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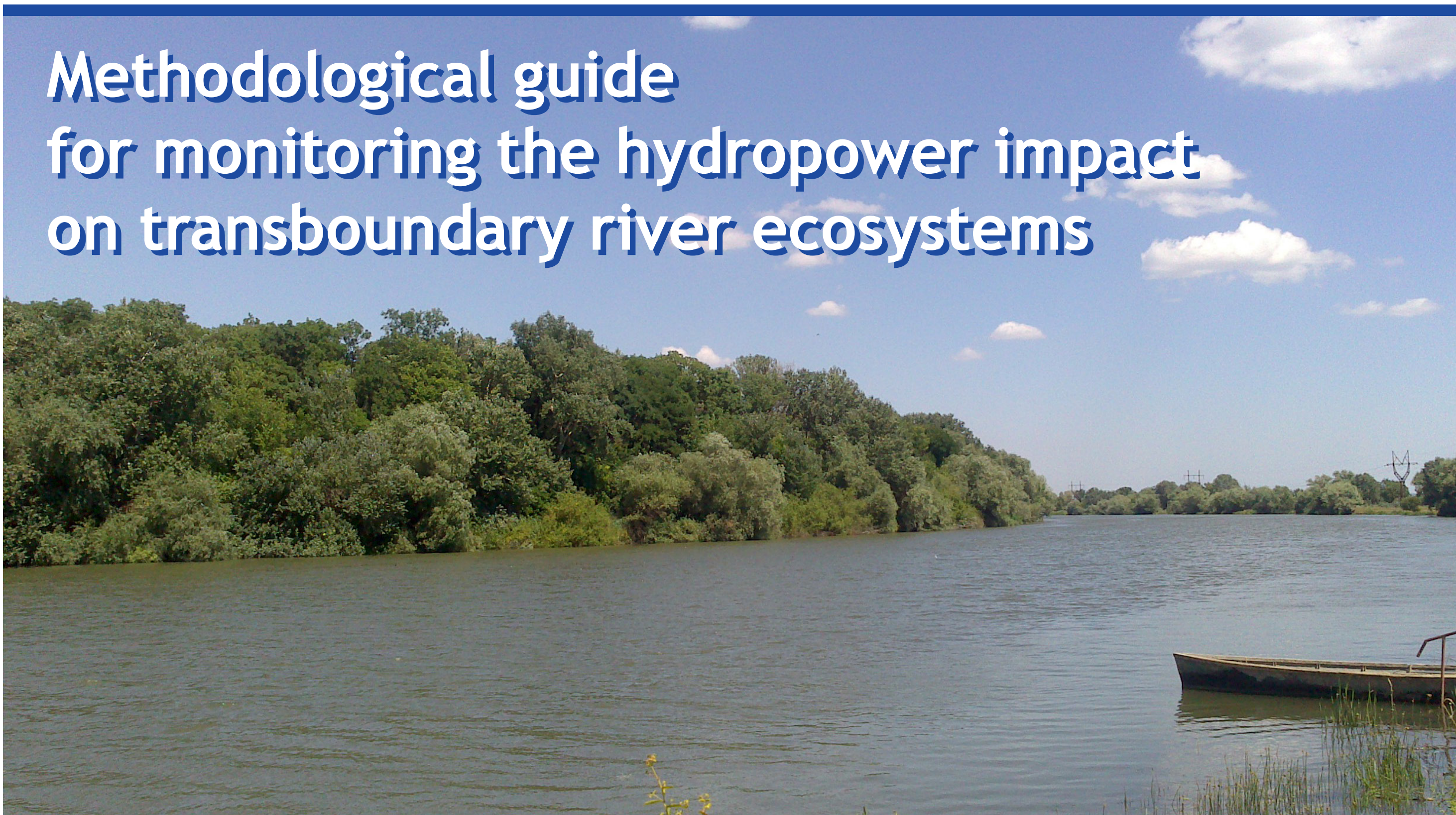
Methodological guide for monitoring the hydropower impact on transboundary river ecosystems



BSB165-HydroEcoNex

Editors:

Elena Zubcov

Lucia Biletchi

Project funded by
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The book will guide the researchers, specialists in aquatic ecology, hydrology, youth (students of all levels of the higher education institutions) and all other interested people in their acquaintance with the theoretical and applied aspects of monitoring the impact of hydropower complex operation and climate change on ecological status and functioning of aquatic ecosystems, including processes occurring in their watersheds.

Project BSB165-HydroEcoNex

Editors:
Elena Zubcov, correspondent member of ASM, doctor habilitat in biological sciences, professor
Lucia Biletchi, doctor in biological sciences, associate professor

Design and layout:
Natalia Dorogan

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INTRODUCTION

Zubcov Elena, Ungureanu Laurentia, Biletchi Lucia
Institute of Zoology

Before presenting the changes in transboundary aquatic ecosystems under the impact of hydropower complexes, it should be noted that for a long time these complexes were included in the list of companies for obtaining of so-called "green energy", what is true if to make a comparison with the impact of thermal power plants. Nevertheless, it has already been proven for several regions that hydropower complexes or power plants, with dams built on riverbeds, although are not polluting sources, destroy running or lotic ecosystems. The public interest in this topic is also demonstrated by the fact that the annual reports for 2000-2021 of the World Dam Commission, which reflect a wide range of interests of all those involved in the debate on river dam issues, are quite visualised. The current situation on the Dnieper and Volga rivers may be an other eloquent example (Dams and development: a new methodological framework for decision making, 2009). This evidence allows us to state that hydropower cannot be on the "green energy" list. The damming of transboundary rivers not only causes ecological and economic problems, but also leads to various conflicts (Field, 2021; Water conflicts and resistance issues and challenges in South Asia, 2021).

Currently, the quantity and quality of inland waters have been already recognized as a major global direct threat to human health, being one of the most pressing human problems, including from the point of view of the human right to safe drinking water. The European directives (Directive 2000/60/EC) lay down several environmental issues, but the majority of population and authorities of different level focus, primarily, on that of pollution. The annexes and regulations of several directives include limit values for the content of heavy metals, petroleum substances, pesticides, detergents and other toxic and hazardous substances, which enter the environment in the process of human activity, less – issues of biodiversity conservation and very little – those of the functioning of lotic and lentic ecosystems.

The following definitions are given in the Water Framework Directive (Directive 2000/60/EC): river means "*a body of inland water flowing for the most part on the surface of the land but which may flow underground for part of its course*"; river basin means "*the area of land from which all surface run-off flows through a sequence of streams, rivers and, possibly, lakes into the sea at a single river mouth, estuary or delta*".

Further it can be read: heavily modified water body means "*a body of surface water which as a result of physical alterations by human activity is substantially changed in character…*" (water speed, discharge, suspensions, etc.).

This means that the monitoring program of a river, as a running water body, refers, in particular, to the assessment of the volume and level or discharge of water, the speed of water flow, the amount of suspended substances and alluvium, as these data are important for appreciation of the ecological, chemical state of the river. They determine the balance in the "water-suspensions-silts" system and the ecological potential of the river.

In some cases, like that of the Dniester river, the impact of the dam is supplemented with the effect of thermal pollution, or change of thermal regime, which is closely related to the gas regime, the processes of development and reproduction of aquatic organisms and, obviously, to the production-destruction processes and the intensity of cycle and migration of chemicals. Deciphering and evaluating these processes means not only finding one or another indicator of water quality or the number of some groups of hydrobionts, but also establishing the characteristic relationships for a lotic ecosystem between chemical or biological components, the ratio between different hydrological, hydrochemical, hydrobiological and ecotoxicological factors.

Impact of damming rivers for hydropower purposes and regulating the flow of water downstream of dams only on the base of the needs of hydropower industry is exacerbated by climate change.

Investigations carried out by the beneficiaries (Institute of Zoology, International Environmental Association of River Keepers Eco-Tiras, Dunarea de Jos University of Galati, Ukrainian Scientific Center of Ecology of the Sea, Hydrometeorological Center for Black and Azov Seas) of the project BSB 165 HydroEcoNex *Creating a system of innovative transboundary monitoring of the Black Sea river ecosystems transformation under impacts of hydropower development and climate change*, financed by the European Union within the Joint Operational Program Black Sea Basin 2014-2020, has actually shown that the Dniester and Prut rivers are ecosystems heavily modified by hydrotechnical constructions.

The aim and objectives of the project are highly relevant and of great resonance. Among arguments in favour of this statement can be the invitation of the Committee on Migration, Refugees and Displaced Persons of the Parliamentary Assembly of the Council of Europe in the meeting from March of 2021, in order to present our vision on the impact of the Dniester Hydropower Complex (DHPC) on the Dniester river ecosystem.

1 Chapter

MODIFICATION OF AQUATIC ECOSYSTEMS AND ECOLOGICAL INDICATORS FOR MONITORING THE IMPACT OF HYDROPOWER COMPLEXES ON CROSS-BORDER AQUATIC ECOSYSTEMS

Zubcov Elena[1], Ungureanu Laurentia[1], Biletchi Lucia[1], Bagrin Nina[1], Andreev Nadejda[1], Zubcov Natalia[1], Ene Antoaneta [2], Jurminskaia Olga[1], Ciornea Victor[1], Subernetkii Igor[1]

[1]Institute of Zoology, [2]Dunarea de Jos University of Galati

1.1. General considerations about the hydropower complexes in the hydrographic basins of the Dniester and Prut rivers

The Dniester river not only means a running water body for Moldova, as it is the main source of supply with drinking water, water for irrigation and for the economy, in general. Many years the Dniester was navigable. The river was a habitat of valuable species of fish (sturgeons, salmonids, carp, etc.) and the banks of the river were the most densely populated territories.

The Dniester begins in the north-western part of the Eastern Carpathians, on the slope of Rozlici Mountain. The river length is equal to 1352 km, the surface of hydrographic basin – 72100 km^2, including within the limits of Moldova – 657 km and, respectively, 19000 km^2. Until the 2000s, the average multiannual flow was around 10 km^3. In the upper course (Carpathian mountainous area), the river has a deep valley and stony riverbed and layers of limestone and sandstone are visible on the banks. In some places, fragments of mountain rocks block the riverbed (so-called thresholds). In the lower course, it is a typical plain river, with a wide and low meadow.

The Dniester crosses the territory of Ukraine, then the territory of the Republic of Moldova from Naslavcea to Palanca and flows into the Dniester liman of the Black Sea, southwest of Odessa city. The Dniester river basin is located on the territory of three countries – Ukraine, Moldova and Poland. From the territory of the latter only a small stream brings its waters into the river *(Fig. 1.1)*.

In 1954, in the lower part of the middle course of the Dniester, between Camenca and Dubasari towns, the Dubasari reservoir was built. Its length is equal to 128 km, the width – from 200 to 1800 m (on average – 528 m), the water surface is of 6570 ha, the average depth – around 7 m and the complete volume – 485.5 million m^3.

In 1981, the Novodnestrovsk reservoir was built on the river sector from Ojevo village, Sochireanskii district, Chernivtsi oblasti, to Ustie village, Borshevsk district, Ternopol oblasti. The reservoir length is of 214 km, the width varies from 200 to 3750 m and the depth – from 3 to 56 m (in the lower sector) *(Fig. 1.2)*.

Later, since 1983, the Dniester water has been discharged downstream of this dam from the depth, through HPP-1 turbines, having a permanent temperature of about 9 ^{o}C, what caused very big changes in the thermal regime of the Dniester *(Fig. 1.3)*.

Figure 1.1. Map of the hydrographic basin of the Dniester river.
Source: https://unece.org/sites/default/files/2021-04/Dniester_English_web.pdf

Figure 1.2. Dnestrovsk (Ukraine): reservoir and the hydropower plant 1 (HPP-1) of the Dniester Hydropower Complex (DHPC) built on the riverbed.
Source: https://uges.com.ua/ru/ content/dnestrovskaya-gaes

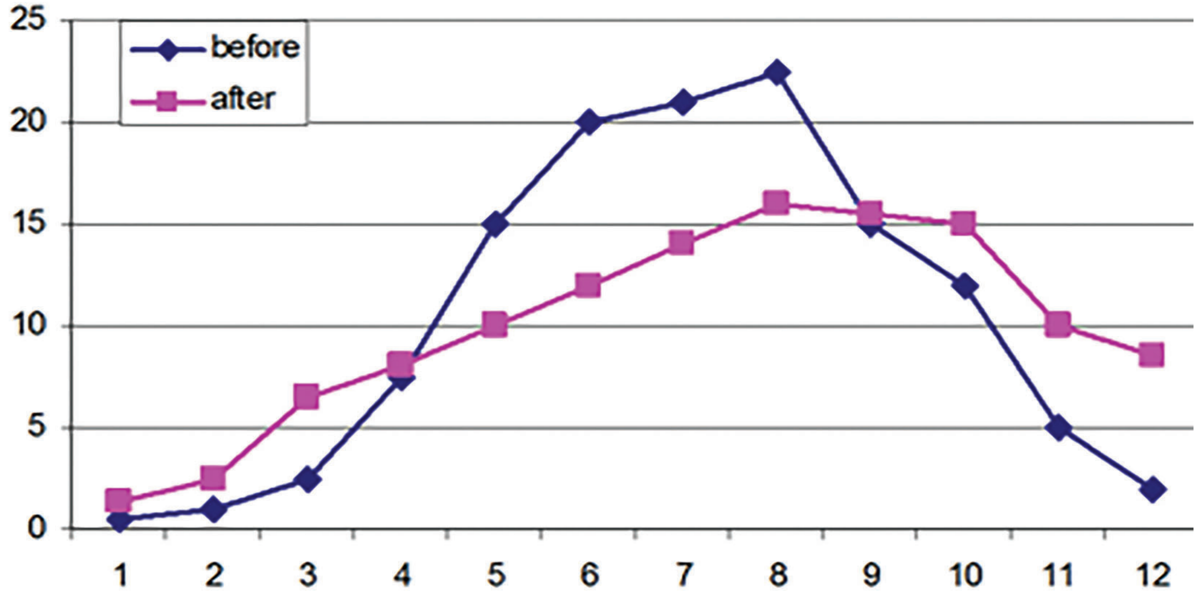


Figure 1.3. Water temperature downstream of Naslavcea station at the entrance of the Dniester river on the territory of the Republic of Moldova (Zubcov, 2007)

As result, the Dniester has not frozen even in one winter on the 20 km river section between Dnestrovsk and Naslavcea since 1983, even at the air temperature of minus 26 ^{o}C. For this territory, fog has become a norm in the autumn-winter-spring period, thus, the local effect of DHPC on climatic parameters is observed. It is known that the thermal regime of aquatic ecosystems also reflects on the gas regime, biochemical and chemical oxygen demand, on the reproduction of aquatic organisms, including fish (Zubcov, 2007).

Based on the working regime of HPP-1, downstream of the CHE-1 dam, the water level in the Dniester could increase or decrease very suddenly – up to 1.5-2.0 m/hour. Sometimes, in 15 minutes, the water level rises sharply by 1 m.

In the 80s and 90s of the last century, as part of the program of ecological, economic and social investigations in the hydrographic basins of the Dnieper, Pripiaty and Dniester rivers, the Institute of Zoology initiated investigations to estimate the impact of HPP-1 on the middle and lower Dniester. Institute also participated, as an expert organization, in detailing of the first operation regulation of HPP-1, which stipulated the importance of this complex for drinking water supply, irrigation, generation of hydroelectric power and mitigation of the negative effects of floods and droughts, characteristic for the Dniester hydrographic basin.

In order to mitigate the rise in water level and temperature, a dam was built upstream of Naslavcea village. In this way, a buffer reservoir was built on a distance of 20 km between HPP-1 dam and Naslavcea dam.

Unfortunately, Ukraine installed three hydropower turbines in this dam between 1991 and 1992, this being the beginning of HPP-2 *(Fig. 1.4)*. In such a way, this reservoir lost its original function.

Construction of the above-mentioned reservoirs and the operation of these two hydropower plants essentially influenced the hydrological, hydrochemical and hydrobiological regimes of the Dniester river, but the construction of the Pumped Storage Hydropower Plant (PSHPP) and a reservoir of this plant on the right side of the buffer reservoir will completely destroy the Dniester *(Fig. 1.5)*. Currently, the buffer reservoir has already been transformed in the technological reservoir of PSHPP.

Figure 1.4. Hydropower plant 2 (HPP-2) with the reservoir of DHPC built on the riverbed upstream Naslavcea.
Source: https://uges.com.ua/ru/ content/ dnestrovskaya-gaes

Figure 1.5. PSHPP with the largest reservoir built on the karst bank of the Dniester near Ocnita town; the reservoir is fed from the Dniester river.
Source: https://uges.com.ua/ru/content/dnestrovskaya-gaes

It should be mentioned that the construction of this complex constitutes a very big danger not only for the Dniester river, but also for at least one third of the territory of the Republic of Moldova. Four turbines have already been in operation here. Each turbine needs 260-280 m^3/s of water – namely, this was the discharge of water in the Dniester at the entrance on the territory of the Republic of Moldova until the construction of PSHPP.

In 2005-2006, the researchers of the Institute of Zoology, after receiving of information from colleagues from the Institute of Hydrobiology (Kyiv, Ukraine) about the starting of PSHPP construction, visited the construction area, where at that time works on widening and deepening the buffer reservoir and building the reservoir on the karst bank of the Dniester were in progress.

With the support of the Presidium of the Academy of Sciences of Moldova (ASM), the Institute of Zoology, together with NGO Eco-Tiras, organized a round table at ASM, after which open letters were sent to the Government and Parliament of both countries, to the scientific community and world organizations. Several discussions were held at different levels, gaining the support of many bodies, including the Presidium of the National Academy of Sciences of Ukraine, led by academician Boris Paton. Also, a commission for negotiations at Government level was created in that period. Unfortunately, the assumptions of researchers from that period have already become a reality: the Dniester, as river, is degrading day by day.

PSHPP pumps water from the Dniester into its reservoir by using electricity, in order to reduce the jump of electricity in the grid. Then the turbines, which are placed deep in the ground directly on the right bank of the Dniester, are fed with water from reservoir to generate electricity. During the visit to this complex in 2019, the chief engineer said that two turbines work 24 hours a day, and one – 12 hours. This year, the forth turbine is put into operation and three more turbines are planned to be installed.

Not being engineers in the field, we can assume that PSHPP already works to obtain energy cheaper than that produced by ordinary HPPs, therefore, the ecological problems of the Dniester downstream of PSHPP does not bother the employees and owners of this complex. Based on the information placed on the websites of hydropower plants, PSHPP has already exceeded the planned volume of electricity production from the initial project, developed in the 80's of the last century.

Prut is the second largest river in Moldova. It springs on the highest peak (Goverla Mountain) of the Ukrainian Carpathians, near the Vorohta village. The length of the river is of 898 km, within the borders of Moldova – 695 km, the surface of the hydrographic basin – 27500 km^2. The presence of a large number of small tributaries and the lack of large ones are characteristic for the Prut river basin. The Prut flows into the Danube at 174 km from its delta, thus representing the last large left tributary of one of the largest rivers in Europe.

According to the character of water supply and the hydrological regime, the Prut river resembles the Dniester river, but, obviously, it has a lower water flow – about 2.9 km^3. Prut has a V-shaped valley with a width of 3 km in the mountainous region, a trapezoidal shape – from Lipcani town and down on the river, with a width from 3-7 km to 12 km in the delta. Atmospheric precipitation is the main source of supply.

In 1978, at a distance of 560 km from the mouth of the Prut river, the Costesti-Stanca reservoir was built, with a length of 60-90 km, the average width – 1 km, surface – 59 km^2, depth near the dam – 41.5 m, average depth – 12.5 m, full volume – 735 million m^3. Reservoir has a seasonal regulation and its water is changed completely every 4 months. The reservoir has many bays, and there are numerous springs along the banks. The Costesti-Stanca hydropower plant is a joint plant of Romania and the Republic of Moldova, the ecological aspects of the Prut river downstream of the dam being solved jointly. Hydrological changes are visibly smaller than in the Dniester hydrographic basin. Consequently, the characteristic properties for the lotic ecosystems have been preserved in the Prut river.

1.2. Methodological aspects

Development of scientific bases for monitoring, estimating the functioning of aquatic ecosystems, with the aim to reduce the technogenic effects on the aquatic environment, has now become a global priority in environmental research. Innovative monitoring tools and gaining in-depth knowledge about the state and processes that take place in the aquatic environment can be ensured by the correct use of methods and techniques, by establishing different regularities of physico-chemical and biological processes in the investigated ecosystems.

In the process of implementing the project, expeditions were made in order to conduct complex investigations of aquatic ecosystems in the hydrographic basins of the Dniester and Prut rivers, including joint expeditions with partners. Field investigations and laboratory modelling were performed. The multi-annual materials of the Institute of Zoology, which began in the 1940s, have also been systematized.

Sampling of water, suspensions and silts, collection of biological samples, laboratory analyses of physico-chemical parameters, chemical-analytical determinations of macro- and microcomponents in water, various field and laboratory modelling, evaluation and determination of the quality of investigated water is carried out in accordance with ISO standards adapted to national ones and summarized in two recently developed and published guides (Hydrochemical and hydrobiological sampling guidance, 2015; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020).

Systematization of information on the state and functioning of ecosystems is made by using the Statistics-10, Excel-10, Paradox programs, dispersive analysis ANOVA etc.

For the field investigations, the Volkswagen Caravelle vehicle, which is equipped with a refrigerator, filtration systems and several accessories for sampling, was used.

The laboratory investigations were conducted by using performant equipment, including: inductively coupled plasma optical emission spectrometer (ICP OES) – ICAP 6000, atomic absorption spectrophotometer AAS Analyst-400, spectrophotometer Specord 230, three gas chromatographs – Clarus 500, Agilent-MS and UHPLC Flexar FX 20, pH-meters, gasometers, digital spectrometers Sartorius PB 11-P11, digestion system Berghof SPEEDWAVE, acid distillation system Berghof, centrifuge Hettich Rotina 420, oven Nabertherm CV3/11/B170, automatic burettes, analytical balances, thermostats, microscope MISMED/2 (LOMO), microscope *Axio Imager A.2 (*Zeiss*)*, microscope *Axio Imager A.2 for epi-fluorescence (*Zeiss), binocular *Stereo Discovery. V8 (*Zeiss*)*, binocular Minimed-502. All equipment and microscopes are computerized, which reduces the possible errors of researchers.

1.3. Status of aquatic ecosystems and ecological monitoring indicators

Physical-geographical factors (composition and condition of soils and mountain rocks, transformation of landscape, character of precipitations, state of groundwater), including the climate change in the river hydrographic basins have a dominant role in the functioning of lotic ecosystems. It should be noted that the reduction of area and poor maintenance of the river protection zones, reduction of forest areas in catchment basins of aquatic ecosystems, damming of rivers, including with hydropower purpose, are factors which affect the formation of discharge of running waters and, as a result, increase the hydrological drought and the extent of floods caused by human activity. Finally, the impact of the discharge of industrial and household wastewater, of runoff from agricultural fields and urbanized territories on the functioning of aquatic ecosystems shall be mentioned (Zubcov et al., 2020).

