TRANSYLVANIAN REVIEW OF SYSTEMATICAL AND ECOLOGICAL RESEARCH

18.2

The Wetlands Diversity

Editors

Angela Curtean-Bănăduc & Doru Bănăduc

Sibiu - Romania
2016

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Published based mainly on some of the scientific materials presented
at the 41st International Association for Danube Research Conference
"Tributaries as Key Elements in Sustainable Management of the Danube River Basin" -
Sibiu/Romania 13-16 September 2016

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IN MEMORIAM

Francis Crick
(1916 – 2004)

Francis Crick was a British molecular biologist, biophysicist, and neuroscientist, born and raised in Weston Favell, a small village near Northampton, in which Crick's father ran a boot and shoe factory.

Walter Crick, his grandfather, was an amateur naturalist. He published a review of local foraminifera, corresponded with Charles Darwin, and had two gastropods named after him.

From childhood, Francis was fascinated by science and by what he could learn about it. , His uncle, also Walter Crick, lived in a small residence in Abington Avenue, near his parent's home; Walter had a shed in his small garden where he instructed Francis do chemical experiments, blow glass, and to make photographic prints. When he was eight Francis moved to Northampton Grammar School. His teacher, Miss Holding, was an inspiring teacher and made all subjects attractive. After 14 years old, he won a scholarship to attend Mill Hill School in London, where he studied physics, mathematics, and chemistry. He shared the Walter Knox Prize for Chemistry in 1933 on Mill Hill School's Foundation Day 7 July, and he asserted that his achievement was stimulated by the quality of teaching he has enjoyed at Mill Hill.

At the age of 21, Crick earned a Bachelor of Science degree in physics from University College London (UCL). He began a PhD at UCL but was interrupted by World War II. He later became a PhD student and Honorary Fellow of Gonville and Caius College, Cambridge and for the most part worked at the Cavendish Laboratory and the Medical Research Council Laboratory of Molecular Biology in Cambridge. Crick was also an Honorary Fellow of Churchill College Cambridge and of University College London.

Crick began his PhD research project on the viscosity of water at high temperatures at University College London. During the second year of his PhD, he was awarded the Carey Foster Research Prize. He did postdoctoral work at the Polytechnic Institute of Brooklyn.

During World War II, he worked for the Admiralty Research Laboratory, from which emerged many notable scientists. He designed magnetic and acoustic mines.

In 1947, Crick started studying biology and was a part of an important migration of physicists into the area of biological research. Crick worked first on the physical properties of cytoplasm at the Strangeways Research Laboratory and Cavendish Laboratory, Cambridge.

Crick was interested mainly in fundamental problems of biology: how molecules make the transition from the non-living to the living, how the brain makes a conscious mind, the origin of life, etc.

He was most famous for being a co-discoverer of the structure of the DNA molecule in 1953 together with James Watson. In addition to his one-third share of the 1962 Nobel Prize, he received many honours and awards, among other things the Royal and Copley Medals of the Royal Society (1972 and 1975), and the Order of Merit (27 November 1991).

He married twice, fathered three children and was grandfather of six grandchildren.

Crick died of colon cancer on the morning of 28 July 2004, he was cremated and his ashes were scattered into the Pacific Ocean.

The Francis Crick Institute biomedical research centre located in north London, United Kingdom, is one of the largest such centres in Europe today.

The Editors

Transylvanian Review of Systematical and Ecological Research

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Preface

In a global environment in which the climate changes are observed from few decades no more only through scientific studies but also through day by day life experiences of average people which feel and understand allready the presence of the medium and long-term significant change in the “average weather” all over the world, the most comon key words which reflect the general concern are: heating, desertification, rationalisation and surviwing.

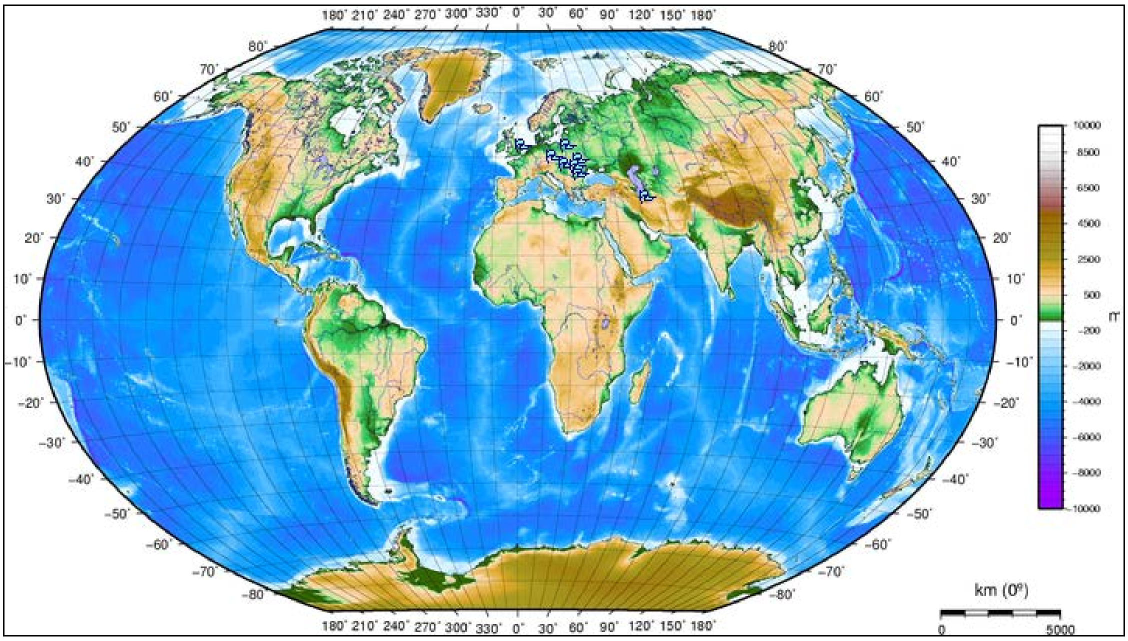
The causes, effects, trends and possibilities of human society to positively intervene to slow down this process or to adapt to it involve a huge variety of aproacess and efforts.

With the fact in mind that these aproaces and efforts shuld be based on genuine scientific understanding, the editors of the *Transylvanian Review of Systematical and Ecological Research* series launch a second annual volumes dedicated to the wetlands, volumes resulted mainly as a results of the *Aquatic Biodiversity International Conference*, Sibiu/Romania, 2007-2015 and The 41st International Association for Danube Research Cnference, Sibiu/Romania, 2016.

The therm wetland is used here in the acceptance of the Convention on Wetlands, signed in Ramsar, in 1971, for the conservation and wise use of wetlands and their resources. **Marine/Coastal Wetlands -** Permanent shallow marine waters in most cases less than six metres deep at low tide, includes sea bays and straits; Marine subtidal aquatic beds, includes kelp beds, sea-grass beds, tropical marine meadows; Coral reefs; Rocky marine shores, includes rocky offshore islands, sea cliffs; Sand, shingle or pebble shores, includes sand bars, spits and sandy islets, includes dune systems and humid dune slacks; Estuarine waters, permanent water of estuaries and estuarine systems of deltas; Intertidal mud, sand or salt flats; Intertidal marshes, includes salt marshes, salt meadows, saltings, raised salt marshes, includes tidal brackish and freshwater marshes; Intertidal forested wetlands, includes mangrove swamps, nipah swamps and tidal freshwater swamp forests; Coastal brackish/saline lagoons, brackish to saline lagoons with at least one relatively narrow connection to the sea; Coastal freshwater lagoons, includes freshwater delta lagoons; Karst and other subterranean hydrological systems, marine/coastal. **Inland Wetlands -** Permanent inland deltas; Permanent rivers/streams/creeks, includes waterfalls; Seasonal/intermittent/irregular rivers/streams/creeks; Permanent freshwater lakes (over eight ha), includes large oxbow lakes; Seasonal/intermittent freshwater lakes (over eight ha), includes floodplain lakes; Permanent saline/brackish/alkaline lakes; Seasonal/intermittent saline/brackish/alkaline lakes and flats; Permanent saline/brackish/alkaline marshes/pools; Seasonal/intermittent saline/brackish/alkaline marshes/pools; Permanent freshwater marshes/pools, ponds (below eight ha), marshes and swamps on inorganic soils, with emergent vegetation water-logged for at least most of the growing season; Seasonal/intermittent freshwater marshes/pools on inorganic soils, includes sloughs, potholes, seasonally flooded meadows, sedge marshes; Non-forested peatlands, includes shrub or open bogs, swamps, fens; Alpine wetlands, includes alpine meadows, temporary waters from snowmelt; Tundra wetlands, includes tundra pools, temporary waters from snowmelt; Shrub-dominated wetlands, shrub swamps, shrub-dominated freshwater marshes, shrub carr, alder thicket on inorganic soils; Freshwater, tree-dominated wetlands; includes freshwater swamp forests, seasonally flooded forests, wooded swamps on inorganic soils; Forested peatlands; peatswamp forests; Freshwater springs, oases; Geothermal wetlands; Karst and other subterranean hydrological systems, inland. **Human-made wetlands -** Aquaculture (e. g., fish/shrimp) ponds; Ponds; includes farm ponds, stock ponds, small tanks; (generally below eight ha); Irrigated land, includes irrigation channels and rice fields; Seasonally flooded agricultural land (including intensively managed or grazed wet meadow or pasture); Salt exploitation sites, salt pans, salines, etc.; Water storage areas, reservoirs/barrages/dams/ impoundments (generally over eight ha); Excavations; gravel/brick/clay pits; borrow pits, mining pools; Wastewater treatment areas, sewage farms, settling ponds, oxidation basins, etc.; Canals and channels, ditches; Karst and other subterranean hydrological systems, human-made.

The editors of the *Transylvanian Review of Systematical and Ecological Research* started and continue this new annual sub-series (*Wetlands Diversity*) as an international scientific debate platform for the wetlands conservation, and not to take in the last moment, some last heavenly "images" of a perishing world …

This 18.1 volume included variated researches from diverse wetlands around the world.



The subject areas (⚑) for the published studies in this volume.

No doubt that this new data will develop knowledge and understanding of the ecological status of the wetlands and will continue to evolve.

Acknowledgements

The editors would like to express their sincere gratitude to the authors and the scientific reviewers whose work made the appearance of this volume possible.

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(ISSN-L 1841 – 7051; online ISSN 2344 – 3219)

A HISTORY OF PEBBLES AND SILT – FLUVIAL SEDIMENT TRANSPORT, HYDROPOWER AND TECHNICAL EXPERTISE AT THE AUSTRIAN DANUBE AND ITS TRIBUTARIES

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DOI: 10.1515/trser-2015-0083

KEYWORDS: hydropower, sediment regime, environmental history, technical experts, Danube tributaries.

ABSTRACT

The paper investigates experts' perceptions of hydropower, sediment regime, and their interaction in the 20th century with an environmental historical approach, based on various case studies at both the Danube River and one of its tributaries, and on a review of contemporary literature authored by engineers. Results show that questions of sediment continuity have engaged planners of hydropower plants since the advent of this technology, and decisions were at any time influenced by multiple interests (navigation, electricity demand, nature conservation). In such an intricate fluvial landscape, phenomena like reservoir sedimentation and riverbed incision can be approached as "legacies" of past technical interventions, which limit the options of current and future river management.

ZUSAMMENFASSUNG: Umwelthistorische Überlegungen zu Sedimenttransport, Wasserkraft und Expertenwissen an der österreichischen Donau und ihren Zubringern.

Der Beitrag untersucht Expertenwahrnehmung von Wasserkraft, Feststoffhaushalt und deren Wechselwirkung im Verlauf des 20. Jahrhunderts, basierend auf Fallstudien an der Donau und an einem ihrer Zubringer, sowie publizierten Arbeiten zeitgenössischer Ingenieure. Die Ergebnisse zeigen, dass sich letztere seit Beginn der Wasserkraftnutzung mit Fragen der Sedimentdurchgängigkeit beschäftigten, und dass Entscheidungen von vielfältigen Interessen (Schifffahrt, Strombedarf, Naturschutz) beinflusst wurden. In einer komplexen Flusslandschaft können Phänomene wie Speicherverlandung und Sohleintiefung als langfristige Nebenwirkungen ("legacies") vergangener technischer Eingriffe betrachtet werden, die den Rahmen für aktuelle wie auch zukünftige wasserwirtschaftliche und flussbauliche Möglichkeiten vorgeben.

REZUMAT: O istorie a pietrișului și nămolului – transportul sedimentelor fluviale, hidroenergia şi expertiza tehnică a Dunării şi a afluenților săi din sectorul austriac.

Lucrarea prezintă percepția specialiștilor în hidroenergie, regimul sedimentelor și interacțiunea lor în secolul 20, cu o abordare istorică a mediului, bazată pe diverse studii de caz, atât pentru fluviul Dunărea cât și pentru unul dintre afluenții săi, precum și o trecere în revistă a literaturii contemporane specializate semnată de ingineri. Rezultatele arată că problemele legate de continuitatea sedimentelor au determinat angajarea de planificatori de centrale hidroelectrice încă de la apariția acestei tehnologii, iar deciziile au fost, în orice moment, influențate de interese multiple (navigație, cererea de energie electrică, dconservarea naturii). Într-un astfel de peisaj fluvial complicat, fenomene precum sedimentarea din rezervor și incizia albiei fluviului pot fi abordate ca „moșteniri" ale intervențiilor tehnice anterioare, care limitează posibilitățile de gestionare curente și viitoare ale râului.

INTRODUCTION

Sediment transport plays a key role for sustainable management of the Danube River. In the upper part of its catchment, “hydromorphological” pressures have been identified as important obstacles for reaching the goals of the EU Water Framework Directive (2000/60/EC), and such pressures can be attributed – amongst other causes – to impoundment by hydropower plants (BMLFUW, 2010, 2015). However, a rivers’ morphology is not only linked with hydrological aspects, but also with its sediment regime, i.e. processes of erosion, transfer, and deposition of solid material, which is transported either at the riverbed (bedload) or in the liquid phase (suspended load) (Mangelsdorf et al., 1990). From the perspective of fluvial ecology, hydromorphology and sediment regime influence abiotic habitat conditions like flow velocities, temperature, and oxygen content (Jungwirth et al., 2003), are therefore important to observe in river restoration projects with the aim to improve the “ecological state” (Hajdu and Kelemen, 2009; Hohensinner and Jungwirth, 2016). Fluvial landscapes with little anthropogenic alterations are generally characterized by a “dynamic equilibrium” of erosion and deposition, which means that in the long term neither aggradation nor degradation of the riverbed occur (Habersack et al., 2012). However, in the Austrian part of the Danube catchment, interruptions of sediment continuity exist, which most likely affect the whole river basin. One of them, dams and weirs of hydropower plants, are a focus of this paper.

In Austria, hydropower plays an important economic role; it contributes to approximately two thirds of the country’s annual electricity generation, and one third comes from large run-of-river plants at the Danube (Wagner et al., 2015). The latter have been built in the second half of the 20th century, later than many of its tributaries where hydropower development started at the end of the 19th century. Dams and weirs of hydropower plants interact with the flow of water and sediments in a river, causing morphological changes like reservoir sedimentation upstream and riverbed incision downstream – or, more generally speaking, leading to a surplus or deficit of material in certain river sections (Brandt, 2000; Schoder, 2013; Habersack et al., 2013). This can have negative consequences for both society (e.g. increased flood risk) and ecosystems (e.g. desiccation of floodplains). Riverbed incision (or riverbed degradation – the two terms will henceforward be used synonymously) has become regarded as a challenge downstream Vienna, in one of the two remaining free-flowing sections of the Austrian Danube (cf. e.g. Klasz et al., 2016). Now the question arises of how to deal with this problem, especially considering that part of the adjacent floodplain is protected as a national park (Nationalpark Donau-Auen) since the prevention of a hydropower plant at this site is a debated issue among river engineers, navigation, and environmentalists (Schoder and Schmid, in press). As the Austrian part of the Danube has a long history of societal use (Schmid, 2013), no single cause of the imbalanced sediment regime can be discerned. It must rather be attributed to multiple interventions in the catchment (e.g. river regulation, torrent control, the construction of storage reservoirs, gravel mining, and land use change) and in the Danube itself. Fluvial landscapes have been transformed in massive river engineering projects in the 19th century, first and foremost by straightening channels and cutting off side arms. How these measures affected the riverbed has been discussed recently by Hohensinner and Jungwirth (2016), who also state that an overall picture of this “third dimension” is still missing, mostly due to a lack of data and sources. About the *additional* effects of hydropower, little research has been carried out so far, apart from the comparatively well-studied river section downstream of Vienna.

It is the aim of this paper to explore the interaction of hydropower and sediment regime at the Austrian Danube in the 20th century, from the viewpoint of technical experts, who decided how the river would function after its second big transformation, i.e. the transformation for its energetic use. Were the changes in sediment transport caused by and affecting hydropower really unexpected – or even unintended – by early planners and engineers, or rather consciously accepted with (technical) solutions already at hand? How do past socio-natural interactions affect the implementation of such solutions at present? These questions are addressed using an environmental historical approach. Environmental history studies the interaction of humans and the rest of nature in the past, reconstructing both environmental conditions and the way they were perceived and interpreted by contemporaries (Winiwarter and Knoll, 2007). Especially the last part of this definition is central to this paper, because environmental conditions are approached "through the eyes" of technical experts due to the chosen historical sources (cf. the next section). Studying changing modes of perception can teach important lessons about environmental change, because how humans perceive a river is decisive for how they intervene. Thus, also perception determines material outcomes. The basic assumption of this paper is that for river management, it makes a difference whether experts and decision makers look at water, energy, biota, or at sediments. By choosing such an approach, oriented rather towards social sciences and humanities, this contribution is in line with the general working program of the expert group on "Long-Term Socio-Ecological Research (LTSER) and Environmental History", recently established with the International Association for Danube Research (IAD) (Schmid and Haidvogl, 2015).

MATERIAL AND METHODS

Historical research starts with identifying sources to answer the research questions. The archival sources used here are compiled in the last column of table 1. They pertain to selected case studies (Fig. 1) – hydropower projects which are analysed in the course of the author's PhD Thesis on the environmental history of hydropower in Austria – and includes reports, protocols, and studies. These have been written by experts associated with energy companies, administrations, and universities. Table 1 also includes information on selected features of the geomorphological and socio-economic background of the investigated hydropower plants, as it is assumed that both affect their interaction with sediment regime, and perceptions of planners, as well as the public. To complement these site-specific sources, a search of Austrian library databases and review of publications by the Austrian association of engineers and architects (Österreichischer Ingenieur- und Architektenverein, ÖIAV, journals and monographs between 1900 and 1925), have been carried out. These sources were interpreted to reconstruct what Austrian engineers and planners wrote on the topic of hydropower and sediment regime in approximately the first half of the 20th century.

In the next step, these sources were systematically evaluated using the methodological framework of historical discourse analysis (Landwehr, 2008). This approach, which cannot be elaborated here in detail, analyzes what certain actors said or wrote about a specific topic, and the (medial, institutional, political, etc.) context in which this was done. The guiding research questions were, what planners of hydropower plants wrote about the interaction of hydropower and fluvial sediment regime, and how they perceived related challenges (sediment surplus or deficit) – if they did so at all. Particular attention was also paid to which interests of certain stakeholders (e.g. riparian and navigation), the engineers and planners had to observe in this regard. The investigated period was divided into three phases, and the results of this analysis are presented in the following section and finally summarized in a schematic drawing (Fig. 4).

Table 1: Overview of case studies, selected features of their geomorphological and socio-economic background, and historical sources.

Name/location of hydropower project	Period of planning[a] and construction[b]	River	Geology, (original) river morphology	Main pre-existing uses of river and floodplain	Used archives
Wienerbruck	[a] 1906-1908 [b] 1908-1911	Erlauf, Lassing-bach	Northern Limestone Alps; constrained/pendulous morphorology	Timber floating, mills and small hydropower, tourism (hiking)	EVN-Archiv, Maria Enzersdorf[d]
Wallsee	[a] oldest plans 1910 [b] 1965-1968	Danube	Anabranching river in alluvial plain (Molasse)	Navigation, agriculture	OÖ Landesarchiv, Linz[e]
Ybbs-Persenbeug	[a] oldest plans 1922 [b] 1942-1944, 1954-1959	Danube	Constrained morphology (Danube cutting through Bohemian Massif)	Navigation, agriculture	ÖNB, Vienna[f]
Vienna	[a] different sites discussed at least since 1910 [b] Freudenau plant built 1992-1998	Danube	Anabranching river in alluvial plain (Molasse)	Navigation, urban infrastructure, national park (since 1996)	Misc. university libraries/archives[g] and Archiv DonauConsult[h], Vienna
Wachau	[a] 1971-1983 [b] plant not built	Danube	Constrained morphology (Danube cutting through Bohemian Massif)	Navigation, viticulture and orchards, tourism	Privatarchiv Arbeitskreis Wachau, Spitz a.d. Donau[i]

[c] Refers to location of the dam or weir, based on: Muhar et al. (2004), Hohensinner and Jungwirth (2016).
[d] Misc. technical reports (acc. no. 845-4 to 845-8) and legal protocols (acc. no. 845-9-8).
[e] "Erkenntnis über die wasserrechtliche Verhandlung zur Errichtung des Donaukraftwerkes Wallsee", 1919 (Bestand Strombauleitung Grein, Schachtel 9, Stammzahl 329).
[f] Grzywienski, 1949.
[g] Universität für Bodenkultur, 1991; Bundesministerium für Land- und Forstwirtschaft, 1996.
[h] Söllner, 1943.
[i] Grubinger, 1979.

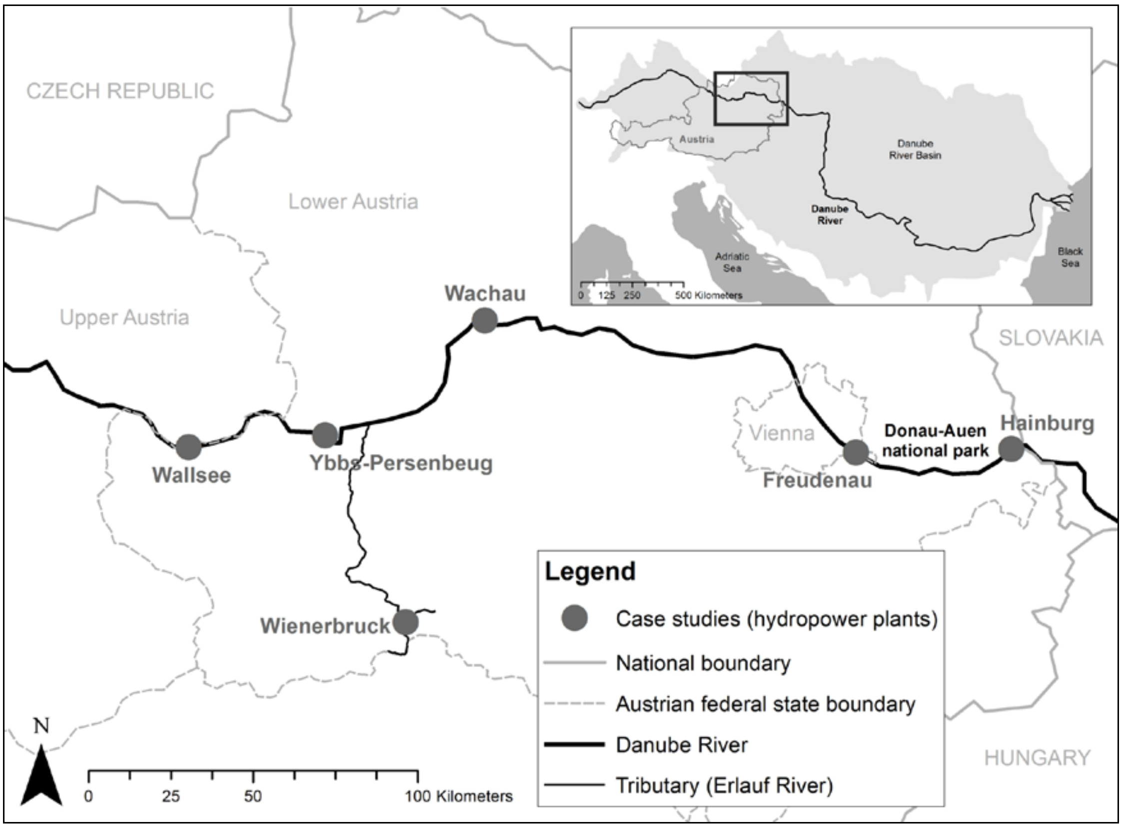

Figure 1: Location of case study sites in the project area in eastern Austria, and within the Danube River basin.

RESULTS AND DISCUSSION

Three distinct phases can be identified for hydropower development at the Austrian Danube. The first phase (“Vision”), covers the period from approximately 1900 to 1940, when first plants at the Danube were planned, but not yet realized. While plants of different types and sizes were already built at the tributaries, various obstacles existed at Austria’s largest river, economic and technical uncertainties being perhaps the most important. Technical experts and investors alike were uncertain whether: enough demand existed for such large amounts of electricity, if storage capacities were lacking, if navigation was strongly opposed to those plans, and if the bedload question was unresolved (Schoder and Schmid, in press). Construction of the first plant at the Austrian Danube (Ybbs-Persenbeug) started during the National Socialist regime. War economy made it necessary to stop the work, which was resumed in the 1950s, only due to unclarified ownership of the project.

A framework plan for the Austrian Danube (Donaurahmenplan) was published in 1955 by Österreichische Donaukraftwerke AG (DoKW), the corporation in charge of developing hydropower along this river since 1947, and four plants were completed by 1970. For this second phase (“Implementation”), approximately from 1940 to 1970, a highly technocratic approach to planning is characteristic.

In the 1970s, for the first time a plant in the planning stage had to be stopped because of civil protests (cf. case study Wachau), which marks the beginning of the third phase (“Ecology”). Its title hints at two interlinked societal phenomena, which is on the one hand an increased ecological awareness in the general public, and on the other hand the need that planners of hydropower plants consider knowledge about ecological processes in rivers and floodplains in order to reach acceptable solutions. This is also reflected in new international agreements (e.g. Ramsar Convention on wetlands), national legislation (e.g. nature protection laws), and the establishment of protected zones (e.g. national parks).

While such a chronological division can never be clear-cut – one must keep in mind that not all plans of the 1940s were actually implemented, and that six more hydropower plants were completed at the Austrian Danube after 1970 – it provides a useful framework particularly to analyse how broader societal circumstances of hydropower development changed during the 20th century.

Phase 1: Vision (1900-1940)

“We should not regard bedload transport as an illness of the river; it can be slowed down by human interventions, but never be brought completely to a standstill”. (Halter in Grünhut, 1919)

Results of literature review (Tab. 2) show that interactions of hydropower plants with rivers’ sediment regime are by no means a recent field of research; the first studies dealing with this topic go back to the advent of hydroelectricity in the late 19th and early 20th century. The literature cited in table 2 includes articles and printed speeches by experts from the fields of hydraulic, agricultural and civil engineering, electricity industry, and navigation, all written to contribute to the contemporary debate on water management and hydropower development in Austria. The authors address their professional peers in the ÖIAV and they also want to reach decision-makers with their recommendations; some want to promote their specific hydropower projects. The Danube as well as its Alpine tributaries are covered, and different aspects of sediment regime which are also nowadays known (except for ecological questions) are addressed: interrupted continuity of bedload and suspended load, upstream challenges (the majority of studies deal with the topic of reservoir sedimentation), but also downstream effects (riverbed degradation is for the first time explicitly addressed by Schoklitsch, 1935). Conclusions or proposed solutions vary according to the focus of the studies. Some articles acknowledge basin-wide processes and challenges of sediment continuity and hydropower (Singer, 1909; Putzinger, 1923) – and come up with interesting suggestions, like using only selected catchments for hydropower and harnessing those completely, starting upstream (Singer, 1909). Others propose solutions for the problem of siltation of Alpine reservoirs or sedimentation in impoundments of run-of-river plants (Halter, 1913; Hauptner, 1914; Putzinger, 1923; Schoklitsch, 1935). It is striking that the majority of currently applied methods (Schoder, 2013) are obviously already known and used. Many studies attempt to quantify sediment transport and bed level changes, although mostly without coming to a precise conclusion. An exception is Schoklitsch (1935), who studied a large number of hydropower plants and included in his comprehensive literature review previous research that dates back to the end of the 19th century. In the contemporary debates (ÖIAV, 1917; Grünhut, 1919, 1922) about hydroelectricity at the Danube and its tributaries (covering also some rivers outside the Danube Basin), many experts acknowledge that sediment transport is an important topic, but they are vague about actual changes that might occur. For the Danube River itself, they mostly address a surplus of bedload and its effects on navigation or reservoir functioning.

Table 2: Results of literature review, studies mentioning hydropower and sediment regime, 1900-1940.

Reference	River (type)	Topics covered [a]							Conclusions and/or proposed solutions
		Bedload	Suspended load	Upstream effects	Downstream effects	Technical challenges	Ecological challenges	Quantification	
Singer, 1909	Alpine streams	(X)	(X)	X	X	X			Use only selected catchments for hydropower and develop those completely, starting upstream.
Halter, 1913	Large rivers (mainly Danube)	X		X		X		X	Flushing and dredging needed, diminishing economic returns of hydropower plants.
Hauptner, 1914	Alpine streams	X	X	X		X			Flushing, dredging, pre-impoundment basins, diversion channels and tunnels, dead storage capacity, dam elevation, training works in reservoir.
ÖIAV, 1917	Danube	(X)				(X)			Bedload transport needs to be observed when a plant is built (referring to Halter, 1913).
Grünhut, 1919	Alpine streams and large rivers	X		X	(X)	X			Large reservoirs should be built in the Bohemian Massif rather than the Alps, because the former is less prone to erosion and sedimentation.
Grünhut, 1922	Danube	X				X		X	Sedimentation will raise the bed in reach with residual flow, interactions with navigation needs to be observed.
Putzinger, 1923	Alpine streams and large rivers	X	(X)	X	X	X			Complete suppression of sediment transport is not desirable; measures should ensure continuity of material (flushing, diversion, dredging with deposition).
Schoklitsch, 1935	Alpine streams and large rivers	X	X	X	X	X	(X)	X	Most current measures of sediment management mentioned, including the impact of reservoir flushing on fish fauna.

