		
Hottonia palustris L.				x		
Hydrocharis morsus–ranae L.	x			x		
Lemna gibba L.	x	x				
Lemna minor L.	x	x	x	x	x	x
Lemna trisulca L.	x	x	x	x	x	x
Lycopus europaeus L.			x			
Lysimachia nummularia L.	x	x				
Lythrum salicaria L.	x			x		
Mentha aquatica L.	x	x	x	x		
Myosotis palustris (L.) L. Emend. Rchb.	x	x		x		x
Myriophyllum spicatum L.	x					x
Nuphar lutea (L.) Sibth. & Sm.	x	x	x	x		
Phalaris arundinacea L.	x	x	x	x		x
Phragmites australis (Cav.) Trin. Ex Steud.	x				x	
Potamogeton crispus L.	x		x			
Potamogeton lucens L.	x					
Potamogeton pectinatus L.	x	x	x	x	x	
Potamogeton perfoliatus L.	x					
Potamogeton praelongus Wulfen					x	
Rorippa amphibia (L.) Besser	x	x	x		x	
Rumex hydrolapathum Huds.	x		x	x		
Sagittaria sagittifolia L.	x	x	x	x		
Schoenoplectus lacustris (L.) Palla	x					
Scirpus sylvaticus L.		x				
Scrophularia umbrosa Dumort.			x			
Sium latifolium L.	x	x	x	x		
Sparganium emersum Rehmann	x	x	x	x	x	x
Sparganium erectum L. Emend. Rchb. S. Str.	x		x	x		x
Spirodela polyrhiza (L.) Schleid.	x	x	x	x		
Stachys palustris L.		x				
Typha latifolia L.	x	x	x	x		
Veronica anagallis–aquatica L.	x					x
Total number of taxa	45	29	31	30	11	13

Table 9. Values of dominance (*D*%) and constancy (*C*%) indices calculated for the mollusc communities in the Wkra River and its tributaries

Species	Wkra River		Sona River		Łydynia River		Raciążnica River		Mławka River		Szkotówka River		Naruszewka River	
	D	*C*	*D*	*C*	*D*	*C*	*D*	*C*	*D*	*C*	*D*	*C*	*D*	*C*
Theodoxus fluviatilis	7.7	25.8	–	–	–	–	–	–	–	–	–	–	–	–
Viviparus contectus	0.3	11.7	0.5	16.0	0.5	8.0	0.1	8.0	–	–	–	–	–	–
Viviparus viviparus	11.9	41.7	16.9	40.0	–	–	0.6	20.0	–	–	–	–	–	–
Bithynia tentaculata	14.1	79.2	16.2	60.0	22.5	84.0	28.6	92.0	9.7	26.7	2.5	80.0	–	–
Potamopyrgus antipodarum	0.2	3.3	–	–	–	–	–	–	–	–	–	–	–	–
Lithoglyphus naticoides	1.1	3.3	–	–	–	–	–	–	–	–	–	–	–	–
Valvata cristata	0.2	2.5	–	–	–	–	–	–	–	–	–	–	–	–
Valvata macrostoma	–	–	0.1	8.0	–	–	–	–	–	–	–	–	–	–
Valvata piscinalis	4.0	43.3	4.5	48.0	10.2	40.0	3.0	40.0	1.5	20.0	1.9	30.0	2.6	60.0
Borysthenia naticina	0.1	5.0	–	–	–	–	–	–	–	–	–	–	–	–
Acroloxus lacustris	0.5	5.0	0.7	8.0	0.4	16.0	2.4	28.0	–	–	–	–	–	–
Galba truncatula	0.2	5.0	0.2	12.0	–	–	–	–	0.9	13.3	0.4	20.0	–	–
Stagnicola palustris	2.5	15.0	–	–	0.1	8.0	–	–	0.6	13.3	6.5	70.0	–	–
Stagnicola corvus	0.1	1.7	0.9	16.0	–	–	0.2	12.0	–	–	–	–	–	–
Radix auricularia	0.2	10.0	1.0	32.0	0.7	16.0	0.3	12.0	–	–	–	–	–	–
Radix balthica	7.4	72.5	4.2	44.0	14.1	72.0	3.8	36.0	19.9	33.3	39.0	90.0	22.7	80.0
Lymnaea stagnalis	1.3	41.7	1.0	44.0	4.2	60.0	2.6	44.0	4.1	20.0	0.7	50.0	1.0	60.0
Physa fontinalis	1.0	22.5	1.4	28.0	3.2	40.0	3.2	52.0	15.0	40.0	2.2	50.0	–	–
Aplexa hypnorum	–	–	0.1	8.0	–	–	–	–	1.2	13.3	0.2	10.0	–	–
Planorbarius corneus	0.6	22.5	17.6	48.0	0.6	12.0	3.6	40.0	2.9	26.7	0.9	50.0	–	–
Planorbis planorbis	0.1	7.5	18.2	16.0	0.4	20.0	1.1	24.0	–	–	0.8	50.0	–	–
Planorbis carinatus	0.1	0.8	–	–	–	–	–	–	–	–	–	–	–	–
Anisus spirorbis	0.1	0.8	0.2	12.0	–	–	0.1	8.0	–	–	0.4	20.0	1.0	40.0
Anisus vortex	3.2	37.5	0.6	28.0	4.7	28.0	4.5	52.0	4.1	26.7	6.9	90.0	–	–
Bathyomphalus contortus	0.6	10.8	0.1	8.0	0.1	4.0	0.1	12.0	–	–	2.4	80.0	–	–
Gyraulus albus	2.2	25.8	2.6	40.0	1.0	28.0	0.6	20.0	–	–	0.1	10.0	–	–
Gyraulus crista	0.1	0.8	–	–	–	–	–	–	–	–	–		–	–
Hippeutis complanatus	–	–	–	–	–	–	–	–	–	–	0.2	10.0	–	–
Segmentina nitida	0.1	0.8	0.8	12.0	–	–	0.1	8.0	0.6	13.3	0.6	20.0	–	–
Ancylus fluviatilis	4.9	35.0	1.6	20.0	17.5	32.0	0.8	8.0	10.6	26.7	0.6	30.0	16.5	100.0
Unio pictorum	2.4	35.8	1.1	32.0	0.8	20.0	0.8	24.0	1.5	20.0	–	–	–	–
Unio tumidus	2.1	40.8	0.2	16.0	0.6	12.0	0.1	8.0	0.6	13.3	–	–	–	–
Unio crassus	0.8	22.5	–	–	–	–	–	–	–	–	–	–	–	–
Anodonta cygnea	0.6	15.8	1.2	36.0	–	–	–	–	1.2	20.0	–	–	–	–
Anodonta anatina	2.1	39.2	1.6	36.0	0.4	20.0	0.6	24.0	1.2	20.0	–	–	–	–
Pseudanodonta complanata	0.1	2.5	–	–	–	–	–	–	–	–	–	–	–	–
Sphaerium corneum	11.5	40.0	3.2	48.0	14.4	60.0	41.3	68.0	19.6	40.0	16.9	80.0	36.6	100.0
Sphaerium rivicola	3.8	33.3	0.6	12.0	–	–	0.1	8.0	–	–	–	–	–	–
Musculium lacustre	–	–	0.4	16.0	–	–	–	–	–	–	11.2	30.0	–	–
Pisidium amnicum	6.9	44.2	0.1	8.0	1.4	12.0	0.5	16.0	2.3	20.0	0.2	10.0	9.3	100.0
Pisidium henslowanum	0.2	4.2	0.1	8.0	–	–	0.2	8.0	–	–	0.5	10.0	2.6	60.0
Pisidium supinum	2.5	30.8	1.9	16.0	2.3	36.0	0.3	8.0	2.6	26.7	1.0	30.0	1.0	40.0
Pisidium subtruncatum	1.7	10.8	0.3	8.0	–	–	0.1	8.0	–	–	2.7	60.0	–	–
Pisidium casertanum	0.2	5.0	–	–	–	–	–	–	–	–	1.2	30.0	6.7	80.0
Total number of species (*S*)	40		32		21		27		19		24		10	

Bithynia tentaculata was eudominant and euconstant, whereas *Viviparus viviparus* and *Sphaerium corneum* were eudominants and accessory species in the mollusc communities in the Wkra River. Four mollusc species were eudominants and euconstants in the tributaries of the Wkra River: *B. tentaculata* (Łydynia, Raciążnica), *Radix balthica* (Szkotówka, Naruszewka), *Ancylus fluviatilis* (Naruszewka) and *S. corneum* (Szkotówka and Naruszewka). Subrecedents and accedents, i.e. *Planorbis carinatus* and *Gyraulus crista*, which are typical of lakes or smaller, permanent water bodies, occurred only in the Wkra River (Table 9). Four species: *Valvata macrostoma* (subrecedent and accedent), *Aplexa hypnorum*, *Hippeutis complanatus* (subrecedent and accedent), and *Musculium lacustre*, which are mainly characteristic of reservoirs, were found only in the tributaries.

Typical fluviatile species, which were found only in the Wkra River, e.g. *U. crassus*, *Pseudanodonta complanata*, *Borysthenia naticina*, are subrecedents and accedents in the mollusc communities. A bivalve, *Sphaerium rivicola*, was recorded both in the Wkra River (subdominant and accessory species) and its tributaries (subrecedent and accedent). These species are legally protected in Poland (Dz.U. 2011a). Some species are included in the Polish Red Data Book of Animals (Głowaciński & Nowacki 2004) and the Red List of Threatened Animals in Poland (Dyduch-Falniowska & Zając 2002, Piechocki 2002) as critically endangered, i.e. *B. naticina*, or endangered, like *Lithoglyphus naticoides*, *P. complanata*, *Anodonta cygnea* and *U. crassus*. Alien species, *L. naticoides* (recedent and accedent) and *Potamopyrgus antipodarum* (subrecedent and accedent), were observed only in the Wkra River.

The maximum number of mollusc species per site (21) was higher at the sampling sites of the Wkra River compared to its tributaries (Table 10). The highest mean number of mollusc species (9.7 ± 2.7) was recorded in the Sona River, whereas the lowest mean number was recorded in the Mławka River (5.2 ± 2.8) (Table 10). The maximum density in the Wkra River was 302 indiv. m^{-2}, whereas the maximum density in the tributaries ranged from 69 indiv. m^{-2} (Naruszewka) to 649 indiv. m^{-2} (Sona). The mean density of molluscs amounted to 89.3 ± 68.2 in the Wkra River and it ranged from 37.9 ± 33.9 to 196.5 ± 182.4 in its tributaries (Table 10).

The median number of mollusc species ranged from 6 in the Naruszewka River to 10 in the Sona River (Fig. 12). A median density of 70 indiv. m^{-2} was recorded in the Wkra River, whereas in the tributaries it ranged from 32 in the Mławka River to 148 indiv. m^{-2} in the Sona River (Fig. 13). The Kruskal-Wallis one-way ANOVA test revealed statistically significant differences in the average number of mollusc species (H = 17.32, p = 0.001) as well as in the average values of mollusc density (H = 16.77, p = 0.01) at the sampling sites. Multiple comparison post hoc tests showed statistically significant differences between certain rivers only, i.e. the Sona River and the Mławka River (Figs 12 and 13).

The maximum values of the H', H_{max} or J' indices were higher in the Wkra River compared to its tributaries (Table 10). The median values of the H' and H_{max} indices ranged from 1.79 to 2.39 and from 2.81 to 3.34 in the Mławka and Sona rivers, respectively (Figs 14 and 15). Differences in the average values of H' (H = 13.70, p = 0.05) and H_{max} (H = 17.32, p = 0.001) indices were statistically significant at the sampling sites. A post hoc test showed statistically significant differences between certain rivers only, i.e. the Sona River and the Mławka River (Figs 14 and 15).

Statistically significant differences in the average number of mollusc species (H = 15.03, p = 0.05) and values of H_{max} (H = 15.03, p = 0.05) were recorded at sampling sites between the years of the survey.

Table 10. Number of mollusc species, density (individuals m^{-2}), values of H', H_{max} and J' of the mollusc communities in the Wkra River and its tributaries at the sampling sites

Index	Wkra River	Sona River	Łydynia River	Raciążnica River	Mławka River	Szkotówka River	Naruszewka River
No of species (ranges)	2–21	4–13	2–11	5–14	1–8	5–13	4–7
No of species (mean±SD)	8.2±3.9	9.7±2.7	6.5±2.3	8.7±2.5	5.2±2.8	9.2±2.3	6.0±1.7
Density (ranges)	5–302	17–649	9–185	32–527	3–98	35–337	62–69
Density (mean±SD)	89.3±68.2	196.5±182.4	75.5±44.7	129.4±133.9	37.9±33.9	98.7±92.2	64.7±3.8
Shannon–Wiener index (H')	0.43–3.71	1.17–3.09	0.76–2.37	1.47–3.57	0.00–2.73	1.11–3.10	2.03–2.38
Maximum Shannon-Wiener index (H_{max})	1.00–4.39	2.00–3.70	1.00–3.46	2.32–3.81	0.00–3.00	2.32–3.70	2.81–2.97
Evenness index (J')	0.18–0.98	0.51–0.95	0.51–0.81	0.42–0.94	0.00–0.99	0.37–0.90	0.74–0.85

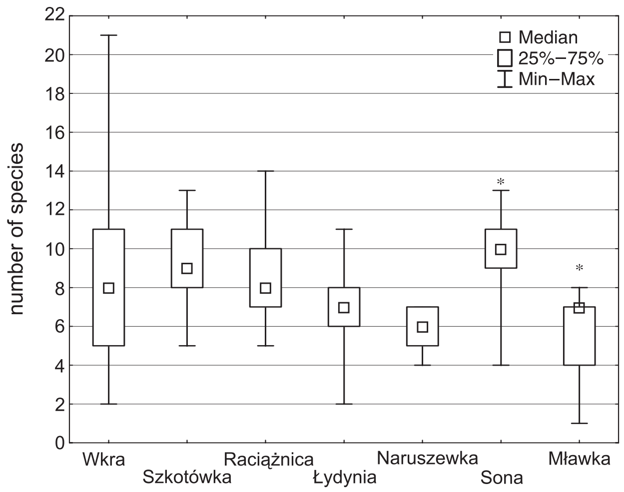


Fig. 12. Box-and-whisker plot showing the number of mollusc species in the Wkra River and its tributaries (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

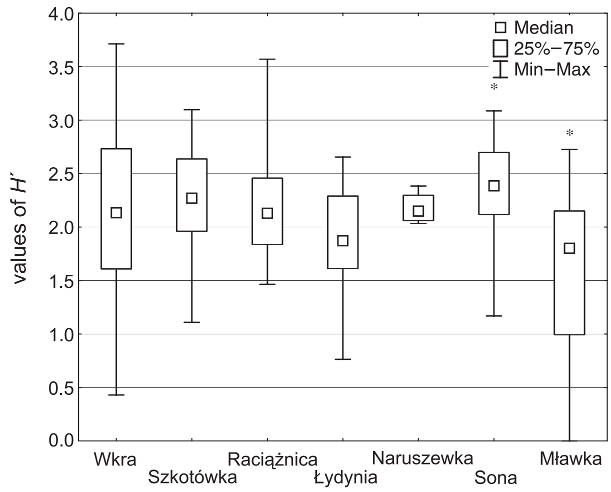


Fig. 14. Box-and-whisker plot showing the values of the *H'* index calculated for the mollusc communities in the Wkra River and its tributaries (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

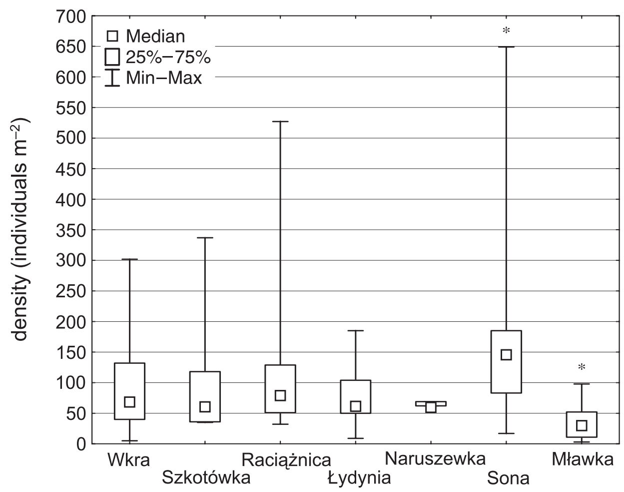


Fig. 13. Box-and-whisker plot showing the density of molluscs in the Wkra River and its tributaries (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

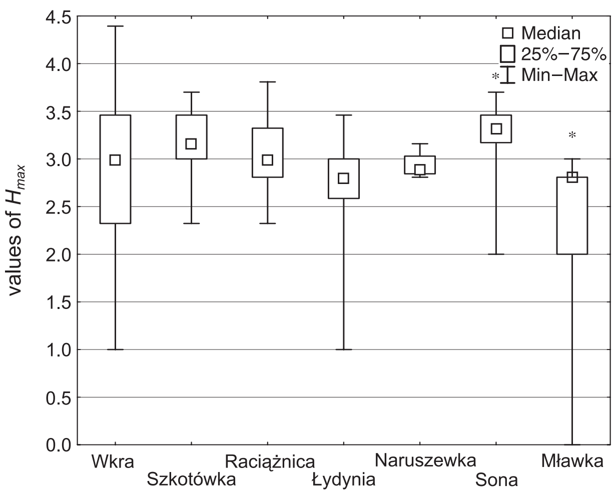


Fig. 15. Box-and-whisker plot showing the values of the H_{max} index calculated for the mollusc communities in the Wkra River and its tributaries (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

4.1.3.1. The Wkra River

Between 2005–2009, 40 mollusc species were observed in the Wkra River (Table 11). The dominance structure of the mollusc community varied among the years of the survey. Two gastropod species, i.e. *V. viviparus* and *R. balthica*, were eudominants in the mollusc communities in 2005, and four species including bivalves, *S. corneum* and *Pisidium amnicum*, in 2006. *B. tentaculata* was eudominant from 2006 to 2009. Subrecedents in the mollusc communities: *Anisus spirorbis*, *G. crista* and *P. complanata*, were recorded only in 2005 and *P. carinatus* in 2007. Prosobranch species, *B. naticina* (subrecedent) and *L. naticoides* (subrecedent or recedent) started to occur from 2006. Unionids: *Unio pictorum*, *U. tumidus* and *A. anatina*, were subdominants or recedents, whereas *U. crassus* and *A. cygnea* were recedents or subrecedents. The number of specimens dramatically decreased from 2008 to 2009 (Table 11).

Tables 12 and 13 show the dominance pattern of mollusc communities at the sampling sites located

Table 11. Values of dominance index (*D*%) calculated for the mollusc communities in the Wkra River between 2005 and 2009

Species	Years of the survey				
	2005	2006	2007	2008	2009
Theodoxus fluviatilis	3.5	3.3	12.0	5.8	17.1
Viviparus contectus	0.3	0.4	0.3	0.5	0.1
Viviparus viviparus	21.6	10.1	5.3	9.7	14.7
Bithynia tentaculata	8.6	15.4	10.7	21.8	14.4
Potamopyrgus antipodarum	0.2	0.1	0.5	0.1	–
Lithoglyphus naticoides	–	0.9	1.9	1.3	1.5
Valvata cristata	–	–	0.2	0.8	–
Valvata piscinalis	4.7	5.0	4.3	1.5	5.3
Borysthenia naticina	–	0.3	0.1	0.1	0.3
Acroloxus lacustris	1.2	1.3	0.1	–	0.1
Galba truncatula	0.1	0.5	0.1	0.1	0.1
Stagnicola palustris	6.4	2.1	1.1	0.7	2.5
Stagnicola corvus	–	–	–	–	0.3
Radix auricularia	0.3	0.2	0.2	0.3	0.4
Radix balthica	12.9	7.9	5.7	5.7	3.9
Lymnaea stagnalis	1.5	0.7	1.4	1.5	2.0
Physa fontinalis	1.0	0.7	0.5	0.8	2.2
Planorbarius corneus	0.9	0.6	0.5	0.6	0.7
Planorbis planorbis	0.3	–	0.2	0.1	–
Planorbis carinatus	–	–	0.1	–	–
Anisus spirorbis	0.1	–	–	–	–
Anisus vortex	4.1	1.6	1.0	5.3	5.1
Bathyomphalus contortus	1.6	0.2	0.6	0.3	0.2
Gyraulus albus	5.8	1.1	1.4	1.6	0.9
Gyraulus crista	0.2	–	–	–	–
Segmentina nitida	–	–	0.1	–	–
Ancylus fluviatilis	3.1	9.1	7.0	1.9	2.1
Unio pictorum	3.7	1.9	1.7	3.1	1.7
Unio tumidus	2.7	0.8	1.9	2.5	3.0
Unio crassus	0.9	0.1	1.1	1.3	0.4
Anodonta cygnea	0.3	0.2	0.9	1.0	0.2
Anodonta anatina	2.4	1.7	1.7	1.5	4.3
Pseudanodonta complanata	0.2	–	–	–	–
Sphaerium corneum	3.7	12.2	17.7	15.0	6.0
Sphaerium rivicola	2.9	2.9	6.4	2.6	4.1
Pisidium amnicum	1.6	13.1	5.4	8.5	4.9
Pisidium henslowanum	–	–	–	0.8	–
Pisidium supinum	1.5	2.6	4.7	1.7	1.4
Pisidium subtruncatum	1.4	2.6	2.6	1.3	–
Pisidium casertanum	0.3	0.2	0.7	0.1	–
No of specimens	2,328	2,475	2,646	2,410	1,478
Total number of species (*S*)	33	31	35	33	29

Table 12. Values of dominance index (*D*%) calculated for the mollusc communities at the sampling sites in the Wkra River between 2005 and 2009 (sampling sites N1–D11)

Species	N1	N2	N3	N4	N5	D6	D7	D8	D9	D10	D11
Theodoxus fluviatilis	–	–	–	–	–	–	–	–	–	–	–
Viviparus contectus	7.0	0.7	2.9	–	–	0.2	–	–	–	–	–
Viviparus viviparus	3.9	–	–	–	–	–	–	–	–	–	–
Bithynia tentaculata	1.3	–	–	–	–	5.0	29.3	27.1	21.9	21.8	30.7
Potamopyrgus antipodarum	–	–	–	–	–	–	–	–	–	1.6	5.3
Lithoglyphus naticoides	–	–	–	–	–	–	–	–	–	–	–
Valvata cristata	–	–	0.3	–	–	–	–	–	4.4	–	0.6
Valvata piscinalis	–	0.3	7.7	–	–	0.4	3.1	10.4	22.9	1.3	17.9
Borysthenia naticina	–	–	–	–	–	–	–	–	–	–	–
Acroloxus lacustris	–	–	–	–	2.2	–	–	–	–	0.6	–
Galba truncatula	–	–	–	1.0	2.2	–	1.5	–	–	2.5	0.9
Stagnicola palustris	–	17.5	24.0	2.1	2.2	–	–	–	–	6.9	–
Stagnicola corvus	–	–	0.3	–	–	–	–	–	0.7	–	–
Radix auricularia	–	–	–	–	–	–	–	–	–	–	0.9
Radix balthica	–	14.1	10.6	3.4	–	7.1	9.0	20.8	10.9	6.3	14.4
Lymnaea stagnalis	0.9	0.3	0.3	–	–	1.5	1.9	–	0.7	2.8	4.4
Physa fontinalis	6.1	0.7	3.4	–	–	1.0	2.5	8.3	–	1.6	0.9
Planorbarius corneus	2.2	0.7	3.9	–	–	0.4	–	–	2.8	–	–
Planorbis planorbis	–	–	0.7	–	–	0.6	0.9	–	–	–	0.6
Planorbis carinatus	–	–	–	–	–	–	–	–	–	–	0.6
Anisus spirorbis	–	–	–	–	–	0.4	–	–	–	–	–
Anisus vortex	5.2	1.5	14.8	5.2	2.2	2.5	4.9	33.3	3.5	40.4	1.6
Bathyomphalus contortus	13.5	0.7	3.9	–	2.2	0.8	0.6	–	–	0.6	–
Gyraulus albus	–	–	–	–	–	2.3	–	–	18.9	9.1	3.1
Gyraulus crista	–	–	–	–	–	–	–	–	0.9	–	–
Segmentina nitida	–	–	0.3	–	–	–	–	–	–	–	–
Ancylus fluviatilis	–	–	–	–	–	37.4	39.8	–	3.0	0.9	10.0
Unio pictorum	–	–	–	–	–	–	–	–	1.2	0.9	1.3
Unio tumidus	–	–	–	–	–	–	0.6	–	0.7	0.3	0.6
Unio crassus	–	–	–	–	–	–	**1.2**	–	–	**1.3**	**1.6**
Anodonta cygnea	–	–	–	–	–	–	–	–	–	0.3	–
Anodonta anatina	–	–	–	–	–	0.2	1.2	–	–	0.6	0.9
Pseudanodonta complanata	–	–	–	–	–	–	–	–	–	–	–
Sphaerium corneum	–	62.7	26.9	86.6	4.3	36.1	2.5	–	3.2	–	–
Sphaerium rivicola	–	–	–	–	–	–	–	–	–	–	–
Pisidium amnicum	–	–	–	–	35.9	1.2	–	–	–	–	3.5
Pisidium henslowanum	–	–	–	–	–	–	0.9	–	–	–	–
Pisidium supinum	–	–	–	1.7	7.6	3.1	–	–	4.4	–	–
Pisidium subtruncatum	59.8	1.0	–	–	26.1	–	–	–	–	–	–
Pisidium casertanum	–	–	–	–	15.2	–	–	–	–	–	–
Total number of species (*S*)	9	11	14	6	10	17	15	5	15	18	19

Table 13. Values of dominance index (*D*%) calculated for the mollusc communities at the sampling sites in the Wkra River between 2005 and 2009 (sampling sites W12–W24)

Species	W12	W13	W14	W15	W16	W17	W18	W19	W20	W21	W22	W23	W24
Theodoxus fluviatilis	–	–	–	–	26.1	16.6	26.5	2.8	–	9.3	4.3	12.4	24.4
Viviparus contectus	–	0.4	–	0.4	–	–	–	–	–	–	–	–	–
Viviparus viviparus	1.1	0.4	–	–	33.0	17.2	24.6	13.1	23.0	6.3	32.4	30.6	5.1
Bithynia tentaculata	8.0	14.3	15.4	9.8	9.6	14.0	2.7	52.8	13.1	21.6	11.4	5.5	4.3
Potamopyrgus antipodarum	–	–	–	–	–	–	–	–	–	–	–	–	–
Lithoglyphus naticoides	–	–	–	–	–	–	–	–	–	–	–	–	20.2
Valvata cristata	–	–	–	–	–	–	–	–	–	–	–	–	–
Valvata piscinalis	5.3	2.9	5.6	–	–	0.7	9.3	0.2	1.1	11.3	6.1	5.1	3.4
Borysthenia naticina	–	–	–	–	–	–	–	–	–	–	–	0.7	1.3
Acroloxus lacustris	–	–	–	11.7	0.1	–	–	–	–	–	–	0.1	–
Galba truncatula	–	–	–	–	–	–	–	0.2	–	–	–	–	–
Stagnicola palustris	1.1	–	1.4	–	–	–	–	–	–	–	–	–	–
Stagnicola corvus	–	–	–	–	–	–	–	–	–	–	–	–	–
Radix auricularia	3.7	0.4	–	–	0.6	–	0.4	0.4	0.6	–	0.5	–	0.3
Radix balthica	23.4	6.6	8.0	24.2	4.0	4.3	8.2	4.9	5.3	2.0	3.5	2.5	6.6
Lymnaea stagnalis	2.1	0.6	–	1.5	2.3	0.7	2.2	0.3	1.7	2.7	3.7	1.1	1.1
Physa fontinalis	–	–	0.7	–	2.5	–	–	0.3	–	1.0	1.9	0.4	–
Planorbarius corneus	–	1.2	0.7	–	–	–	1.6	–	0.6	1.3	0.8	0.2	–
Planorbis planorbis	–	–	0.7	–	–	–	–	–	–	–	–	0.2	–
Planorbis carinatus	–	–	–	–	–	–	–	–	–	–	–	–	–
Anisus spirorbis	–	–	–	–	–	–	–	–	–	–	–	–	–
Anisus vortex	10.6	0.4	–	0.8	1.1	–	0.2	–	0.4	1.0	0.8	0.1	–
Bathyomphalus contortus	–	–	–	–	–	–	–	–	–	–	–	–	–
Gyraulus albus	4.3	–	0.7	7.5	–	0.2	0.9	–	2.1	2.3	0.8	3.8	0.8
Gyraulus crista	–	–	–	–	–	–	–	–	–	–	–	–	–
Segmentina nitida	–	–	–	–	–	–	–	–	–	–	–	–	–
Ancylus fluviatilis	9.0	3.1	11.9	15.0	–	2.4	0.5	1.8	1.5	1.7	–	–	–
Unio pictorum	4.3	1.2	3.5	0.4	0.8	4.1	1.1	0.6	6.5	11.6	10.4	6.9	1.9
Unio tumidus	13.8	4.6	2.4	0.8	0.7	1.2	0.9	0.3	6.5	1.3	5.3	6.0	4.7
Unio crassus	**5.3**	**0.2**	–	–	**0.1**	**0.3**	**0.2**	**0.2**	**0.6**	**3.0**	**1.9**	**1.2**	**4.7**
Anodonta cygnea	–	1.4	0.7	0.2	0.5	1.6	3.3	–	0.4	2.0	1.3	0.6	–
Anodonta anatina	–	3.3	5.6	2.7	2.2	7.4	3.6	0.3	2.7	4.7	8.5	2.8	1.6
Pseudanodonta complanata	2.7	–	–	–	–	–	–	–	–	–	–	0.2	–
Sphaerium corneum	1.6	1.0	2.1	5.4	0.2	0.7	0.5	18.2	6.9	2.3	0.5	0.6	–
Sphaerium rivicola	–	1.7	12.6	17.7	3.2	12.1	4.0	0.5	7.8	1.0	1.3	4.1	13.3
Pisidium amnicum	3.7	53.5	21.3	1.0	12.4	8.1	8.2	1.5	4.4	9.3	3.2	7.6	3.0
Pisidium henslowanum	–	0.4	–	–	0.3	–	–	0.7	–	–	–	–	0.6
Pisidium supinum	–	1.5	6.6	0.4	0.4	8.1	0.9	0.8	14.7	1.7	1.3	4.6	2.7
Pisidium subtruncatum	–	–	–	–	–	0.3	–	–	–	2.7	–	1.9	–
Pisidium casertanum	–	0.8	–	0.4	–	–	–	–	–	–	–	0.9	–
Total number of species (*S*)	16	21	17	17	19	18	20	19	19	21	20	25	18

along the Wkra River between 2005 and 2009 (sampling sites N1–W24). The planorbid *Bathyomphalus contortus* and the fingernail clam *Pisidium subtruncatum* were eudominants in the mollusc communities at sampling site N1 located in the upper course of the Wkra River (Table 12). *Stagnicola palustris, S. corneum, Pisidium subtruncatum and P. casertanum* were eudominants at some sampling sites located mainly in the upper course (N1–N5).

Typical fluviatile species, e.g. *B. naticina*, *A. fluviatilis*, *U. crassus*, *S. rivicola* or *Pisidium henslowanum*, were completely absent from the mollusc communities at sampling sites located in the upper course, whereas some fingernail clams were eudominant (*P. amnicum*, sampling site N5), recedent (*Pisidium supinum*, sampling site N4) or dominant (*P. supinum*, sampling site N5).

The prosobranch species, *B. tentaculata* was eudominant at most sampling sites in the middle and lower course, whereas *V. viviparus* was eudominant only at the sampling sites located in the lower course (sampling sites W16–W20, W22–W23) (Tables 12 and 13).

Segmentina nitida, which was subrecedent in the mollusc communities, was recorded at one sampling site in the upper course (sampling site N3). *P. antipodarum* (recedent at site D10, dominant at site D11), *P. carinatus* (subrecedent at site D11) and *A. spirorbis* (subrecedent at site D6) were recorded in the middle course (Table 12). *B. naticina* was observed only at two sampling sites, i.e. W23 (subrecedent) and W24 (recedent) in the lower course. *L. naticoides* occurred only at one site in the lower course (W24) where it was eudominant (Table 13). Unionid species occurred at some sampling sites located in the middle course and all of the sites located in the lower course. Five unionid species occurred in the middle course (except site D8): *U. pictorum* and *A. anatina* were recedents or subrecedents in the mollusc communities; *U. tumidus* and *A. cygnea* were subrecedents (sites D7, D9–D11) and *U. crassus* was recedent (sites D7, D10–D11). Typical fluviatile species: *L. naticoides*, *A. fluviatilis*, *S. rivicola*, *P. amnicum* or *P. supinum*, were eudominants or dominants at some sampling sites located in the lower course. The prosobranch *Theodoxus fluviatilis* (eudominant at sites W16–W18, W23, W24), and the sphaeriid *S. rivicola* (eudominant at sites W14, W15, W17, W24), occurred in the lower course only. The number of species increased along the river from the sources to the mouth. The number of mollusc species ranged from 6 to 14 in the upper course, from 5 to 19 in the middle course, while there were up to 25 species (site W23) in the lower course (Tables 12 and 13).

Wide ranges of the density of *T. fluviatilis* (2–136 indiv. m^{-2}), *V. viviparus* (2–150 indiv. m^{-2}), *B. tentaculata* (2–162 indiv. m^{-2}), *S. corneum* (2–213 indiv. m^{-2}), and *P. amnicum* (2–147 indiv. m^{-2}) were recorded in the Wkra River (Figs 16A, B). The density of the alien species, *L. naticoides,* which was recorded at only one sampling site (W24), ranged from 22 to 50 indiv. m^{-2}, whereas in the case of *P. antipodarum* it ranged from 2 to 13 indiv. m^{-2}. The density of legally protected species, i.e. *B. naticina* and *P. complanata*, was low and amounted to only 4 and 5 indiv. m^{-2}, respectively, in contrast to the density of *S. rivicola* (2–56 indiv. m^{-2}). The maximum density of *U. pictorum*, *U. tumidus* or *A. anatina* was above 20 indiv. m^{-2} (Figs 16A, B).

The mean density and mean number of mollusc species varied from among the sites, starting from the headwaters to the mouth and was reflected in the mean values of the H', H_{max} and J' indices (Figs 17 and 18). The highest mean density (217 indiv. m^{-2}), mean number of species (16.8), H' (3.20) and H_{max} (4.04) were obtained at site W23, located in the lower course of the Wkra River. Generally, the mean values of the H', H_{max} and J' indices for the mollusc communities in the lower course of the river were higher than those in the upper course, excluding site N3.

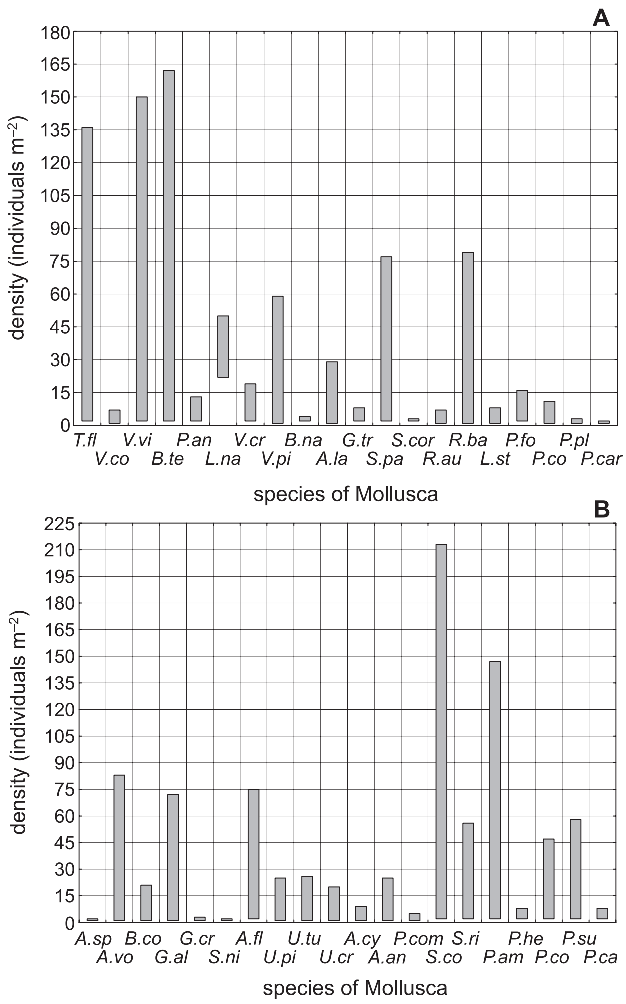


Fig. 16A, B. Density (ranges) of the mollusc species in the Wkra River (2005–2009)

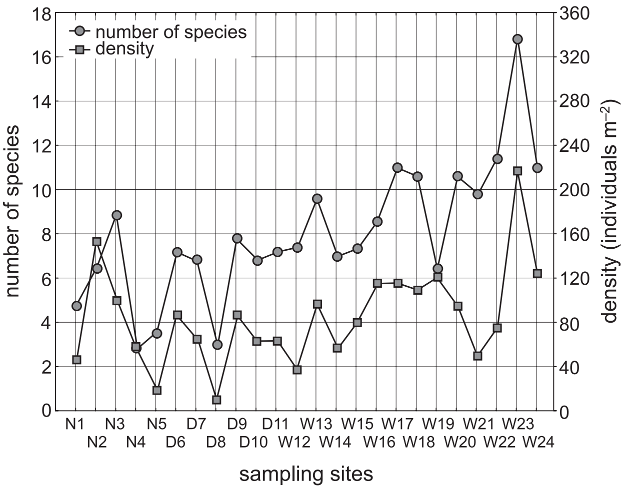


Fig. 17. Mean density (indiv. m^{-2}) and mean number of mollusc species at sampling sites in the Wkra River (2005–2009)

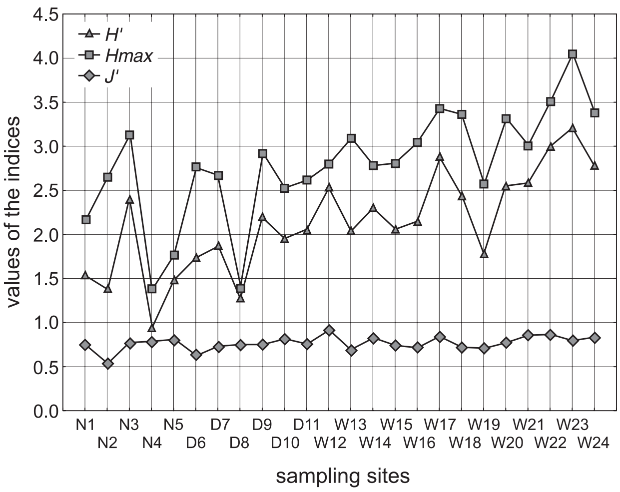


Fig. 18. Mean values of indices *H'*, H_{max} and *J'* calculated for the mollusc communities at sampling sites in the Wkra River (2005–2009)

The highest mean number of mollusc species of 9.8 was recorded in 2007, whereas the highest mean density of 107.6 indiv. m^{-2} was recorded in 2006. From 2007 to 2009, both the number of mollusc species and the density showed a decreasing tendency (Table 14).