There are different methods of analysis and evaluation of hydrological data, most of them are grouped according to different indicators over a long period. Real data starting with 1976 (seven years until the commissioning of HPP-1) and until 2017, what means a period of 41 years are presented in *Figure1.6*. It can be observed that in those 7 years before the construction of HPP-1 the water discharge oscillated within the limits of 324-610 m^3/s, being on average of 422 m^3/s; in the 34 years after the commissioning of HPP-1 the average

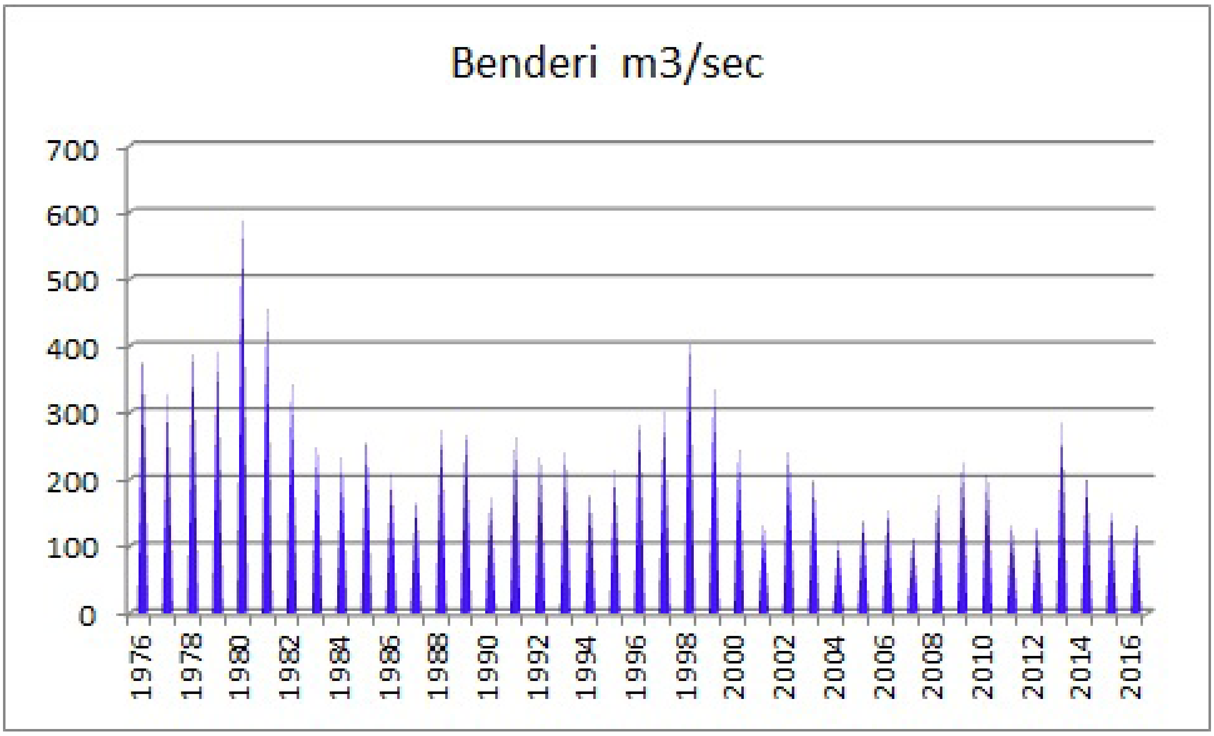


Figure 1.6. Water discharge of the Dniester river at Bender hydrological post, annual average, m^3/s. Source: Hydrometeo USSR,1976-1992 and data from Bender

value was of 218 m^3/s and in only 1 year it was more than 400 m^3/s, in 2 years – more than 300 m^3/s, in 17 years – more than 200 m^3/s and in 14 years – more than 100 m^3/s. In Camenca and Moghiliov these values are even lower, but we do not have data for all years. Thus, the impact of DHPC is quite visible.

In most of appraisals made by other specialists, the years when HPP-1 was put into operation are excluded, or comparisons are made with the data obtained at Zaleshciki post, which is 200 km upstream of the HPP-1 dam, without analysing the water volume of tributaries flowing into the reservoir. In some analyses, the period of 1980s-1990s or years during the installation of PSHPP turbines are excluded, or the average values are calculated separately for the years with an increased flow and for the years with drought. We consider that the latter data are extremely important for the analysis of climatic processes.

An indicator of the state of the Dniester is also the volume of the river flow at Mayaki post *(Fig. 1.7)*.

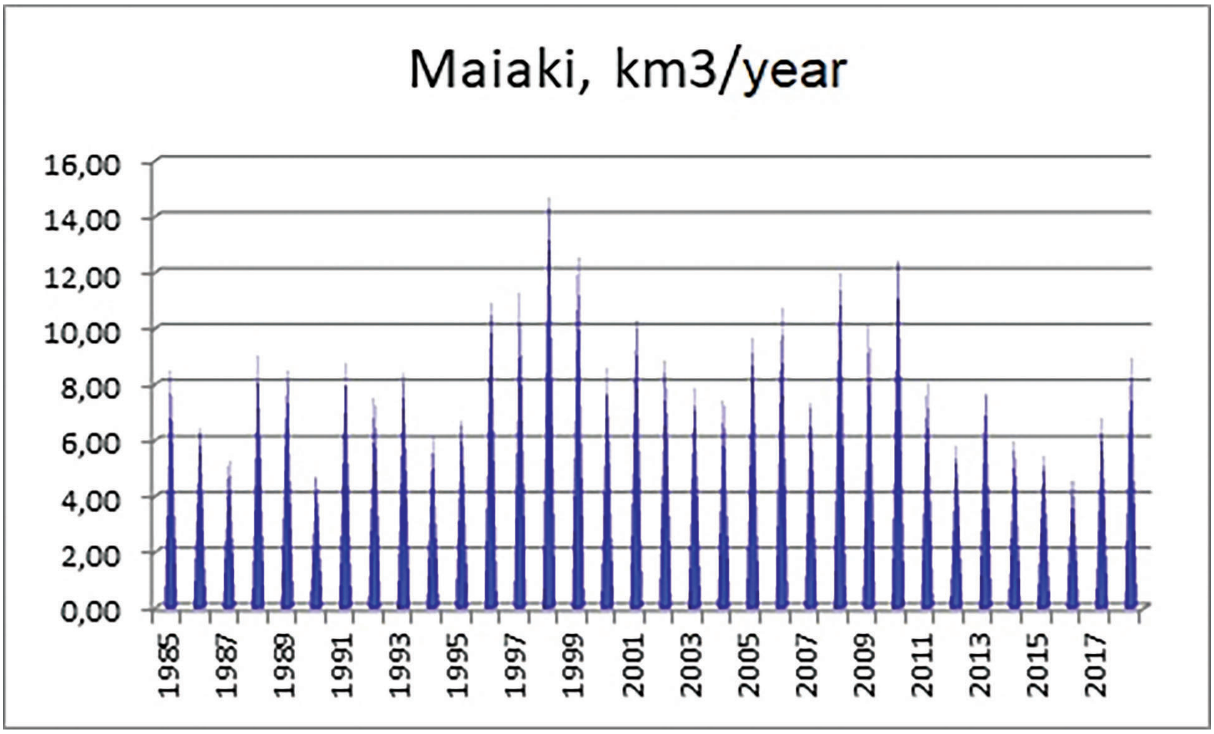


Figure 1.7. Annual flow of the Dniester at Mayaki, km^3.
Source: data calculated by partners from HydroEcoNex project from the Hydrometeorological Center for Black and Azov Seas

Unfortunately, no official multiannual data on water speed along rivers are available. However, being almost monthly in expeditions, we noticed more than once, especially in the middle sector of the river, an extremely low water speed. The same is regularly recorded in the area of Palanca, but at this station the coming up of water from the lower sector of the river, especially during the low water, is observed. A long period, the Institute of Zoology had hydrologists among its research staff, who were making all hydrological measurements when collecting hydrochemical samples, were systematizing the results of Hydrometeo, which were published annually in registers of restricted usage, available to researchers in scientific libraries.

Water level, speed, discharge are the necessary target indicators for assessing the influence of hydropower complexes on the state of dammed running ecosystems. Obviously, hydrological measurements downstream of dams are needed. Taking into account the problems we face today in estimating the ecological situation of the Dniester, it is very important to have a hydrological station, which would work online, in any area downstream of DHPC, but upstream of Dubasari reservoir. Hydrological data (water speed, level, discharge, and temperature) are part of the category of main data, mandatory for monitoring and evaluating the impact of DHPC on the state of the lower and middle Dniester. It is known that modification of these indicators can cause serious and often irreversible changes in running water.

In natural river ecosystems, which not undergo an anthropogenic transformation (reference ecosystems), there is a balance between physico-chemical, chemical and biological parameters, more exactly, there is a system “water – suspensions – bottom sediments – hydrobionts”, which is very mobile, but, at the same time, specific and stable for each water body. Namely, this balance determines the processes of functioning of aquatic ecosystems. The latter serve as a ground for monitoring the processes that also take place in the hydrographic basin.

Speed of water flow, the origin of suspended solids and alluvium, their particle size, mineralogical and chemical composition were in the past and remain today necessary parameters for assessing the cycle and migration capacity of chemicals in running ecosystems and the erosion-denudation processes in a hydrographic basin.

For example, the dependence of the concentration of metals in water and in suspensions from the quantity of suspensions in water (S, mg/l) and the water discharge (Q, m^3/s) was established for a range of metals (Al, Cu, Zn, Ni, Pb, Ti, Sn, Fe, Mn etc.). In the case of copper, this dependence is described by the equation (Zubcov, Zubcov, 2013; Zubcov et al., 2016):

$$Cu = 0{,}069 \cdot S + 0{,}099 \cdot Q - 16{,}4, \quad R = 0{,}88.$$

In the 1990s, the values of the correlation coefficient decreased to values that indicate a weak correlation (0.5), but in the last 20 years, such correlations no longer exist. Moreover, the concentrations of most investigated metals in filtered water through membrane filters with a pore size of 45 µm is higher than their concentration in suspensions (Zubcov, 2007). This demonstrates the radical modification of migration cycles, in this case, of metals, reduction of processes of sedimentation, self-cleaning and others in the Dniester river.

Previously, the assessment of solids flow was one of the fundamental criteria for appreciating the status of the river hydrographic basin and the river itself. Due to their sorbent nature, the suspended substances have the role of filters for aquatic ecosystems. Self-cleaning processes, buffer capacity of aquatic ecosystems, intensity of production and destruction processes, secondary pollution of ecosystems and formation of bottom sediments depend, to a large extent, on the adsorption potential, composition and structure of suspensions and bottom sediments.

Currently, an imbalance of “adsorption-sedimentation-desorption” processes is observed in the Dniester ecosystems, which, in turn, determines the processes of self-cleaning and those of secondary pollution in aquatic ecosystems. Unfortunately, these measurements are not included in any normative act, which regulate the assessment of the status of aquatic ecosystems.

Decrease of the amount of solid suspensions in the Dniester is directly caused by the operation of the Dniester hydropower plants. If until the construction of the Dubasari dam the flow of suspensions in the lower Dniester oscillated between 4000 and 5500 tons/year, then after its construction it decreased to 2600-2800 tons/year. Flow of suspension was only of 700 tons/year after the commissioning of the Novodnestrovsc hydropower plant in 1983. It decreased to 267-403 tons/year in 1986-1987 and reached only 50-70 tons/year in 2015-2019, being tens of times lower than before the commissioning of DHPC (Zubcov et al., 2019a).

Analysis of the long-term results of the dynamics of the content and flow of suspended substances indicates that such a dynamic is typical for stagnant water bodies, but not for lotic ecosystems. In this case, there is practically no seasonal dynamics and no relationship between the water flow and the physico-chemical parameters.

The adsorption capacity of the Dniester water for allogenic chemicals is close to zero, hence the sharp decline in self-cleaning processes and the increased role of secondary pollution for the river. These factors are also fundamental in changing river hydrobiocenoses, reducing the buffer capacity of the ecosystem and the tolerance of aquatic organisms.

Thus, the decrease of the amount of solids in river suspensions causes:

- decrease of the processes of absorption and sedimentation of chemicals (these processes are dominant in the migration and cycle of chemicals, in increasing the self-cleaning and reducing the secondary pollution of the ecosystem);

- modification of the structure and composition of bottom sediments – already many sectors of the river with sandy bottom are replaced by grey or grey-black muds, which are not characteristic for rivers, but are more characteristic for stagnant waters and swamps;
- increase of transparency – leads to the abundant development of aquatic higher plants and influences the gas regime and the structure of the river hydrobiocenosis.

Analysis of the quantity and composition of at least the mineral and organic components in suspensions is also important for evaluating the changes of lotic ecosystems under the influence of hydropower systems and decreasing the amount of suspensions downstream of dams more than twice can serve as an indicator of fundamental change of the running water body.

Ratio between the dynamics of the suspension content and the dynamics of oxygen concentration in the waters of the Prut River is classic for running aquatic ecosystems in the geographical area of Moldova, when increasing the content of suspensions causes the decrease in dissolved oxygen *(Fig. 1.8)*. In

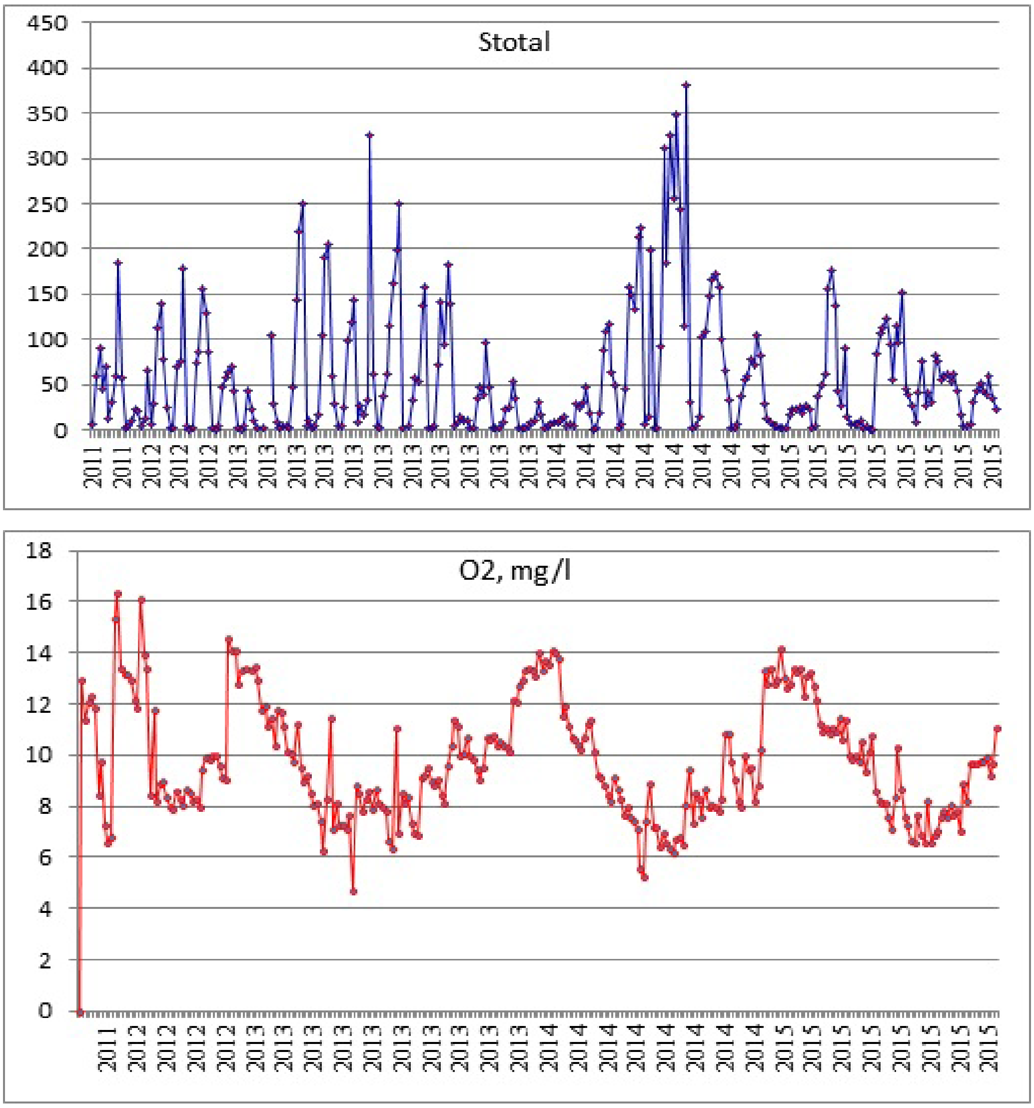
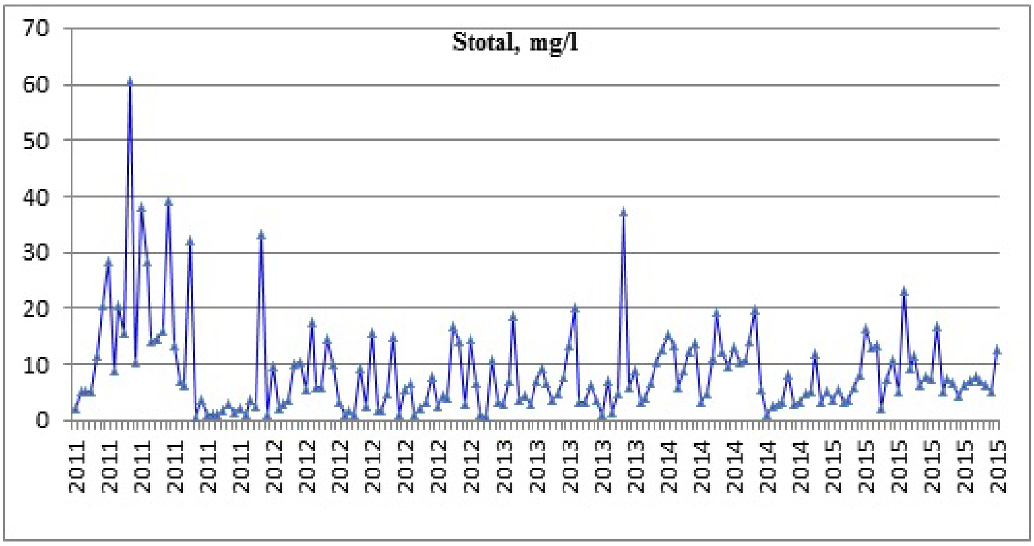


Figure 1.8. Dynamics of suspensions (Stotal, mg/l) and of dissolved oxygen (O_2, mg/l) in the Prut waters (Zubcov et al., 2020)



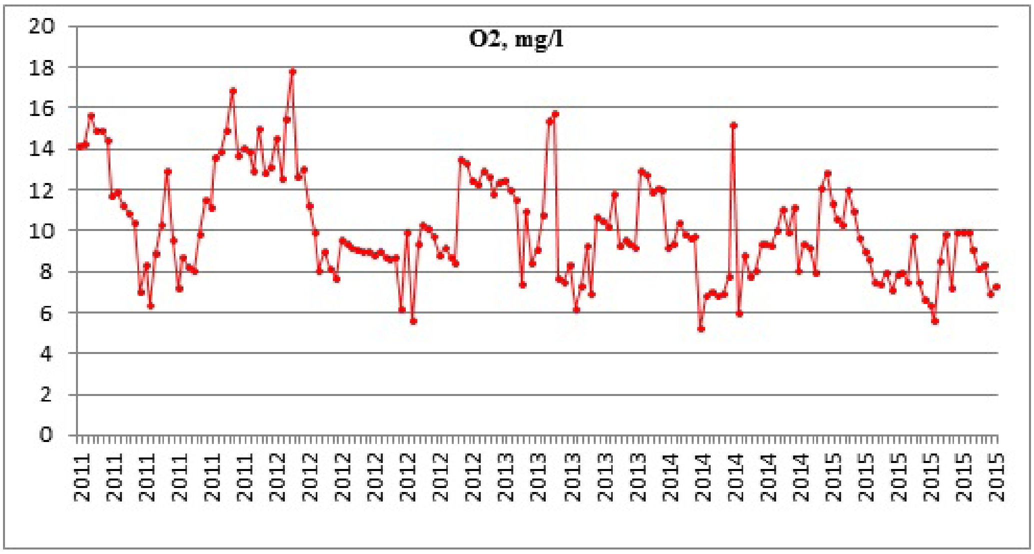


Figure 1.9. Dynamics of suspensions (Stotal, mg/l) and of dissolved oxygen (O_2, mg/l) in the Dniester waters (Zubcov et al., 2020)

the Dniester waters, unfortunately, no longer exists such a correlation *(Fig. 1.9)*.

Both rivers are located in the same physical-geographical area, spring in the same region of the Carpathian Mountains. The fact that in the Prut river the properties of a running ecosystem have been preserved, but the Dniester river is transformed into an ecosystem with stagnant water allows ascertaining the impact of the operation of DHPC and the need to solve this problem between two neighbouring countries, through a balanced management, with the aim to preserve the ecosystems of the Dniester downstream of the Naslavcea dam.

Gas regime, as well as the biochemical and chemical oxygen demand depend a lot on the thermal regime and the content of suspensions (Jurminskaia et al., 2020). At Naslavcea station, downstream of the dam, the tendency of decrease of the water saturation with dissolved oxygen is obvious – up to 58–62%. Perished hydrobionts have been repeatedly observed in places with intense macrophyte development. Extremely low level of oxygen (40% of saturation) and the presence of hydrogen sulphide (H_2S) in the water layers is recorded at Unguri station. In the past, such cases were not observed in the Dniester even in the areas of wastewater discharge.

Modification of the thermal regime (in spring and autumn the water temperature is with 5-7 °C higher, but in summer – much lower *(Fig. 1.3)* and not exceeds 16 °C in the Naslavcea-Unguri sector even at air temperatures of 37 °C) affects not only the gas regime, but also the production-destruction processes, the production potential of the hydrobiont communities. It is established that up to 50-60% of the females of valuable fish species had lost their reproductive capacity under modified living conditions (Bulat, 2017).

The fact that the mineralization of water in the river in spring has become higher than in the summer-autumn period *(Fig. 1.10)* shows that irreversible and unpredictable processes take place in the Dniester basin both for the functioning of the aquatic ecosystem and the entire river basin, what can cause its intensive desertification (Zubcov et al., 2019b). An inverse correlation is no longer observed between the mineralization indicators and the water flow and level in the Dniester river. In recent years, the highest values of mineralization are recorded in spring (Zubcov et al., 2019b).