[a] X signifies that topic is more extensively covered, (X) that it is only briefly or implicitly mentioned.

This proves that processes and challenges associated with hydropower and fluvial sediment regime were known among technical experts at least since the early 20th century, especially those deriving from a surplus of material. However, what was the role of this knowledge in actual project planning? The engineers designing the Wienerbruck power station, which was the first genuine storage plant in the Austrian monarchy and completed in 1911, had to deal with various challenges, from applying untried technology in a harsh Alpine environment to negotiating with other water users (e.g. timber floating). Sediment continuity is not an issue they addressed in their plans, nor do the preserved legal documents suggest that authorities regarded interrupted sediment transport as problematic, as no respective measures were prescribed. However, later archival material shows that the two reservoirs connected with this plant had to be dredged or flushed several times over the following decades (the larger reservoir at least in the 1940s and 1970s, the smaller one was dredged only a few years ago, Fig. 2). These measures are necessary to maintain the functioning of the reservoir, but are associated with reduced electricity production over a certain period of time and can have negative effects on aquatic organisms (Jungwirth et al., 2003).

Figure 2: Reservoir siltation at the Erlauf River, a Danube tributary.
The water level is drawn down to allow dredging
of deposited gravel, sand, and fine material.

In contrast to this storage plant located at an Alpine tributary, the *Wallsee* hydropower project, a run-of-river plant at the Danube for which the first plans were made in 1910, involved a lot of debate about sediment transport. Technical experts expected that deposition of bedload would occur in the Danube upstream of the weir, in the impounded tributaries, and in the residual flow reach. Measures including dredging and flushing were prescribed, as well as additional surveys of the riverbed. The protocol of water law negotiations (1918-1919) demonstrates that local residents and farmers regarded upstream bed level changes and sediment deposition as a threat to their property (due to increased flood risk, impact on water supply and sanitation infrastructure, and change of groundwater levels). Navigation was another important stakeholder that apprehended grievances caused by bedload depositions. As actual implementation in the 1960s differed considerably from these early plans (construction of a river power plant, within the Danube reservoir chain), the actual impacts of this single plant cannot be evaluated here.

Hence, comparing the site-specific plans with more general studies of hydropower and sediment transport in this phase reveals that basin-wide interactions, which in theory were known, were often not observed at the plant scale. If it was addressed at all, the disruption of sediment continuity was regarded merely as a technical problem. The connection of sediment regime and river ecology (e.g. fish fauna) was nowhere considered in the reviewed literature, except for Schokolitsch (1935), who mentions that flushing a reservoir might have negative impacts on the fish population due to increased concentration of suspended sediments. The reason for the identified gap between theoretical knowledge of sediment transport and applied practice of hydropower planning might be a pragmatic one. If the warnings from more cautious experts (e.g. Singer, 1909), had been observed, this would have considerably slowed down hydropower development in Austria – and most engineers at that time agreed that Austria was already late in exploiting this valuable resource, compared to other countries, e.g. Switzerland (cf. Schoder, in press). Moreover, siltation of Alpine reservoirs, which was the most important known impact of disrupted sediment continuity, comes with a considerable time lag, which might explain why it was not considered as urgent problem in the planning stage. The Danube itself remained still free of hydropower plants in this phase, and unclarified issues of bedload transport (especially its interaction with navigation) were an important obstacle. Nevertheless, debates about the first schemes and projects also reveal that the river's sediment regime was already affected by plants at the tributaries and 19th century river regulation – but exactly to what extent remained mostly obscure to technical experts.

Phase 2: Implementation (1940-1970)

"May the construction of large river power plants, which could bring about unwanted or even dangerous effects without mastering the bedload problem, provide us again with occasion for new successful research (…)?". (Lanser, 1953)

Phase II witnessed a comprehensive framework plan which aimed at complete exploitation of the Austrian Danube's energetic potential, the completion of the first four plants, but also saw (at least) one unsuccessful scheme.

Plans for a diversion plant at Fischamend, shortly downstream of Vienna and the later constructed plant Freudenau at the south-eastern edge of the city (Fig. 1), to some extent cover the topic of sediment regime. Those plans were submitted in 1943 by the engineer Karl Söllner, who estimated that annually transported quantities of bedload amounted to 350,000 to 450,000 m³, and that thereof 250,000 to 350,000 m³ would have to be dredged, unless they would be flushed during floods. Interestingly, he assumes an "almost stationary state of the riverbed" due to the Danube's regulation, in the river stretch at Vienna and downstream (Söllner, 1943). This statement corresponds to recently published data showing that the effects of regulation were in fact temporally highly variable. Iinitially, a large amount of bedload was released from the regulated stretch at Vienna and deposited in the immediately following river sections. Later, this material was transported further downstream, leading to rather stable bed levels for some decades (since 1930 to 1940) in those sections, followed by conditions of riverbed incision (Hohensinner and Jungwirth, 2016; Klasz et al., 2016; based on Schmautz et al., 2002). Apparently Söllner, at his time, saw only a part of this bigger picture; he missed the necessary long-term perspective and in this sense came up with a false estimation of the general challenge.

Deviations can be observed in the figures for bedload transport (and also in the methods to determine them), when only a few years later quantities at Ybbs-Persenbeug (upstream, Fig. 1) were estimated at an annual average of 400,000 m³ and a maximum of 600,000 m³ (Grzywienski, 1949). The author studying this site, where construction of the first Danube plant had started in 1942, concluded that the majority of this material would need to be dredged, although part of it might be used for construction purposes. He recommended bedload traps at the beginning of the impoundment due to the irregular input of material. It must be noted that both studies did not yet regard the Danube as a chain of plants, although Grzywienski (1949) mentioned that downstream erosion would be reduced by another plant at Melk.

A few years later, the Donaurahmenplan (DoKW, 1955) hardly mentioned sediment transport in its accompanying descriptive text. This document merely states "avoiding aggradation of the riverbed" as a general planning principle (DoKW, 1955); apart from that, it explains how an uninterrupted chain of hydropower plants would exploit the Danube's hydropower potential most efficiently, and how navigation would benefit from the increased water depths. An incised riverbed was regarded as positive to increase useable head and should for some plants be achieved "artificially" with dredging (DoKW, 1995). After all, hydropower at that time was regarded as a means to reconstruct the war-torn country and to overcome energy shortages, and plants at the Danube enjoyed priority due to their large contribution to this end (Vas, 1956). However, in the eyes of other experts, questions about quantifying processes at the riverbed, let alone ensuring continuity of sediments, were far from being resolved. Two years earlier, civil engineer Otto Lanser, who had been involved in many hydropower projects before, pointed out the need to develop quantification methods further, especially in the light of building hydropower plants at the Danube, but he also expressed a lot of optimism about technical and scientific progress in this regard (Lanser, 1953). Despite the lack of accurate insights into processes at the riverbed, the first two plants of the Donaurahmenplan (Jochenstein at the Austrian-German border, and Ybbs-Persenbeug; Fig. 3), were finished by 1960, and two more plants started operation by 1970.

Figure 3: Impoundment of Ybbs-Persenbeug, one of the ten run-of-river plants at the Austrian Danube (picture taken by the author in October 2014). Prior to "rectification" in the 19th century, the river had at this site anabranching channel morphology.

Phase 3: Ecology (1970-2000)

"Donaukraft (hydropower operator) has to maintain only the existing riverbed which is defined as the reference bed before impoundment. Beyond this, improvements of navigability are not in the responsibility of Donaukraft". (Bundesministerium für Land- und Forstwirtschaft, 1996)

With six more plants built after 1970, two free-flowing sections remain at the Austrian Danube. They warrant a closer look, because processes at the riverbed were central in the argumentation for and against hydropower plants at these sites, and they challenge experts up to the present. In these sections, flow velocities and sheer stress are higher than in the impoundments, and due to the lack of material input from either upstream or the banks, the river can degrade its bed (Habersack et al., 2012; Klasz et al., 2016).

Riverbed incision was initially used by the electricity industry as an argument to build a power plant in the Wachau section of the Danube, which should ensure that the bed was stabilized and one of the remaining bottlenecks for navigation was removed. Although foreseen in the Donaurahmenplan, this plant was controversial from the beginning, not least because of the importance of tourism and viticulture in the region. Locals feared that the scenery would be impaired by dams, and that vineyards and orchards might be affected by microclimatic changes; they were supported in their protests against the plant by many visitors of the region, and also by some technical experts (Hirtzberger, 1995). Herbert Grubinger (geologist, engineer, and professor at ETH Zürich) came to the preliminary conclusion that riverbed incision would not be a major issue in this river section due to its geomorphology (Tab. 1). Therefore, navigability could be ensured also with conventional river engineering measures (Grubinger, 1979). He recommended that further studies about the riverbed should be made, but in 1983 representatives of DoKW and the Austrian government officially announced that no plant would be built at that site (Hirtzberger, 1995). The region was declared UNESCO cultural heritage in 2000, the Wachau is still one of the "bottlenecks" for navigation along the Danube (via donau, 2005), but it seems that riverbed incision indeed is not a major problem, in contrast to the Danube's second free-flowing section east of Vienna.

At that location, another hydropower plant (Hainburg) was stopped due to civil protests in the early 1980s, and a national park was established in 1996 to protect one of the last remaining floodplain forests at the upper Danube (Nachtnebel, 1995; Schmid and Veichtlbauer, 2006; Schoder and Schmid, in press). However, DoKW set out to build another plant upstream, where the impoundment would not interfere with any protected or ecologically sensitive areas (Freudenau plant, completed in 1998). In the meantime, first studies had been carried out detecting and quantifying riverbed incision at the Danube east of Vienna (e.g. Kresser, 1984). Ecological issues related to Danube hydropower plants were investigated in different research projects (e.g. Hary and Nachtnebel, 1989; ARG, 1989; Expertengruppe Untere Donau, 1996). Amongst other insights, they established that changes of the riverbed, water tables both in the Danube and the adjacent groundwater bodies, and floodplain ecology were closely linked. This meant that sediment continuity was important to consider for the planners of the Freudenau plant, in order to not further aggravate the already existing problem of riverbed degradation. Artificial bedload addition was prescribed as a downstream measure, and the average amount of annual material input was estimated at 160,000 m³ in the respective legal document (Bundesministerium für Land- und Forstwirtschaft, 1996). This figure is considerably lower than the amount of sediment transport experts had calculated before the first Danube plants were built, which might be explained with the mostly completed hydropower chain and the different plant sizes and types – after all, the prescribed measure was only to offset the effect of this single plant. However, the cited document also states that the actual amount was to be determined based on detailed riverbed surveys and the calculation of mass balances. Artificial gravel addition at the Danube east of Vienna is still ongoing and riverbed incision remains a challenge not yet entirely resolved, although in the meantime revised projects have been proposed based on ecologically oriented measures of hydraulic engineering (cf. e.g. Habersack et al., 2012).

To sum up the developments in this last phase, changes of the riverbed and associated problems, which had been to some extent anticipated by experts in earlier phases, became evident in certain river sections. At the same time, ecological properties of rivers and floodplains emerged as an additional (also public and thus political) interest to be observed by planners, and the designation of protected areas excluded certain options (such as "hard" engineering measures or an additional hydropower plant at the Danube east of Vienna).

Synthesis. The fluvial landscape of the Austrian Danube results from the overlaid effects of different phases of river management. Along with the material transformation, also knowledge and expertise of how to initiate and shape this transformation evolved (Fig. 4). Already in Phases I and II, studies of sediment transport and insights into bed level changes at the Austrian Danube and some tributaries (e.g. HZB, 1937; HZB, 1948, cited in Klasz et al., 2016), and experts and planners of hydropower also drew from knowledge of channel morphology dating back to river regulation (e.g. Schoklitsch, 1935; Söllner, 1943). Hydropower development provided a stimulus for more research. In Phase III, methods of quantification were improved in the course of more detailed studies of specific river sections, where issues of sediment regime were perceived as problematic. Monitoring and modelling sediment transport and river morphological changes related to hydropower have of course further evolved in the 21st century (Habersack et al., 2013). However, if and to what extent this knowledge is considered in actual river management and decision making depends on constellations of interests in the specific social, political and cultural context of each period. These interests in the fluvial landscape have considerably diversified since Phase III, when in addition to long established stakeholders (navigation, electricity demand, agriculture, infrastructure and settlements in the floodplains) issues like nature protection, cultural heritage, as well as tourism and recreation have to be observed. Some other interests, however, had disappeared from the landscape at a much earlier stage: timber floating, mills, and commercial fishery can be seen as "losers" in the transformation of the Austrian Danube (Haidvogl, 2010).

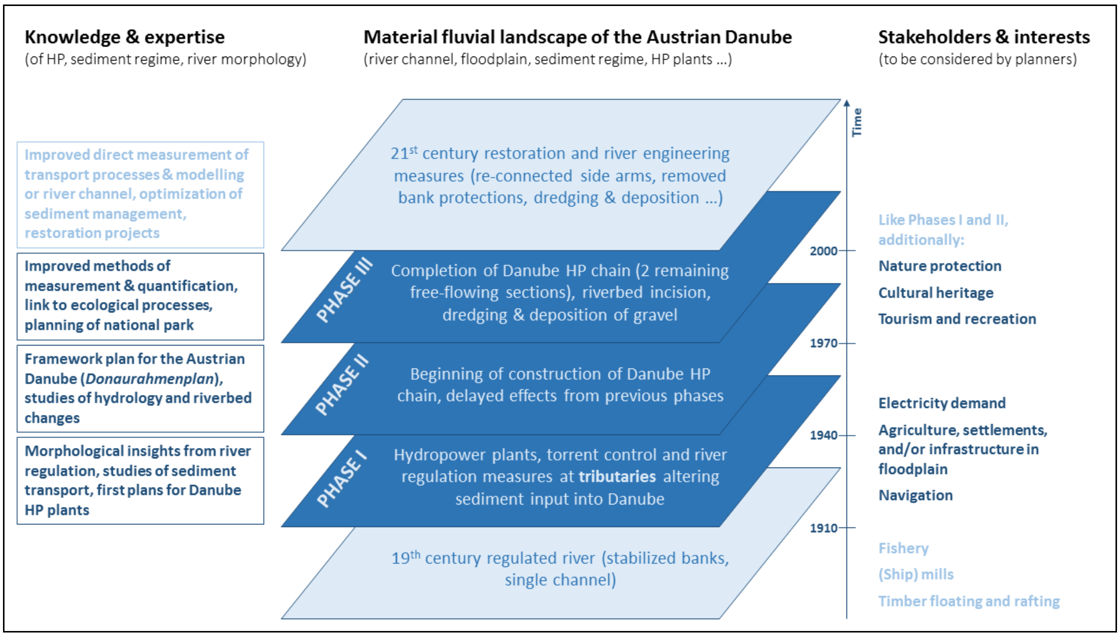

Figure 4: Three dimensions of the Danube's transformation – knowledge and expertise, stakeholders and interests, and different states of the material fluvial landscape.

Figure 2 provides a synthesis of the findings from all phases, each layer representing a historical state of the river; the long-term (side) effects of this state are passed on to later phases and generations as "legacies". The regulated river of the 19th century must be regarded as a merely arbitrary starting point, as the Danube's sediment regime has already been altered in pre-industrial times (Giosan et al., 2012). In any case, such a physically and institutionally complex landscape means that the attribution of individual responsibilities to compensate environmental problems has become increasingly difficult. If DoKW (respectively its legal successor) has to offset only the effects of single plants, who is then responsible to implement measures to tackle the problem of riverbed incision that had already existed or had been initiated before that plant was built? How can demands of navigation, which had been the dominant interest to transform the Danube for centuries, be reconciled with recently emerged needs of nature conservation and a national park? What is an adequate "reference state" of the riverbed that should be the baseline for restoration projects? These are only some of the questions posed by a fluvial landscape which have been so thoroughly transformed like the Austrian Danube. Socio-cultural drivers, like different perceptions of the river and its ideal state, played and still play an important role in shaping this landscape, and environmental history can help to unravel them.

CONCLUSIONS

This paper focused on the interaction between hydropower and the Danube’s sediment regime, and analysed how it changed in the course of the 20th century. Approaching this link through the writings of engineers and planners of hydropower plants, three dimensions of the Danube’s transformation were distinguished. At the level of knowledge and expertise, it became clear that management options (e.g. flushing and dredging) did not change significantly over that time period, but were optimized due to improved methods of quantification and monitoring. In the material fluvial landscape, the effects of different interventions (e.g. river regulation, plants at tributaries, Danube hydropower chain) can hardly be disentangled. As “legacies”, they are passed on and limit the options for present and future river management; but on the other hand, they enable certain practices, such as electricity generation and navigation. Looking at the two remaining free-flowing sections of the Austrian Danube has revealed that also site-specific geomorphology and properties of the riverbed matter for actual outcomes (e.g. riverbed incision). Changing modes of perception were central to this study, and it can be concluded that those were closely related to specific constellations of interests in any of the observed phases. For example, interests of electricity generation and navigation caused planners to envisage the Danube as an uninterrupted chain of impoundments after World War II. Some case studies have illustrated how a lack of long-term observation can mean that experts neglect or misjudge processes at the riverbed and their temporal variability. Also nowadays, such a long-term perspective is needed to understand and manage sediment dynamics in the catchment of a large river like the Danube.

ACKNOWLEDGEMENTS

The author is recipient of a DOC Fellowship of the Austrian Academy of Sciences at the Institute of Social Ecology, and this research is part of her ongoing dissertation project entitled “Flow regimes – the nature of Austrian hydropower, technical expertise, and environmental discourses in the 20th century”. Travel costs for participating in the IAD Conference in Sibiu, Romania have been partially financed by Österreichische Forschungsgemeinschaft (ÖFG, grant 06/14960). The author gratefully acknowledges financial support from AAU Research Council and advice from supervisor Assoc. Prof. Mag. Dr. Martin Schmid, and wants to thank all people and institutions providing access to archival sources.

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TROPHIC STATUS AND PHYTOPLANKTON LIMITATION CONDITIONS IN A FEW BULGARIAN AND HUNGARIAN DANUBE RIVER WETLANDS

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DOI: 10.1515/trser-2015-0084

KEYWORDS: Secchi depth, chlorophyll-a, phosphorus, nitrogen macrophytes.

ABSTRACT

The phytoplankton limitation conditions in a few Bulgarian and Hungarian wetlands and in two more sampling sites, one for each Danube stretch, were investigated by means of trophic state index of Carlson (1977) for phytoplankton chlorophyll-a (CHL), Secchi disk depths (SD), total phosphorus (TP), and total nitrogen (TN) concentrations. Phytoplankton of both river sites was not limited by non-algal turbidity, nor was they phosphorus and nitrogen limited. In summer months the studied wetlands were predominantly nitrogen limited, while in spring and autumn limitation by non-algal turbidity prevailed.

RESUMEN: Estado trófico e condiciones de limitación de fitoplancton en dos humedales del río Danubio búlgaro y húngaro.

Las condiciones de limitación de fitoplancton en dos humedales búlgaros y húngaros y en dos más sitios de muestreo para cada tramo del Río Danubio, fueron estudiados por medio de los índice de estado trófico de Carlson (1977) para la clorofila-a (CHL) del fitoplancton, profundidad del disco de Secchi (SD), y por las concentraciones de total de fósforo (TP) y total de nitrógeno (TN). El fitoplancton en ambos sitios del río no estaba limitado por turbidez no algal pero tampoco por fosforo ni nitrógeno. En los meses de verano, los humedales estudiados fueron predominantemente limitados por nitrógeno, mientras en primavera y otoño prevaleció la limitación por turbidez no-algal.

REZUMAT: Starea trofică și condiții de limitare a fitoplanctonului din câteva zone umede de pe Dunăre, din Bulgaria și Ungaria.

Condițiile de limitare a fitoplanctonului în câteva zone umede din Bulgaria și Ungaria și alte două situri de prelevare a probelor, unul pentru fiecare zonă a Dunării, au fost investigate folosind indicele de stare trofică Carlson (1977) pentru clorofila-a din fitoplancton (CHL), adâncimea discului Secchi (SD), concentrația totală de fosfor (TP) și cea totală de azot (TN). Fitoplanctonul din ambele râuri a fost limitat de turbiditatea non-algală, dar nu a fost limitat de concentrațiile de azot și fosfor. În lunile de vară factorul limitativ în zonele umede studiate a fost azotul, în timp ce în primăvară și toamnă factorul limitativ prevalent a fost turbiditatea non-algală.

INTRODUCTION

According to Lieth (1975) swamps, moors, and similar extremely shallow aquatic ecosystems (i.e. wetlands) are among the most productive territories on the earth. The Danube River adjacent wetlands belong to this category and their high productivity on one hand means utilization of nutrients, but depending on hydrology the primary production could be transformed into fish yield. Probably as a result of considerable wetland reduction and increasing isolation from the main river the fish productivity is decreasing (Kalchev et al., 2007) and depending on connectivity to the main river and prevailing hydrology conditions wetlands are functioning as sinks for nutrient load brought in the floodplain by the river and as filter between terrestrial and the river areas (Hein et al., 2005; Bondar et al., 2007). The capacity of wetlands to act as filter or sink for nutrients depends on the intensity, i.e. on limitation conditions of process of primary production, accomplished by two producers − phytoplankton and macrophytes. While the methodology for measuring phytoplankton productivity is advanced and simplified long time ago (Vollenweider, 1969; Carlson, 1977, 1991) the unified estimation of productivity of both phytoplankton and macrophytes together is still a difficult task (Canfield, 1984) due to the tedious nature of macrophyte measurements, heterogonous, diverse character of wetland biotopes and strong influence of varying hydrology. Nevertheless, the compiled data of nutrient concentrations, main photosynthetic pigment, and water column transparency for several years by application of wide spread methodology for trophic status estimation (Havens, 1994; Gibson et al., 2000; Jarosiewicz et al., 2011; Tosheva and Traykov, 2012; Pęczuła et al., 2014) give the opportunity to get insight into main trophic characteristics and differences between Hungarian (middle Danube) and Bulgarian (the Lower Danube) wetlands.

MATERIALS AND METHODS

Sampling sites in the Lower Danube River encompassed three marshes located on Belene Island (Murtvo blato, Peschin, and Dyulova bara, forming the wetland group of lake type), the middle of the river side arm located in front of Belene locality, and three other wetlands (Kalimok Canal, Kalimok Marsh, and Brushlen Canal – the wetland group of canal type) in the Kalimok-Brushlen protected area (Fig. 1).

Water samples were taken in spring, summer, and autumn between autumn 2009 and spring 2012. Total nitrogen (TN), total phosphorus (TP), and turbidity at 550 nm as absorbance were determined colorimetrically with Nova 60 photometer and ready to use kits from Merck. The chlorophyll-a samples after filtering through 0.7 μm glass fibre filter and storage in liquid nitrogen were analysed in the laboratory according to ISO 10260 standard (ISO, 1992). The water column transparency was measured by means of Secchi disk. In some cases such measurements were not possible due to low depth and high transparency and therefore we calculated the missing Secchi disk values by means of the regression equation based on absorbance at 550 nm and Secchi disk readings from the same wetlands when such measurements were possible ($\lg10\,(SD) = -0.536 - 0.653{*}\lg10\,(\text{Absorbance})$, $n = 14$, $R^2 = 0.772$, $P < 0.00005$).

The sampling campaign of Hungarian wetlands includes Mohács Danube River, Riha and Mocskos wetland sites (Fig. 2) in years 2012 (six visits), 2013 (four visits), and 2014 (three visits). Standard analytical methods (Golterman et al., 1978) were used for determination of suspended particular matter (SPM), total phosphorus (TP), and chlorophyll-a (Chl). Total nitrogen (TN) was determined by TOC analyser (Elemetar-liqui-TOC).

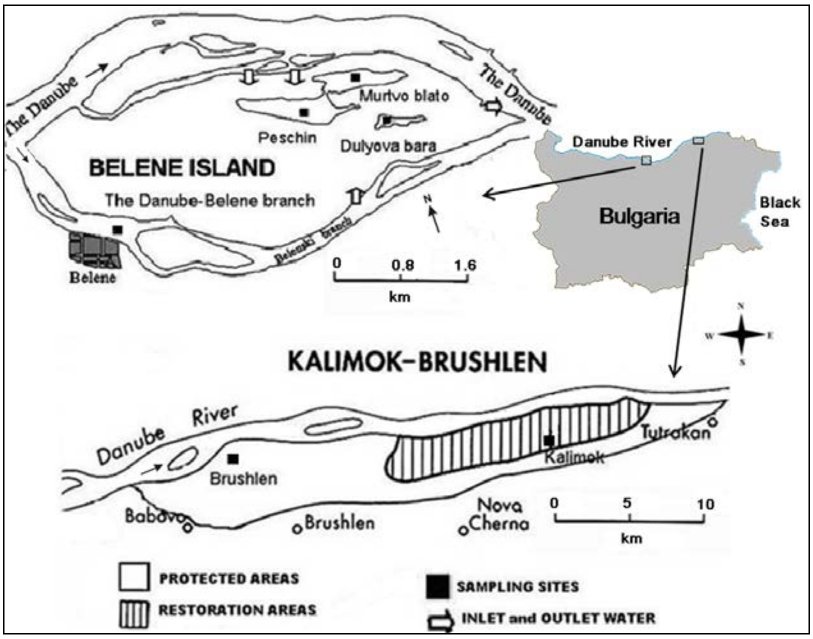

Figure 1: Lower Danube wetland sites on Belene Island (river km 561-576): Murtvo blato, Peschin and Dyulova bara marshes; Danube River sampling site in front of Belene locality; in Kalimok-Brushlen area river km 440-465: Kalimok Canal, Kalimok Marsh, Brushlen Canal.

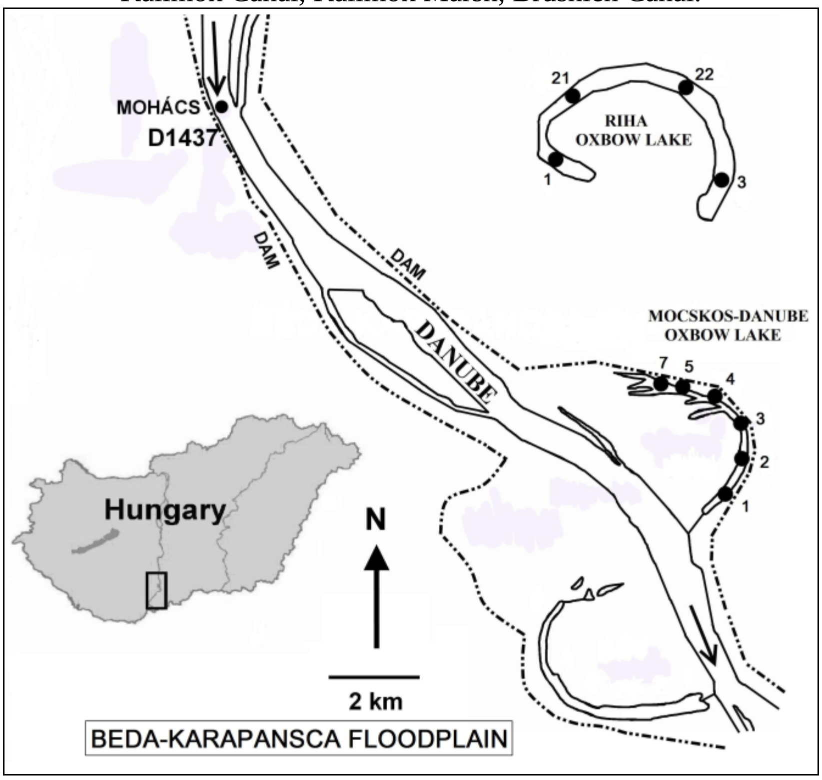

Figure 2: Middle Danube sites: Mohács Danube River, Riha, and Mocskos wetlands.

Because the SPM data were not appropriate for calculation of trophic state index of Carlson (1977) and no local data were available to convert SPM values into Secchi depth readings for this purpose we used relationships published by Dahlgren et al. (2004). They derived regression equations for California streams and waterways, which were shallow and streaming like most of Hungarian wetlands along the Danube River, which are temporary streaming and temporary stagnant. Due to the shallow nature of such water bodies Dahlgren et al. (2004) measured the water column transparency by means of a tube and calculated the regressing equation between SPM and tube transparency (TT) $\log_{10}$ (SPM) = 1.17*$\log_{10}$ (TT) + 3.13, $R^2 = 0.51$, $P < 0.001$. However, the obtained TT values are not identical with Secchi disk readings (SD) and therefore Dahlgren et al. (2004) provided a second equation, converting TT (in cm) into SD (in cm) – SD = 1.09 *TT + 15.6, $R^2 = 0.71$, $P < 0.001$. In both studies (Bulgarian and Hungarian) the degree of coverage of water surface by macrophytes (floating and submerged) was also estimated visually as a percentage.

The compiled values of chlorophyll-a (CHL), Secchi disk readings (SD), total phosphorus (TP), and total nitrogen (TN) were converted into trophic state index (TSI) of Carlson using formulas published by Carlson (1977) and Havens (1994). The deviations between four trophic state indices were used to evaluate the kind and degree of nutrient limitation and composition and particle size of seston after Havens (1994). As shown in figure 3 negative differences of TSI of Chl and TSI of nutrients (TP, TN) on the ordinate axis indicates a lack of nutrient limitations and vice versa, while the gradient from negative to positive differences between TSI of Chl and TSI of SD on abscissa axis indicates transition from small to large algae and from non-algal turbidity to zooplankton grazing.

The statistical analysis includes linear regression calculation which was carried out by PAST statistical package (Hammer et al., 2001).