The maximum number of mollusc species for the upper course (Nida River) was 11, and the median value was 5; the respective values for the lower course (Wkra River) were 21 and 9 (Fig. 19). The Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests revealed statistically significant differences in the average number of mollusc species ($H = 24.11$, $p = 0.0001$) between the sampling sites located in the upper (sites N1–N5), middle (sites D6–D11) and lower course (sites W12–W24) in the Wkra River (Fig. 19). The median density ranged from 48 indiv. m^{-2} in the upper course to 73 indiv. m^{-2} in the lower course. Differences in the average values of mollusc density ($H = 7.14$, $p = 0.028$) were statistically significant; the post hoc test showed statistically significant differences between the upper and lower courses (Fig. 20).

The median values of the *H*' index ranged from 1.50 (upper course) to 2.42 (lower course) and the maximum values of the *H*' index – from 2.66 to 3.71 (Fig. 21). The median values of the H_{max} index ranged from 2.32 (upper course) to 3.17 (lower course) (Fig. 22). The Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests revealed statistically significant differences in the average values of *H*' ($H = 25.67$, $p = 0.0001$) and H_{max} ($H = 24.11$, $p = 0.0001$) indices between the sampling sites located in the upper, middle and lower course of the Wkra River (Figs 21 and 22).

Table 14. Number of mollusc species and density (individuals m^{-2}) in the Wkra River and its tributaries recorded from 2005 to 2009 (mean±SD)

River	Index	Years of the survey				
		2005	2006	2007	2008	2009
Wkra River	No of species	7.7±3.2	8.1±3.8	9.8±4.1	9.0±4.1	6.4±3.7
	Density	80.2±57.7	107.6±83.4	105.8±69.1	96.5±69.9	59.1±52.7
Sona River	No of species	10.5±0.7	12.5±0.7	7.6±3.1	10.8±1.3	9.4±2.5
	Density	268.0±171.1	256.0±138.6	246.8±214.8	228.4±241.2	90.4±61.2
Łydynia River	No of species	4.3±4.0	6.0±4.6	7.3±1.4	7.9±1.6	5.8±1.6
	Density	55.3±42.2	72.3±54.9	76.8±56.9	89.0±51.3	63.7±36.9
Raciążnica River	No of species	5.5±0.7	6.0±1.0	10.4±3.1	8.4±2.3	7.8±1.3
	Density	51.0±14.1	56.0±18.5	121.0±141.6	162.6±205.9	116.4±47.7
Mławka River	No of species	4.0±1.4	5.0±1.4	6.5±2.1	6.3±2.1	3.0±3.5
	Density	42.5±6.4	52.5±30.4	60.7±33.8	39.0±38.6	14.0±18.2
Szkotówka River	No of species	8.5±2.1	11.5±0.7	10.3±2.5	8.0±1.0	8.0±3.0
	Density	76.5±40.3	119.0±104.6	147.0±165.4	65.7±32.7	70.0±4.8

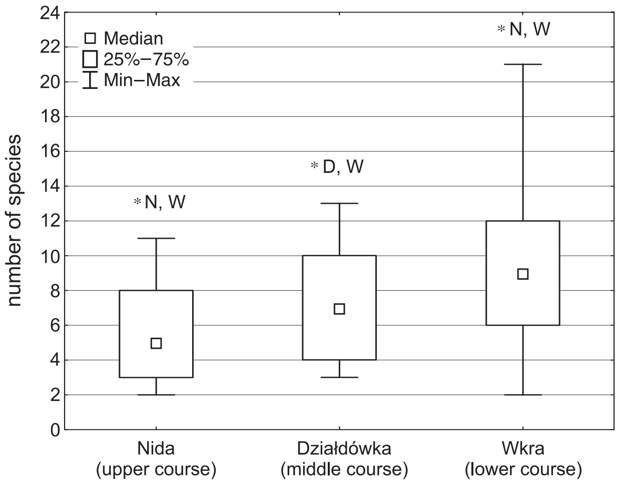


Fig. 19. Box-and-whisker plot showing the number of mollusc species in the upper, middle and lower course of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

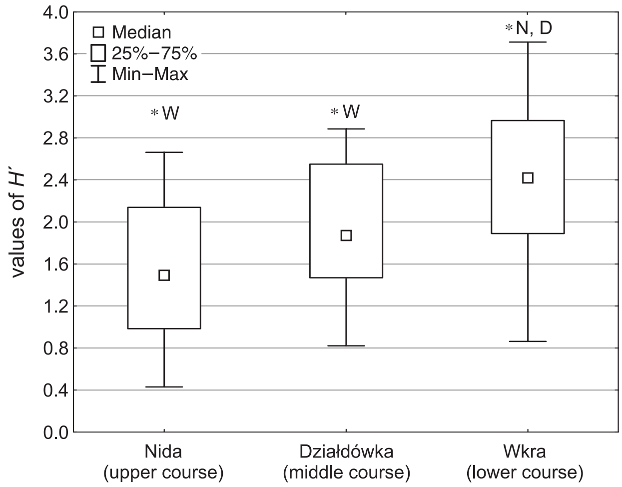


Fig. 21. Box-and-whisker plot showing the values of the *H'* index calculated for the mollusc communities in the upper, middle and lower course of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

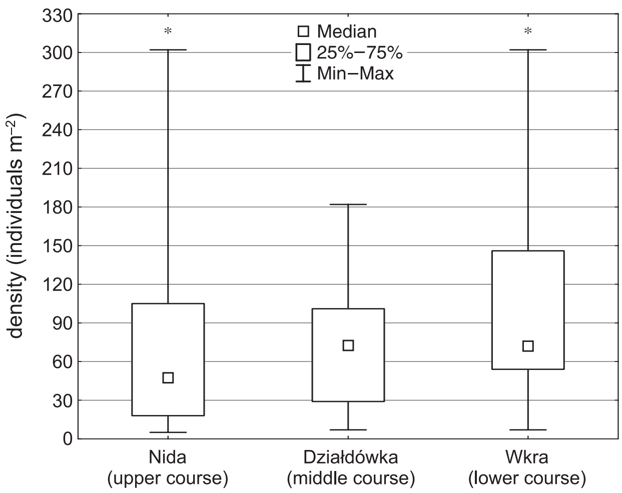


Fig. 20. Box-and-whisker plot showing the density of molluscs in the upper, middle and lower course of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

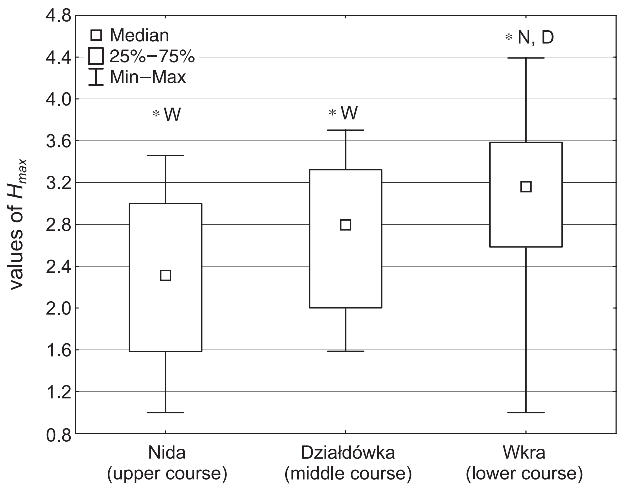


Fig. 22. Box-and-whisker plot showing the values of the H_{max} index calculated for the mollusc communities in the upper, middle and lower course of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

The median number of mollusc species, median density as well as median values of the *H'* and H_{max} indices increased along the river from the upper to the lower course (Figs 19–22).

4.1.3.2. The Sona River

In the Sona River, in comparison to the Wkra River, the density of *V. viviparus* (5–225 indiv. m^{-2}) and *B. tentaculata* (3–179 indiv. m^{-2}) varied widely. In contrast, the density of two planorbid species, i.e. *Planorbarius corneus* (up to 288 indiv. m^{-2}) and *Planorbis planorbis* (up to 279 indiv. m^{-2}), was very high (Fig. 23A). The density of bivalve species, especially *U. pictorum* (up to 19 indiv. m^{-2}), *U. tumidus* (up to 3 indiv. m^{-2}), *S. rivicola* (up to 18 indiv. m^{-2}) or *P. amnicum* (up to 2 indiv. m^{-2}), was lower than that in the Wkra River (Fig. 23B).

The highest mean number of species (11.2) was recorded at site S4, whereas the highest mean density

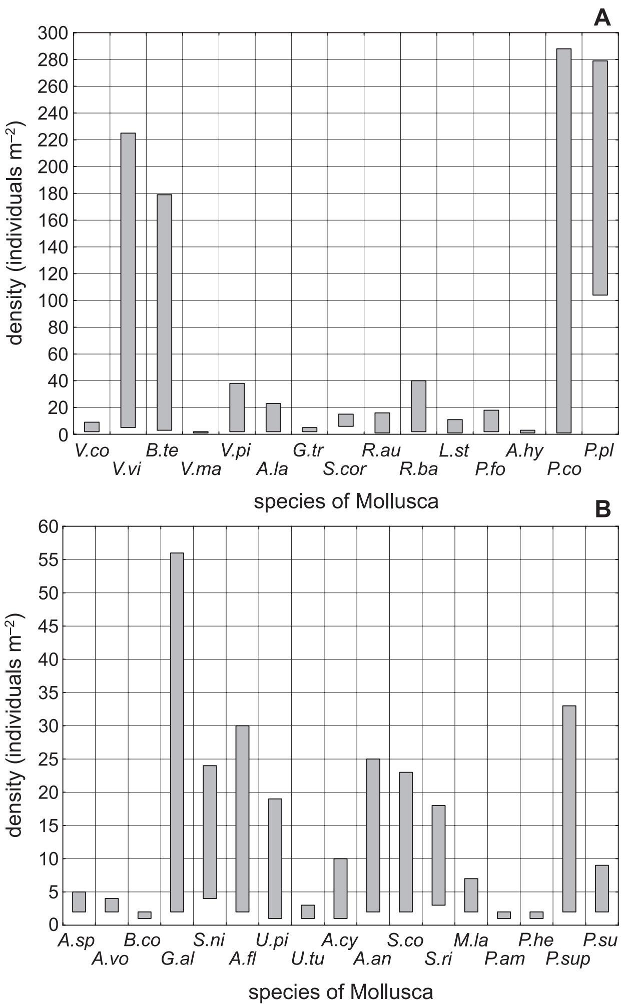


Fig. 23A, B. Density (ranges) of the mollusc species in the Sona River (2005–2009)

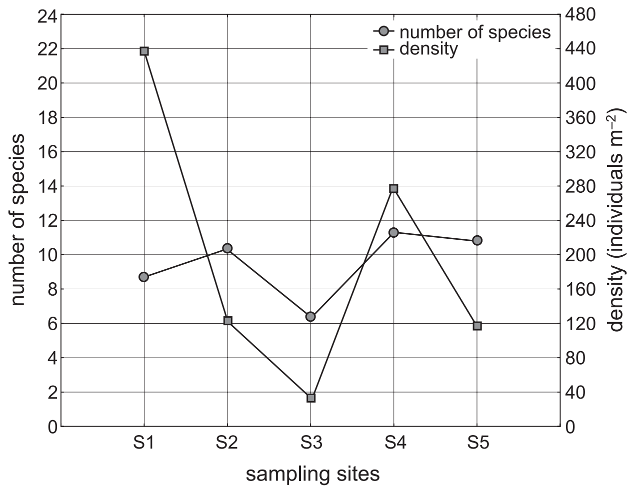


Fig. 24. Mean density (indiv. m^{-2}) and mean number of mollusc species at sampling sites in the Sona River (2005–2009)

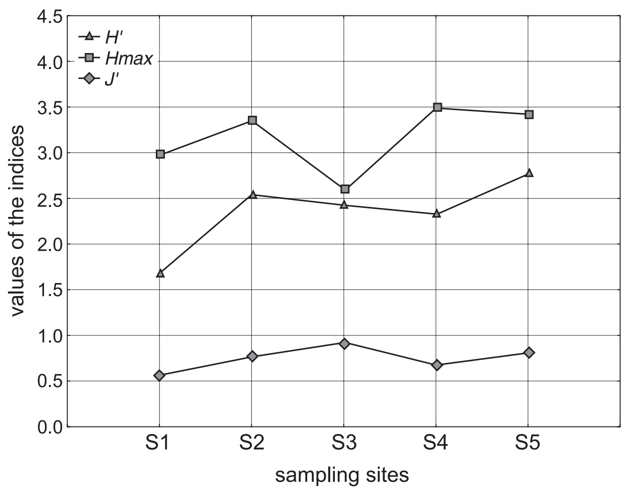


Fig. 25. Mean values of indices *H*', H_{max} and *J*' calculated for the mollusc communities at sampling sites in the Sona River (2005–2009)

(437.0 indiv. m^{-2}) was observed at site S1, which is located in the upper course of the river. However, the highest mean value of the *H*' index calculated for mollusc communities amounted to 2.77 at site S5 (lower course of the Sona) (Figs 24 and 25). The mean species diversity as measured by the Shannon-Wiener index (*H*') ranged from 1.60 to 2.77 among the sampling sites and increased significantly from the upper to the lower course of the river.

The mean number of mollusc species varied within 2005–2009, but from 2008 it decreased. The mean density showed a decreasing tendency from 268.0 indiv. m^{-2} in 2005 to 90.4 indiv. m^{-2} in 2009 (Table 14).

4.1.3.3. The Łydynia River

The widest density range of *A. fluviatilis* (from 4 to 129 indiv. m^{-2}) was recorded in the Łydynia River in comparison to the Wkra River and the other tributaries (Fig. 26). Only three unionid species occurred there, i.e. *U. pictorum* (1–5 indiv. m^{-2}), *U. tumidus* (3–4 indiv. m^{-2}) and *A. anatina* (1–3 indiv. m^{-2}), whose density was lower than in the Wkra River and the

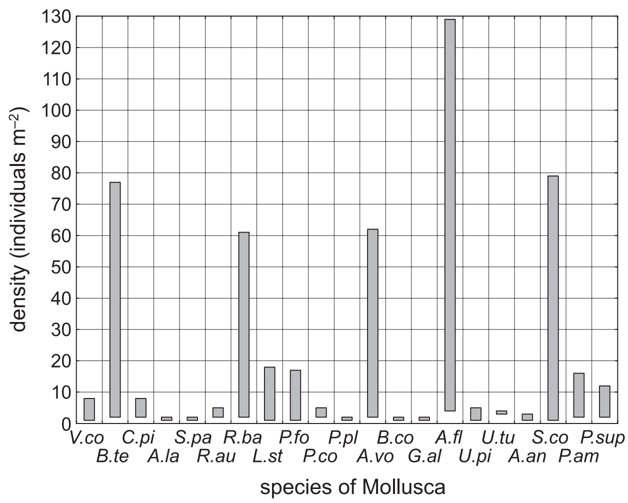


Fig. 26. Density (ranges) of the mollusc species in the Łydynia River (2005–2009)

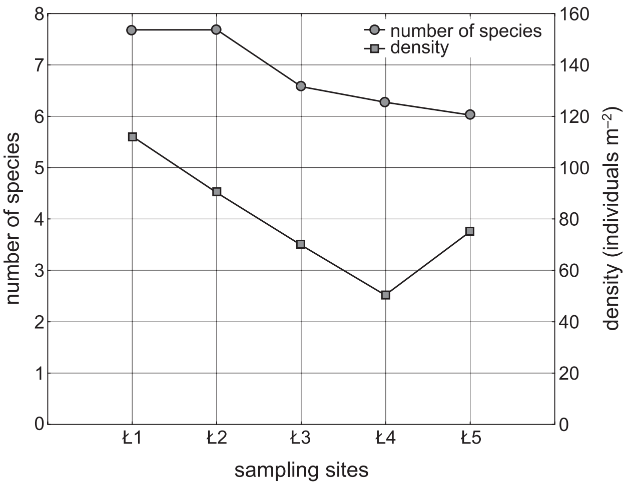


Fig. 27. Mean density (indiv. m^{-2}) and mean number of mollusc species at sampling sites in the Łydynia River (2005–2009)

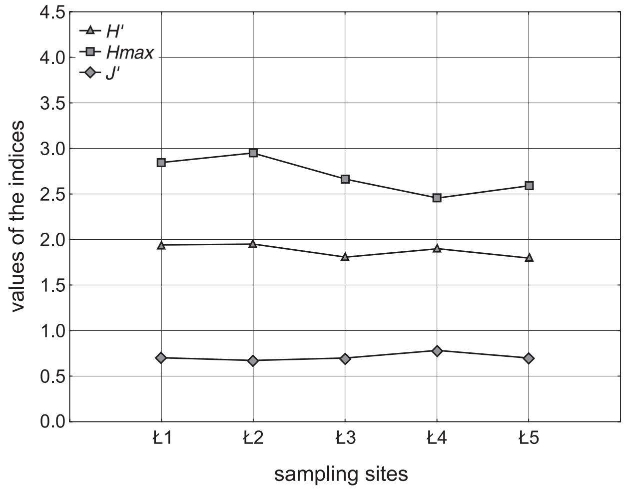


Fig. 28. Mean values of indices *H'*, H_{max} and *J'* calculated for the mollusc communities at sampling sites in the Łydynia River (2005–2009)

Sona River. The density of typical fluviatile fingernail clams, *P. amnicum* (up to 16 indiv. m^{-2}) was higher, whereas that of *P. supinum* (up to 12 indiv. m^{-2}) was lower than those recorded in the Sona River.

The mean number of species amounted to 7.7 at site Ł1 and than showed a decreasing tendency from site Ł2 to site Ł5. The highest mean density of molluscs (112 indiv. m^{-2}) was recorded at a sampling site located in the upper course of the river (Ł1). From the upper to the middle course, the mean density decreased to 50.0 indiv. m^{-2} (site Ł4) and then increased to 74.8 indiv. m^{-2} (site Ł5) (Fig. 27). The mean values of the *H'* and H_{max} indices were higher in the upper course compared to the lower course of the Łydynia River. The mean value of *J'* calculated for the mollusc communities ranged from 0.66 at site Ł2 to 0.77 at site Ł4 (Fig. 28).

In 2005–2008 the mean number of species and the mean density increased to 7.9 and 89.0 indiv. m^{-2}, respectively, and then showed a decreasing tendency (Table 14).

4.1.3.4. The Raciążnica River

In contrast to the Wkra River and the other tributaries, the widest density ranges of *B. tentaculata* (2–206 indiv. m^{-2}) and fingernail clam, *S. corneum* (2–286 indiv. m^{-2}) were recorded in the Raciążnica River (Figs 29A, B). Only three unionid species were observed there: *U. pictorum*, *U. tumidus* and *A. anatina*, whose density did not exceed 5 indiv. m^{-2}. The density of *S. rivicola* was also very low (from 1–2 indiv. m^{-2}) as well as that of other fingernail clam species (*P. amnicum*, *P. henslowanum*, *P. supinum* and *P. subtruncatum*) which was up to 5 indiv. m^{-2} except of *S. corneum* (Fig. 29B). However, the mean number of mollusc species (11) and the density (358.3 indiv. m^{-2}) were highest at site R2, the values of the

H' calculated for the mollusc communities was the lowest (1.64) in comparison to the other sampling

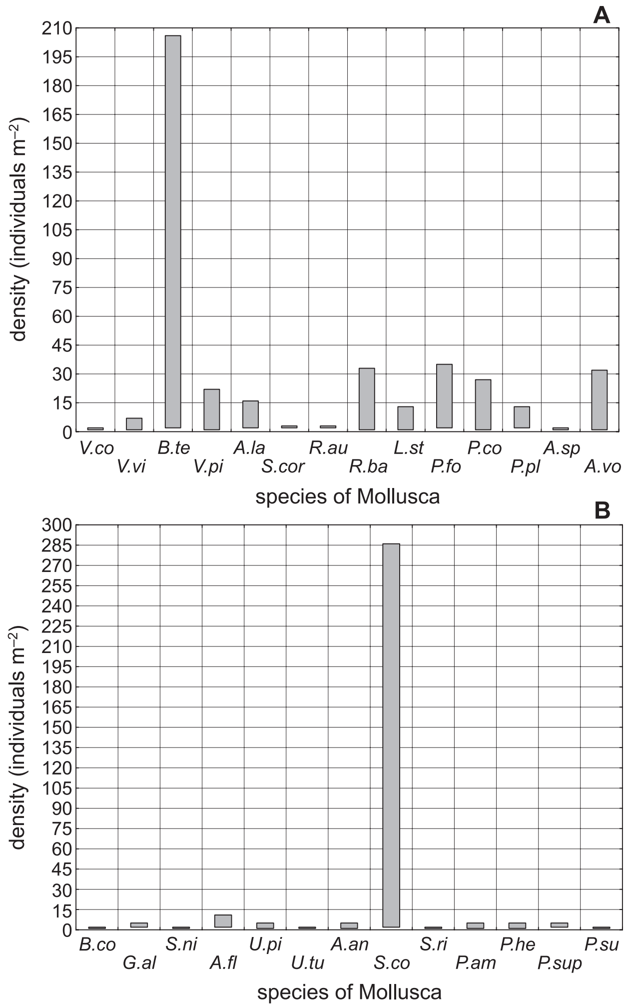


Fig. 29A, B. Density (ranges) of the mollusc species in the Raciążnica River (2005–2009)

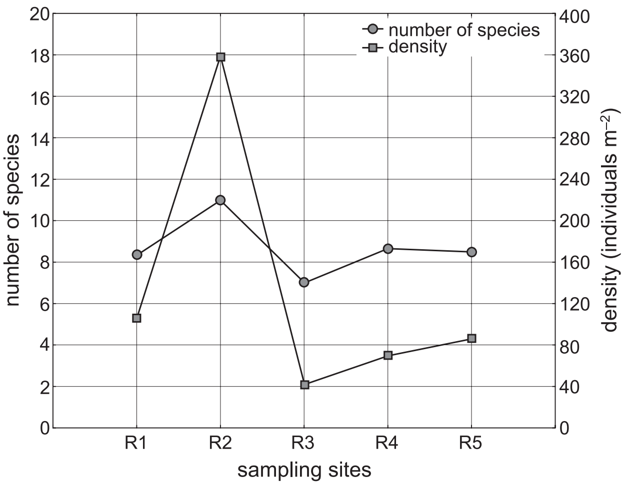


Fig. 30. Mean density (indiv. m^{-2}) and mean number of mollusc species at sampling sites in the Raciążnica River (2005–2009)

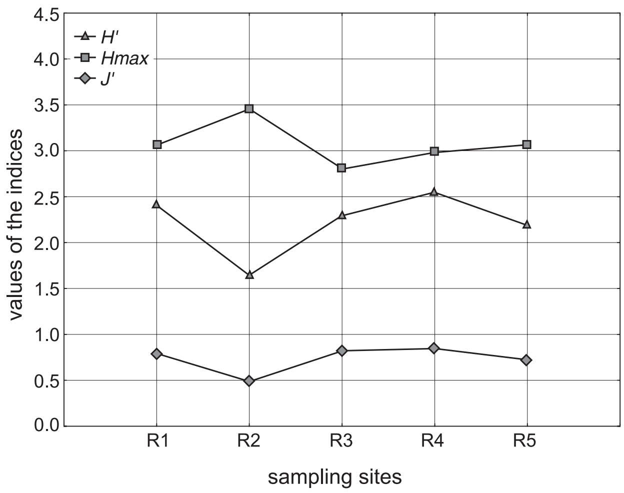


Fig. 31. Mean values of indices *H'*, H_{max} and *J'* calculated for the mollusc communities at sampling sites in the Raciążnica River (2005–2009)

sites. At site R2, the quantity ratio (the number of specimens of each species) was poorly balanced in the mollusc community (few dominant species, the Evenness index (*J'*) was very low, amounting to 0.48), and therefore reflected very low values of the *H'* index (Figs 30 and 31). In 2005–2009, the mean number of mollusc species ranged from 5.5 to 10.4, whereas the mean density ranged from 51.0 to 162.6 indiv. m^{-2}. The mean number of species showed a decreasing tendency from 2007, whereas the mean density showed a decreasing tendency from 2008 (Table 14).

4.1.3.5. The Mławka River

The density of *R. balthica* ranged from 4 to 55 indiv. m^{-2}. The maximum density of most of the species, including unionids, did not exceed 5 indiv. m^{-2} (Fig. 32). The mean number of mollusc species

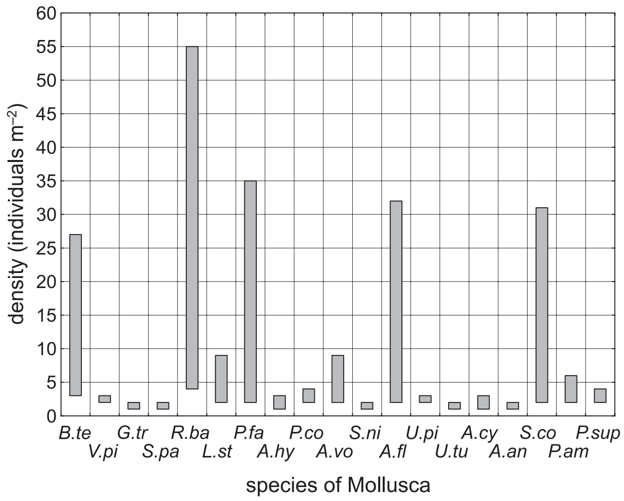


Fig. 32. Density (ranges) of the mollusc species in the Mławka River (2005–2009)

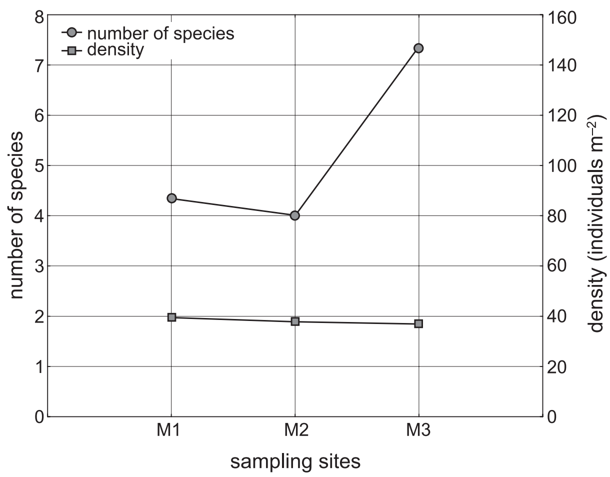


Fig. 33. Mean density (indiv. m^{-2}) and mean number of mollusc species at sampling sites in the Mławka River (2005–2009)

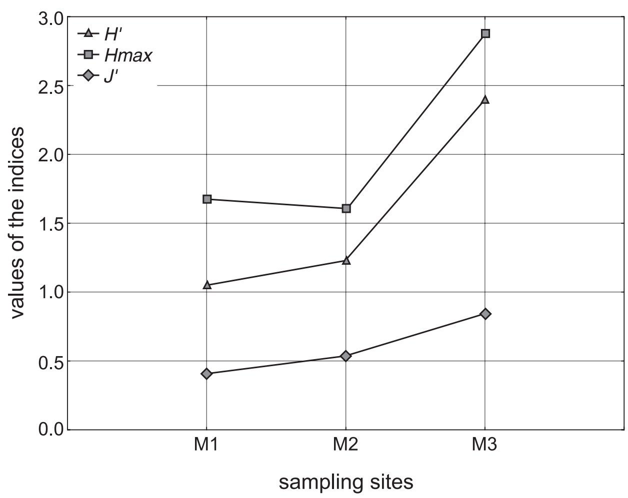


Fig. 34. Mean values of indices *H'*, H_{max} and *J'* calculated for the mollusc communities at sampling sites in the Mławka River (2005–2009)

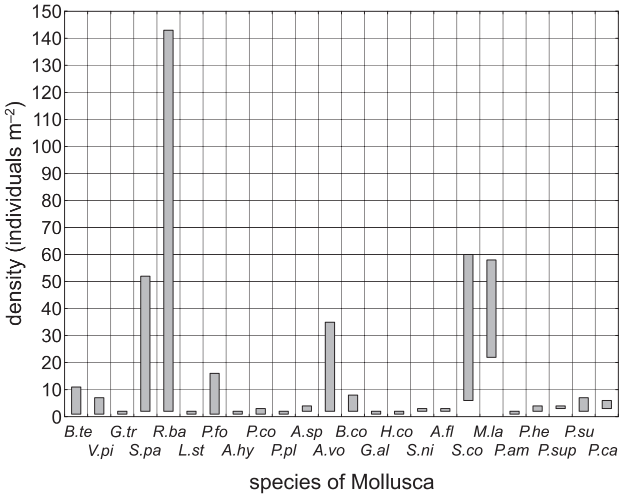


Fig. 35. Density (ranges) of the mollusc species in the Szkotówka River (2005–2009)

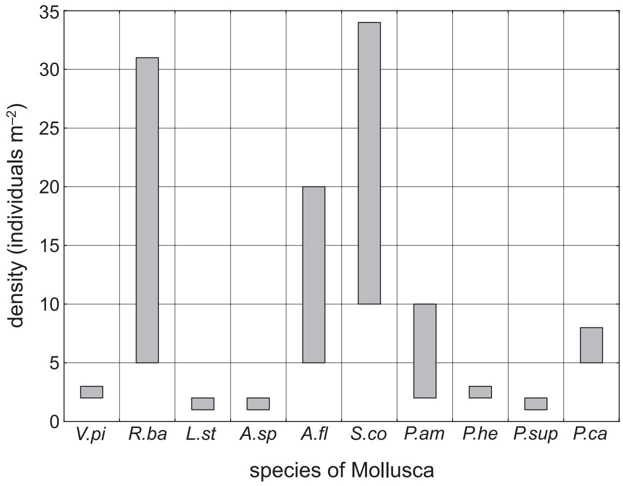


Fig. 36. Density (ranges) of the mollusc species in the Naruszewka River (2005–2009)

increased from 4.3 at site M1 to 7.3 at site M3, whereas the density decreased from 39.3 indiv. m^{-2} to 36.7 indiv. m^{-2} (Fig. 33). At site M3, the quantity ratio (the number of specimens of each species) was evenly balanced in the mollusc community (no dominance of one or few species). Therefore, the values of the *H*', H_{max} and *J*' indices were relatively high (2.29, 2.87 and 0.84, respectively) (Fig. 34).

The mean number of mollusc species and the density increased from 2005 to 2007, and after that showed a decreasing tendency (Table 14).

4.1.3.6. The Szkotówka River

The widest density ranges of *R. balthica* (2–143 indiv. m^{-2}) and *M. lacustre* (22–58 indiv. m^{-2}), were recorded in the Szkotówka River in contrast to the Wkra River and the other tributaries (Fig. 35). The densities of other fingernail clams, excluding *S. corneum*, were low and did not exceed 7 indiv. m^{-2}. The mean number of species and the mean density were 8.4 and 125.9 indiv. m^{-2}, respectively, at site Sz1 (mean *H*' = 1.99, H_{max} = 3.04, *J*' = 0.66), and 10.5 and 51.2 indiv. m^{-2} at site Sz2 (mean *H*' = 2.70, H_{max} = 3.36, *J*' = 0.81). The highest mean number of mollusc species (11.5) was recorded in 2006, whereas the highest mean density of 147.0 indiv. m^{-2} was obtained in 2007. From 2006 to 2009 the mean number of mollusc species showed a decreasing tendency (Table 14).

4.1.3.7. The Naruszewka River

The density of three out of 10 species (*R. balthica*, *A. fluviatilis* and *S. corneum*) exceeded 10 indiv. m^{-2}. The density of *P. amnicum*, a typical fluviatile species, ranged from 2 to 10 indiv. m^{-2}, whereas that of *P. henslowanum* and *P. supinum* was lower than 5 indiv. m^{-2} (Fig. 36). The mean number of species and the mean density amounted to 7.0 and 64.7 indiv. m^{-2} at site Na1. The species diversity measured by the Shannon-Wiener index (*H*'), the maximum Shannon-Wiener index (H_{max}) and the Evenness index (*J*') were 2.17, 2.81, and 0.77, respectively.

4.1.4. OCCURRENCE OF *UNIO CRASSUS* PHILIPSSON, 1788 IN RELATION TO THE MOLLUSC COMMUNITIES (THE WKRA RIVER)

From 2005 to 2009, the thick shelled river mussel *U. crassus* was recorded at three sites located in the middle course of the Wkra River: D7, D10 and D11, and at eleven sites located in the lower course in the following years:

- 2005 (W12, W19, W20, W21, W22, W23, W24),
- 2006 (W16, W19, W23),
- 2007 (D10, W12, W13, W21, W24),
- 2008 (D7, W12, W17, W18, W20, W21, W23, W24),
- 2009 (D10, D11, W23, W24).

In consecutive years between 2005 and 2009, a few more specimens of *U. crassus* were recorded only at five out of the 14 sampling sites, i.e. D10, W12, W21, W23 and W24. The thick shelled river mussel was recedent in the mollusc communities at sites D10 and D23, subdominant at sites W21 and W24, whereas it was dominant at site W12 (Tables 12 and 13). In total, 25 mollusc species occurred at site W23, including *U. crassus* and other legally protected species: *B. naticina* (subrecedent), *A. cygnea* (subrecedent), *P. complanata* (subrecedent) and *S. rivicola* (subdominant). At site W24, which is located below the area of the Natura 2000, 18 mollusc species were recorded. Among them *T. fluviatilis*, *L. naticoides* and *S. rivicola* were eudominants, whereas *B. naticina* was recedent in the mollusc communities. The number of mollusc species including *U. crassus* ranged from 18 (site D10) to 25 (site W23) (Tables 12 and 13).

Table 15. Density, shell measurements (ranges), age of *Unio crassus* and some environmental factors recorded at selected sampling sites in the Wkra River (2005–2009)

Parameter	D10 (Bądzyń)	W12 (Bieżuń)	W21 (Joniec)	W23 (Śniadówko)	W24 (Pomiechówek)
Density (individuals m^{-2})	2–3	3–4	2–4	2–7	4–20
Shell length (mm)	31.7–46.5	25.1–60.4	41.6–66.9	23.5–56.2	15.7–53.8
Shell height (mm)	19.1–27.5	15.3–33.2	25.3–36.5	13.1–33.5	9.5–30.4
Age (years)	2–4	1–5	3–6	1–4	1–5
Conductivity (μS cm^{-1})	390–490	440–480	500–520	480–550	490–540
Nitrites (mg NO_2^- dm^{-3})	0.06–0.50	0.02–0.53	0.10–0.36	0.03–0.50	0.07–0.43
Nitrates (mg NO_3^- dm^{-3})	5.76–87.71	1.33–50.95	6.64–27.91	3.99–65.12	18.16–33.63
Phosphates (mg PO_4^{3-} dm^{-3})	0.26–1.06	0.45–1.08	0.11–1.73	0.12–1.63	0.05–1.42
Velocity (m s^{-1})	0.22–0.30	0.40–0.50	0.33–0.60	0.09–0.30	0.06–0.33
Predominate particle size (mm)	0.25–0.50 (medium sand)	0.10–0.25 (fine sand)	0.25–0.50 (medium sand)	0.25–0.50 (medium sand)	0.50–0.80 (coarse sand) 0.25–0.50 (medium sand)

The density of *U. crassus* ranged from 2–3 indiv. m^{-2} (site D10) to 4–20 indiv. m^{-2} (site W24) in the Wkra River (Table 15). The maximum shell length (66.9 mm), shell height (36.5 mm) and age (6-year-old specimens) of *U. crassus* were recorded at site W21 (Table 15).