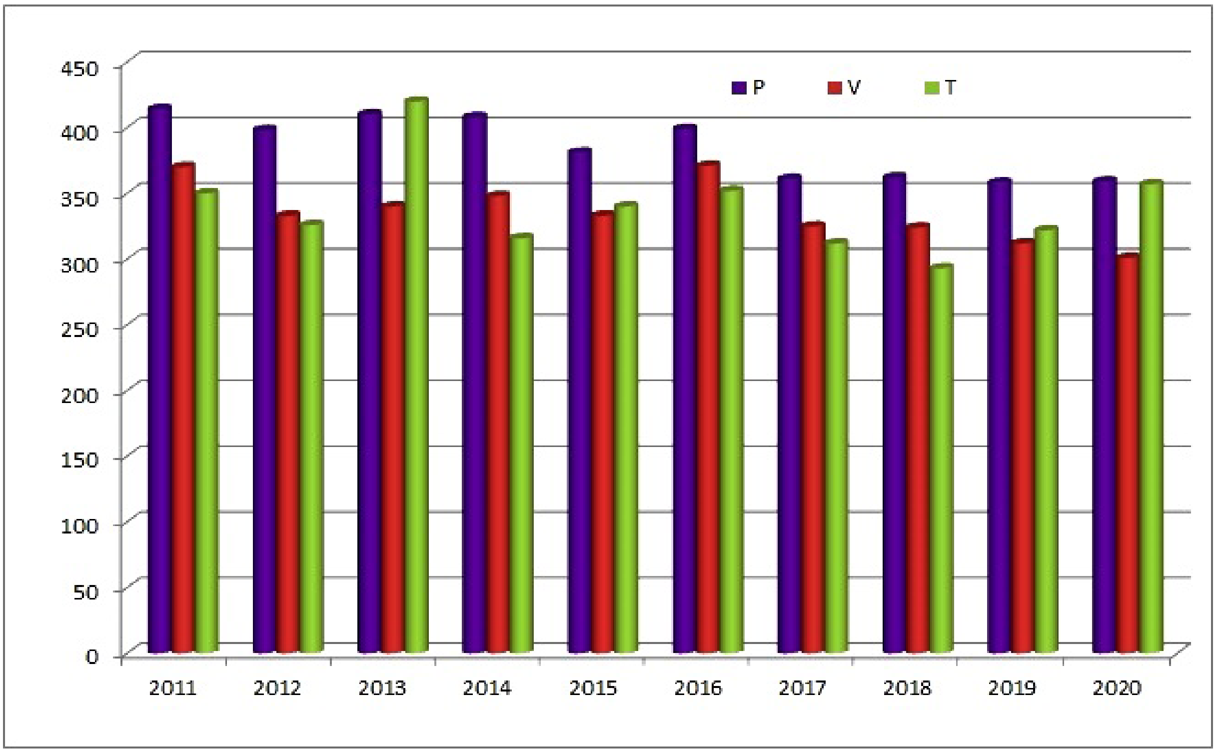


Figure 1.10. Dynamics of mineralization of the Dniester water in 2011-2020, mg/l (P – spring, V- summer, T – autumn)

The ratio between the main ions changes – it was established the replacement of calcium ions with potassium ions, without increasing the total mineralization, this phenomenon indicating the metamorphosis of the chemical composition, especially the ratio between the main ions – components of mineralization in the Dniester waters. Metamorphose of the Dniester river water type testifies that the water flow in the middle and lower sector of the river is mainly formed by local sources (tributaries and groundwater). Nowadays, when the volume of water has an obvious tendency to decrease downstream of DHPC, these processes can determine the occurrence of desertification processes in the river basin, especially in its lower part.

These changes affect the biodiversity, population and production processes of hydrobiocenoses. Replacement of rheophilous species with limnophilous ones, the spread of invasive hydrobiont species are observed (Lebedenco et al., 2021; Munjiu, Andreev, 2021). There are obvious changes in the ichthyocenoses of the middle Dniester and of Dubasari reservoir, where small and economically non-valuable species dominate completely (Bulat et al., 2020).

The surface covered by macrophytes until the construction of DHPC was of 0.7-1%, in the 1980s – 10-15%, in recent years – about 85% of waters *(Fig. 1.11)*. On the Dniester river downstream of the Naslavcea dam, rheophilous plant species are replaced by those characteristic for swamps and stagnant waters: *Myriophillum spicatum, Elodea canadensis, Ceratophyllum demersum, Potamogeton lucens, P. crispus, P. pectinatus, Najas marina, Salvinia natans, Polygonum amphibium, P.submersum* etc. The increase of abundance of planktonic algae belonging to *Cyanophyta (Aphanizomenon flos-aquae, Oscillatoria lacustris, Microcistis aeruginosa), Pyrrophyta (Ceratium hirundinella)* and *Euglenophyta (Euglena polymorpha, Trachelomonas hispida)*, which are more characteristic for stagnant waters is recorded (Zubcov et al., 2019b).

Figure 1.11. Macrophytes in the middle Dniester. Photo: Zubcov Elena

1.4. Conclusions and recommendations

Hydrological regime of the Dniester downstream of DHPC is distinguished by sudden daily fluctuations and lowering of the water level until the exposure of the river bottom, by the imbalance of thermal and gas regime. Volume of water flow has an obvious tendency to decrease downstream of Naslavcea.

The decrease of the content of suspensions of mountainous origin caused the intensification of the swamping or limnization processes of the Dniester along the entire course downstream of the Naslavcea dam, caused by the permanent lack of release of water with a natural speed and in the necessary volume downstream HPP-2. This modified the cycle and migration processes of chemicals in the Dniester river and the intensity of their migration in the river hydrographic basin.

The anthropic impact and, first of all, the operation of the cascade of reservoirs built in the last 20 years on the Dniester river has caused significant changes in the hydrological and hydrochemical regime, degree of eutrophication, organic pollution etc., which have considerably influenced the state of biodiversity and the quantitative structure of the main hydrobiont communities. In the last years, the communities of rheophilous hydrobionts have been replaced by limnophilous hydrobionts.

Complex multi-annual investigations allow us to propose the exclusion of hydropower complexes, in particular, those of pumped storage from the list of so-called "green enterprises", because they destroy the functioning of river ecosystems. Construction of pumped storage hydroelectric power plants (PSHPP) on large rivers, which are a source of drinking water and are used for fish farming, should be banned, as they destroy all living things in running water and damage the functioning of lotic aquatic ecosystems.

We propose the following indicators for assessing the impact of hydropower complexes and climate change on running aquatic ecosystems:

- hydrological (water discharge, speed, temperature in river ecosystems, quantity, composition and distribution of suspensions and alluvium, hydromorphological modifications of the hydrographic basin, quantitative assessment of the river waters originated from atmospheric precipitation, including snowmelt in mountains, and from groundwater, to prevent the drainage of the hydrographic basin, especially downstream of the HPP dams);
- hydrochemical (gaseous regime (O_2, CO_2, CODMn, CODCr, BOD), the ratio between the main ions and their correlation with the hydrological parameters, the processes of migration of the chemical substances in the water-suspensions-silt system);
- hydrobiological (indicators of biodiversity, number and productivity of planktonic and benthic organisms (bacteria, algae, invertebrates), ichthyofauna status, their reproductive potential, biological pollution);
- ecotoxicological and of the ecosystem functioning (level of tolerance of hydrobionts, buffer potential of the ecosystem, its trophicity and saprobity, level of eutrophication, intensity of self-cleaning and secondary pollution processes, production-destruction processes and reproduction of aquatic organisms, including ichthyofauna).

These indicators should also base the assessment of the impact and, conversely, of the socio-economic benefits of HPPs.

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2 Chapter HYDROLOGIC MONITORING OF RIVER ECOSYSTEMS

Onishchenko Eduard, Matygin Alexander
Hydrometeorological Center of Black and Azov Seas

The need to organize a complete monitoring of certain hydrological parameters of river systems in conditions of anthropogenic impact and climate change is determined by the environmental, recreational, biological and water management aspects, which are foreseen in both EU directives and national regulations (Instructions for hydrometeorological stations and posts, 1978; Methodical recommendations on hydrometeorological monitoring of surface water masses of the category "Rivers", 2019; Monitoring Protocol for assessment of the impact of hydropower on river ecosystem functioning, 2020; Regulation on environmental quality requirements for surface waters, 2013).

Observations should reflect all major phases of the hydrological regime. Volume of observations should be sufficient to determine the statistical relationships between the water level and water discharge and such parameters as turbidity, salinity, temperature, transparency, colour (water chromaticity), etc.

Periodic daily observations on water level

Standard hydrological observations are carried out at hydrological gauging stations (g/s) twice a day – at 08:00 and 20:00. There are Mogilev-Podolskiy, Bender, Dubasari and Mayaki gauging stations on the Dniester River, downstream of the Dnestrovsk HPP-1 dam, where periodic standard observations are carried out, such as: level (water discharge) measurements, flow of suspended matters (turbidity), water and air temperature registration, and accounting for precipitation.

During spring high waters and floods, when the water level rises sharply, observations of the level are made more frequently than in the case of standard hydrological observations, at regular intervals: after each 1, 2 or 4 hours (Instructions for hydrometeorological stations and posts, 1978).

At the automatic measuring stations, the observations are made continuously and data are transmitted via the internet channel with 1-hour discreteness. These measuring stations are located downstream of HPP-2 on the Dniester river within the borders of the Republic of Moldova: Naslavcea – 658 km, Soroca – 545 km, Vadul lui Voda – 286 km from the confluence of the river with the Dniester estuary. The data can be found on the website http://nistru.meteo.gov.ua/en/autoposts_operational_data/.

The basic hydrological characteristics of the river runoff are:

- water discharge (Q), is usually expressed in m^3/s or in L/s – for small streams;
- flow volume (W) – in km^3 or m^3;
- runoff modulus (q), $L/(s \cdot km^2)$ for large rivers or $m^3/(s \cdot km^2)$ – for small rivers and streams with a small catchment area;
- drainage layer (h), mm;
- water level (H), cm.

Time data series of these hydrological characteristics are usually used to describe the features of the river runoff dynamics for different time periods (intervals). Hydrometric data series of sufficient duration provide the ability to determine the rated hydrological characteristics using analytical and empirical exceedance probability distribution functions (probability curves). The main regularities of the of hydrological quantities distribution have been identified on the basis of numerous studies in the field of hydrological calculations. The construction of an empirical probability curve for the hydrological characteristic precedes the calculation of the probability of each member of the ranked series.

To assess hydrological data for statistical homogeneity, the following criteria are mainly used:

- criteria for sharply deviating extreme values in the empirical distribution (Smirnov-Grubbs and Dixon criteria);
- criteria for the homogeneity of sample variances (Fisher's test);
- a criterion for checking the significance of the difference in the mean values of two data samples (Student's test).

1. The empirical probability of exceeding the rated hydrological characteristic Pm (%) is determined by the formula:

$$P_{m,\%} = \frac{m}{n+1} 100, \qquad (1)$$

where: *m* is the ordinal number of the members of a series of hydrological characteristics, arranged in descending order; *n* is the total number of members of the data series.

Probability curve constructed according to formula (1) makes it possible to determine the value of the rated hydrological characteristic of the required probability of exceeding. In particular, the value of the level (H) with a 1%-probability indicates the possibility of its occurrence once every 100 years.

Empirical curves of the annual values of distribution of the probabilities can be generated with special probability papers or the empirical curves can be generated using special modern software. The type of probability papers is selected in accordance with the accepted (selected) analytical probability distribution function and the obtained ratio of the asymmetry coefficient Cs to the coefficient of variation Cv.

If a series of hydrometric data turns out to be heterogeneous (which, first of all, may indicate different genetic conditions for the formation of runoff at different time intervals), it is allowed to use truncated and composite curves of the distribution of annual exceedance probabilities.

2. The parameters of the analytical distribution curves are: 1) the average long-term value of the calculated hydrological characteristic, for example, discharge; 2) coefficient of variation Cv; 3) the ratio of the asymmetry coefficient to the coefficient of variation Cs/Cv, which are established from the hydrometric series of observations of the considered calculated hydrological characteristic by the method of maximum likelihood or the method of moments.

The coefficient of variation Cv and the coefficient of asymmetry Cs for the three-parameter Kritsky-Menkel gamma distribution can be determined by the method of maximum likelihood, depending on the statistics λ_2 and λ_3, which are calculated by the formulas:

$$\lambda_2 = \left(\sum_{i=1}^{n} \lg k_i\right)/(n-1), \qquad (2)$$

$$\lambda_3 = \left(\sum_{i=1}^{n} k_i \lg k_i\right)/(n-1), \qquad (3)$$

where: k_i is the modular coefficient of the considered hydrological characteristic (in this example – *Q*), determined from the ratio

$$k_i = \frac{Q_i}{\overline{Q}}, \qquad (4)$$

where: *Qi* – the average annual water flow rate for a separate *i*-th year; $\overline{Q}$ – the arithmetic mean of water discharge over the observation period:

$$\overline{Q} = \sum_{i=1}^{n} Q_i / n, \qquad (5)$$

According to the obtained values of statistics λ_2 and λ_3, the coefficients of variation and asymmetry are determined according to special nomograms.

The calculated coefficients of variation can also be determined by the method of moments using the formulas:

$$\tilde{C}_v = \sqrt{\frac{\sum_{i=1}^{n}(k_i - 1)^2}{n-1}}; \quad (6)$$

$$\tilde{C}_s = \frac{\left[n\sum_{i=1}^{n}(k_i - 1)^3\right]}{\left[\tilde{C}_v^3 (n-1)(n-2)\right]}, \quad (7)$$

The coefficient of variation (variability) $\tilde{C}_v$ (6) characterizes the intensity of fluctuations of the rated hydrological characteristic relative to the average value. The more $\tilde{C}_v$, the greater the amplitude of the oscillation. $\tilde{C}_s$ (7) characterizes the degree of asymmetry of the distribution. For example, if positive deviations from the average long-term value (high-water years) repeat less often than negative deviations (low-water years), but at the same time have a more significant range, then the asymmetry is considered positive.

Monitoring of the waters of rivers under the influence of hydropower constructions

The current network of observations on the hydrological regime and water quality indicators, as well as the system for conducting observations on certain water bodies cannot fully ensure the obtaining of accurate data, which are later processed and analysed. The reason is the rare and irregular collection of samples, as well as the insufficient number of observation points. In principle, this problem can only be solved by organizing adequate monitoring.

The annual hydrograph of the monthly average values of the flow, based on the daily average values, allows to:

- estimate the time of occurrence and the quantitative characteristics of the highest and lowest values of flow during the high water, low water, floods;
- compare the modification of the annual flow before and after the regulation of watercourse, in order to determine the efficiency of the redistribution of the flow over the seasons;
- evaluate the efficiency of the ecological releases of the Dniester Hydropower Complex, which are meant to maintain the ecological conditions at a favourable level for the life of aquatic organisms.

In the case of detailed studies on phenomena of shorter duration (floods, melting snow, sudden withdrawing large volumes of water from the river, etc.), it is necessary to consider the hydrograph based on the average values for a 10-5 days periods or for 24 hours.

On the one hand, dams of hydropower plants (HPP) regulate the volume of flow, preventing catastrophic floods, and on the other hand, they are obliged to ensure the normal life of the river ecosystem along its entire length. In this respect, observations of the level (flow) in important stretches of the river should be analysed with discretion from 1 to 24 hours, depending on the dynamics of level changes caused by a flood wave or ecological release.

Indicator „impact of daily flow changes" describe the impact of artificial constructions (dams) or water capture on the diversity of types of water flow (Methodical recommendations on hydrometeorological monitoring of surface water masses of the category "Rivers", 2019).

Frequent daily variations of flow usually occur downstream of the HPP, where turbine operation changes frequently (often daily). Dramatic increases in water levels can result from:

- water releases, which may increase or decrease the level by more than 5 cm/hour;
- HPP operation (daily changes in releases), when changes may occur gradually and the water level rises or falls at a speed of less than 5 cm/hour.

Tables 2.1a, b and the explanatory figures demonstrate the quantitative and qualitative estimates of the daily fluctuations of water flow and level, which characterize the river flow dynamics.

In addition to standard observations on water levels, in order to describe the impact of hydropower constructions and climate change on the ecosystem of the Dniester river, the following characteristics of water must be included in the monitoring program: water temperature, turbidity, transparency, colour, speed, mineralization. This information, in addition to the description of the anthropogenic and climatic influence on the hydrological regime, will allow coordinating of the volume and period of water flow downstream of the dam, based on weather conditions in the hydrographic basin of the middle and lower Dniester. Thus, it will help to ensure the functionality of the river ecosystem, creating favourable conditions for the development of hydrobionts and preserving the diversity of river hydrobiocenosis.

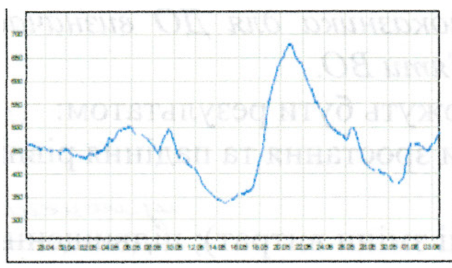
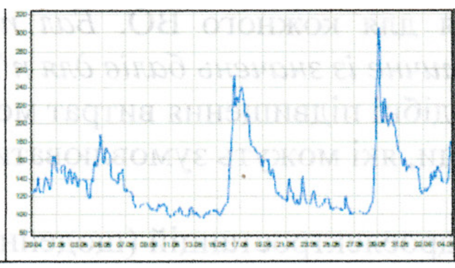
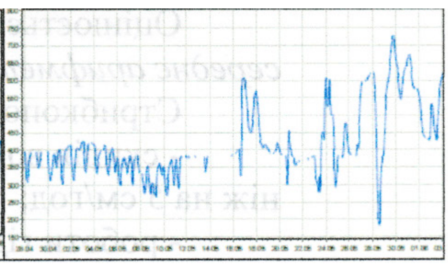
River water temperature

Observations on water temperature are made daily at standard hours. Anthropogenic thermal pollution of water and climate change significantly affect the oxygen regime and the intensity of self-purification processes in the middle and lower Dniester (Zubcov, 2007; Zubcov, 2012). As a result, the natural balance of the Dniester is disturbed, often ir-

Table 2.1a. Estimation of the indicator "Impact of daily flow changes". Quantitative evaluations

1	2	3	4	5
There are no daily flow disturbances or changes, or the impact of hydrotechnical constructions is manifested by flow changes lasting <2% days per year (7 days), leading to at least a double increase or decrease in flow, or changes in water level	Impact of hydrotechnical constructions is exhibited by changes in flow, which last from 2% to 5% of time during a year, leading to at least a double increase or decrease in flow or changes in water level	Impact of hydrotechnical constructions is exhibited by changes in flow, which last from 5% to 20% of time during a year, leading to at least a double increase or decrease in flow or changes in water level	Impact of hydrotechnical constructions is exhibited by changes in flow, which last from 20% to 40% of time during a year, leading to at least a double increase or decrease in flow or changes in water level	Impact of hydrotechnical constructions is exhibited by changes in flow, which last >40% of time during a year, leading to at least a double increase or decrease in flow or changes in water level
> 5 cm/hour	>5 cm/hour	>5 cm/hour	>5 cm/hour	>5 cm/hour

Table 2.1b. Qualitative evaluations and the appearance of the corresponding graphs

1	2	3
No sudden fluctuations in flow are observed (<5% of the time during the year)	Sudden fluctuations in flow are rarely observed (from 5% to 20% of the time during the year)	Sudden fluctuations in flow are often observed (20% of the time during the year)
		

reversibly, and special ecological conditions are formed, which negatively affect, in general, the processes of hydrobiocenosis functioning (Zubcov et al., 2020; Zubcov et al., 2019b; Jurminskaia et al., 2020).

The temperature of river waters, which are not affected by human activity, depends generally on natural conditions (Analysis of the impact of the reservoirs of the Dniester HPPs on the state of the Dniester River, 2019). However, in the immediate proximity of the dam, where the water is released, several temperature observation points should be located, with the purpose of monitoring its change along the riverbed. As reason is the fact that in summer the water released downstream of the HPP-1 dam has a temperature of up to 8 °C lower, and in winter – of up to 6 °C higher than the natural seasonal water temperature in this river stretch (Analysis of the impact of the reservoirs of the Dniester HPPs on the state of the Dniester River, 2019).

To measure the water temperature in the surface layer, different thermometers are used, the classic being the TM-10 thermometer in an OT-51 metal frame. The thermometer is immersed to a depth of 0.4 – 0.5 m and maintained for 5-10 min. The water temperature is measured with an accuracy of 0.1 °C.

To describe the daily change in at least water temperature, content of dissolved oxygen, values of pH, turbidity, water level and speed and other physico-chemical parameters, it is necessary to install the equipment downstream of the HPP-2 dam, which could provide information on-line, with a discretion from 15 min to 4-6 hours.

This equipment would provide the opportunity to highlight the processes that take place in the Dniester river under the influence of the Dniester Hydropower Complex, the given information being necessary for the preservation and sustainable capitalization of the resources of the main transboundary water streams, of vital value, for Moldova and Ukraine.

It should also be mentioned that the Hydrometeorological Center for Black and Azov Seas (5th project beneficiary), in the process of implementing the BSB165 project, strengthened the technical endowment of the institution for hydrometeorological investigations, including hydrological ones in Ukraine.

Turbidity of river waters

Turbidity of water is due to the presence in it of different types of mechanical impurities in suspension: particles of sand, clay, mud, organic and inorganic substances in suspension, plankton and various microorganisms. The size of particle that determines the turbidity of water is 0.004-1.0 mm.

At hydrological stations the single standard samples for determining the turbidity of water are usually collected daily during hydrological observations. In case of sudden daily fluctuations of the water level, it is recommended to make observations several times during 24 hours.

The optimal time for observing the turbidity in periods of high water and floods is chosen based on the study of the daily dynamics of turbidity, which require frequent observations. During the high water, at least 8-10 measurements are performed, preferably evenly distributed according to the amplitude of the water level and the phases of increase and decrease of the high water. During rain floods, at each peak of the water level a turbidity measurement is performed at both increase and decrease of the level (Instructions for hydrometeorological stations and posts, 1978).

Regulation of the Dniester flow led to a sudden decrease of the amount of suspended particles (approximately by 10 times) downstream of the dam, as well as to a change in the spatial and temporal structure of turbidity along the river. Consequently, the intensity of water self-cleaning processes was reduced, the composition of suspended substances and bottom sediments was negatively influenced, what led to excessive growth of macrophytes along the river downstream of the Dniester Hydropower Complex (Zubcov, 2012; Zubcov et al., 2019a). Therefore, for optimal monitoring of turbidity in the regulated sections of the river, the rules for measuring turbidity at the extreme values of rising and falling water levels must be followed (Instructions for hydrometeorological stations and posts, 1978).

When determining the turbidity in the laboratory, the amount of suspended particles (flow of suspended solids) is measured by filtering a known volume of a water sample through filters with a pore diameter not exceeding 45 μm (Monitoring Protocol for assessment of the impact of hydropower on river ecosystem functioning, 2020; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020). The measurement result is expressed in mg/dm^3. In the case of more detailed classical hydrochemical investigations, the amount of mineral and organic suspensions shall be determined.