RESULTS AND DISCUSSION

Limitation conditions

The deviations between four calculated TSI for Bulgarian wetlands and Danube River in figure 4 (A-C) show that phosphorus never limits the phytoplankton growth. The scatter of Danube River samples in figure 4A demonstrates prevailing limitation by non-algal turbidity except in one to two summer samplings in which the phytoplankton seems to consist of nitrogen limited large algae. The non-algal turbidity limitation effect in Danube is still a natural situation despite its considerable reduction during the last decades.

In wetlands of lake type on Belene Island situated in the middle of the river the nitrogen limitation is stronger than in the river (Fig. 4B). Four summer samplings have nitrogen limited large algae, while two spring and one summer samplings have small slightly nitrogen limited algae. However, the majority of samplings from these lake type wetlands in autumn and spring are characterized by considerable non-algal turbidity limiting the phytoplankton growth. This turbidity might have two origins: the connection to the river which is more pronounced in spring and autumn and the strong wind influence on these shallow waters (about one m depth). The zooplankton grazing might also be of importance due to low number or complete fish absence as a result of restricted connection to the river and reduced oxygen concentration during the night, caused by intense decay of plant organic matter.

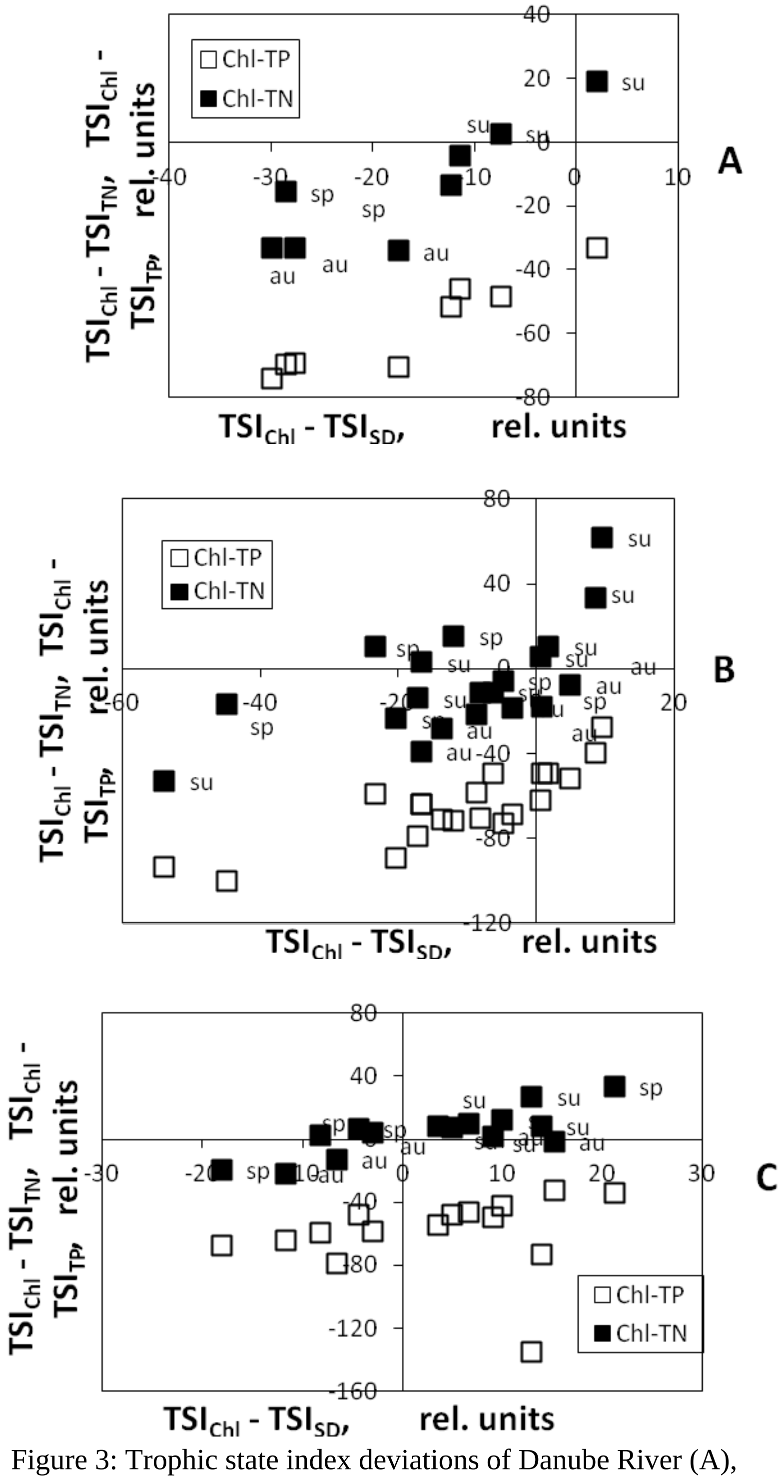

Figure 3: Trophic state index deviations of Danube River (A), Belene (B), and Kalimok-Brushlen Wetlands (C) sampling sites. Abbreviations indicate spring (sp), summer (su), and autumn (au) samplings.

The Bulgarian wetlands of canal type, considerably remote from the river with little or no connection to the river and bordering on cultivated farmland, are distinguished by more clearly expressed nitrogen limited algae (Fig. 4C). Obviously considerable quantities of phosphorus are entering the wetlands by surface run off and ground waters, which leads to nitrogen limitation again more pronounced during the summer months and one spring month of sampling. The non-algal turbidity influence seems to be limited to a few samplings due to greater depth (up to four-five meters at high river levels) and shelter against wind provided by strong plant growth frequently covering both canal banks.

In the middle Danube River similarly to the Lower Danube the majority of samplings are dominated by non-algal turbidity, which is usual for flowing waters and still valid for the Danube River despite the large number of reservoirs build on its tributaries and similar facilities built in the river itself (Fig. 4A). With very few exceptions phytoplankton of almost all samplings is limited neither by phosphorus nor by nitrogen and the non-algal turbidity might be the only factor affecting algae because the zooplankton seems to have no noticeable effect in this river stretch.

Riha Wetland like the Danube River are neither phosphorus nor nitrogen limited (Fig. 4B). However, the difference is that only spring and part of summer samplings are under the influence of non-algal turbidity. The rest of samplings are exposed to zooplankton grazing, which, as shown, was completely absent in this part of the Danube River. Most probably during summer-autumn months the fish press on zooplankton in this isolated from the river wetland is weakened, which could explain the occurrence of zooplankton grazing.

The Mocskos Wetland demonstrates a weak but clear nitrogen limitation of phytoplankton occurring in some of late summer and autumn months (Fig. 4C). The reason for this might be a lower supply of nitrogen because its catchment is exposed to a lesser agricultural influence than the Riha Wetland. In some of the spring and summer months we observed zooplankton grazing, but there are enough summer samplings in which the non-algal turbidity was the limiting factor. This heterogeneity of limitations might be the result of different distance to and as a consequence of different exposure of Mocskos Wetland sampling sites to varying Danube influence, while Riha being completely isolated from the river did not demonstrate such variability in limitation conditions.

Trophic status estimation

Considering the obtained TSI values for all four variables (Chl, SD, TP, and TN) we have to bear in mind that TSI_{Chl} is closer to a true, generated productivity (trophy), while the other three, especially when their values are higher than that of TSI_{Chl}, present potential trophy. By means of the classification scale providing approximate trophic ranges published by Gibson et al. (2000) the TSI_{Chl} of Bulgarian Danube and Belene Wetlands seem to be mesotrophic, while Kalimok Wetland and all Hungarian sites are eutrophic (Tab. 1). However, except the Danube River where no macrophytes are available, the TSI of both Bulgarian and Hungarian wetlands underestimate the real value of whole ecosystem trophy because beside the phytoplankton they also have a strong aquatic macrophyte growth, with considerable and difficult to measure share of total ecosystem primary production.

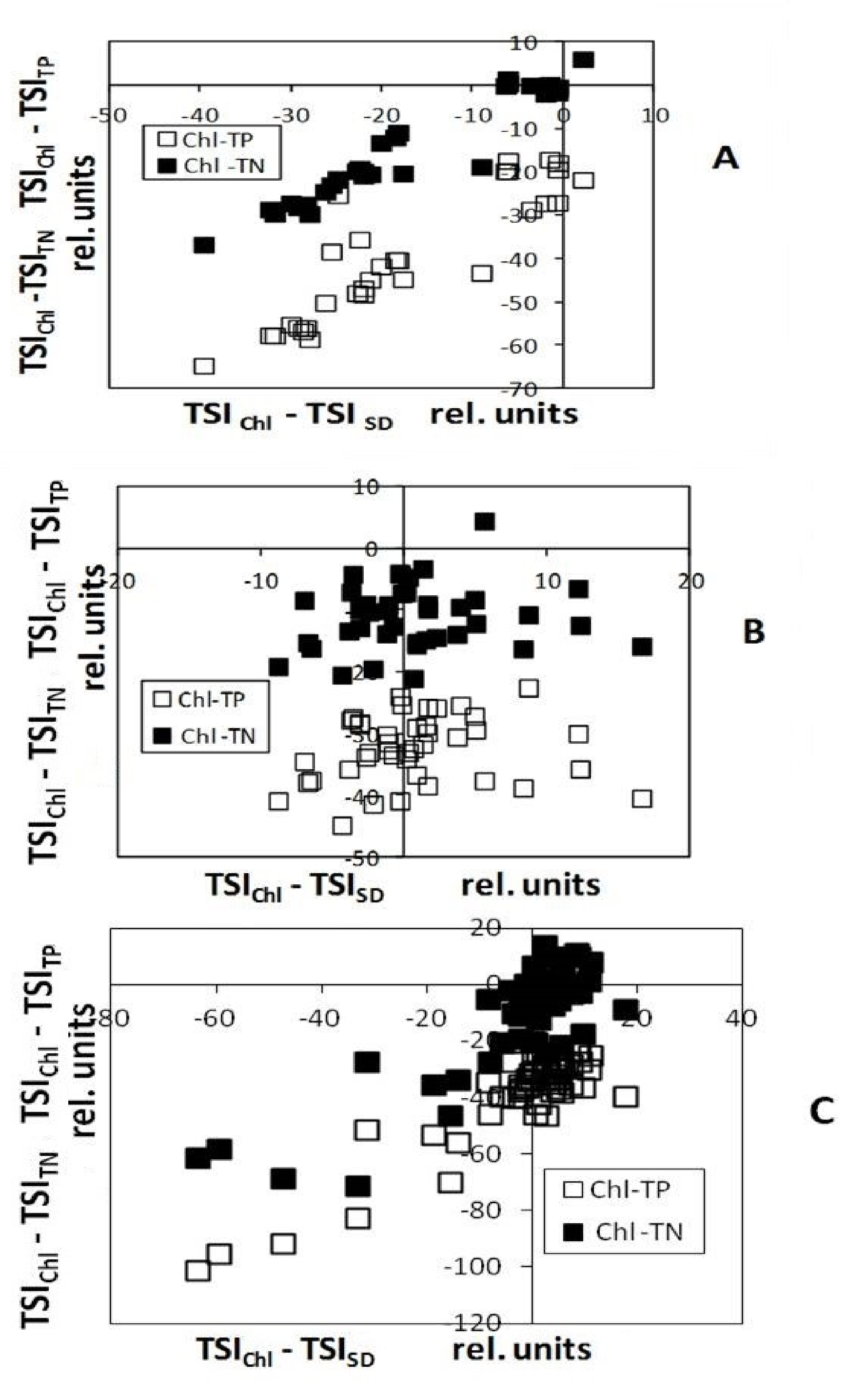

Figure 4: Trophic state index deviations of Mohács Danube (A), Riha (B), and Mocskos (C) sampling sites.

Table 1: Arithmetic means (AM) and standard deviations (STD) of trophic state indices of chlorophyll (TSI_{Chl}), total phosphorus (TSI_{TP}) and total nitrogen (TSI_{TN}).

Sampling sites	Indices	TSI_{Chl}	TSI_{SD}	TSI_{TP}	TSI_{TN}
Belene Danube	AM	43.7	60.3	101.6	57.7
	STD	14.3	10.0	7.6	7.0
Belene Wetlands	AM	44.4	53.4	106.9	48.7
	STD	15.0	8.6	10.8	17.7
Kalimok Wetlands	AM	53.6	52.4	103.7	49.4
	STD	14.9	6.3	15.0	10.2
Mohács Danube	AM	54.0	64.3	93.8	69.4
	STD	11.3	15.0	6.7	2.4
Riha Wetland	AM	57.0	56.6	93.9	65.8
	STD	13.5	9.4	5.8	3.3
Mocskos Wetland	AM	60.9	60.1	93.4	71.9
	STD	9.7	10.9	9.8	5.9

This fact has been acknowledged from Porcella et al. (1979), Canfield et al. (1983a, b), Canfield and Jones (1984). Canfield et al. (1983a, b) assume that real trophic state estimated by phosphorus is a sum of quantities in water column and aquatic macrophyte tissue, despite that it is known that macrophytes take up nutrients from both water column and sediment (Porcella et al., 1979; Tosheva and Traykov, 2012). After extensive time and labour intensive investigations Canfield et al. (1983a) showed a considerable increase in lake trophy after adding phosphorus from macrophytes to water column concentrations. Applying the percentage of macrophyte cover, TN, and TP Canfield et al. (1984) obtained multiple regression predicting chlorophyll-a concentration, while Porcella et al. (1979) incorporated the percent macrophyte area covered into a lake evaluation index.

Based on the well-known negative relationship between macrophyte growth and phytoplankton chlorophyll-a, our experience that macrophyte development varies between years and seasons, we tested the relationship between four derived trophic state indices and percentage area covered by aquatic macrophytes in the hope, that at zero macrophyte cover we will be able to read the real trophic state. We obtained only two statistically significant linear regressions. In Mocskos Wetland, which is nitrogen limited, we obtained $TSI_{TN} = 71 - 0.078{*}\%Area_{Macrophytes}$, $R^2 = 0.32$, $P < 0.002$, which means at zero macrophytes the trophic state might be close to hypereutrophic i.e. TSI = 71. This is bigger than 60.9 for TSI_{Chl} and close to AM of TSI_{TN} in table 1 (71.9). The second linear regression is $TSI_{Chl} = 80.6 - 0.325{**}\%Area_{Macrophytes}$, $R^2 = 0.24$, $P < 0.05$ in Bulgarian Kalimok-Brushlen Wetland of canal type. The TSI at zero macrophytes would be 80.6, which is considerably higher than the table mean value for TSI_{Chl} of 53.6. The lack of statistically significant relationships between percentage area covered by macrophytes and other trophic state indices might be due to incomplete data collected about macrophytes. As practiced by Porcella et al. (1979) and Canfield et al. (1983a) we could supplement them with density of macrophyte stocks and utilize recent technical advancements to apply flying drones for better visual estimation of area covered by macrophytes.

CONCLUSIONS

This paper indicates once more the lack of phosphorus, or very weak and seldom occurring nitrogen limitation in the Middle and Lower Danube River, which indirectly support the previous suppositions in the literature for the limiting role of silica for river phytoplankton (Kalchev et al., 2008). The phosphorus load on wetlands is also considerable and seems even higher than on the river, despite the diminishing effect of abundant macrophyte growth. This leads to nitrogen limitation, which is more expressed in wetlands of the Lower than the Middle Danube. More precise determination of macrophyte share from primary production of wetland ecosystems but also accounting for morphology, hydrology, and possible toxic influences e.g. by cadmium brought into water by phosphorus fertilizers will allow more accurate estimation of wetland trophic status and their capacity to diminish the anthropogenic load on the river.

ACKNOWLEDGMENTS

This study was funded by the National Scientific Fund of Bulgaria within the project DO 02-352, by the Deutsche Bundesstiftung Umwelt, AZ 24050 project and by the program for collaboration and visit exchange of Hungarian and Bulgarian Academies of Science. We are also very grateful to Buzetzky Győző for his help in the fieldwork, to Gábor Horváth and Bernadett Garad for the chemical analyses.

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NANOPARTICLES ECOTOXICITY ON *DAPHNIA MAGNA*

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DOI: 10.1515/trser-2015-0085

KEYWORDS: acute toxicity, titanium dioxide, copper oxide, zinc oxide.

ABSTRACT

In recent years, development of nanotechnology as well as the toxicity potential of nanomaterials on the environment has received much attention. In order to assess the potential toxic impact of nanoparticles on aquatic environments, we used three kinds of nanoparticles, including titanium dioxide (TiO_2), copper oxide (CuO), and zinc oxide (ZnO) on an aquatic model species, *Daphnia magna*. In fact, *Daphnia magna* was exposed to different concentrations for 24, 48, 72, and 96 h at 20-25°C. All the important water quality parameters, such as temperature, pH, and dissolved oxygen (DO) were controlled to meet the standard requirements during the experiment. The LC_{50} values for 24, 48, 72, and 96 h were estimated statistically using Probit methods. The LC_{50} 48 h values for TiO_2, CuO, and ZnO were 171.88 mg/l, 6.62 mg/l, and 3.23 mg/l, respectively.

ZUSAMMENFASSUNG: Die Ökotoxizität von Nanopartikeln an *Daphnia magna*.

Die Entwicklung der Nanotechnologie in den letzten Jahren sowie das Toxizitätspotential von Nanomaterial werden in dieser Arbeit behandelt. Um die potentiellen toxischen Einwirkungen von Naopartikeln in aquatischer Umwelt zu ermitteln, wurden drei verschiedene Nanopartikeln von Titan Dioxyd (TiO_2), Kupferoxyd (CuO) und Zink Oxyd (ZnO) an einer aquatischen Modellart *Daphnia magna* untersucht. *Daphnia magna* wurde unterschiedlichen Konzentrationen für 24, 48, 72 und 96 Stunden bei 20-35°C ausgesetzt. Sämtliche der Wasserqualitätsparameter wie Temperatur, pH und DO wurden kontrolliert, um die Standardansprüche während des Experiments kennenzulernen. Die LC_{50} Werte für 24, 48, 72 und 96 Stunden wurden statistisch mit Hilfe der Probit Methode geschätzt. Die LC_{50} 48 Stunden Werte für TiO_2, CuO und ZnO betrugen jeweils 171.88 mg/l, 6.62 mg/l, 3.23 mg/l.

REZUMAT: Ecotoxicitatea nano particulelor la *Daphnia magna*.

Dezvoltarea nanotehnologiei în ultimii ani şi potentialul de toxicitate a acestora constituie subiectul prezentei lucrari. Pentru a evalua potenţialul impact toxic al nanoparticulelor în mediu acvatic, au fost folosite trei feluri de nanoparticule cum sunt dioxidul de titan (TiO_2), oxidul de cupru (CuO) şi oxidul de zinc (ZnO), pe o specie-model acvatică, *Daphnia magna*. *Daphnia magna* a fost expusă la diferite concentraţii timp de 24, 48, 72 și 96 de ore la 20-35°C. Parametri de calitate a apei ca: temperatura, pH-ul şi oxigenul dizolvat au fost controlaţi pentru a cunoaşte cerinţele standard în timpul experimentului. Valorile LC_{50} pentru 24, 48, 72 şi 96 ore au fost estimate statistic cu ajutorul metodei Probit. Valorile de LC_{50} la 48 de ore pentru TiO_2, CuO și ZnO au fost de 171.88 mg/l, 6.62 mg/l, 3.23 mg/l la respectivul experiment.

INTRODUCTION

In the past decade, there was less concern and attention about nanoparticles (NPs) and nanomaterials (Lovern and Klaper, 2006; Masciangioli and Zhang, 2003). In recent years, however, by finding more evidence about the potential impact of NPs, they have come under scrutiny. It seems that NPs can enter into the aquatic ecosystem as a result of their proposed use in industries (Fabrega et al., 2013).

Daphnia is known as a bio-indicator used by various organizations around the world and is suggested by U.S. Environmental Protection Agency (U.S.EPA) for bioassay test. *D. magna* can be utilized as an aquatic bio-indicator because this species is able to filter an average of 16.6 ml/h (McMahon and Rigler, 1965), and consequently has a great potential to be affected by pollutant particulates such as NPs. Nanoparticles have the capability to be absorbed during filtration, hence have an impact on feeding ability. Because *D. magna* has a key role in the aquatic food chain between the algae and fish which feed on them, it is necessary to find out the toxic response and effect of NPs on *D. magna* (Lovern and Klaper, 2006).

Several studies performed on the eco-toxicity of TiO_2 NPs have already showed that these NPs have low toxicity (Ozkan et al., 2015; Johari and Asghari, 2015; Zhu et al., 2010). In particular, Heinlaan et al. (2008) studied the toxicity of NPs, such as ZnO, CuO, and TiO_2 on *Vibrio fischeri*, *Daphnia magna*, and *Thamnocephalus platypus*, and demonstrated that TiO_2 NPs did not have any toxicity in low concentrations (below 20 mg/l). Similarly, Griffitt et al. (2008) could not find any evidence for TiO_2 toxicity in standard eco-toxicological tests with zebra fish, *Daphnia*, and algae.

Considering this fact that *D. magna* has a key role in the aquatic food chain, studying the toxicity of NPs on this organism would offer important insights into a broad impact of NPs in the aquatic environment.

The main aim of this study was to assess the potential toxic effects of NPs on freshwater environments. For this purpose, three different NPs including titanium dioxide (TiO_2), copper oxide (CuO), and zinc oxide (ZnO) were used on an aquatic model species, *Daphnia magna*.

MATERIAL AND METHODS

Zinc oxide (ZnO), titanium dioxide (TiO_2), and copper oxide (CuO) nanoparticles were purchased from U.S. Research Nanomaterial's Inc., Houston, TX, USA. The physical properties of NPs are listed in table 1. The NPs stock solutions were prepared by suspending selected NPs powders in deionized water at a stock concentration of 20% (w/v). Vortexes for 20 s at 2000 rpm were used for homogenizing the suspension (Ozkan et al., 2015; Ates et al., 2013). The stock suspension was transferred immediately into the 500 ml beaker which contained *D. magna*.

Table 1: Size distribution and other characteristics of nanoparticles.

Nano Particles	APS	SSA	Purity	Color
CuO	40 nm	~ 20 m^2/g	99%	black
ZnO	10-30 nm	20-60 m^2/g	+ 99%	milky white
TiO_2	20 nm	10-45 m^2/g	+ 99%	white

SSA: Specific Surface Area, APS: Average Particle Size.

Daphnia magna was obtained from the health faculty of Shahid Beheshti University. *Daphnia magna* was maintained at a constant temperature (22 ± 2ºC) with natural light-dark cycle.

Some of the water quality parameters, including pH, DO, EC, and temperature were measured in each test during the experiment. The temperature was measured using a digital thermometer. The solution pH was measured using a ColeParmer Model 5398-00 digital pH meter. DO was measured using a sensefon 378 digital model. The physicochemical characteristics of the test water are presented in table 2.

Acute toxicity test was conducted according to the Organization for Economic Cooperation and Development testing guidelines 202 guidelines (OECD, 2004). In the primary test, to quantify the NPs concentration, *D. magna* was exposed to 0.05, 0.01, 1.00, 10.00, 100.00, and 200.00 mg/l in 96 h and the mortality was recorded in each treatment. In this stage, we used 15 individuals of *D. magna* in 0.5 litter beaker. This method was used for all three NPs. Then, totally 45 *D. magna* were exposed to different concentrations of the NPs for 24 h, 48 h, 72 h, and 96 h. After that, based on mortality rate in the first stage, different concentration was made for each NP.

Based on the primary test, the concentrations of ZnO and CuO NPs were same as of 1, 3, 5, 7, 9, and 10 mg/l. However, for TiO_2 NPs, the test concentrations were 100, 120, 140, 160, 180, and 200 mg/l. A control group was also set up without the test compound, using only fresh water. Exposures concentrations were carried out in triplicate groups in 500 ml fresh water.

Slight aeration was provided through the bottom of beaker for *D. magna*, to prevent settling of NPs in the beaker. For all test groups, the light pattern was 16:8 h light: dark. All tests were conducted in the absence of food in all-time periods. *D. magna* was exposed to selected NPs for 24, 48, 72, and 96 h. At the end of each period, live *D. magna* were counted.

The LC_{50} values (with 95% confidence) were calculated by the Probit Software (Zhu et al., 2008; Strigul et al., 2009; Ranjbar et al., 2011; Khoshnood et al., 2014). Significant differences between the control and all experimental samples were determined using the Bonferroni nonparametric post hoc tests, where $p < 0.05$ was considered to be significantly different. SAR (Safe Application Rate) and SAFE (Safety Factor) coefficients were also calculated by 96 h acute toxicity test. The SAFE formula was used as indicated in Formula 1 (Basak and Konar, 1977; Jaafarzadeh et al., 2013).

Formula (1):

$$SAFE = \frac{\mathrm{LC}_0 \ at\ 96\ hours}{\mathrm{LC}_{100} \ at\ 96\ hours}$$

Formula (2): $\mathrm{SAR} = (96 \text{ hours } \mathrm{LC}_{50}) \times \mathrm{SAFE}$

Table 2: Physico-chemical properties of the test.

Characteristics/Parameter	Range	Mean ± SD
Room Temperature	27.8 – 29.1 (˚C)	28.3 ± 0.5
Water Temperature	22 – 22.3 (˚C)	22.1 ± 0.1
Dissolved Oxygen	6.9 – 7.2 (mg/l)	7.2 ± 0.12
pH	7 – 7.4	7.1 ± 0.1

RESULTS

No mortality was observed in the control group during the experiment. The toxicity of NPs to *Daphnia magna* was increased with increasing concentration of selected NPs and duration the exposure ($p < 0.05$). Mortality percent of the *D. magna* in each test are presented in table 3.

Table 3: Mortality Percent (%) of the *D. magna* after exposure to NPs.

NPs	Concentration	Time			
		24 h	48 h	72 h	96 h
ZnO	Control	0.0	0.0	0.0	0.0
	1	11.1	28.8	48.9	68.9
	3	26.7	42.3	71.1	95.6
	5	35.5	53.3	77.8	97.8
	7	48.9	64.4	86.7	100
	9	64.4	77.8	95.6	100
	10	64.4	82.2	95.6	100
CuO	Control	0.0	0.0	0.0	0.0
	1	11.1	13.3	17.8	26.7
	3	22.3	22.3	33.4	44.5
	5	33.4	37.8	46.6	55.5
	7	46.7	51.1	60.0	68.8
	9	55.6	60.0	66.7	82.2
	10	60.0	66.7	77.8	91.2
TiO_2	Control	0.0	0.0	0.0	0.0
	100	6.6	11.1	22.2	28.8
	120	13.3	20.0	33.3	42.3
	140	17.7	26.6	46.5	60.0
	160	35.6	44.5	64.5	77.8
	180	46.6	55.5	73.3	93.3
	200	55.5	64.5	86.6	100

The results of 57 acute toxicity tests (54 cases and three controls), performed with TiO_2, CuO, and ZnO, expressed as LC_{10}, LC_{50}, and LC_{90} values, are summarized in table 4 and figures 1 to 3.

The controls showed no mortality in 24, 48, 72, and 96 h. As mentioned above, all the exposures tests were conducted in the absence of feeding. No mortalities in the control treatment could describe the fact that food lack did not induce any lethal effects on this species even up to 96 h. In 24 h, the average mortality ranged from 11.1% (one mg/l) to 64.4% (9, 10 mg/l) for ZnO NPs, from 11.1% (one mg/l) to 60% (10 mg/l) for CuO NPs, and from 6.6% (100 mg/l) to 55.5% (200 mg/l) for TiO_2 NPs.

The average mortality in 96 h exposure was about 68.9% in one mg/l suspensions of the ZnO NPs and increased to 100 % in 7, 9, 10 mg/l suspensions. Likewise, the mortality rate of CuO NPs was 26.7% in one mg/l suspensions and increased to 91.2% in 10 mg/l suspensions. For TiO_2 NPs, the average mortality was about 28.8% in 100 mg/l suspensions and increased to 100% in 200 mg/l suspensions.

These results of mortality percent, point to the fact that both TiO_2 and CuO NPs showed moderate toxicity to *Daphnia magna* at 96 h of exposure compared to CuO regardless of their size and concentration. The toxicity pattern in mortality percent of metal oxides to *D. magna* at 24 h of exposure was in the order of ZnO > CuO > TiO_2.

In the present study, CuO NPs were found to have a 24 h LC_{50} of 7.85 mg/l, for ZnO LC_{50} at 24 h was found to be 6.58 mg/l, and for TiO_2 was 188.99 mg/l which was around 28 times higher than that of ZnO NPs and around 25 times higher than that of CuO NPs.

Formula 1 and 2 were used to calculate the safety factor of SAFE and SAR coefficients. For CuO NPs, SAFE and SAR were 0.001 and 0.002, respectively. By considering the obtained value of SAR (0.002 mg/l), it could be seen that effluents with this SAR could enter the water body without any concerning about mortality in some sensitive organisms such as *D. magna*. SAFE and SAR values were 0.010 and 0.006 for ZnO NPs, respectively; showing it could not be very harmful at this level for some aquatic organisms such as *D. magna*. Table 5 shows SAFE and SAR values.

Table 4: The toxicity (mg/l) of NPs on *D. magna*.

Nano Particles	Time	LC_{10}	LC_{50}	LC_{90}
CuO	24 h	1.16	7.85	53.23
	48 h	1.06	6.62	41.32
	72 h	0.74	4.66	29.30
	96 h	0.53	2.99	16.74
ZnO	24 h	1.12	6.58	38.66
	48 h	0.40	3.23	25.99
	72 h	0.28	1.19	7.71
	96 h	0.20	0.64	2.05
TiO_2	24 h	113.18	188.99	315.50
	48 h	100.36	171.88	294.35
	72 h	85.69	139.14	225.91
	96 h	76.92	123.79	176.30

Table 5: SAFE and SAR Coefficient's values.