4.1.5. MOLLUSC COMMUNITIES IN THE WKRA RIVER AND ITS TRIBUTARIES IN RELATION TO SELECTED ENVIRONMENTAL FACTORS

Canonical correspondence analysis (CCA) based on the species data and environmental variables showed that the first and second axes explained 20.6% of the variance in the species data and 54.9% of the variance in the species and environment relationship (the first axis explained 13.2% of the variance in the species data and 35.0% of the variance in the species and environment relationship). The eigenvalues of axes 1, 2, 3 and 4 were 0.749, 0.425, 0.302 and 0.151, respectively. Species-environmental correlations of axes 1, 2, 3 and 4 were: 0.951, 0.849, 0.734 and 0.626.

Conductivity, hardness, water velocity, river width, temperature, nitrites, nitrates, phosphates and substrate particles (0.006–0.02 mm, 0.02–0.05 mm, 0.05–0.10 mm and 0.25–0.50 mm) were the parameters most associated (statistically significant according to the forward selection results) with the distribution of mollusc species (Fig. 37). The forward selection results showed that the concentration of dissolved oxygen, macrophytes and other particle fractions of the bottom sediments (2.0–5.0 mm and 5.0–10.0 mm) also exerted a significant influence on the distribution of mollusc species.

Some patterns of mollusc distribution were identified. *A. fluviatilis*, *T. fluviatilis*, *U. crassus*, *S. rivicola*, *U. pictorum*, *U. tumidus*, *P. amnicum*, *A. anatina*, *A. cygnea*, *V. viviparus* and *P. supinum* were positively influenced by velocity, river width or temperature and negatively influenced by nitrite concentration. *P. corneus* and *P. planorbis* occurred at the sampling sites with a higher conductivity, phosphate concentration in the water, hardness and substrate particles ranging from 0.006 to 0.02 mm. *B. tentaculata* and *Valvata piscinalis* occurred at the sampling sites with a higher dissolved oxygen concentration in the water and a lower concentration of phosphates. *R. balthica*, *Acroloxus lacustris*, *Anisus vortex* were associated with the distribution of macrophytes. *P. subtruncatum* and *B. contortus* were negatively influenced by substrate particles ranging from 0.25 to 0.50 mm (Fig. 37).

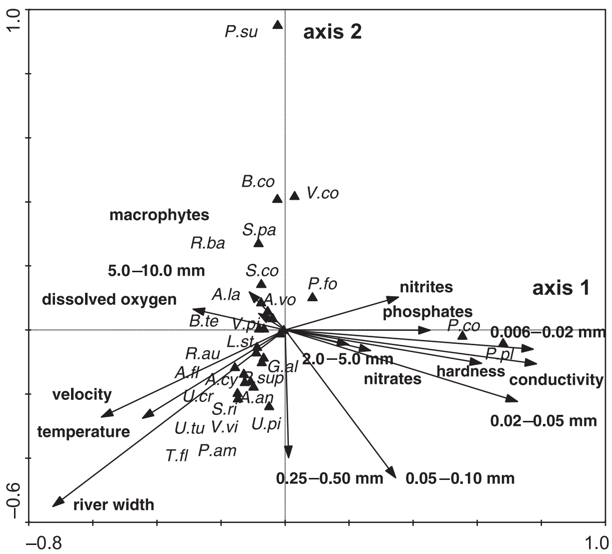


Fig. 37. Ordination diagram (biplot) based on canonical correspondence analysis (CCA) of the Mollusca data and environmental variables. Long arrows representing selected (statistically significant) environmental variables emphasize their impact on the structure of the mollusc communities in the Wkra River and its tributaries

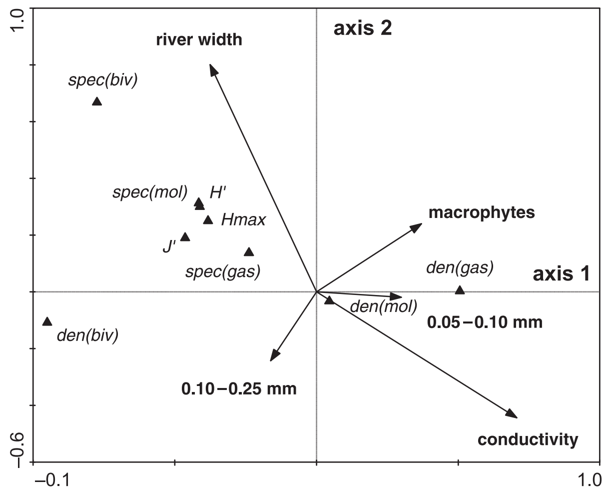


Fig. 38. Ordination diagram (biplot) based on canonical correspondence analysis (CCA) of the indices and selected environmental variables (the Wkra River and its tributaries). Abbreviations: *spec*(*mol*) – number of mollusc species; *spec*(*gas*) – number of gastropod species; *spec*(*biv*) – number of bivalve species; *den*(*mol*) – density of molluscs; *den*(*gas*) – density of gastropods; *den*(*biv*) – density of bivalves; *H'* – values of the Shannon-Wiener index; H_{max} – values of the maximum Shannon-Wiener index; *J'* – values of the Evenness index

The relationship between the species composition of Mollusca and the environmental variables was significant (Monte Carlo test of significance of the first canonical axis: *F*-ratio = 17.749, *p*-value = 0.002; test of significance of all canonical axes: *F*-ratio = 4.409, *p*-value = 0.002).

The number of gastropod and bivalve species and the values of *H*', H_{max} and *J*' were positively correlated with the width of the rivers and negatively correlated with conductivity. The density of bivalves was positively correlated with the substrate particles ranging from 0.10 to 0.25 mm, while the density of gastropods was associated with the macrophytes. The number of mollusc species and the values of the *H*', H_{max} and *J*' indices decreased with increasing conductivity, whereas the density of Mollusca increased (Fig. 38).

The relation between the values of the metrics and the environmental variables was statistically significant (Monte Carlo test of significance of the first canonical axis: *F*-ratio = 34.937, *p*-value = 0.002, test of significance of all canonical axes: *F*-ratio = 7.689, *p*-value = 0.002).

4.1.6. MOLLUSC SPECIES IN RELATION TO THE VALUES OF THE STATISTICALLY SIGNIFICANT ENVIRONMENTAL FACTORS

Six gastropod species: *V. piscinalis*, *Lymnaea stagnalis*, *Physa fontinalis*, *P. corneus*, *P. planorbis*, *Gyraulus*

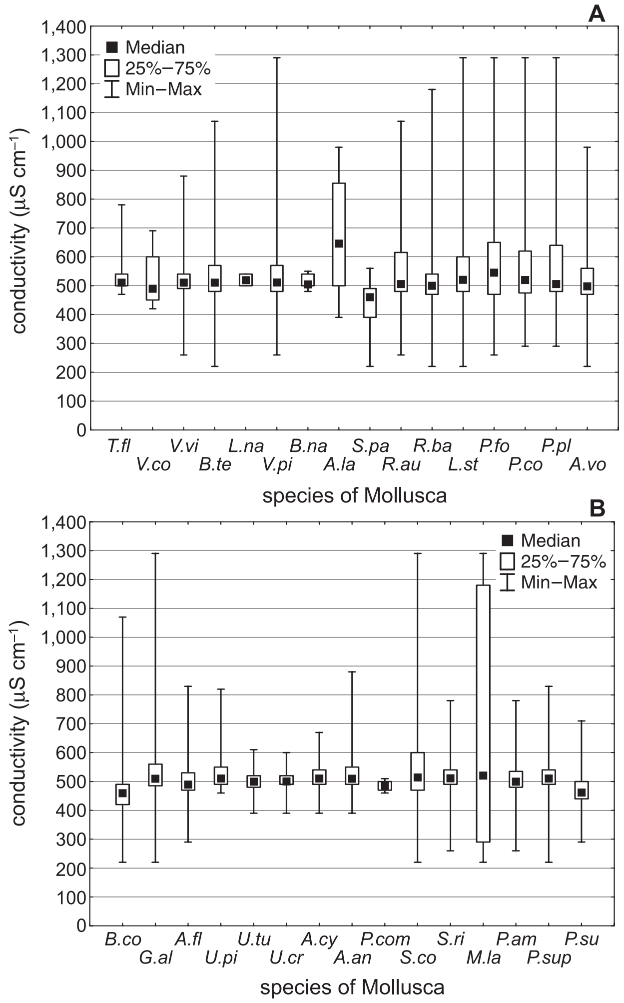


Fig. 39A, B. Box-and-whisker plot showing the conductivity of the water (µS cm^{-1}) (medians, interquartile ranges, minimum and maximum values) within which the mollusc species were recorded at sampling sites of the Wkra River and its tributaries

albus and two sphaeriid species, *S. corneum* and *M. lacustre*, occurred in the water of maximum conductivity of up to 1,290 µS cm^{-1}. The unionids, except *P. complanata* (460–510 µS cm^{-1}) and *U. crassus* (390–600 µS cm^{-1}), were recorded within 390–880 µS cm^{-1} (Figs 39A, B). Most of the mollusc species occurred at sampling sites with median values of conductivity that ranged from 400 to 600 µS cm^{-1}. *A. lacustris* and *M. lacustre* occurred within relatively wide ranges of the upper and lower quartiles (25th and 75th percentiles) (Figs 39A, B).

Only three species, *L. naticoides*, *Radix auricularia* and *M. lacustre*, were recorded at sampling sites with nitrate concentration below 50.00 mg NO_3^- dm^{-3} (Figs 40A, B). *L. stagnalis*, *P. corneus* and *P. planorbis* occurred in the water with the widest range of nitrate concentration and the maximum value up to 142.20 mg NO_3^- dm^{-3}. Among the bivalve species, *P. complanata* tolerated nitrate concentration of 7.09–65.12 mg NO_3^- dm^{-3}, *A. anatina* and *S. rivicola* up to 68.67 mg

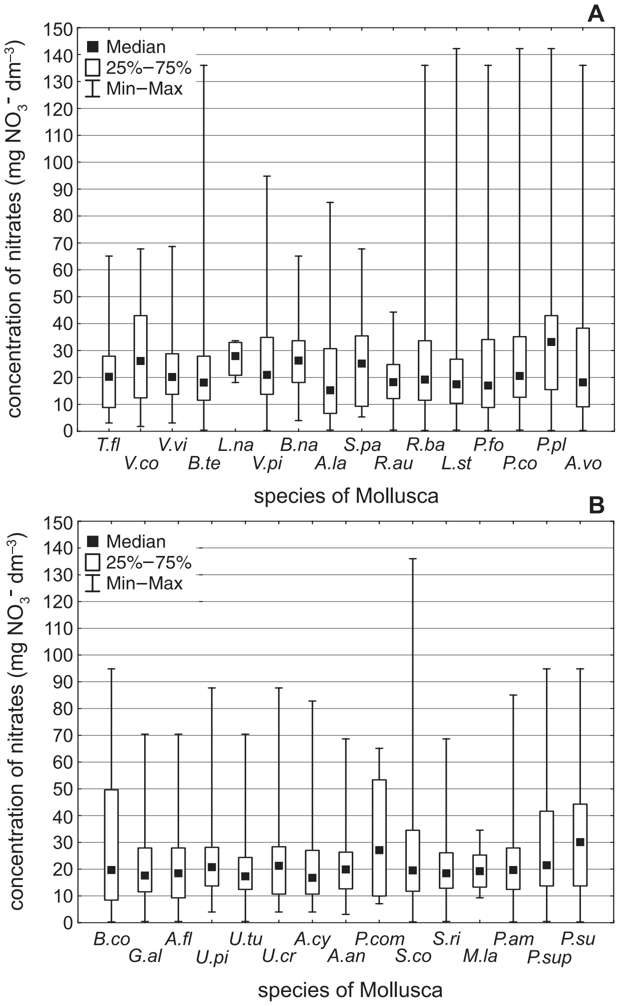


Fig. 40A, B. Box-and-whisker plot showing the concentration of nitrates in the water (mg NO_3^- dm^{-3}) (medians, interquartile ranges, minimum and maximum values) within which the mollusc species were recorded at the sampling sites of the Wkra River and its tributaries

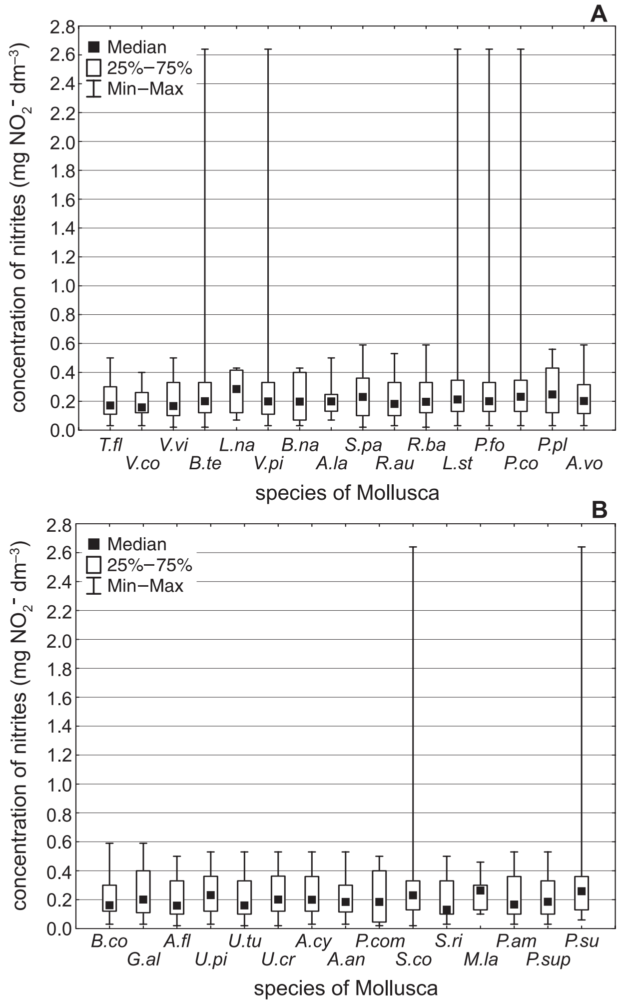


Fig. 41A, B. Box-and-whisker plot showing the concentration of nitrites in the water (mg NO_2^- dm^{-3}) (medians, interquartile ranges, minimum and maximum values) within which the mollusc species were recorded at the sampling sites of the Wkra River and its tributaries

NO_3^- dm^{-3}, and *U. pictorum* up to 87.71 mg NO_3^- dm^{-3}. What is more, *A. cygnea* was recorded at a sampling site with a relatively high concentration of nitrates of 82.84 mg NO_3^- dm^{-3} (site S3). *U. crassus* was recorded at sampling sites with a nitrate concentration up to 65.12 mg NO_3^- dm^{-3} (W23), but unexpectedly, a few specimens occurred at a site with a very high nitrate concentration of 87.71 mg NO_3^- dm^{-3} (site D10) (Figs 40A, B). Most of the mollusc species occurred at sites with a median nitrate concentration below 30.00 mg NO_3^- dm^{-3} with the exception of *P. planorbis* (median nitrate concentration 33.22 mg NO_3^- dm^{-3}). *U. crassus* was recorded at sampling sites with a median nitrate concentration of 21.26 mg NO_3^- dm^{-3} (Figs 40A, B).

Most mollusc species tolerated a nitrite concentration below 0.60 mg NO_2^- dm^{-3}, whereas unionid species tolerated concentrations of up to 0.50 mg NO_2^- dm^{-3}. However, *U. crassus* occurred in sites with the nitrite concentration of 0.02–0.53 mg NO_2^- dm^{-3}. Some species, including two prosobranch species, i.e. *B. tentaculata* and *V. piscinalis*, were recorded at sites with the maximum nitrite concentration of 2.64 mg NO_2^- dm^{-3}. Some species (*L. naticoides*, *S. palustris*, *P. corneus*, *P. planorbis*, *M. lacustre* or *P. subtruncatum*) occurred at sites with the median nitrate concentration above 0.20 mg NO_2^- dm^{-3} (Figs 41A, B).

Only three species: *L. naticoides*, *B. naticina* and *P. complanata*, occurred in water with the maximum phosphate concentration below 1.50 mg PO_4^{3-} dm^{-3} (Figs 42A, B). *B. tentaculata*, *V. piscinalis*, *A. lacustris*, *L. stagnalis*, *Ph. fontinalis*, *P. corneus*, *P. planorbis*, *A. vortex* and *S. corneum* were recorded at sites with a very high phosphate concentration in the water, exceeding 7.00 mg PO_4^{3-} dm^{-3} (site R3). A unionid, *U. tumidus* and fingernail clams, *S. rivicola* and *P. amnicum*, occurred in waters with a very high phosphate concentration of up to 3.01 mg PO_4^{3-} dm^{-3}. *U. crassus* was observed at sites with a relatively high phosphate concentration,

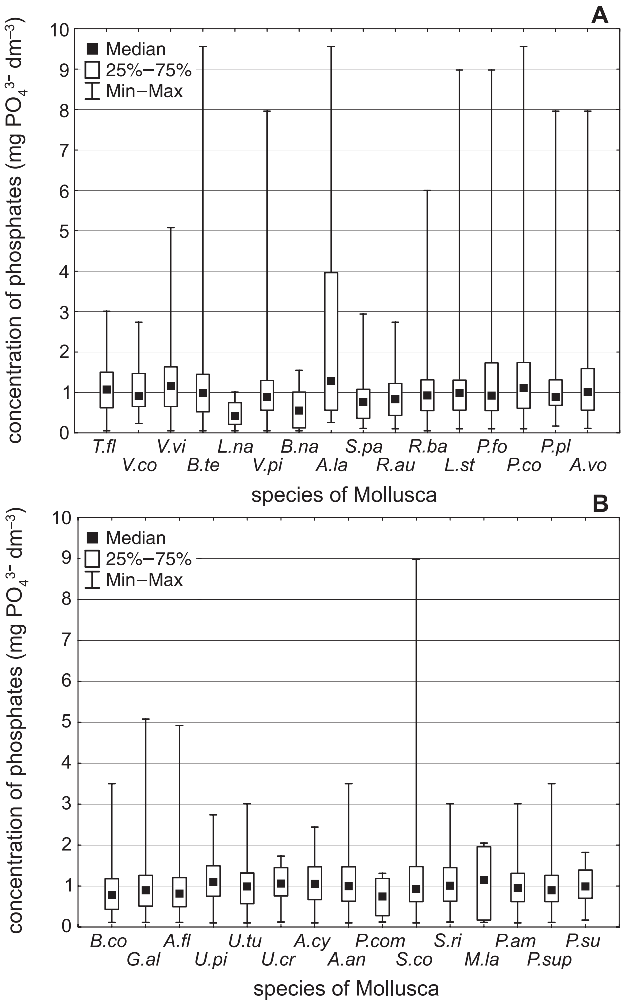


Fig. 42A, B. Box-and-whisker plot showing the concentration of phosphates in the water (mg PO_4^{3-} dm^{-3}) (medians, interquartile ranges, minimum and maximum values) within which the mollusc species were recorded at the sampling sites of the Wkra River and its tributaries

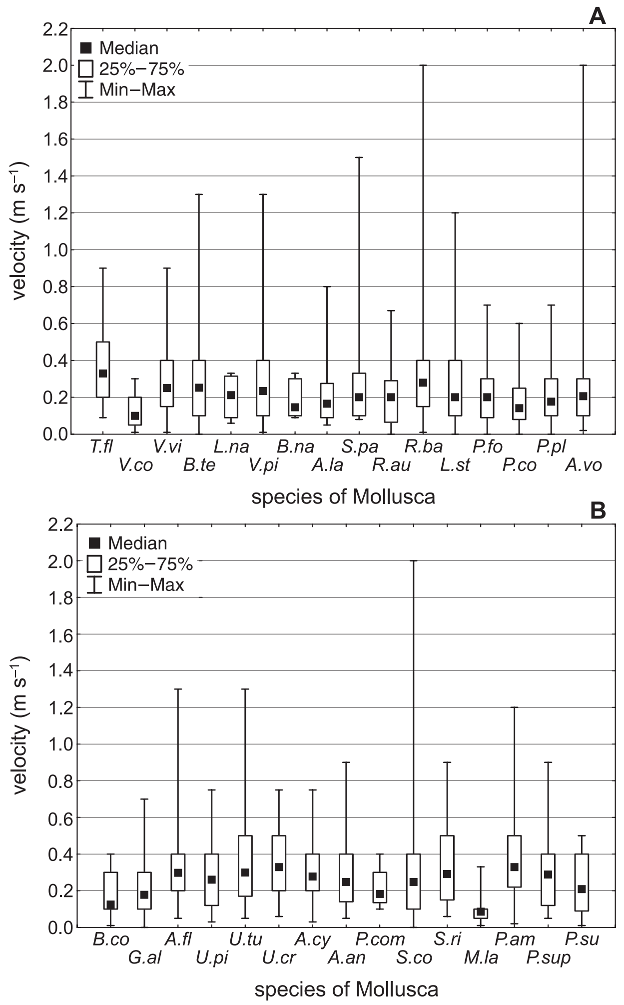


Fig. 43A, B. Box-and-whisker plot showing the velocity of water (m s^{-1}) (medians, interquartile ranges, minimum and maximum values) within which the mollusc species were recorded at the sampling sites of the Wkra River and its tributaries

up to 1.73 mg PO_4^{3-} dm^{-3} (Figs 42A, B) (site W21). Some species occurred at sites with a median phosphate concentration of up to 1.00 mg PO_4^{3-} dm^{-3}. *T. fluviatilis*, *V. viviparus*, *A. lacustris*, *P. corneus*, *U. pictorum* and *M. lacustre* occurred at sites with median phosphate concentrations exceeding 1.00 mg PO_4^{3-} dm^{-3}. *M. lacustre* and *A. lacustris* were recorded at sites with relatively wide ranges of lower and upper quartiles (25th and 75th percentiles) up to 2.00 mg PO_4^{3-} dm^{-3} and 4.00 mg PO_4^{3-} dm^{-3}, respectively (Figs 42A, B).

Some gastropods (*B. tentaculata*, *V. piscinalis*, *S. palustris*, *R. balthica*, *A. vortex*, *A. fluviatilis*) and bivalves (*U. tumidus*, *S. corneum* and *P. amnicum*) were found to tolerate wider ranges of water velocity, up to 1.3 m s^{-1} or even 2.0 m s^{-1} (Figs 43A, B). Some typical fluviatile species, e.g. *T. fluviatilis*, *A. fluviatilis* and *U. crassus*, occurred at sites with the median velocity ranging from 0.30 to 0.33 m s^{-1}. Unionids were recorded at sites with the median velocity ranging from 0.19 to 0.33 m s^{-1}. *M. lacustre*, which is typical of reservoirs, was recorded at sites with the lowest median velocity, i.e. 0.08 m s^{-1} (Figs 43A, B).

Prosobranchs: *T. fluviatilis*, *L. naticoides*, *B. naticina* and a unionid, *P. complanata*, were associated with river sites which were about 15 to 40 m wide, whereas a fingernail clam, *M. lacustre* was associated with a smaller width of rivers (up to 6 m). Most of the mollusc species were recorded at sites with widths of up to 60 m (Figs 44A, B). Unionid mussels were observed at sites where the median width of the river ranged from 15 to 30 m, whereas most gastropod species were recorded at sites where the median width of the river ranged from 10 to 30 m (Figs 44A, B).

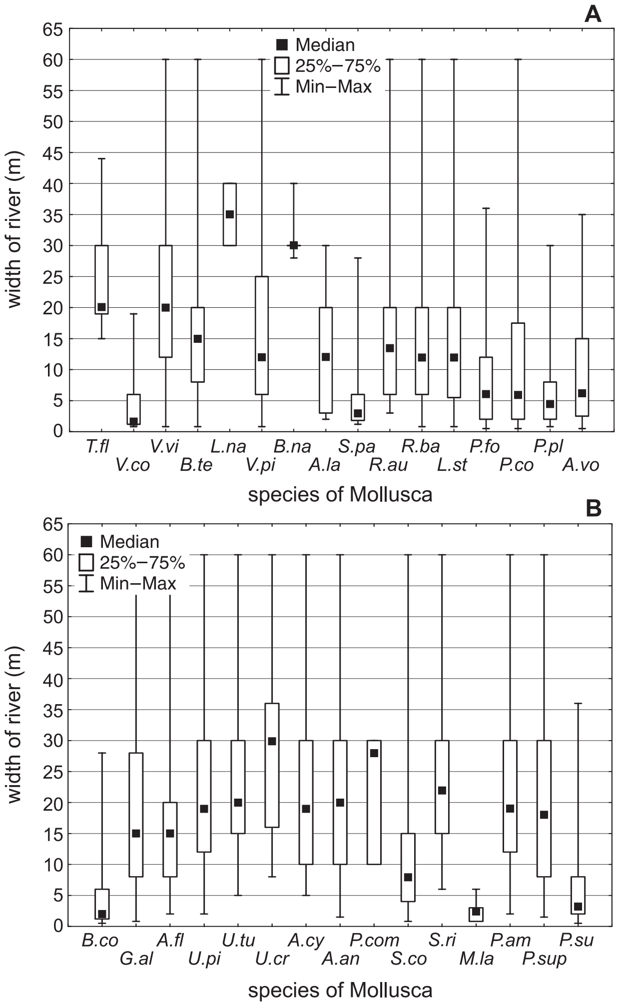


Fig. 44A, B. Box-and-whisker plot showing the width of river (m) (medians, interquartile ranges, minimum and maximum values) within which the mollusc species were recorded at the sampling sites of the Wkra River and its tributaries

4.2. OXBOW LAKES OF THE WKRA RIVER

4.2.1. PHYSICAL AND CHEMICAL PARAMETERS OF THE WATER, ORGANIC MATTER CONTENT IN BOTTOM SEDIMENTS AND GRAIN SIZE COMPOSITION

Data summarising the physical and chemical parameters of the water, the organic matter content in the bottom sediments and the grain size composition in the oxbow lakes of the Wkra River are given in Table 16.

The water temperature was 14.0–25.0°C. The values of conductivity and the total dissolved solids ranged from 380 (oxbow lakes Ox1 and Ox4) to 880 $\mu S\ cm^{-1}$ (Ox5) and from 180 to 420 mg dm^{-3}, respectively (Table 16). Lower pH values and less alkaline water were recorded in the oxbow lakes of the Wkra River (6.4–7.9), compared to the Wkra River (6.7–9.1). The concentration of dissolved oxygen in the water was relatively low at some of the sampling sites, particularly in oxbow lake Ox7. The waters of oxbow lakes Ox1–Ox6 showed lower nutrient concentrations than the Wkra River with the exception of oxbow lake Ox7; the concentration of nitrites and nitrates amounted up to 0.59 mg NO_2^- dm^{-3} and up to 73.10 mg NO_3^- dm^{-3} (oxbow lake Ox5). The maximum concentration of both nitrites and nitrates in oxbow lake Ox7 was very high and amounted to 3.30 mg NO_2^- dm^{-3} and 103.22 mg NO_3^- dm^{-3}, respectively. The concentration of phosphates ranged from 0.11 mg PO_4^{3-} dm^{-3} (oxbow lakes Ox6 and Ox7) to 4.64 mg PO_4^{3-} dm^{-3} (Ox5) and was lower compared to the Wkra River (up to 6.00 mg PO_4^{3-} dm^{-3}). The water hardness ranged from 190 mg $CaCO_3$ dm^{-3} (medium hard water, Ox4) to 415 mg $CaCO_3$ dm^{-3} (hard water, Ox5) and was higher than in the Wkra River (145–395 mg $CaCO_3$ dm^{-3}). The maximum chloride concentration in some oxbow lakes (up to 49 mg Cl^- dm^{-3}) was higher than in the Wkra River. The Kruskal-Wallis one-way ANOVA test revealed statistically significant differences in the average values of the environmental variables, i.e. conductivity ($H = 14.08$, $p = 0.02$), total dissolved solids ($H = 13.81$, $p = 0.02$), hardness ($H = 15.88$, $p = 0.007$), alkalinity ($H = 13.93$, $p = 0.02$) in sampling sites between the oxbow lakes. Statistically significant differences only in the values of nitrites ($H = 11.60$, $p = 0.009$) were obtained in the years of the survey between the sampling sites.

The organic matter content in the bottom sediments ranged from 0.54% (low organic matter content, oxbow lake Ox4) to 8.89% (high organic matter content, Ox6), whereas it ranged from 0.29 to 26.45% in the Wkra River (Tables 6 and 16). The differences in the average organic matter content at the sampling sites were not statistically significant either between the oxbow lakes (Kruskal-Wallis one-way ANOVA, $H = 9.19$, $p = 0.10$) or the years of the survey ($H = 1.09$, $p = 0.78$).

Fine sand (0.10–0.25 mm) constituted the highest percentage share (52.2–69.4%) in the grain size composition of bottom sediments in oxbow lakes Ox1–3 and Ox6–7. Medium-grained sand (0.25–0.50 mm) constituted the highest percentage (47.8–77.6%) in the grain size composition in the bottom sediments of oxbow lakes Ox4 and Ox5. Particles above 20.0 mm constituted from 0 to 16.5% in the grain size composition of the bottom sediments (Table 16). The Kruskal-Wallis one-way ANOVA test revealed statistically significant differences in the average grain size above 20.0 mm ($H = 14.24$, $p = 0.03$), 10.0–20.0 mm ($H = 16.76$, $p = 0.01$), 5.0–10.0 mm ($H = 16.79$, $p = 0.01$), 2.0–5.0 mm ($H = 19.37$, $p = 0.004$) and 1.0–2.0 mm ($H = 17.46$, $p = 0.008$) at sampling sites between the oxbow lakes. The differences in the average grain size at the sampling sites were not statistically significant between the years of the survey.

Table 16. Physical and chemical parameters of the water (ranges), organic matter content (%) in bottom sediments and grain size composition (%) in the oxbow lakes of the Wkra River

Parameter	Ox1	Ox2	Ox3	Ox4	Ox5	Ox6	Ox7
Temperature (°C)	18.5–22.5	19.0–23.0	18.0–22.0	22.0–24.0	21.0–25.0	14.0–22.0	16.0–21.5
Conductivity (μS cm^{-1})	380–430	390–450	400–460	380–450	430–880	480–610	580–690
Total dissolved solids (mg dm^{-3})	180–210	190–220	200–220	180–220	210–420	230–300	290–340
pH	7.1–7.9	7.0–7.6	6.8–7.7	6.4–7.3	7.2–7.6	6.8–7.5	6.6–7.9
Dissolved oxygen (mg dm^{-3})	3.1–6.3	4.2–6.0	3.4–6.7	3.8–5.8	2.5–4.9	1.4–6.5	0.6–6.2
Nitrites (mg NO_2^- dm^{-3})	0.10–0.53	0.07–0.33	0.10–0.43	0.17–0.36	0.10–0.59	0.03–0.36	0.03–3.30
Nitrates (mg NO_3^- dm^{-3})	7.08–21.54	1.89–38.98	6.20–33.20	21.60–43.86	19.05–73.10	7.10–45.19	13.73–103.22
Phosphates (mg PO_4^{3-} dm^{-3})	0.53–1.14	0.32–0.94	0.42–0.85	0.50–0.73	0.44–4.64	0.11–0.90	0.11–1.29
Hardness (mg $CaCO_3$ dm^{-3})	196–220	220–255	210–270	190–210	280–415	221–265	220–357
Alkalinity (mg $CaCO_3$ dm^{-3})	170–225	160–210	145–185	160–180	190–215	160–230	185–255
Chlorides (mg Cl^- dm^{-3})	17–32	18–31	22–30	17–40	18–22	21–49	23–42
Organic matter (%)	0.82–3.54	0.70–2.35	0.67–1.35	0.54–2.80	1.53–3.10	1.36–8.89	0.57–6.10
Grain size (diameter):							
above 20.0 mm	0%	0.5–3.4%	0.2–2.8%	0.3–1.8%	0.8–6.2%	5.5–16.5%	0.0–15.7%
10.0–20.0 mm	0.1–0.8%	1.1–3.2%	0.1–1.5%	2.1–2.8%	1.6–5.7%	1.0–19.8%	0.8–38.5%
5.0–10.0 mm	0.1–1.5%	1.6–4.7%	0.1–1.5%	3.0–4.6%	1.7–11.4%	3.7–18.5%	6.6–14.6%
2.0–5.0 mm	2.8–3.6%	0.6–5.2%	0.2–2.1%	1.5–3.6%	0.5–11.9%	2.8–11.7%	0.7–10.3%
1.0–2.0 mm	3.6–5.6%	0.3–3.9%	0.5–1.2%	0.8–6.7%	1.8–6.8%	0.6–12.4%	2.2–4.9%
0.80–1.0 mm	0.6–3.5%	0.4–4.1%	1.0–5.4%	0.4–12.8%	1.5–3.5%	0.2–7.9%	1.1–4.2%
0.50–0.80 mm	0.7–25.8%	0.2–9.2%	0.7–42.2%	2.6–28.2%	3.3–17.6%	0.4–15.5%	0.8–10.4%
0.25–0.50 mm	18.4–43.7%	20.0–59.6%	29.0–44.6%	23.2–77.6%	21.3–47.8%	9.1–30.2%	10.3–42.7%
0.10–0.25 mm	17.5–60.9%	25.2–69.4%	2.2–66.7%	6.0–60.3%	13.2–37.1%	5.9–53.3%	7.0–52.2%
0.05–0.10 mm	0.2–8.5%	2.4–4.6%	1.1–7.7%	0.4–9.6%	0.8–3.6%	1.4–4.6%	0.8–7.8%
below 0.05 mm	0.7–2.3%	0.1–0.7%	0.5–5.5%	0.1–1.3%	0.6–16.1%	0.7–1.9%	1.6–2.4%

4.2.2. MACROPHYTES

In total, 28 macrophyte taxa were recorded at the sampling sites in the oxbow lakes during the survey (Table 17). The number of taxa ranged from 8 (oxbow lake Ox1) to 15 (Ox7). *Stratiotes aloides*, which is typical of lentic water bodies, was found at only one oxbow lake (Ox4). Among Lemnaceae, *Lemna minor*, and *Spirodela polyrhiza*, were recorded in all of the oxbow lakes. Free-floating macrophytes, which are typical of oxbow lakes, e.g. *Hydrocharis morsus-ranae* and *Lemna gibba*, occurred in five out of seven reservoirs. The submersed macrophytes, e.g. *Ceratophyllum demersum* and *Elodea canadensis*, were recorded in most of the oxbow lakes, excluding *Myriophyllum spicatum*. The nymphaeid, *N. lutea*, which is legally protected in Poland (Dz.U. 2012a), was recorded in all of the oxbow lakes (Table 17).

4.2.3. STRUCTURE OF MOLLUSC COMMUNITIES AND VALUES OF INDICES IN THE OXBOW LAKES OF THE WKRA RIVER

In total, 30 mollusc species were recorded in the oxbow lakes: 19 gastropod species and 11 bivalve species. Three common species, usually found in still waters, *L. stagnalis*, *P. corneus* and *A. vortex*, were eudominants (D > 10.0%) and euconstants (C = 75.1–100.0%) in the mollusc communities of the oxbow

Table 17. Macrophyte occurrence in the oxbow lakes of the Wkra River

Species	Ox1	Ox2	Ox3	Ox4	Ox5	Ox6	Ox7
Acorus calamus L.	x	x	x	x	x		
Alisma plantago-aquatica L.	x					x	
Butomus umbellatus L.							x
Ceratophyllum demersum L. S. Str.				x	x	x	x
Elodea canadensis Michx.		x	x	x	x	x	x
Equisetum sp.							x
Glyceria fluitans (L.) R. Br.					x		
Glyceria maxima (Hartm.) Holmb.	x		x	x		x	x
Hydrocharis morsus-ranae L.		x	x	x	x		x
Lemna gibba L.	x		x		x	x	x
Lemna minor L.	x	x	x	x	x	x	x
Lemna trisulca L.		x		x	x		x
Lysimachia vulgaris L.				x			
Myriophyllum spicatum L.							x
Nuphar lutea (L.) Sibth. & Sm.	x	x	x	x	x	x	x
Phalaris arundinacea L.				x			
Phragmites australis (Cav.) Trin. Ex Steud.		x	x		x		
Rumex hydrolapathum Huds.					x		x
Sagittaria sagittifolia L.			x			x	
Schoenoplectus lacustris (L.) Palla	x	x			x		
Solanum dulcamara L.				x			
Sparganium emersum Rehmann						x	
Sparganium erectum L. Emend. Rchb. S. Str.			x				
Spirodela polyrhiza (L.) Schleid.	x	x	x	x	x	x	x
Sium latifolium L.			x			x	x
Stratiotes aloides L.				x			
Typha angustifolia L.		x					
Typha latifolia L.				x		x	x
Σ of species	8	10	12	14	13	12	15

lakes of the Wkra River. Twelve out of the 30 mollusc species were subrecedents ($D \leq 1.0\%$) and accedents ($C \leq 25.0\%$), including six bivalve species (Table 18). Among them three gastropod species are characteristic of permanent water bodies and are intolerant to desiccation, i.e. *Stagnicola corvus*, *B. contortus* and *H. complanatus*. Typical fluviatile species, *A. fluviatilis*, *P. amnicum* and *P. supinum*, were accedents in the mollusc community. Two bivalve species, *A. cygnea* (subdominant, accessory species) and *S. rivicola* (subrecedent, accedent), are legally protected in Poland (Dz.U. 2011a).