Currently, instruments called nephelometers are used to determine the turbidity, including *in situ*. The official measurement unit of turbidity is Formazin Nephelometric Unit (FNU). Numerically, the turbidity expressed in FNU differs from that measured in Kaolin units – mg/dm^3 (1 FNU = 0.58 mg/dm^3 of Kaolin) (EN ISO 7027-1-2016).

Transparency

Transparency of natural waters is due to their colour and turbidity, i.e. of their content of various coloured and suspended organic and mineral substances. The simplest methods for determining the optical properties of water include determining the depth of disappearance of the visibility of a white disk (Secchi disk) and appreciating the colour of water according to the colour scale.

When determining the transparency directly in the water body (*in situ*), a white metal disc with a diameter of 300 mm is used. Observations on the transparency of water with the help of the white disk are made from the shadow side, while the direct rays of the sun should illuminate the disk itself. Transparency is determined as follows: the disk is slowly lowered into the water and, at the limit of visibility, the depth of its disappearance is recorded, then, lowering it by 0.5 – 1.0 m, it is raised and the depth of its visibility is noted again. The average value of two depths is the measure of transparency, i.e. the height of the water column in cm, at which the white disk is still observed.

The transparency in measured in laboratory by the Snellen method, the essence of which is to read the standard font from above through a column of water. A standard font is placed under a cylinder of 60 cm high and 3-3.5 cm in diameter at a distance of 4 cm from the bottom, the analysed sample is poured into the cylinder. The method for the quantitative determination of transparency is based on the determination of the height of the water column, at which it is still possible to visually distinguish (read) a black font with a height of 3.5 mm and a line width of 0.35 mm on a white background. The method used is unified and corresponds to ISO 7027 (EN ISO 7027-1-2016 Water quality; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020). The results are expressed in centimetres *(Tab. 2.2)*.

Table 2.2 Characteristic of water according to transparency

Transparency	Measure unit, cm
Transparent	More than 30
Little turbid	More than 25 and up to 30
Medium turbidity	More than 20 and up to 25
Turbid	More than 10 and up to 20
Very turbid	Less than 10

Water colour

Colour of natural waters is mainly due to the presence of humic substances, iron compounds and synthetic colourants. The amount of these substances in the water depends on the geological conditions, the characteristics of soils in the river basin, etc. Industrial wastewater can also create an intense water colour.

There is a standard technique for visual comparison of the colour of water sample with the colour of standard solutions, the quantitative result being expressed in Hazen units of colour (degrees) according to the platinum-cobalt scale (ISO 2211:1973 Liquid chemical products). This standard is applied to pure, slightly coloured liquids, the colour of which corresponds to the brown-yellow colour of the platinum-cobalt scale. The technique requires a rather complicated preparation of the reagents and the availability of the appropriate equipment. Colour of water sample is observed according to the generally accepted scale, which characterizes the colour of water. Lately, these investigations are performed by spectrophotometric methods. Likewise, *in situ* investigations are performed by using spectrophotometry-based equipment, by comparing water samples with a number of standards.

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3 Chapter

MONITORING OF PHYTOPLANKTON AND IDENTIFICATION OF ITS ROLE IN THE FUNCTIONING OF AQUATIC ECOSYSTEMS

Ungureanu Laurenția[1], Grandova Maria[2], Kovalyshyna Svetlana[2], Tumanova Daria[1], Ungureanu Grigore[1]

[1]Institute of Zoology, [2] Ukrainian Scientific Center of Ecology of the Sea

Study of regularities of the functioning of aquatic ecosystems, their productivity and water quality are of particular importance in conditions of eutrophication and continuous pollution. Interpretation of ecological processes and forecast of changes in aquatic ecosystems are impossible without multilateral investigations of formation and restructuring mechanisms of communities of planktonic algae – basic producers of organic matter and important factors of natural water quality. Establishment of the regularities of phytoplankton formation and functioning, and putting into evidence of the influence of environmental factors on this process, in the conditions of intensifying anthropogenic pressure, contribute to the elaboration of the theory of biological productivity, methods of directing the functioning and sustainable capitalization of aquatic ecosystems. In time, the study methods were continuously improved, which allowed the algologists to obtain internationally competitive results (Ungureanu, 2011).

The complexity of phytoplankton research derives from the multitude of addressed issues regarding the functioning of planktonic algae communities in aquatic ecosystems of different types (Ungureanu et al., 2018; 2020; 2020a).

In terms of monitoring and evaluating the influence of hydropower and climate change on algae communities, it is important to divide algae species into two groups – characteristic for running water and for stagnant water. The relationship between these groups or the dominance of one of these groups denotes changes caused by river dams, modification in the ecological status of aquatic ecosystems according to hydrological characteristics.

Limnophilous algae species:

Microcystis aeruginosa f. *flos-aquae* (Wittr.) Elenc., *Anabaena spiroides* f. *Woronichiniana* Elenk., *Anabaena verrucosa* B.Peters. f. *verrucosa*, *Oscillatoria granulata* Gardner f. *granulata*, *Navicula lacustris* Greg., *Navicula hungarica* Grun., *Navicula placentula* (Ehr.) Grun. f. *placentula*, *Navicula gastrum* Ehr. var. *gastrum*, *Gyrosigma fasciola* Ehr., *Amphora venata* Kutz. var. *venata*, *Cymbella amphicephala* Nag. var. *amphicephala*, *Gomphonema parvulum* (Kutz.) Grun. var. *parvulum*, *Gomphonema lanceolatum* Ehr. var. *lanceolatum*, *Gomphonema gracile* Ehr. var. *gracile*, *Gomphonema ventricosum* Greg. f. *ventricosum*, *Nitzschia apiculata* (Greg.) Grun., *Nitzschia constricta* (Greg.) Grun. f. *constricta*, *Nitzschia amphibia* Grun var.*amphibia*, *Nitzschia hantzschiana* Rabenh., *Surirella turgida* W.Sm. var.*turgida*, *Surirella robusta* var. *splendida* Ehr., *Goniochloris spinosa* Pasch., *Glenodinium berolinense* (Lemm.) Lind. var. *berolinense*, *Trachelomonas armata* (Ehr.)Stein var. *armata*, *Trachelomonas Playf*. var. *scabra*, *Strom-*

bomonas planctonica (Wolosz.) Popova var. *planctonica*, *Euglena oblonga* Schmitz, *Euglena ehrenbergii* Klebs var. *ehrenbergii*, *Lepocinclis ovum* var. *major* (Hub.-Pestol.) Conr., *Lepocinclis teres* (Schmitz.) France, *Chlamydomonas globosa* Snow., *Chlamydomonas reinhardtii* Dang., *Carteria klebsii* (Dang.) France, *Eudorina elegans* Ehr., *Volvox aureus* Ehr., *Ankyra ancora* F.*issajevii* (Kissel), *Pediastrum borianum* var. *longicorne* Reinsch., *Pediastrum duplex* var. *reticulatum* Lagerh., *Dictyosphaerium chlorelloides* (Naum.) Komarek et Perman, *Scenedesmus sempervirens* Chodat., *Closterium braunii* Reinsh., *Closterium acerosum* f. *elongatum* (Breb.)Kossinsk, *Closterium venus* Kutz. var.*venus*, *Cosmarium undulatum* Corda.

Rheophilous algae species:

Romeria elegans (Wolosz.) Koczw, *Rhizosolenia longisseta* Zacharias, *Navicula cuspidata* f. *primigena* Dipp., *Pinnularia viridis* (Nitzsch.) Ehr., *Surirella robusta* Ehr. var. *robusta*, *Tetraedriella spinigera* Skuja, *Tetraplektron acutum* f. *laevis* (Bourr.) Ded. Stscheg., *Ophiocytium lagerheimii* Lemm., *Trachelomonas incerta* var. *punctata* Lemm., *Strombomonas tambowica* (Swir.) Defl., *Strombomonas gibberosa* (Playf.) Defl. var. *gibberosa*, *Euglena fenestrata* Elenk., *Euglena tripteris* var. *major* Swir., *Monomorphina splendens* (Pochm.) Popova, *Desmatractum indutum* (Geitl.) Pasch., *Ankyra ancora* F. *spinosa* (Korsch.)Fott, *Closterium lanceolatum* Kutz., *Staurastrum crenulatum* (Nag.) Delp.

Investigation of phytoplankton and the evaluation of the primary production of phytoplankton and the destruction of organic matter are the basis for determining the level of trophicity, eutrophication of water bodies, the knowledge of which is necessary for the evaluation of self-purification and secondary pollution processes (Ungureanu et al., 2019; 2020).

The main principles of sampling and further processing of phytoplankton samples are expounded in appropriate guides (Hötzel, Croome, 1999), including those elaborated recently in correspondence with EU requirements (Hydrochemical and hydrobiological sampling guidance, 2015; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020).

The samples of phytoplankton should be collected once a season. For additional monitoring of short-time effects of hydropower constructions, the samples should be collected before the great releases, during the release and weekly during the next month after release.

Sampling sites should be chosen according to common principles, based on the set objectives: upstream and downstream of a point source such as a sewage treatment plant, weir pool or tributary; upstream and downstream of a source of major ecological impact, such as a reservoir built with hydropower purpose; and at certain intervals along the studied river stretch, in order to explore longitudinal distribution of phytoplankton.

It is preferable to sample from the main current by boat or from a bridge. As usual, the samples are taken from surface and bottom layer (with the help of Niskin bottle); if the depth is less than 3 m, the samples may be taken only from surface layer. In this case, sample is better to take at mid-stream 0.5 m below the surface. If a boat is not available, a sample may be taken from the shore with either a dip stick sampler (more than 3 m long), carrying a glass sample bottle at the end, or entering the river at a distance of at least 3 m from the shore (Hydrochemical and hydrobiological sampling guidance, 2015; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020).

A better statistical result may be achieved by collecting several samples from slightly different spots at one site and counting them to a lower level of precision.

Good practice is simultaneous phytoplankton sampling together with hydrochemical samples for the main factors for phytoplankton development: dissolved inorganic phosphorus, dissolved inorganic nitrogen, oxygen concentration, pH, silicates (Ungureanu et al., 2018; 2020; 2020a).

For chlorophyll-a analysis, a separate sample of 0.5 L to 1 L is taken. A live sample taken with a plankton net (mesh size of 25 µm to 35 µm), in addition to a whole water sample, may aid the identification of the larger species. It is advisable to use a standardized phytoplankton field sampling sheet to ensure all samples and measurements taken in the field are properly recorded in the field. The field sampling sheet will also facilitate sample registration in the laboratory and later data reporting. In addition to taking water samples for phytoplankton and water quality parameters, it is useful to record observations such as water color, smell and scum formation as well as wind direction and strength (Hötzel, Croome, 1999).

For quantitative samples of phytoplankton, 1-2 L of water are fixed with buffered formaldehyde. To make a 20% aqueous solution of formaldehyde (HCHO), mix equal parts of formalin (40% HCHO) and concentrated acetic acid. For fixation, add 100 mL of the water sample to 2 mL of the acidified formaldehyde (the final concentration of HCHO should be 0.4%). Then the samples are allowed to settle for approximately two weeks, and then slowly decanted to the volume of 20-30 ml. The sedimentation is carried out in cylinder of suitable size, allowing six hours for each 1 cm of water column at 20°C (Hötzel, Croome, 1999).

After the appropriate period of sedimentation, the top 90% of volume is carefully siphoned off without disturbing the sedimented algae, the remainder is shaken gently and put into another cylinder with appropriate volume, than the sedimentation is repeated. These two sedimentation give a possibility to obtain the concentrated sample with a volume of 20-30 ml. Before the analysis, the sample is shaken gently and a subsample of appropriate volume is transferred to the counting chamber and allowed to settle before counting.

Structural and functional characteristics of phytoplankton community

The parameters to be investigated: 1) abundance; 2) biomass; 3) taxonomic composition; 4) structure of phytoplankton community: the contribution of top five ranked phyla in the total biomass.

Identification of species and cell counting is carried out under a light microscope Mikmed-5 (600x) in the drop of 0.05 ml and existing identification guides.

The biomass of phytoplankton is calculated by the method of geometric similarity equating shapes of cells to corresponding geometrical shapes and assuming that the cell volume of 1 mm^3 is equal to 1pg. The water quality is estimated based on the parameters of total phytoplankton biomass and the species of algae, which are indicators of saprobity (CSN EN 15204 Water quality – Guidance standard on the enumeration of phytoplankton using inverted microscopy (Utermöhl technique).

The definition of water quality classes of the ecosystem is carried out in accordance with the in force regulations on the environmental quality requirements for surface water quality at the national level (Regulation on environmental quality requirements for surface waters, 2013; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020).

Knowledge of the regularities of the functioning of phytoplankton communities and aquatic biocenoses, in general, contributes substantially to solving a range of problems aimed at the sustainable use of aquatic resources. Phytoplankton is one of the primary producers of aquatic ecosystems. Through their vital activity, planktonic algae contribute to the biological productivity of aquatic ecosystems, regardless of whether its production share is high or low. On the other hand, they indirectly participate in the total biological productivity of aquatic ecosystems, as they are part of animal feed at different trophic levels. Primary production and destruction of organic substances determine the nature of the efficiency of production processes in aquatic ecosystems (Mineeva, 2009; Ungureanu et al., 2020a).

The A/R ratio changes during pollution and self-purification processes. Therefore, it can be used to characterize the level of organic pollution in ecosystems and the composition of the substances that pollute it. The higher the level of penetration into the ecosystem of non-native organic substances is, the more they influence the balance of production-destruction processes.

It has been demonstrated that the degree of illumination of water layers, temperature, transparency, hydrological regime of water influence the development of phytoplankton, the level of primary production and destruction of organic substances in different types of aquatic ecosystems, but many aspects of these phenomena require thorough investigation (Ungureanu et al., 2018; 2020; 2020a).

According to the values of phytoplankton biomass, primary production in the photic layer (A), in the water column (A/m^{-2}) and the ratio of production and destruction processes (A/R), the Lower Dniester can be attributed to the category of eutrophic, periodically mesotrophic ecosystems, and the Dubasari reservoir and the Middle Dniester – to the category of eutrophic, periodically polytrophic ecosystems (Ungureanu et al., 2020a).

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4 Chapter

MONITORING OF PLANKTONIC INVERTEBRATES AND RECOMMENDATIONS FOR ITS IDENTIFICATION

Lebedenco Liubovi[1], Nabokyn Mykhailo[2], Kovalyshyna Svetlana[2],
[1]Institute of Zoology, [2] Ukrainian Scientific Center of Ecology of the Sea

In lotic ecosystems, all processes of development of planktonic invertebrate communities (zooplankton), designed to ensure a structural and functional order, are systematically disrupted by disturbances. Depending on their nature, the changes have a different character. The changes are classified according to the nature of their origin – natural hydrological (deviations of the moment and power of floods, rain floods and droughts, sediment formation, etc.), anthropogenic (changes in channel morphometry, pollutant load, etc.). Concomitantly, these changes may last short time, such as rising of water levels due to heavy rainfall, floods, or pollutant discharges, but they can also be stable when a sudden change in conditions is accompanied by the subsequent maintenance of a new state. Zooplankton community is a dynamic system that, due to its increased sensitivity, filtration type of nutrition, as well its relatively short life cycle, reacts relatively quickly to changes in environmental conditions (habitat). This is expressed in change of the structure and functional indicators (Lebedenco, 2020).

Fluctuations in the hydrological regime throughout the year are reflected in the development of zooplankton communities of the Dniester and Prut rivers. The species composition and quantitative parameters of zooplankton in a flowing ecosystem remain approximately constant over a period of time, so that the sudden appearance or disappearance of some species may indicate changes in water quality. From this point of view, a long-term monitoring program of the structure of zooplankton communities can elucidate aspects related to the ecological status of flowing ecosystems and can distinguish between the normal effects induced by the succession of seasons in the zooplankton community and changes of anthropogenic nature (Lebedenco, 2020; Lebedenco et al., 2021).

For the quantitative and qualitative sampling of zooplankton in the river, the Apstein net is most often used. It is made from mill gas. The recommended mesh size is 100 µm (Aleksandrov et al., 2014, Hydrochemical and hydrobiological sampling guidance, 2015; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020).

Quantitative sampling is carried out by filtering a known amount of water through the planktonic net, according to ISO standards and recently elaborated guides (SM SR ISO 5667-6:2011; SM SR EN 15110:2012; Hydrochemical and hydrobiological sampling guidance, 2015; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020). It is recommended to filter from 50 to 500 liters, however, the specialist carrying out the sampling may change this volume both up and down.

Sampling may be carried out by pouring river water through the net (Tevyashova, 2009). Preferably to use a Patalas bottle, or, in the absence of it, a bucket of a known volume. Another way of sampling is horizontal or vertical pulling of the net in the reservoir. In this case, it is recommended to install a mechanical flow meter on the net for a more accurate measurement of the volume of water that has passed through the net. It is important that in order to avoid the loss of the most active organisms during sampling, the net must be pulled with a speed of at least 1 m/s. When flow speed is sufficiently fast (more than 1 m/s), you may simply hold the net in the river for a certain known time to estimate the volumes of filtered water. In the case of qualitative sampling, an accurate measurement of the filtered water is not required, however, the volume of filtered water should be several times larger than the volumes filtered for quantitative samples.

Before zooplankton sampling, a specialist visually assesses the number and area of all biotopes present at the sampling site. It is recommended to take 1-6 samples from each biotope, depending on the area they occupy. In flowing water bodies, sampling from areas with obvious flowing is mandatory (with water speed of more than 20 cm/s). Sampling is recommended with a frequency of 10-14 days.

Immediately after sampling, the samples are preserved with Lugol's solution or formaldehyde (SM SR ISO 5667-6:2011; SM SR EN 15110:2012; Hydrochemical and hydrobiological sampling guidance, 2015). The concentration of formaldehyde in the final solution shall not exceed 2-4% (i.e., 5-10 ml of 40% formalin per 100 ml of the sample).

In the laboratory, samples are concentrated to a volume of 20-200 ml, depending on the abundance in the sample, and analyzed using modern microscopes and identification guides. To achieve a statistically reliable result, it is advisable to analyze the volume of each sample, which will contain at least 100 specimens of each of the 4-5 main species (Kojova, Melinik, 1978). Taxonomy should be brought in line with the site WoRMS (http://www.marinespecies.org/).

Usually, the counting of zooplankton organisms is performed with the help of the Bogorov chamber, in two or three repetitions, using sophisticated equipment, such as the ZEISS Discovery V8 stereo zoom binocular. The density (N) of planktonic invertebrates is expressed in the number of individuals per 1 m^3 and is an essential parameter in the quantitative characterization of biotic communities in aquatic ecosystems – those retained by the net for zooplankton sampling. Identification of zooplankton species is performed using the Axio Imager A2 microscope (ZEISS), by using identification guides and specialized literature (Hydrochemical and hydrobiological sampling guidance, 2015; Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020).

Zooplankton organisms are identified to the maximum possible lowest systematic category: immature forms of copepods – to the level of suborder (*Cyclopoida, Calanoida* or *Harpacticoida*), adult cladocerans and copepods – the species level, rotifers – up to the genus and, if possible – to the species level. Biomass (B, mg/m^3) of zooplankton communities is calculated by multiplying the density by the average individual masses of each species. The estimation of the investigated aquatic ecosystems and water quality is performed by saprobiological analysis, based on the principles proposed by the saprobic system. Evaluation of water quality classes of the Dniester and Prut rivers on the base of zooplankton communities is carried out according to the limit values presented in the national regulation (Lebedenco, 2020).

Currently, there are used several computerized methods for investigation of plankton-

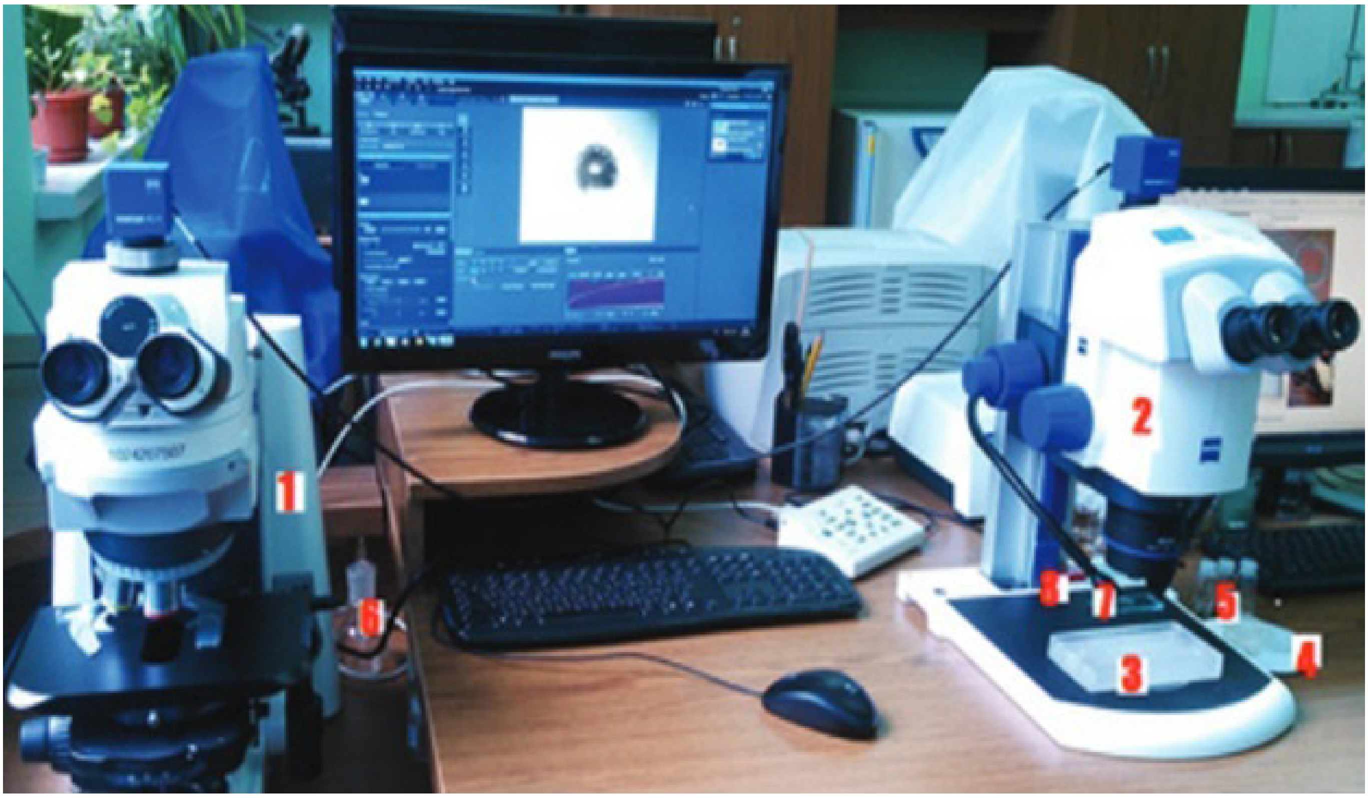

Figure 4.1. 1 – microscope; 2 – binocular; 3 – Bogorov chamber; 4 – Petri dishes; 5 – glass bottles; 6 – vessel with alcohol; 7 – microscope glass slide; 8 – cover glass

Figure 4.2. Computerized microscope Axio Imager A.2 (Zeiss)

ic invertebrates *(Fig. 4.1-4.2)*, which reduce possible deviations due to the human factor (Guidance on the monitoring of water quality and assessment of the ecological status of aquatic ecosystems, 2020).