Nano Particles	SAFE	SAR
CuO	0.001	0.002
ZnO	0.010	0.006
TiO_2	0.270	33.420

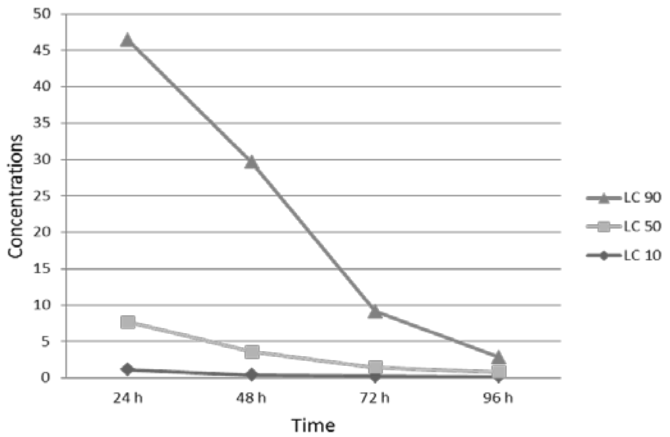

Figure 1: Acute toxicity (mg/l) of ZnO NPs in *Daphnia magna*.

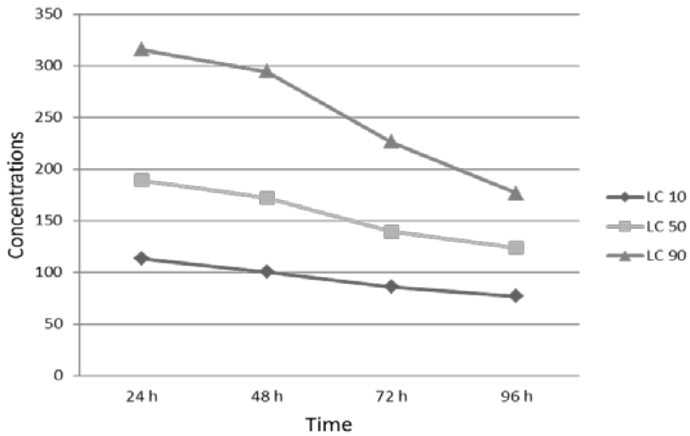

Figure 2: Acute toxicity (mg/l) of TiO_2 NPs in *Daphnia magna*.

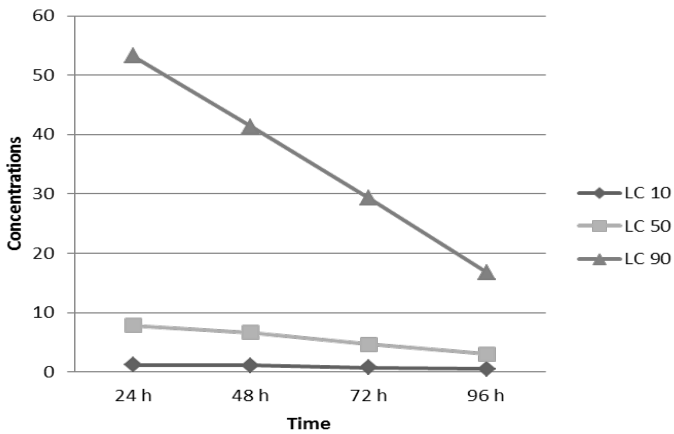

Figure 3: Acute toxicity (mg/l) of CuO NPs in *Daphnia magna*.

Table 6 shows the results of some other studies about the toxicity of NPs on *D. magna* which can be used to compare with our findings. In the present study, the LC_{50} value for the acute toxicity tests of ZnO NPs on *D. magna* at 48 h was 3.23 mg/l. For the same species, Heinlaan reported a 48 h LC_{50} acute toxicity value of 3.20 mg/l for the acute toxicity of Zn (Heinlaan et al., 2008), which is close to the LC_{50} values obtained in the present study. On the other hand, Liu et al. (2014) reported toxicity value for 48 h LC_{50} as 6.32 mg/l, that are two times more than the results of this study and those obtained by Heinlaan et al. (2008). Also, Zhu et al. (2008), Wiench et al. (2009), and Lopes et al. (2014) found 48 h LC_{50} value of 1.511 mg/l, one mg/l, and 1.10 mg/l respectively, which are about three times lower than the present results.

About TiO_2, the 48 h LC_{50} of *D. magna* was 171.877 mg/l. In addition, Zhu et al. (2008) (Zhu et al., 2008) reported this value as of 143.387 mg/l which is close to the results of the present study.

For CuO NPs, the calculated 48 h LC_{50} was 6.624 mg/l in the present study, however Luiet al. (2014) and Heinlaan et al. (2008) reported this value as 5.56 mg/l and 3.2 mg/l, respectively (Liu et al., 2014; Heinlaan et al., 2008).

Based on our results, any difference between toxicity thresholds may be related to the particle size differences, analyzing and preparation methods, or test designs and inconsistent test conditions such as pH, photoperiod, and dissolved oxygen (DO) (Wang et al., 2016). For example, the diameter of the TiO_2 used in Zhu et al. (2008) study was 20 nm, but in the present study, it was 10-30 nm, and about other NPs, the actual size of the TiO_2 NPs in this study was 20 nm and in Heinlaan study was 50-70 nm. The toxicity of metal oxide NPs may be partially due to the release of metal ions. Blinova et al. (2010) reported that the toxicity of *n*CuO (nano CuO) and *n*ZnO (nano ZnO) in natural waters resulted mainly from dissolved metal ions (Blinova et al., 2010).

In some cases, NPs were observed in the *Daphnia* intestine. Aquatic animals such as *D. magna* may consume or eat NPs, mistaking them for food. In this situation, NPs agglomerate further in the digestive system, and finally block it and cause death (Strigul et al., 2009; Wiench et al., 2009).

Cytotoxicity of metal oxide NPs to mammalian cells is strongly influenced by the dissolution of those NPs (Brunner et al., 2006). However, Baek et al. (2011) reported the toxicity of four metal oxide NPs and found that the toxicity induced by the dissolved ions was negligible (Baek and An, 2011). The toxicity mechanism of metal oxide NPs is very complex and has not been completely defined (Hai-zhou et al., 2012).

Table 6: Comparing acute toxicity test of nanoparticles on *D. magna*.

			ZNO (ML^{-1})	TIO_2 (ML^{-1})	CUO (ML^{-1})	
D. magna	Acute toxicity Nano size and bulk	TiO_2, CuO, TiO_2	3.2	LC > 2000	3.2	(Heinlaan et al., 2008)
D. magna	Acute toxicity	ZnO, TiO_2, Al_2O_3, C^{60}, SWCNTs*, MWCNTs**	1.511	143.387	–	(Zhu et al., 2008)
D. magna	Acute toxicity, chronic toxicity, mobility, reproduction	TiO_2, ZnO (Nano and non-Nano)	1	LC > 50	–	(Wiench et al., 2009)
D. magna	Acute toxicity	TiO_2, Al, B	–	LC > 250	–	(Strigul et al., 2009)
D. magna	Acute toxicity, chronic toxicity, NPs size, mobility	ZnO	1.10	–	–	(Lopes et al., 2014)
D. magna	Acute toxicity, Size	ZnO, CuO, Au, TiO_2	6.73	–	5.66	(Liu et al., 2014)
D. magna	Acute toxicity, Size	TiO_2	–	LC > 100	–	(Johari and Asghari 2015)
D. magna	Chronic exposure	CuO, $CuCl_2$, $_2H_2O$	–	–	1.041	(Adam et al., 2015)
D. magna	Acute toxicity	TiO_2, CuO, TiO_2	3.232	171.877	6.624	This study

* Single-walled carbon nanotube

** Multiple-walled carbon nanotube

CONCLUSIONS

The results of this study showed that LC_{50} 96 h TiO_2 nanoparticles were much less toxic to *D. magna* than CuO and ZnO NPs. Based on our results in this study, it can be concluded that:

- The selected NPs in the present study (CuO, ZnO, and TiO_2) may have acute dose-dependent eco-toxicological effects on *D. magna*.
- NPs with different compositions or different size show different toxicities effects on aquatic organisms.
- NPs toxicity effect could be as a result of the NPs properties, dissolution methods, and NPs agglomerates which were developed during the test.
- The results of this study indicated that the potential eco-toxicity and environmental health effects of NPs should be given due attention.

ACKNOWLEDGEMENTS

The authors are grateful to Dr. Zareie and Mrs. Moebed and other KWPA Central Laboratory staff for their valuable assistance in laboratory experiments and especial thanks due to the staff of all laboratories who participated in the inter-laboratory test and data generation.

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TRAPPING ALIENS: UNDERSTANDING THE COMPLEXITIES OF CONTROLLING INTRODUCED FRESHWATER CRAYFISH IN THE UNITED KINGDOM

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DOI: 10.1515/trser-2015-0086

KEYWORDS: *Pascifastacus leniusculus*, signal crayfish, invasive, management, trapping, stakeholder, crayfish.

ABSTRACT

Invasive non-native species pose practical and ethical problems for the people tasked with their management. Invasive freshwater crayfish species in the UK threaten rare native crayfish and freshwater habitats, yet their control is beset with social, practical and environmental barriers to success. This paper draws on an interdisciplinary study of stakeholders involved in crayfish management in East Anglia to explore the management of non-native freshwater crayfish in the UK. It concludes that when standard methods of control fail, stakeholders are willing to consider unusual management suggestions such as commercial trapping, whilst recognising that these may bring their own problems.

ZUSAMMENFASSUNG: Fremdländische Arten einfangen: zum Verständnis der Komplexität der Kontrolle zugewanderter Süßwasserkrebse in Großbritannien.

Invasive, zugewanderte Arten stellen für diejenigen, die mit ihrem Management betraut sind, praktische und ethische Probleme dar. So gefährden die invasiven Süßwasserkrebsarten Großbritanniens seltene, heimische Flusskrebsarten und deren Habitate und dennoch ist deren Kontrolle und Erfolg umgeben von Hindernissen aus dem sozialen, praktischen und dem Umweltbereich. Vorliegende Arbeit umfasst eine interdisziplinäre Studie von Interessenvertretern, die in das Management des Flusskrebses in Ost-Anglia eingebunden sind, um die Behandlung der invasiven, fremden Arten zu erforschen. Aus den durchgeführten Untersuchungen folgern sie, dass bei Versagen der Standard-Kntrollmethoden auch unübliche Managementvorschläge in Betracht gezogen werden können, wie zum Beispiel wirtschaftliches Fangen, solange zugegeben wird, dass diese auch ihre eigenen Probleme mit sich bringen können.

REZUMAT: Prinderea în capcană a străinilor: înţelegerea complexităţii controlului introducerii racilor de apă dulce în Regatul Unit.

Speciile invazive pun probleme practice și etice persoanelor responsabile cu gestionarea acestora. Speciile invazive de raci din Marea Britanie amenință racii de apă dulce nativi, foarte rari, şi habitatele lor, şi totuşi controlul lor este blocat de bariere sociale, practice și de mediu. Această lucrare se bazează pe un studiu interdisciplinar al părților interesate implicate în managementul racilor din Anglia de est, pentru a explora gestionarea racilor de apă dulce non-nativi, din Marea Britanie. Se ajunge la concluzia că, atunci când metodele standard de control eșuează, părțile interesate sunt dispuse să ia în considerare sugestii de gestionare neobișnuite, cum ar fi capturarea comercială, recunoscând în același timp că aceasta poate aduce propriile ei probleme.

INTRODUCTION

Managing invasive non-native species (INNS), species which have been introduced by human action outside of their range and which pose a threat to biodiversity in their new locale (Defra, 2008), is a key environmental management challenge, as they are regarded as an internationally important issue for conservation (Millennium Ecosystem Assessment, 2005; Hall, 2003). Invasive NNS are a primary driver of ecological degradation through competitive interactions such as predation, resource competition, habitat alteration, disease transmission and hybridisation (Manchester and Bullock, 2000; Clout, 2002). INNS cause economic as well as nature conservation problems, through reduced harvests of commercial species and the costs of combating invasions through quarantine, control and eradication (Manchester and Bullock, 2000; Mack et al., 2000). INNS are therefore a major focus for environmental management with increasing investment in local, national and regional programmes, such as the Great Britain Non-Native Species Secretariat (NNSS) and the EU's Delivering Alien Invasive Species Inventories for Europe programme (DAISIE) (Roy et al., 2014).

Managing invasive species typically requires co-operation between many stakeholder groups, which can be challenging as they can have different priorities and interests, making collaboration difficult (Stokes et al., 2006). Co-operation between stakeholders (scientists, conservation managers, commercial, industrial and public sectors) is therefore necessary for such efforts to be sustainable in the long term (Stokes et al., 2006). This issue is especially pertinent to freshwater environments, which are particularly vulnerable to invasion (Moorhouse and Macdonald, 2015), both due to the interconnected nature of many aquatic habitats (Ramsar, 2003) and because they often have complex management systems as a result of multiple stakeholder and user groups with different rights and responsibilities (Stokes et al., 2006; EA, 2009).

Therefore, understanding attitudes and perceptions to INNS management is very important. However, scientific discourses often ignore factors such as values and interests, despite the fact that these often shape discussions and drive engagement with such topics (Gozlan et al., 2013; Selge et al., 2011). Public attitudes towards management of wildlife in the UK indicate a general support for eradicating INNS posing a threat to human health, native species or causing economic damage, although the level of support is context-dependent and tends to be higher for high-profile species such as red squirrels (Bremner and Park, 2007; Defra, 2009). However, public opposition to INNS control can delay interventions to the point that eradication becomes impossible (Selge et al., 2011).

Managing invasive crayfish in the United Kingdom

The UK has well-established populations of non-native crayfish species, dating back decades. Freshwater crayfish are a diverse group of decapod crustaceans, many of which are endangered (Crandall and Buhay, 2008). However, a small number of species are widely distributed outside their native range, primarily through the aquaculture, pet and live bait trades (Holdich et al., 2009). Their roles as key organisms in many food webs make their introduction significant (Lindqvist and Huner, 1999). In Europe, invasive non-native crayfish are responsible for major shifts in ecosystem structure and dynamics (e.g. Gutierrez-Yurrita et al., 1999). The UK has one native crayfish species, the white-clawed crayfish (*Austropotamobius pallipes*), currently protected under Annex II of the EU Habitats Directive (JNCC, 2010) and seven invasive species, the most widespread of which is the signal crayfish (*Pacifastacus leniusculus*) (Holdich et al., 2009).

Collapse of commercially important native European crayfish from crayfish plague (*Aphanomyces astaci*) in the 1970's led to widespread introduction of resistant non-native species as replacements (Lowery and Holdich, 1988). The UK lacked a commercial fishery for crayfish but signal crayfish were introduced for export to European markets (Lowery and Holdich, 1988; Holdich et al., 1999a). Most farms failed and were abandoned. Signal crayfish populations adapted well to the UK, expanding rapidly through well-used and interconnected waterways (Lowery and Holdich, 1988). Declining habitat quality was also identified as an important reason for the wide spread of signal crayfish in the UK, as it is recognised that poor-quality habitats are more vulnerable to invasion by INNS (Manchester and Bullock, 2000; Mack et al., 2000).

Signal crayfish today rank as one of the best-known and most regularly mentioned INNS by environmental managers in freshwater environments (Williams et al., 2010; Gozlan et al., 2013), and are thought to cost the UK (in terms of lost revenue from water-users and anglers, costs for removal and for habitat restoration and protection for the native white-clawed crayfish) around £1,502,000 per annum (Williams et al., 2010). In addition, their population continues to expand across the UK (NNSS, 2011). Thus, signal crayfish are a major issue for freshwater environmental managers.

This paper explores the complexities of managing signal and other invasive crayfish species in the UK from a stakeholder perspective. It focuses on three key challenges for stakeholders involved in the management process which range across social and natural themes: selecting the most appropriate control method; implementing and enforcing existing legislation; and building partnerships between different stakeholder groups. The paper concludes with a brief discussion about the potential for using harvesting as a control mechanism, following the growing cultural interest in "wild foods" (Reyes-Garcia et al., 2015).

METHODS

This paper draws on fifteen in-depth interviews carried out between June and August 2010 with river managers, crayfishers and academics working on freshwater environments in East Anglia. This area was selected because non-native crayfish are a significant issue in the region and several local angling clubs and other organisations have worked on crayfish management.

A local researcher and angler was approached to act as gatekeeper for contacts, and the researcher met them several times in order to discuss the project and its aims (Valentine, 2005). A snowball technique was used to recruit contacts, from which the author selected a sample of interviewees including national and local government bodies (n = 3), navigation authorities (n = 1), local authorities (n = 2), local angling or river clubs (n = 3), academic NNS researchers (n = 3) and unaffiliated stakeholders including crayfishers and landowners affected by crayfish (n = 3), who could provide insights into the issues and opinions surrounding management of non-native crayfish.

A list of key questions was used to guide interviews, provided in advance if requested (Valentine, 2005; Robson, 2007). During the interview, if a new topic was introduced, it was followed up, and the key questions for subsequent interviews adjusted to include it. Most interviewees were interviewed in their offices or homes, with two interviewed by telephone. All interviews were recorded on a digital voice recorder, and stored as mp3 files on a computer. Transcription by the author took place as soon as possible after the interview, to allow notes, ideas and clarification to be included on the transcript, and to familiarise the author with the material (Crang, 2005). The format of the transcript was an interview report, rather than a verbatim transcript (Crang, 2005) permitting interviewees to more easily see whether their ideas had been well recorded (Jones, 2008). The analysis and results are presented using quotations from interviews woven into a synthetic narrative about signal crayfish management and stakeholder interactions (Adams et al., 2004; Jones, 2008). Where necessary, quotes have been edited with square brackets to maintain anonymity or coherence.

Note on language: Interviewees used different phrases to describe NNS, e.g. “invasive species” or “alien species”, and these have been quoted verbatim. The author has used non-native species (NNS) or invasive non-native species (INNS) throughout the rest of the text. This latter term refers explicitly to species that, on their arrival in a new area, are regarded as causing significant issues or other species, or which expand rapidly.

RESULTS

Choosing methods of control

The first major challenge in controlling signal crayfish is selecting a method to control them with. Signal crayfish are tough, omnivorous and mobile, tolerant to the UK’s temperate climate and able to live in a diverse range of habitats (NNSS, 2011). A variety of methods are used to control non-native crayfish, including mechanical removal, poisoning or biocide and physical barriers (Holdich et al., 1999b; Freeman et al., 2010; Stebbing et al., 2014).

Biocide applications have not been widely used in the UK, but early trials in enclosed water bodies like ponds have been successful (Peay et al., 2006). It is possibly the only method that will completely eradicate crayfish as long as 100% of crayfish are killed (Peay, 2001). Unfortunately, signal crayfish are resistant to many biocides and pesticides (NNSS, 2011). Two stakeholders (both regional environmental management officials) who favoured biocide as a control method generally agreed that biocide applications are only appropriate in certain places, generally as a spot treatment, because the toxins used are not crayfish-specific, and in large water bodies like rivers the collateral damage is high (Holdich et al., 1999b; Peay, 2001). However, other stakeholders, particularly anglers, were not happy about the use of biocides with one angler commenting that using biocides was deliberately “creating an ecological incident” (interview with angler). Stakeholders who were interested in native crayfish conservation (a mix of officials, anglers and researchers) also expressed concern that biocide would have no effect on crayfish plague (carried as a latent infection by signal crayfish but lethal to white-clawed crayfish, NNSS, 2011), so its use in creating white-clawed crayfish refuges might be limited.

Several stakeholders mentioned that work was proceeding on finding a control method based on pheromone trapping, although no one mentioned any successes (Holdich et al., 1999b; Stebbing et al., 2002). This is an area recommended in the NNSS risk assessment for signal crayfish as an area in need of work. In terms of a wider focus on biological controls on signal crayfish, native predators are known to have a critical role to play in controlling populations of NNS (Rabeni, 1992; Hill and Lodge, 1994; Juliano et al., 2010). Hein et al., (2007) found that reducing fishing pressure on predatory fish contributed to crayfish population reductions. Non-native crayfish in Spain and France support populations of predatory species like otters, eels and bitterns, which in turn act as a control on crayfish population sizes (Musseau et al., 2015; Barrientos et al., 2014; Poulin et al., 2006; Blanco-Garrido et al., 2008). In the UK, native crayfish predators include otters (*Lutra lutra*) and carnivorous fish like eels (*Anguilla anguilla*), both of which were identified by stakeholders as potentially important factors in controlling signal crayfish populations, with one trapper saying "to some extent, crayfish have been good news for otters, as they eat them … if otter populations got big enough, it could be very useful" (interview with crayfish trapper). However, otters also pose a threat to fishing operations, as an environmental management official explained: "They (crayfish) can supplement the diet of otters when they fail to catch fish, so the energy pressure on otters which acted as a (population) control is lost, so otters can keep chasing fish" (interview with regional environmental management official).

The use of barriers to prevent expansion of signal populations up- and downstream has been applied in a few cases (e.g. Reeve, 2004; Kerby et al., 2005), but is complex, especially over long time periods, and can cause other issues such as access for migratory fish and other species. Other habitat alteration ideas include creating chemical barriers to signal crayfish expansion, e.g. by altering water acidity/alkalinity. One such case study on the River Clyde, Scotland (Reeve, 2004), was widely cited by stakeholders interviewed. Most acknowledged that there would probably be side effects on other species, but felt that this negative would be outweighed by the benefits of a reduction in crayfish populations: "If, by changing the pH, we reduce the number of crayfish, and everything else improves, overall this is good" (interview with local environmental authority official).

Studies on the dewatering of burrows (Peay and Dunn, 2014) suggest that whilst it can remove signal crayfish from an area, they are able to cross dry land, and dewatering encourages such movement, potentially increasing their colonisation of new areas.

The easiest method to control crayfish is via baited traps laid on riverbeds. Trapping is widely accepted not to eradicate crayfish populations, as traps can be selective for large males, and removal of large males increases the recruitment of juveniles, leading to a sustained population (NNSS, 2011; Momot, 1998; Hein et al., 2007). Most stakeholders cited this research, with one neatly summarising it by saying that trapping: "Removes the large cannibalistic individuals and promotes dispersal of smaller individuals" (interview with national invasive species advisor).

However, intensive trapping over several years causes a decline in catch rates over time, which, whilst not eradicating signal crayfish, does maintain a low population and allow macrophyte, invertebrate and fish populations to increase, which are all negatively affected by large crayfish populations (Hein et al., 2007). This effect was observed by many of the anglers and trappers interviewed for this study. Members of a local angling club which has trapped signal crayfish intensively for some years have observed improvements to their river environment, with more fish fry, insects and small fish. Individuals involved in long-term trapping commented that "I have trapped this area solidly for five years, and I have noticed a reduction in population and size … I do think I have made a huge difference to the area" (interview with trapper) and "If I do a second day (of trapping), I tend to notice a falling off. I have fished for three days in a row, and had very few on the third day" (interview with trapper).

Trapping is a long-term control method – unless trapping pressure is maintained, populations will recover and continue to expand (Hein et al., 2007). An angler involved with his local angling club's trapping project indicated that they had thought hard about starting, because "when we started, we knew it was not going to be a short term fix, but a long term commitment" (interview with angler).

The conclusion from this section, and from reports such as the NNSS risk assessment for signal crayfish (NNSS, 2011), is that there is no one method that will successfully deal with signal crayfish. Instead, each affected area has to work within environmental and practical limits – if possible, to use extreme methods such as biocide to remove signal crayfish, but where that is not practical to focus on developing new methods such as pathogen-based controls (Freeman et al., 2010) or use time and personnel-intensive methods like trapping.

Bureaucracy, licensing and enforcement

Carrying out control measures against NNS is often fraught with uncertainty, but failure to act quickly following the initial arrival of INNS can result in serious problems later on (Coblentz, 1990). In the case of signal crayfish, the legislative environment in which it was first introduced was also an important factor, as one NNS expert commented: “The lesson has been learnt from signal crayfish – there are not always happy endings with invasive species – they need to be caught early and hard to be effective” (interview with national invasive species advisor).

INNS legislation is often piecemeal and designed in response to a particular problem, and thus of variable effectiveness (Fasham and Trumper, 2001). Following criticisms around existing UK legislation on INNS, a Defra policy review was carried out in 2003 (Defra, 2003), from which the GB NNS Strategy was developed (Defra, 2008). The Strategy adopts a precautionary approach, aiming to better balance reactive management with preventative work such as species alerts and horizon scanning (Defra, 2008), and has also focused on awareness raising campaigns to reduce inadvertent introductions of potentially invasive species by focussing on good management practices, e.g. “Be Plant Wise” for invasive pond weeds (NNSS, 2009).

Rapid response infrastructure in the UK is being developed, especially for invasive aquatic species like the top-mouth gudgeon (NNSS, 2010), and would be potentially valuable against new invasive crayfish species, e.g. *Procambarus fallax* forma *virginalis* (Scholz et al., 2003; CABI, 2015). However, this is too late for signal crayfish: “If we had had a rapid response capacity with signal crayfish (in the 1970s), we could have eradicated them when they were still only in ponds” (interview with regional invasive species co-ordinator).

Signal crayfish legislation is still somewhat confusing, as one stakeholder explained: “Requests for (signal) crayfish trapping go to the EA (Environment Agency) … requests to do research trapping (e.g. capture-mark-recapture) (on white-clawed crayfish) have to go to Natural England, who implements the Wildlife and Countryside Act 1981 … (and) enforcement of the Act (Wildlife and Countryside Act 1981 which makes it illegal to introduce non-native crayfish to the UK) is through the EA. Additionally, CEFAS and Defra are involved” (interview with regional environmental manager).

The Environment Agency requires individual trapping for crayfish to have a license (EA, 2010a), designed to prevent illegal trapping, or trapping in sensitive areas. Stakeholders holding licences were critical of the process, saying that the relevant sections on the Environment Agency website were hard to find, and that the licence form was hard to fill in: “On the form there is a bit for landowner permission and location … I contacted the EA Properties department to ask how I could find out about landowners. They had no idea and suggested I went knocking on doors” (interview with angler). Others commented that enforcement of license renewal was poor: “I am meant to have a Defra license, and did have one for two years, but now I can’t be bothered, as no one has said anything” (interview with trapper).

Further, trappers who had complied with the license requirement to give catch returns to the Environment Agency (EA, 2010b) were frustrated by a lack of feedback, disincentivising them to renew their license: “I sent the EA my catch data and never got any acknowledgement, so what is the point in doing it? ... I would have been very interested to see what it showed” (interview with trapper).

Currently, trapping is only licensed in certain “go” areas where there are no populations of the protected white-clawed crayfish (*Austropotamobius pallipes*) (HMSO, 1996). A major concern for many stakeholders was the potential for confusion between white-clawed and signal crayfish. Stakeholders mentioned that many people didn’t know about white-clawed crayfish, and indicated that because the licensing system does not require training in crayfish identification, and does not clearly indicate the “go” and “no-go” areas for fishing, mistakes were more likely to happen (Peay, 2010). Improved identification skills were seen as key to solving this: “I have no problem with public fishing, as long as the people have had a degree of education. It should be conditional on getting a license, but it isn’t” (interview with angler).

Officials dealing with signal crayfish management also raised concerns over the association of trapping with bycatch of protected crayfish predators like otters: “We have a big problem with illegal fishing and otter bycatch. Whenever EA staff go out and remove traps, they will find dead otters” (interview with regional invasive species co-ordinator).

These issues are recognised by the fishing and trapping communities, with one trapper suggesting that the EA could improve its trap licensing process to reduce the risk of by-catch and illegal trapping by having: “a website that says where you can fish crayfish, what sort of trap you should use, they could even sell the nets” (interview with trapper).

All stakeholders concluded that stronger trapping regulation was necessary, with one official working on national invasive species strategy commenting that “the licensing system, whilst it is clunky, does allow some sort of regulation” (interview with national NNS co-ordinator).

The issues around dealing with legislation and enforcement of legislation are made more difficult by the diverse range of stakeholders involved in crayfish management. Engagement of local stakeholders (e.g. land-owners, anglers, etc.) can be really beneficial for government bodies strained financially: for example, the Local Action Groups organised and co-ordinated by the NNSS have been a successful model for public engagement in INNS control (NNSS, 2015). However, issues arise when enforcement is weak/ineffective (as observed here) or when opinions diverge on how things should be done. The following section delves into these issues further.

Expertise, partnership and interests

Improved co-operation and co-ordination helps to achieve long-term effective protection against invasions (Taugbøl and Skurdal, 1999; Stokes et al., 2006). Stakeholders mentioned increased local and national co-operation over NNS work as positive. Nationally, the establishment of the GB Non-Native Species Secretariat (NNSS) and Strategy has: “Cut through debates about whose responsibility it was to deal with invasive species, particularly between the EA and NE, and laid out how the work should be done” (interview with national advisor on invasive species).

The UK has an advantage over much of Europe, as its freshwater habitats are geographically isolated from mainland Europe, and has a body like the NNSS, which can co-ordinate horizon scanning and surveillance activities for potential INNS (Defra, 2008). This places the UK in a good position to react to new INNS arriving, although does not prevent new species arriving (10-12 new INNS become established every year in the UK, NNSS, 2015).

However, despite the advantages of a co-ordinated approach to INNS management, there remain uneasy relationships between some stakeholders. Some trappers felt that attitudes towards individuals like themselves within the policy world were not always positive: "The EA's attitude to fishermen (and their ability to identify things) is patronising" (Interview with angler). They felt that their expertise and experience were not fully appreciated: "There is a certain amount of elitism which has been a problem – so much information on crayfish has come from academics that it is only more recently that they are willing to listen to other people" (interview with trapper).

One trapper felt that many people working for official organisations had lots of qualifications but relatively little experience compared to people without an academic record but who have spent a long time living or working in an area. This is not a phenomenon restricted to the case study area: generally, those in policy favour information coming from scientific and academic sources over that from other sources (Beck, 1992; Leach and Scoones, 2005). In contexts where the success of a project may depend on the good-will and engagement of non-scientists such as trappers and anglers, this sort of attitude can lead to problems (e.g. Wynne, 1992).