In the oxbow lakes of the Wkra River, the number of mollusc species ranged from 4 to 21, whereas the density of molluscs ranged from 20 to 157 indiv. m^{-2}. The highest mean number of mollusc species (16.0 ± 6.3) was recorded in oxbow lake Ox3, whereas the lowest mean number was recorded in oxbow lake Ox1 (5.2 ± 0.4) (Table 19). The mean density of molluscs ranged from 33.6 ± 15.5 (oxbow lake Ox1) to 94.4 ± 45.2 (Ox5). The mean density of gastropods was 49.3 indiv. m^{-2}, and the mean density of bivalves was 11.5 in the lentic oxbow lakes of the Wkra River. The maximum value of the H' index – 4.01 – for mollusc communities was obtained in Ox6, whereas the maximum value of H_{max} (4.39) was recorded in Ox3 (Table 19).

In the oxbow lakes of the Wkra River, the density of mollusc species varied from a few specimens per square metre (*S. palustris*, *P. supinum* or *U. tumidus*) to over a dozen (*P. corneus*, *S. corneum*) (Figs 45A, B). The density of *A. lacustris* (dominant, accessory species), a limpet which is mainly characteristic of lentic waters, *P. corneus*, which has poor powers of passive dispersal, or *A. vortex*, were highest in the oxbow lakes of the Wkra River and ranged from 2 to 31 indiv. m^{-2}, from 2 to 54 indiv. m^{-2}, and up to 34 indiv. m^{-2}, respectively (Fig. 45A). The density of typical fluviatile species: *A. fluviatilis* and *P. supinum* was low (2–3 indiv. m^{-2}), whereas the density of *P.*

Table 18. Values of the dominance (*D*%) and constancy (*C*%) indices calculated for the mollusc communities in the oxbow lakes of the Wkra River

Species	*D*%	*C*%
Viviparus contectus	8.1	94.3
Bithynia tentaculata	8.9	65.7
Valvata piscinalis	2.1	31.4
Acroloxus lacustris	5.4	37.1
Stagnicola palustris	0.2	5.7
Stagnicola corvus	0.5	5.7
Radix auricularia	1.4	25.7
Radix balthica	5.1	62.9
Lymnaea stagnalis	10.7	85.7
Physa fontinalis	3.5	57.1
Planorbarius corneus	14.8	82.9
Planorbis planorbis	1.4	22.9
Anisus vortex	15.0	97.1
Bathyomphalus contortus	0.4	11.4
Gyraulus albus	2.0	25.7
Gyraulus crista	1.0	17.1
Hippeutis complanatus	0.4	11.4
Segmentina nitida	1.1	22.9
Ancylus fluviatilis	0.1	2.9
Unio pictorum	2.3	25.7
Unio tumidus	0.1	2.9
Anodonta cygnea	3.2	37.1
Anodonta anatina	1.2	22.9
Sphaerium corneum	7.8	80.0
Sphaerium rivicola	0.2	2.9
Musculium lacustre	0.5	11.4
Pisidium amnicum	1.7	14.3
Pisidium henslowanum	0.5	8.6
Pisidium supinum	0.1	2.9
Pisidium subtruncatum	0.2	2.9
Σ of specimens	2,104	

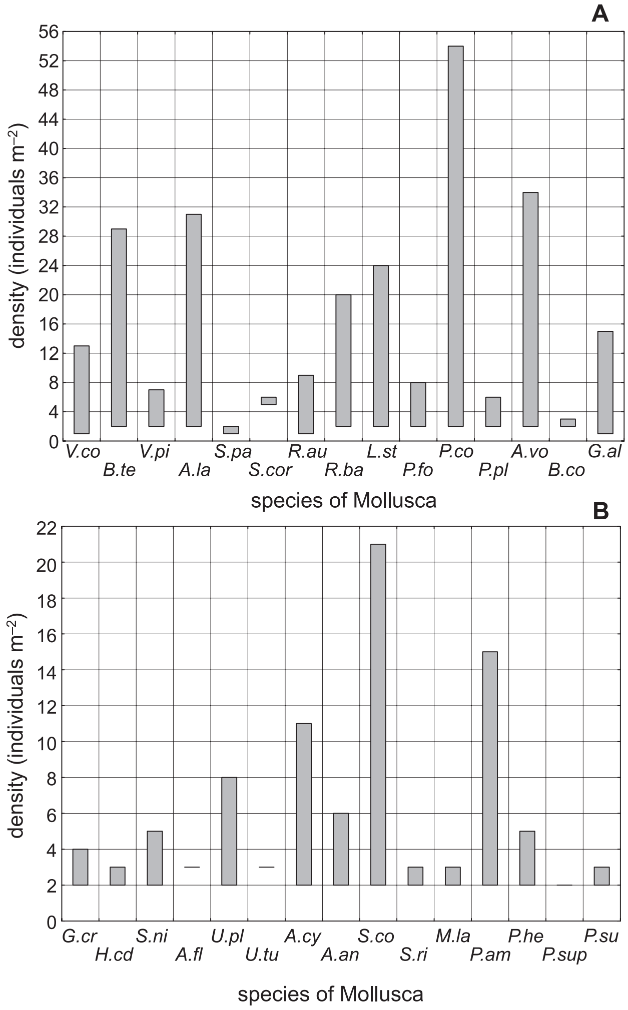


Fig. 45A, B. Density (ranges) of the mollusc species in the oxbow lakes of the Wkra River (2005–2009)

amnicum was higher and ranged from 2 to 15 indiv. m^{-2} (Fig. 45B).

The density of protected species, *A. cygnea* and *S. rivicola*, in the oxbow lakes ranged from 2 to 11 indiv. m^{-2} and from 2 to 3 indiv. m^{-2}, respectively (Fig. 45B).

In individual oxbow lakes of the Wkra River, the total number of mollusc species ranged from 7 (Ox1) to 23 (Ox6) (Table 20). Five species: *Viviparus*

Table 19. Number of mollusc species, density (individuals m^{-1}), values of *H*', H_{max} and *J*' of the mollusc communities in the oxbow lakes of the Wkra River at the sampling sites

Index	Ox1	Ox2	Ox3	Ox4	Ox5	Ox6	Ox7
No of species (ranges)	5–6	7–9	5–21	9–12	4–9	10–17	4–10
No of species (mean±SD)	5.2±0.4	7.4±0.9	16.0±6.3	11.0±1.2	7.8±2.2	15.0±2.8	6.0±2.4
Density (ranges)	20–58	27–49	24–117	48–65	46–157	45–149	22–51
Density (mean±SD)	33.6±15.5	37.4±8.0	84.6±35.4	53.6±6.8	94.4±45.2	76.0±42.8	41.2±1.3
Shannon-Wiener index (*H*')	2.11–2.45	2.42–2.87	2.12–3.99	2.72–3.52	1.79–2.93	3.08–4.01	1.80–2.99
Maximum Shannon-Wiener index (H_{max})	2.32–2.59	2.81–3.17	2.32–4.39	3.17–3.58	2.00–3.17	3.32–4.09	2.00–3.32
Evenness index (*J*')	0.91–0.96	0.86–0.97	0.84–0.92	0.86–0.98	0.83–0.92	0.86–0.98	0.88–0.98

contectus, *L. stagnalis*, *Ph. fontinalis*, *P. corneus* and *A. vortex* occurred in all the oxbow lakes. *V. contectus*, *A. lacustris*, *S. corvus*, *A. vortex*, *S. nitida* and *M. lacustre* are mainly characteristic of still waters.

Typical fluviatile species, *A. fluviatilis* and *P. supinum*, were subrecedents, whereas *P. amnicum* was dominant in the mollusc communities in oxbow lake Ox6. *H. complanatus*, which is especially characteristic of small water bodies, was subdominant in the mollusc community in oxbow lake Ox3. *A. lacustris*, a limpet that usually inhabits reservoirs and attaches itself to stiff leaves and stems of macrophytes, was dominant (oxbow lakes Ox2 and Ox6), eudominant (Ox3) and recedent in the mollusc communities of oxbow lake Ox1. Two legally protected bivalve species were found, i.e. *A. cygnea*, which was dominant in oxbow lakes Ox2–Ox4, and *S. rivicola*, which was recedent in oxbow lake Ox3 (Table 20).

The median number of mollusc species ranged from 5 (Ox1) to 18 (Ox3), whereas the median density ranged from 29 indiv. m^{-2} to 93 indiv. m^{-2}, respectively (Figs 46 and 47). The Kruskal-Wallis one-way ANOVA test revealed statistically significant differences in the average number of mollusc species (H = 22.22, p = 0.001) as well as the average values of mollusc density (H = 17.25, p = 0.008) in the sampling sites between the oxbow lakes. A multiple comparison post hoc test showed statistically significant differences in the average number of mollusc species between oxbow lakes Ox1, Ox3 and Ox6 and in the average density between oxbow

Table 20. Number of mollusc species and values of the dominance index (D%) calculated for the mollusc communities in the oxbow lakes of the Wkra River

Species	Ox1	Ox2	Ox3	Ox4	Ox5	Ox6	Ox7
Viviparus contectus	20.8	12.8	3.8	10.4	4.9	5.3	12.1
Bithynia tentaculata	–	3.7	3.3	12.3	15.7	13.2	4.4
Valvata piscinalis	–	–	3.1	–	4.2	3.2	–
Acroloxus lacustris	1.8	6.4	18.4	–	–	5.5	–
Stagnicola palustris	–	–	0.5	–	–	0.5	–
Stagnicola corvus	–	2.7	–	–	–	–	2.9
Radix auricularia	–	–	4.7	3.7	–	–	–
Radix balthica	–	5.3	3.1	4.8	3.4	14.2	0.9
Lymnaea stagnalis	7.7	17.7	7.8	14.2	7.4	8.7	19.4
Physa fontinalis	8.3	4.3	2.6	4.1	1.9	4.5	1.5
Planorbarius corneus	24.4	16.6	5.9	11.6	26.1	0.5	28.2
Planorbis planorbis	–	–	4.2	–	0.8	2.1	–
Anisus vortex	27.4	21.4	12.1	6.7	22.7	2.6	20.9
Bathyomphalus contortus	–	–	–	–	–	1.8	0.9
Gyraulus albus	–	–	1.9	–	–	8.4	–
Gyraulus crista	–	–	1.4	–	–	3.4	–
Hippeutis complanatus	–	–	2.1	–	–	–	–
Segmentina nitida	–	–	1.7	0.8	–	2.6	1.5
Ancylus fluviatilis	–	–	–	–	–	0.8	–
Unio pictorum	–	1.1	5.2	9.0	–	–	–
Unio tumidus	–	–	–	–	–	0.8	–
Anodonta cygnea	–	8.0	8.8	6.0	–	–	–
Anodonta anatina	–	–	2.8	4.8	–	–	–
Sphaerium corneum	9.5	–	5.4	10.4	12.9	5.8	7.3
Sphaerium rivicola	–	–	1.2	–	–	–	–
Musculium lacustre	–	–	–	1.1	–	1.8	–
Pisidium amnicum	–	–	–	–	–	9.5	–
Pisidium henslowanum	–	–	–	–	–	2.9	–
Pisidium supinum	–	–	–	–	–	0.5	–
Pisidium subtruncatum	–	–	–	–	–	1.3	–
No of species (S)	7	11	21	14	10	23	11

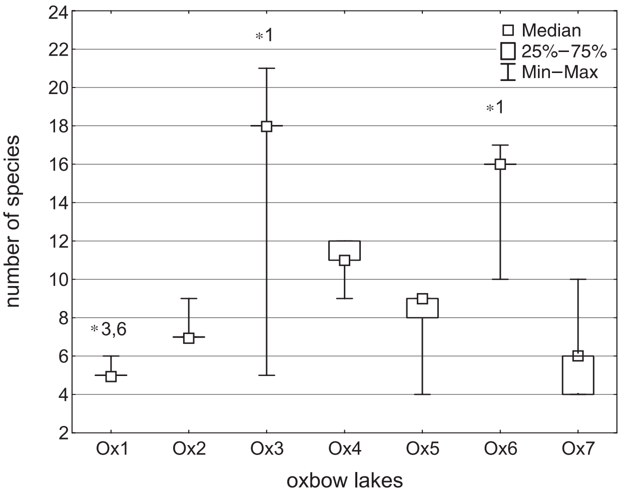


Fig. 46. Box-and-whisker plot showing the number of mollusc species in the oxbow lakes of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

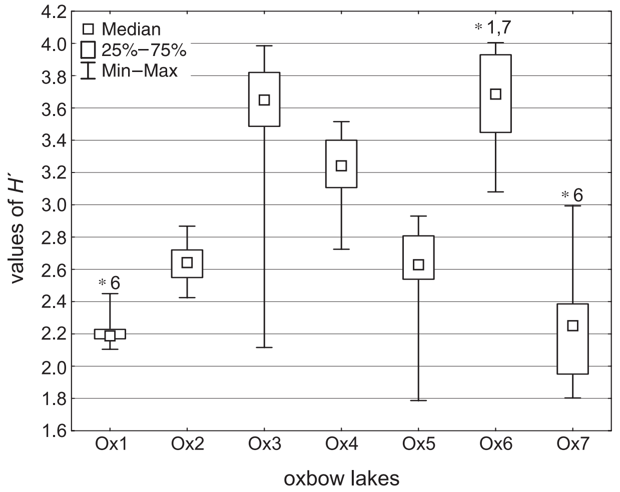


Fig. 48. Box-and-whisker plot showing the values of the H' index calculated for the mollusc communities in the oxbow lakes of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

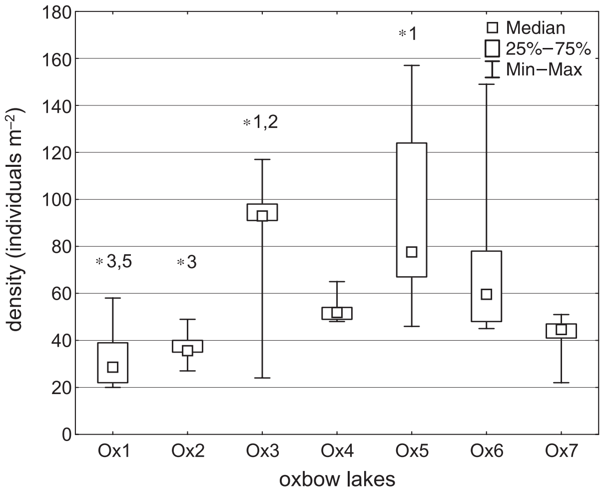


Fig. 47. Box-and-whisker plot showing the density of molluscs in the oxbow lakes of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

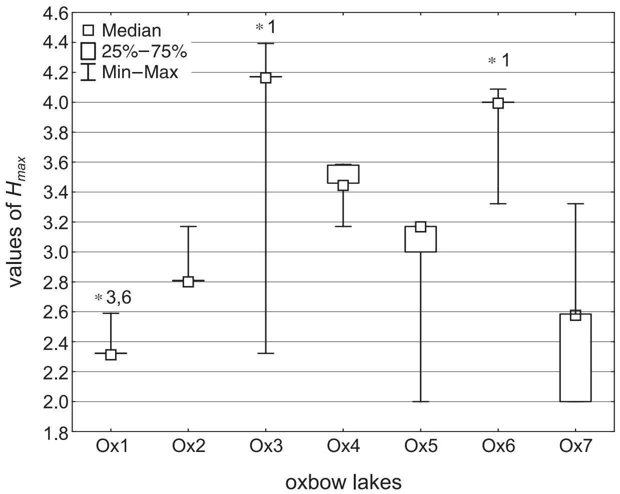


Fig. 49. Box-and-whisker plot showing the values of the H_{max} index calculated for the mollusc communities in the oxbow lakes of the Wkra River (medians, interquartile ranges, minimum and maximum values). Asterisks over a whisker denote significant differences between rivers (the Kruskal-Wallis one-way ANOVA and multiple comparison post hoc tests)

lakes Ox1, Ox2, Ox3 and Ox5 (Figs 46 and 47). The median values of the H' and H_{max} indices ranged from 2.20 (Ox1) to 3.70 (Ox6) and from 2.32 (Ox1) to 4.18 (Ox3). Statistically significant differences in the average values of the H' ($H = 21.63$, $p = 0.001$) and H_{max} ($H = 22.02$, $p = 0.001$) indices were recorded in the sampling sites between the oxbow lakes (Kruskal-Wallis one-way ANOVA test). A post hoc test showed statistically significant differences in the average values of the H' between oxbow lakes Ox1, Ox6, Ox7 as well as significant differences in the average values of the H_{max} between oxbow lakes Ox1, Ox3 and Ox6 (Figs 48 and 49). The differences in the average values of the J' in the sampling sites between the oxbow lakes were not statistically significant. The differences in the average values of the number of mollusc species, the average values of mollusc density, the H', H_{max} and J' were not statistically significant at sampling sites between the years of the survey.

4.2.4. MOLLUSC COMMUNITIES IN THE OXBOW LAKES OF THE WKRA RIVER IN RELATION TO SELECTED ENVIRONMENTAL FACTORS

Canonical correspondence analysis (CCA) based on the species data and environmental variables showed that the first and second axes explained 35.5% of the variance in the species data and 76.7% of the variance in the species and environment relationship (the first axis explained 23.6% of the variance in the species data and 44.9% of the variance in the species and environment relationship). The eigenvalues of axes 1, 2, 3 and 4 were 0.494, 0.297, 0.178 and 0.036, respectively. Species-environmental correlations of axes 1, 2, 3 and 4 were: 0.908, 0.791, 0.833 and 0.773.

Nitrates, water temperature and substrate particles ranging from 10.0 mm to 20.0 mm and from 0.25 mm to 0.50 mm as well as above 20.0 mm were the parameters that were most associated (statistically significant according to the forward selection results) with the distribution of mollusc species in the oxbow lakes of the Wkra River. The concentration of dissolved oxygen also exerted a significant influence on the distribution of mollusc species (Fig. 50).

The following patterns of mollusc distribution were observed: *V. piscinalis*, *Ph. fontinalis*, *P. planorbis*, *G. albus*, *G. crista*, *S. corneum*, *B. tentaculata* and *S. nitida* were negatively influenced by the nitrate concentration as well as substrate particles higher than 20.0 mm. *R. auricularia*, *U. pictorum*, *A. cygnea* and *A. anatina* were positively influenced by the substrate particles ranging from 0.25 mm to 0.50 mm or water temperature and negatively influenced by substrate particles ranging from 10.0 mm to 20.0 mm. *P. amnicum* was associated with the substrate particles ranging from 10.0 to 20.0 mm (medium and coarse gravel) and concurrently negatively correlated with the water temperature and the substrate particles ranging from 0.25 to 0.50 mm (medium-grained sand) (Fig. 50).

The relationship between the species composition of Mollusca and environmental variables was significant (Monte Carlo test of significance of the first canonical axis: *F*-ratio = 5.261, *p*-value = 0.002; test of significance of all canonical axes: *F*-ratio = 3.227, *p*-value = 0.002).

The number of mollusc species was negatively correlated with nitrite and nitrate concentrations, whereas the values of the *H*', H_{max} and *J*' were negatively correlated with the water temperature (Fig. 51). The relation between the values of the metrics and the environmental variables was statistically significant (Monte Carlo test of significance of the first canonical axis: *F*-ratio = 19.265, *p*-value = 0.002; test of significance of all canonical axes: *F*-ratio = 7.0, *p-value* = 0.002).

4.2.5. MOLLUSC SPECIES IN RELATION TO THE VALUES OF THE STATISTICALLY SIGNIFICANT ENVIRONMENTAL FACTORS

Four gastropod species, *V. contectus*, *L. stagnalis*, *P. corneus* and *A. vortex*, were recorded at sampling sites with relatively high nitrate concentrations of up to 103.22 mg NO_3^- dm^{-3} (Fig 52). Among unionid mussels, *U. tumidus* tolerated a nitrate concentration of

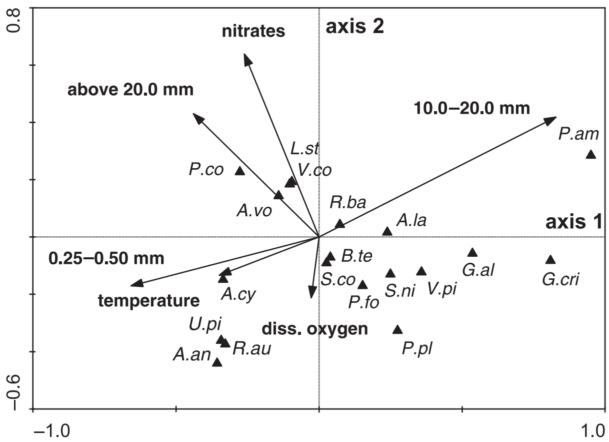


Fig. 50. Ordination diagram (biplot) based on canonical correspondence analysis (CCA) of the Mollusca data and environmental variables. Long arrows representing selected (statistically significant) environmental variables emphasize their impact on the structure of the mollusc communities in the oxbow lakes of the Wkra River

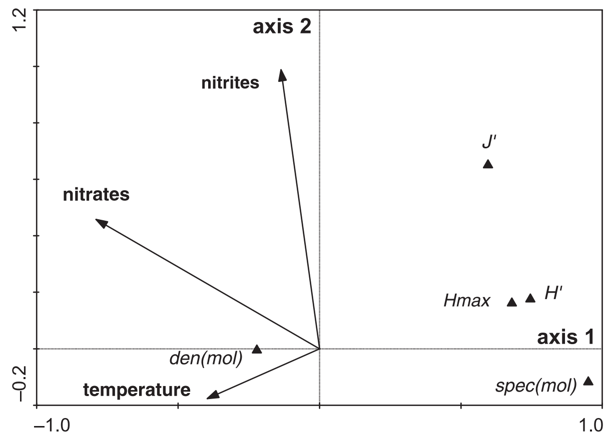


Fig. 51. Ordination diagram (biplot) based on canonical correspondence analysis (CCA) of the indices and selected environmental variables (oxbow lakes of the Wkra River). Abbreviations: *spec(mol)* – number of mollusc species, *den(mol)* – density of molluscs; *H'* – values of the Shannon-Wiener index, H_{max} – values of the maximum Shannon-Wiener index, *J'* – values of the Evenness index

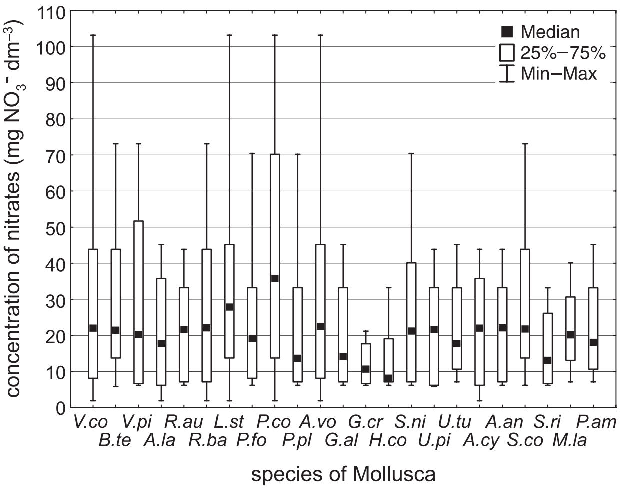


Fig. 52. Box-and-whisker plot showing the concentration of nitrates in the water (mg NO_3^- dm^{-3}) (medians, interquartile ranges, minimum and maximum values) within which the mollusc species were recorded at sampling sites in the oxbow lakes of the Wkra River

up to 45.19 mg NO_3^- dm^{-3}. A sphaeriid, *S. corneum*, was recorded at sampling sites with the maximum nitrate concentration of up to 73.10 mg NO_3^- dm^{-3} (75th percentile up to 43.86 mg NO_3^- dm^{-3}), whereas *S. rivicola* was recorded at sites with the maximum nitrate concentration of 33.20 mg NO_3^- dm^{-3} (75th percentile up to 26.12 mg NO_3^- dm^{-3}) (Fig. 52).

Most mollusc species occurred at sampling sites with the median nitrate concentration of about 20.00 to 30.00 mg NO_3^- dm^{-3}. *P. corneus* was recorded at sites with the median nitrate concentration of 35.74 mg NO_3^- dm^{-3} and relatively wide range of lower and upper quartiles (25th and 75th percentiles) of up to 70.00 mg NO_3^- dm^{-3} (Fig. 52). *H. complanatus* was observed at sites with the lowest median nitrate concentration of 8.10 mg NO_3^- dm^{-3}.

DISCUSSION

5.1. THE WKRA RIVER AND ITS TRIBUTARIES

5.1.1. STRUCTURE OF MOLLUSC COMMUNITIES AND VALUES OF INDICES IN LOWLAND RIVERS

A long-term survey on the mollusc communities revealed the occurrence of 44 mollusc species: 30 gastropod species and 14 bivalve species in the Wkra River and its tributaries (Ecoregion 16, Eastern Plains). Until 2009, no alien bivalve species had been recorded in the Wkra River and its tributaries. There were up to 40 mollusc species in the Wkra River and from 10 to 32 species in its tributaries. This result is in contrast to Korycińska's (2002) survey, which showed a much smaller number of mollusc species, i.e. 21 species in a lowland river of Ecoregion 16, as well as to Strzelec & Królczyk (2004) and Raczyńska & Chojnacki (2009) who recorded 14 gastropod species and 8 mollusc species, respectively, in the lowland rivers of Ecoregion 14 (Central Plains). Considering Polish lowland rivers of a similar length and adjacent catchment area, the number of mollusc species was higher in the Wkra River (40 species) compared to the selected pristine sampling sites of the Pilica River (33 species) (Ecoregion 14) (Nijboer et al. 2006). For comparison, in very large rivers, i.e. the Odra River, the Vistula River and their selected tributaries, 23 and 27 mollusc species, including 13 and 10 bivalves, respectively, were recorded (Lewandowski 2004, Zettler 2012). In contrast, 55 mollusc species, including eight alien species, were found in the middle and lower sections of the Odra River in 2009–2011 (Piechocki & Szlauer-Łukaszewska 2013).

Typical fluviatile, legally protected species were found in the Wkra River and its tributaries: *P. complanata*, *B. naticina*, *S. rivicola* and *U. crassus*. However, they were not recorded in the selected pristine sampling sites of Polish lowland rivers (central Poland), or only single (*R. auricularia*, *S. palustris*, *A. cygnea*, *U. crassus*) or few specimens (*A. anatina*, *U. pictorum*, *U. tumidus*) were observed (Nijboer et al. 2006). The density of gastropod species and Unionidae in the Wkra River and its tributaries was much higher than in those pristine sampling sites.

Six unionid species out of the seven recorded from Poland to date, including one non-indigenous (Piechocki 2008), were found in the Wkra River. Similar to the results of Piechocki (1969), *S. nitida* was recorded only in the upper course of the Wkra River, whereas *P. complanata* and *B. naticina*, species that are typical of larger rivers, occurred in the lower course. *B. tentaculata* was euconstant in the mollusc communities in the Wkra River and some of its tributaries. *B. tentaculata* was also euconstant in the Serbian stretch of the Danube (Martinovic-Vitanovic et al. 2013). The authors explained its occurrence at most of their sampling sites by its feeding behaviour, because it can both graze scraping material from the substratum and filter phytoplankton using ctenidium. However, filter feeding may be more efficient than scraping, and this might explain the high abundance of *B. tentaculata* especially in nutrient-rich habitats. The present results are in contrast to PIECHOCKI's (1969), who obtained a decrease in the number of mollusc species along the Grabia River (Ecoregion 14) from 32 in the upper course to 24 in the lower course. Statistically significant differences in the number of mollusc species as well as in their density and species diversity measured by the values of the Shannon-Wiener index (*H*') among the upper, middle and lower courses, which showed a decreasing tendency from the head to the mouth in a large river in an agricultural area, were obtained by MCRAE et al. (2004). The number of mollusc species, their density and the values of the *H*' index showed an opposite tendency in the Wkra River. The lowest number (from 6 to 11 mollusc species) was recorded in the upper course of the Wkra River, whose stretch has been regulated to a great degree. The highest number of species or values of *H*' were revealed at site W23, which has almost no regulation and is located in the lower course.

According to PIECHOCKI (1992), some sphaeriid species, e.g. *S. corneum*, *P. amnicum*, *P. henslowanum* or *P. supinum*, occur with a high frequency in lowland rivers. *P. subtruncatum* and *P. casertanum* predominate in smaller rivers, and *S. rivicola* in larger rivers mainly. The present results confirm the finding of PIECHOCKI (1992): sphaeriid species, *S. corneum*, *P. amnicum* and *P. supinum*, occurred at most of the sampling sites in the Wkra River and its tributaries, whereas *P. subtruncatum* and *P. casertanum* were eudominants in the mollusc communities only in the upper course of the Wkra River. *S. rivicola* inhabited the lower course of the Wkra River and also some of its tributaries.

The mollusc samples were collected five times during five years (each August), from the head to the mouth from each sampling site in the Wkra River and 40 species of Mollusca were recorded. By contrast, VIRBICKAS et al. (2011) obtained only 13 mollusc species during his survey (agricultural area, Lithuania) in a lowland river of a similar length and catchment area to the Wkra River. However, he sampled only once at each sampling site. Thirty three mollusc species (18 gastropod species and 15 bivalve species including 5 alien species) were recorded during a six-year survey of the 396.6 km long Serbian stretch of the Danube, whereas only 26 mollusc species were obtained during a one-year survey of the same stretch of the Danube River (MARTINOVIC-VITANOVIC et al. 2013, TUBIĆ et al. 2013). In comparison, 22 bivalve species including five alien species occurred in the Hungarian stretch of the Danube River and its tributaries (samples were collected four times a year) (BÓDIS et al. 2011). However, 85 mollusc species were found in the entire course of the Danube River (MARTINOVIC-VITANOVIC et al. 2013). This survey showed that the number of mollusc species per site ranged from 1 to 21. Similarly, VON OHEIMB et al. (2007) recorded 3–19 mollusc species per site in lowland rivers; however, the number of species was lower (29 species) than in the Wkra River catchment area. Some authors obtained a lower number of mollusc species, number of species per site or mean density in European rivers compared to the Wkra River catchment area: 21 gastropod species (rivers in Switzerland) and from 15 to 24 (including 3–5 alien species) in the rivers of the Iberian Peninsula (BAUR & RINGEIS 2002, SOUSA et al. 2007, PÉREZ-QUINTERO 2007, 2011). In contrast, the number of mollusc species ranged from 3 to 28 at sampling sites in the Serbian stretch of the Danube (a very large river) (MARTINOVIC-VITANOVIC et al. 2013). According to PÉREZ-QUINTERO (2007), larger drainage areas support a higher number of mollusc species. These results confirmed his findings: 40 mollusc species were recorded in the Wkra River (a large river according to the catchment area), whereas 10–32 species were recorded in its tributaries (medium-sized rivers). Therefore, the number of mollusc species not only depends on the size of the catchment area, but also on how intensive (frequent) the sampling is. By comparison, 44 mollusc species, including 10 alien species, were collected in the Rhine River (a very large river according to the catchment area), one of the largest rivers in Europe (VERBRUGGE et al. 2012).

5.1.2. RARE, THREATENED AND ALIEN SPECIES IN THE MOLLUSC COMMUNITIES OF LOWLAND RIVERS

The decline of the world's freshwater mollusc species can be attributed to anthropopressure (e.g. acidification, salination, siltation from agriculture and logging, pesticide and heavy metal load, agricultural development) or life history traits (STRONG et al. 2008). About 44% (373 species) of freshwater Mollusca are threatened in Europe and about 50% at the level of the 27 Member States of the European Union (CUTTELOD et al. 2011). The major threats to European freshwater mollusc, which lead to population decline, are the intensification of agriculture including the increased use of chemical fertilisers and pesticides and poor sewage management, which result in a decreasing water quality in rivers and lakes, as well as a modification of the water sources and changes in the flow regime. Therefore, monitoring the population size, distribution and trends (possibly through monitoring the habitats) should be undertaken, including those of threatened and Data Deficient species.

The Wkra River catchment area holds five mollusc species which are protected by the Polish legislation: *B. naticina*, *U. crassus*, *A. cygnea*, *P. complanata* and *S. rivicola* (DZ.U. 2011a). Among these *B. naticina* belongs to the rarest and most threatened (Critically Endangered, CR) freshwater mollusc species in

Poland according to the Polish Red Data Book of Animals (Piechocki 2004a). *B. naticina* mainly inhabits large and very large lowland rivers, especially their middle and lower courses (Piechocki 1979, Zettler 2012). *B. naticina*, which occurred only in the lower course in the Wkra River, was a subrecedent and accedent species in the mollusc communities, whereas it was a constant species in a very large river, the Serbian stretch of the Danube (Martinovic-Vitanovic 2013). According to the European Red List of Non-marine Molluscs (Cuttelod et al. 2011), *B. naticina* is listed as a Least Concern (LC) species, of equal rank (status) with e.g. *V. piscinalis*, *B. tentaculata* or *V. viviparus*. Contrary to its status as shown by Cuttelod et al. (2011), I support the opinion of Zettler (2012) that *B. naticina* is at least endangered in some parts of its distribution area.

A. cygnea, a species typical of still waters or slow-flowing lowland rivers, was an accedent or accessory species in the mollusc communities of the Wkra River and its tributaries. In comparison, the swan mussel was accedent in large rivers, i.e. in the middle and lower Odra River (Piechocki & Szlauer-Łukaszewska 2013) as well as in the Serbian stretch of the Danube (Tubić et al. 2013). *A. cygnea* is rare or absent from degraded European rivers (Michalik-Kucharz et al. 2000, von Oheimb et al. 2007). Its average density ranged from 0.4 to 4.5 with a maximum density of even up to 28 indiv. m^{-2} in European lowland rivers (Abraszewska-Kowalczyk 2002, Sousa et al. 2007). In contrast, the maximum density of *A.cygnea* was 10 indiv. m^{-2} in the Wkra River and its tributaries.

The depressed river mussel, *P. complanata*, was observed at only two out of the 24 sampling sites with the maximum density of up to 5 indiv. m^{-2} in the Wkra River. It was also rare and occurred at eight out of 46 sampling sites with the maximum density of 6 indiv. m^{-2}; the sites were located in the mainstream of the Pilica River (Ecoregion 14) (Abraszewska-Kowalczyk 2002). *P. complanata*, a typical rheophilous species, is rare in Polish lowland rivers (Zając 2004a). In large European rivers, it was accedent in the Serbian stretch of the Danube (Tubić et al. 2013). The mean density of the depressed river mussel was lower than 0.4 indiv. m^{-2} in the middle Danube catchment area in Hungary (Bódis et al. 2011). The range of *P. complanata* extends across most European countries: Finland, Sweden, Russia, Germany, Austria, Switzerland, France, the Netherlands and the United Kingdom. The depressed river mussel has disappeared from approximately 30% of its historical sites in England and Scotland (Kileen et al. 2004). Although the decline in the occurrence of *P. complanata* was observed in recent years, the United Kingdom is home to some of the largest populations in the world (McIvor & Aldridge 2007). Currently, according to the European Red List of Non-marine Molluscs (Cuttelod et al. 2011), *P. complanata* is classified as Near Threatened (NT) at the European and at the EU 27 levels.

The river orb mussel, *S. rivicola*, is a rheophilous species that is protected by the Polish legislation (Dz.U. 2011a) as well as regarded as Vulnerable (VU) according to the Polish Red List of Species (Dyduch-Falniowska & Zając 2002). In contrast, according to the European Red List of Non-marine Molluscs (Cuttelod et al. 2011), *S. rivicola* is listed as a Least Concern (LC) species at the European and at the EU 27 levels of equal rank (status) with *S. corneum*. The river orb mussel lives in large, slow-flowing lowland rivers or canals; it appears to require deeper water than other *Sphaerium* and *Pisidium* species (Killeen et al. 2004). *S. rivicola* was an accessory species in the Wkra River, whereas it was accedent in its tributaries. It was an accessory species in the Serbian stretch of the Danube (Tubić et al. 2013) and in the middle and lower Odra River (Piechocki & Szlauer-Łukaszewska 2013).