Structural and functional characteristics of zooplankton community

The main parameters that need to be recorded for planktonic invertebrates:

1) total abundance and biomass of zooplankton;
2) number of species and diversity indicators;
3) Saprobity index;
4) the proportion and ratio of the main systematic groups of zooplankton.

Investigations of changes in community structure and populations of certain species and taxonomic groups require an individual set of data and criteria for each region. However, the following indicators of zooplankton seem to be the most promising in the Dniester water area:

1) % Rotifera;
2) % Copepoda;
3) Shannon-Weaver diversity index;
4) Saprobity index.

Total biomass, Shannon-Weaver diversity index and saprobity index are good indicators of ecological status and eutrophication. The lowest biomass values were observed in the period of the 70s, during the period of the so-called. hypereutrophication, and the saprobity index in that period was increased.

When comparing the current state with the historical data of the Dniester delta, one can see a decrease in the proportion of rotifers from 74% on average per year in 1949-1952 (Iaroshenko, 1957) up to 31% in 2016-2020 (Nabokin, 2020; Lebedenco et al., 2021). A decrease in the rotifer proportion may be associated with the consequences of hydro-construction, i.e. low water level and silt accumulation that has become unsuitable for this group, since their filter apparatus is filled with silt particles.

Zooplankton, as the most dynamic component of aquatic invertebrates, is characterized by uneven development, disturbances of population in terms of both quantitative and qualitative structure and their dynamics. The specificity of reactions of zooplankton to environmental change, in particular, to water level fluctuations has been expressed by restructuring the composition of species diversity and the fluctuation of quantitative parameters of zooplankton development in aquatic ecosystems.

Development of zooplankton in the Dniester and Prut rivers is strictly influenced by the conditions of the hydrological regime of these rivers. In the monitoring of aquatic bodies, due to constantly changing conditions, accompanied by climate change and the intensification of the anthropogenic factor, the composition and quantitative parameters of zooplankton denote the stability of aquatic ecosystems. The ecological status of the investigated ecosystems, according to the parameters of zooplankton communities, corresponds to the ß-mesosaprobe zone, and the water quality is characterized as moderately polluted (Lebedenco, 2020).

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5 Chapter

RECOMMENDATIONS FOR MONITORING AND IDENTIFICATION OF BENTHOS

Kovalyshyna Svetlana[1], Chuzhekova Tatyana[1], Munjiu Oxana[2]
[1]Ukrainian Scientific Center of Ecology of the Sea, [2]Institute of Zoology

For quantitative and qualitative sampling of macro benthic invertebrates, the following sampling devices are used depending on bottom type and depth:

1) Petersen and Ekman grabs (capture area of 0.025 m²) are used on soft sediments (silt, sand, detrit) and depth from 1.5 m;
2) Surber sampler (capture area of 0.025 m²) are used on all kinds of substrate and depth up to 1.5 m;
3) Rectangular dredge with the width of mouth 0.35-0.50 m and capture distance up to 3 m are used on all kinds of substrate, and depth from 1.5 m;
4) Frames (25x25 cm) are used on all kinds of substrate, mainly in low water and contact zone water-bank;
5) Silk or nylon nets with mesh size of 333 µm are used on soft sediments and macrophytes assemblage;
6) Bottom scrapers (mesh cloth of 500 – 1000 µm);
7) Drift trap.

If a net with square opening of 25x25 cm is used, than the total picked up area is equal to 0.075 m². Bottom scrapers are used for sampling on mixed and hard substrates, including the artificial constructions.

Sampling is done at three transects across the stream lying about 10 m apart. Transects should be placed diagonally in an upstream direction. Sampling is started at the downstream transect and progresses upstream. At each transect five minutes are given for hand-picking from submerged stones and large wooden debris. The animals collected by hand-picking are kept separately from the kick sample.

The drift sampling could be taken using the method of “mowing”, when a net is dragged for three minutes opposite the water stream or on a one meter distance, or the method of by setting a drift trap for 15-30 minutes depends on current velocity. In both cases the procedure is repeated three times. Also, it possible to use kick samplings from the area 0.5 m² for 15 seconds for the same purposes (as an example of kick samplings method please see the video https://www.youtube.com/watch?v=yoFK4hCu42c)

Important to remember:

- Sampling must cover all possible bottom biotopes on each site (stones, large wooden debris, sand, silt, man-made constructions etc.)
- The number of quantitative samplings must be at list 3 per each biotope with the minimal area of a sample of 0.075 m² per biotope.
- The of area of qualitative sampling should exceed the area of quantitative samplings and cover at list a 1 m².

Taxonomic determination of species is carried out with the use of modern microscopes and identification guides and the systematics should be checked with Fauna Europea https://fauna-eu.org/.

The main parameters to be collected for benthic invertebrates are:

1) number of phyla present;
2) phyla dominance - top five ranked phyla in terms of % contribution to total biomass;
3) rate of reophilous to limnophylous species;
4) guild structure.

Changes in the structure of the community are associated with the dynamics of populations of individual species and require a set of data for each individual region. However, a number of authors from Europe and North America (Feld et al. 2012; Hering et al., 2010; Hershkovitz et al., 2015; Langlois et al., 2018; Lawrence et al., 2010; Li et al., 2012; Woznicki et al., 2016; Xiaocheng et al., 2008; Arkansas Aquatic Nuisance Species Management Plan, 2013) recommend the following parameters, which are suitable for investigation of both temperature and flow changes:

- % EPT (sum of Ephemeroptera, Plecoptera and Trichoptera) both in alpha diversity and in total abundance;
- $\frac{\%EPT\ (eurithermic\ \&\ warm\ water)}{\%EPT\ (stenothermic\ \&\ cold\ water)}$ to analyze the effect of the actual temperature change in the presence of long-term data;
- Diversity Coleoptera, Diptera, Odonata;
- % of Insecta from total abundance;
- % of primary aquatic organisms (like flat worms, Oligochaeta, Hirudines, Bivalvia, Gastropoda with gills, Crustacea);
- % Bivalvia diversity/Gastropoda diversity;
- % predators;
- % shredders (like Tipulidae, Limnepelidae and other);
- % scrapers (like Ephemeroptera, Orthocladiinae and other);
- % stenobiont species.

To estimate the impact of hydropower on the Dniester ecosystem it is possible to use the indexes based on flow preference of aquatic invertebrates.

Potamon type index (PTI) (Schöll, Haybach, 2000; Schöll et al., 2005) is designed for large watercourses for which reference conditions are rarely available:

$$PTI = \frac{\sum_{i=1}^{T}(W_i * G_i \sum_{j=1}^{N} A_{i,j})}{\sum_{i=1}^{T}(G_i * \sum_{j=1}^{N} A_{i,j})},$$

where $A_{i,j}$ is the relative abundance of taxon i ($1 \le i \le T$) in sample j ($1 \le j \le N$), $G_i = 2^{(5-Wi)}$ and $Wi = 6\text{-}ECO_i$.

ECO_i is the ecological class of taxon i and ranges between 1 (weakly indicative of Potamon) and 5 (highly indicative of Potamon). Ecological classes of 317 taxa are available from Table 6 in Schöll et al. (Schöll et al., 2005), who defined classes of ecological status according to PTI values with class I = 1.00 < PTI < 1.90 (very good ecological status), class II = 1.91 < PTI < 2.60 (good), class III = 2.61 < PTI < 3.40 (average), class IV = 3.41 < PTI < 4.10 (mediocre) and class V = 4.11 < PTI < 5.00 (bad).

Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence, 1998; Turley et al., 2016, Review of hydropower plants influence on water quantity and quality in Venta, 2017):

$$LIFE = \frac{\sum fs}{n}.$$

This index is based on division of macroinvertebrate families into one of six flow groups. Each flow group is associated with different flow requirements: I (rapid flows), II (moderate to fast flows), III (slow to sluggish flows), IV (slow flowing and standing waters), V (standing waters) and VI (drought impacted sites). This index can be easily calculated using historical data and are good and inexpensive option to assess the influence of flow alterations.

Danish Stream Fauna Index (DSFI) (Lundsfryd et al., 2017; Skriver et al., 2000) is calculated basing on sum of positive (Tricladida, *Gammarus*, every genus of Plecoptera, every family of Ephemeroptera, Elmis, Limnius, Elodes, Rhyacophilidae, every family of case-bearing Trichoptera, *Ancylus*) and negative (Oligochaeta >100, *Helobdella, Erpobdella, Asellus, Sialis,* Psychodidae, *Chironomus,* Eristalini, *Sphaerium, Lymnaea*) indicator groups. The index value (fauna class) is a function of occurrence of selected indicator taxa in combination with the number of diversity groups (See Annex 1).

Table 5.1. Correlation between DSFI and ecological status

Status	High	Good	Moderate	Poor	Bad
DSFI	7	5-6	4	3	1-2

Desktop-Software *ASTERICS* (=AQEM/STAR Ecological River Classification) (Version 4.04) covers nearly all of recommended parameters and indexes (https://gewaesser-bewertung.de/files/asterics_4.0.4-setup.zip). On the other hand it is possible to use online database https://www.freshwaterecology.info/ to investigate autecology of species. An example of selected parameters listed above calculated within ASTERICS for the Middle and Lower Dniester is given in the Annex 1, Table 2.

Annex 1

Table 1. Danish Stream Fauna Index (DSFI)

Indicator groups (IG)	Number of taxa	Diversity groups			
		< -2	-1 to 3	4 to 9	10
Indicator Group 1 (IG 1):					
Brachyptem, Capnia, Leuctra, Isoperla, Isoperla, Isoptena, Perlodes, Protonemura Siphonoperla, Ephemeridae,	> 2 taxa	-	5	6	7
Limnius, Glossosomacidae, Sericostomatidae.	1 taxon	-	4	5	6
Indicator Group 2 (IG 2):					
Amphinemura, Taeniopreryx, Ametropodidae, Ephemerellidae, Heptageniidae, Leptophlebiidae, Siphlonuridae, *Elmis, Elodes,* Rhyacophilidae, Goeridae, *Ancylus If Asellus>5* go to IG 3. If Chironomus >5 go to IG 4		4	4	5	5
Indicator Group 3 (IG 3):					
Gammarus >10, Caenidae, Other Trichoptera >5 If Chironomus > 5 go to IG 4		3	4	4	4
Indicator Group 4 (IG 4):					
Gammarus >10, *Asellus,* Caenidae	>2 taxa	3	3	4	
Sialis, Other Trichoptera	1 taxon	2	3	3	-
Indicator Group 5 (IG 5):					
Gammarus <10 Baetidae	>2 taxa	2	3	3	-
Simuliidae >25 If Oligochaeta >100 go to IG 5, 1 taxon If Eristalini > 2 go to IG 6	1 taxon or if Oligochaeta >100	2	2	3	-
Indicator Group 6 (IG 6):					
Tubificidae, Psychodidae Chironomidae Eristalini		1	1	-	-

Table 2. Macrozoobenthos metrics calculated under selected requirements

Metrics	Mayaky (Ukraine)	Palanca (Moldova)	Total
EPT-Taxa % in total abundance	0	5.051	4.63
EPT-Taxa species number	0	5	5
EPT [%] (abundance classes)	0	5.051	4.63
Taxonomic group (number of taxa)			
Turbellaria	0	1	1
Nematoda	0	1	1
Nematomorpha	0	1	1
Gastropoda	6	21	22
Bivalvia	3	6	8
Polychaeta	0	1	1
Oligochaeta	6	16	18
Hirudinea	0	2	2
Crustacea	2	10	11
Araneae	0	0	0
Ephemeroptera	0	1	1
Odonata	0	8	8
Plecoptera	0	0	0
Heteroptera	0	4	4
Trichoptera	0	4	4
Coleoptera	0	2	2
Diptera	3	20	23
Bryozoa	0	1	1
EPT-Taxa	0	5	5
EPT/ Oligochaeta	0	0.312	0.278
EPT/Diptera	0	0.25	0.217
EPTCBO (Eph., Ple., Tri., Col., Bivalv., Odo.)	3	21	23
Diversity Coleoptera, Diptera, Odonata	3	30	33
% of Insecta from total abundance	15%	39%	39%
Feeding types			
[%] Grazers and scrapers	13.5	15.354	14.537
[%] Miners	0	1.111	1.019
[%] Shredders	1	4.848	4.63
[%] Gatherers/Collectors	35	27.475	27.037
[%] Active filter feeders	19	15.354	15.926
[%] Passive filter feeders	6.5	0.202	1.389
[%] Predators	3.5	17.879	17.037
[%] Parasites	0	2.222	2.037
[%] Other Feeding types	1.5	3.434	3.426
[%] no data available	20	12.121	12.963
[%] (Grazers + Scrapers)/(GatherersCollectors + FilterFeeders)	0.223	0.357	0.328
[%] Xyloph. + Shred. + ActFiltFee. + PasFiltFee	26.5	20.404	21.944
[%] Shredders (scored taxa* = 100%)	1.25	5.517	5.319
[%] Gatherers/Collectors (scored taxa = 100%)	43.75	31.264	31.064
Active/Passive filter feeders (all taxa)	2.923	76	11.467
LIFE Index	6.6	6,357	6,4
DSFI Diversity Groups	2	2	3
% Bivalvia/Gastropoda diversity	50%	29%	36%

*Scored taxa are given according their presence in ASTERICS database

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6 Chapter

ICHTHYOFAUNA IN THE CONDITIONS OF THE IMPACT OF HYDROTECHNICAL CONSTRUCTIONS ON RIVER ECOSYSTEMS

Bulat Denis, Bulat Dumitru, Usatii Marin
Institute of Zoology

Construction of Dubasari (1953) and Novodnestrovsk (1980) reservoirs on the Dniester river and of Costești-Stanca reservoir (1978) on the Prut river caused the rupture of the longitudinal connectivity, what resulted in the disturbance of the hydrological, thermal, hydrochemical and hydrobiological regimes, and, as consequence, have had a major negative impact on taxonomic diversity and fish productivity in riverbed sectors.

Thus, for the evaluation of the impact of hydrotechnical constructions on the fish fauna from lotic ecosystems, various methods of fish catching (passive or active) are used, through series of fishing, which are performed with a frequency of three times per year and coincide with the main phases of the life cycle of fish:

- spring fishing (seine net, trammel net, gill net and trawl net) (March – April) with the beginning of schooling and migration to reproduction sites;
- summer-autumn fishing (seine net, trammel net, gill net and trawl net) (August-September) during the fattening of juvenile age groups as well as mature age groups, which are spread more or less uniformly on the surface of waters;
- late autumn fishing (seine net, trammel net, gill net and trawl net) (October – November) at the time when fish migrate to deep places for wintering.

Fishing in these seasons allows highlighting the peculiarities of seasonal behaviour and of the distribution of fish, which can be greatly modified by the hydromorphological and hydrometeorological factors.

The construction of the Novodnestrovsk dam had an extremely negative influence on the reproduction conditions of fish in the middle and lower sector of the Dniester River. Disturbances in the thermal regime appeared, which consist of low temperatures in the spring-early summer and high – in the autumn-winter periods. This thermal influence downstream of the hydropower plant causes various dysfunctions in the life cycles of hydrobionts, and at the cellular level – in the process of gametogenesis. The onset of the reproduction period in fish species, such as *Rutilus rutilus, Rutilus heckelii, Esox lucius, Abramis brama* and *Sander lucioperca*, has started later than usual, which has led to the reduction in the fattening period of the juveniles and their entry into the wintering period with a poor physiological status (Bulat, 2017; Usatii et al., 2017).

The most common changes in the reproduction system in freshwater fish species are: asymmetric development of ovaries and testicles, their atypical shape, change in the duration of ovogenesis and spermatogenesis, shift in the timing of reproduction, cases of mass resorption of gametes in the last phases of trophoplasmic growth, reduction of egg portions and decreased prolificacy, disorders in the process of vitellogenesis, decreased fertilization capacity, decreased share of individuals capable of reproduction, abortion of eggs by damaging follicular membranes, etc. (Chepurnova,1972).

Table 6.1 can be used to assess the stages of maturation of sexual products in fish species.

At the mature stages of development, sex determination is easy to achieve: in adult females the eggs are easily distinguished in the ovaries and in males the testes are smooth, whitish, apparently ungranulated. Many species of fish have sexual dimorphism, which is more obvious during the reproduction periods, being highlighted in the change of colour (brighter in males), body proportions (usually larger in females), the appearance of growths (nuptial tubercles in males of some cyprinids).

Other indicators used to assess the reproduction potential (Pricope, 2013):

- absolute (individual) prolificacy – the total amount of eggs in the fish ovaries;
- relative prolificacy - the number of eggs per unit of fish weight;
- gonosomatic ratio or *gonadosomatic index* (GSI) - it is based on the correlation between the weight of the ovaries and the body weight of the female and is calculated according to the formula:

$$GSI = \frac{OW \times 100}{W}, \qquad (1)$$

where: *OW* – ovaries weight, *W* – fish body weight.

At the individual and population level, under the influence of anthropogenic pressure, there are various dysfunctions in fish, which can serve as strong indicators in the process of assessing the quality of the environment.

In the ichthyological analysis at individual level, a range of gravimetric measurements, indices and coefficients are used.

The following biometric parameters are determined by direct measurements:

- total body length (L) – the distance from the tip of the snout to the tip of the caudal fin (cm);

Table 6.1. Gonad maturity stages (Meyer scale correlated after Holden-Raitt)

Stages		Characteristics
I	Immature	- very small gonads (ovaries and testes), in the form of long threads; - are located close to the spine; - light pink to white ovaries, no sex cells are distinguished
II	Early maturation	- the thin gonads are gray-pink, more or less symmetrical; - in adult individuals, this is the resting stage that is installed after spawning; - at a future maturation the development of the gonads starts from stage II
III	Development	- gonads occupy almost two-thirds of the abdominal cavity; - the ovaries are red-orange, the oocytes are distinguished by the naked eye; - testes are creamy white, no milt is obtained at stripping
IV	Maturation	- gonads occupy almost two-thirds of the abdominal cavity; - testes are creamy white, leave milt at stripping; - ovaries are red-orange, the oocytes are spherical and visible to the naked eye
V	Reproduction	- sexual products are released at gentle touch of the abdomen; - eggs are transparent and large, there are also opaque, immature eggs in the ovaries
VI	After reproduction	- soft, wrinkled gonads occupy about half of the abdominal cavity; - ovaries are highly vascularized, they may also contain degenerated eggs; - testes with some unreleased sperm

- standard body length (l) - distance from the tip of the snout to the base of the caudal fin (cm);
- head length (hl) - distance from the tip of the snout to the posterior edge of the opercular bone (cm);
- height (H) - distance from the edge of the back to the line of the abdomen, in the widest region of the body (cm);
- body thickness (BT) - maximum distance between the flanks of the fish (cm);
- body circumference (BC) - measured in the area of the maximum thickness of the fish (cm).

The gravimetric characters are determined using the electronic scale, which is chosen according to the size of the fish and the desired accuracy. The total weight (W) is the mass of the fish expressed in grams. The individual weight is determined on live specimens (normal or anesthetized), on recently dead specimens or on frozen specimens. When assessing the weight on preserved material, certain corrections are made, depending on the method of preservation, as losses of up to 5% of weight can be registered. In addition to the value of the total weight, the weight of the body without viscera, the weight of the intestinal contents and the weight of the gonads are calculated, values which will later be used to assess the following indices and coefficients:

- fattening coefficient (Fulton)

$$Fc = \frac{W}{l^3} * 100 \qquad (2),$$

- the increase of absolute growth

$$W_{final} - W_{initial} \qquad (3),$$

- bowel filling index (BFI)

$$BFI = \frac{Weigt\ of\ intestinal\ content(g)}{Weight\ of\ gutted\ fish\ (g)} * 10\,000 \qquad (4).$$

Determining the age of fish from a natural or anthropogenic aquatic ecosystem has a special theoretical and practical importance. Studies on ichthyofauna in an aquatic ecosystem require accurate knowledge of the age of fish, in order to establish the annual growth rate and age structure of populations of each species. These data, along with other important indicators, reveal the reproductive capacity of the population and show the well-being of the population in that ecosystem. To determine the age of fish, the principle of the anatomical method is most often used, which consists in the fact that in fish in the temperate zone, the calcium impregnation of some bone organs (scales, otoliths, rays, vertebrae) is not uniform during the year, which determines the appearance of growth rings in these organs (Pricope, 2013).

Bertalanffy equation is used to characterize the growth of different fish species. The calculation of the growth parameters *k* and t_0 can be performed by pre-setting the value of as the input value (Shibaev, 2007).

This method is used more for the short-lived species, when the highest empirical gravimetric values are not affected by selective fishing and correspond to the realities, the populations having a complete and well-balanced structure. Alternatively, this value can be taken from other unanimously recognized scientific sources, by making the corresponding references (for example, fishbase.org) (Fish Base. A Global Information System on Fishes).

In the present study, the Ford-Walford relation was applied, which requires the prior calculation of the value l_∞, used to describe the Bertanlanffy equation and allows, in real conditions, based on empirical data, to estimate the maximum theoretical physiological growth. Thus, the length of fish of age t will be:

$$l_t = l_\infty(1 - e^{-k(t-to)}) \qquad (5)$$

and, respectively, the fish body mass:

$$w_t = w_\infty(1 - e^{-k(t-to)})^3 \qquad (6)$$

where:

l(t) - standard length of fish at age t;

w(t) - body weight of fish at age t;

l_∞ - the maximum theoretical length of the fish;

w_∞ - the maximum theoretical weight of the fish, g;

k - constant of growth;

t_0 - theoretical age at which the length of the fish is «0»;

e - base of the natural logarithm.

Following the application of a series of mathematical transformations, the given equations can be brought to the following final equations: $l_{t+1} = a + bl_t$ and respectively, $w_{t+1}^{\frac{1}{3}} = a + bw_t^{\frac{1}{3}}$.

For the calculation of the coefficients *a* and *b*, the method of least squares was used:

$$a = \bar{y} - b\bar{x} \quad (7)$$

$$b = \frac{n\sum xy - \sum x \sum y}{n\sum x^2 - (\sum x)^2} \quad (8)$$

Correlation between the body length and body weight - from an analytical point of view, it is described by the equation:

$$w = a \cdot l^b \quad (9),$$

where:

w - body weight, g;

l - standard length of the fish, cm;

a - constant equal to *w* when *l* = 1;

b - exponential coefficient.