However, it is not just about attitudes to engagement: sometimes agreeing on how to manage the environment can be difficult between different stakeholders (Selge et al., 2011). A good example of this is the potential conflicts of interest between anglers and environmental managers over freshwater INNS management options: the UK has seen an increase in "big game" angling, which has led to increasing demands for stocking of fish types (Hickley and Chare, 2004). Fish stocking both presents one of the key routes for signal crayfish to spread to new areas (the larvae and eggs can be transported with fish, NNSS, 2011), and the investment in this industry also becomes an issue when populations of predatory species like otters increases. The support offered to otter populations by large signal crayfish populations has not been seen positively by some anglers in the case study area: "(it is) very problematic – there was a case of two carp killed by otters … which weighed 35 and 39 lb, and cost £5000 each – the fisherman lost £10,000 of investment in total" (interview with regional environmental management official).

Officials said that anglers have demanded that otter populations be controlled, despite their positive effects on freshwater habitats and fish populations through their control of crayfish (Barrientos et al., 2015): "They (anglers) have run big campaigns against otters, especially over the last two-three years" (interview with regional environmental management official).

Thus, some of the solutions to controlling crayfish that are low-impact on the wider environment (i.e. supporting native predators, rather than using harmful methods like biocide or habitat alteration) can be as unpopular as the presence of the INNS themselves, providing serious challenges to those trying to manage the environment for many different users (Hickley and Chare, 2004).

Harvesting invasive crayfish – solution or not?

Having studied some of the attitudes and approaches of stakeholders charged with managing crayfish, I now return to the question of why signal crayfish were ever introduced, and whether it offers a possible solution going forwards. As discussed in the Introduction, signal crayfish were introduced to the UK as an aquaculture species for export to Europe for consumption (Holdich et al., 1999a). In the UK, the last decade or so has experienced a surge in interest in so-called "foraging" or wild harvest of species for consumption, encouraged by TV shows and haute cuisine (Reyes-Garcia et al., 2015; NNSS, 2011). Signal crayfish, as a numerous and widely-present crustacean with a history of consumption in other countries (Taugbøl, 2004), are an obvious target for foragers (e.g. Clay, 2011). The fact that removal of signal crayfish could be regarded as an environmental benefit enhances their attraction (along with consumption of other INNS like grey squirrels, e.g. Davies, 2008). Such an approach has been considered by Defra to control Chinese mitten crabs (*Eriocheir sinensis*) in the river Thames (NHM, 2008).

In this study, three of the six trappers interviewed trap because they enjoy eating crayfish: "I am fishing for food, not to conserve the white-clawed crayfish" (interview with angler). Stakeholders suggested that a market for wild-harvested crayfish might provide a financial incentive to trap, whilst simultaneously securing labour for projects trying to control crayfish populations.

Whilst in concept attractive, the reality of setting up a commercial enterprise is more difficult, for three reasons: firstly, the market for wild-caught freshwater crustaceans in the UK remains relatively undeveloped, making it hard to find local consumers. As one angler commented, "the biggest problem with them is not catching them but getting rid of them" (interview with angler).

Secondly, there is a contradiction between trapping for conservation, where the main focus is on reducing populations significantly, and trapping for commercial sale, where the interest is on large individuals that are saleable. As one trapper put it, "anything less than eight cm is not really commercial and not worth it" (interview with trapper).

A trapping project for conservation reasons which encourages people to participate by selling their catch is likely to fail over time as catch volume and crayfish size decline and the market value declines in parallel. As one angler pointed out, "a (commercial) trapper at that point would move on" (interview with angler). This is not an option for conservationists who need to maintain low population numbers in order that freshwater habitats can recover. Thus, co-operation and potential subsidies to compensate for declining body sizes and profit would be required for such a project to work long-term.

Finally, and most problematic, is the issue of signal crayfish spread, and the legal aspects of this. As a species listed under the Wildlife and Countryside Act 1981, signal crayfish are not permitted to be released into the wild, yet encouraging commercial harvesting brings the risk of deliberate introduction of signal crayfish to new areas (Gutierrez-Yurrita et al., 1999; Peay, 2010). The NNSS risk assessment for signal crayfish outlines a few case studies where signal crayfish have been introduced deliberately, or where people inadvertently trapped for the protected white-clawed crayfish (NNSS, 2011). Anything that encourages the expansion of signal crayfish or which affects white-clawed crayfish should be very carefully implemented, if at all (NNSS, 2011). Thus, it seems likely that although individuals may choose to consume locally-caught crayfish, the creation of a market that will act to fund the control of signal crayfish is likely to fail.

CONCLUSIONS

This paper has outlined a series of institutional and practical challenges and opportunities for signal crayfish management in the UK. There is no single method that will successfully control signal crayfish, reflecting both their habitat (interlinked freshwater environments with many sensitive species present) and their robust life-history. Therefore, as one angler commented: “We have no choice but to accept that the signal crayfish is here to stay” (interview with angler).

However, most stakeholders did not feel that ignoring signal crayfish is an option – loss of protected species like white-clawed crayfish, and continued degradation of freshwater habitats as a consequence of growing signal crayfish populations is likely to be problematic for the UK under the EU Habitats and Water Framework Directives (JNCC, 2010; EU, 2000).

The improved policy and legislative environment around INNS management has no doubt benefited national and regional efforts to control them, in part by drawing together government stakeholders, and supporting public awareness of INNS issues. However, this paper has also identified ongoing issues around stakeholder interactions, not least perceptions of groups such as anglers and trappers, who act often from environmentally-oriented reasons but who feel separated from policy and that their knowledge about river environments is not well-regarded (Van Eijck and Roth, 2007). Further, other parts of their communities can directly inhibit the efforts of environmental managers to control INNS, providing new sources of conflict. This small-scale case study is not alone in this, but provides a good example of the challenges facing environmental managers and those wishing to support the environment when trying to work together, not least that the science behind control is difficult and uncertain.

Commercial exploitation of signal crayfish through trapping is an unusual suggestion, and although trapping is widely held not to eradicate signal crayfish, intensive, organised trapping can benefit freshwater ecosystems, reducing signal crayfish numbers and allowing macrophyte, fish and invertebrate populations to recover (Hein et al., 2007; West, 2009). Whilst a system where trapped crayfish are sold to raise funds for control efforts, there are also a number of serious risks associated with such an idea. These are not helped by a confusing legislative environment. Should a commercial approach be used in controlling signal crayfish, it would need to be strictly regulated, and could be designed as a way of encouraging environmental citizenship, an approach successfully used in Europe to protect native crayfish from introductions of invasive species (Taugbøl and Skurdal, 1999).

The issues that emerged from this paper are relevant to wider biodiversity management. INNS are a part of wider issues facing conservation of freshwater ecosystems. Further work could seek to put perspectives on INNS management into a wider freshwater management context.

ACKNOWLEDGEMENTS

I’d like to thank everyone who kindly gave up their time and allowed me to interview them. I’d also like to thank Peter Jones at UCL Geography Department for supervising my MSc thesis which was the basis for this paper, and Carl Sayer, Sophie des Clers and Sam Randalls at UCL for useful discussions about different aspects of the project, especially during its development stages. This paper benefitted from discussions with Bill Adams, Anne Toomey, Beth Brockett, Anni Fiebig and the Lancaster Environment Centre writing group.

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THYMALLUS THYMALLUS (LINNAEUS, 1758), ECOLOGICAL STATUS IN MARAMUREŞ MOUNTAINS NATURE PARK (ROMANIA)

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DOI: 10.1515/trser-2015-0087

KEYWORDS: Grayling, habitats, human impact, assessment.

ABSTRACT

Thymallus thymallus is considered a species of significant protective importance within the Vişeu Watershed. The state of habitats characteristically inhabited by *Thymallus thymallus* within the Maramureş Mountains Nature Park is balanced between reduced (one third of the lotic sectors where the species was identified), average (one third) and good (one third). The excellent conservation status is currently missing for populations of this fish in the Vișeu Basin.

Human impact types identified as contributing towards the decreasing state of *Thymallus thymallus* habitats and therefore populations in the studied area in comparison with its natural potential are: poaching, minor riverbeds morphodynamic changings, solid and liquid natural flow changes, destruction of riparian trees and bush vegetation, habitat fragmentation/isolation of population, organic and mining pollution, and displaced fish that are washed away during flood periods in the lotic sectors uniformized by humans.

RESUMEN: Estado ecológico de *Thymallus thymallus* (Linnaeus, 1758) en el Parque Montañas Maramureș (Rumania).

Las características del habitat de *Thymallus thymallus* dentro del Parque natural Montañas Maramures es un balance entre estados degradado (un tercio de los sectores donde se ha identificado a la especie), promedio (un tercio) y bueno (un tercio). Para las poblaciones de este pez en la cuenca Vișeu, a la fecha, no existe un estado excelente de conservación.

Los impactos humanos que reducen las poblaciones de *Thymallus thymallus*, en relación con su potencial natural son: pesca ilegal, cambios morfodinámicos menores en las cuencas fluviales, cambios en el flujo natural de sólidos y líquidos, fragmentación de hábitat/aislamiento de poblaciones, contaminación orgánica y de minería y pérdida de peces durante los periodos de inundación en los sectores loticos que han sido uniformizados.

REZUMAT: *Thymallus thymallus* (Linnaeus, 1758), situația ecologică în Parcul Munții Maramureșului (România).

Starea habitatelor caracteristice pentru *Thymallus thymallus* din Parcul Natural Munţii Maramureşului oscilează între precară (o treime din sectoare în care specia a fost identificată), medie (o treime) și bună (o treime). Statutul de conservare excelent lipsește pentru moment la populațiile acestei specii în zona de referinţă – Bazinul Vişeului.

Tipurile de impact antropic identificate a fi responsabile de înrăutățirea stării habitatelor populației de *Thymallus thymallus* în zona studiată, comparativ cu potențialul său natural, sunt: braconajul, modificări ale morfodinamicii albiei minore, modificări ale debitelor naturale solid și lichid, distrugerea vegetației arbustive şi arboricole ripariene, fragmentarea habitatelor/izolarea populațiilor, poluare organică și cu ape de mină, evacuarea peștelui în perioadele de inundații în sectoarele lotice uniformizate antropic.

INTRODUCTION

The mountainous water is usualy of high quality in the condition of modest development of anthropic activities (Romanescu, 2016), activities which should be assessed in the local conservation issues context.

Stream and river ecosystems of the Maramureș Mountains Nature Park mostly belong to the Vișeu River basin and very little to the Bistrița Aurie Basin (Fig. 1), in the northern part of Romania. The Vișeu Basin is bound by the Maramureș Mountains in the northeast, by the Maramureș Hills in the west and southwest, and by Rodna Mountains in the south. The lowest area of this basin is at 303 m above the sea level at the point of confluence of the Tisa and Vișeu rivers, while the highest area reaches 2,303 m altitude in the Pietrosul Rodnei Peak in the Rodna Mountains. Due to the tectonic, geological, and geographical diversity within this basin (karst, exokarst, glacial relief forms, etc.), the studied area is diversified in landscapes, inevitably having a high diversity of biotopes, biocoenosis, and among other elements ichtyocoenosis. (Curtean-Bănăduc et al., 2008; Bănăduc et al., 2011)

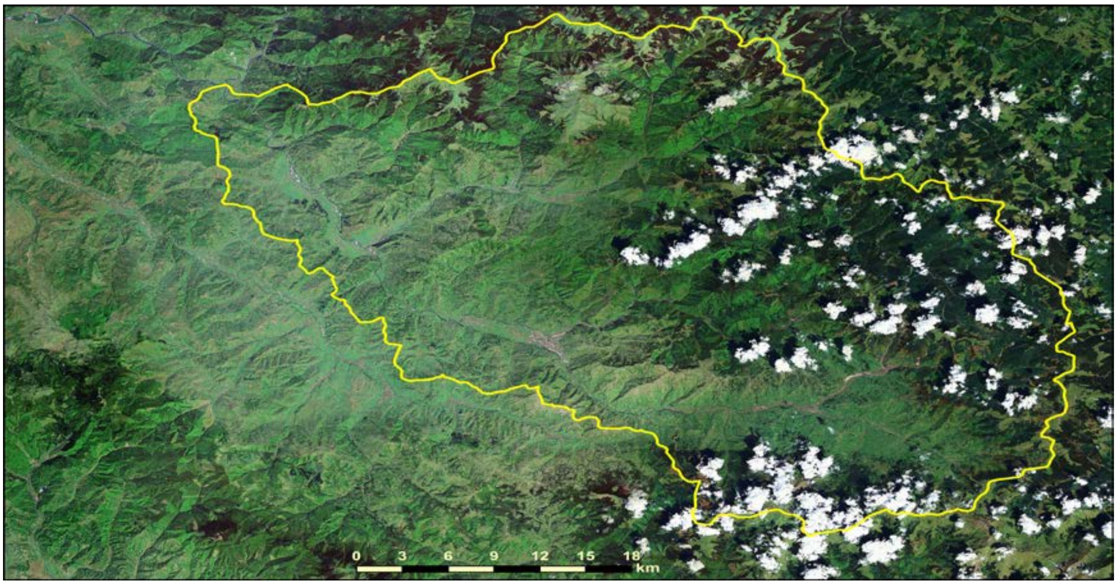

Figure 1: Vișeu River basin.

The Vişeu River is a second degree tributary of the Danube River, streaming into the much larger Tisa River. It has a length of 80 km and a multiannual average flow of 30.7 m^3/s at its end near the confluence with Tisa. The spring area is placed in Prislop Pass (1,416 m) and it stream into Tisa River in the proximity of Valea Vișeului Village, the basin covering 1,606 km^2. In its upper sector, from springs area to the Moisei locality, the Vișeu River has a relatively considerable abruptness (20-50 m/km) and is named Borșa or Vișeuț. The Vișeu enters at Moisei in the Maramureș Depression where the river valley is broader, even if some narrow gorge-like passages exist: Rădeasa Gorges between Moisei and Vișeu de Sus, Oblaz Gorges between Vișeu de Jos and Leordina, and Vișeu Gorges between Bistra and Vișeului Valley. The Vișeu River hydrography is of Eastern-Carpathian-Moldavian kind in its upper sector and of Eastern-Carpathian-Transylvanian type in its lower part. Its discharge is significant in the springtime (39.4% of the annual discharge) afterwards declining in the summer (27% of the annual discharge) likewise during the autumn (18.6% of the annual discharge) and with its lowest period during wintertime (15% of the annual discharge). (Curtean-Bănăduc et al., 2008; Bănăduc et al., 2011)

Due to fact that the Vişeu Basin is located for the most part in mountainous areas (67%) induce a significant density of the hydrographic network (0.7-1 km/km^2) and to one of the biggest specific discharges in Romanian territory, as a result of rain and snowfall of over 1,100 mm/year. In the upper sector, the tributaries formed in the glacial-type Rodna Mountains, have a discharge of approximative five m^3/s. The most substantial Rodna-originating tributaries of Vişeu are: Fântânilor Valley (seven km length), Negoiasa Valley (six km), Repedea Valley (10 km), Pietroasa Valley (seven km), Vremeșu Valley, Hotarului Stream, Dragoș's Valley (11 km) and Izvorul Negru (seven km). From the Maramureș Mountains, the right side tributaries are: Hășmașul Mic, Cercănel (11 km), Țâșla (20 km), Vaser (52 km in length and catchment area of 422 km^2, with an average flow of nine m^3/s contributing by 27% to the total flow of Vişeu), Novăț (16 km, 88 km^2 tributary of the Vaser), Ruscova (39 km in length and 435 km^2 catchment area, average discharge of 11.3 m^3/s), Socolău (13 km in length and 72 km^2 catchment area, tributary of the Ruscova), Repedea (19 km in length and 87 km^2 catchment area, tributary of the Ruscova), Bardi (11 km in length and 32 km^2 catchment area, tributary of the Ruscova), Covașnița (11 km in length and 34 km^2 catchment area, tributary of the Ruscova), Frumușeaua (14 km in length) and Bistra (nine km in length). From the Maramureș Hills originate the left-side tributaries, small and with insignificant water input: Drăguiasa, Cocicoi, Spinului, Plăiuț, Neagră and Luhei. (Curtean-Bănăduc et al., 2008; Bănăduc et al., 2011)

In the Vişeu River watershed, the water feature is influenced in some areas in a natural way by the existence of mineral springs (around five in Maramureş Hills; about six in Rodnei Mountains, and 150 in Maramureş Mountains) with a rather diverse content (ferrous, bicarbonate, sulphurous and saline) (Curtean-Bănăduc et al., 2008).

In the Maramureş and Rodnei mountains the streams and rivers are occasionally obstructed by considerable waterfalls and successions of rapids, we remark the most important of these waterfalls from the Maramureş Mountains: Criva, Tomnatic and Bardău, and from the Rodnei Mountains: Cailor, Cimpoioasa Valley, Repedea Valley and Izvorul Verde. Lentic water ecosystems also occur. Glacial lakes from Rodnei Mountains are placed at an altitude over 1,900-1,950 m and were formed at the heels of some deposits: Iezer Lake, Gropi Lake, Buhăiescu Lake, Rebra Lake, Negoiescu Lake and Cimpoieş Lake. The local wetlands are eutrophic and oligotrophic marshes: Strungi, Tăul Obcioarei, Tăul Ihoasa, Jneapănul Hânchii, Pietrosul Barcăului, Tăul Băiţii, Preluca Meşghii, Vârtopul Mare, Tăul cu Muşchi and Bedreasca. The lakes from Maramureş Mountains are Lutoasa, Bârsănescu, Budescul Mare, Măgurii and Vinderel. On the Vişeu River valley near Petrova locality, there are some small bodies of water/ponds. (Curtean-Bănăduc et al., 2008; Bănăduc et al., 2011)

The variety of aquatic and semi-aquatic habitats and their linked endangered, rare and endemic species from Vişeu Watershed are heterogeneous and very valuable under preservation context. The fish species of this area are no exception, as recorded by various ichthyologists rather constant over more than a century of specific research (Bănărescu, 1964; Staicu et al., 1998; Curtean-Bănăduc et al., 2008). Around 50% of the local fish species are of significant protective importance.

Thymallus thymallus (Linnaeus, 1758) is one such fish species of preservation importance, where populations within the Vişeu Watershed have declined. The distribution and abundances of this endangered taxa are relatively unknown and specific information for the a proper management is highly needed.

MATERIAL AND METHODS

Research on populations of *Thymallus thymallus* within the Maramureşului Mountains Nature Park was accomplished in January – July 2015, consisting of 370 sampling sectors (Fig. 2; Tab. 1). This research included population mapping, evaluation of the current preservation status, and identification of factors responsible for the current declines in populations.

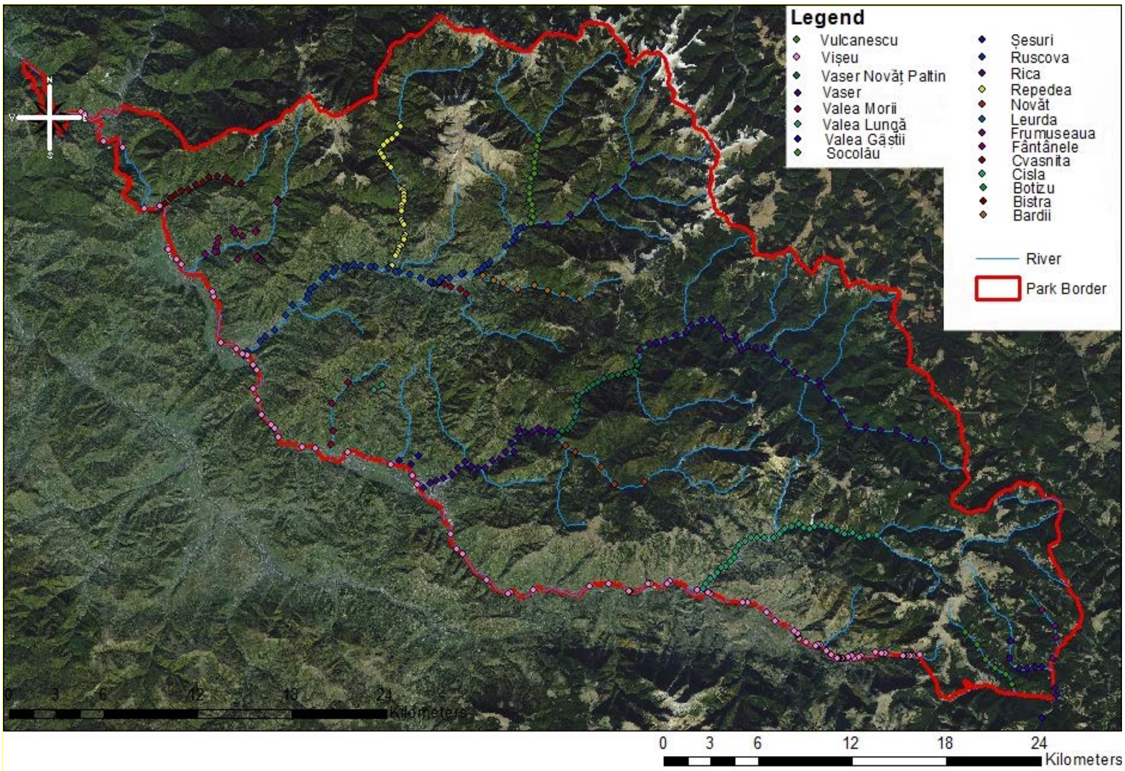

Figure 2: Locations of the 370 sampling stations (GIS support – Danci O.).

To evaluate the conservation status and population decline of *Thymallus thymallus* within the Maramureşului Mountains Nature Park, quantitative samples were collected from sampling stations of approximately three kilometres between two consecutive sectors on all watercourses with appropriate habitats for this species. The locations of the sampling stations allows the evaluation of the effects of human influence on the target populations, including the biotope situation transformation, presence of substratum exploitation, pollution sources, hydrotechnical works, and last but not least chaotic recreational fishing and poaching.

Quantitative sampling of fish fauna was accomplished by the electronarcosis, per unit of time and effort per section (two hours on Vişeu River, one hour on Ruscova River, 30 minutes on the other rivers of the references zone), on five longitudinal sections of 100 m length. After species identification and counting individuals, fish were immediately released in their natural habitat.

The number of individual fish caught in the unity of time and effort can be converted through correspondence in the following categories: (C) – common fish species, (R) – rare fish species, or (V) – very rare fish species, according to the guidelines for Natura 2000 standard data form filling, “In mammals, amphibians, reptiles and fishes, no numeric information can be indicative and then the size/density of the population is evaluated as (C) – common species, (R) – rare species, or (V) – very rare species”.

The criteria used to evaluate the population status are the following: balanced distribution of individuals by age classes, population size, distribution areal size and the percent of fish individuals of the species of interest in the structure of fish communities.

According to the Natura 2000 guidelines, standard data form filling the following criteria "The conservation degree of specific habitats" contain subcriteria: i) the degree of conservation of the habitat features which are important for the species; ii) possibilities for recovery.

The criteria i) need a comprehensive assessment of the characteristics of the habitat regarding the needs of the species of interest. **"**The best expertise" is used to rank this criterion in the following way: I. elements in excellent condition, II. well preserved elements, III. elements in average or partially degraded condition.

In the cases in which the subclass I is granted "I: elements in excellent condition" or "II: well preserved elements," the criteria B (b) should be classified entirely as "A: excellent conservation" or "B: good conservation", regardless of the other sub-criterion classification.

In the case of this sub-criterion ii) which is taken into account only if the items are averagely or partially degraded, an evaluation of the viability of the analysed population is necessary. The obtained ranking system is: I. easy recovery; II. restoration possible with average effort; III. restoration difficult or impossible.

The combination practiced for classification is based on two sub-criteria: A – excellent conservation = elements in excellent condition, regardless of classification of recovery possibility; B – very good conservation = well preserved elements, regardless of classification of recovery possibility; B – good conservation = elements in average or partially degraded condition and easy to restore; C – average or reduced conservation = all other combinations.

In every sampled area, were evaluated: condition, pressures/threats of habitats and populations of interested fish species.

The sampling sections to assess fish population and the conservation status of *Thymallus thymallus* in the monitoring area occurred in sectors where these populations are constant, with a favourable conservation status and well preserved characteristic habitats, as well as river sectors located at the limit of the dispersion area for this species, which include sectors under human impact that can endanger the studied populations condition – the so called Representativity Criteria.

The economical criterion was treated for constituting the monitoring sectors, in this manner an average number was set to supply the needed data for the management decision mechanism in order to be able to conserve a favourable status for the interest fish species in the research area.

Thymallus thymallus (Linnaeus, 1758), Actinopterygii – Salmoniformes – Salmonidae – Thymallinae (Fig. 4), was found in the researched area in the last century (Bănărescu, 1964; Staicu et al., 1998; Telcean and Bănărescu, 2002; Bănăduc et al., 2012; Homei, 1963).

The two sections of the body are evenly convex or, the dorsal one, slightly more convex; the maximum height is located in the insertion of the dorsal or in front of the first quarter of the dorsal. The eyes, placed in the anterior part of the head, look sideways. Interorbital space, convex. Small nostrils, at an equal distance from the tip of the snout and from the anterior margin of the eye. Small mouth, subterminal, transversal, protractile; its opening located anteriorly from the nostrils. The rear edge of the upper jaw reaches far below the anterior margin of the eye, insertion of the mandible behind the middle of the eye. Margins of jaws, straight and sharp. Teeth small, pointed, arranged in a single row on the intermaxillary, maxillary and dental; a group of tiny teeth on prevomer and palate. Wide gill openings, brachial membrane fixed by isthmus only with the anterior end. Dorsal margin rounded anteriorly, straight posteriorly. Posterior radials of the dorsal fin slightly longer than the anterior ones but strongly inclined backwards so that the fin height decreases slightly toward the rear. Pectorals sharp, ventral fins rather rounded. Anal margin anteriorly convex, slightly concave posteriorly. Ventral fins are inserted under the posterior side of the dorsal; anal fin behind the posterior tip of the dorsal and ventral fins, and the adipose fin above the posterior half of the anal. Caudal deeply cupped. Lateral line complete, almost straight. The scales well fixed, absent on the head, on the pectoral base, isthmus and irregular areas of the anterior part of the body ventral side. Ventral scales small. Head and backside are brown, body sides yellowish with a metallic luster or silvery with purple reflections, ventral side silvery white, sometimes reddish. On the sides of the anterior body half a series of dots or thin strips dark brown, very obvious and persistent. Dorsal dark, with a series of brick-red spots and black, less displayed like longitudinal stripes. Caudal gray, with reddish or blueish tint, and brown spots. Pectorals yellowish, purplish-pink ventrals and anal pink-violaceous. It can reach 30-35 cm, up to 50 cm in length. (Bănărescu, 1964)

RESULTS

The lotic sectors where the species *Thymallus thymallus* (Fig. 3) was found during the research are shown in table 1 (Fig. 4), for each such sectors the catch index values were specified (individuals number per time and effort unit).

Figure 3: *Thymallus thymallus* (Linnaeus, 1758).

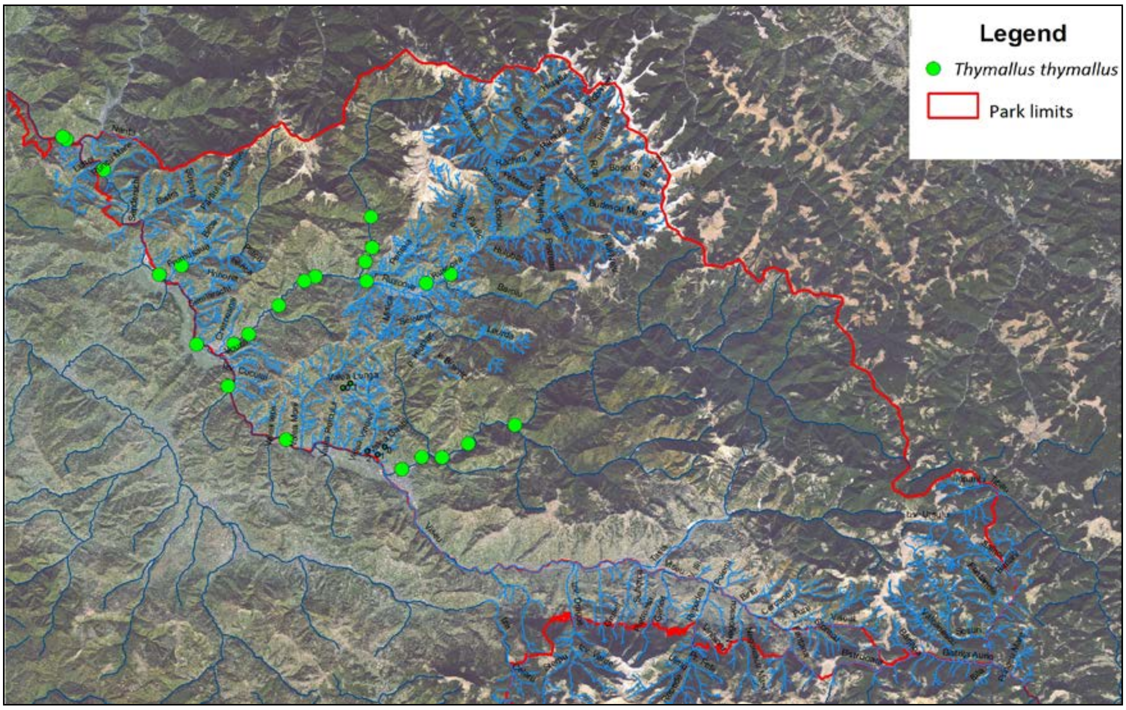

Figure 4: *Thymallus thymallus* sampling stations location (GIS support – Danci-Kast O.).

Table 1: *Thymallus thymallus* sampling points in Maramureș Mountains Nature Park.