An alien species, the gravel snail, *L. naticoides*, which was a recedent and simultaneously accedent species in the mollusc communities in the Wkra River, is not protected in Poland. However, it was included in both the Polish Red List of Species and the Polish Red Book of Animals as Endangered (EN) (Piechocki 2002, 2004b). In Poland, *L. naticoides*, which originates from the western Black Sea, occurs in large and medium lowland rivers (Falniowski 1987, Piechocki 1996, Lewandowski 2004, Kołodziejczyk 2011, Piechocki & Szlauer-Łukaszewska 2013). Its density was up to 50 indiv. m^{-2} in the Wkra River; it ranged from 1 to more than 500 indiv. m^{-2} in the Odra River (Piechocki & Szlauer-Łukaszewska 2013). Its strong decline, which has been attributed to water pollution and river regulations, has been observed within the last 40–50 years. In contrast, the gravel snail was euconstant in the mollusc communities in the 396.6 km long Serbian stretch of the Danube (Martinovic-Vitanovic et al. 2013). Though its density reached 10,608 indiv. m^{-2}, an extreme decrease of down to 74 indiv. m^{-2} was recorded during the next year. The density of the gravel snail in the Saône River varied from 243 to 1,090 indiv. m^{-2} and its decreasing tendency down to 16 indiv. m^{-2} was also observed (Mouthon 2007). In this case, the decline in the population of *L. naticoides* was explained not by pollution, but by an increase in the temperature of the water as a result of the global warming as well as possible interspecific competition between *L. naticoides* and *V. piscinalis*.

Some species which were found in the Wkra River catchment area, e.g. *U. crassus* and *S. rivicola*, are included in the IUCN Red List of Threatened Species (IUCN 2012) as Endangered or Vulnerable (EN, V), whereas *H. complanatus*, *L. naticoides*, *A. cygnea*, *A. anatina* and *U. tumidus* belong to the Least Concern species (LC). The highly invasive alien species, *P. antipodarum*, which is able to achieve upstream movement of 0.66 to 1.20 m per day (Kappes & Haase 2012) and reproduces throughout the year in the Polish climatic conditions (Lewin 2012), belongs to the subrecedents in the mollusc communities in the Wkra River catchment area.

5.1.3. *UNIO CRASSUS* PHILIPSSON, 1788 AS A SPECIES OF THE EUROPEAN COMMUNITY INTEREST

From the beginning of the 20th century, the thick-shelled river mussel (*U. crassus*) was regarded as the most common unionid species in central Europe, reaching a density of more than 700 indiv. m^{-2}. Its populations declined dramatically in western and central Europe during the second half of the 20th century and a further, significant decline was observed within the last 50 years (Zettler & Jueg 2007, Douda et al. 2012, Denic et al. 2013). The dramatic decrease in the population density has led to a consideration of this species within the European law regulations. Therefore, under Annexes II and IV to Council Directive 92/43/EEC (1992) of EU and its consolidation version of 01.01.2007 which includes the latest version of the annexes and the Polish legislation (Dz.U. 2012b), *U. crassus* is one of species of Community interest whose conservation requires designation of special conservation areas within the Habitats Directive Natura 2000.

Currently, according to the European Red List of Non-marine Molluscs (Cuttelod et al. 2011), *U. crassus*, which mainly inhabits flowing waters from large rivers to small streams, is classified as Vulnerable (VU) at the European and at the EU 27 levels, whereas on the IUCN Red List of Threatened Species, it is listed as Endangered (EN) (Iucn 2012). The gradual extinction of *U. crassus* resulting from water pollution, eutrophication or degradation of the natural river valleys has been observed in Poland, like in other European countries (Piechocki & Dyduch-Falniowska 1993, Abraszewska-Kowalczyk 2002, Dyduch-Falniowska & Zając 2002, Zając 2004b).

This survey showed that more than a few specimens of *U. crassus* were continuosly recorded only at five out of 14 sampling sites in the Wkra River in consecutive years between 2005 and 2009. In contrast, the population size of the thick-shelled river mussel varied between one hundred and six hundred thousand with a maximum density up to 50 indiv. m^{-2} in the rivers of the Mecklenburg-Vorpommern (Germany) (Zettler & Jueg 2007). The maximum density of *U. crassus* was to 20 indiv. m^{-2} in the Wkra River, and only 3 indiv. m^{-2} in the Pilica River (Ecoregion 14) (Abraszewska-Kowalczyk 2002), and from 5 to 50 indiv. m^{-2} in the Cedron River (foreland of the Polish Carpathians, southern part of Ecoregion 16) (Zając & Zając 2011). A long-term survey showed that *U. crassus* was an accedent species in the mollusc communities in the Wkra River and was also accedent in the 396.6 km long stretch of the Serbian Danube (Martinovic-Vitanovic et al. 2013).

This survey revealed (CCA analysis) that from among the physical parameters, river width, velocity and water temperature were most associated (statistically significant) with the distribution of *U. crassus*. The thick-shelled river mussel occurred at sites with water velocity of 0.06–0.60 m s^{-1} and a sandy bottom (fine, medium or coarse sand). This is in accordance with the results of Abraszewska-Kowalczyk (2002) who highlighted the importance of the physical factors, e.g. the type of bottom and the sediment grain size, in addition to the physical and chemical parameters of the water, which affect the occurrence of unionid species in rivers, including *U. crassus*. According to Abraszewska-Kowalczyk (2002), *U. crassus* occurs both near the shores in the shallows and on a sloping bottom as well as in the current. The thick-shelled river mussel avoids organic-rich muddy substratum and is more abundant in sandy tributaries. Adult specimens of *U. crassus* are able to move actively towards more preferable environmental conditions (Zając & Zając 2011). *U. crassus* prefers fine sediments of a steep bank in a pool with a minimum of water flow. If individuals were present in the riffle, they were burrowed under the gravel armour of the bottom. It was shown (Zając & Zając 2011) that *U. crassus* occurred in pools with the maximum water velocity up to 0.59 m s^{-1} which is comparable to those recorded in the Wkra River.

The recruitment of juveniles (< 30.0 mm shell length or 1–3 years old specimens) was recorded in the Wkra River. Moreover, individuals up to six years old with the maximum shell length of 66.9 mm were also observed there. The maximum shell length and age of specimens varied among rivers and amounted to 47.8 mm (10 years old) in the Płociczna River (Poland, Ecoregion 14), 68.0 mm (18 years old) in the Szeszupa River (Poland, Ecoregion 16), 79.1 mm (6 years old) in the Pilica River and its tributaries (Poland, Ecoregion 14), 85.0 mm (25 years old) in the Höjea River (Sweden) as well as 94.0 mm (32 years old) in the Suså River (Denmark) (Zettler 2000, Abraszewska-Kowalczyk 2002).

The larvae of *U. crassus* (glochidia) like other unionoid species, must undergo a period of metamorphosis as obligate ectoparasites on specific host fishes. The glochidia become encysted by host tissues and then undergo a metamorphosis into juvenile mussels living independently by burrowing in the sediments. As a result, Unionoidea require a host fish in order to complete their successful recruitment. Therefore, the potential threats to the endangered *U. crassus* result from a deterioration in its host resources, i.e. fishes, especially in Central Europe. The local extinction of the thick-shelled river mussel is probably associated with the impaired status of fishes and with the absence of the primary host fishes, i.e. *Phoxinus phoxinus* (L.), *Scardinius erythrophtalmus* (L.), and *Cottus gobio* L., in European rivers (DOUDA et al. 2012). Thus, not only does the pollution of rivers and habitat quality affect this species directly but also indirectly, i.e. by influencing the composition of ichtyofauna as well as the loss of the obligate host fish species (ZETTLER & JUEG 2007).

KÖHLER (2006) showed two patterns of nitrate nitrogen concentrations in rivers which are inhabited by *U. crassus*. The first type refers to a low annual concentration of NO_3–N dm^{-3} (about 2.00 mg, which corresponds to 8.86 mg NO_3^- dm^{-3}) during the whole season and the second type refers to a relatively high concentration of NO_3–N dm^{-3} during the winter, while during every vegetation period (summer) the concentration of NO_3–N dm^{-3} does not exceed 2.00 mg dm^{-3}. The concentration of nitrates (or nitrate nitrogen) in the Wkra River did not conform to this pattern. However, the nitrate (or nitrate nitrogen) concentration was recorded during every vegetation period (summer), and its values exceeded 2.00 mg NO_3–N dm^{-3}, which was much higher compared to the results obtained by KÖHLER (2006).

The mortality of juveniles is related to the concentration of nitrate nitrogen in the water. In rivers with a nitrate nitrogen concentration below 2.00 mg NO_3–N dm^{-3}, successful growth of juveniles is observed. In moderately polluted rivers with a nitrate nitrogen concentration between 2.00 and 10.00 mg NO_3–N dm^{-3} (which corresponds to 44.30 mg NO_3^- dm^{-3}), no or only a very limited recruitment was observed. In waters with up to 20.00 mg NO_3–N dm^{-3} (which corresponds to 88.60 mg NO_3^- dm^{-3}), the population of *U. crassus* has already become extinct or nearly extinct (ZETTLER & JUEG 2007). Successful growth of juveniles and recruitment was recorded in non-polluted rivers with a very low concentration of nitrites (up to 0.11 mg NO_2^- dm^{-3}), nitrates (up to 2.60 mg NO_3^- dm^{-3}) or phosphates (up to 0.35 mg PO_4^{3-} dm^{-3}) in the foreland of the Polish Carpathians (southern part of Ecoregion 16) (HUS et al. 2006). However, the authors of the survey stated that *U. crassus* preferred waters with slightly elevated concentration of nutrients to non-polluted rivers in the foreland of the Polish Carpathians. It was shown that *U. crassus* reproduced in the Lužnice River (the Czech Republic) where the upper limit concentration of NO_3–N dm^{-3} was 2.00 mg (median values) (DOUDA 2010). This value corresponds to about 8.86 mg NO_3^- dm^{-3}. DENIC et al. (2013) found a recent recovery of *U. crassus* in a stream with elevated nitrate nitrogen mean concentration regimes of 6.40 mg NO_3–N dm^{-3} (which corresponds to about 28.35 mg NO_3^- dm^{-3}) in free-flowing water and 4.10 mg NO_3–N dm^{-3} in the substratum. In contrast, recruitment of juveniles was recorded at sampling sites with the maximum concentration of nitrates ranging from 27.91 to 87.71 mg NO_3^- dm^{-3} in the Wkra River (sampling sites D10, W12, W21, W23, W24), which exceeded the currently proposed threshold value of 2.00 mg NO_3–N dm^{-3} (8.86 mg NO_3^- dm^{-3}) (KÖHLER 2006, ZETTLER & JUEG 2007).

The novel finding of this survey is that *U. crassus* is more tolerant to a relatively high concentration of nitrates in the water than was previously thought; it is also more tolerant to nitrites and phosphates and may survive in rivers with relatively high nutrient concentrations. However, a dramatic decline in the abundance, density and number of mollusc species was recorded during the last year in the Wkra River.

5.1.4. SELECTED ENVIRONMENTAL FACTORS AFFECTING MOLLUSC COMMUNITIES

5.1.4.1. Mollusc communities in relation to physical and chemical parameters of the water

NIJBOER et al. (2006) showed that some lowland rivers in central Poland might constitute reference conditions for Dutch rivers because of their pristine character. Even within the same Ecoregion, for example Ecoregion 16, Polish lowland rivers show relatively low phosphate and nitrate concentrations compared to the Wkra River catchment area (ZIELIŃSKI et al. 2009). In contrast, a very high concentration of phosphates (up to 11.00 mg dm^{-3}), nitrites (up to 2.64 mg dm^{-3}) or nitrates (up to 142.20 mg dm^{-3}) was recorded at sampling sites in the Wkra River catchment area. According to the survey by the Voivodeship Inspectorate (REPORT 2009, 2011) in 2008–2010, 92.3% of the sampling sites of rivers in the central part of Poland (Mazovian Voivodeship) were prone to eutrophication and showed high level of nutrients in the water. This is a result of the anthropogenically elevated nutrient concentrations from both diffuse and point sources in this region.

It was shown that pollution in the water that originated from an agricultural area (nitrites, nitrates, ammonia, phosphates, phenol) caused genotoxicity (micronucleus and nuclear abnormalities) and lysosomal damage in haemocytes as well as a high level of the glutathione S-transferase (GST), which is the enzyme of phase II detoxifying systems in the gills of *A. cygnea* (FALFUSHYNSKA et al. 2010). The anthropogenic input of nitrogen from point and diffuse sources into the environment results in inorganic nitrogen pollution. In rivers, the most abundant forms of anthropogenic dissolved inorganic nitrogen in freshwater ecosystems are nitrates (NO_3^-), whereas ammonium (NH_4^+) and nitrites (NO_2^-) account for a much smaller fraction of this pool (CAMARGO & ALONSO 2006, SOUCEK & DICKINSON 2012). This survey revealed that the physical and chemical parameters of water, i.e. conductivity, hardness, concentration of dissolved oxygen, nitrites, nitrates, phosphates, and temperature were most associated (statistically significant) with the distribution of mollusc species. Both nitrites and nitrates negatively influenced mollusc communities; however, a more visible negative correlation was observed between the distribution of some gastropod species, unionid mussels, fingernail clams and the nitrite concentration in contrast to the nitrate concentration. The main toxic effects of nitrites on higher taxa of freshwater macroinvertebrates, which include gastropods, unionids, fingernail clams and crayfish, is due to the conversion of the oxygen-carrying blood pigments (haemoglobin and haemocyanin) into forms that cannot carry oxygen, such as methaemoglobin and methaemocyanin, thus causing hypoxia and ultimately death (CAMARGO & ALONSO 2006, SOUCEK & DICKINSON 2012). Branchial permeability to nitrates is low, and therefore nitrate uptake in aquatic animals is more limited than the uptake of nitrites and ammonium. Among freshwater molluscs, *P. antipodarum* showed a high tolerance to the lethal effects of ammonium and nitrate concentrations in the water. This phenomenon may be a consequence of detoxification processes, such as a low branchial Cl^-/NO_2^- uptake rate or a low nitrite affinity for the uptake mechanism that is regarded as a protective mechanism for aquatic macroinvertebrates. Nitrites can be taken up across the gill epithelium and accumulated in body fluids (ALONSO & CAMARGO 2006). The results from a survey of nitrite and nitrate toxicity showed that a nitrite concentration of 0.08–0.35 mg dm^{-3} and a maximum nitrate concentration of 2.00 mg dm^{-3} might be adequate to protect sensitive freshwater species (CAMARGO et al. 2005, CAMARGO & ALONSO 2006). ALONSO & CAMARGO (2009) stated that the nitrite concentration in unpolluted waters ranged from 1.00 to 3.00 μg dm^{-3} and might exceed 0.90 mg dm^{-3} in polluted surface freshwaters. The occurrence of unionid species was significantly reduced in a stretch of the Lužnice River (the Czech Republic) with elevated nitrite-nitrogen concentrations (DOUDA 2010). DOUDA (2010) showed that *P. complanata* occurred at river sites with an upper limit of NO_3–N concentration of up to 2.00 mg dm^{-3} (median value), whereas *U. tumidus*, *U. pictorum* and *A. anatina* had an upper limit of to 4.30 mg dm^{-3} (median values; corresponds to about 19.05 mg NO_3^- dm^{-3}). In contrast, the present survey showed much higher maximum values of the nitrate concentration in the Wkra River and its tributaries. *P. compalanata* was observed at sampling sites with the maximum nitrate concentration of 65.12 mg NO_3^- dm^{-3}, whereas *U. tumidus*, *U. pictorum* and *A. anatina* tolerated the maximum values of 70.44, 87.71 and 68.67 mg NO_3^- dm^{-3}, respectively and the median values from 17.28 to 27.24 mg NO_3^- dm^{-3}. In a field survey, MOUTHON (1996b) obtained the threshold values of nitrite and nitrate concentrations in rivers above which an inhibiting effect on mollusc communities was observed, i.e. 0.30 mg NO_2^- dm^{-3} and 20.00 mg NO_3^- dm^{-3}. The density of molluscs and the number of species increased with nitrate concentration up to 20.00 mg NO_3^- dm^{-3} and decreased above this value, thus indicating that it had an inhibiting effect on mollusc communities. In comparison, the concentration of nitrites and nitrates in the Wkra River catchment area reached 2.64 mg dm^{-3} and 142.20 mg dm^{-3}, respectively. My results confirmed and supported the results of the surveys which indicated the negative effect of nitrites and nitrates on freshwater species that are caused by their toxicity. Thus, the negative correlations between nitrite and nitrate concentrations and the distribution of Mollusca in the Wkra River catchment area may be explained by their physiological make up. However, the concentration of nitrites and nitrates in the Wkra River catchment area was much higher in comparison to those that may be adequate to protect sensitive freshwater species as was shown by CAMARGO et al. (2005) and CAMARGO & ALONSO (2006). A novel finding of this survey is that some freshwater mollusc species can tolerate higher nitrite and nitrate concentrations in rivers than the ranges indicated by toxicological research.

The results of the CCA ordinations also suggested that conductivity and hardness as well as phosphate concentrations were the parameters most associated (statistically significant) with the distribution of *P. corneus* and *P. planorbis*. DILLON (2000) showed that planorbid adults and juveniles could exploit different resources. According to PIECHOCKI (1979) and TSIKHON-LUKANINA et al. (1998), adult specimens of *P. corneus* mainly consume the decaying tissue of vascular plants, rarely ingesting living macrophyte tissue or algae, whereas juvenile individuals consume periphytic algae. *P. planorbis* feeds on periphitic algae mainly. In streamwater, phosphorus occurs as orthophosphate (PO_4^{3-}) dissolved in water and attached to inorganic particles in suspension, as well as dissolved organic molecules and in particulate organic form mainly in bacteria and detrital particles (ALLAN & CASTILLO 2007). Algae directly assimilate orthophosphates, whereas other forms of phosphorus must be transformed into orthophosphates by phosphomonoesterase before its assimilation (KAWECKA & ELORANTA 1994). Not only orthophosphate or the total phosphorus concentration in the water, but also conductivity that reflected watershed processes and nutrient enrichment in agricultural areas, were shown to be the most important environmental factors having an influence on the periphitic algal assemblages in rivers (HWANG et al. 2011, RUSANOV et al. 2012). Therefore, the increase in the densities of *P. corneus* and *P. planorbis* with an increasing concentration of orthophosphates may be explained by their feeding preferences. A higer concentration of orthophosphate in the water has an influence on the increase in algae abundance, which in turn provides food to these planorbids.

According to MOUTHON (1996a), a few mollusc species, e.g. *V. piscinalis*, *V. viviparus* and *S. corneum*, show an affinity to phosphate concentrations of up to 4.00 mg dm^{-3} or nitrites up to 1.50 mg dm^{-3}. However, MOUTHON (1996b) also obtained a threshold value of phosphate concentration in the water above which an inhibiting effect on molluscs was observed, i.e. 0.30 mg PO_4^{3-} dm^{-3}. At values above 0.30 mg, phosphates and nitrates favoured proliferation of algae that are liable to have a highly limiting effect on mollusc populations. In contrast, my survey showed a decreasing density of *V. piscinalis* and *B. tentaculata* at values exceeding 1.10 mg PO_4^{3-} dm^{-3} and only two species showed an affinity to a higher phosphate concentration: *P. corneus* and *P. planorbis*. In the Wkra River catchment area, *V. viviparus* was associated with sites with lower nitrite concentrations, whereas *S. corneum* was more influenced by the substratum than by the physical or chemical properties of the water. It was shown that an increase in the concentration of nitrates, nitrites and phosphates in the water could reduce the period of activity of *U. tumidus* by 82 to 28% (WEBER 2005). The following threshold values were shown: nitrates up to about 12.41 mg NO_3^- dm^{-3}, nitrites up to about 0.46 mg NO_2^- dm^{-3} and phosphates up to about 0.16 mg PO_4–P dm^{-3}. In contrast, *U. tumidus* was recorded in sampling sites of the Wkra River and its tributaries with much higher nutrient concentrations, of up to 70.44 mg NO_3^- dm^{-3}, 0.50 mg NO_2^- dm^{-3} and 3.01 mg PO_4^{3-} dm^{-3}. However, *U. tumidus* was simultaneously a subdominant and an accessory species in mollusc communities in the Wkra River (lower nutrient concentrations than in the tributaries), and was absent from or only simultaneously subrecedent and accedent in the tributaries (higher nutrient concentration than in the mainstream).

This survey found that the density of molluscs increased to 649 indiv. m^{-2}, whereas the number of mollusc species decreased (from 21 to 11) with a conductivity increase above 820 μS cm^{-1}. Thus, the values of indices *H*' and H_{max} (which include the density and the number of species) were highest at lower values of conductivity, i.e. up to 520 μS cm^{-1}. This result is consistent with those of MOUTHON (1996b) who showed that the preferable values of conductivity for molluscs ranged from 300 to 600 μS cm^{-1}, and also with SHIEH et al. (1999) who obtained a positive correlation between conductivity and density, and a negative one between conductivity and the number of mollusc species. For example, the density of *V. piscinalis* amounted to 59 indiv. m^{-2} in the Wkra River catchment area and was dependent on the conductivity, whereas in the Saône River, its density ranged from 5.8 to 872.7 indiv. m^{-2} and was dependent on an increase in annual water temperature or the availability of resources (competition) (MOUTHON & DAUFRESNE 2008). This survey supports the results of PIRES et al. (2000) who found that Planorbidae occurred at higher values of conductivity because in the Wkra River catchment area *P. corneus* and *P. planorbis* were recorded at sites which not only had a higher phosphate concentration, but also higher conductivity. Conductivity was shown to be an indicator of water quality, especially nutrient enrichment; a positive correlation between conductivity and nutrients was recorded in relation to agricultural areas (MARET et al. 2010). It was demonstrated (HERBST et al. 2008) that a significant reduction in the survival and growth of gastropods was observed in calcium-free river waters with conductivity below 300 μS cm^{-1}. In such waters, insufficient dissolved mineral content for growth and shell-building impedes their development. In the rivers of an agricultural area in Poland, BIS et al. (2000) observed a higher density of gastropods, including *B. naticina*, *R. balthica*, *A. spirorbis* or *G. albus*, at sites with a high nutrient concentration. However, the concentration of nitrates at these sites was incomparably lower than those recorded at the sampling sites of the Wkra River catchment area. They also showed a higher density of gastropods at higher values of conductivity, i.e. from 304 to 438 μS cm^{-1}.

The results of the CCA ordinations showed that *B. tentaculata* and *V. piscinalis* occurred at the sites with a higher dissolved oxygen concentration in the water. According to BOYCOTT (1936), *B. tentaculata* and *V. piscinalis*, which are gill breathers and are unable to come to the water's surface to gulp in air like pulmonates, require well-oxygenated water. Thus, the clear relationship between the distribution of these prosobranch species and dissolved oxygen may be explained by their physiological make up. The concentration of dissolved oxygen (besides substratum particle size) was the parameter that also affected the distribution of some unionid species most, including *U. pictorum* and *U. crassus* in the Danube River and its tributaries (BÓDIS et al. 2011).

Freshwater molluscs are considered to be good indicators of the quality of running waters because they are sensitive to biodegradable pollution, acidification, concentrations of nitrites and nitrates, and alkalinity levels (Dillon & Benfield 1982, Clarke & Scruton 1997, Mouthon & Charvet 1999, Soucek & Dickinson 2012). Thus, the decrease in populations of molluscs may be the result of the large-scale deterioration in the health of freshwater ecosystems (Baur & Ringeis 2002). In the Wkra River catchment area, molluscs can be biological indicators of nutrient enrichment, especially the nitrite concentration, conductivity and dissolved oxygen in the water. The number of mollusc species decreased with an increase in conductivity, whereas the density of *P. corneus* and *P. planorbis* increased when the conductivity and concentration of phosphates increased. *B. tentaculata* and *V. piscinalis* may be indicators of a higher concentration of dissolved oxygen.

5.1.4.2. Mollusc communities in relation to velocity, river width and substratum

Abiotic factors including current velocity, substratum or water temperature are often the most important variables that influence the distribution and abundance of organisms in fluvial environments (Allan & Castillo 2007). In the lowland rivers of agricultural areas, the most important environmental variables that affect the structure and trophic organisation of stream communities including molluscs are the riparian canopy cover, stream hydraulic processes, morphometry of rivers and nutrient inflow (Bis et al. 2000, Parr & Mason 2003, Probst et al. 2005, Virbickas et al. 2011). Ecological studies of freshwater ecosystems within the landscape framework of agricultural areas have important management implications. In the lowland rivers of agricultural areas, large-scale properties, e.g. nutrient input, riparian cover and hydraulics, are strong predictors of the functional organisation of macroinvertebrates including molluscs (Bis et al. 2000).

The results of the CCA ordinations revealed that among the environmental factors, velocity, river width and temperature were positively correlated with the distribution of gastropod species, unionid mussels and fingernail clams. This finding is consistent with the surveys of Pires et al. (2000) and Probst et al. (2005) who consider these factors to be the most important. In rivers within agricultural areas, the number of species increased with an increase in stream width. The significance of this variable may be explained by the tendency of larger areas of stream channel to provide more diverse instream habitats and niches; therefore, wider rivers support a higher number of species. A higher number of mollusc species was recorded in the Wkra River, the width of which ranges from 0.5 to 60.0 m, than in its tributaries (width 0.8–15.0 m). According to Pérez-Quintero (2007, 2011) and Maloney et al. (2012), mussels, especially Unionidae, are sensitive to variation in hydrological conditions (lower-higher flows, velocity, river depth, bed stability) and prefer the permanent lowland watercourses of rivers. Thus, the higher number of unionid mussel species in the Wkra River may be explained by its less variable conditions compared to its tributaries.

The present survey revealed that in addition to the physical and chemical parameters of the water, the current velocity also influenced the distribution of mollusc species, especially Unionidae: *U. tumidus*, *U. pictorum*, *A. cygnea* and *U. crassus*. This finding is consistent with Weber (2005) who explained this phenomenon as a relationship between the velocity and the substratum. Unionids, especially juveniles, which inhabit the upper stratum of the sediments, are dependent on the interstitium conditions. A higher current velocity ensures the better ventilation of the sediments, prevents the accumulation of pollution and reduces the sedimentation of fine-grained material, which may clog the interstitium. What is more, when the current velocity increases, the influence of dissolved oxygen and the concentration of ions on mussel populations is less visible. Flow velocity influences the burrowing behaviour of unionids, i.e. an increase in the flow velocity causes mussels to burrow into the sediments (Jonsson et al. 2013).

This research found that some particle size fractions of the bottom sediments and the occurrence of macrophytes were also predictive environmental variables of mollusc distribution, e.g. *R. balthica*, *A. lacustris*, *S. corneum*, *S. palustris* and *A. vortex*. Gastropod species may either feed on vascular plants (macrophytes) directly or may graze on the microorganisms that colonise the macrophytes (macrophyte algae) (Dillon & Benfield 1982, Chertoprud & Udalov 1996). It was shown that pulmonate gastropods could utilise the dissolved organic matter (DOM) from living macrophytes (Thomas & Kowalczyk 1997). On the other hand, the growth of macrophytes, e.g. *Ceratophyllum*, may be enhanced by the presence of pulmonate gastropods. Pulmonate gastropods, macrophytes and the associated epiphytic algae and bacteria should be regarded as four components of mutualistic relationships (Thomas & Kowalczyk 1997). Bivalves mostly live on the bottom and filter phytoplankton, bacteria and particulate organic matter from the water column, though *S. corneum* or some *Pisidium* spp. often climb up macrophytes which aids their filter-feeding. Therefore, the association between pulmonate species, *S. corneum* and macrophytes in the Wkra River catchment area may be explained by their feeding mode and the filtration process. This result is consistent with the survey done by Bódis et al. (2011) who, based on a

CCA analysis, showed a positive correlation between the distribution of *S. corneum* or other fingernail clams and the macrophyte occurrence.

The distribution of molluscs is highly correlated with the substratum because they utilise the substratum as a surface to crawl upon, burrow into and as a food source (organic matter in the substratum and algae growing on it). Therefore, *A. cygnea*, *U. crassus*, *T. fluviatilis*, *R. auricularia* for example, prefer fine sediments and fine gravels (Økland 1983, Dillon 2000, Minton et al. 2008). Crowl & Schnell (1990) claim that gastropods are associated with a larger substratum particle size in riffles. The abundance of Mollusca increases with an increase in substratum particle size up to cobble size (diameter of 64–256 mm) and decreases as the substratum achieves boulder or rock size (Beisel et al. 1998). This survey supports the previous research of Crowl & Schnell (1990) who found that the mollusc density was positively correlated with substratum particle size (density of both gastropods and bivalves). McRae et al. (2004) found significant relationships between fine gravel and some unionid species. Bódis et al. (2011) also highlighted the importance of substratum particle size in the distribution of bivalve species. She showed that *P. casertanum*, *P. nitidum* or *P. complanata* were associated with a coarse fraction of sediments, whereas *P. supinum* was associated with sand in very large and large rivers (the Danube River and its tributaries). *Pisidium* species show preferences in relation to the substratum. Certain species are associated with coarser substrata (above 8 mm) because large-pored interstitial spaces enable them to take up oxygen-rich water but also prevent them from sinking during crawling on the surface of the substratum (Meier-Brook 1969). Substratum preferences by *Pisidium* species in small and medium-sized rivers (Ecoregion 14) were observed by Piechocki (1987), and Piechocki & Strzelec (1999). *P. subtruncatum*, *P. casertanum* or *P. supinum* predominated in small and medium-sized rivers with a very fine loess fraction of sediments (which corresponds to a particle size of 0.02–0.05 mm), whereas some species were associated with very coarse sand (particle size 0.50–1.0 mm) as well as medium-grained sand (particle size 0.25–0.50 mm). *P. supinum* was associated with medium-grained sand (particle size 0.25–0.50 mm), whereas *P. subtruncatum* avoided this sediment fraction in the Wkra River and its tributaries.

Vaughn & Hakenkamp (2001) found that the filtration rate of bivalves varied and depended on the bivalve species and its size, particle size and concentration, flow regime or water temperature of a river. The filtration rate increases with temperature because the bivalve metabolic rate increases. A possible explanation of the significant correlation between river width, temperature, particle size or velocity and mussel distribution in the Wkra River catchment may be that these environmental factors jointly promote favourable conditions for the filtration process.

It is evident that the mollusc distribution is rarely controlled by a single environmental variable, but results from several variables acting together (Kerney 1999). In rivers, it may be the result of the interaction between alkalinity and the stream drainage area (Dillon & Benfield 1982), algal biomass, flow rate, substrate particle sizes, depth (Crowl & Schnell 1990), hydrological conditions, or bed stability (Maloney et al. 2012). Distribution may also be a function of food supply (Dillon & Benfield 1982), conductivity, permanence, river width, turbidity (Pérez-Quintero 2011), environmental calcium concentration (Briers 2003) or nutrient enrichment and periphyton abundance (Harding et al. 1999). In this context, the most predictive environmental factors in the occurrence of molluscs in the Wkra River catchment were a few of the physical and chemical parameters of the water, selected abiotic factors including river width, velocity, particle size of bottom sediments as well as the occurrence of macrophytes.

5.2. OXBOW LAKES OF THE WKRA RIVER

5.2.1. CONCENTRATION OF NUTRIENTS AND WATER CONDUCTIVITY IN OXBOW LAKES

Small water bodies and watercourses can create important biogeochemical barriers which effectively restrict the free migration of minerals and organic substances. It was shown (Joniak et al. 2006) that the conductivity of water in small mid-field water bodies located in agricultural areas with different levels of anthropogenic transformation amounted to 1,318 μS cm^{-1}, and that the concentration of phosphates and nitrates reached 0.14 mg dm^{-3} and 2.35 mg dm^{-3}, respectively (Ecoregion 14). Very similar values of conductivity (up to 1,294 μS cm^{-1}), concentration of phosphates (up to 0.95 mg dm^{-3}) and nitrates (up to 2.48 mg dm^{-3}) were found by Pasztaleniec & Poniewozik (2013) in the lentic oxbow lakes of the Bug River which had no permanent connection to the main river channel (Ecoregion 16). The conductivity of the water was lower but the concentrations of phosphates and nitrites were much higher, of up to 4.64 mg dm^{-3} and 103.22 mg dm^{-3}, respectively, in the oxbow lakes of the Wkra River, which are located in an agricultural area of Ecoregion 16.

The high values of conductivity as well as the concentration of chlorides and nitrates in the water of small oxbow lakes (especially in summer), where 30% of the catchment area consists of arable fields, indicates that the reservoirs must have sources of water other than the main river channel which contains these ions, i.e. the Bug River (DAWIDEK & TURCZYŃSKI 2006). The agricultural activities in the fields surrounding the oxbow lake could account for the increase of chloride and nitrate ions in the groundwater which supplies the reservoir during the isolation phase. Thus, following this pattern, the agricultural activities in the fields, pastures or meadows surrounding the lentic oxbow lakes of the Wkra River could account for the considerable concentrations of nitrites, nitrates and phosphates in the water as well as for its conductivity.

Lower concentrations of nitrates and a higher concentration of phosphates were obtained in the lentic oxbow lakes of both the Słupia River (Ecoregion 14) and in the Łyna River (Ecoregion 16) than in the main river channel (GLIŃSKA-LEWCZUK 2009, OBOLEWSKI & GLIŃSKA-LEWCZUK 2011). This survey showed lower maximum concentrations of nitrates and phosphates in the water of most of the oxbow lakes of the Wkra River than in the main river channel. i.e. the Wkra River. The relatively higher concentration of phosphates and concentrations of other nutrients in the Wkra River in comparison to it lentic oxbow lakes may be explained by their extremely high concentration in the tributaries, i.e. the Raciążnica River (concentration of phosphates up to 11.00 mg PO_4^{3-} dm^{-3}, nitrites up to 2.64 mg NO_2^- dm^{-3}) and the Sona River (concentration of nitrates up to 142.20 mg NO_3^- dm^{-3}). However, the concentrations of nitrites, nitrates and phosphates in the oxbow lakes of the Wkra River were higher than those obtained in the lentic oxbow lakes of both the Słupia River and the Łyna River. The concentration of nutrients was also lower in other oxbow lakes of very large rivers in Ecoregion 16, for example in the oxbow lakes of the Bug River (URBAN & WÓJCIAK 2004, PASZTALENIEC et al. 2006), the San River (MICHALSKA-HEJDUK et al. 2009) or in Ecoregion 14, e.g. in the oxbow lakes of the Warta River (PENCZAK et al. 2004) and the Odra River (SPAŁEK 2008a). What is more, no such extremely high concentrations of nutrients as those recorded in the oxbow lakes of the Wkra River or in the Wkra River and its tributaries have been recorded to date (DUSSART 1976, STRZELEC 1993, WILLIAMS et al. 2003, PENCZAK et al. 2004, GALLARDO et al. 2008, 2012, SUREN et al. 2008, KOC et al. 2009, JURKIEWICZ-KARNKOWSKA 2011, CHALUPOVÁ et al. 2012, RISTAU et al. 2012, JIANG et al. 2013, PASZTALENIEC & PONIEWOZIK 2013). However, HASSALL et al. (2011) showed a maximum concentration of orthophosphates of up to 5.87 mg dm^{-3} in lentic water bodies – ponds (United Kingdom).

Small oxbow lakes are especially sensitive to the local disturbances that are caused by human activity, primarily water management, land use changes and agriculture (GALBARCZYK-GĄSIOROWSKA et al. 2009). Oxbow lakes, located within agricultural areas, regulate the nutrient transfer towards rivers, mainly through the retention of matter. Thus, oxbow lakes prevent the rivers from a decline in water quality. It was shown that in lentic oxbow lakes most of the substances were withdrawn from the system and had a tendency to accumulate in the bottom sediments; they were also intensively taken up by aquatic organisms. The effectiveness of the withdrawal of the total inorganic nitrogen (including nitrites, nitrates) from lentic oxbow lakes is estimated as 75%, HCO_3^- as 50%, SO_4^- by 82%, K^+ as 80% and Na^+ as 72% compared to lotic oxbow lakes (GLIŃSKA-LEWCZUK 2009). The concentration of nutrients depends on their input from the river catchment area and the nutrients that are cycling in oxbow lakes. Macrophytes take up nutrients from autochthonous or allochthonous sources that originate from the river catchment area. According to SZMEJA (2006), macrophytes, especially Lemnaceae that cover the surface of reservoirs, accumulate substantianal amounts of nutrients. A thick layer of Lemnaceae species covered the water surface in the oxbow lakes of the Wkra River. Thus, the lower concentration of nutrients in the oxbow lakes compared to the Wkra River and its tributaries may be partially explained by their uptake by macrophytes. The primary role of mussels, especially Unionidae, is to increase the retention of phosphorus in reservoirs by removing particles from the water column, and, on the other hand, depositing particles on the bottom. The phosphorus, which has accumulated in the body of mussels, is withdrawn from the cycling of nutrients during a certain period (lifetime of mussels) (STAŃCZYKOWSKA 1983, JURKIEWICZ-KARNKOWSKA 2002). The maximum density of Unionidae, e.g. *U. pictorum* and *A. cygnea*, was 8 and 11 indiv. m^{-2}, respectively, in the oxbow lakes of the Wkra River. These species may play a significant role in phosphorus cycling in the oxbow lakes. However, the uptake of phosphates by mussels constitutes a smaller fraction of the total uptake (NALEPA et al. 1991).