The r_{xy} correlation coefficient was calculated according to the equation:

$$r_{xy} = \frac{n\sum xy - (\sum x)(\sum y)}{\sqrt{[n\sum x^2 - (\sum x)^2][n\sum y^2 - (\sum y)^2]}} \quad (10)$$

If in the process of growing the species geometric similarities (harmonic balance) of body shape are maintained, then *b*=3. But if *b*>3, then the positive allometry is found, and *b*<3 indicates a negative allometry (favoring the increase in length). The length-weight correlation can be expressed in logarithmic form:

$$lgw = a + b \cdot lgl \quad (11)$$

Following the calculations, a series of data are obtained regarding the growth of fish, which characterize the type of growth in a given ecosystem, allowing comparisons between populations of the same species in different aquatic ecosystems (different ecological conditions), or between populations of different species in the same aquatic ecosystem (similar ecological conditions).

The confidence limits (CL) for *b* parameter define the upper and lower value of the range, within which the calculated parameter value is, with a certain probability. In general, as the probability threshold the value of 95% (a=0.05) is accepted. At this threshold, it can be considered that there is a 95:5 (or 19 to 1) probability that the value of *b* is placed between the given values. The confidence limits are calculated according to the equation:

$$LC = b \pm t\alpha\sqrt{\frac{S^2}{n}} \quad (12)$$

where:

b - exponential coefficient;

S^2 - variance; *n* - number of analyzed specimens;

ta - tabulated value for the Student's t-distribution.

It is considered that the population sex structure of most fish species in natural conditions is close to the ratio 1♀:1♂, being optimal for ensuring the highest productive parameters. Usually, males predominate in young age groups, and with age the sex ratio becomes more balanced due to males' higher natural mortality. It has been found that in populations of fish species where big-sized age groups are actively extracted, smaller males get advantaged.

Following the construction of numerous reservoirs in the riverbeds, a spatial niche, uncharacteristic for river species, was formed and expanded - the pelagic and littoral area, where, in the condition of water stagnation, a vertiginous development of phytoplankton, zooplankton and macrophytes occurs. This can also lead to a positive response from consumers at higher trophic levels, such as fish. Thus, the exaggerated growth of the density of euribiont species of fish and their biomass (*Carassius gibelio* sensu lato, *Rutilus rutilus*, *Alburnus alburnus*, *Rhodeus amarus*, *Perca fluviatilis*, etc.) cannot serve as a beneficial indicator for riparian ecosystems, being a sign of their active eutrophication. On the background of the biological progression of some euritope species of fish, as a rule, the numerical depression and impoverishment of the diversity of native rheophilous fish species occur. As result, an important indicator of the impact of hydrotechnical constructions on lotic ecosystems is the change in potential fish productivity, which is assessed on the basis of natural trophic resources until the construction of these dams and after their construction.

The following input data are used to assess the potential fish productivity (Kitaev, 2007):

- average multiannual biomasses from the vegetative period of the main groups of fodder hydrobionts: phytoplankton (g/m^3), zooplankton (g/m^3), zoobenthos (g/m^2), macrophytes (g/m^2);
- production (P) of fodder organisms based on the P/B coefficients for each group is calculated: phytoplankton – varies between 175 and 353, zooplankton – 30-45, zoobenthos – 1.4 – 2.2, macrophytes – 1.5-3;
- obtained production is added up and converted from grams and meters to kilograms per hectare;
- coefficient of recovery by fish (K3) of the production of fodder organisms is considered 0.3 (or 30%) for phytoplankton; 0.54 (or 54%) for zooplankton and 0.45 (or 45%) for zoobenthos;
- finally, knowing the feed conversion coefficient (K2) for each group of hydrobionts consumed by phytoplanktonophagous, zooplanktonophagous and zoobenthosophagous fish, the total potential fish productivity can be assessed (according to the *Instruction on assessing the damage caused to fishery resources in the water bodies of the Republic of Moldova*, approved by the Ministry of Ecology, Construction and Territorial Development of the Republic of Moldova, October 7, 2003, no. 206, the value of the trophic coefficient (K2) is: for phytoplankton – 30, for zooplankton – 10, for macrozoobenthos – 8, macrophytes – 40);
- to adjust the values of potential fish productivity, the share of predators in ichthyocenosis (%) is taken into account, multiplying the final value by the pressure coefficient of ichthyophagous species ($K_{predator}$) *(Table 6.2)*.

For example, if there was found a potential fish productivity of 52 kg/ha and the share of ichthyophagous is 20% in ichthyocenosis, then the final value is the product between 52 kg/ha and 0.56 ($K_{predator}$ for 20%), obtaining the value of the final potential fish productivity of 29.12 kg/ha.

In the analysis of the quantitative values of fish communities, such as biomass – B (kg/ha) and density – ρ (indiv./ha), the method of test surfaces was used, by applying the necessary correction coefficients (depending on the tool used and the area of action, the value of the catchability coefficient (q) varies from 0.1 to 0.6) (Usatii et al., 2017; Kitaev, 2007; Kotlear, 2004; Shibaev, 2007).

Table 6.2. Evidence of the influence of ichthyophagous species on the potential fish productivity (after Kitaev, 2007)

	The share of fish ichthyophagous species (%)								
	0	5	10	20	30	40	50	70	90
$K_{predator}$	1.0	0.83	0.71	0.56	0.45	0.39	0.33	0.26	0.22
Decrease (%)	0	17	29	44	55	61	67	74	78

Figure 6.1. Abundance of macrophytes in the Dniester riverbed (Criuleni) (Photo: Bulat D.)

As previously mentioned, in the current ecological conditions of river fragmentation, substantial changes in the structure of ichthyocenoses are recorded. As a result of these hydrotechnical works, the speed of water has been reduced, significantly accelerating the negative processes of siltation and weeding of the riverbed sectors (limnification process).

In this way, the habitats of the typical river species were strongly altered: *Barbus barbus, Vimba vimba, Chondrostoma nasus, Ballerus sapa, Squalius cephalus, Zingel zingel, Zingel streber, Gymnocephalus acerina*, species from genera *Gobio* and *Romanogobio, Alburnoides bipunctatus etc.*

Also, the effect of riverbed fragmentation led to the decline of migratory and semi-migratory species, such as species from the *Acipenseridae* family, *Salmo labrax, Anguilla anguilla, Pelecus cultratus* etc., as the access to the upstream spawning grounds was restricted and the remaining spawning grounds located downstream the hydrotechnical constructions were degraded by clogging.

Figure 6.2. *Cynocephalus acerina* (Gmelin, 1789) – typical river species, which reached the numerical decline in the Dniester river (Photo: Bulat D.)

Figure 6.3. *Pelecus cultratus* (Linnaeus, 1758) (CR RM – VU) is a semi-migratory species with a vulnerable status in the Dniester river (until the middle of the 20th century being very numerous) (Photo: Bulat Dm.)

Thus, a very important indicator for assessing the impact of hydrotechnical constructions on lotic ecosystems is the diversity of fish species found before and after their construction. In order to assess the ichthyofaunal diversity with the identification of taxa up to species rank, specialized identification guides are used (Fish Base. A Global Information System on Fishes; Kottelat, Freyhof, 2007).

The meristic characters used to determine taxonomic affiliation are:

- number of scales in the lateral line;
- transversal rows of scales (counted in the highest part of the body);
- number of rays in the fins;
- formula of pharyngeal teeth;
- number of gill spines.

It should be mentioned that the structure of the "core species" for a certain type of ecosystem (for example, for the Dniester river – *Barbus barbus, Vimba vimba, Chondrostoma nasus, Gymnocephalus acerina, Ballerus sapa, Squalius cephalus,* familia *Acipenseridae, Pelecus cultratus*), allows reconstructing of the history of environmental conditions and highlighting of the limiting factors. Currently, the representatives of the families *Petromyzontidae, Acipenseridae, Thymallidae, Salmonidae, Lotidae, Cottidae* have practically disappeared (or are found sporadically).

Figure 6.4. *Umbra krameri* Walbaum,1792 (RB RM – EN) – a typical pond species that has declined numerically due to the destruction of wetlands (Photo: Bulat Dm.)

In addition to the typical rheophilous and cryophilic taxa, the populations of lacustric and palustric stenobiont species were affected, such as *Carassius carassius, Tinca tinca, Umbra krameri, Misgurnus fossilis,* which were vitally dependent on the biotopes of ponds and small floodplain lakes affected by the massive drying and chemicalization in the years' 50-'80 of the XXth century.

On the other hand, on the background of the reduction of diversity of the stenobiont species of fish, the advancement and proliferation of the small native euritope species, such as *Alburnus alburnus, Rhodeus amarus, Perca fluviatilis, Rutilus rutilus, Blicca bjoerkna*, invasive allogenic, such as *Carassius gibelio* sensu lato, *Pseudorasbora parva, Lepomis gibbosus, Perccottus glenii*, and opportunistic intervenient species, such as *Neogobius melanostomus, Neogobius fluviatilis, Babka gymnotrachelus, Proterorhinus semilunaris, Ponticola kessleri, Syngnathus abaster, Pungitius platygaster, Gasterosteus aculeatus, Atherina boyeri, Clupeonella cultriventris* is observed.

Figure 6.5. Allogenic *Lepomis gibbosus* and species of *Gobiidae* are currently in the phase of biological progression in the Dniester river (Photo: Bulat D.)

Thus, an important indicator which characterizes the degree of fish bioinvasion, following the changes in abiotic conditions in the ecosystem of fragmented rivers, is the Branch index, which represents the ratio between the number of allogenic species and the total number of species in an ecosystem and its modified form, which expresses the share of abundance of allogenic species (*Table 6.3*) (Skolka, Gomoiu, 2004).

Table 6.3 Analysis of invasion indices in ichthyocenoses of the Dniester and Prut rivers

No	Ecosystem	Invasive index (Branch, 1994),%	Invasive index (by abundance),%
1.	Dniester river	3	4
2.	Prut river	3	2

Note: 0 – there is no biocontamination;
1 – low biocontamination (> 0 – <10%);
2 – moderate biocontamination (> 10–20%);
3 – high biocontamination (21–50%);
4 – severe biocontamination (> 50%).

It should be mentioned that, in unstable ecological conditions, the share of interspecific hybrids has increased in the ichthyocenoses of the aquatic ecosystems of the Republic of Moldova. Unfavourable conditions during the reproductive period for a species can cause disturbances in the process of gametogenesis and, respectively, the modification of the spawning periods. As a result, when favourable conditions return, there may be overlaps in the reproduction of several species in the same spawning areas (especially in case of shortage of such areas), and as a result - the emergence of hybrids (a phenomenon with increasing frequency after the construction of the Novodnestrovsk and Dubasari dams). The most numerous hybrids in the aquatic ecosystems of the Dniester and Prut are between *Abramis brama* x *Rutilus rutilus*, *Abramis brama* x *Blicca bjoerkna*, *Blicca bjoerkna* x *Alburnus alburnus* and *Alburnus alburnus* x *Scardinius erythrophthalmus*. Usually, the overlap of the breeding period in the spawning areas occurs when the water level is low and the temperature rises slowly (a phenomenon frequently observed in the middle Dniester), or when the water level is high and the temperatures rise suddenly.

According to some researches (Djimova, 2009; Moshu, 2014), the study of parasitoses in biotic relationships in ichthyocenoses can provide useful data for assessing ecosystem welfare. Significant anthropogenic pressure leads to the accumulation of pollutants in hydrobionts, what, in turn, reduces their degree of resistance in the host-parasite relationship, often causing epizooties.

The possibility of using parasites in fish as bioindicators is justified by the double influence exerted on them: from the external environment and from the host organism. Thus, one of the key factors influencing and determining the degree of parasitic invasion in fishing communities is the physiological state of the host organism.

Among the most significant factors, which currently stimulate the spread of ichthyozooanthropocenoses in the conditions of the Republic of Moldova can be listed:

1. active limnification of lotic aquatic ecosystems leading to an increase in the number of final, intermediate and complementary hosts (planktonic crustaceans, molluscs, oligochaetes, fish, ichthyophagous birds, etc.);
2. decrease of floodplain areas and concentration of birds (hosts) on limited areas, causing contact of affected individuals with healthy ones;
3. overfishing of large fish and excessive numerical development of small and medium-sized species, which subsequently serve as basic vectors in the transmission of parasitoses;
4. expansion and active proliferation of alien and intervenient fish species;
5. deplorable sanitary-ecological state of water bodies used for fish culture (Djimova, 2009).

In order to decipher the relationships established between different species within the ecosystem, the hierarchies that are consolidated within the ichthyocenoses, a set of mathematical methods known under the generic name of synecological analysis is used (Sirbu, Benedek, 2012; Lebedeva et al., 1999).

This type of analysis allows us to accurately identify the species that have the largest share in the ecosystem in terms of energy exchange with the environment, which are the species characteristic for a biotope, or species that have accidentally arrived in the researched area.

Also, the interrelationships between the species that make up the biocenosis can be established with sufficient precision. Depending on how they are calculated, two distinct categories are used: 1. analytical ecological indices (calculated based on the raw data collected in the field); 2. synthetic ecological indices (calculated based with analytical indices), which are used to highlight interrelationships between species, communities or cenoses (Monitoring of the water quality and assessing the ecological status of aquatic ecosystems, 2015).

6.1. Analytical ecological indices

Numerical abundance (A) – represents the absolute number of individuals of a species in the research area. Five classes are used to estimate the abundance: 0 – absent; I – rare; II – relatively rare; III – abundant; IV – very abundant (Davideanu, 2013).

The abundance of the populations of a species is an important criterion in prioritizing the species of interest for conservation, especially if we have some comparative information, respectively data on their abundance/density from the past. In this situation, the tendency (or rate) of increase or, on the contrary, of the decrease of the size of the species can be revealed.

Relative abundance (Ar) – represents the share (%) of each species in the studied biocenosis and is estimated according to the equation:

$$Ar = \frac{n}{N}100, \qquad (13)$$

where:

n = number of individuals of species A,

N = total number of individuals of all species.

And in this case, the method of abundance classes is used, marked by conventional signs:

0	between 0 and 10%
I	between 11 and 30%
II	between 31 and 50%
III	between 51 and 70%
IV	between 71 and 100%.

Often the relative abundance (Ar) is expressed by *dominance* (D), having the same ecological meaning. Depending on the value of dominance, the species are assigned to the following classes:

D1 - subrecedents	- less than 1.1%
D4 - dominants	- between 5.1 and 10%
D2 - recedents	- between 1.1 and 2%
D5 - eudominants	- over 10%.
D3 - subdominants	- between 2.1 and 5%

This indicator can also be used in the form of the share of species reflected by the biomass of catches. The values of dominance expressed in the form of number and biomass will be totally different in the case of 10 *Abramis brama* and 100 *Gasterosteus aculeatus*.

Frequency (F) – indicates the percentage of samples in which a species is present compared to the total number of samples collected in the research area (biotope). By frequency, the species are classified into:

common	- frequency over 70%
rare	- frequency 10 - 29%
relatively common	- frequency of 50 - 69%
very rare	- frequency below 10%.
relatively rare	- frequency of 30 - 49%

In the case of studying a heterogeneous habitat, the preferences of the given population for its certain characteristics can be estimated with the help of this parameter. However, its usage requires a lot of caution, as it can have different meanings if reported at different scales. Knowing that species demonstrate a high affinity for certain characteristic habitats, to which they are best adapted, if we take samples only from them, we can quickly conclude that a particular species is extremely common, and vice versa, if the samples include areas that do not meet the conditions necessary for the survival of that species, the value of the parameter will obviously be small.

It is important that the samples are collected in sufficient numbers and at different times of the year. When there are a small number of samples, we may not be able to identify the rare populations in a habitat. At a time analysis, we can conclude that, within a year, a population has a high frequency, but, in fact, the samples were collected only during the period of its maximum migration.

Also, the frequency (F) can be expressed by *constancy* (C), which shows the continuity of the appearance of a species in a given biotope and its importance in achieving the structure of the biocenosis.

Constancy is estimated via equation:

$$C = \frac{p}{P} 100, \qquad (14)$$

where:

p = number of samples in which the species A is found,

P = total number of collected samples.

Depending on the values of constancy, four categories of species were established:

C1 <25% - accidental species
C3 = 50.1-75% - constant species
C2 = 25.1-50% - accessory species
C4 > 75% - euconstant species.

The constancy of a local endemic or of a stenotope species can be high within the range, respectively, within its typical habitat, being characterized as constant. However, by increasing the studied area, the conditions tolerated or preferred by the respective species will be very quickly overcome, which will, thus, become accessory and, ultimately, accidental. Therefore, caution is also required in interpreting the values of constancy.

According to N. Botnariuc and A. Vadineanu (1982), *fidelity* expresses the strength of the links of a species with other species of the biocenosis or of a given ecosystem. Thus, species can be divided into characteristic, preferential, random and ubiquitous (indifferent).

6.2. Synthetic ecological indices

Ecological Significance Index (W) represents the relationship between the structural indicator (constancy) and the productive indicator (dominance), reflecting the position of a species in the biocenosis (Davideanu, 2013).

It is calculated according to the equation:

$$W = \frac{C_A \cdot D_A \cdot 100}{10000}, \qquad (15)$$

According to the values obtained for this index, the species are divided into the following classes:

W1 < 0.1%,
W2 0.1 - 1%,
W3 1.1 - 5%,
W4 5.1 - 10%,
W5 >10%.

Class W1 corresponds to accidental species, classes W2 and W3 – to accessory species (accompanying) and classes W4 and W5 – to species characteristic for the given biocenosis.

It is not desirable to use an unconditional scale of the type: euconstant, constant, accessory or random species, depending on the values of this index, due to the same criticisms that were mentioned earlier regarding the frequency and constancy. From the other hand, one and the same value can be achieved by high abundance and low frequency, or vice versa, what have different meanings in ecology.

Specific Similarity Index expresses the degree of similarity between two samples/communities/biocenoses in terms of the presence of common species. It is estimated according to the values of the Sorensen coefficient:

$$S = \frac{2c}{a+b} * 100, \qquad (16)$$

where:

a - number of species in A sample;

b - number of species in B sample;

c - number of common species in A and B samples.

The values of this index vary between 0 and 1.

The analysis of similarity can be performed with the help of several indices that can be found in the literature. However, the value obtained by the Sörensen index includes any other qualitative information reflected by them, especially, since it is possible to include equivocal results by using indices that do not vary on a standardized scale (Sirbu, Benedek, 2012).

Cenotic Affinity Index (*q*) allows highlighting the existing affinities between species of a group in a cenosis, affinities established based on common preferences for living environment:

$$q = \frac{c}{a+b-c} 100, \qquad (17)$$

where:

a - number of samples in which A species is found;

b - number of samples in which B species is found;

c - number of samples containing both species simultaneously.

The result of calculating this index is presented in the form of a dendrogram, which shows the degree of affinity between species and how they are grouped according to affinities.

Diversity Index (Shannon-Wiener Index) is used for information purpose and is calculated according to the following equation:

$$H(S) = -k \sum_{i=1}^{S} p_i \cdot lgp_i, \qquad (18)$$

$$p_i = \frac{N_i}{N}, \qquad (19)$$

where:

k - conversion factor for changing the base of the logarithm from 10 to 2, having the value of 3.321928;

N - total number of individuals;

N_i - number of individuals of species *i*;

S - total number of species;

p_i - species dominance.

The use of this index allows performing of comparative studies, regardless of the sample size. Its value is directly proportional to the number of species and their share of representation. The greater the diversity and share of stenobiont species in an ecosystem, the higher the value of this index. This index is also preferential from the point of view of error theory, especially in the case of rare species, which changes insignificantly its value. Therefore, it can also characterize the functional aspect of the biocenosis, because the species that have become rare usually play an insignificant functional role. However, this fact does not deny the importance of rare species in establishing the faunal heritage. The negative correlation between the degree of trophicity of the ecosystem and the value of this index has been demonstrated, serving as an indicator of organic pollution.

Equitability (e) (Lloyd-Gheraldi) varies between 0 and 1. It tends to 0 when most individuals belong to a single species and to 1 if each species is represented by the same number of individuals:

$$e = \frac{S`}{S} \text{ sau } e = \frac{H(S)}{S}, \qquad (20)$$

where:

S' - theoretical number of species expressed by *H(S)*;

S - observed number of species.

Simpson Index (Is) measures the probability that two individuals taken randomly from a sample or a series of samples belong to the same species. It is very sensitive to changes in the abundance of dominant species (shows the "concentration" of dominance):

$$I_S = \sum p_i^2, \tag{21}$$

In conclusion, it should be emphasized that, beyond the fact that most studies on specific richness target almost exclusively the number of species, this number cannot be the only criterion for assessing the biodiversity quality of a studied unit. For example, two ichthyocenoses from two types of aquatic ecosystems from the same geographic region are examined. Ichthyocenosis A has 50 species and biocenosis B – 30 species. If we absolutize the criterion "number of species", than the biocenosis A seems to have a higher specific biodiversity than biocenosis "B" with 20 species. If biocenosis B has 19 species from the category of those with various rarity status, and biocenosis "A" has only 10 species from these categories, the more complex structure of biocenosis "B" is obvious. Thus, this example suggests the need to take into account several elements when assessing the specific biodiversity of a water body.

Currently, the index that uses aquatic vertebrates to assess the water quality is the Biotic Index of Fish Integrity, which has been introduced in the US for the first time (Index of Biotic Integrity, IBI, Karr, 1981), with multiple subsequent changes both in the country of origin and in Europe (Florea, 2007; Manual for application of the European Fish Index (EFI); Pricope, 2010; Semenchenko, 2004).

An important advantage of applying the Index of Biotic Integrity consists in the possibility of analysing the fishing community by taking in account the parameters that integrate the three structural levels of organisation of living things: the individual, population and fishing community level. In Europe, the system for assessing and classifying the water bodies based on fish fauna is applied in a modified form called EFI+ (European Fish Index) (Manual for application of the European Fish Index (EFI)).

The metrics selected and used for the calculation of EFI+ are related to two major categories: salmon water bodies and cyprinid water bodies. In some particular situations, it is difficult to delimit those two types of water bodies. In these cases, the importance of the specialist's opinion and competence, based on the level of knowledge of the history and ecological characteristics of the ecosystem, increases.

It was found that the EFI is sensitive to pressures on water quality and is not a very good indicator to highlight the hydromorphological pressures, which are so evident in the conditions of the Republic of Moldova. Application of IBI method also revealed deficiencies in identifying the correct thresholds for separating quality classes within a type of ecosystem (upper and lower river sector) and between different types of ecosystems (rivers of different sizes, lakes, ponds).

Considering the regional specificity of the ichthyofauna, the pressures exerted on the ichthyocenoses, as well as the particularities of reaction to these threats, the IBI was adapted for the lotic ecosystems in the conditions of the Republic of Moldova (Bulat, 2017) *(Table 6.4)*.