No. crt.	River	Station code	Lat. (N')	Long. (E')	Catch index no. ind./100 m × 30 min	Characteristic habitat state
1.	Vișeu	56	47 43 55,5	24 20 02,7	1	reduced
2.	Vișeu	62	47 45 53,3	24 16 53,9	1	reduced
3.	Vișeu	68	47 47 24,0	24 15 12,6	1	reduced
4.	Vișeu	71	47 49 57,1	24 13 09,3	2	reduced
5.	Vișeu	76	47 53 47,4	24 10 06,8	7	good
6.	Vișeu	78	47 54 53,0	24 08 06,6	5	good
7.	Vișeu	79	47 54 58,8	24 07 54,2	1	reduced
8.	Ruscova	7	47 50 03,2	24 28 46,7	1	average
9.	Ruscova	12	47 49 43,0	24 27 28,3	5	average
10.	Ruscova	21	47 49 46,7	24 24 16,7	7	good
11.	Ruscova	29	47 49 55,6	24 21 31,7	6	good
12.	Ruscova	32	47 49 45,4	24 20 56,8	3	average
13.	Ruscova	38	47 48 51,2	24 19 35,0	1	average

Table 1 (continued): *Thymallus thymallus* sampling points.

No. crt.	River	Station code	Lat. (N’)	Long. (E’)	Catch index no. ind./100 m × 30 min	Characteristic habitat state
14.	Ruscova	41	47 47 47,4	24 17 58,4	1	reduced
15.	Ruscova	43	47 47 25,6	24 17 12,2	1	reduced
16.	Repedea	18	47 52 08,7	24 24 28,7	3	good
17.	Repedea	25	47 51 00,3	24 24 33,4	4	good
18.	Repedea	30	47 50 29,6	24 24 11,5	2	good
19.	Vaser	42	47 43 49,4	24 29 46,9	1	good
20.	Vaser	46	47 43 18,4	24 28 23,2	6	average
21.	Vaser	49	47 43 18,9	24 27 17,6	5	average
22.	Vaser	52	47 42 52,6	24 26 15,2	2	average
23.	Frumușeaua	18	47 50 17,0	24 14 21,6	1	reduced
24.	Novăț	32	47 44 31,2	24 32 16,2	1	average

DISCUSSION

Based on the present research results, correlated with *Thymallus thymallus* species biological and ecological necessities, some risk elements were diagnosed (pressures and threats): poaching, minor riverbed morphodynamic changes, liquid and solid natural flow disruption, destruction of riparian vegetation, and habitats/fish populations fragmentation.

Poaching. During the field research chaotic recreational fishing and poaching was observed with electricity from vehicles accumulators and from other categories of rechargeable devices. Poachers were seen during their activities also using diverse substances (natural and/or synthetic) to slaughter fish and gather them downstream. By questioning many people who live in Maramureș Mountains Nature Park, at least from these sources poaching is a somewhat constant activity around the year in the Vișeu River basin. The inability to stop this anomaly may cause a decline in the number of *Thymallus thymallus* individuals.

Minor riverbed morphodynamic changings. Characteristic diverse microhabitat and habitat necessities for this species, in accordance with its life cycle stages, include a natural variability of riverbed morphodynamics. Dams, dikes, sills, changed riverbeds, roads in riverbeds, riverbed mineral exploitation (Fig. 5), changed dynamics of liquids and solids flow, etc., all contributed towards modifying the natural morphodynamics of major and minor riverbeds. These changes also affected the habitats and/or microhabitats needed for the life cycle stages of the *Thymallus thymallus*, which could contribute towards the decline in abundances of this species.

Watercourse obstructions, and water resource development activities on the studied watercourses (for example: dams, dikes, high sills, microhydropowerplants, modifications in the riverbeds, water extractions, riverbed mineral overexploitations, etc.) should not be admitted by the Park Administration without an appropriate/explicit ichthyologic study for this valuable fish species.

Figure 5: Overexploitation of minerals in the banks and terraces of the lower Vișeu River.

Solid and liquid and natural flow changings. The modification of natural flow and riverine morphology prevent the generation of specific microhabitats, habitats, and environmental conditions necessary for the continued presence of *Thymallus thymallus*. These changes to the riverbed natural morphodynamics may be contributing towards the decreasing population size of this fish species. Atypical events, where water turbidity is artificially increased due to careless forestry activities within the vicinity of riverbeds, are examples of activities causing disturbance to the natural balance of the solid and liquid flow.

The solid and liquid natural flow can be preserved as similar as desirable to the natural condition if the riverbed gravel exploitations and/or forestry practices do not significantly disturb the watershed capacity to have a proper self-sustainable function. This can be accomplished by coordinating the human activities with the periods of the year when the natural circumstances are very much alike to those to be generated (e.g. high water turbidity). Proposed in-channel structures and modifications, such as dams, thresholds (Fig. 6) embankments, crossings, water extractions, bank modifications and roads in the waterbed (Fig. 7), thalweg changes by exploitations of construction materials from the riverbed, etc., should be not allowed by the administrator of the protected site without the consent of experts studying the species, based on the integration of the identified local stress factor and the biological and ecological needs of the fish species of interest. In this peculiar case, no crossing should be higher than 15 cm in the shallow water sectors and dry season. We also propose the monitoring of the forestry regulation surveillance containing the boycott of dragging and storage lumber through/in the lotic systems. We also suggest the inspection of the development works for lumber storage and exploitation terraces, (Fig. 8) and the compulsory demand to fast reforestation. In this situation, the rotation of forest exploitations in the sub-basins of the Vișeu Basin is desirable.

Figure 6: Concrete threshold of three m height, with no fish ladder on the upper Vişeu River.

Figure 7: Frumuşeaua River concrete riverbank/completely modified and road in the riverbed.

Figure 8: Logs transported and deposited on the Vaser River banks and in the riverbed.

Figure 9: Destroyed riparian vegetation on the Ruscova River islet.

Destruction of riparian trees and bush vegetation. The clearing and destruction of riparian and islet vegetation (Fig. 9) can cause a reduction in the abundances of fish species, including the *Thymallus thymallus*, due to alterations in the in-stream microclimate and changes to trophic resources (Curtean-Bănăduc et al., 2014). Where feasible, arboreal and shrubs riparian vegetation must remain undamaged.

Habitat fragmentation/isolation of populations always leads to genetic isolation, reduced gene diversity, inbreeding, and local or regional extinction. Unobstructed upstream and downstream movement, including connectivity of the diverse sub-drainage basins of the Vișeu Basin, is a very critical element for the management of this species.

We propose considering the potential investments situated on the streams and rivers courses carefully, as some developments may weaken or interrupt water course connectivity, not only by making diversified crosswise barriers in the riverbed, but also by diminishing the water flow or draining of some river sectors.

Pollution caused by mining activities. The historical pollution generated by heavy metal mining activities in the Țâșla River basin are not influencing only the Țâșla River habitats and species but also the downstream habitats and species of the upstream Vișeu River. The effect of the meteoric washing waters of the abandoned mine galleries and greened refuse heaps is rated as considerable on the Țâșla River and important on the upstream Vișeu River.

The effects of meteoric waters washing the abandoned mine galleries and rehabilitated refuse heaps can be considerably diminished by isolating/filling the old mine galleries and by insulating (not greening) the refuse heaps from the water courses in the Țâșla River basin.

Combined human impacts affect numerous lotic sectors in the area of interest (Figs. 10 and 11), and consequently the studied fish species in the area compared to its natural potential.

Minimal management elements that should be enforced in the researched area include: development of lotic systems buffer zones; wise management of water use; wise management of sewage and waste water and surface water pollution; adapt to the local conditions of the hydroenergetic use of streams and rivers; enforcement of integrated water management at the Vișeu Watershed level; create ecological networks; streams and rivers connectivity restoration; back specific high scientific quality assessments and monitoring and basin integrated management oriented research.

Organic pollution is an old persistent problem, where sewage and wastewater treatment is simultaneously connected with the farm activities, in the majority of the Vișeu Basin, principally on the Vișeu River (Curtean-Bănduc, 2008), which is a lasting stress source for fish populations. Complete sewage systems must be created in the Vișeu Watershed and the wastewaters of all the localities alongside the main water courses must be properly treated.

Displacement of fish washed away during flood periods in the human uniformized lotic sectors. In the lotic sectors uniformized by humans, fish are washed away in flood periods. In these sectors shelters should be created with a maximum high of 15 cm.

A **diet overlapping** among *Thymallus thymallus* and *Salmo trutta* (Kruzhylina and Didenko, 2011) can be another local pressure on the European Grayling populations, but such studies in the research area are still needed.

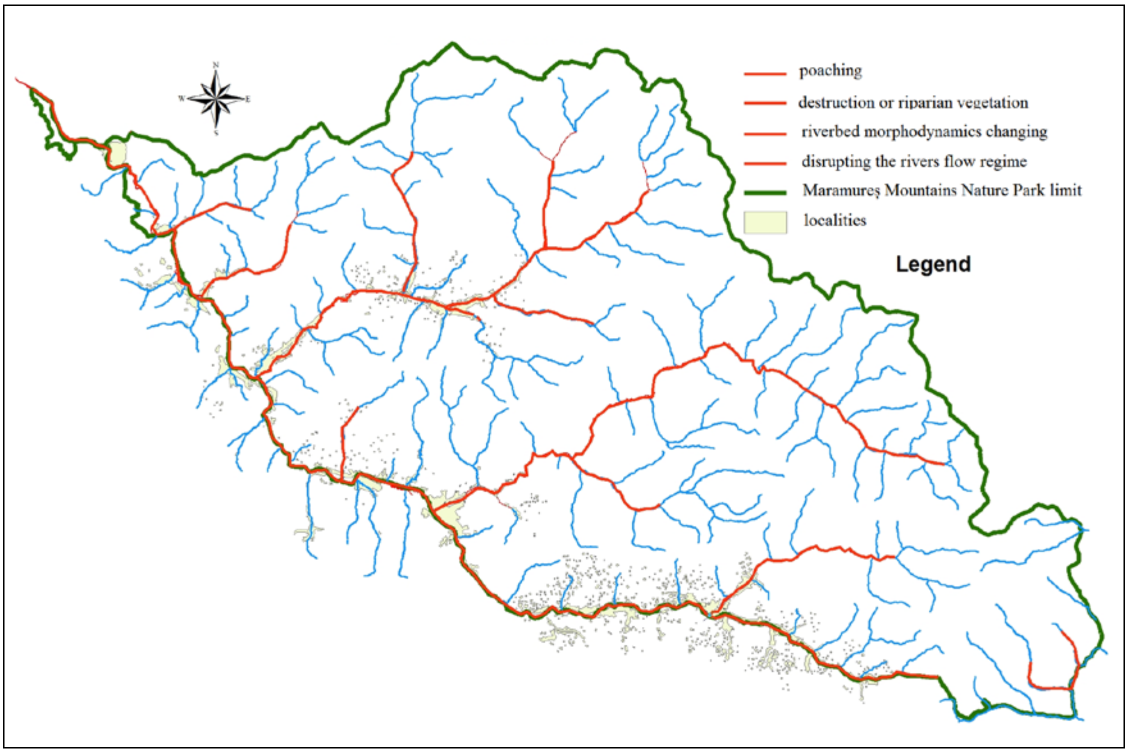

Figure 10: Diagnosed mixed pressures and threats for *Thymallus thymallus* in the studied area.

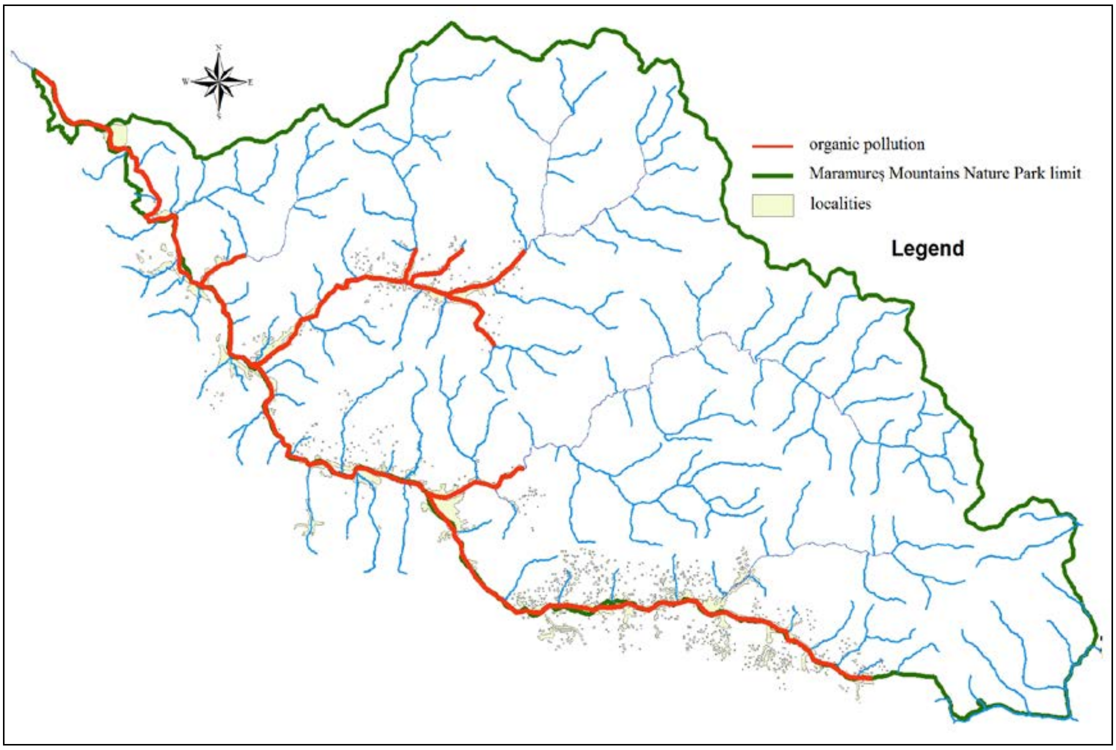

Figure 11: Lotic sectors influenced by organic pollution (GIS support – Danci-Kast O.).

CONCLUSIONS

Thymallus thymallus from the researched area has stable populations but not achieve their natural potential due to anthropic activities, in the following basins: Vișeu – middle and lower sectors, Ruscova – middle and lower, Vaser – middle and lower, Frumușeaua – lower and Bistra – lower. The preferred habitat for this species is large enough within this area to preserve the actual average state of the Grayling studied populations.

Thymallus thymallus can be considered in the present as a relatively rare species in the studied area and there where a specificaly low abundance was registered exist a suplementary limitation on their use for restorative goals.

The conditions of the Vișeu and Frumușeaua lotic systems are the most degraded, and do not meet the habitat quality requirements of *Thymallus thymallus* species.

ACKNOWLEDGEMENTS

These data were obtained in the project “Inventarierea, cartarea și evaluarea stării de conservare a speciilor de pești din Parcul Natural Munții Maramureșului (ROSCI 0124 Munții Maramureșului)/ Inventory, mapping and assessment of the conservation status of fish species of Munții Maramureșului Nature Park (ROSCI 0124 Maramureșului Mountains)”. Special thanks for the continuous support of the Munții Maramureșului Natural Park Administration and Scientific Council members especially to: Bogdan C., Bucur C., Szabo B., Brener A. and Mărginean M. Also thanks for Mrs. Danci O. for the GIS support.

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PROPOSAL TO ACHIEVE FLOODPLAIN CONNECTIVITY IN ALȚÂNA SECTOR ON HÂRTIBACIU RIVER (TRANSYLVANIA, ROMANIA)

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DOI: 10.1515/trser-2015-0088

KEYWORDS: lotic system, floodplain connectivity, fish conservation.

ABSTRACT

The process of supplying water to the new anthropogenic wetland is achieved gravitationally, and the excess water in the wetland will be directed towards the Hârtibaciu River in a similar natural way. The fish and fauna of the Hârtibaciu River have a disrupted lateral connectivity due to its banks embanking including in the proximity of the Alţâna locality. The newly proposed anthropogenic wetland would improve habitat quality for the fish species of conservation interest, *Rhodeus amarus* (Bloch, 1782), and increase its population numbers. A new fish species, *Chondrostoma nasus*, was identified for the first time in the Hârtibaciu River.

RÉSUMÉN: Propuesta para lograr la conectividad de llanuras inundables para el sector Altana del Río Hartibaciu (Transilvania, Rumania).

El proceso para suministrar agua al nuevo humedal antropogénico se consigue gravitacionalmente, y el exceso de agua en el humedal se dirigirá hacia el río Hârtibaciu de una manera natural similar. La fauna de peces del río Hârtibaciu tiene una conectividad lateral interrumpida debido a sus bancos de embarque incluyendo en la proximidad de la localidad de Alţâna. El humedal antropogénico recientemente propuesto puede mejorar la calidad del hábitat de las especies de peces de interés conservador *Rhodeus amarus* (Bloch, 1782) y puede aumentar el número de individuos de esta población. Una nueva especie de pez, *Chondrostoma nasus*, fue identificada por primera vez en el río Hârtibaciu.

REZUMAT: Propunere de realizare a unei conexiuni laterale în sectorul Alţâna pe râul Hârtibaciu (Transilvania, România).

Procedeul pentru alimentarea cu apă a noii zone umede antropice se realizează gravitaţional, iar excesul de apă din zona umedă va fi direcționată spre râul Hârtibaciu într-un mod natural similar. Peștii din râul Hârtibaciu au o conectivitate laterală perturbată datorită malurilor sale îndiguite, inclusiv în proximitatea localității Alțâna. Zona umedă propusă în articol poate îmbunătăți calitatea habitatului pentru speciile de pești de interes conservativ cum ar fi *Rhodeus amarus* (Bloch, 1782), și poate crește numărul de indivizi ai acestei populații. O nouă specie de peşte *Chondrostoma nasus* a fost identificată pentru prima dată în râul Hârtibaciu.

INTRODUCTION

Floodplain connectivity adds the necessary habitat for conservation target areas, moderates variable regular and irregular categories of disturbance elements, ameliorates groundwater access, ameliorates water and soils, increases sediment depository potential, supplements nutrient flux, disipates flood energy distribution, invigorates riparian and floodplain biocoenoses, and ensures ecosystem functionality, etc. (Schneider-Binder, 2008, 2009; Orlov and Vovk, 2011; Bănăduc et al., 2016)

The Water Framework Directive sets new standards for waters in the European Union, standards that can be achieved, and the deterioration of aquatic and semi-aquatic ecosystems avoided. Only if rivers, floodplains and wetlands are managed with an integrated permanent approach, can a good ecological and hydro-morphological status be achieved (Peacock, 2003).

Figure 1a: Human impact on the Hârtibaciu River studied sector on banks and riverbed.

In the study area, the floodplain connectivity was identified as a necessary obtainable outcome, to provide favorable conditions that would meet the goals of restoring connectivity and avoiding/diminishing the present associated challenges/risks, inclusive of the local habitats and biocoenoses, including the fish community.

The proposed wetland creation is more than needed for the local fish and fauna in a sector where the river was chanelised many years ago, and inapropriate work (cleaning the river bed and the river banks with heavy machines, cutting the riverine vegetation, etc.) was carried out (Fig. 1a, b). This sector is very different from the upstream semi-natural sector where fishing takes place (Fig. 2).

Figure 1b: Human impact on the Hârtibaciu River studied sector on banks and riverbed.

Figure 2: Semi-natural sector of Hârtibaciu River in the studied sector.

MATERIAL AND METHODS

To assess the need for the proposed investment the fish communities' structure was studied in the local Hârtibaciu River sector, in the Alţâna locality proximity.

The fish individuals were sampled with a mountain fishing net, in time and effort unit, identified and immediately released back into their habitat.

RESULTS AND DISCUSSION

Local fish community challenges and need for habitat restoration

The Hârtibaciu River fish fauna includes a relatively high number of fish species, such as the following: *Squalius cephalus* (Linnaeus, 1758), *Alburnus alburnus* (Linnaeus, 1758), *Alburnoides bipunctatus* (Bloch, 1782), *Rhodeus amarus* (Bloch, 1782), *Gobio obtusirostris* Valencienes, 1842, *Romanogobio kessleri* (Dybowski, 1862), *Barbus meridionalis* Risso, 1827, *Barbatula barbatula* (Linnaeus, 1758), *Misgurnus fossilis* (Linnaeus, 1758), *Cobitis taenia* Linnaeus, 1758, *Sabanejewia romanica* (Băcescu, 1943), and *Sabanejewia aurata* (De Filippi, 1863) (Bănărescu, 1964).

In the Alţâna locality sector of Hârtibaciu River five fish species were sampled: *Squalius cephalus*, *Alburnoides bipunctatus*, *Chondrostoma nasus* (Linnaeus, 1758), *Rhodeus sericeus amarus*, and *Gobio gobio*.

In the period of this study *Chondrostoma nasus* was recorded for the first time in the Hârtibaciu River (Fig. 3).

The creation of the proposed wetland would provide an important buffer zone for fish, especially in the dry and cold (with frozen periods) seasons.

Figure 3: sampled individual of *Chondrostoma nasus* in the study sector.

In the studied river sector on the Hârtibaciu River only five fish species were recorded: *Alburnus bipunctatus* with a relative abundance of 40%, *Rhodeus amarus* 28%, *Gobio gobio* 20%, *Squalius cephalus* 8%, and *Chondrostoma nasus* 4%.

The newly proposed wetland area should also be favourable for the colonisation of the bilalve species *Anodonta cygnea* (Linnaeus, 1758), which is present in the studied water course (Curtean et al., 1999; Sîrbu et al., 1999). This is a species which can provide a symbiotic role for species reproduction of the *Rhodeus amarus* (Bănărescu and Bănăduc, 2007) and also for the water self-cleaning processes.

It is also important to note that *Rhodeus amarus* (Fig. 4) is a protected species under the Habitats Directive (92/43/EEC). The creation of the new proposed wetland can increase its abundance in the area as it provides this species with a habitat characterised by stagnant or semi-stagnant water with soft sediment bottom (Bănărescu and Bănăduc, 2007).

Figure 4: sampled individual of *Rhodeus sericeus amarus* in the study sector.

The creation of the new proposed wetland can provide a buffer zone for fish, especially in the dry and cold (with long frozen periods) seasons, and also an easy route for passage. Aquatic and semi-aquatic birds, amphibians, mollusks, and aquatic and semi-aquatic vegetation can also benefit by this wetland in the future. It will continue to evolve, becoming stronger and naturally more diverse, healthy and self-supporting over time.

Ecologic restoration and new wetland area creation

To achieve floodplain connectivity in the study area, we propose the development of a wetland (Fig. 5) downstream of a bridge (Figs. 6a, b) which will be supplied by capturing water from the Hârtibaciu River through a water intake situated upstream of the bridge or upstream from the village of Alțâna.

The wetland to be created is segmented as follows: i) village road capping (upstream), ii) 200 m dyke on the left bank of the Hârtibaciu River, iii) current channel for water discharge protected by the earth dike and, iv) an earth dam to be built on a pasture around 200 m from the bridge. The earth dikes proposed to be built are trapezoidal with 2.5 m height, small base of 2.5 m and large base of 4.0 m (Figs. 7, 8).

In order to ensure functionality of the newly created ecosystem, it is recommended that the water level in the wetland will be approximately 1.0 m. Populating the wetland with various native species is recommended, taking into account the species characteristic to aquatic lentic ecosystems with a depth of between 0.8 m and 1.0 m.

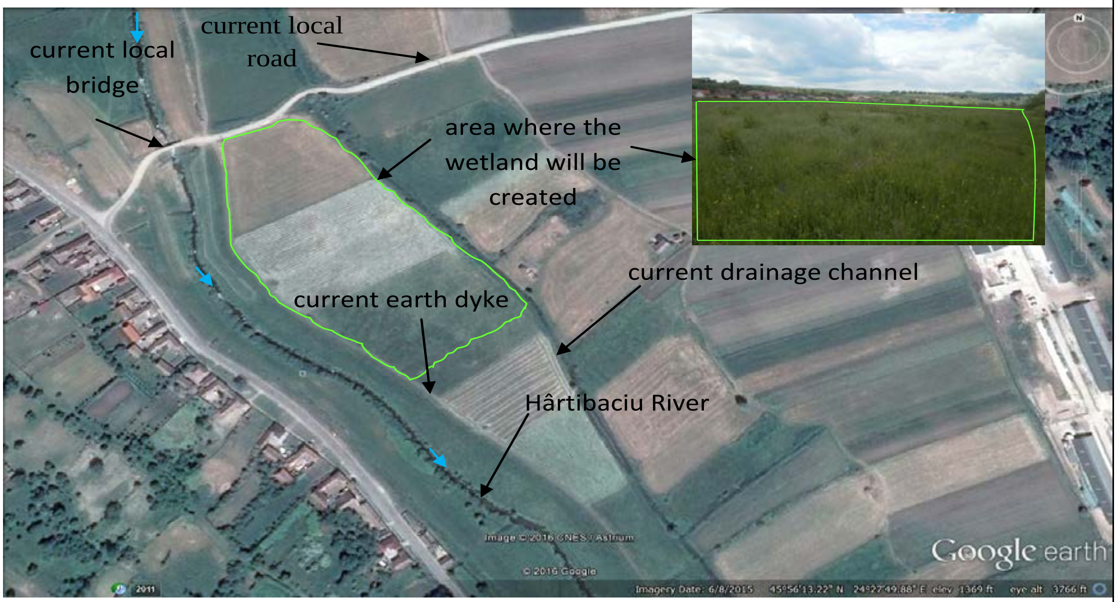

Figure 5: Area where the wetland will be created downstream of bridge (www.google.com).

Figure 6: The Hârtibaciu River downstream of the bridge in the village of Alțâna.

Steps in creating the new anthropogenic wetland.

The first step is aimed at creating habitat conditions that can later support the development of the biotic component (communities of plants and aquatic animals). In this case, in the study area (Figs. 5, 6), it is recommended that the wetland interior is situated near the village of Alțâna and its water supply system.

The second step is the process of populating the wetland with certain species of aquatic organisms (e.g. macrophytes, some species of fish). Thusly, the complexity of interrelations between these two components (biotic and abiotic) will ensure the functionality of the newly created ecological system/wetland.

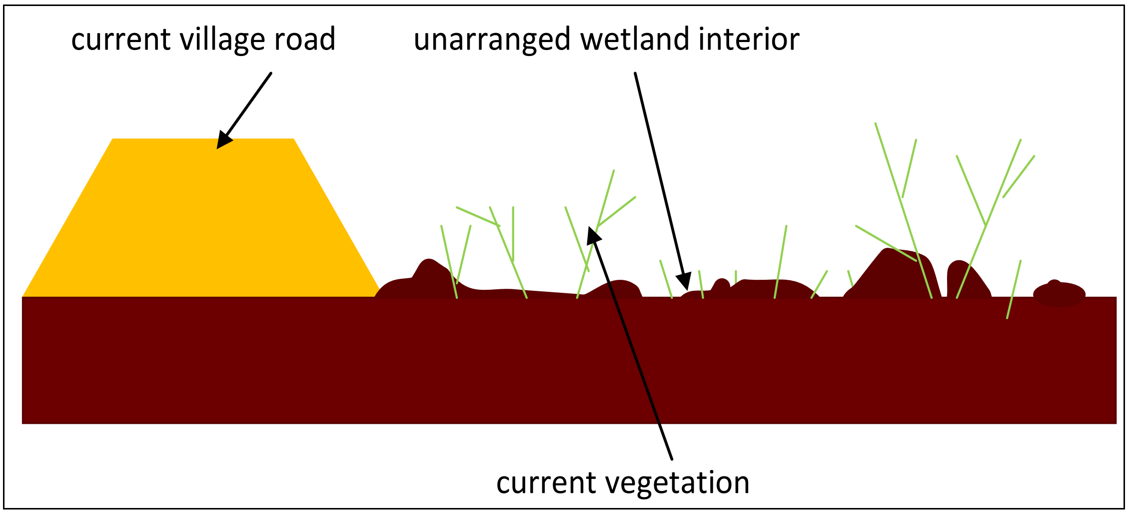

Figure 7: Surface where the wetland is to be built (cross section) – indicative scheme.

The process of habitat wetland arrangements consists of scraping a 30 cm layer of soil (removing all existing vegetation in the area) and levelling the internal surface (Figs. 7, 8 – wetland arranged inside newly built earth dyke).

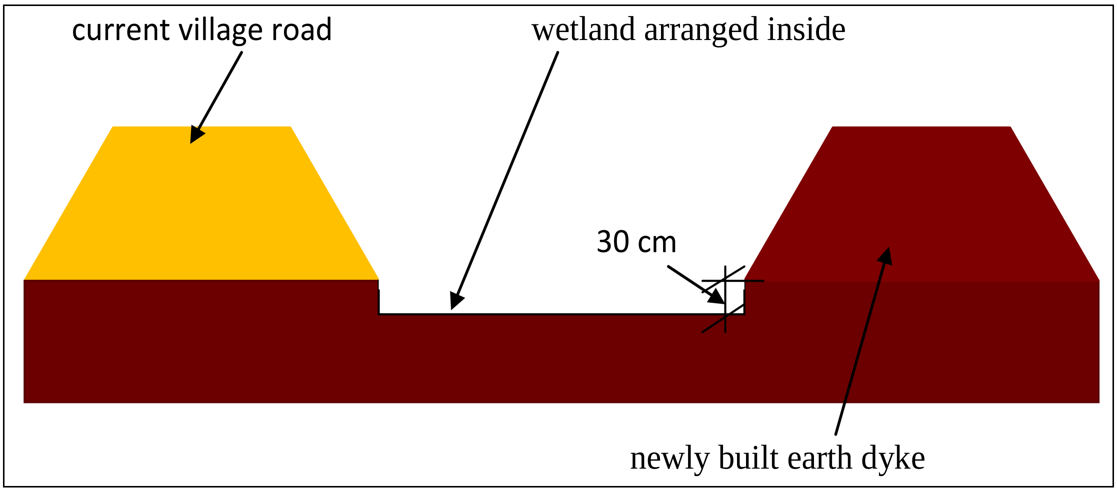

Figure 8: Construction of the wetland (cross section) – indicative scheme.