According to DODDS (2002), a typical way to control non-point nutrient input, besides reducing fertiliser applications or the proper timing of their application, is keeping livestock out of streams and ponds with fences and by providing stock tanks. During grazing, livestock have unrestricted access to the oxbow lakes of the Wkra River, and in some cases to the main river channel, for drinking water. These results showed that the concentration of nitrates in the water in two out of seven oxbow lakes of the Wkra River was much higher than the threshold value according to the Nitrates Directive (i.e. 50 mg NO_3^- dm^{-3}) (COUNCIL DIRECTIVE 91/676/EEC). ZABLOTOWICZ et al. (2010) obtained significant differences in the concentration of nutrients in the water of oxbow lakes in different agricultural areas which were dependent on the implementation the Rules of Good Agricultural Practice. When the Rules of Good Agricultural Practice were applied in the watershed of oxbow lakes, the concentrations of nitrate nitrogen and phosphates significantly decreased to 0.94 mg and 0.99 mg, respectively.

5.2.2. STRUCTURE OF MOLLUSC COMMUNITIES AND VALUES OF INDICES IN OXBOW LAKES

In total, 30 mollusc species were recorded in the oxbow lakes of the Wkra River (Ecoregion 16, Eastern Plains): 19 gastropod species and 11 bivalve species. The mollusc density ranged from 20 to 157 indiv. m^{-2} (mean: 60.1 ± 34.6). For comparison, within the same Ecoregion 16, only 12 mollusc taxa: eight gastropods and three bivalves, were recorded in the oxbow lakes of the Łyna River whose length and catchment area are similar to the Wkra River (large river according to Directive 2000/60/EC, catchment area of 1,000–10,000 km^2) (Obolewski et al. 2009). However, the mollusc density was higher than that in the oxbow lakes of the Wkra River and amounted to 358 indiv. m^{-2}. In the oxbow lakes of the Vistula River (very large river, catchment area > 10,000 km^2), Stańczykowska (1960a) recorded 19 mollusc species: 12 gastropod and seven bivalve species (Konfederatka oxbow lake) with a density of 70 to 3,000 indiv. m^{-2}. However, bivalves within the genus *Pisidium* were not identified to the species level and therefore they were excluded from the analysis. Kołodziejczyk (2000) and Kołodziejczyk & Dołęga (2004) found 23 taxa: 18 gastropod and five bivalve species in another oxbow lake (individuals within subgenus *Radix*, *Galba* and family Sphaeriidae were not identified to the species level).

This survey showed the occurrence of 30 mollusc species in seven of the oxbow lakes situated within a large river catchment area, while 43 mollusc species were recorded in the 21 oxbow lakes situated within a very large river catchment area (the Bug River) (Jurkiewicz-Karnkowska 2006). Jurkiewicz-Karnkowska (2009) showed that the maximum density of molluscs was 620 indiv. m^{-2} and recorded 51 species: 33 gastropods and 18 bivalves in permanent floodplain water bodies. However, the molluscs were sampled from 121 floodplain water bodies within a large sector (over 100 km) of the Bug River valley (very large river according to its catchment area).

This survey showed that the density of same fingernail clam species, i.e. *P. henslowanum* and *P. subtruncatum*, was up to 5 indiv. m^{-2}; whereas the density of unionid mussel species was up to 11 indiv. m^{-2}. Jurkiewicz-Karnkowska (2006) obtained a similar density of the same fingernail clam species as well as unionid mussel species in the oxbow lakes of a very large river. A much higher density of unionid mussels than those in the oxbow lakes of the Wkra River was recorded by Abraszewska-Kowalczyk (2002) in the oxbow lakes of a large river (Ecoregion 14) or by Lewandowski (2006) in the oxbow lakes of a very large river (Ecoregion 16) (up to 200 indiv. m^{-2}).

The differences in the average density of molluscs between the oxbow lakes of the Wkra River were statistically significant. This result is consistent with the survey of Gallardo et al. (2008) who obtained statistically significant differences in the average density of molluscs between oxbow lakes.

Within Ecoregion 14 (Central Plains), the number of mollusc species was 37 in the oxbow lakes of the Grabia River (medium river, catchment area of ca. 1,000 km^2) (Piechocki 1969) and only six mollusc species were recorded in an oxbow lake of the Słupia River (large river, catchment area of 1,000–10,000 km^2). This survey showed a similar total number of gastropod species, i.e. 19 species, as that recorded in the oxbow lakes of medium and large rivers (e.g. Przemsza River, Biała Przemsza River, Kłodnica River, Brynica River) in the southern part of Ecoregion 14 (Strzelec 1993).

In contrast to oxbow lakes of very large rivers, i.e. the Danube or the Bug River, the Vistula River of Ecoregion 16 (Kołodziejczyk & Dołęga 2004, Jurkiewicz-Karnkowska 2006, 2009, Reckendorfer et al. 2006) or the large and medium rivers of Ecoregion 14 (Strzelec 1993, Obolewski et al. 2009), i.e. Słupia, Przemsza, Biała Przemsza, Kłodnica, Brynica, no alien mollusc species were recorded in the oxbow lakes of the Wkra River.

The total number of mollusc species recorded in the oxbow lakes of the Wkra River (30 species) comprised about 34% of the Polish species list (88 species) (Piechocki 2008). Both the number of mollusc species and their density varied among the oxbow lakes of the Wkra River, which may be dependent on various environmental factors: lack of connectivity with the main channel or physical and chemical parameters of the water or substratum.

Up to 21 mollusc species and a value of the Shannon-Wiener index (H') of up to 4.01 were recorded from individual sampling sites in the lentic (isolated) oxbow lakes of the Wkra River. Jurkiewicz-Karnkowska (2006) found up to 12 mollusc species and a value of the H' index of up to 3.20 in the lentic (isolated) oxbow lakes of the Bug River as well as 15 species and a value of index of up to 3.50 in oxbow lakes that were connected with the main river channel. Values of the H' index below 1 were obtained for mollusc communities of shallow water bodies with a thick layer of dark muddy sediments with a coarse detritus admixture which were located on a former floodplain, far from the embankment. In contrast, Obolewski et al. (2009) recorded up to five gastropod species, no bivalve species and a value of the *H*' index of up to 0.50 in lentic (isolated) oxbow lakes. According to Obolewski et al. (2009), such low values of the *H*' index may be explained by advanced succession processes and terrestrialisation, which leads to a decline in these habitats.

These results showed that the number of mollusc species and the value of the *H*' index recorded in the isolated oxbow lakes of the Wkra River (large river according to catchment area) are comparable to those obtained in the lentic oxbow lakes of the Bug River (very large river) by Jakubik et al. (2006).

Species diversity indicates the status of the ecosystem or community. A high species diversity contributes to the stability of the ecosystem (Głowaciński 1996). Higher values of *H*' indicate a quite high biodiversity in some of the oxbow lakes of the Wkra River despite the considerable anthropogenic pressure resulting from agricultural practices.

5.2.3. RARE, THREATENED AND PROTECTED SPECIES IN THE MOLLUSC COMMUNITIES OF OXBOW LAKES

Six of the mollusc species which were recorded in the oxbow lakes of the Wkra River are included in the Red List of Threatened Animals in Poland: *S. palustris*, *S. corvus*, *H. complanatus*, *A. cygnea*, *S. rivicola* and *M. lacustre* (Dyduch-Falniowska & Zając 2002, Piechocki 2002). Two of them, *A. cygnea* and *S. rivicola*, are protected by the Polish law (Dz.U. 2011a). Both *S. palustris* and *S. corvus,* whose density ranged from 1 to 2 indiv. m^{-2} and from 5 to 6 indiv. m^{-2}, respectively, were accedents ($C \leq 25.0\%$) in the mollusc communities in the oxbow lakes of the Wkra River. *S. palustris* and *S. corvus*, whose density was up to 10 indiv. m^{-2}, were also accedents in the mollusc communities in the permanent water bodies of the Bug River, whereas their density was up to 13 and 66 indiv. m^{-2}, respectively, in the oxbow lakes in the southern part of Ecoregion 14 (Strzelec 1993, Jurkiewicz-Karnkowska 2006, 2009).

The drainage of wetlands and riverside meadows and agricultural drainage constitute a threat to various species including *H. complanatus* (rare species in Poland) and *M. lacustre* (vulnerable species, VU), which are characteristic of small closed ponds or swampy ditches (Kerney 1999, Piechocki 2008). According to Killeen et al. (2004), the lake orb mussel, *M. lacustre,* is the most common species in swamps, ponds, marsh drains as well as in the well vegetated margins of rivers and canals. The lake orb mussel was recedent in mollusc communities in oxbow lakes Ox4 and Ox6 of the Wkra River. *H. complanatus*, which was subdominant in the mollusc communities, occurred only in one out of seven oxbow lakes of the Wkra River. For comparison, *M. lacustre* and *H. complanatus* were subrecedents in mollusc communities in water bodies of the lower Bug River floodplain (Jurkiewicz-Karnkowska & Karnkowski 2013).

According to the European Red List of Nonmarine Molluscs (Cuttelod et al. 2011), 75 species (8.8% of European mollusc species) are classified as Near Threatened (NT), including *A. cygnea.* In Poland, in addition to the protection provided by law, *A. cygnea* is listed in both the Red Book and the Red List of Threatened Animals as Endangered (EN) (Dyduch-FalniowskA & Zając 2002, Zając 2004c). *A. cygnea,* which is a Palaearctic species, occurs primarily in shallow eutrophic water bodies, oxbow lakes and lakes, slowly flowing rivers, artificial ponds, canals and dam reservoirs in the lowland part of the country. According to Abraszewska-Kowalczyk (2002), *A. cygnea* is a typical species of floodplain water bodies. In Poland, *A. cygnea* is a rare species that is characteristic mainly of stagnant waters, and a decline in its population has been observed since the 1950s. The major threats to *A. cygnea* are habitat degradation which results from the destruction of small shallow reservoirs including oxbow lakes, and water pollution as well as the decline in the quality of habitats in rivers and lakes (Zając 2004c, Piechocki 2008). This survey showed that the density of *A. cygnea*, which was subdominant in the mollusc communities, was up to 11 indiv. m^{-2}. The density of *A. cygnea* amounted to 10 indiv. m^{-2} in the permanent water bodies of the Bug River (very large river, Ecoregion 16) which is characterised by a low nutrient concentration and higher concentration of calcium and dissolved oxygen, or 16 indiv. m^{-2} in the oxbow lakes of the Pilica River (large river, Ecoregion 14) (Abraszewska-Kowalczyk 2002, Jurkiewicz-Karnkowska 2006, 2009, 2011). *A. cygnea* was eudominant in the mollusc community in an isolated oxbow lake of the Vistula River (Jeziorko Czerniakowskie, landscape nature reserve, Ecoregion 14) (Kołodziejczyk 2000, Kołodziejczyk & Dołęga 2004). In contrast, the density of *A. cygnea* was up to 65 indiv. m^{-2} in a young artificial pond on a peat excavation site, surrounded by extensively utilised meadows (Ożgo & Abraszewska 2009).

5.2.4. SELECTED ENVIRONMENTAL FACTORS AFFECTING OXBOW LAKE MOLLUSC COMMUNITIES

The physical and chemical parameters of the water including pH, concentration of calcium, total phosphorus, conductivity, alkalinity or hardness have usually been cited as major factors that influence the structure of mollusc communities in natural lentic habitats (Dussart 1976, Bendell & McNicol 1993, Briers 2003, Heino 2000, Hassall et al. 2011). The distribution of freshwater molluscs in these habitats not only depends on the physical and chemical parameters of the water, but is also a function of abiotic and biotic factors, e.g. substratum, organic matter content and types of bottom sediments, habitat and food selection, interspecific competition, predation, pollution and surface area of the water body, all of which act together (Kołodziejczyk 1984, Lodge et al. 1987, Savage & Gazey 1987, Chertoprud & Udalov 1996, Carlsson 2001, Oertli et al. 2002). Different studies also stressed the importance of food (Brönmark 1989, Kornijów 1996, Kornijów & Ścibor 1999a, Peeters et al. 2004).

Among the many environmental variables which were considered in canonical correspondence analysis (CCA), only nitrates, nitrates, dissolved oxygen, water temperature and three different particle size classes of the substratum were the parameters most associated (statistically significant according to the forward selection results) with the distribution of mollusc species in the oxbow lakes of the Wkra River. Some authors pointed out that the main environmental factors that acccounted for most of the variability in the mollusc community structure in floodplain water bodies including oxbow lakes were the physical and chemical parameters of the water, i.e. concentration of dissolved oxygen, chlorides, nutrients, permanence of the water body, its size and depth, types of bottom sediments, insolation and hydrological connectivity or even complexity of woody debris (Abraszewska-Kowalczyk 2002, Schneider & Winemiller 2008, Jurkiewicz-Karnkowska 2009, Obolewski 2011a, Funk et al. 2013). Suren et al. (2008) found that gastropods and bivalves (only Sphaeriidae) were restricted to wetlands (including ponds) with a higher pH and conductivity combined with a lower nitrite-nitrogen concentration (mean concentration of up to 3.50 μg N-NO_3 dm^{-3} in the water). This concentration of nitrate-nitrogen is incomparably lower than that recorded in the oxbow lakes of the Wkra River.

This survey revealed that nitrites, nitrates, dissolved oxygen and water temperature were the parameters most associated (statistically significant) with the distribution of mollusc species, e.g. *V. piscinalis*, *A. lacustris*, *Ph. fontinalis*, *P. planorbis*, *G. albus*, *G. crista*, *S. nitida* or *S. corneum* in the oxbow lakes of the Wkra River. The concentration of both nitrites (up to 3.30 mg NO_2^- dm^{-3}) and nitrates (up to 103.22 mg NO_3^- dm^{-3}) negatively influenced the mollusc communities in these water bodies. Although the concentration of nitrites and nitrates was relatively high in the Wkra River and its tributaries as well as its oxbow lakes, the total number of mollusc species was negatively correlated with both nitrites and nitrates of the latter water bodies. For example, Dussart (1979) obtained a negative correlation between the density of *B. tentaculata* and the nitrate concentration and a positive one with the density of *G. albus*. In contrast to the oxbow lakes of the Wkra River, the mean concentration of nitrates in the waters surveyed by Dussart (1979) amounted to only 5.70 mg NO_3^- dm^{-3}. The permanence, depth, flow and altitude were the main environmental variables that explained the composition of the macroinvertebrate communities, including molluscs, in the water bodies of an agricultural area in southern England, even when the concentration of nitrates was up to 38.30 mg NO_3^- dm^{-3} in ponds or up to 19.80 mg NO_3^- dm^{-3} in rivers and streams (Williams et al. 2003). The main toxic effects of nitrites and nitrates on macroinvertebrates including gastropods, unionids and fingernail clams was discussed earlier in relation to running waters, i.e. the Wkra River and its tributaries; therefore, similar toxic effects of these parameters may be expected in relation to the mollusc communities of the oxbow lakes of the Wkra River. In the oxbow lakes of the Wkra River, the concentration of nitrites and nitrates as well as its threshold values, above which an inhibiting effect on mollusc communities is seen, were higher than the threshold values given by Mouthon (1996b) for rivers, i.e. 0.30 mg NO_2^- dm^{-3} and 20.00 mg NO_3^- dm^{-3}. This novel finding is in contrast to Mouthon's (1996b) results, because in the oxbow lakes of the Wkra River the threshold values above which the number of mollusc species decreased amounted to 0.43 mg NO_2^- dm^{-3} and 38.22 mg NO_3^- dm^{-3}. These negative correlations between nitrite and nitrate concentrations and the distribution of Mollusca in the oxbow lakes of the Wkra River may be explained by a similar pattern of toxic effects as in the Wkra River and its tributaries. Gallardo et al. (2008) showed that a threshold nitrate concentration of 40.00 mg NO_3^- dm^{-3} caused apparent changes in composition, density and diversity of macroinvertebrates.

This survey revealed a negative correlation between the density of *P. amnicum* and the water temperature or the diameter of sand particles (medium: 0.25–0.50 mm), whereas there was a positive correlation with medium and coarse gravel (10.0–20.0 mm). *P. amnicum* (the river pea mussel) is a lowland species typical of clean, moderately hard, running water in rivers; it occasionally inhabits lakes (Dyduch-Falniowska 1982, Kerney 1999, Killeen et al. 2004). Some studies reported that *P. amnicum* preferred sites with finer sediments and high organic matter content (2.8–16.1%) in rivers, but that it also occurred on coarser sediments and at a lower organic matter content (0.6–3.0%). In contrast to rivers, the river pea mussel was recorded on coarser bottom sediment (i.e. medium and coarse gravel) and at a medium to high (1.36–8.89%) organic matter content in the bottom sediment in the oxbow lakes of the Wkra River. On the other hand, ALLAN & CASTILLO (2007) stated that sand was generally considered to be unsuitable for macroinvertebrates due to its instability and the tight packing of sand grains which reduced the trapping of detritus and could limit the availability of oxygen. This phenomenon may explain the negative correlation between *P. amnicum* and medium sand and the positive one with medium and coarse gravel in the oxbow lakes of the Wkra River.

Environmental variables, i.e. water temperature or dissolved oxygen, depend on each other and it is impossible to consider them separately. Water temperature has an influence on metabolism, growth, feeding rates, fecundity and survival behaviour of macroinvertebrates. In lentic oxbow lakes, the water temperature and the concentration of dissolved oxygen vary to a great degree depending on the season and fluctuations in the main river water table. Therefore, the highest concentration of dissolved oxygen is observed during spring, just after snow-melting flooding and in periods of high hydrological intensity water cycling, whereas the lowest values are observed in summer (KALINOWSKI et al. 2011). Thus, in this survey, the negative correlation between the density of *P. amnicum* and the water temperature or the diameter of the particles (medium-grained sand) and the positive one with medium and coarse gravel may be explained by the physiological make up of this species.

Benthic macroinvertebrates including molluscs are dependent on the substratum that they crawl on or to burrow into. Hard inorganic substrata, such as stones or pebbles, are usually covered by algae and therefore many species can feed on their surface. An increasing diversity in the substratum is correlated with an increasing mollusc diversity (ØKLAND 1990). Because gastropods are more mobile, they can fix themselves anywhere with their adhesive feet in contrast to bivalves which are more dependent on the substratum than gastropods (BOYCOTT 1936). The highest density of Mollusca, of up to 1,144 indiv. m^{-2}, was recorded on the muddy bottom in contrast to the sandy-muddy and sandy bottom in the Konfederatka oxbow lake of the Vistula River (Ecoregion 16) (STAŃCZYKOWSKA 1960a). PAN et al. (2012) confirmed that in alluvial floodplains, which include oxbow lakes, a gravel substratum supported a higher density of macroinvertebrates including Mollusca than a sandy substratum. In the oxbow lakes of the Wkra River, three size classes of the substratum were most associated with the distribution of molluscs: medium and coarse gravel, medium sand as well as coarse and very coarse gravel. These results revealed that the density of unionid species, e.g. *A. cygnea*, *A. anatina* and *U. pictorum*, was correlated with medium-grained sand (particle diameter 0.25–0.50 mm) and with a higher water temperature. This finding is in accordance with the survey of KERNEY (1999) and KILLEEN et al. (2004) who showed that *A. cygnea* avoided gravelly and rocky substrates and preferred a firm sandy bottom in which it could burrow. Besides, *A. cygnea* is often the sole large mussel species inhabiting a silt-rich substratum because it is able to lie on its side and remain at the sediment-water interface without sinking. These results also confirm the survey of CLEMENTS et al. (2006) who found that unionid bivalves exhibited a preference for a sandy substratum. In reservoirs, the parameters that were most associated with a high mollusc richness, besides pH, were the substrate grain sizes. ABRASZEWSKA-KOWALCZYK (2002) also showed that *A. cygnea*, which preferred muddy or sandy sediments with superficial layer of mineralised silt and fine-grained sediments of 0.063–0.123 mm, was a typical species of floodplain water bodies. According to BOYCOTT (1936), *A. cygnea* and *U. pictorum* abound in reservoirs without any pulmonates or operculates which illustrates the dependence of bivalves on the types of bottom. These results support his survey, because *A. anatina*, *U. pictorum* and *A. cygnea* were the species associated with the sandy substratum. However, *R. auricularia* with its adhesive foot can efficiently crawl on sand. *A. cygnea* was recorded in three out of seven oxbow lakes, in which the upper range of nitrates concentration in the water was lower than that in other oxbow lakes. Thus, the occurrence of *A. cygnea* and other unionid species in oxbow lakes was affected by the temperature and the sandy substratum rather than by nitrates.

The type of substratum influences Gastropoda when parts of it serve as food sources. Gastropods mainly feed on decayed remains of macrophytes or algae which they scrape off leaves of macrophytes, mud, stones and other surfaces (BOYCOTT 1936, ØKLAND 1990). *L. stagnalis* and *P. corneus* inhabit all categories of substratum and consume a wide variety of food. Moreover, they are more abundant in reservoirs surrounded by cultivated fields, pastures and lands (ØKLAND 1990). Viviparids not only feed on detritus but also on diatoms, some filamentous algae and blue-green algae (TSIKHON-LUKANINA et al. 1998, DILLON 2000). Viviparids were observed on both rocky sediments and fine particulate sediments, depending on the diatom patches. They aggregated at one point if only diatom patches were available as source of food. Thus, the distribution of viviparids may be more a function of the availability of food in the substratum than its texture (DILLON 2000). JAKUBIK (2012) observed seasonal dynamics in the density of *V. contectus* in different habitats including both lentic oxbow lakes and oxbow lakes connected with the main river. In summer, *V. contectus* formed aggregations in the near-shore zones that were larger in comparison to those formed in spring or autumn. Thus, in the oxbow lakes of the Wkra River, the relationship of *L. stagnalis*, *P. corneus* and *V. contectus* with coarse and very coarse gravel between or among which they can scrape off food may be explained by their feeding habits.

The organic matter content in bottom sediments ranged from 0.54% to 8.89% in the oxbow lakes of the Wkra River. These values comprised classes of organic matter content from low (< 1.0%) to high (4.1–10.0%) according to the classification of VERDONSCHOT (2001). KOC et al. (2009) reported a maximum organic matter content in bottom sediments higher than those in the oxbow lakes of the Wkra River (up to 24.3%, oxbow lakes of a large river); the same is true of JURKIEWICZ-KARNKOWSKA (2009) and JURKIEWICZ-KARNKOWSKA & KARNKOWSKI (2013) (10–80%, oxbow lakes of a very large river).

The relationship between the species composition of Mollusca and the organic matter content in the bottom sediments was not statistically significant in the oxbow lakes of the Wkra River. JURKIEWICZ-KARNKOWSKA & KARNKOWSKI (2013) showed a negative influence of the organic matter content in the bottom sediments on the species richness in the water bodies of the lower Bug River floodplain. This negative correlation was explained by a high proportion of transformed organic matter, which was resistant to degradation and biologically unavailable in water bodies that represented advanced successional stages.

The organic matter content may vary depending on the hydrological connectivity of oxbow lakes with the main river channel, for example from 0.23% to 3.97% (from low to medium organic matter content) in oxbow lakes connected with the main river channel or from 0.07% to 31.39% (from low to very high organic matter content) in the lentic oxbow lakes of the Bug River (JAKUBIK et al. 2006, JAKUBIK 2012).

In the oxbow lakes of the Wkra River, 28 taxa of macrophytes were recorded. Most of these species, including *N. lutea*, *H. morsus-ranae* and other pleustonic species, *S. aloides*, *A. calamus* or free submersed species like *E. canadensis* and *M. spicatum*, are typical of eutrophic as well as lentic (isolated) oxbow lakes (PIECHOCKI 1969, PODBIELKOWSKI & TOMASZEWICZ 1996, KOŁODZIEJCZYK 2000, OBOLEWSKI & GLIŃSKA-LEWCZUK 2011, MRÓZ 2012, PASZTALENIEC & PONIEWOZIK 2013).

Four of the six species of Lemnaceae known in Poland (WÓJCIAK & URBAN 2009) occurred in the oxbow lakes of the Wkra River. The most common duckweed, which is typical of oxbow lakes, i.e. *L. minor*, was recorded in all of the studied oxbow lakes. *L. gibba*, which is rarely found in oxbow lakes, is a nitrophilous species that is known from scattered sites in Poland (WÓJCIAK & URBAN 2009). According to SZMEJA (2006), *L. gibba* not only prefers waters with a higher nitrogen concentration, but also with a higher conductivity and pH above 7.0. It was recorded in five of the seven roxbow lakes of the Wkra River which had a relatively high concentration of nitrites and nitrates in the water as well as higher values of conductivity.

In contrast to many authors (STAŃCZYKOWSKA 1960b, DVOŘÁK & BEST 1982, DVOŘÁK 1987, 1996, KORNIJÓW et al. 1990, KORNIJÓW 1996, KORNIJÓW & ŚCIBOR 1999b) who highlighted the significant relationship between macrophytes and macroinvertebrates including molluscs in oxbow lakes or small and shallow lakes, the results of the CCA analysis in this survey showed that macrophytes were not the environmental variable that was most associated (statistically significant according to the forward selection results) with the distribution of mollusc species. For example, BROCK & VAN DER VELDE (1996) found that nymphaeid macrophytes played an important role for macroinvertebrates by providing conditions and microhabitats that do not exist in and above macrophyte-free sediments. They showed that the number of macroinvertebrate taxa including molluscs, their density and biomass were considerably higher in a Nymphoides-site in contrast to an open water site which was devoid of macrophytes (open water, macrophyte-free biotope). The sediment compartment that is rooted by nymphaeid macrophytes may be favourable to macroinvertebrates including molluscs because the roots stabilise the sediment structure and thus protect them from predation by benthic-feeding fish. In addition, the roots of some species release oxygen into the sediments. Moreover, fine detritus particles and bacteria are important food sources for mussels including *U. pictorum*, *A. anatina* and *U. tumidus* (Unionidae) and these types of food are more available in nymphaeid-dominated sites than in open water sites (BROCK & VAN DER VELDE 1996). Nymphaeid species, e.g. *N. lutea*, were recorded in all of the oxbow lakes of the Wkra River, but Unionidae occurred only in four of the seven oxbow lakes. However, all of the unionid species were sampled in the nympaheid-dominated sites in the oxbow lakes of the Wkra River.

Erosive flooding and flow pulses, acting at different time-scales, control connectivity in the riverine landscape which includes aquatic habitats, floodplain surface and riparian zone (TOCKNER et al. 2000). Hydrological connectivity between the river and its associated water bodies including oxbow lakes is the principal factor which influences biogeochemical fluxes, the fod-web stucture and is often considered to be the potential driver structuring benthic macroinvertebrates including molluscs (TOCKNER et al. 1999, RECKENDORFER et al. 2006, GALLARDO et al. 2008, OBOLEWSKI 2011b, PAN et al. 2012). Periodical flood events cause changes in lentic oxbow lakes that include the physical and chemical parameters of the water and sediments and also the accessibility of allochthonous matter and aquatic organisms, thereby regulating the aerobic conditions in water (Glińska-Lewczuk 2009, Kalinowski et al. 2011). The flood events affect the ecological function of oxbow lakes for the biota (Heimann et al. 2011) as well as the structure of mollusc communities (Maltchik et al. 2005). According to Jakubik et al. (2006), lentic (isolated from the river) oxbow lakes show a lower species diversity of benthic macroinvertebrates including molluscs than oxbow lakes that are connected with the main channel. They recorded from 13 to 14 mollusc species in oxbow lakes connected with the mainstream and 8–13 in lentic oxbow lakes.

According to Piechocki (1969), the isolation of oxbow lakes from the main channel causes disappearance of rheophilous species. *A. vortex*, *Ph. fontinalis*, *L. stagnalis*, *P. corneus*, *B. contortus*, *S. corneum*, *V. contectus* and *B. tentaculata* were the most numerous species in the lentic oxbow lakes of the Grabia River, whereas the percentage of typical fluviatile species, e.g. *A. anatina*, *U. tumidus*, *P. amnicum* and *P. henslowanum*, in the mollusc communities was smaller (Piechocki 1969). The results of the present study showed that common species, which are usually found in still waters: *L. stagnalis*, *P. corneus*, and *A. vortex*, were eudominants, whereas *V. contectus*, *B. tentaculata*, *A. lacustris* and *R. balthica* were dominants in the mollusc communities in the lentic oxbow lakes of the Wkra River. Typical fluviatile species, such as *A. fluviatilis*, *P. amnicum* and *P. supinum*, were also recorded, but only in one out of seven oxbow lakes, which was located closer to the Wkra River than the other oxbow lakes (Ox6). However, they were subrecedents or recedents in the mollusc communities. Thus, these results are consistent with the survey of Piechocki (1969).

According to Piechocki (1969), *V. contectus* is a typical species in lentic oxbow lakes. These results confirmed his survey. In the lentic oxbow lakes of the Wkra River *V. contectus* was dominant in the mollusc communities in contrast to semi-lotic or lotic oxbow lakes, where *V. viviparus* or *B. naticina* and *V. piscinalis* were eudominant (Stańczykowska 1959, 1960a, c, Obolewski et al. 2009). This survey showed that *V. contectus* and four other species, i.e. *L. stagnalis*, *P. corneus*, *A. vortex*, and *S. corneum*, were euconstants in the mollusc community of lentic oxbow lakes (C = 75.1–100.0%). Only one species, *A. vortex*, was euconstant in the lentic oxbow lakes of the Grabia River (Piechocki 1969) and three species: *B. tentaculata*, *L. stagnalis* and *P. corneus*, were euconstants in the mollusc community of permanent water bodies of the Bug River valley (Jurkiewicz-Karnkowska 2009).

Reckendorfer et al. (2006) claimed that bivalve species, for example *U. tumidus* and *U. pictorum*, were common in mollusc communities in water bodies connected with the main stream, whereas such pulmonate species as *P. corneus*, *L. stagnalis* or *A. vortex* were common in isolated (disconnected) water bodies. What is more, species that are typical of isolated water bodies are characterised by a higher resistance to desiccation, better adaptation to higher water temperature or lower dissolved oxygen concentration (physiological make-up). These results partially confirm the survey by Reckendorfer et al. (2006). In the isolated oxbow lakes of the Wkra River *L. stagnalis*, *P. corneus* and *A. vortex*, which are more resistant to desiccation than other species, were observed in all of the oxbow lakes (eudominants or dominants). Bivalve species, e.g. *U. pictorum*, *A. cygnea* or *A. anatina*, were also recorded (3 out of 7 oxbow lakes). However, *U. tumidus* occurred only in one oxbow lake Ox6, which is situated in the vicinity of the Wkra River. According to Jakubik (2012), the density of *V. contectus* was similar in both types of oxbow lakes: isolated and connected with river (up to 30 indiv. m^{-2}). In contrast, the maximum density of *V. contectus* recorded in the oxbow lakes of the Wkra River was only half the number that was obtained by Jakubik (2012).

The values of the H' indices calculated for the mollusc communities of the oxbow lakes of the Wkra River (Ox3, Ox6) were even higher than those recorded for semi-lotic or lotic as well as permanent water bodies within a 140 km section of the lower Bug River valley (Jurkiewicz-Karnkowska 2006, 2011). The number of mollusc species was relatively high (21–23 species) and the quantity ratio (number of specimens of each species) was relatively evenly balanced in the mollusc communities in oxbow lakes Ox3 and Ox6 of the Wkra River, and therefore, this reflected high values of the H' index. The relatively high total number of mollusc species reflecting higher values of the H' index in the oxbow lakes Ox6 or Ox3, which are located in the vicinity of the river, may be explained by the flooding and flow pulses concept (Tockner et al. 1999, 2000, Reckendorfer et al. 2006). Flooding and flow pulses, acting at different time-scales, may control temporary connectivity between the main river channel and the oxbow lakes of the Wkra River, and therefore exchange of the mollusc species is possible. For example, the same 12 mollusc species were recorded both in the oxbow lake Ox6 and in the main river channel (site W15).

5.2.5. OXBOW LAKE HABITATS AS REFUGES AND THEIR SIGNIFICANCE FOR BIODIVERSITY CONSERVATION

Under Annex 1 to Council Directive 92/43/EEC (1992) of the EU and its consolidation version of 01.01.2007, which includes the latest version of the annexes as well as the Polish legislation (Dz.U. 2012b), oxbow lakes are protected natural habitats (code 3150). According to these Directives, oxbow lakes, which constitute natural habitat types that are of interest to the EU Community, require designation as special conservation areas within the Habitats Directive Natura 2000 as the subtype "Eutrophic oxbow lakes and natural, small reservoirs" (code 3150–2) (Mróz 2012). The phytosociological indicators of such a type of habitat include submerged unattached macrophytes (*Potamion* and partly *Nymphaeion*) and floating attached as well as floating unattached macrophytes (*Lemnetea*). Although this survey was not carried out in terms of the natural habitat types that are of interest to the EU Community, some macrophyte species that are characteristic of these phytosociological indicators of the habitat were recorded in the oxbow lakes of the Wkra River.

It was shown that oxbow lakes provided unique habitats for many macrophyte species, e.g. *Nymphaea candida* C. Presl, *Trapa natans* L. S. L., *Salvinia natans*, *N. lutea*, *Utricularia vulgaris* L. and *Nymphaea alba* L. (Jakubska & Kazuń 2005, Spałek 2008a, 2008b, Michalska-Hejduk et al. 2009), which are legally protected (Dz.U. 2012a). This type of reservoir also supports uncommon species that are of a specific conservation interest, e.g. *Azolla filiculoides* Lam., a free-floating fern, which only occurs in a few localities including oxbow lakes (Szczęśniak et al. 2009) or *Ceratophyllum submersum* L. (Proćków & Proćków 2008). Moreover, stratiotes beds (*Stratiotes aloides*) may create a specific habitat and refuge for planktonic and bentho-planktonic crustaceans (Strzałek & Koperski 2009), for epiphytic macrofauna including *S. corvus*, *L. stagnalis* and *A. lacustris* (Obolewski 2005) as well as for other mollusc species (Obolewski et al. 2013). In the oxbow lakes of the Wkra River, rare and legally protected macrophyte species were also recorded, while stratiotes bed only occurred in one oxbow (Ox4).

Oxbow lakes and other small water bodies can contribute significantly to regional biodiversity; they maintain river valleys biodiversity and constitute habitats for many precious, rare or disappearing species. It was shown (Williams et al. 2003) that the number of macroinvertebrate species was higher in small water bodies of agricultural areas compared to lakes, streams and rivers. Small water bodies offer patches for high aquatic biodiversity including invertebrates and macrophytes even within intensely farmed agricultural areas (Sayer et al. 2012). Clements et al. (2006) recorded high species richness in small reservoirs, and therefore they highlighted their importance for the conservation of molluscs providing that the water pH was near neutral. This survey showed opposite results, i.e. a higher number of mollusc species was recorded in the Wkra River (40 species) in comparison to the oxbow lakes of the Wkra River (30 species). The phenomenon may be explained by the hydrological isolation of the oxbow lakes of the Wkra River along with the increased nutrient and salt concentration in the water which may be responsible for the lower diversity of Mollusca than in the Wkra River. This result confirms the survey of Gallardo et al. (2012) who showed that the lack of hydrological connectivity and eutrophication were the main drivers of low macroinvertebrate biodiversity and abundance, including Mollusca.

On the other hand, newly created wetlands (oxbow lakes), for example in degraded areas of floodplains, increase heterogeneity and provide new habitats that play key roles in the ecosystem from nutrient removal to carbon storage, pollutant removal, water storage during floods or water provision during droughts, as well as constituting wildlife refuges (Gallardo et al. 2012). More mobile species occur in the newly created oxbow lakes, and after that they begin to appear in existing natural oxbow lakes and vice versa. Less mobile species appear in newly created wetland after a certain period of time thus enhancing the local biological diversity of the floodplain. For example, at first, dragonflies, *Trithemis annulata* (P. de Beauv.) and *Coenagrion scitulum* (Ramb.) occurred in a newly created oxbow lake and after that they appeared in the existing natural oxbow lakes (Gallardo et al. 2012).

Oxbow lakes are also habitats for biota that have been restored after an intense high water-level period (Pakulnicka & Nowakowski 2012). These water bodies, which are continuously or periodically connected with the main river channel, are indispensable for maintaining high biodiversity; even during low discharge many species can successfully complete their life cycle in them (Penczak et al. 2004, Pakulnicka & Nowakowski 2012). In summary, oxbow lakes together with rivers and hydrological catchment areas may create a unique valuable ecosystem which contributes to the natural diversity of such areas.

SUMMARY AND CONCLUSIONS

A larger river (the Wkra River) provides more diverse instream habitats and niches than medium-sized or small rivers (tributaries of the Wkra River). Consequently, a higher number of mollusc species (40 species), including some that are rare, vulnerable, threatened or legally protected, e.g.

Borysthenia naticina, *Unio crassus* or *Pseudanodonta complanata*, was recorded in the Wkra River than in its tributaries (32 species).