The Biotic Integrity Index does not claim to replace physico-chemical control or to have a predictive role. Because it is forced to fill in the missing information, IBI may seem imperfect and unsatisfactory at first, but, on the other hand, we cannot expect data generated by fundamental research while natural heritage degradation continues and urgent action needs to be taken to limit, as far as possible, the impact on aquatic ecosystems (Florea, 2007).

Table 6.4. Criteria proposed for determining the Biotic Integrity Index (IBI) of lotic (riparian) aquatic ecosystems from the Republic of Moldova

Category of metrics	Proposed metrics	Score 5	Score 3	Score 1
Specific structure of ichthyocenosis (qualitative aspect)	1. Share of native species (in relation to those allogenic and intervenient)	>67%	33-67%	<33%
	2. Share of salmonid and acipenserid species (for small rivers - of salmonids and cottids)	>5%	2-5 %	<2%
	3. Share of native rheophilous species	>40%	20-40%	<20%
	4. Total extinct (Ex) or endangered (E) species	0	1 - 2	> 2
Trophic and reproductive metrics of ichthyocenosis	5. Relative abundance of phytophilic species	<30%	30-60%	>60%
	6. Relative abundance of obligatory ichthyophagous species	>10%	3-9%	<3%
	7. Relative abundance of omnivorous species (polyphagous)	<20%	20-40%	>40%
Demographic structure and health of ichthyocenosis	8. Relative abundance of individuals with $l_{stand.}$ > 15 cm	>20%	10-20%	<10%
	9. Relative abundance of intervenient and invasive alien fish species	<5%	5-10%	>10%
	10. Share of hybrids and individuals with abnormalities, tumors and parasitic diseases	<0.1%	0.1-1%	>1%
Total score				

Table 6.5 Biotic Integrity Classes

Score: Medium and large ecosystems	Score: Small ecosystems	Biotic Integrity Class		Quality category in accordance with Directive 2000/60 EC
47-50	37-40	I	Excellent	High (I)
40-46	32-36	II	Good	Good (II)
8-39	22-31	III	Fair	Moderate (III)
19-27	16-21	IV	Poor	Poor (IV)
10-18	8-15	V	Very poor	Bad (V)

The operation of the hydrotechnical constructions causes sudden and frequent decreases in the water level, which lead to the mass loss of eggs and fry left on land. The most disastrous effect is found in species with a unitary mode of reproduction, in which the whole generation of that year can be compromised (most economically valuable species belong to this group). Therefore, for state institutions empowered in the protection and sustainable management of the fish stock, it is very important to assess, at fair value, the damage caused to aquatic biological resources by the construction of hydrotechnical/hydroenergetical complexes (Instruction on assessing the damage caused to fishery resources from the water bodies of the Republic of Moldova, 2003).

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7 Chapter

METHODOLOGY AND ECONOMIC VALUATION OF ECOSYSTEM SERVICES AND THEIR LOSSES

Cazanteva Olga
International Environmental Association of River Keepers Eco-Tiras

INTRODUCTION

Hydropower plants (HPPs), dams and reservoirs are usually built to generate electricity and to store water for compensating river flow fluctuations, thereby providing a measure of human control over water resources, or to raise the level of water upstream of the HPP in order to either increase hydraulic head or to enable diversion of water into a canal to mitigate flooding, as well as to supply water for agriculture, industries, municipalities, etc.

However, the effectiveness of dam technology in delivering these services is currently being hotly debated, especially from ecological points of view due to their biological effects. The magnitude and extent of hydropower, associated dams and reservoirs construction result in water diversion, exploitation of groundwater aquifers, stream channelization and inter-basin water transfer. Overall, these factors are often capable to cause hydrological alterations having global-scale environmental effects. Hydrological alteration, which can be defined as any anthropogenic disruption in the magnitude or timing of natural river flows and the fragmentation of river channels caused by dams and reservoirs, can profoundly affect biological populations over a substantial area.

Generally, the HydroEcoNex project aims to analyze the effects of hydropower on the ecological status of aquatic ecosystems and ecosystem services they provide. Here, status expresses the quality of the structure and functioning of aquatic ecosystems; ecosystem services refer to the benefits that people obtain from them, expressed as their direct and indirect contributions to human well-being. On this background, this chapter aims to present a methodology for the economic evaluation (EV) of services provided by aquatic ecosystems. The proposed methodology tries to address ecosystem services at different scales, to present effects on them of main stressors under study and thus to support EV implementing in the integrated river basin management (IRBM).

7.1. Methodology

7.1.1 General provisions

Generally, the value of an ecosystem service in monetary terms depends on who is the potential payer, as well as on a number of other factors, including whether it will be possible to use this service on a sustainable basis in the long term. Within any scheme involving the application of market mechanisms to ecosystem services one of the main tasks is to determine their ‘true’ value. There is no universal method for this, and in practice a number of approaches are used. Specific information on the various valuation methods is contained in different documents (GEF, 2018; Secretariat of the Convention on Biological Diversity, 2007; The Economics of Ecosystems and Biodiversity, 2010).

In the development of the HydroEcoNex project's methodology for economic valuation of ecosystems service two approaches were combined.

The first approach included selection of a conceptual framework for assessing and valuing ecosystem services of water ecosystems for specific applications in the Black Sea basin, based on the literature review and on-going initiatives in Europe (DEFRA, 2007; GEF, 2018; Grizzetti et al., 2015)

The second approach included an experience, knowledge and needs of the project partners to select the relevant ecosystem services and target methodology.

Thus, the research described in the presented document can be considered as a learning process where previous experience and information available through literature review on EV of ecosystem service had to be combined with the knowledge and expertise of the project partners. The integrated outcomes of these activities should base a methodology both to address the project's objectives and to be applicable in a wider practice.

Generally, any economic valuation is a resource-intensive activity, and significant expert´s knowledge is needed for its conducting. In cases where such knowledge and resources are limited, GEF Guidance recommends to use a "benefit transfer" method based on transferring available information from the studies already completed in another location and context (GEF, 2018). Benefit transfer method is also used when there is too little time available to conduct an original valuation study. Economic valuations in such situations are referred by GEF Guidance as "tier 1" projects (GEF, 2018). Valuation studies with more resources at hand, i.e. those which have adequate funds and time, are referred as "tier 2" projects that are based on more detailed and more comprehensive studies. In other words, depending on the available resources, EVs could differ, necessitating to conduct a rather "rough" screening of the ecosystems, or to prioritize some ecosystem services above others. Alternatively, the specific objectives of EV could make a necessity to concentrate on a very specific, localized ecosystem of high value (e.g., a biodiversity "hotspot"), or on particular pressure affecting any region or system.

This guidance, proceeding from its goal and potential users, considers mainly a "screening analysis", assessing the overall value of some ecosystem services in a transboundary river basin without conducting resource-intensive in-depth analyses. In most cases such a screening could likely be conducted, using the tier 1 methodology and mainly for communication and awareness raising purposes. However, because such "screening" also forms some basis for an in-depth analysis that follows "tier 2" methodology, an economic valuation based on a "hotspot analysis" was also used in this research. The in-depth analysis of very biodiversity rich and important ecosystems or areas (in particular, wetlands) was applied as well.

Based on analyses of the scope of ecosystems services assessment, Grizzetti et al. (2015) identified some requirements to the methodology of this process, which can be formulated as follows:

- define the ecosystem services relevant for aquatic ecosystems and water resource management;
- provide quantitative information on the benefits people obtain from nature, including economic value, with a focus on biophysical quantification and monetary valuation;
- be sufficiently simple and flexible (not site-specific) to be applied for analyses at different spatial scales and by different users;

- capture the effect of multiple stressors and scenarios on ecosystem services delivery;
- to be linked to valuation (cost-benefit analysis, trade-off analysis) and proves effective in communication with stakeholders involved in river basin management planning.

Based on the analysis of different methods of ecosystems services EV and taking into account the specific of this document, the proposed methodology uses mainly recommendations for a tier 1 project. This methodology entails the following steps:

- Setting the Scene: Determination of the spatial boundaries of the area to be studied, i.e. deciding on whether to include some areas and exclude others;
- Setting the Scene: Identification of ecosystems and ecosystem services present in the site to be studied/assessed;
- Setting the Scene: Determine the size of ecosystems present in the area under investigation;
- Identification of which ecosystem services can be accessed directly via market prices and which need a benefit transfer;
- Assess the values of provisioning services via local market prices;
- Assess the values of other ecosystem services using the simplified Benefit Function Transfer and other approaches;
- Summing up the values and determining the ecosystems total value.

Such so-named "screening analysis", based on tier1 methodology, in some cases will be supplemented by an in-depth analysis of very biodiversity rich and important ecosystems. Economic valuation of these areas follows the tier 2 methodology.

7.1.2 Setting the Scene

Setting spatial boundaries

The determination of spatial boundaries of areas to be studied and to decide whether to exclude some of them and include others should define the scope and scale of the assessment. This initial step in EV depends on its specific aims and objectives. In general, at this step, according to the GEF Guidance (GEF, 2018), the following, slightly modified, questions should be answered:

- Do you aim to assess the value of natural and undisturbed ecosystems in your project's area?
- Are significant urban agglomerations in the study area, which provide ecosystem services (e.g., recreation benefits)? If yes, they should be included in the valuation or treated separately.
- Are other areas that are very strongly affected by human activities (e.g., intensive agriculture)? If yes, they should be excluded or treated separately.
- What are relations with regard to size between natural ecosystems and heavily impacted areas, i.e. are the latter significant in the overall study (say more than 5 or 10%)?

As a result of this exercise, a map of the entire study area should be produced, clearly showing where its boundaries are located and which its parts are possibly to be excluded from an economic valuation. As alternative, a textual description detailing the decisions taken with regard to spatial boundaries will work equally well. Both a map and textual description can act as a basis for the whole analysis. In particular, Grizzetti et al. (2015) proposed a methodological framework for the ecosystem service assessment and economic valuation of European water resources. This framework includes three spatial scales: water body, catchment and the European one.

As an example *(Fig. 7.1)*, in the HydroEcoNex project, the water body scale is presented by the Dniester and Prut HPPs reservoirs, the catchment scale – by the Dniester and Prut river basins within the territories of Moldova and Ukraine, and the European scale – by the whole territory of the Project's activity, including the north-western Black Sea coast.

Identification of ecosystems and ecosystem services

At this step it is necessary to identify ecosystems that are located within spatial boundaries, which were set at step 1, and ecosystem services they provide. Generally, the water ecosystems and ecosystems service are those related to the water bodies covered by the WFD and relevant for a river basin management. A large variety of such services have been addressed under different projects and assessments; partially, they are discussed, for example, in (GEF, 2018; Grizzetti et al., 2015).

In this study the preference was given to the GEF Guidance *(Table 7.1)*.

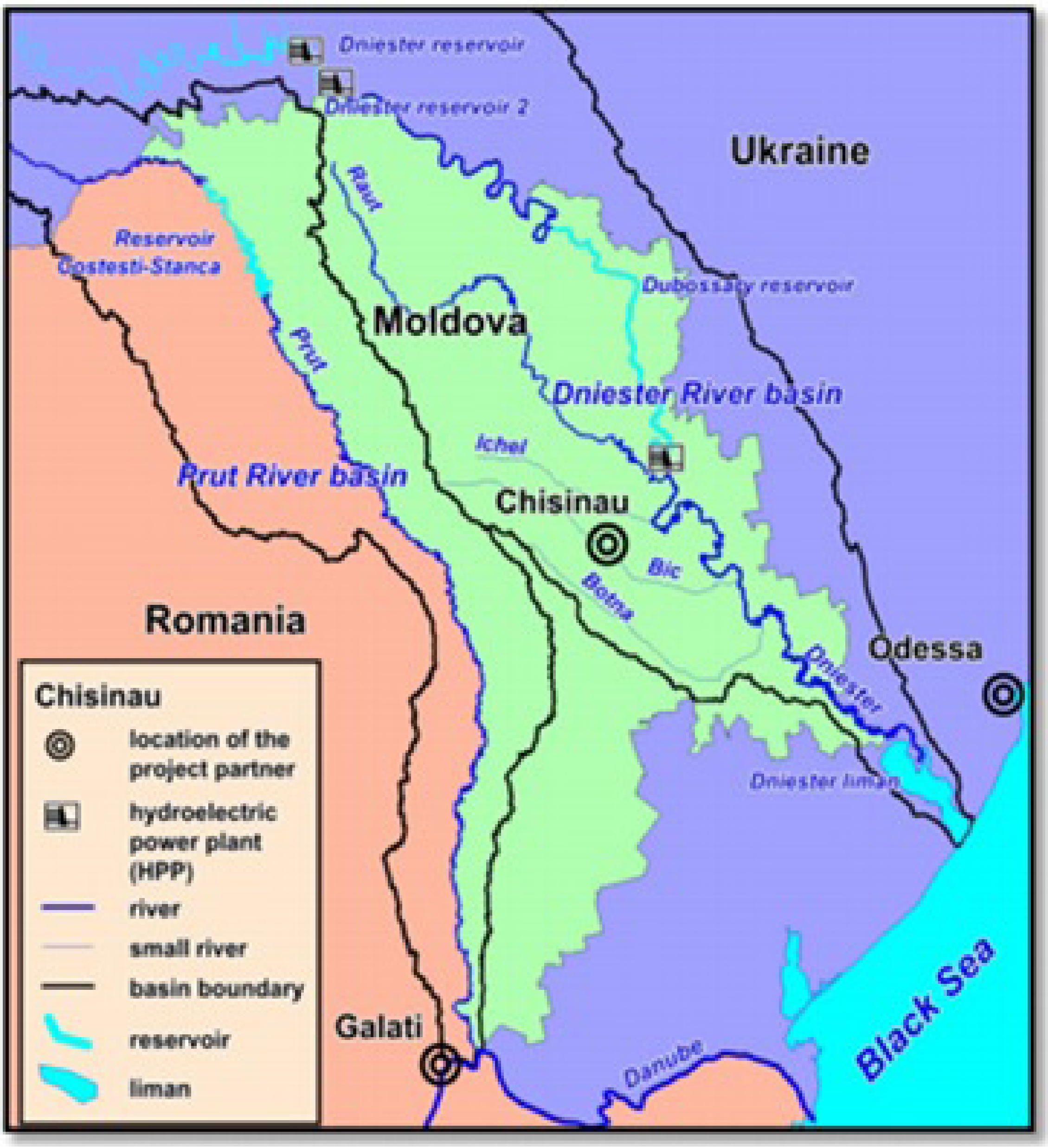

Figure 7.1. Setting the scene in the HydroEcoNex project

Table 7.1. Template of ecosystem services and freshwater ecosystems providing them (in green – ecosystem services covered under this publication). Adapted from GEF (2018)

Type of ecosystem service	Ecosystem services	Category of value	Provided by which ecosystem
Provisioning Services	Food • Fish • Aquaculture • Other product • Genetic and medical resources	Direct use	Rivers, lakes, inland wetlands
	Forestry: fiber, timber, fuel		Inland wetlands
	Water: drinking water, irrigation, cooling		Rivers, lakes
Regulating Services	Air quality regulation	Indirect use	Inland wetlands
	Climate regulation (Carbon sequestration)		
	Moderation of extreme events (e.g. floods)		
	Water treatment		
	Erosion prevention		
	Nutrient cycling and maintenance of soil fertility		Rivers, lakes, inland wetlands
Habitat Services	Maintenance of life cycles of migratory species (nursery service for fish species)		
	Maintenance of biodiversity		
Cultural Services	Opportunities for tourism/recreation	Direct use	
	Aesthetic inspiration	Non-use	
	Spiritual experience		
	Education		

Determination of area and size of ecosystems to be valuated

Determining the area of ecosystems selected for economic valuation follows the previous steps. If no quantitative information is available for any ecosystem type in a studied area, the reliable estimates based on expert judgment can be used. Also, in the case when the scale of the economic valuation of ecosystem services is quite large, e.g. a river basin, the estimated territory can be subdivided into smaller sections. An example of such approach is given in *Fig. 7.2* and *Table 7.2*. Here, the Dniester River's floodplain from the Dniester hydropower complex (DHPC) to this river mouth was subdivided into seven parts, with their own sets (clusters) of ecosystems.

Figure 7.2. Breakdown of the Dniester floodplain into clusters to study the ecosystems and their services

Table 7.2 Area of ecosystem types (km²) in the Moldavian part of the Dniester River[1]

Ecosystem		Clusters						
	CHEN-Dubasari	Dubasari reservoir	Dubasari - Raut mouth	Raut mouth - Ichel mouth	Ichel mouth - Bic mouth	Bic mouth - Botna mouth	Botna mouth – Dniester Liman	**Total**
Aquatic	23,6	64,1	1.5	4,6	20,8	5,2	17,8	137,6
Lakes				0,1	0,3	0,5	4,9	5,8
Wetland	0,7	5,2			0,2	0,8	32,0	38,9
Forest	2,8	3,8	0,3	2,6	32,8	7,1	29,4	78,8
Grassland	25,9	13,8	3,1	22,9	95,3	46,2	135,2	342,4
Perennial	0,7	1,8	0,1	8,7	12,8	11,1	13,1	48,3
Arable							82,1	82,1
Localities	2,5	5,0		2,04	16,6	3,8	21,6	51,6
Total:	56,1	93,7	5,0	40,9	178,8	74,7	336,3	785,6

[1]According to Ecosystem types of Europe – version 3.1. Available at: https://www.eea.europa.eu/data-and-maps/data/ecosystem-types-of-europe-1

Distribution and fragmentation of natural ecosystems

Across the world a variety of ecosystems are spread, each with distinctive interacting characteristics and components. They range from small (e.g., a freshwater pond) to global (e.g., a taiga biome). While, the distribution of large-scale ecosystems (biomes) is determined by climate, the distribution of small-scale undisturbed ecosystems is determined mainly by a local climate. Any changes in this climate in common with any anthropogenic intervention lead to their transformation.

Once the scene of economic valuation is set, the following steps should include a quantification and valuation of ecosystem services and their losses under observed impacts. According to Fahrig (2003), the concept 'ecosystem loss' refers to the disappearance of an ecosystem or an assemblage of organisms and the physical environment in which they exchange energy and matter. As one indicator of an ecosystem's losses, there is considered a fragmentation of its initial distribution. Thus, the current condition of any territory is results of its exposure to long-term impacts of natural or anthropogenic loads that leads finally to transformation and fragmentation of its natural complexes and reducing their biological diversity and ecological stability as a whole. Therefore, any EV of ecosystem services should be preceded by the assessment of relevant ecosystems current distribution.

The assessment of fragmentation is an extremely important element in the economic valuation of ecosystems services because it identifies areas that are in need of protection and restoration. Already now numerous terrestrial and riverine habitats are becoming increasingly fragmented, which threatens the viability of the species and their ability to adapt, for example, to climate change (Secretariat of the Convention on Biological Diversity, 2010).

The fragmentation of ecosystems, combined with an increase in the area of disturbed lands, weakens the material-energy bonds between individual landscapes. The notion of fragmentation is best understood as certain subdivision of a formerly contiguous landscape into smaller units, thus reducing its continuity and interfering with species dispersal and migration, isolating the populations and disrupting the flow of individual plants and their genetic material across a landscape (Secretariat of the Convention on Biological Diversity, 2007, 2010). For example, Moldova lies in the zone of likely large-scale extinction of species under unfavorable conditions for adaptation: the excessive fragmentation of natural ecosystems and deformed hydrological regime of its main rivers, first of all the Dniester River, against the background of general flow instability (Corobov et al., 2014).

However, assessing the fragmentation is not only the assessment of the ecosystem's loss and vulnerability. It is also assessing the territorial distribution of all services provided by ecosystems.

Quantitatively, the degree of fragmentation is estimated, using various indices (e.g., McGarigal and Marks, 1994). In the Moldavian studies, for example in the latest (Cazanteva et al., 2019), as a quite informative index, the Coefficient of fragmentation (CF), calculated as a ratio of an ecosystem's perimeter to its area was used: the higher this ratio, the more pronounced the fragmentation. Concurrently, the ecosystems' average area and their number were also used.

7.2. Economic valuation of ecosystem services and their losses

7.2.1 Selection of methodology

Economic Valuation as a common approach, taken from the field of environmental economics (Plottu, Plottu, 2007), aims to create a single monetary metric combining all activities within an area, and to express the level of each activity in a common monetary measure, e.g., US dollar. As such, it is a useful tool for exploring what types of values each ecosystem service provides and, accordingly, it helps to determine a cost required to conserve these values (DEFRA, 2007).

Differences in the problems to be studied require differentiation of approaches to their solution. Any ecosystem is the interacting and dynamic system consisting of biotic and abiotic elements, which are not in a static composition. In every ecosystem the animals, plants, micro-organisms, mineral resources, climatic and other factors interact. The provision by an ecosystem of ecological services is a result of specific interactions of these components, and only a healthy ecosystem can provide the full set of its potential services. Thus, the task of economic valuation is not only to assess a potential value of these services, but mainly to assess their real value resulting from certain losses caused by different impacts.

The value of an ecosystem service in monetary terms depends also on who is the potential payer, as well as on a number of other factors, including whether it will be possible to use this service on a sustainable basis in the long term. Within any scheme involving the application of market mechanisms to ecosystem services, one of the main tasks is to determine their 'true' value.

There is no universal method for this, and in practice a number of approaches are used. Relevant information on the various valuation methods is contained in different documents (GEF, 2018; Secretariat of the Convention on Biological Diversity, 2007; The Economics of Ecosystems and Biodiversity, 2010).

Although most ecosystem services are not traded on markets, there are some that are. In particular, the latter may include products that are derived directly from the ecosystem (e.g., food), or some other services, e.g. tourism. If products are directly traded on markets, their value is best assessed using the local market prices. Although they significantly differ from country to country or from region to region, it is relatively easy to obtain and provide them as a local value as well. In particular, in a screening analysis (tier 1) methodology, the ecosystem services traded on local/national markets are not accessed via a benefit transfer, but using local market prices. For provisioning services, it is highly recommended to use such prices; for other services (e.g., tourism and recreation) this approach is optional. Moreover, market prices are relatively easy to obtain, and they provide fairly exact estimates of ecosystem services value for a local community. That is why, it is strongly recommended to use local market prices as much as possible in the economical valuation of ecosystem services.

Concerning the selection of methodology of EV of freshwater ecosystems, GEF (2018) proposes the following methodology *(Table 7.3)*.

7.2.2 Economic valuation of provisioning services

Water

The total cost of providing water includes its full economic cost and environmental externalities, associated with public health and ecosystem maintenance. In this duality, the first component consists from water supply cost, e.g., operating and maintenance expenditures and capital charges. In turn, ecosystems maintenance depends on water availability. The most difficult element in EV of water services is to determinate their market price, usually taken as average price for 1 m^3 of drinking still water.