In order for water to maintain a constant level of 1.0 m, the newly created habitat should be dammed entirely with earth dikes of around 2.5 m height and 4.0 m thickness (Fig. 6) and continuously supplied with water. Creating or arranging a wetland requires ways to supply water and storing it so as to maintain a constant water level. The newly created wetland area will be supplied with water by tankers during its construction phase, and then the supply system shown below will ensure the water supply within the created wetland.

A main aspect is that the wetland water supply is achieved gravitationally and that excess water from the wetland will be directed into the Hârtibaciu River. Water captured from the Hârtibaciu River comes back to the same source. The need for access to water is due to the proposed wetland being 1.0 m deep, and where there will surely be some water loss through a series of processes such as infiltration and biological consumption, the highest quantity will be lost in the evaporation process.

Equipping the system with water level sensors and automatic valves reduces the cost of fuel system verification by specialists. If there are heavy rains, the water level in the wetland will remain constant due to the concrete channels built in order to handle the excess water and discharge it back into the Hârtibaciu River. The automatic valve allows flows of 10 l/s into the stainless steel pipe, regardless of the increase or decrease of the Hârtibaciu River water level.

Description of the wetland water supply system

Water supply to the wetland will be directly provided from the river upstream of Hârtibaciu bridge (approximately one km from the bridge). Captured water will be transported under pressure by a stainless steel pipe with an adjusted flow of 10 l/s. The annual average flow of the Hârtibaciu River in the village of Alțâna is about one m^3/s, or approximately 1,000 l/s. Therefore, the undertaken flow (10 l/s) is not significant and will not affect the aquatic organisms or the abiotic structure of the Hârtibaciu River. This pipeline will be fitted with an automatic valve at its upstream end (Fig. 9) through which a flow of 10 l/s will be maintained in the pipe, regardless of water level fluctuations (water level increase or decrease) in the Hârtibaciu River. It is also proposed that the automatic valve be connected to a level sensor located on a gauge within the wetland. At the upstream end of the pipe, a metal grill will be installed in order to avoid clogging and blockage (Fig. 9). The pipeline is underground and circular-shaped, and will pass under the unpaved road on the bridge.

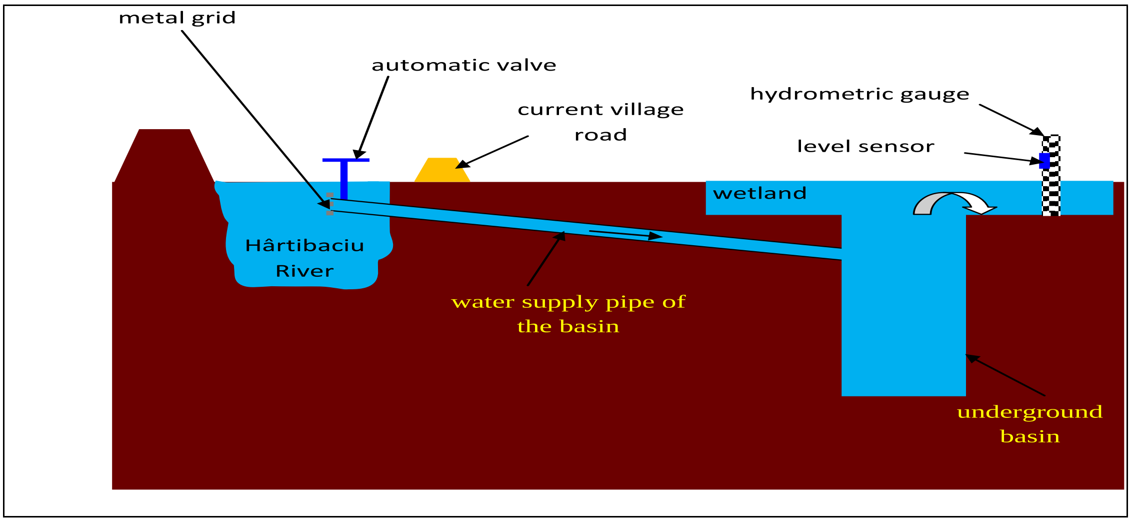

Figure 9: Schematic representation of the wetland location – indicative scheme.

The water is then discharged into a basin built into the wetland with the following dimensions: 5.0 m depth, 5.0 m length and 6.0 m width. The wetland will be directly connected through a channel to another basin (pond) built with the following dimensions: 20.0 m length, 3.0 m height and 10.0 m width (Fig. 10). This basin will be used to propagate fish populating the Hârtibaciu River (chub, barbell, broad snout, loach, carp, perch, etc.). This pond has a direct connection with the Hârtibaciu River through a rectangular channel (Fig. 11). Water quality in the pond will be better than in the Hârtibaciu River because the wetland functions as a filter. In case of river pollution the pond will be suitable habitat for many creatures inhabiting the wetland.

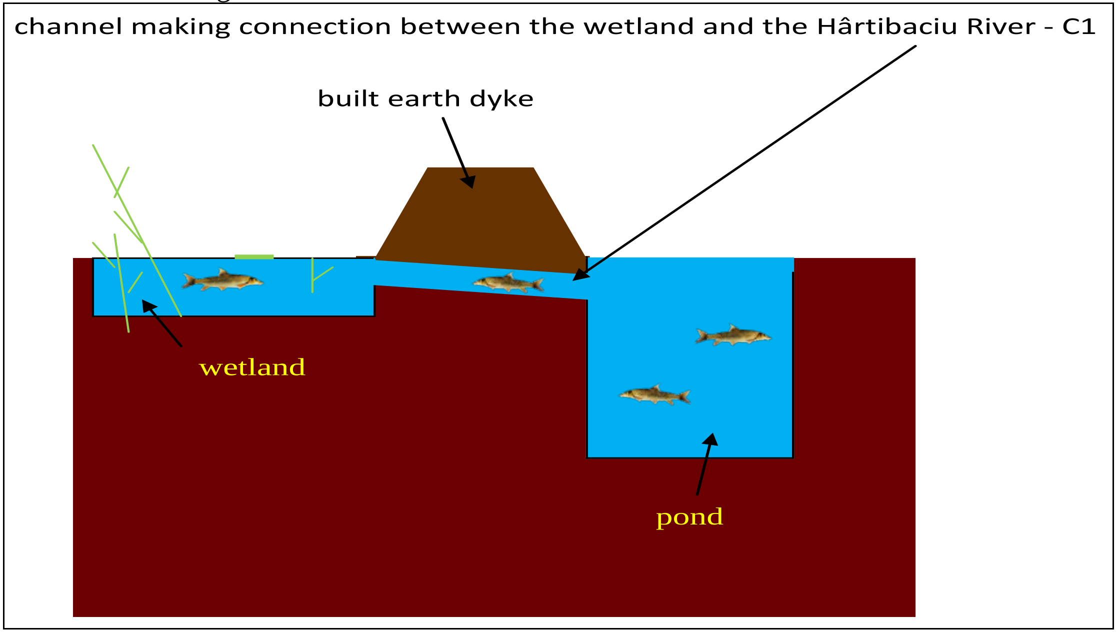

Figure 10: Positioning C1 channel – indicative scheme.

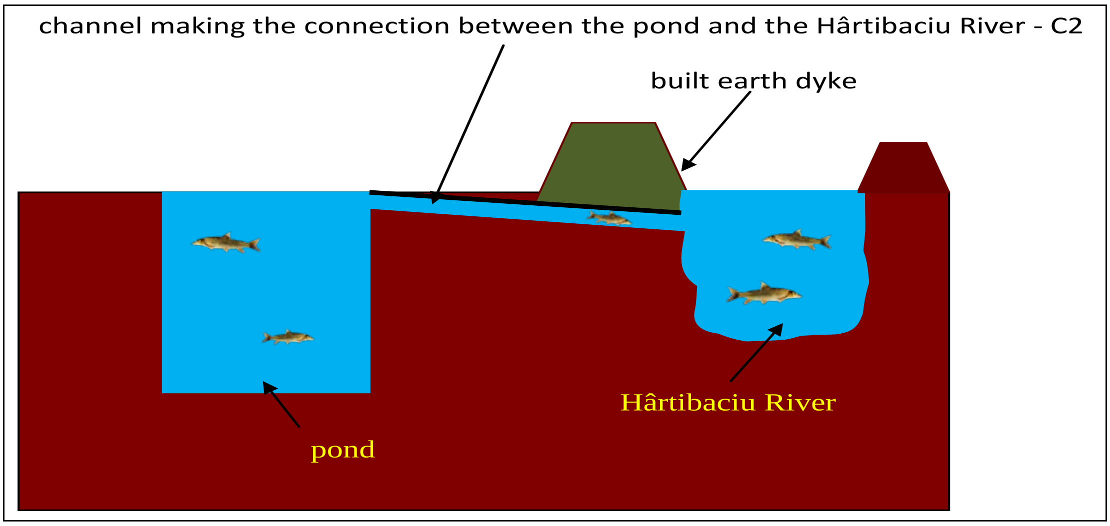

Figure 11: Positioning C2 channel – indicative scheme.

After filtration has reduced pollution, they can return to the wetland. The two channels connecting the wetland and pond (C1) and the pond and the Hârtibaciu River (C2), will be equipped with mobile vertical grids driven by mechanical gears (Fig. 12), and will be made of concrete plated with wood piling and river stones (Fig. 13). Biologists, along with specialists, will monitor the proper functioning of the engineering components in the newly created wetland and will specify when to open and close the metal grids for fish to reach the wetland or the Hârtibaciu River. The two connection channels have different slopes: the channel connecting the wetland and the pond (C1) has a tighter slope favourable for all species to pass into the wetland, whilst the channel connecting the pond and the Hârtibaciu River (C2) has a slope which does not facilitate fish passage from the river up to the pond.

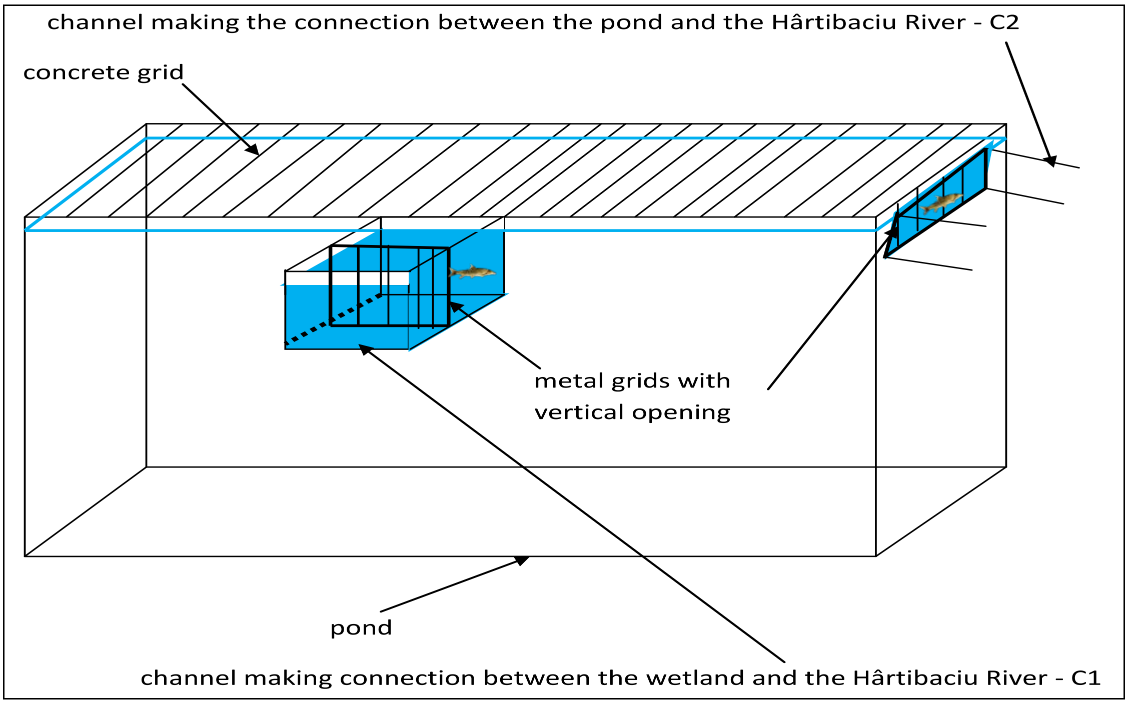

Figure 12: The positioning of the two channels C1 and C2 towards the pond – scheme.

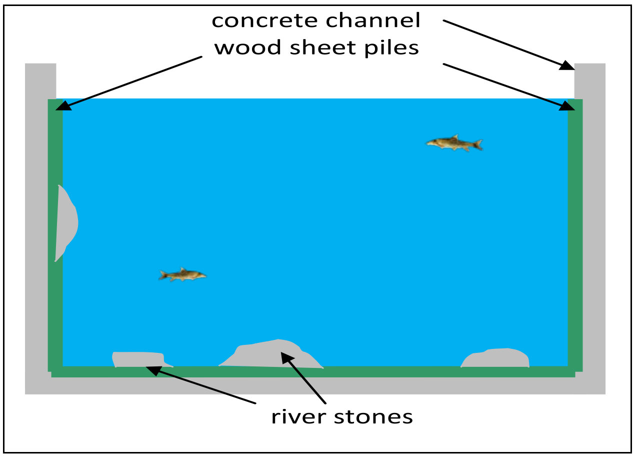

Figure 13: Concrete channel plated with wood piling and river stones – scheme.

The second channel (C2) is strictly built for populating the Hârtibaciu River with different fish species. If the water level exceeds 1.0 m, it would be advisable to build a concrete channel (sewer) in the existing dyke along the Hârtibaciu River, designed to take the excess water over 1.0 m in the wetland, which will then be discharged into the Hârtibaciu River (Fig. 14). We estimated that the sewer should be able to handle a higher flow rate of 10 l/s.

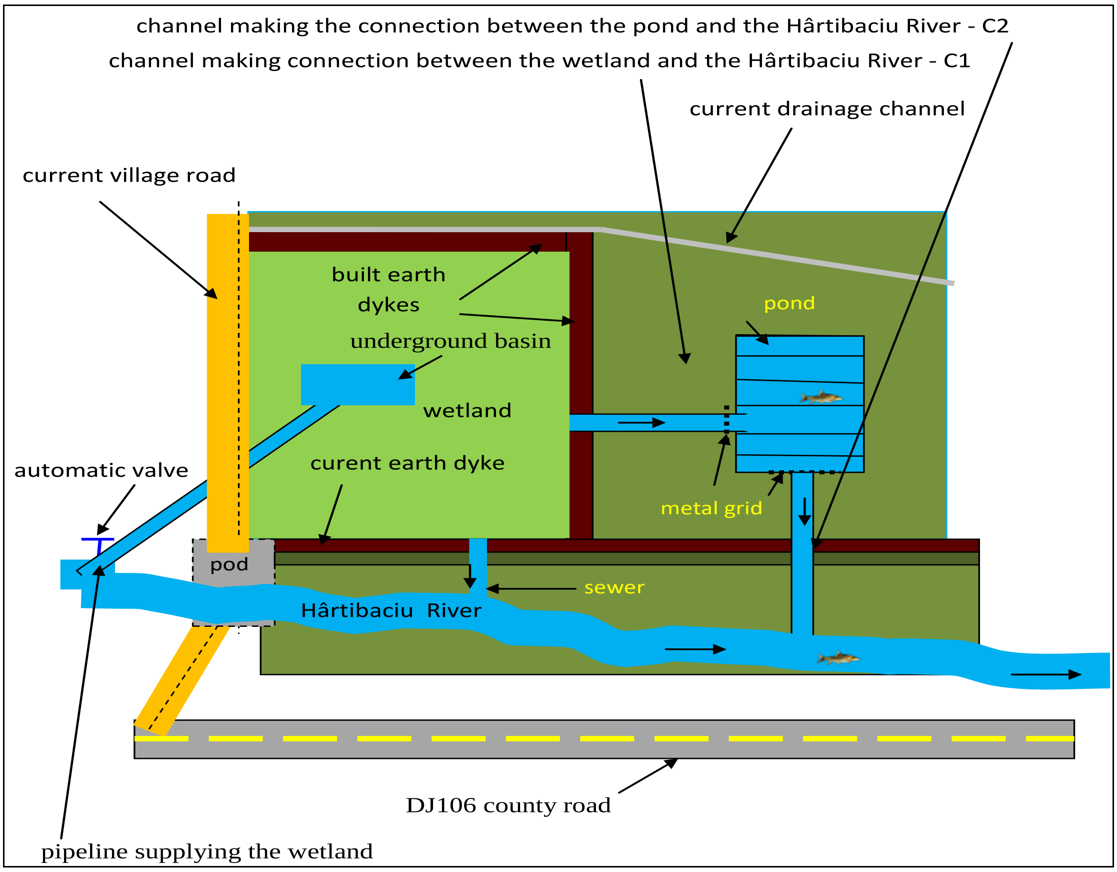

Figure 14: General scheme proposed for achieving wetland – indicative scheme.

An important aspect to stress is that the process to supply water to the wetland is achieved gravitationally, and the excess water in the wetland will be directed towards the Hârtibaciu River in a similar way.

CONCLUSIONS

All the fish species presently experience major human impacts along the Hârtibaciu studied sector, in the proximity of the Alţâna location.

The newly proposed constructed wetland sets in place the foundation for the rehabilitation of the lateral connectivity and the habitat quality for the single fish species of conservation interest sampled there *Rhodeus sericeus amarus*, and would increase the population numbers of this species.

The creation of the new proposed wetland would constitute an important buffer zone for fish, especially in the dry and cold (with frozen periods) seasons.

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A TOURISTIC CONCEPT OF LAND MANAGEMENT OF ZAKLIKÓW'S RESERVOIR BASED ON ITS NATURAL VALUES (SOUTH-EASTERN POLAND)

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DOI: 10.1515/trser-2015-0089

KEYWORDS: natural and tourist values, concept of development.

ABSTRACT

The aim of the study was to determine actions towards enrichment of natural aspects of the water reservoir in Zaklików, with consideration of nature protection and social needs. For this purpose, physical and chemical parameters of the reservoir, floristic and physiognomic characteristics of the area, cartographic analysis of land use, natural and touristic valorisation, and the concept of tourism development of the area were considered. The reservoir, based on the ESMI index, rated as moderate ecological status. The highest natural and tourist values, as well as the highest intensity of conflict between them, were located in the north-eastern and central part of the study area. Based on all analyzes, a concept of development of the study area was created, consistent with the local development plan.

RÉSUMÉN: Un concepto turístico de manejo de tierras en el reservorio Zaklikow, en el contexto de valores naturales (sureste de Polonia).

El objetivo de este estudio fue determinar acciones para enriquecer de forma natural las aguas del reservorio Zaklików, tomando en cuenta la protección a la naturaleza y las necesidades sociales. Para tal fin, se muestrearon parámetros físicos y químicos en el reservorio, se midieron características florísticas y fisionómicas del área, se llevaron a cabo análisis cartográficos y de uso de suelo, se realizaron valoraciones turísticas y del capital natural así como también estudios del desarrollo turístico del área. Sobre la base del índice ESMI, el reservorio se catalogó con un grado de impacto ecológico moderado. Los valores más altos de capital natural y turístico, así como la más alta intensidad de conflictos, se localizaron en las partes noreste y central del área de estudio. De acuerdo con los análisis se logró crear el concepto de desarrollo en el área de estudio, el cual es consistente con los planes locales de desarrollo.

REZUMAT: Un concept turistic de gestionare a terenurilor din zona lacului de acumulare Zaklików din punctul de vedere al valorilor sale naturale (sud-estul Poloniei).

Scopul studiului a fost acela de a elabora măsurile care să ducă la îmbogățirea naturală a lacului de acumulare Zaklików, luând în considerare protecţia naturii şi nevoile sociale. În acest sens, au fost evaluaţi parametrii fizici şi chimici ai lacului, caracteristicile floristice şi fizionomice ale zonei, analiza cartografică de utilizare a terenului, evaluarea elementelor naturale şi turistice, precum şi conceptul de dezvoltare turistică a zonei. Pe baza indicelui ESMI s-a stabilit o stare ecologică moderată a lacului de acumulare. Cele mai mari valori atât naturale cât şi turistice, precum şi conflictul cu cea mai mare intensitate s-au constat în partea de nord-est şi centrală a zonei de studiu. Pe baza tuturor analizelor, a fost creat conceptul de dezvoltare a zonei de studiu, în concordanţă cu planul de dezvoltare locală.

INTRODUCTION

Water is one of the most important components of the natural environment. Its suitable availability and quality is essential for the proper functioning of all organisms on the Earth (Poskrobko et al., 2007).

Poland is a country poorly resourced in water; its average abundance of surface waters is 62 km³. This indicator is three times smaller than the average value for the European Union and almost five times less than the average value of the world. In connection with the development of agriculture, industry, and other sectors of the economy, the demand for water has rapidly increased, while a very large part of it was contaminated. (Poskrobko et al., 2007)

According to the recommendation of the Water Framework Directive, the state and ecological potential of waters are defined by three basic criteria: biological (the composition and abundance of aquatic flora, benthic invertebrate fauna and composition, abundance and age structure of fish fauna), hydromorphological (hydrological system, the continuity of the watercourse, morphological conditions) and physico-chemical (salinity, state acidification, thermal conditions, biogenic, oxygenation, and pollution of priority substances identified as being discharged into water bodies, and contamination by other substances identified as being discharged in significant quantities water bodies) (Water Framework Directive 2000/60/EC, 2000).

In determination of the class of ecological potential of the water reservoirs, the importance of the lowest classified part of their quality obtained in the evaluation procedure is taken into account and accepted as the ultimate result (Water Framework Directive 2000/60/EC, 2000). Assessment of the state and ecological potential of waters allows activities to improve the quality of surface waters. It also allows their rational management, which is a key element in the struggle against the deterioration of water quality (Poskrobko et al., 2007).

Besides satisfying the needs of production and consumption, water is also used in tourism. Nowadays, properly arranged reservoirs represent one of the most commonly visited places by tourists (Kożuchowski, 2005).

Touristic values are specific elements and features of the geographical environment or various types of expressions of human activity – which are actual, available attractions for tourists (Bieńczyk, 2003). Their concentration is unequally distributed in the area (Migoń, 2012). The basic condition for discretion of an object or phenomenon as a touristic value, apart from having the characteristics of stimulating interest and permanent or temporary access for tourists, is its clarity in the landscape that allows for sensory perceptions.

The most important division of tourist values deemed to be their differentiation based on the origin. In this way, they are divided on natural values: created by nature and man-made created by human (anthropogenic). They complement, overlap, and depend on each other, and during their use it is quite difficult to distinguish between natural and anthropogenic components of space tourist's resources (Kożuchowski, 2005).

In tourism, natural values are features of elements of the environment that are assessed by tourists, arousing their interest and attracting them to a particular place (Boud-Bovy, 1985; Gaworecki, 2007; Lijewski et al., 2008). The size and direction of tourism are primarily shaped by natural values, constituting a major factor in attracting tourists. A diversified terrain, a mild climate, a presence of surface waters, and a presence of rich vegetation have a particular importance. However, the highest attractiveness is guaranteed by the contrast of natural landscape, treated as a set of these factors (Kowalczyk, 2000).

The second type of touristic values are the anthropogenic structure – material objects, closely associated with life, work, and human activity, produced by him in the process of historical development and being of interest to tourists. Among these anthropogenic values, historical monuments of architecture and engineering play the main role (Lijewski et al., 2008). Apart from tourist and natural values, one of the main factors attracting tourists is providing them with appropriate conditions for rest is suitable touristic management (Kowalczyk, 2000; Rogalewski, 1979; Sessa, 1983; Kruczek, 2012).

Surface waters are a very important natural value that affects the tourist attractiveness of an area. In many cases, their occurrence generates a choice of summer holidays by tourists (Młynarczyk and Zajadacz, 2008; Kowalczyk, 2000). Water reservoirs, together with the adjacent areas, create conditions for life of many flora and fauna species not found in terrestrial ecosystems, often under protection. In addition, surface waters are a valuable part of the visual landscape, increasing its aesthetic values (Kożuchowski, 2005).

Due to its tourist attractiveness, water bodies are particularly vulnerable to degradation because of the increase of tourism concentration in their surroundings. Therefore, they are very vacillating ecosystems, sensitive to external impacts. The main reasons of a negative impact of tourism on the water reservoirs are: water pollution by wastewater from touristic houses facilities; the lack of sewers; leaking septic tanks; leaking gasoline and oil from boats; as well as direct littering. The consequence is a whole range of negative effects, which can be divided into indirect and direct. Indirect groups include reducing conditions of aquatic ecosystems, and changing the composition of water and eutrophication. Among the direct effects there is a discernible increase in water turbidity, increasing the amount of nutrients, growth of harmful bacteria and other organisms, change in water quality and an introduction of alien species (Młynarczyk and Zajadacz, 2008).

Limiting the risks associated with the development of tourism requires undertaking actions, among which valorisation of the area is important key. It means evaluating and comparing with each other individual elements of a particular area. The aim of the tourist valorisation is to prepare the basis for finding the optimal method of natural resources use and to ensure the adequate development of land space use for recreational purposes (Cowell, 2007; Chmielewski, 2012). The obtained results approximately allow determining the probable impact of tourist events and tourism traffic on the environment (Kożuchowski, 2005).

The aim of the study was to determine actions to natural enrichment of the water reservoir in Zaklików and changes in the infrastructure and its development, with the consideration of nature protection and social needs. For this purpose, the following research was conducted: physical and chemical parameters of the reservoir; floristic and physiognomic characteristics of the area; cartographic analysis of land use; natural and touristic valorization, as well as the concept of tourism development of the area.

MATERIAL AND METHODS

The study area is located in the most northern part of the Podkarpacie Voivodship and Stalowa Wola District, in the city and municipality named Zaklików (Fig. 1). Due to the hydrological location, it is situated in a basin of the Vistula and the San rivers. In the area under the study, there are no protected areas or natural monuments. At the nearest neighborhood, there is the Natura 2000 called “Janowskie Forests”. It was created to protect and preserve the unique landscape character and is one of the biggest forest complexes in Poland – Solska Primeval.

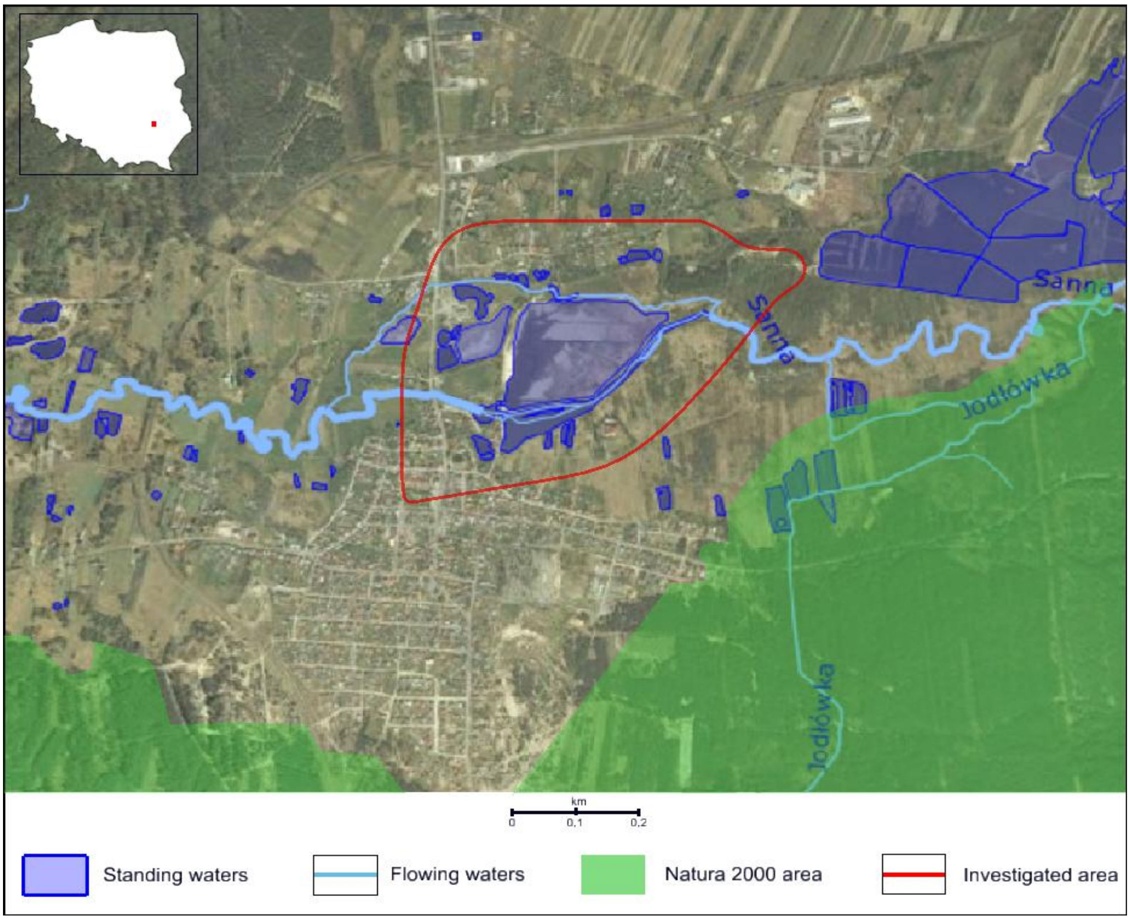

Figure 1: Study area location on the orthophotomap (http://mapy.geoportal.gov.pl/).

Within the analyzed area, the largest water reservoir is the one in Zaklików (Fig. 2), whose surface is approximately 20 ha. The reservoir was built in 1935 as a recreational object. Currently, its main function is retention – operation of flood control and recreation. The second tank is a fishing water body of approximately four ha, called by the inhabitants as the Giant Ferris Pond, because its waters supply a small private hydroelectric power. At the beginning of the XX century, it fuelled the local mill, as well as provided energy for the street lighting of Zaklików locality. Both the water reservoir and the fishing water body are powered by the Sanna River. The study included a buffer zone of the reservoir, amounting to 500 m zone around the reservoir.