The mollusc communities in the Wkra River catchment area, including *Unio crassus*, one of the species of Community interest whose conservation requires designation of special conservation areas within the Habitats Directive Natura 2000, are influenced by several environmental factors acting together. In addition to the nutrient concentration, conductivity, hardness, water temperature or dissolved oxygen, hydrological conditions, size of sediment particles and the occurrence of macrophytes are important (statistically significant). Among environmental factors, anthropogenically elevated concentrations of nitrites and nitrates in the water are the most predictive parameters that negatively (stressors) influence the structure of mollusc communities.

Compared to the Wkra River, its tributaries are more influenced by nutrient input from diffuse and point sources of pollution (higher concentration of phosphates, nitrites and nitrates in the water). Despite the establishment of one of 21 areas of Nitrate Vulnerable Zones (NVZs) designated in Poland in the Sona River catchment area in accordance with the requirements of the COUNCIL DIRECTIVE 91/676/EEC (called the Nitrates Directive), the concentration of nitrates is still high, up to 142.20 mg NO_3^- dm^{-3}. The present results also show a relatively high concentration of nitrates not only in the Sona River, but in the Wkra River (up to 87.71 mg NO_3^- dm^{-3}) and the other tributaries (up to 136.00 mg NO_3^- dm^{-3}) as well as in some of the oxbow lakes of the Wkra River (up to 103.22 mg NO_3^- dm^{-3}). The concentration of nitrates of an agricultural origin in these habitats is still higher than the trigger threshold value set in the Nitrates Directive, e.g. 50 mg NO_3^- dm^{-3}.

The long-term survey made it possible to ascertain the ranges of the environmental variables, which influenced the structure of the mollusc communities in the Wkra River catchment area. A novel finding of this survey is that some mollusc species tolerate wider ranges of some environmental factors than was expected, however only up to certain, threshold values.

The present survey shows that some gastropod species are more tolerant to relatively high nutrient concentrations in the water in comparison to unionid mussels or fingernail clams excluding *Sphaerium corneum*. Pulmonate gastropod species, e.g. *Planorbarius corneus*, *Planorbis planorbis* and *Lymnaea stagnalis* occur at river sites with an upper limit of nitrate concentration of up to 142.20 mg NO_3^- dm^{-3} and unionid mussels with an upper limit of up to 87.71 mg NO_3^- dm^{-3}. Some gastropod species were recorded at sampling sites with an extremely high phosphate concentration (up to 9.56 mg PO_4^{3-} dm^{-3}). Unionidae are less tolerant to a relatively high concentration of nutrients in the water compared to gastropods or sphaeriids. Among the sphaeriid species recorded in the Wkra River catchment area, *Sphaerium corneum* is the most tolerant and was recorded at river sites with the upper limit of nitrates (up to 136.00 mg NO_3^- dm^{-3}), nitrites (up to 2.64 mg NO_2^- dm^{-3}) and phosphates (up to 8.98 mg PO_4^{3-} dm^{-3}). The present results revealed the maximum values of some physical and chemical parameters of the water within which mollusc species are able to survive. A novel finding of this survey is that some freshwater mollusc species can tolerate higher nitrite and nitrate concentrations in rivers than the ranges shown by both toxicological research and field surveys.

The present results confirmed both the field and the toxicological research of the negative effect of nitrites and nitrates, which is caused by their toxicity to freshwater molluscs. However, the concentrations of nitrites and nitrates in the Wkra River catchment area were much higher in comparison to those that may be adequate to protect sensitive freshwater species as indicated by toxicologists. Even though high nutrient concentrations in the water were recorded, which can lead to reduction in aquatic biodiversity that favours invasive species over native species, this phenomenon has not yet been observed in the Wkra River catchment.

A novel finding of this survey is that *Unio crassus* is more tolerant to a relatively high concentration of nitrates and also to nitrites and phosphates in the water than was previously expected and that it may survive in rivers with relatively high nutrient concentrations. However, a decline in the abundance, density and number of mollusc species was recorded in the Wkra River and its tributaries in the last year of the survey. The present results confirm the global trend in a decline in the mollusc populations, especially mussels. Therefore, under Annexes II and IV to COUNCIL DIRECTIVE 92/43/EEC (1992) of the EU and its consolidation version of 01.01.2007, which includes the latest version of the annexes and the Polish legislation, some mollusc species, including *Unio crassus*, are regarded as species that are of Community interest whose conservation requires designation of special conservation areas within the Habitats Directive Natura 2000.

Most unionid species are relatively long-lived organisms. *Unio crassus* may live even up to 75 years. Relic populations can be found many years after recruitment has stopped, which has been attributed to the pollution of the water that has especially affected juveniles or the loss of the obligate host fish species. Due to the fact that juveniles (< 30.0 mm shell length) of *Unio crassus* were observed in the Wkra River, which confirms that recruitment is still occurring, conservation measures should be undertaken not only in terms of *Unio crassus* but also other endangered or rare mollusc species, paying special attention to unionid species. These conservation measures should include minimising management operations in the stretches of the river in which larger unionid populations are present; appropriate fish farming in order to minimise the forms of management, that may lead to the disappearance of host fishes or removing large mussels; reducing pollution from both point and non-point sources of nutrient input into surface freshwaters and groundwaters and implementing a wider distribution of the Rules of Good Agricultural Practice with the objectives of reducing nitrate pollution.

The highest mean value of the Shannon-Wiener index (*H*') of 3.20 was recorded at a sampling site located in the lower course of the Wkra River (site W23), above the stretch of the river which is included into both the area of Natura 2000 (PLH 140005) and the nature reserve "Dolina Wkry". A relatively high species diversity as measured by the Shannon-Wiener index was recorded at most of the sampling sites located in the lower course of the Wkra River which is almost unregulated or is regulated to a lesser degree in comparison to sites that are located in the heavily morphologically transformed upper and middle course.

Molluscs can be biological indicators of nutrient enrichment, especially the nitrite and phosphate concentrations in the water, conductivity and dissolved oxygen in the lowland rivers of agricultural areas, e.g. in the Wkra River and its tributaries.

Both the number of mollusc species, including two protected species, i.e. *Anodonta cygnea* and *Sphaerium rivicola*, and their density vary among the oxbow lakes of the Wkra River and may be dependent on various environmental factors, including lack of connectivity with the main channel, physical and chemical parameters of the water or grain sizes of the substratum.

Although relatively high concentrations of nitrates, nitrites and phosphates were recorded in the oxbow lakes of the Wkra River, which may indicate a considerable anthropogenic pressure from agriculture, the species diversity in some oxbow lakes as measured by the Shannon-Wiener index (*H*') was higher than the species diversity recorded in some oxbow lakes of large rivers, e.g. the Elba River and the Bug River. The oxbow lakes of the Wkra River provide a refuge not only for protected mollusc species, but also for many other rare and vulnerable mollusc species as well as for macrophytes and other wildlife.

The number of mollusc species and density decreased dramatically in the Wkra River and its tributaries in the last year of the survey. This phenomenon appeared to be a direct result of nutrient enrichment in the Wkra River catchment area and the regulation of riverbeds or an indirect result, e.g. the lack of primary or appropriate host fishes for unionid species. The global trend of declining mollusc populations has probably begun to occur in the Wkra River catchment.

Biodiversity conservation constitutes an essential component of the responsible management of environment and natural resources. Knowledge about sustainable management of environment in order to achieve better biodiversity outcomes is extremely important. Therefore, the results of this survey may be directly applicable for the rehabilitation of river ecosystems, monitoring these ecosystems and programmes for the conservation of biodiversity in running waters and reservoirs (oxbow lakes). The results of these long-term surveys can be useful for institutions, local and regional authorities and for the Voivodeship Inspectorates for Environmental Protection in terms of monitoring species and their habitats, and providing protection and management in accordance with the requirements of COUNCIL DIRECTIVE 92/43/EEC (1992) of EU and its consolidation version of 01.01.2007 including the latest version of the annexes and the Polish legislation (DZ.U. 2012b).

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REFERENCES

ABRASZEWSKA-KOWALCZYK A. 2002. Unionid bivalves of the Pilica River catchment area. Folia Malacol. 10: 99–173.

ALDRIDGE D. C. 2000. The impacts of dredging and weed cutting on a population of freshwater mussels (Bivalvia: Unionidae). Biol. Conserv. 95: 247–257. http://dx.doi.org/10.1016/S0006-3207(00)00045-8

ALLAN J. D., CASTILLO M. M. 2007. Stream ecology. Structure and function of running waters. Second edition. Springer, Dordrecht, the Netherlands. http://dx.doi.org/10.1007/978-1-4020-5583-6

ALONSO A., CAMARGO J. A. 2006. Toxicity of nitrite to three species of freshwater invertebrates. Environ. Toxicol. 21: 90–94. http://dx.doi.org/10.1002/tox.20155

ALONSO A., CAMARGO J. A. 2009. Effects of pulse duration and post-exposure period on the nitrite toxicity to a freshwater amphipod. Ecotox. Environ. Safe. 72: 2005–2008. http://dx.doi.org/10.1016/j.ecoenv.2009.06.008

ALTERRA 2007. Assessment of the designation of Nitrate Vulnerable Zones in Poland. Contract 2006/441164/MAR/B1. Implementation of the Nitrates Directive (91/676/EEC). Task 3. Alterra, Environmental Sciences, Wageningen University and Research Centre, Wageningen.

AQEM CONSORTIUM 2002. Manual for the application of the AQEM system. A comprehensive method to assess European streams using benthic macroinvertebrates, developed for the purpose of the Water Framework Directive. Version 1.0, February 2002. Available from: www.aqem.de

BAUR B., RINGEIS B. 2002. Changes in gastropod assemblages in freshwater habitats in the vicinity of Basel (Switzerland) over 87 years. Hydrobiologia 479: 1–10. http://dx.doi.org/10.1023/A:1021008010386

BEISEL J. N., USSEGLIO-POLATERA P., THOMAS S., MORETEAU J. C. 1998. Stream community structure in relation to spatial variation: the influence of mesohabitat characteristics. Hydrobiologia 389: 73–88. http://dx.doi.org/10.1023/A:1003519429979

BENDELL B. E., MCNICOL D. K. 1993. Gastropods from small northeastern Ontario Lakes: their value as indicators of acidification. Can. Field. Nat. 107: 267–272.

BEONE G. M., CATTANI I., FONTANELLA M. C., RAVERA O. 2011. Relationship between element concentrations and body size in the lake Maggiore population of *Unio pictorum mancus* (Mollusca, Bivalvia). J. Limnol. 70: 283–292. http://dx.doi.org/10.4081/jlimnol.2011.283

BIS B., MIKULEC A. 2013. Przewodnik do oceny stanu ekologicznego rzek na podstawie makrobezkręgowców bentosowych. Inspekcja Ochrony Środowiska. Biblioteka Monitoringu Środowiska, Warszawa.

BIS B., ZDANOWICZ A., ZALEWSKI M. 2000. Effects of catchment properties on hydrochemistry, habitat complexity and invertebrate community structure in lowland river. Hydrobiologia 422/423: 369–387. http://dx.doi.org/10.1023/A:1017002923173

BŁACHUTA J., CZOCH K., KULESZA K., PICIŃSKA-FAŁTYNOWICZ J. 2005. Typologia rzek i strumieni Polski. I Ogólnopolska Konferencja Naukowa. Wdrażanie Ramowej Dyrektywy Wodnej: ocena stanu ekologicznego wód w Polsce: 5–7.

BODIN N., N'GOM-KÂ R., KÂ S., THIAW O. T., TITO DE MORAIS L., LE LOC'H F., ROZUEL-CHARTIER E., AUGER D., CHIFFOLEAU J. F. 2013. Assessment of trace metal contamination in mangrove ecosystems from Senegal, West Africa. Chemosphere 90: 150–157. http://dx.doi.org/10.1016/j.chemosphere.2012.06.019

BOGAN A. E. 2008. Global diversity of freshwater mussels (Mollusca, Bivalvia) in freshwater. Hydrobiologia 595: 139–147. http://dx.doi.org/10.1007/s10750-007-9011-7

BOYCOTT A. E. 1936. Habitats of fresh-water Mollusca in Britain. J. Anim. Ecol. 5: 116–186. http://dx.doi.org/10.2307/1096

BÓDIS E., NOSEK J., OERTEL N., TÓTH B., HORNUNG E., SOUSA R. 2011. Spatial distribution of bivalves in relation to environmental condition (middle Danube catchment, Hungary). Community Ecol. 12: 210–219. http://dx.doi.org/10.1556/ComEc.12.2011.2.9

BRACCIA A. J., VOSHELL J. R. JR. 2007. Benthic macroinvertebrate responses to increasing levels of cattle grazing in Blue Ridge Mountain Streams, Virginia, USA. Environ. Monit. Assess. 131: 185–200. http://dx.doi.org/10.1007/s10661-006-9467-3

BRIERS R. 2003. Range size and environmental calcium requirements of British freshwater gastropods. Global Ecol. Biogeogr. 12: 47–51. http://dx.doi.org/10.1046/j.1466-822X.2003.00316.x

BROCK T. C. M., VAN DER VELDE G. 1996. Aquatic macroinvertebrate community structure of a *Nymphoides peltata*- dominated and macrophyte-free site in an oxbow lake. Neth. J. Aquat. Ecol. 30: 151–163. http://dx.doi.org/10.1007/BF02272235

BRÖNMARK C. 1989. Interactions between epiphytes, macrophytes and freshwater snails: a review. J. Mollus. Stud. 55: 299–311. http://dx.doi.org/10.1093/mollus/55.2.299

CALLIL C. T., JUNK W. J. 2001. Aquatic gastropods as mercury indicators in the Pantanal of Poconé region (Mato Grosso, Brasil). Water Air Soil Poll. 319: 319–330. http://dx.doi.org/10.1023/A:1005230716898

CAMARGO J. A., ALONSO A. 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: a global assessment. Environ. Int. 32: 831–849. http://dx.doi.org/10.1016/j.envint.2006.05.002

CAMARGO J. A., ALONSO A., SALAMANCA A. 2005. Nitrate toxicity to aquatic animals: a review with the data for freshwater invertebrates. Chemosphere 58: 1255–1267. http://dx.doi.org/10.1016/j.chemosphere.2004.10.044

CARLSSON R. 2001. Species-area relationships, water chemistry and species turnover of freshwater snails on the Åland Islands, southwestern Finland. J. Mollus. Stud. 67: 17–26. http://dx.doi.org/10.1093/mollus/67.1.17

CHALUPOVÁ D., HAVLÍKOVÁ P., JANSKÝ B. 2012. Water quality of selected fluvial lakes in the context of the Elbe River pollution and anthropogenic activities in the floodplain. Environ. Monit. Assess. 184: 6283–6295. http://dx.doi.org/10.1007/s10661-011-2419-6

CHERTOPRUD M. V., UDALOV A. A. 1996. Ecological groups of freshwater Gastropoda in the central part of European Russia- the influence of the type of water body and substrate. Zool. Zh. 75: 664–676.

CLARKE K. D., SCRUTON D. A. 1997. The benthic community of stream riffles in Newfoundland, Canada and its relationship to selected physical and chemical parameters. J. Freshwater Ecol. 12: 113–121. http://dx.doi.org/10.1080/02705060.1997.9663514

CLEMENTS R., KOH L. P., LEE T. M., MEIER R., LI D. 2006. Importance of reservoirs for the conservation of freshwater molluscs in a tropical urban landscape. Biol. Conserv. 128: 136–146. http://dx.doi.org/10.1016/j.biocon.2005.09.023

COSGROVE P. J., HASTIE L. C. 2001. Conservation of threatened freshwater pearl mussel populations: river management, mussel translocation and conflict resolution. Biol. Conserv. 99: 183–190. http://dx.doi.org/10.1016/S0006-3207(00)00174-9

COUNCIL DIRECTIVE 91/676/EEC. Council Directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (91/676/EEC).

COUNCIL DIRECTIVE 92/43/EEC. Council Directive of 21 May 1992 on the conservation of natural habitats and wild fauna and flora.

CROWL T. A., SCHNELL G. D. 1990. Factors determining population density and size distribution of a freshwater snail in streams: effects of spatial scale. Oikos 59: 359–367. http://dx.doi.org/10.2307/3545147

CUTTELOD A., SEDDON M., NEUBERT E. 2011. European Red List of Non-marine Molluscs. Publications Office of the European Union, Luxembourg.

DAWIDEK J., TURCZYŃSKI M. 2006. Charakterystyka hydrologiczna jezior. In: WOJCIECHOWSKA W. (ed.). Jeziora rzeczne doliny środkowego Bugu. Różnorodność biologiczna i krajobrazowa. Wydawnictwo KUL, Lublin, pp. 19–28.

DENIC M., STOECKL K., GUM B., GEIST J. 2013. Physicochemical assessment of *Unio crassus* habitat quality in a small upland stream and implication for conservation. Hydrobiologia. http://dx.doi.org/10.1007/s10750-013-1467-z

DILLON R. T. 2000. The ecology of freshwater molluscs. Cambridge University Press, Cambridge. http://dx.doi.org/10.1017/CBO9780511542008

DILLON R. T. JR., BENFIELD E. F. 1982. Distribution of pulmonate snails in the New River of Virginia and North Carolina, U.S.A.: interaction between alkalinity and stream drainage area. Freshwater Biol. 12: 179–186. http://dx.doi.org/10.1111/j.1365-2427.1982.tb00612.x

DIRECTIVE 2000/60/EC. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy.

DIRECTIVE 2009/147/EC. Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds (codified version).

DODDS W. K. 2002. Freshwater ecology. Concepts and environmental applications. Academic Press, USA.

DODDS W. K. 2006. Eutrophication and trophic state in rivers and streams. Limnol. Oceanogr. 51: 671–680. http://dx.doi.org/10.4319/lo.2006.51.1_part_2.0671

DOMINGO C. Y. J., CASAO P., ASUNCION M. B. 2012. Cryptosporidium oocysts (Apicomplexa: Cryptosporidiidae) in the freshwater Asiatic clam, *Corbicula fluminea* O. F. Müller, 1774 (Bivalvia: Corbiculidae) from selected municipalities of Aurora, Philippines. Philipp. J. Vet. Med. 49: 82–87.

DOUDA K. 2010. Effects of nitrate nitrogen pollution on Central European unionid bivalves revealed by distribution data and acute toxicity testing. Aquat. Conserv. 20: 189–197. http://dx.doi.org/10.1002/aqc.1076

DOUDA K., HORKÝ P., BILÝ M. 2012. Host limitation of the thick-shelled river mussel: identifying the threats to decline affiliate species. Anim. Conserv. 15: 536–544. http://dx.doi.org/10.1111/j.1469-1795.2012.00546.x

DUSSART G. B. J. 1976. The ecology of freshwater molluscs in north west England in relation to water chemistry. J. Mollus. Stud. 42: 181–198.

DUSSART G. B. J. 1979. Life cycles and distribution of the aquatic gastropod molluscs *Bithynia tentaculata* (L.), *Gyraulus albus* (Muller), *Planorbis planorbis* (L.) and *Lymnaea peregra* (Muller) in relation to water chemistry. Hydrobiologia 67: 223–239. http://dx.doi.org/10.1007/BF00023179

DVOŘÁK J. 1987. Production-ecological relationships between aquatic vascular plants and invertebrates in shallow waters and wetlands – a review. Arch. Hydrobiol. Beih. Ergebn. Limnol. 27: 181–184.

DVOŘÁK J. 1996. An example of relationships between macrophytes, macroinvertebrates and their food resources in a shallow eutrophic lake. Hydrobiologia 339: 27–36. http://dx.doi.org/10.1007/BF00008910

DVOŘÁK J., BEST E. P. H. 1982. Macro-invertebrate communities associated with the macrophytes of Lake Vechten: structural and functional relationships. Hydrobiologia 95: 115–126. http://dx.doi.org/10.1007/BF00044479

DYDUCH-FALNIOWSKA A. 1982. Oscillations in density and diversity of *Pisidium* communities in two biotopes in southern Poland. Hydrobiol. Bull. 16: 123–132. http://dx.doi.org/10.1007/BF02255366

DYDUCH-FALNIOWSKA A., ZAJĄC K. 2002. Bivalvia Małże. In: GŁOWACIŃSKI Z. (ed.). Czerwona Lista Zwierząt Ginących i Zagrożonych w Polsce. Red List of Threatened Animals in Poland. Polish Academy of Sciences, Institute of Nature Conservation Cracow, pp. 23–26.

DZ.U. 2002. Dziennik Ustaw nr 241, poz. 2093. Rozporządzenie Ministra Środowiska z dnia 23 grudnia 2002 r. w sprawie kryteriów wyznaczania wód wrażliwych na zanieczyszczenie związkami azotu ze źródeł rolniczych.

Dz.U. 2011a. Dziennik Ustaw nr 237, poz. 1419. Rozporządzenie Ministra Środowiska z dnia 12 października 2011 r. w sprawie ochrony gatunkowej zwierząt.

Dz.U. 2011b. Dziennik Ustaw nr 25, poz. 133. Rozporządzenie Ministra Środowiska z dnia 12 stycznia 2011 r. w sprawie obszarów specjalnej ochrony ptaków.

Dz.U. 2012a. Dziennik Ustaw poz. 81, rozporządzenie Ministra Środowiska z dnia 5 stycznia 2012 w sprawie ochrony gatunkowej roślin.

Dz.U. 2012b. Dziennik Ustaw Rzeczypospolitej Polskiej nr 0, poz. 1041. Rozporządzenie Ministra Środowiska z dnia 9 sierpnia 2012 r. zmieniające rozporządzenie w sprawie siedlisk przyrodniczych oraz gatunków będących przedmiotem zainteresowania Wspólnoty, a także kryteriów wyboru obszarów kwalifikujących się do uznania lub wyznaczenia jako obszary Natura 2000.

Eaton A. D., Franson M. A. H. 2005. Standard methods for the examination of water and wastewater. American Public Health Association.

Elliott J. M. 1993. Some methods for the statistical analysis of samples of benthic invertebrates. Freshwater Biological Association, Scientific Publication no 25. Titus Wilson & Son, Kendal, England.

Falfushynska H. I., Gnatyshyna L. L., Farkas A., Vehovszky Á., Gyori J., Stoliar O. B. 2010. Vulnerability of biomarkers in the indigenous mollusk *Anodonta cygnea* to spontaneous pollution in a transition country. Chemosphere 81: 1342–1351. http://dx.doi.org/10.1016/j.chemosphere.2010.08.016

Falniowski A. 1987. Hydrobioidea of Poland (Prosobranchia: Gastropoda). Zeszyty Naukowe Akademii Górniczo-Hutniczej im. Stanisława Staszica nr 1096. Folia Malacol. 1: 1–121.

Flessas C., Couillard Y., Pinel-Alloul B., St-Cyr L., Campbell P. G. C. 2000. Metal concentrations in two freshwater gastropods (Mollusca) in the St. Lawrence River and relationships with environmental contamination. Can. J. Fish. Aquat. Sci. 57, Suppl. 1: 126–137. http://dx.doi.org/10.1139/f99-229

Fovler J., Cohen L., Jarvis P. 1998. Practical statistics for field biology. John Wiley & Sons Ltd., England.

Frantsevich L. I., Pankov I. V., Ermakov A. A., Kornyushin A. V., Zakharchuk T. N. 1995. Molluscs as indicators of environmental pollution by radionuclides. Russ. J. Ecol. 26: 47–52.

Funk A., Schiemer F., Reckendorfer W. 2013. Metacommunity structure of aquatic gastropods in a river floodplain: the role of niche breadth and drift propensity. Freshwater Biol. 58: 2505–2516. http://dx.doi.org/10.1111/fwb.12228

Gabriels W., Lock K., DePauw N., Goethals P. L. M. 2010. Multimetric Macroinvertebrate Index Flanders (MMIF) for biological assessment of rivers and lakes in Flanders (Belgium). Limnologica 40: 199–207. http://dx.doi.org/10.1016/j.limno.2009.10.001

Galbarczyk-Gąsiorowska L., Gąsiorowski M., Szeroczyńska K. 2009. Reconstruction of human influence during the last two centuries on two small oxbow lakes near Warsaw (Poland). Hydrobiologia 631: 173–183. http://dx.doi.org/10.1007/s10750-009-9809-6

Gallardo B., Cabezas Á., González E., Comín F. A. 2012. Effectiveness of a newly created oxbow lake to mitigate habitat loss and increase biodiversity in regulated floodplain. Restor. Ecol. 20: 387–394. http://dx.doi.org/10.1111/j.1526-100X.2010.00766.x

Gallardo B., García M., Cabezas Á., González E., González M., Ciancarelli C., Comín F. A. 2008. Macroinvertebrate patterns along environmental gradients and hydrological connectivity within a regulated river-floodplain. Aquat. Sci. 70: 248–258. http://dx.doi.org/10.1007/s00027-008-8024-2

Gangloff M. M., Feminella J. W. 2007. Stream channel geomorphology influences mussel abundance in southern Appalachian streams, USA. Freshwater Biol. 52: 64–74. http://dx.doi.org/10.1111/j.1365-2427.2006.01673.x

Glińska-Lewczuk K. 2009. Water quality dynamics of oxbow lakes in young glacial landscape of NE Poland in relation to their hydrological connectivity. Ecol. Eng. 35: 25–37. http://dx.doi.org/10.1016/j.ecoleng.2008.08.012

Glöer P. 2002. Mollusca I. Süsswassergastropoden. Nord- und Mitteleuropas Bestimmungsschlüssel, Lebensweise, Verbreitung. ConchBooks, Hackenheim.

Glöer P., Meier-Brook C. 1998. Süsswassermollusken. Ein Bestimmungsschlüssel für die Bundesrepublik Deutschland. Deutscher Jugendbund für Naturbeobachtung DJN, Hamburg.

Głowaciński Z. 1996. Różnorodność gatunkowa – jej interpretacja i obliczanie. Species diversity – its interpretation and calculation. Różnorodność biologiczna: pojęcia, oceny, zagadnienia ochrony i kształtowania. Zeszyty Naukowe Komitetu "Człowiek i Środowisko" 15: 57–70.

Głowaciński Z., Nowacki J. 2004. Polska czerwona księga zwierząt. Bezkręgowce. Polish Red Data Book of Animals. Invertebrates. Institute of Nature Conservation PAS, Cracow, The August Cieszkowski Agricultural University of Poznań.

Górny M., Grüm L. 1981. Metody stosowane w zoologii gleby. Państwowe Wydawnictwo Naukowe, Warszawa.

Grabow K., Martens A., Eggers T. O., Wimmer W. 2000. Die Großmuschlen (Bivalvia: Unionidae) in fer Fließgewäs der Stadt Braunschweig – heute und vor einhundert Jahren. Unionidae (Mollusca: Bivalvia) in running waters of Braunschweig (Germany) – today and a hundred years ago. Braunschw. Naturkdl. Schr. 6: 39–55.

Graczyk T. K., Conn D. B., Marcogliese D. J., Graczyk H., De Lafontaine Y. 2003. Accumulation of human waterborne parasites by zebra mussels (*Dreissena polymorpha*) and Asian freshwater clams (*Corbicula fluminea*). Parasitol. Res. 89: 107–112. http://dx.doi.org/10.1007/s00436-002-0729-x

Harding J. S., Young R. G., Hayes J. W., Shearer K. A., Stark J. D. 1999. Changes in agricultural intensity and river health along a river continuum. Freshwater Biol. 42: 345–357. http://dx.doi.org/10.1046/j.1365-2427.1999.444470.x

Hassall C., Hollinshead J., Hull A. 2011. Environmental correlates of plant and invertebrate species richness in ponds. Biodivers. Conserv. 20: 3189–3222. http://dx.doi.org/10.1007/s10531-011-0142-9

Hauer F. R., Lamberti G. A. 2007. Methods in stream ecology. 2nd ed., Academic Press, Elsevier.

Heimann W., Sylvester M., Seiler T. B., Hollert H., Schulz R. 2011. Sediment toxicity in a connected oxbow lake of the Upper Rhine (Germany): EROD induction in fish cells. J. Soil. Sediment. 11: 1279–1291. http://dx.doi.org/10.1007/s11368-011-0416-6

Heino J. 2000. Lentic macroinvertebrate assemblage structure along gradients in spatial heterogeneity, habitat size and water chemistry. Hydrobiologia 418: 229–242. http://dx.doi.org/10.1023/A:1003969217686

Herbst D. B., M. T., Lusardi R. A. 2008. Low specific conductivity limits growth and survival of the New Zealand mud snail from the Upper Owens River, California. West. N. Am. Naturalist 68: 324–333.

Hermanowicz W., Dojlido J., Dożańska W., Koziorowski B., Zerbe J. 1999. Fizyczno-chemiczne badanie wody i ścieków. Wydawnictwo Arkady, Warszawa.

Hornbach D. J., Deneka T. 1996. A comparison of qualitative and a quantitative collection method for examining freshwater mussel assemblages. J. N. Am. Benthol. Soc. 15: 587–596. http://dx.doi.org/10.2307/1467809

Hus M., Śmiałek M., Zając K., Zając T. 2006. Occurrence of *Unio crassus* (Bivalvia, Unionidae) depending on water chemistry in the foreland of the Polish Carpathians. Pol. J. Environ. Stud. 15: 169–172.

Hwang S.-J., Kim N.-Y., Yoon S. A., Kim B.-H., Park M. H., You K.-A., Lee h. Y., Kim H. S., Kim y. J., Lee J., Lee o. M., Shin J. K., Lee E. J., Jeon S. L., Joo H. S. 2011. Distribution of benthic diatoms in Korean rivers and streams in relation to environmental variables. Ann. Limnol. – Int. J. Lim. 47: 15–33. http://dx.doi.org/10.1051/limn/2011017

IUCN 2012. The IUCN Red List of Threatened Species: (IUCN 2012. IUCN Red List of Threatened Species. Version 2012.1).

Jackiewicz M. 1998. European species of the family Lymnaeidae (Gastropoda: Pulmonata: Basommatophora). Genus 9: 1–93.

Jakubik B. 2012. Life strategies of Viviparidae (Gastropoda: Caeonogastropoda: Architaenioglossa) in various aquatic habitats: *Viviparus viviparus* (Linnaeus, 1758) and *V. contectus* (Millet, 1813). Folia Malacol. 20: 145–179. http://dx.doi.org/10.2478/v10125-012-0013-3

Jakubik B., Kufel L., Lewandowski K. 2006. Macrobenthos differentiation among oxbow lakes of the River Bug within the Bug River Valley Landscape Park. Teka Kom. Ochr. Kszt. Środ. Przyr. 3: 55–59.

Jakubska A., Kazuń A. 2005. Stanowisko kotewki orzech wodny *Trapa natans* L. (Trapaceae) w starorzeczu koło Zakrzowa (powiat oławski). Acta Bot. Siles. 2: 155–158.

Jarvie H. P., Haygarth P. M., Neal C., Butler P., Smith B., Naden P.S., Joynes A., Neal M., Wickham H., L., Armstrong L., Harman S., Palmer-Felgate E. J. 2008. Stream water chemistry and quality along an upland-lowland rural land-use continuum, south west England. J. Hydrol. 350: 215–231. http://dx.doi.org/10.1016/j.jhydrol.2007.10.040

Jarvie H. P., Withers P. J. A., Bowes M. J., Palmer-Felgate E. J., Harper D. M., Wasiak K., Wasiak P., Hodgkinson R. A., Bates A., Stoate C., Neal M., Wickham H. D., Harman S. A., Armstrong L. K. 2010. Streamwater phosphorus and nitrogen across a gradient in rural-agricultural land use intensity. Agr. Ecosyst. Environ. 135: 238–252. http://dx.doi.org/10.1016/j.agee.2009.10.002

Jiang X., Xiong J., Song Z., Morse J. C., Jones F. C., Xie Z. 2013. Is coarse taxonomy sufficient for detecting macroinvertebrate patterns in floodplain lakes? Ecol. Indic. 27: 48–55. http://dx.doi.org/10.1016/j.ecolind.2012.11.015

Joniak T., Kuczyńska-Kippen N., Nagengast B. 2006. Chemistry of waters of small water bodies in the agricultural landscape of the Western Wielkopolska region. Teka Kom. Ochr. Kszt. Środ. Przyr. 3: 60–65.

Jonsson A., Bertilsson A., Rydgård M. 2013. Spatial distribution and age structure of the freshwater unionid mussels *Anodonta anatina* and *Unio tumidus*: implications for environmental monitoring. Hydrobiologia 711: 61–70. http://dx.doi.org/10.1007/s10750-013-1455-3

Jurkiewicz-Karnkowska E. 1994. Mięczaki a metale ciężkie w środowiskach słodkowodnych i lądowych. Molluscs and heavy metals in freshwater and terrestrial environments. Wiad. Ekol. 40: 127–141.

Jurkiewicz-Karnkowska E. 2002. Differentiation of phosphorus concentration in selected mollusc species from the Zegrzyński Reservoir (Central Poland): implications for P accumulation in mollusc communities. Pol. J. Environ. Stud. 11: 355–359.

Jurkiewicz-Karnkowska E. 2006. Communities of aquatic molluscs in floodplain water bodies of lowland river (Bug River, East Poland). Pol. J. Ecol. 54: 253–266.

Jurkiewicz-Karnkowska E. 2009. Diversity of aquatic malacofauna within a floodplain of a large lowland river (Lower Bug River, Eastern Poland). J. Mollus. Stud. 75: 223–234. http://dx.doi.org/10.1093/mollus/eyp017

Jurkiewicz-Karnkowska E. 2011. Effect of habitat conditions on the diversity and abundance of molluscs in floodplain water bodies of different permanence of flooding. Pol. J. Ecol. 59: 165–178.

Jurkiewicz-Karnkowska E., Karnkowski P. 2013. GIS analysis reveals the high diversity and conservation value of mollusc assemblages in the floodplain wetlands of the lower Bug River (East Poland). Aquatic Conserv: Mar. Freshw. Ecosyst. 23: 952–963. http://dx.doi.org/10.1002/aqc.2351

Kalinowski A., Burandt P., Glińska-Lewczuk K. 2011. Wpływ czynników hydrologicznych na warunki termiczno-tlenowe starorzeczy na przykładzie doliny Drwęcy. Proc. ECOpole 5: 245–250.

Kappes H., Haase P. 2012. Slow, but steady: dispersal of freshwater molluscs. Aquat. Sci. 74: 1–14. http://dx.doi.org/10.1007/s00027-011-0187-6

Kawecka B., Eloranta P. V. 1994. Zarys ekologii glonów wód słodkich i środowisk lądowych. Wydawnictwo Naukowe PWN, Warszawa.

Kerney M. 1999. Atlas of the land and freshwater molluscs of Britain and Ireland. Harley Books, England.

KILLEEN I., ALDRIDGE D., OLIVER G. 2004. Freshwater Bivalves of Britain and Ireland. FSC, AIDGSP Occasional Publication 82.

KOC J., KOBUS S., GLIŃSKA-LEWCZUK K. 2009. The significance of oxbow lakes for the ecosystem of afforested river valleys. J. Water Land Dev. 13a: 115–131. http://dx.doi.org/10.2478/v10025-010-0023-8

KODEKS 2004. Kodeks Dobrej Praktyki Rolniczej. Ministerstwo Rolnictwa i Rozwoju Wsi, Ministerstwo Środowiska. Fundacja Programów Pomocy dla Rolnictwa, Warszawa.

KOŁODZIEJCZYK A. 1984. Occurrence of Gastropoda in the lake littoral and their role in the production and transformation of detritus. I. Snails in the littoral of Mikołajskie Lake- general characteristic of occurrence. Ekol. Pol. 32: 441–468.

KOŁODZIEJCZYK A. 2000. Rezerwat krajobrazowy Jeziorko Czerniakowskie. In: KOT H., DOMBROWSKI A. (eds). Strategia ochrony fauny na Nizinie Mazowieckiej. Mazowieckie Towarzystwo Ochrony Fauny, Siedlce, pp. 201–216.

KOŁODZIEJCZYK A. 2011. *Lithoglyphus naticoides* (C. Pfeiffer 1828). In: GŁOWACIŃSKI Z., OKARMA H., PAWŁOWSKI J., SOLARZ W. (eds). Alien species in the fauna of Poland. I. An overview of the status. Instytut Ochrony Przyrody PAN, Kraków, pp. 70–75.

KOŁODZIEJCZYK A., DOŁĘGA A. E. 2004. Molluscs of selected water bodies in Warsaw. Folia Malacol. 12: 63–71.

KONDRACKI J. 2002. Geografia regionalna Polski. Wydawnictwo Naukowe PWN, Warszawa.

KORNIJÓW R. 1996. Znaczenie bezkręgowców jako konsumentów zielenic nitkowatych i moczarki kanadyjskiej w litoralu jeziornym. Seria Wydawnicza- Rozprawy Naukowe 188. Wydawnictwo Akademii Rolniczej, Lublin.

KORNIJÓW R., GULATI R. D., VAN DONK E. 1990. Hydrophyte-macroinvertebrate interactions in Zwemlust, a lake undergoing biomanipulation. Hydrobiologia 200/201: 467–474. http://dx.doi.org/10.1007/BF02530364

KORNIJÓW R., ŚCIBOR R. 1999a. Seasonal changes in macrofaunal feeding groups associated with *Nuphar lutea* (L.) Sm. leaves in a small eutrophic lake. Pol. J. Ecol. 47: 135–147.

KORNIJÓW R., ŚCIBOR R. 1999b. Inverebrate herbivores and their impact on damages of water lily (*Nuphar lutea*) floating leaves in a small eutrophic lake. Pol. Arch. Hydrobiol. 46: 155–160.

KORYCIŃSKA M. 2002. Molluscs of the Liwiec River (South Podlasie and Middle Mazovian Lowlands). Folia Malacol. 10: 17–23.

KÖHLER R. 2006. Observations on impaired vitality of *Unio crassus* (Bivalvia: Najadae) populations in conjunction with elevated nitrate concentration in running waters. Acta Hydrochim. Hydrobiol. 34: 346–348. http://dx.doi.org/10.1002/aheh.200500635

KRYGER J., RIISGÅRD H. U. 1988. Filtration rate capacities in 6 species of European freshwater bivalves. Oecologia 77: 34–38. http://dx.doi.org/10.1007/BF00380921

KUPIEC J., ŁAWNICZAK A. E., ZBIERSKA J. 2008. Action reducing the outflow of nitrates from agricultural sources to waters on the nitrate vulnerable zone in the catchment of the Samica Stęszewska river. Ann. Warsaw Univ. of Life Sci. – SGGW Land Reclam. 40: 3–13.

KZGW 2008. Wszystko, co wato wiedzieć o wdrażaniu dyrektywy azotanowej w Polsce. Krajowy Zarząd Gospodarki Wodnej, Warszawa.

LABROT F., NARBONNE J. F., VILLE P., SAINT DENIS M., D. RIBERA D. 1999. Acute toxicity, toxicokinetics, and tissue target of lead and uranium in the clam *Corbicula fluminea* and the worm *Eisenia fetida:* comparison with the fish *Brachydanio rerio*. Arch. Environ. Con. Tox. 36: 167–178. http://dx.doi.org/10.1007/s002449900457

LECHOWSKI L., BUCZYŃSKI P. 2006. Aquatic and semiaquatic bugs (Heteroptera: Nepomorpha et Gerromorpha) of water bodies in the middle reach of the River Bug and its valley. Acta Biol. Univ. Daugavp. 6: 109–115.

LEWANDOWSKI K. 2004. Mięczaki (Mollusca) w dorzeczach Wisły i Odry. Biuletyn Monitoringu Przyrody 1: 5–9.

LEWANDOWSKI K. 2006. Bivalves of the family Unionidae in ox-bow lakes of the Bug River. Fragm. Faun. 49: 69–74.

LEWANDOWSKI K., STAŃCZYKOWSKA A. 2000. Rola małża *Dreissena polymorpha* (Pall.) (racicznica zmienna) w ekosystemach słodkowodnych. The role of *Dreissena polymorpha* (Pall.) (zebra mussel) in freshwater ecosystems. Przegl. Zool. 43: 13–21.

LEWIN I. 2012. Occurrence of the invasive species *Potamopyrgus antipodarum* (Prosobranchia: Hydrobiidae) in mining subsidence reservoirs in Poland in relation to environmental factors. Malacologia 55: 15–31. http://dx.doi.org/10.4002/040.055.0102

LODGE D. M., BROWN K. M., KLOSIEWSKI S. P., STEIN R. A., COVICH A. P., LEATHERS B. K., BRÖNMARK C. 1987. Distribution of freshwater snails: spatial scale and the relative importance of physicochemical and biotic factors. Am. Malacol. Bull. 5: 73–84.

LUKASHEV D. V. 2010. Accumulation of heavy metals by *Anodonta anatina* (L.) in the place of household sewage release in the river ecosystem. Hydrobiol. J. 46: 67–77. http://dx.doi.org/10.1615/HydrobJ.v46.i4.50

ŁAWICKI Ł., WYLEGAŁA P., WUCZYŃSKI A., SMYK B., LENKIEWICZ W., POLAKOWSKI M., KRUSZYK R., RUBACHA S., JANISZEWSKI T. 2012. Rozmieszczenie, charakterystyka i status ochronny noclegowisk gęsi w Polsce. Ornis Polonica 53: 23–38.

MALONEY K. O., LELLIS W. A., BENNETT R. M., WADDLE T. J. 2012. Habitat persistence for sedentary organisms in managed rivers: the case for the federally endangered dwarf wedgemussel (*Alasmidonta heterodon*) in the Delaware River. Freshwater Biol. 57: 1315–1327. http://dx.doi.org/10.1111/j.1365-2427.2012.02788.x

MALTCHIK L., FLORES M. L. T., STENERT C. 2005. Benthic macroinvertebrate dynamics in a shallow floodplain lake in the South of Brazil. Acta Limnol. Bras. 17: 173–183.

MARET T. R., KONRAD C. P., TRANMER A. W. 2010. Influence of environmental factors on biotic responses to nutrient enrichment in agricultural streams. J. Am. Water Resour. Ass. 46: 498–513.

MARTINOVIC-VITANOVIC V. M., RAKOVIC M. J., POPOVIC N. Z., KALAFATIC V. I. 2013. Qualititative study of Mollusca communities in the Serbian Danube stretch (river km 1,260–863.4). Biologia 68: 112–130. http://dx.doi.org/10.2478/s11756-012-0137-3

MCCUNE B., GRACE J. B. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, Oregon, USA.

MCIVOR A. L., ALDRIDGE D. C. 2007. The reproductive biology of the depressed river mussel, *Pseudanodonta complanata* (Bivalvia: Unionidae), with implications for its conservation. J. Mollus. Stud. 73: 259–266. http://dx.doi.org/10.1093/mollus/eym023

MCRAE S. E., ALLAN J. D., BURCH J. B. 2004. Reach- and catchment-scale determinants of the distribution of freshwater mussels (Bivalvia: Unionidae) in south-eastern Michigan, U.S.A. Freshwater Biol. 49: 127–142.

MEIER-BROOK C. 1969. Substrate relations in some *Pisidium* species (Eulamellibranchiata: Sphaeriidae). Malacologia 9: 121–125.

MEWES M. 2012. Diffuse nutrient reduction in the German Baltic Sea catchment: cost-effectiveness analysis of water protection measures. Ecol. Indic. 22: 16–26. http://dx.doi.org/10.1016/j.ecolind.2012.01.006

MICHALIK-KUCHARZ A., STRZELEC M., SERAFIŃSKI W. 2000. Malacofauna of rivers in Upper Silesia (Southern Poland). Malak. Abh. 20: 101–109.

MICHALSKA-HEJDUK D., KOPEĆ D., DROBNIEWSKA A., SUMOROK B. 2009. Comparison of physical and chemical properties of water and floristic diversity of oxbow lakes under different levels of human pressure: A case study of the lower San River (Poland). Ecohydrol. Hydrobiol. 9: 183–191. http://dx.doi.org/10.2478/v10104-010-0016-x

MINTON R. L., WHITE J. D., HAYES D. M., CHENOWETH M. S., HILL A. M. 2008. Diversity and distribution of freshwater gastropods in the Bayou Bartholomew drainage, Arkansas, U.S.A. Am. Malacol. Bull. 26: 171–177. http://dx.doi.org/10.4003/006.026.0217

MIREK Z., PIĘKOŚ-MIRKOWA H., ZAJĄC A., ZAJĄC M. 2002. Flowering plants and pteridophytes of Poland. A checklist. Krytyczna lista roślin naczyniowych. W. Szafer Institute of Botany, Polish Academy of Sciences, Kraków.

MOUTHON J. 1996a. Molluscs and biodegradable pollution in rivers: proposal for scale of sensitivity of species. Hydrobiologia 317: 221–229. http://dx.doi.org/10.1007/BF00036472

MOUTHON J. 1996b. Molluscs and biodegradable pollution in rivers: studies into the limiting values of 11 physico-chemical variables. Hydrobiologia 319: 57–63. http://dx.doi.org/10.1007/BF00020971

MOUTHON J. 2007. *Lithoglyphus naticoides* (Pfeiffer) Gastropoda: Prosobranchia): distribution in France, population dynamics and life cycle in the Saône river at Lyon (France). Ann. Limnol. – Int. J. Lim. 43: 53–59. http://dx.doi.org/10.1051/limn/2007027

MOUTHON J., CHARVET S. 1999. Compared sensitivity of species, genera and families of Molluscs to biodegradable pollution. Ann. Limnol. – Int. J. Lim. 35: 31–39. http://dx.doi.org/10.1051/limn/1999009

MOUTHON J., DAUFRESNE M. 2008. Population dynamics and life cycle of *Pisidium amnicum* (Müller) (Bivalvia: Sphaeriidae) and *Valvata piscinalis* (Müller) (Gastropoda: Prosobranchia) in the Saône river, a nine-year study. Ann. Limnol. – Int. J. Lim. 44: 241–251. http://dx.doi.org/10.1051/limn:2008008

MRÓZ W. 2012. Monitoring siedlisk przyrodniczych. Przewodnik metodyczny. Część II. GIOŚ, Warszawa.

MYŚLIŃSKA E. 2001. Grunty organiczne i laboratoryjne metody ich badania. Wydawnictwo Naukowe PWN, Warszawa.

NAIMO T. J. 1995. A review of the effects of heavy metals on freshwater mussels. Ecotoxicology 4: 341–362. http://dx.doi.org/10.1007/BF00118870

NALEPA T. F., GARDNER W. S., MALCZYK J. M. 1991. Phosphorus cycling by mussels (Unionidae: Bivalvia) in Lake St. Clair. Hydrobiologia 219: 239–250. http://dx.doi.org/10.1007/BF00024758

NEAL C., JARVIE H. P., WITHERS P. J. A., WHITTON B. A., NEAL M. 2010. The strategic significance of wastewater sources to pollutant phosphorus levels in English rivers and to environmental management for rural, agricultural and urban catchments. Sci. Total Environ. 408: 1485–1500. http://dx.doi.org/10.1016/j.scitotenv.2009.12.020

NIJBOER R., VERDONSCHOT P., PIECHOCKI A., TOŃCZYK G., KLUKOWSKA M. 2006. Characterisation of pristine Polish river systems and their use as reference conditions for Dutch river systems. Wageningen, Alterra, Alterra-rapport 1367.

OBOLEWSKI K. 2005. Epiphytic macrofauna on water soldiers (*Stratiotes aloides* L.) in Słupia River oxbows. Oceanol. Hydrobiol. St. 34: 37–54.

OBOLEWSKI K. 2011a. Macrozoobenthos patterns along environmental gradients and hydrological connectivity of oxbow lakes. Ecol. Eng. 37: 796–805. http://dx.doi.org/10.1016/j.ecoleng.2010.06.037

OBOLEWSKI K. 2011b. Functioning of aquatic invertebrate communities in oxbow lakes with various connection to riverbed. In: GLIŃSKA-LEWCZUK K. (ed.). Issues of Landscape Conservation and Water Management in Rural Areas. Contemporary Problems of Management and Environmental Protection, No. 7, pp. 249–265.

OBOLEWSKI K., GLIŃSKA-LEWCZUK K. 2011. Effects of oxbow reconnection based on the distribution and structure of benthic macroinvertebrates. Clean – Soil, Air, Water 39: 853–862. http://dx.doi.org/10.1002/clen.201000491

OBOLEWSKI K., GLIŃSKA-LEWCZUK K., KOBUS S. 2009. Effect of hydrological connectivity on the molluscan community structure in oxbow lakes of the Łyna River. Oceanol. Hydrobiol. St. 38: 75–88. http://dx.doi.org/10.2478/v10009-009-0045-1

OBOLEWSKI K., STRZELCZAK A., GLIŃSKA-LEWCZUK K. 2013. Does hydrological connectivity affect the composition of macroinvertebrates on *Stratiotes aloides* L. in oxbow lakes? Ecol. Eng. http://dx.doi.org/10.1016/j.ecoleng.2013.05.009

OERTLI B., AUDERSET JOYE D., CASTELLA E., JUGE R., CAMBIN D., LACHAVANNE J. B. 2002. Does size matter?

The relationship between pond area and biodiversity. Biol. Conserv. 104: 59–70. http://dx.doi.org/10.1016/S0006-3207(01)00154-9

O'Hare M. T., Clarke R. T., Bowes M. J., Cailes C., Henville P., Bissett N., McGahey C., Neal M. 2010. Eutrophication impacts on a river macrophyte. Aquat. Bot. 92: 173–178. http://dx.doi.org/10.1016/j.aquabot.2009.11.001

Oheimb P. V. von, Heiler K. C. M., Albrecht C. 2007. The mollusc fauna of a degraded European lowland river system: the lower Hawel (Brandenburg, Germany). Mollusca 25: 153–163.

Ożgo M., Abraszewska A. 2009. The importance of peat excavation water bodies for biodiversity and conservation: a case of three unionidae (Bivalvia) mussel species. Pol. J. Ecol. 57: 793–798.

Økland J. 1983. Factors regulating the distribution of fresh-water snails (Gastropoda) in Norway. Malacologia 24: 227–288.

Økland J. 1990. Lakes and snails. Environment of Gastropoda in 1,500 Norwegian lakes, ponds and rivers. Universal Book Services/Dr. W. Backhuys, Oestgeest, Norway.

Pakulnicka J., Nowakowski J. J. 2012. The effect of hydrological connectivity on water beetles fauna in water bodies within the floodplain of lowland river (Neman river, Belarus). Oceanol. Hydrobiol. St. 41: 7–17. http://dx.doi.org/10.2478/s13545-012-0012-4

Pan B. Z., Wang Z. Y., Xu M. Z. 2012. Macroinvertebrates in abandoned channels: assemblage characteristics and their indications for channel management. River Res. Appl. 28: 1149–1160. http://dx.doi.org/10.1002/rra.1515

Parr L. B., Mason C. F. 2003. Long-term trends in water quality and their impact on macroinvertebrate assemblages in eutrophic lowland rivers. Water Res. 37: 2969–2979. http://dx.doi.org/10.1016/S0043-1354(03)00115-5

Pastuszak M., Stålnacke P., Pawlikowski K., Witek Z. 2012. Response of Polish rivers (Vistula, Oder) to reduced pressure from point sources and agriculture during transition period (1988–2008). J. Marine Syst. 94: 157–173. http://dx.doi.org/10.1016/j.jmarsys.2011.11.017

Pasztaleniec A., Poniewozik M. 2013. The impact of free-floating plant cover on phytoplankton assemblages of oxbow lakes (The Bug River Valley, Poland). Biologia 68: 18–29. http://dx.doi.org/10.2478/s11756-012-0129-3

Pasztaleniec A., Poniewozik M., Solis M., Wojciechowska W. 2006. Hydrobiologiczna charakterystyka jezior rzecznych. In: Wojciechowska W. (ed.). Jeziora rzeczne doliny środkowego Bugu. Różnorodność biologiczna i krajobrazowa. Wydawnictwo KUL, Lublin, pp. 35–53.

Peeters E. T. H. M., Gylstra R., Vos J. H. 2004. Benthic macroinvertebrate community structure in relation to food and environmental variables. Hydrobiologia 519: 103–115. http://dx.doi.org/10.1023/B:HYDR.0000026497.48827.70

Penczak T., Galicka W., Głowacki Ł., Koszaliński H., Kruk A., Zięba G., Kostrzewa J., Marszał L. 2004. Fish assemblage changes relative to environmental factors and time in the Warta River, Poland, and its oxbow lakes. J. Fish Biol. 64: 483–501. http://dx.doi.org/10.1111/j.0022-1112.2004.00316.x

Pérez-Quintero J. C. 2007. Diversity, habitat use and conservation of freshwater molluscs in the lower Guadiana River basin (SW Iberian Peninsula). Aquat. Conserv. 17: 485–501. http://dx.doi.org/10.1002/aqc.796

Pérez-Quintero J. C. 2011. Distribution patterns of freshwater molluscs along environmental gradients in the southern Guadiana River basin (SW Iberian Peninsula). Hydrobiologia 678: 65–76. http://dx.doi.org/10.1007/s10750-011-0821-2

Piechocki A. 1969. Mięczaki rzeki Grabi i jej terenu zalewowego. Fragm. Faun. 15: 111–197. http://dx.doi.org/10.3161/00159301FF1969.15.10.111

Piechocki A. 1979. Mięczaki (Mollusca). Ślimaki (Gastropoda). Fauna Słodkowodna Polski, Zeszyt 7. Państwowe Wydawnictwo Naukowe, Warszawa, Poznań.

Piechocki A. 1987. Mięczaki (Mollusca) rzek lessowego obrzeżenia Gór Świętokrzyskich. Fragm. Faun. 31: 169–181.

Piechocki A. 1992. The Polish Sphaeriidae (Bivalvia, Eulamellibranchia) and their ecological references. Proceedings of the 9th International Malacological Congress, Edinburgh: 235–239.

Piechocki A. 1996. Present state of knowledge on the Polish freshwater molluscan fauna. Aktualny stan poznania mięczaków słodkowodnych. Acta Univ. Lodz. Folia limnol. 6: 3–17.

Piechocki A. 2002. Gastropoda aquatica. Ślimaki wodne. In: Głowaciński Z. (ed.). Czerwona Lista Zwierząt Ginących i Zagrożonych w Polsce. Red List of Threatened Animals in Poland. Polish Academy of Sciences, Institute of Nature Conservation, Cracow, pp. 34–37.

Piechocki A. 2004a. *Borysthenia naticina* (Menke, 1845). In: Głowaciński Z., Nowacki J. (eds). Polska czerwona księga zwierząt. Bezkręgowce. Polish Red Data Book of Animals. Invertebrates. Institute of Nature Conservation PAS, Cracow, The August Cieszkowski Agricultural University of Poznań, pp. 317–318.

Piechocki A. 2004b. *Lithoglyphus naticoides* (C. Pfeiffer, 1828). In: Głowaciński Z., Nowacki J. (eds). Polska czerwona księga zwierząt. Bezkręgowce. Polish Red Data Book of Animals. Invertebrates. Institute of Nature Conservation PAS, Cracow, The August Cieszkowski Agricultural University of Poznań, pp. 318–319.

Piechocki A. 2008. Mięczaki Mollusca. Characteristic and checklist of species. In: Bogdanowicz W., Chudzicka E., Pilipiuk I., Skibińska E. (eds). Fauna of Poland. Volume 3. Muzeum i Instytut Zoologii PAN, Warszawa, pp. 365–425.

Piechocki A., Dyduch-Falniowska A. 1993. Mięczaki (Mollusca). Małże (Bivalvia). Fauna Słodkowodna Polski 7A. Wydawnictwo Naukowe PWN, Warszawa.

PIECHOCKI A., STRZELEC M. 1999. Ecology of Sphaeriidae (Bivalvia: Heterodonta) in sandy bottom section of the Grabia River (C-Poland). Malacol. Rev. Suppl. 8: 37–58.

PIECHOCKI A., SZLAUER-ŁUKASZEWSKA A. 2013. Molluscs of the middle and lower Odra: the role of the river in the expansion of alien species in Poland. Folia Malacol. 21: 73–86. http://dx.doi.org/10.12657/folmal.021.008

PIELECH R., KISIEL P. 2010. Raport. Prace utrzymaniowe jako zagrożenie dla osiągnięcia środowiskowych celów Ramowej Dyrektywy Wodnej oraz funkcjonowania sieci ekologicznej Natura 2000 w Polsce.

PIRES A. M., COWX I. G., COELHO M. M. 2000. Benthic macroinvertebrate communities of intermittent streams in the middle reaches of the Guadiana Basin (Portugal). Hydrobiologia 435: 167–175. http://dx.doi.org/10.1023/A:1004003726283

PODBIELKOWSKI Z., TOMASZEWICZ H. 1996. Zarys hydrobotaniki. Wydawnictwo Naukowe PWN, Warszawa.

PROBST M., BERENZEN N., LENTZEN-GODDING A., SCHULZ R., LIESS M. 2005. Linking land use variables and invertebrate taxon richness in small and medium-sized agricultural streams on a landscape level. Ecotox. Environ. Safe. 60: 140–146. http://dx.doi.org/10.1016/j.ecoenv.2004.04.003

PROĆKÓW J., PROĆKÓW M. 2008. Plant cover of the "Łacha Farna Old River-bed" area of ecological use (NW Wrocław, Dolnośląskie voivodeship). Acta Bot. Siles. 3: 83–105.

RACZYŃSKA M., CHOJNACKI J. C. 2009. The structure of macrozoobenthic communities in the Tywa River, a right-bank tributary of the Oder River (northwest Poland). Oceanol. Hydrobiol. St. 38: 31–42. http://dx.doi.org/10.2478/v10009-009-0029-1

RAVERA O., RICCARDI N. 1997. Biological monitoring with organisms accumulator of pollutants. Mar. Chem. 58: 313–318. http://dx.doi.org/10.1016/S0304-4203(97)00057-1

RECKENDORFER W., BARANYI C., FUNK A., SCHIEMER F. 2006. Floodplain restoration by reinforcing hydrological connectivity: expected effects on aquatic mollusc communities. J. Appl. Ecol. 43: 474–484. http://dx.doi.org/10.1111/j.1365-2664.2006.01155.x

REPORT 2005. Raport dla Obszaru Dorzecza Wisły z realizacji art. 5 i 6, zał. II, III, IV Ramowej Dyrektywy Wodnej 2000/60/We. Rzeczpospolita Polska, Ministerstwo Środowiska, Warszawa.

REPORT 2007. Stan środowiska w województwie mazowieckim w 2006 roku. The Voivodeship Inspectorate for Environment Protection in Warsaw.

REPORT 2009. Stan środowiska w województwie mazowieckim w 2008 roku. The Voivodeship Inspectorate for Environment Protection in Warsaw.

REPORT 2011. Stan środowiska w województwie mazowieckim w 2010 roku. The Voivodeship Inspectorate for Environment Protection in Warsaw.

REPORT EU 2011. Report from the Commission to the Council and the European Parliament on implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2004–2007.

RISTAU K., FAUPEL M., TRAUNSPURGER W. 2012. The effects of nutrient enrichment on a freshwater meiofaunal assemblage. Freshwater Biol. 57: 824–834. http://dx.doi.org/10.1111/j.1365-2427.2012.02750.x

ROLAUFFS P., STUBAUER I., ZAHRÁDKOVÁ S., BRABEC K., MOOG O. 2004. Integration of the saprobic system into the European Union Water Framework Directive. Case studies in Austria, Germany and Czech Republic. Hydrobiologia 516: 285–298. http://dx.doi.org/10.1023/B:HYDR.0000025271.90133.4d

RUSANOV A. G., STANISLAVSKAYA E. V., ÁCS É. 2012. Periphytic algal assemblages along environmental gradients in the river of the Lake Ladoga basin, Northwestern Russia: implication for the water quality assessment. Hydrobiologia 695: 305–327. http://dx.doi.org/10.1007/s10750-012-1199-5

SAVAGE A. A., GAZEY G. M. 1987. Relationships of physical and chemical conditions to species diversity and density of gastropods in English Lakes. Biol. Conserv. 42: 95–113. http://dx.doi.org/10.1016/0006-3207(87)90017-6

SAYER C., ANDREWS K., SHILLAND E., EDMONDS N., EDMONDS-BROWN R., PATMORE J., EMSON D., AXMACHER J. 2012. The role of pond management for biodiversity conservation in an agricultural landscape. Aquat. Conserv. 22: 626–638. http://dx.doi.org/10.1002/aqc.2254

SCHNEIDER K. N., WINEMILLER K. O. 2008. Structural complexity of woody debris patches influences fish and macroinvertebrate species richness in temperate floodplain-river system. Hydrobiologia 610: 235–244. http://dx.doi.org/10.1007/s10750-008-9438-5

SHIEH S. H., KONDRATIEFF B. C., WARD J. V., RICE D. A. 1999. The relationship of macroinvertebrate assemblages to water chemistry in a polluted Colorado plains stream. Arch. Hydrobiol. 145: 405–432.

SHUHAIMI-OTHMAN M., NUR-AMALINA R., NADZIFAH Y. 2012. Toxicity of metals to a freshwater snail, *Melanoides tuberculata*. Sci. World J. http://dx.doi.org/10.1100/2012/125785

SOUCEK D. J., DICKINSON A. 2012. Acute toxicity of nitrate and nitrite to sensitive freshwater insects, mollusks, and a crustacean. Arch. Environ. Con. Tox. 62: 233–242. http://dx.doi.org/10.1007/s00244-011-9705-8

SOUSA R., ANTUNES C., GUILHERMINO L. 2007. Species composition and monthly variation of the molluscan fauna in the freshwater subtidal area of the River Minho estuary. Estuar. Coast. Shelf S. 75: 90–100. http://dx.doi.org/10.1016/j.ecss.2007.02.020

SOUSA R., ILARRI M., SOUZA A. T., ANTUNES C., GUILHERMINO L. 2011. Rapid decline of the greater European peaclam at the periphery of its distribution. Ann. Limnol. – Int. J. Lim. 47: 211–219. http://dx.doi.org/10.1051/limn/2011041

SOUSA R., MORAIS P., ANTUNES C., GUILHERMINO L. 2008. Factors affecting *Pisidium amnicum* (Müller, 1774; Bivalvia: Sphaeriidae) distribution in the River Minho estuary: consequences for its conservation. Estuar.

Coast. 31: 1198–1207. http://dx.doi.org/10.1007/s12237-008-9090-3

Spałek K. 2008a. *Nymphaeetum candidae* Miljan 1933 in Poland. Acta Soc. Bot. Pol. 77: 339–343. http://dx.doi.org/10.5586/asbp.2008.044

Spałek K. 2008b. *Salvinia natans* (L.) All. in fishponds and oxbow lakes in Lower and Opole Silesia (SW Poland). In: Szczęśniak E., Gola E. (eds). Club mosses, horsetails and ferns in Poland – resources and protection. Polish Botanical Society and Institute of Plant Biology, University of Wrocław, Wrocław, pp. 147–160.

Stańczykowska A. 1959. Rozmieszczenie i dynamika liczebności żyworódki paskowanej *Viviparus fasciatus* Müll. na terenie łachy Konfederatka. Ekol. Pol. B 5: 55–60.

Stańczykowska A. 1960a. Rozmieszczenie i dynamika liczebności mięczaków dennych na łasze wiślanej Konfederatka pod Wyszogrodem. Ekol. Pol. A 8: 155–168.

Stańczykowska A. 1960b. Obserwacje nad skupieniami *Viviparus fasciatus* Müll. na terenie łachy wiślanej Konfederatka. Ekol. Pol. A 8: 21–48.

Stańczykowska A. 1960c. Charakter występowania mięczaków na kilku gatunkach roślin wodnych. Ekol. Pol. B 6: 333–338.

Stańczykowska A. 1983. Małże a eutrofizacja wód. Wiad. Ekol. 29: 127–129.

Statistical Yearbook of Agriculture 2011. Central Statistical Office. Statistical Publishing Establishment, Warsaw.

Strayer D. L. 2013. Understanding how nutrient cycles and freshwater mussels (Unionoida) affect one another. Hydrobiologia. http://dx.doi.org/10.1007/s10750-013-1461-5

Strong E. E., Gargominy O., Ponder W. F., Bouchet P. 2008. Global diversity of gastropods (Gastropoda; Mollusca) in freshwater. Hydrobiologia 595:149–166. http://dx.doi.org/10.1007/s10750-007-9012-6

Strzałek M., Koperski P. 2009. The *Stratiotes aloides* L. stand as a habitat in oxbow lake Bużysko. Aquat. Bot. 90: 1–6. http://dx.doi.org/10.1016/j.aquabot.2008.04.010

Strzelec M. 1993. Ślimaki (Gastropoda) antropogenicznych środowisk wodnych Wyżyny Śląskiej. Prace Naukowe Uniwersytetu Śląskiego w Katowicach nr 1358, Uniwersytet Śląski, Katowice, pp. 1–104.

Strzelec M., Królczyk A. 2004. Factors affecting snail (Gastropoda) community structure in the upper course of the Warta river (Poland). Biologia, Bratislava 59: 159–163.

Suren A. M., Lambert P., Image K., Sorrell B. K. 2008. Variation in wetland invertebrate communities in lowland acidic fens and swamps. Freshwater Biol. 53: 727–744. http://dx.doi.org/10.1111/j.1365-2427.2007.01931.x

Szafer W., Kulczyński S., Pawłowski B. 1986. Rośliny Polskie. Państwowe Wydawnictwo Naukowe, Warszawa.

Szczęśniak E., Błachuta J., Krukowski M., Picińska-Fałtynowicz J. 2009. Distribution of *Azolla filiculoides* Lam. (Azollaceae) in Poland. Acta Soc. Bot. Pol. 78: 241–246. http://dx.doi.org/10.5586/asbp.2009.031

Szmeja J. 2006. Przewodnik do badań roślinności wodnej. Wydawnictwo Uniwersytetu Gdańskiego, Gdańsk.

Ter Braak C. J. F., Šmilauer P. 2002. CANOCO Reference manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 4.5), 2nd ed., Microcomputer Power, Ithaca, New York.

Tessier L., Vaillancourt G., Pazdernik L. 1996. Laboratory study of Cd and Hg uptake by two freshwater molluscs in relation to concentration, age and exposure time. Water Air Soil Poll. 86: 347–357. http://dx.doi.org/10.1007/BF00279166

The National Environmental Policy 2008. The National Environmental Policy for 2009–2012 and its 2016 outlook. Council of Ministers Republic of Poland, Warsaw.

Thomas J. D., Kowalczyk C. 1997. Utilization of dissolved organic matter (DOM), from living macrophytes, by pulmonate snails: implications to the "food web" and "module" concepts. Comp. Biochem. Phys. A 117: 105–119. http://dx.doi.org/10.1016/S0300-9629(96)00377-5

Tockner K., Malard F., Ward J. V. 2000. An extension of the flood pulse concept. Hydrol. Process. 14: 2861–2883.

Tockner K., Schiemer F., Baumgartner C., Kum G., Weigand E., Zweimüller I., Ward J. V. 1999. The Danube restoration project: species diversity pattern across connectivity gradients in the floodplain system. Regul. Rivers: Res. Mgmt. 15: 245–258.

Townsend C. R., Hildrew A. G., Francis J. 1983. Community structure in some southern English streams: the influence of physicochemical factors. Freshwater Biol. 13: 521–544. http://dx.doi.org/10.1111/j.1365-2427.1983.tb00011.x

Tsikhon-Lukanina E. A., Reznichenko O. G., Lukasheva T. A. 1998. Diet composition and food diversity in marine and freshwater gastropods. Zool. Zh. 77: 270–277.

Tubić B. P., Simić V. M., Zorić K. S., Gačić Z. M., Atanacković A. D., Csányi B. J., Paunović M. M. 2013. Stream section types of the Danube River in Serbia according to the distribution of macroinvertebrates. Biologia 68: 294–302. http://dx.doi.org/10.2478/s11756-013-0152-z

Ulén B., Bechmann M., Fölster J., Jarvie H. P., Tunney H. 2007. Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review. Soil Use Manage. 23: 5–15. http://dx.doi.org/10.1111/j.1475-2743.2007.00115.x

Urban D., Wójciak H. 2004. Water and rush plant associations of the Bug River valley old river bed (Kryłów-Kodeń section) vis-a-vis the habitat conditions. Teka Kom. Ochr. Kszt. Środ. Przyr. 1: 293–300.

Vaughn C. C. 2012. Life history traits and abundance can predict local colonisation and extinction rates of freshwater mussels. Freshwater Biol. 57: 982–992. http://dx.doi.org/10.1111/j.1365-2427.2012.02759.x

Vaughn C. C., Hakenkamp C. C. 2001. The functional role of burrowing bivalves in freshwater ecosystems. Freshwater Biol. 46: 1431–1446. http://dx.doi.org/10.1046/j.1365-2427.2001.00771.x

VERBRUGGE L. N. H., SCHIPPER A. M., HUIJBREGTS M. A. J., VAN DER VELDE G., LEUVEN R. S. E. W. 2012. Sensitivity of native and non-native mollusc species to changing river water temperature and salinity. Biol. Invasions 14: 1187–1199. http://dx.doi.org/10.1007/s10530-011-0148-y

VERDONSCHOT P. F. M. 2001. Hydrology and substrates: determinants of oligochaete distribution in lowland streams (The Netherlands). Hydrobiologia 463: 249–262. http://dx.doi.org/10.1023/A:1013132514610

VIRBICKAS T., PLIŪRAITĖ V., KESMINAS V. 2011. Impact of agriculture land use on macroinvertebrate fauna in Lithuania. Pol. J. Environ. Stud. 20: 1327–1334.

WAYKAR B., DESHMUKH G. 2012. Evaluation of bivalves as bioindicators of metal pollution in freshwater. B. Environ. Contam. Tox. 88: 48–53. http://dx.doi.org/10.1007/s00128-011-0447-0

WEBER E. 2005. Population size and structure of three mussel species (Bivalvia: Unionidae) in a northeastern German river with special regard to influences of environmental factors. Hydrobiologia 537: 169–183. http://dx.doi.org/10.1007/s10750-004-2839-1

WILLIAMS P., WHITFIELD M., BIGGS J., BRAY S., FOXA G., NICOLET P., SEAR D. 2003. Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. Biol. Conserv. 115: 329–341. http://dx.doi.org/10.1016/S0006-3207(03)00153-8

WŁOSIK-BIEŃCZAK E. 1997. Funkcje i zachowanie mięczaków w zbiornikach wodnych. The function and behaviour of molluscs in water reservoirs. Idee Ekologiczne 10: 157–169.

WOOD P. J., GREENWOOD M. T., AGNEW M. D. 2003. Pond biodiversity and habitat loss in the UK. Area 35: 206–216. http://dx.doi.org/10.1111/1475-4762.00249

WOODWARD G., GESSNER M. O., GILLER P. S., GULIS V., HLADYZ S., LECERF A., MALMQVIST B., MCKIE B. G., TIEGS S. D., CARISS H., DOBSON M., ELOSEGI A., FERREIRA V., GRAÇA M. A. S., FLEITUCH T., LACOURSIÈRE J. O., NISTORESCU M., POZO J., RISNOVEANU G., SCHINDLER M., VADINEANU A., VOUGHT L. B. M., CHAUVET E. 2012. Continental-scale effects of nutrient pollution on stream ecosystem functioning. Science 336: 1438–1440. http://dx.doi.org/10.1126/science.1219534

WÓJCIAK H., URBAN D. 2009. Rzęsowate (Lemnaceae) i ich fitocenozy w starorzeczach Bugu na odcinku Kryłów-Kostomłoty. Woda-Środowisko-Obszary Wiejskie 9: 215–225.

ZABLOTOWICZ R. M., ZIMBA P. V., LOCKE M. A., KNIGHT S. S., LIZOTTE JR. R. E., GORDON R. E. 2010. Effects of land management practices on water quality in Mississippi Delta oxbow lakes: Biochemical and microbiological aspects. Agr. Ecosyst. Environ. 139: 214–223. http://dx.doi.org/10.1016/j.agee.2010.08.005

ZAJĄC K. 2004a. *Pseudanodonta complanata* (Rossmässler, 1835). In: GŁOWACIŃSKI Z., NOWACKI J. (eds). Polska czerwona księga zwierząt. Bezkręgowce. Polish Red Data Book of Animals. Invertebrates. Institute of Nature Conservation PAS, Cracow, The August Cieszkowski Agricultural University of Poznań, pp. 351–353.

ZAJĄC K. 2004b. *Unio crassus* Philipsson, 1788. In: GŁOWACIŃSKI Z., NOWACKI J. (eds). Polska czerwona księga zwierząt. Bezkręgowce. Polish Red Data Book of Animals. Invertebrates. Institute of Nature Conservation PAS, Cracow, The August Cieszkowski Agricultural University of Poznań, pp. 353–355.

ZAJĄC K. 2004c. *Anodonta cygnea* (Linnaeus, 1758). In: GŁOWACIŃSKI Z., NOWACKI J. (eds). Polska czerwona księga zwierząt. Bezkręgowce. Polish Red Data Book of Animals. Invertebrates. Institute of Nature Conservation PAS, Cracow, The August Cieszkowski Agricultural University of Poznań, pp. 349–351.

ZAJĄC K., ZAJĄC T. 2011. The role of active individual movement in habitat selection in the endangered freshwater mussel *Unio crassus* Philipsson 1788. J. Conchol. 40: 446–461.

ZETTLER M. L. 2000. Weitere Bemerkungen zur Morphologie von *Unio crassus* Philipsson 1788 aus dem nordeuropäischen Vereisungsgebiet (Bivalvia: Unionidae). Malak. Abh. 20: 73–78.

ZETTLER M. L. 2012. A remarkable record of a very rare freshwater snail *Borysthenia naticina* (Menke, 1845) in north-east Germany compared with three Lithuanian records. Folia Malacol. 20: 105–110. http://dx.doi.org/10.2478/v10125-012-0016-0

ZETTLER M. L., JUEG U. 2007. The situation of the freshwater mussel *Unio crassus* (Philipsson, 1788) in north-east Germany and its monitoring in terms of the EC Habitat Directive. Mollusca 25: 165–174.

ZIELIŃSKI P., GÓRNIAK A., PIEKARSKI M. K. 2009. The effect of hydrological drought on chemical quality of water and dissolved organic carbon concentrations in lowland rivers. Pol. J. Ecol. 57: 217–22.

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