Studied area was classified as area with favourable environmental conditions, slightly transformed by man. Among the risks is the lack of sewers and sewage treatment plants in the municipality, causing diffuse pollution of both water and soil through incorrect exploitation of septic reservoirs. Another is the lack of organizing and regularity in the collection and export of municipal waste from across the whole municipality. The consequence is the formation of illegal dumps affecting negatively on the whole environment. A negative phenomenon is also the illegal exploitation of sand dunes near the reservoirs. This results in the devastation of land and irrational management of resources. In the spring and autumn, there is a significant increase in the level of groundwater, rivers, as well as reservoirs, which in 2012 year contributed to the breaking of the dam and flooded parts of the city (Fig. 2) (Gurdak et al., 2013).

Figure 2: Burst dam on a fish pond in Zaklików in 2012 (http://wiadomosci.onet.pl/Kielce).

The study of physical and chemical properties of water and floral research were conducted in 2015 at four research sites located in two reservoirs in Zaklików. Research sites were taken: from fishing pond (1), whereas the other three from water reservoir: sequentially from the beach side (2), from the old Sanna riverbed (3) and at the mouth of the river to the reservoir (4) (Fig. 3). The selection of these research sites was according to variety of habitats and surroundings usage.

From each research site, water in quantities of about 100 ml was taken. Parameters that were analyzed: potential of hydrogen (pH); electrolytic conductivity (µS/cm), total hardness of water (mg $CaCO_3/dm^3$); and phosphate (mg $P\text{-}PO_4/dm^3$) and nitrates content (mg $N\text{-}NO_3/dm^3$). In addition, visibility of studied water using Secchi disc was investigated.

Vegetation studies were carried out in horizontal transects extending from the shoreline to the maximum depth of their occurrence (Sender, 2012a, b). Plant communities were examined and identified on the basis of the phytosociological method of Braun-Blanquet (Dzwonko, 2007). The syntaksonomic system was adopted from Matuszkiewicz (2013). The phytolittoral surface and the length of the shoreline inhabited by macrophytes were determined on the basis of real vegetation maps of the lakes created by the software Macrostation, version 8. Assessment of the ecological status of lakes was made based on Polish Ecological State Macrophyte Index ESMI officially recognized and accepted the task to monitor the stagnant reservoirs in Poland (Ciecierska et al, 2010):

$ESMI = 1 - exp\frac{-H*Z}{H_{max}}$, where:

H – plant diversity index;

H_{max} – maximum plant diversity;

Z – colonization index.

Figure 3: Distribution of investigated sites.

The analysis of physiognomy and land cover was based on field observations and interpretation of cartographic studies. On the basis of the way these maps were created they carried out the valorisation of the study area, where the system of Chmielewski's (2012) methodological assumptions was adopted. On the basis of the borders of land cover and terrain, the area was divided into natural-landscape units. Each of units was subjected to natural and touristic valorisation.

A natural valorisation was based on an assessment of:

- Ecosystem diversity. For each natural ecosystem, one point was added; for semi natural – 0.5 point (water reservoirs, wet meadows) and 0.25 point (drained meadows). If any ecosystem covered more than 50% area of unit, its points were doubled.
- Species diversity. For each flora and fauna point, the unit has one point.
- Threats like roads, agriculture, industrial objects. For each such object – one point was added.

The total sum points formed the basis of the environmental and tourism assessment. These values gave a basis to the calculation of the intensity of the conflict within each individual unit with the following formula:

$$K = \frac{(WP + WT) + 1}{(WP - WT) + 1}$$

where: K – intensity of the conflict; WP – sum points obtained from environmental valorization; WT – sum points obtained from tourism valorization.

The grading scale of the conflict's intensity:

From – 7 to – 3.8 – one point – very low;
From – 3.7 to – 0.6 – two points – low;
From – 0.5 to 2.6 – three points – moderate;
From 2.8 to 5.8 – four points – high;
From 5.9 to nine – five points – very high.

The next step was touristic valorization of all structural units of the investigated area, in which a five-point scale classification was prepared to rated them in categories such as:

- The diversity of terrain. Within each separated unit horizontal and vertical lines were designated. Then the sum of their intersections with the contour lines was calculated and their length in centimeters. These data are substituted into:

$$\frac{sum\ of\ intersections}{sum\ of\ length}$$

- Presence of standing waters. Each unit having within its borders received one point. If they occupied more than 50% of the unit area, the number of points was doubled.

- Presence of forests. Each unit having within its borders received one point. If they occupied more than 50% of the unit area, the number of points was doubled.

- Presence of cultural objects. Each object gave one point to the unit.

- Cultural objects. Each object gave one point to the unit.

- Viewpoints. Each object gave one point to the unit.

- Touristic management (accommodation and gastronomy objects, touristic routes). Each object gave one point to the unit.

- Destructive objects (roads, high voltage lines, agriculture, industrial objects). Each such an object gave one point to the unit.

Each assessment, presented as a map with different intensities of color, showed a richness of the data values of the analyzed area.

The next step was the calculation of intensity of conflict between them. After comparison of the results obtained for the individual structural units and the analysis of the zoning plan (Gurdak et al., 2013), in aim of organizing spatial structure, a division of the study area into functional zones have been made: nature protection zones, natural enrichment zones, and development of natural infrastructure zones. All analysis allowed the concept of tourism development of the study area.

RESULTS

Analysis of physico-chemical parameters allowed clearer determination of the seasonal changes in examined indicators (Tab. 1). Most of them increased in the autumn. The exception was nitrates, whose concentration in three research sites was much higher in the summer. The reason for this was mainly the impact of the intensity of precipitation, and hence runoff, variability of plants, and the varying intensity of the development area. By comparing the results of all the research sites, it could be noted that the high values for most indicators showed the fishing pond, which could be due to the close proximity to dense building, as well as intensive use of the pond for fishing purposes and energy targets. Other research sites located on the big reservoir were characterized by similar values, but the highest were noted in the site number 4, located at the mouth of the river to the reservoir. This suggested that the river was a carrier of many impurities, what was evidenced by the high degree of electrolytic conductivity at this site, it was also a carrier of nutrients, in particular nitrates. The best physico-chemical conditions were found in water taken from areas located below the beach. In these parts, the value of the pH of water excited close to neutral, low level of impurities, total hardness, nutrients, as well as the highest visibility of water. It could suggest that this was due to the wide angle of that place of water intake from the river mouth to the reservoir. Before water reached the other side of the reservoir, pollutants and nutrients were accumulated in large quantity by the plants in the tank. It proved the self-cleaning capacity of the tank. Water at both reservoirs had low values of total hardness, so their characters were very soft, probably influenced by the absence of chalky substrate.

Table 1: Results of physico-chemical analysis of water of Zaklików's reservoirs (S – summer, A – autumn).

Parametres	water reaction (pH)		electrolytic conductivity (μS/cm)		total hardness of water (mg $CaCO_3/dm^3$)		Phosphate (mg $P\text{-}PO_4/dm^3$)		Nitrates (mg $N\text{-}NO_3/dm^3$)	
Season / site	S	A	S	A	S	A	S	A	S	A
1	7.1	7.4	361.2	358.4	8.3	7.7	0.03	0.69	1.10	0.74
2	6.8	7.3	339.0	367.9	3.4	7.6	0.12	0.14	1.10	0.47
3	6.6	7.3	343.7	385.6	4.3	7.5	0.52	0.56	0.10	0.32
4	6.9	7.3	356.2	380.0	5.0	7.6	0.09	0.37	1.08	0.64

In the study area, there was a high variety of plant species (Tab. 2). They primarily constituted cosmopolitan plants, rush and aquatic vegetation, as well as woody water vegetation.

Table 2: identified plant species.

No.	species	quantity
1.	*Plantago major* L.	+
2.	*Heracleum sphondylium* L.	+
3.	*Heracleum sosnowskyi* Manden.	+
4.	*Hedera helix* L.	2
5.	*Geranium pratense* L.	2
6.	*Artemisia dracunculus* L.	1
7.	*Artemisia vulgaris* L.	3
8.	*Chelidonium majus* L.	1
9.	*Nuphar lutea* L.	1
10.	*Sambucus nigra* L.	2
11.	*Crataegus oxyacantha* L.	3
12.	*Carpinus betulus* L	1
13.	*Malus* Mill.	+
14.	*Acer platanoides* L.	+
15.	*Trifolium repens* L.	1
16.	*Oenanthe aquatica* (L.) Poir.	+
17.	*Lythrum salicaria* L.	+
18.	*Arctium* L.	1
19.	*Glyceria maxima* (Hartm.) Holmb.	2
20.	*Sonchus oleraceus* L.	2
21.	*Elodea canadensis* Michx.	1
22.	*Alnus* Mill.	2
23.	*Cirsium arvense* (L.) Scop.	+
24.	*Juglans regia* L.	+
25.	*Typha* L.	3
26.	*Tussilago farfara* L.	+
27.	*Aegopodium podagraria* L.	+
28.	*Urtica dioica* L.	2
29.	*Convolvulus arvensis* L.	2
30.	*Potamogeton natans*	2
31.	*Robinia pseudoacacia* L.	+
32.	*Eupatorium cannabinum* L.	+
33.	*Equisetum arvense* L.	+
34.	*Rumex hydrolapathum* Huds.	+
35.	*Ulmus minor* Mill.	+
36.	*Salix alba* L.	+
37.	*Salix alba* L.	+
38.	*Chamaenerion palustre* Scop.	+
39.	*Carex limosa* L.	+
40.	*Phragmites australis* (Cav.) Trin. ex Steud	4

Water vegetation did not create typical for lake distribution. The main component of water vegetation was rushes; submerged plants occurred sporadically (Fig. 4).

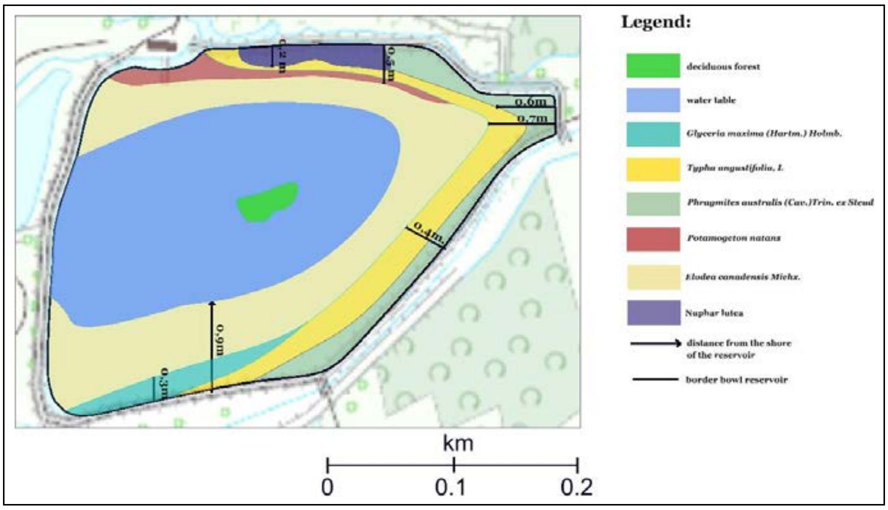

Figure 4: Distribution of plant vegetation in Zaklików Reservoir.

The ecological status of waters on the basis of ESMI ratio has been assessed as moderate. It means that the structure of the vegetation in the reservoir can be under rapid degradation, if any adverse factors will affect.

$$ESMI = 1 - exp\frac{-1{,}337 * 0{,}8075}{1{,}792} = 0{,}103$$

According to Chmielewski (2012), the research area was divided into 33 natural – landscape units. Within the units, based on the documentation (Gurdak et al., 2013) and field observations, the sites of protected species occurrence of fauna and flora were delimitated (Fig. 5).

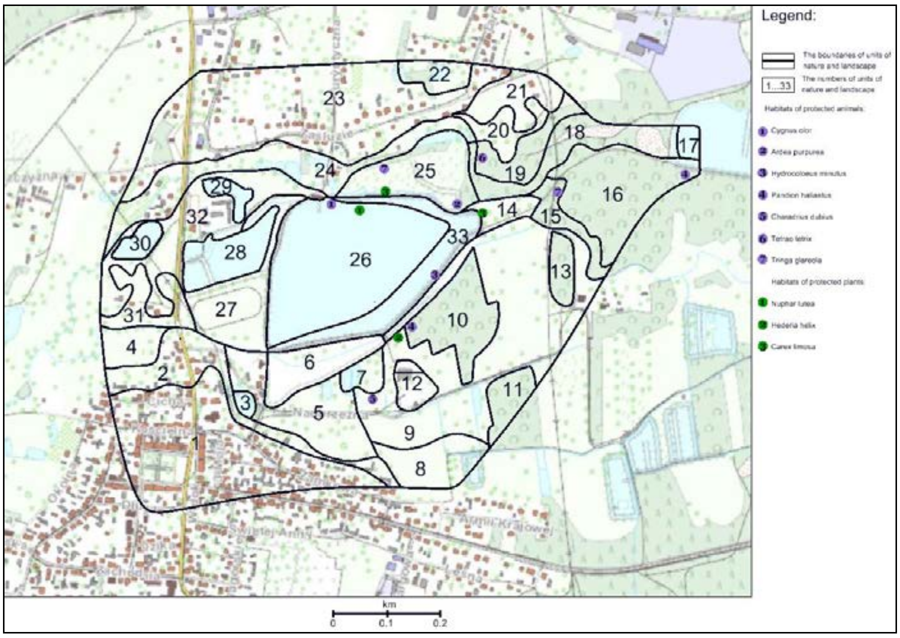

Figure 5: Protected flora and fauna species occurrence within borders of landscape units.

In terms of the occurrence of natural values, the most diverse unit was unit number 25, which is mostly, occupied by old the river Sanna bed (Fig. 6). This type of ecosystem favored the occurrence of many flora and fauna habitats. High natural values occurred in units with the following numbers: 10, 15, 16, 18, 19, 26 and 33, in the coverage of which dominated mainly forests, standing water, and rushes. The lowest natural values occurred in units' no. 1, 2, 20, 21, 23, 27 and 32. The main reason was low diversity, as well as the significant share of the various threats affecting negatively on the environment, such as roads or industrial objects.

The highest tourist values occurred in units' no. 18 and 19 (Fig. 7). This was due to steep slopes covered with forests, attractive for tourists and valuable points of presence of protected birds. High values were also characterized units with the numbers 5, 12 and 24, which were influenced by location within the elements of tourist infrastructure and diversity of the terrain. The lowest touristic values were characterized units number 20, 21 and 23, located in the north of the studied area. This affected the occurrence of unbalanced building, roads, agriculture and the absence of protected species, low diversity terrain, and a number of tourist infrastructure's elements.

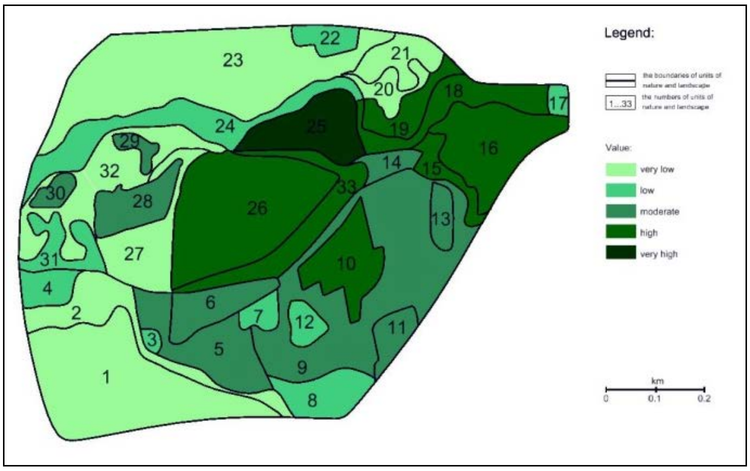

Figure 6: Natural valorization of area under the study.

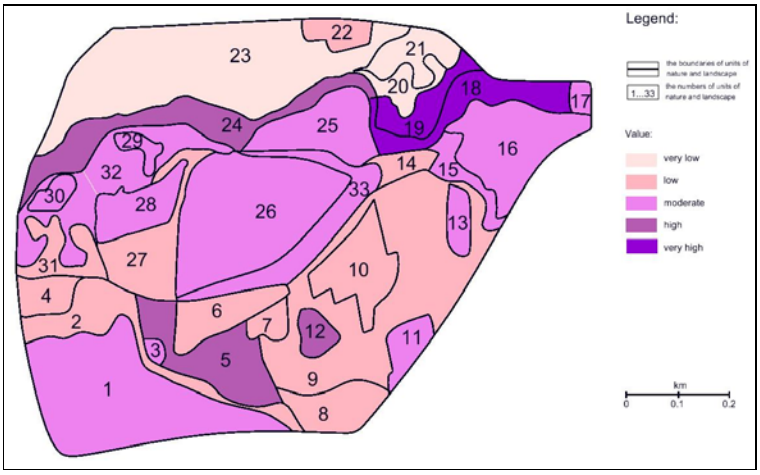

Figure 7: Touristic valorization of investigated area.

The highest intensity of a conflict was characterized by unit numbers 10, 15, 16, 26 and 33 (Fig. 8, Tab. 3), due to the presence in their borders valuable natural ecosystems and occurrence of protected plants and animals, as well as very high tourism values, encouraging people to visit these areas. A high intensity of conflict occurred also within the units with the numbers 6, 8, 14, 20, 21, 28, 29, and 30. They were characterized by comparable value of both nature and tourism. The lowest intensity of conflict appeared in the unit numbers 3, 5, 9, and 13, where the number of tourist attractions prevailed over the amount of natural values.

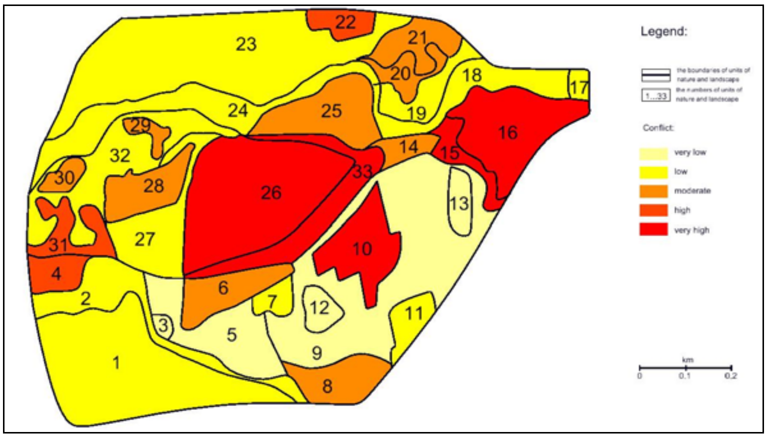

Figure 8: The intensity of the conflict between natural and touristic values of investigated area.

Table 3: The intensity of the conflict between natural and touristic values.

Number of unit	Natural valorisation	Touristic valorisation	Conflict	Scale bonitation
1	– 2	3	– 0.5	2
2	– 1	1	– 1	2
3	1	3	– 5	1
4	1	1	3	4
5	1.75	4	– 5.4	1
6	2	1	2	3
7	0	2	– 3	2
8	0.25	1	9	5
9	0.25	2	– 1.18	2
10	3	3	7	5
11	2	3	0	2
12	0.25	5	– 1.4	1
13	2	4	– 7	1
14	2.25	1	1.9	3
15	3	3	7	5
16	3	3	7	5
17	0.5	3	– 2.3	2
18	3	7	– 3.7	2
19	3	7	– 3.7	2
20	– 1	0	0	3
21	– 2	– 1	0	3

Table 3 (continued): The intensity of the conflict between natural and touristic values.

Number of unit	Natural valorisation	Touristic valorisation	Conflict	Scale bonitation
23	– 1.5	– 1	– 3	2
24	0.5	5	– 1.9	2
25	6	3	2.5	3
26	3	3	7	5
27	– 0.5	2	– 1.7	2
28	2	3	0	3
29	2	3	0	3
30	2	3	0	3
31	0.5	1	5	4
32	– 0.5	3	– 1.4	2
33	3	3	7	5

On the basis of the conflict intensity's analysis, one zone for the protection of nature, located in the north-eastern part of the study area, was delimitated (Fig. 9). Zones of enrichment of nature were separated in north-eastern and north-western part of the area. The other two areas were classified as areas dedicated to the development of tourism infrastructure.

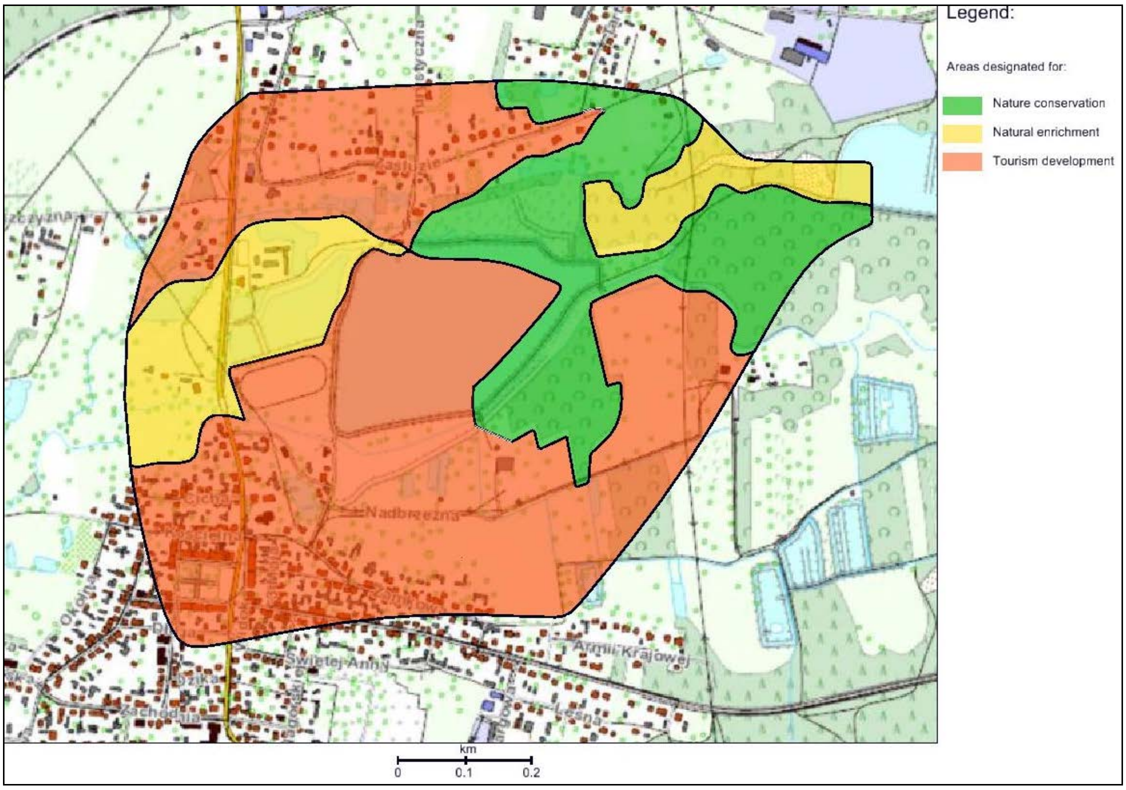

Figure 9: Designations of three type zones: zone of natural conservation (green), natural enrichment (yellow) and tourism development (orange).

On the basis of designated zones, as well as documentations (Gurdak et al., 2013), the concept of study area's development was developed (Fig. 10).

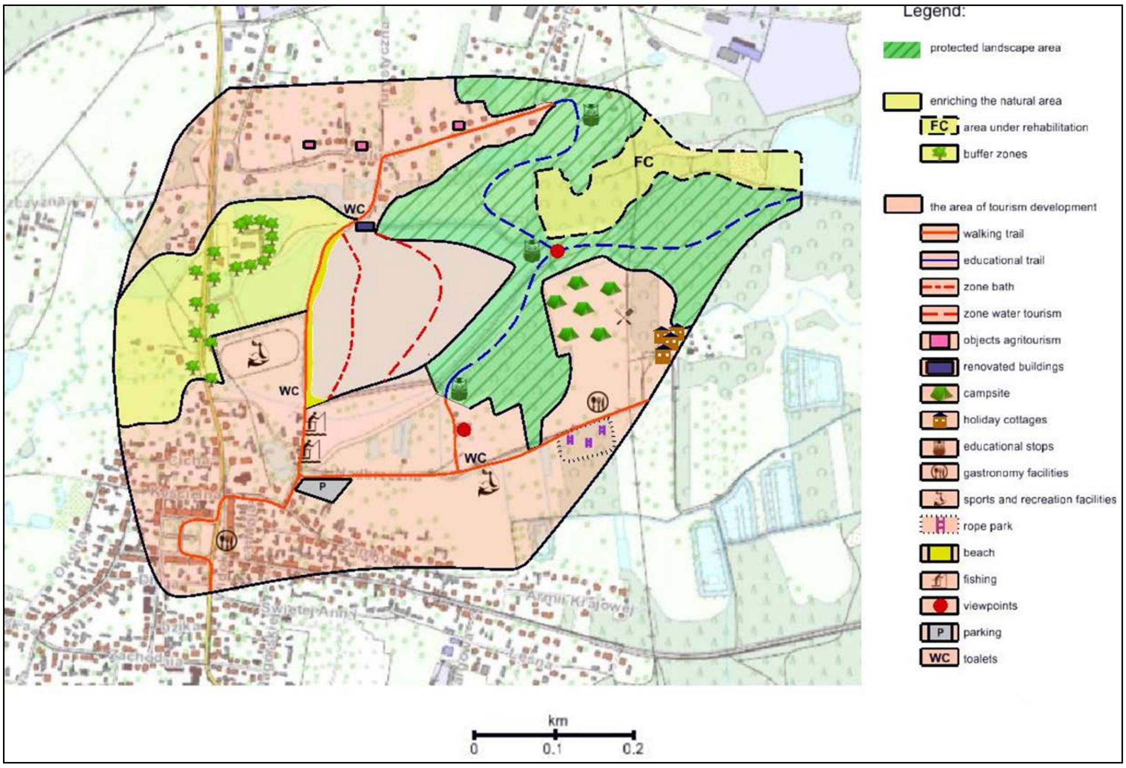

Figure 10: The concept of study area's development.

The natural protection zones comprised mainly of the eastern part of the study area, where deciduous forests, mixed forests, old river Sanna bed, rushes in the bowl of the tank, as well as peat excavations on the waterlogged meadows with the surrounding vegetation, occurred. These valuable ecosystems conducive to the occurrence of many plant habitats, as well as habitats of protected nesting birds. Due to the very high natural value, this part of the site should be taken under the legal protection. According to the study (Gurdak et al., 2013), it could be a protected landscape area. In order to satisfy social needs related to this area, it is suggested to appoint the nature path with educational stops and viewpoints, which would provide the association of residents and tourists with the local nature. The path would combine rich natural values of studied area with the nature reserve named "Łęka", localized in the "Janów Forests" Landscape Park.

Zones of natural enrichment concerned mainly areas with high and moderate natural values. In the eastern part of the area, they covered mixed forest, sand dunes, and part of the water tank. Due to the exploitation of sand, this area should be subjected to reclamation. In the western part, these zones were delimitated in the area including wet meadows, peat excavations with the surrounding vegetation and small water reservoirs. Also, an industrial object and the main road leading to the Zaklików city are in such zone. It was suggested to use plant vegetable protective belts as buffer zones to reduce pollution inputs into surface water, as well as the protection of natural habitats of wet meadows.

The development of a tourism infrastructure was recommended for the rest of the region. In the northern part, the agrotourism development was suggested where a compact farm and distributed buildings were located, which would constitute the base both to catering and accommodation facilities of the area. It was recommended to preserve green areas, pits peat land and small water bodies, affecting not only positively on the environment, but also enhancing its aesthetic value of observers. The zone designated for tourist development also included the reservoir, which would constitute the basis for the development of tourism of the whole analyzed area.

Due to their natural values, waters of the Sanna River that inflows into the reservoir and carries many contaminants, should be primarily clean. On the surface of the reservoir, it was recommended to separate zones for active tourism. Around the reservoir, it was suggested to make a path for tourists (walking, cycling, etc.). At the eastern shore, it is recommended to pour sand in order to expand the provided beach; a resort object located on the reservoir should be renovated and equipped with equipment for water sports. It would also be a good idea to research availability located near the sports field for tourists. In areas located in the southern part of the area (city Zaklików), it was recommended to create greenery areas and objects of small architecture. In the south eastern part of the study area, according to documentation (Gurdak et al., 2013), construction of another water reservoir was suggested. However, in our opinion, this is not a good idea because it requires felling approximately 10 hectares of deciduous forests and takes a large area of meadows. A better solution would be the construction of summer cottages, campsites, catering facilities and sanitation objects, which were in a little quantity over the study area. Also, it was recommended to create objects and sports and recreational facilities such as an outdoor gym or ropes course, as well as the construction of the parking near the reservoir.

CONCLUSIONS

Analysis of physical and chemical parameters showed clear seasonality of studied parameters.

The highest concentration of physic-chemical parameters shows the water in the fish pond, and the lowest, water from the site surrounded by meadows.

Floristic studies showed a large variety of vegetation, especially in the buffer zone. However, the reservoir, based on the ESMI index, rated as moderate ecological status.

On the analyzed area, there were many valuable habitats of plant and breeding birds. There were also a lot of unique morphological forms, as well as several interesting elements of tourist infrastructure.

The highest both natural and tourist values were located in the north-eastern and central part of the study area, the lowest in the north and south.

The highest intensity of the conflict occurred in the area with high tourist and natural values. They were located in the central and south-eastern part of the area. The lowest intensity of the conflicts occurred within the unit, where the tourist values prevailed over the natural values. They were distributed mainly in the southeastern part of the area.

Based on all analyzes, the developmental concept of the study area is consistent with the local development plan. A contentious issue is the construction of an artificial reservoir in the southeastern part of the area.

ACKNOWLEDGEMENTS
The authors would like to thank the reviewers for their input.

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