Table 7.3 Methodologies that can be used for economic valuation (EV) of freshwater ecosystems services (source: adapted from GEF (2018))

<table>
<tr><th>Type of ES</th><th>Ecosystem service</th><th>Category of use</th><th>Methodology for EV</th></tr>
<tr><td>Provisioning Services</td><td>Fish
Aquaculture
Other products
Timber, fuel
Water (drinking, irrigation)</td><td>Direct use</td><td>Market prices</td></tr>
<tr><td>Regulating Services</td><td rowspan="2">Carbon sequestration
Moderation of extreme events
Water/Sewage treatment
Erosion prevention
Nursery services
Maintenance of life cycles of migratory species
Maintenance of genetic diversity</td><td rowspan="2">Indirect use</td><td rowspan="2">Benefit transfer</td></tr>
<tr><td>Habitat Services</td></tr>
<tr><td rowspan="2">Cultural Services</td><td rowspan="2">Tourism
Recreation
Aesthetic information
Spiritual experience
Education</td><td>Direct use</td><td>Market prices, Benefit transfer</td></tr>
<tr><td>Non-use</td><td>Benefit transfer</td></tr>
</table>

Such approach, as useful for EV of impacts on water resources, was applied to evaluate losses of the Dniester River provisioning services due to the Dniester Hydropower Complex (DHPC) operation. The estimations were based on comparing the streamflow volume *(Q)* at hydrological posts Zalishchyky, located upstream DHPC, and Mohyliv-Podilskyi and Bender – downstream in periods before (1951-1980) and after (1991-2015) DHPC construction *(Table 7.4)*.

Table 7.4. The Dniester annual runoff (km^3) before and after DHPC construction

Post	Periods		Change
	1951-1980	1991-2015	
Zalishchyky	7.03	7.28	0.25
Mohyliv	8.89	8.33	-0.56
Bender	10.22	9.15	-1.07

Q decrease downstream the DHPC in 1991-2015, against its increase upstream, indicates its undoubted impact that results in annual economic losses of $30 million in Mohyliv and above twice more – in Bender (at a water price of $25/m^3).

Fishery

The long-term dynamics of the volumes of commercial fisheries in the Dniester River indicates its significant reduction *(Fig. 7.3)*. This reduction is undoubtedly associated with HPPs construction: the first sharp reduction took place in the 1950s and was caused by the Dubasari HPP construction; the second reduction, occurred in the 1990s, was due to the commissioning of the Dniester hydropower complex.

Along with a general decrease of fish stocks, the stock of commercially valuable species is especially significant *(Table 7.5)*.

For EV of the fishery losses, two approaches have been used:

1. cost of direct losses: based on the world price of freshwater fish ($2.35/kg in 2019), the annual losses in Dnies-

ter part, e.g. from Rybnitsa to Palanca, were more than $172 thousand;

2. costs of maintaining the fish habitat: the cost of 150.2 tons of various fish species fries, launched in 1998-2018 in Dubasari reservoir for maintaining its fish stock, amounted ~360,4 USD; this figure can be considered as an equivalent of EV of fish losses.

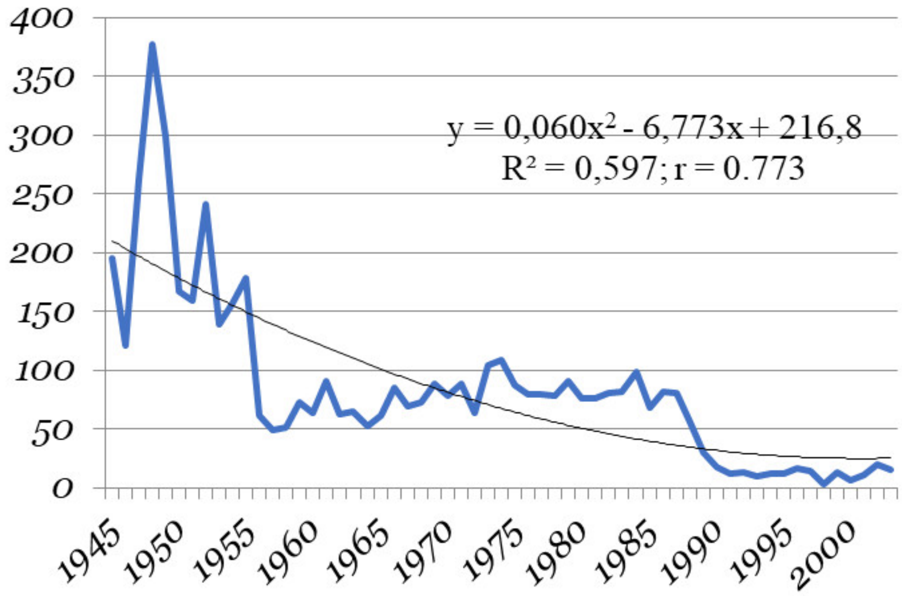

Figure 7.3. The second-degree polynomial trends of fish catches in the Dniester River

Table 7.5 Dniester's catches of various values fish in different time periods, tons

Statistics	Fish values			Total
	Mean	Low	Other	
		1946 -1953		
Mean	83.1	34.8	107.8	227.1
Max	174.0	93.1	191.0	376.8
Min	14.0	10.1	28.7	120.8
		1954 - 1983		
Mean	10.7	58.0	14.8	83.5
Max	43.8	89.4	75.7	178.3
Min	2.2	27.4	0.0	49.5
		1984-2005		
Mean	2.1	28.4	1.1	31.7
Max	11.0	84.1	3.8	98.5
Min	0.0	2.9	0.0	2.9

Forestry

The calculation of the current (annual) economic value (Ri) for forest ecosystems is carried out according to the following equation (Shchegolev et al., 2016).

After recalculating these estimates for the whole Lower Dniester forests area, the total economic value of their ecosystems provisioning services was amounted about 25.1 million MDL (~ 1.5 million USD at the national currency rate of 17.2 lei for 1 USD, or an average of 162 USD per ha). At the same time, significant territorial differences are observed due to the uneven distribution and qualitative composition of forests across this area.

Grass ecosystems

The calculations results showed that grass ecosystem services value in the Lower Dniester amounts to about 17.9 million MDL that is equivalent to about 1.05 million USD (at the currency rate of 17 lei for one USD), or on average 231 USD per ha. At the same time, due to the uneven distribution of grass ecosystems over this area, the significant territorial differences in their values are observed. Presented in mapping units they vary spatially from six to more than 30 thousand USD. The grass ecosystems with the highest provisioning services value are located in the north-western and south parts of this area, primarily due to the significant plots of high-quality grass communities still surviving here.

7.2.3. Economic valuation of regulating ecosystems services

Economic valuation of carbon deposit services

The HydroEcoNex project, due to its goals and objectives, examined the regulating services that to one degree or another relate to climate change and river streamflow.

Carbon deposit by the Low Dniester forest ecosystems: annual CO_2 accumulation for Moldova's main forest-forming species (oak, poplar, white acacia and other species) is 7.7, 10.7, 8.4 and 4.1 ton/ha, respectively. In March 2020 an average price of CO_2 allowance was 24.1 EUR. Based on forest species composition and area that each occupies in the Lower Dniester, the resulting current EV of their annual carbon deposit service is 1.53 million USD, varying across the territory from <5 to 105 thousand USD.

Carbon deposit by swamp ecosystems: valuation of the annual carbon dioxide absorption by swamp ecosystems is determined by the same equations that were used for forest ecosystems. However, in this case the absorption of CO2 by these ecosystems equals 0.705 ton/year (TCP, 2011). Economic value of CO2 deposit service of the very limited swamp ecosystems in the Lower Dniester amounts 25,000 USD (on average – 21.5 USD per ha), varying by cartographic units from less than 0.5 to 7.5 USD thousand.

Economic valuation of the assimilation potential of water-related forest ecosystems

The economic valuation of main forest species assimilation potential is based on estimation of maximum content of pollutants in their phytomass. In particular, the economic value of the assimilation potential (*Eap*) of water related forest ecosystems is calculated as the sum of corresponding estimates for individual pollutants (fluorine compounds, sulfur dioxide, nitrogen oxides, hydrocarbons, etc.). The value of an assimilation potential of the Lower Dniester Forest ecosystems was obtained. It is about 28.2 million lei that is equivalent to ~1.7 million USD, or 182 USD per ha on average. However, significant territorial differences are observed due to the uneven distribution of different forest species with their differing level of the maximum possible pollutants content.

Economic valuation of the sorption function of wetlands

As the sediments, excess nutrients and chemicals flow off of the land, the wetlands filter them before they reach open water. Nutrients are stored and absorbed by plants and microorganisms. Sediments are settling at the bottom after reaching an area with slow water flow. Additionally, CO_2 and other greenhouse gases are stored in wetland sinks instead of being released into the atmosphere. Economic valuation of the sorption (water-cleaning) function of swamps is based on a comparison of the filtering ability of their ecosystems with the filtering capacity of an industrial treatment plant. Based on the swamps area of the Lower Dniester Ramsar site, the economic value of their absorption services is about 107 USD or 91 USD per ha on average. However, this value varies from 1,000 to more than 30,000 USD.

Water protection and water regulation services

This service consists in equalizing seasonal fluctuations in a river runoff, preventing its sharp reductions, to reduce floods intensity by redirecting a surface runoff into ground. So, depending on a sloping forest area in the Lower Dniester, the underground water accumulation here is ~485,000 m^3. With a payment for water for industrial enterprises of ~32 MDL/m^3, the total economic effect of such accumulation is about 11.9 million MDL.

7.2.4. Economic valuation of habitat services

Habitats provide everything that flora or fauna need to survive. In this framework, each ecosystem provides different habitats that can be essential for a species' lifecycle, while the **habitat services** highlight their importance to provide such habitats both for local and migratory species. Along with these tasks, the habitats promote to maintenance of bio- diversity within species populations. EV of biodiversity is usually carried out, using the replacement cost method.

The water-regulating DHPC has changed the volume and seasonal distribution of the Dniester's streamflow, often causing its delta draining. Such destructive impact on main representatives of the delta's natural ecosystems has resulted in a catastrophic reduction in populations (by 70-99%) of almost 80% of its fauna.

So, a glossy ibis *(Plegadis falcinellus)*, which is listed in the Red Books of Moldova and Ukraine, was the most widespread bird in the Dniester delta, where 2,500-3,000 of its adult individuals' nest steadily in 1970-1982. However, already in 1988-2002, the number of breeders decreased manyfold here, ranging from 100-350 adults' individuals; the decrease was continuing further and in 2010-2015 this bird has almost disappeared from the delta as a breeding species.

According to the Ukrainian legislation the penalty for the death of one glossy ibis is about 434 USD. Considering this fine as a kind of compensation for the loss of this environmental service, the economic value of glossy ibis disappearance due to hydropower adverse impacts on the Dniester delta can be estimated of 1.0-1.3 million USD.

7.2.5. Economic valuation of cultural ecosystem services

Cultural ecosystem services encompass the „non-material benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences" (Millennium Ecosystem Assessment, 2005). Consideration of the ecosystems' cultural benefit and values is a distinguishing feature of their service-based approaches to natural resource management. As a class of services, the cultural ecosystem services represent a concept that allow understanding the ecosystems in terms of their life- enriching and life-affirming contributions to human well-being. They also give an example of an approach that is more generally embraced as an important component in the work of environmental managers and planners (Fish et al., 2016).

Generally, cultural ecosystem services include both some measurable services, for example, health outcomes or direct economic benefits, as well as other services that are more intangible and experiential, such as spiritual experiences, education, and aesthetics. However, approaches to understanding and measuring the cultural ecosystem services remain the subject of ongoing debate. For a correct economic valuation of past and future losses (in the absence of the necessary preventive measures), the procedure of bringing multiple damages to the same time interval (discounting) is used.

Recreation service in the Dniester floodplain. Changes in a river flow and temperature-humidity conditions in its basin, caused by climate change and DHPC operation, affected attractiveness of recreational areas and decreased income of the Dniester floodplain inhabitants. In the 1990s, in a 1 km-wide river strip downstream the DHPC about 6,000 families lived, from which every tenth took summer residents, having potential income ~ 5,000 MDL (based on estimations of Bruma and Zubarev (1998)). On total, EV of this ecosystem service could be 26,000 MDL a year, or 5,600 USD at the 1997 exchange rate. At present, a gradual accumulation of losses due decrease of this service can potentially amount to ~32,000 USD.

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8

Chapter

METHODOLOGY FOR ASSESSING A CLIMATE CHANGE FACTOR IN THE HYDROPOWER IMPACTS RESEARCH

Corobov Roman, Sirodoev Ghenadii

International Environmental Association of River Keepers Eco-Tiras

INTRODUCTION

Influence of hydropower on surface water resources has acquired new aspects on the background of changing climate due to the undoubted impact of the latter on the rivers' adaptive capacity. This reality introduces additional dimensions in the concept of relationships between renewable energy and water resources (IHA, 2019). Transformation of a river hydrological cycle, caused by climate change, leads to a variety of impacts and risks on the water and riverine ecosystems through the complex interaction of climatic and non-climatic factors.

In turn, modification of the rivers' hydrological regime increases severity of issues associated with water security in their basins (Laušević et al., 2016; MacQuarrie, Wolf, 2013; WaterAid, 2012; UNU, 2013). This makes current water challenges, which all countries face, more severe because water is that sector where most climatic impacts are especially felt and where climate resilience must be developed first of all. The recent reports on this issue (WWF, AB InBev, 2019) highlights a central role of healthy rivers in adapting to climate change and shows that freshwater conservation must be at the heart of adaptation agendas and efforts. Somewhat earlier, this conclusion was emphasized in other works (UNECE, 2015; Yan, Pottinger, 2013).

Changing temperature and precipitation are likely to strengthen problems in water supplies and demands, affecting human well-being, economy, and especially, ecosystems and their services (WB, 2016). Moreover, some climate change consequences, such as ecosystems loss, may be long-lasting or even irreversible. Higher temperatures lead to more evapotranspiration, thus decreasing a surface runoff, while changes in quantity and timing of precipitation affect the viability of agricultural operations (Fischer et al., 2002), increasing the demands of water for irrigation or directly impacting flow-rates in freshwater and riverine ecosystems. That is why, the scarcity of freshwater in new climatic conditions is increasingly perceived as a global systemic risk, and its essence is considered as a global geographical and temporal mismatch between the needs for water and its availability (WaterAid, 2012). As a challenges should be also considered the fact that large spatial and temporal variability in water demands and its availability leads to water scarcity in different ways in different regions and time periods.

Moreover, global warming is accompanied by an increase of climatic extremes, and in temperate continental climates the heavy rains create favorable conditions for extreme floods (Santato et al., 2013). On the other hand, along with excess rainfall periods, the number of extremely dry seasons is also increasing. Consequently, meeting new water needs and protecting ecosystems by making them sustainable are the most difficult, but also the most important challenges of this century (Mekonnen, Hoekstra, 2016).

Due to great contribution to providing a wide range of public goods and services, the freshwater, in general, and rivers, in particular, occupy a special place in the assessments of ecosystems resilience to climate change (Yan, Pottinger, 2013). Since most rivers are within watersheds, which already are stressed by human activities, the observed changes in climate will add to or magnify present risks through its potential to alter air temperature, precipitation and runoff patterns, and correspondingly disrupting biological communities and their ecological linkages. As a result, many communities will face shrinks in their water supplies with dramatic consequences through threatening public health, weakening economies and decreasing quality of life. Transformation of a hydrological cycle leads to a variety of impacts and risks caused by interaction of climatic and non-climatic stimuli with their responses to water resources management. Coordination efforts between water, energy and environment sectors are especially challenging due to numerous evidences that climate change is strengthening.

At last, climate change introduces new dimensions in the environment and water ecosystems relationships (Pequegnat, 2009; UNECE, 2015) because it results in the "death" of the so called conception of *stationarity*. This conception assumes that climate, and dependent on it hydrology, are predictable, and as such their future can be based on past historical data; accordingly, the current environment and water ecosystems relationships can be reliable in the future. However, in reality, the direction and magnitude of changes in climatic elements will inevitably be different, with unknown impacts on quantity and quality of water, and consequent effects on aquatic ecosystems.

According to Abell et al. (2002), the adverse global warming effects on freshwater ecosystems, which should be taken in conducting the biological assessments and developing the biodiversity visions for ecosystems conservation, are:

- Climate change may alter the composition of water and riparian vegetation.
- Distributions of species will change since some of them can invade the higher latitude habitats or can disappear from the limits of their lower latitude distribution due warming of freshwater habitats. Projected increases in air temperature will be transferred, with local modifications, to ground waters, resulting in elevated temperatures and reduced oxygen concentrations.
- In a warmer and drier climate many streams fed by runoff might become intermittent because of their high flow variability; when streams are drying, the mobile organisms are concentrated and the biotic interactions intensify. Small, shallow habitats will first express effects of changed precipitation, and of the greatest concern are habitats now occupied by threatened and endangered species.
- The cyclic swelling and drying of rivers directly affects aquatic organisms in terms of basic habitat availability, oxygen levels, turbidity, and food resources. Some habitats (swamps, lagoons, floodplain pools), which are considered marginal in dry seasons, become isolated from the main river channel and can dry up. The availability of marginal habitats during wet seasons and the severity of conditions in those habitats during dry seasons are equally dependent on the hydrologic regime, which in turn is dependent on precipitation.

Changes in the physical habitats and food bases deeply impact biological communities from a river source to its mouth. With their banks, floodplains, pits and fords, the rivers are among the richest ecological systems due to their biological diversity and, as such, they are subjected to serious destruction by climate change. A river is also an agent that brings most of these impacts to nature and societies. Although water passes through the global hydrological cycle, it is nonetheless a locally variable natural resource, and vulnerabilities, associated with water hazards, such as floods and droughts, vary between regions, depending on local, often non-climatic anthropogenic drivers. Despite some common features, every river basin has its own particular features, requiring their careful thorough study and consideration, especially in the process of a river flow transboundary monitoring (Pegram et al., 2013). The monitoring of shortcomings caused but climate change is especially important when to consider the compounding impact of climate change with detrimental impacts of hydropower (Casale et al., 2020; Smith et al., 2017). The complexity of coordination increases substantially in transboundary river basins where these impacts spread from one country to another, and trade-offs and externalities may cause frictions between riparian countries. Hereof, a basinwide approach, used in this chapter for considering the climate change issues in hydropower impacts of water ecosystems, is one of the principal dimensions in river basin management.

Climatic definitions and parameters

The World Meteorological Organization (WMO) in its Technical Regulations (WMO, 2017) recommends using the following definitions in climate description:

Average. The mean of monthly values of climatological data over any specified period of time, not necessarily starting in a year ending with the digit; in this case these averages are referred to as "provisional normals".

Element. An aspect of climate, which can be statistically described, for example, air temperature or precipitation.

Parameter. A statistical descriptor of a climate element, which is commonly the arithmetic mean, but can also include values such as the standard deviation, percentile points, number of extreme values, etc.

WMO defines ***three categories*** *of* climatological surface parameters: principal, secondary and other parameters.

Principal, or the most important parameters include monthly mean values of maximum, minimum and daily mean temperatures *(°C)*, and precipitation total *(mm)*. Air temperature is the basic physical factor that affects many natural processes and human activities. Warmer temperatures alter precipitation and runoff patterns, affecting the availability and abundance of aquatic ecosystems and their services as well as leading to a wide range of other impacts, including changes in species' geographic distribution, the timing of their life cycle events, etc. Trends in air temperature and precipitation can also increase the risk of severe weather and hydrological events, such as heat waves or intense floods. Understanding of these trends is important for refining future climate projections in terms of the climate sensitive environment and ecosystems.

The other principal parameters include number of days with precipitation ≥ 1 mm, mean value of sea-level pressure, mean vapor pressure and total number of sunshine hours. However, their use as well as the use of the *secondary* and *other* climatological parameters depends on the available observation data and the tasks to be addressed.

Climate, in its narrow sense, is usually defined as the average weather, or more rigorously — as the statistical description of key climatic elements in terms of their means and variability over a certain period of time (IPCC, 2018b). In particular, WMO (WMO, 2017) defines the *climatological standard normals* as averages of climatological data computed for the following consecutive periods of 30 years, e.g., 1 January 1981 — 31 December 2010, 1 January 1991 —31 December 2020, and so forth, updated every ten years. So, the period from 1961 to 1990 has been retained as a *standard reference (baseline) period* for long-term climate change assessments (WMO, 2017); the latest Assessment Reports of the Intergovernmental Panel on Climate Change (IPCC) used such approach in projecting the likely future climate for two time horizons: 2021-2050 and 2071-2100 (IPCC, 2013, 2018a).

Based on this definition the two thirty-years (1961-1990 and 1991-2018) periods were compared to define changes in the Dniester and Prut basin climates (Corobov et al., 2019; 2021a). These periods reflect, respectively, the relatively “stationary” regional climate of the second part of the 20th century and the climate of intensive global warming that was observing over the last three decades. Some objective “shortening” of the second period (28 years), caused by the timing of these studies, can be neglected.

The selection of the correct period of averaging is very important since its duration is one of potential sources of uncertainty and bias in the monitoring results (Mohammed, Scholz, 2019). This moment is important not only when choosing a “baseline” time period, from which the potential climate change projections are estimated, but also in identifying any change in the current climate. Sometimes, in a number of works, including some of those, which will be cited below, the choice of time periods for averaging has often been governed by availability of observations data.

According to the WMO Regulations (WMO, 2017, p. 1), in this kind of research, one should use a term ‘*climate normals*’ that serve only “…as a benchmark against which recent or current observations can be compared”. Such normals are also used as a prediction of the conditions most likely to be experienced in a given location.

8.1. Content of climate change research

The specific of tasks to be solved in the assessment of climate change influence on water ecosystems determines the choice of a corresponding methodic.

In our opinion, the **methods** to assess changes in climate should include the main following components:

1. Study of time trends in historical data.
2. *Descriptive analysis* to describe and compare the basic features of temperature-humidity conditions in climatic periods under comparison. The ***descriptive statistics***, at least, should include annual and seasonal reference normals and standard deviations *(Sd)* of mean *(Tmean)*, maximum *(Tmax)* and minimum *(Tmin)* air temperatures, as well as analogous statistics for precipitation totals *(P)*.
3. Assessment of *statistical significance* of the observed differences between estimated statistics for the compared periods, considered as a sound evidence of presence/absence of reliable changes in the river basin climate.
4. Aassessment of the ***likely future*** climate.

Practically all statistical analyses can be performed, using appropriate tools provided by the *Microsoft Excel*. The more powerful software, for example, Statgraphics (2014) is necessary for estimating the statistical significance of calculation results.

Trends analysis in air temperature and precipitation

The time trends in climatic elements provide useful information for understanding the changes in climate, which are associated with global warming. First of all, a trend analysis concerns air temperature and precipitation as two principal meteorological elements, which present the most important aspects of climate. In a number of the most recent publications, relevant to hydropower impact, the trends of these elements are analyzed either individually or in various combinations with river flow characteristics that are important for water resources management. So, Ge et al. (2019) assessed trends and variability in surface air temperature over the Indochina Peninsula; Jeganathan et al. (2019) – for one State of India. Zhao et al. (2019) explored linear trends to analyze mean and extreme precipitation under climate change within the Yellow River Basin (China), while Szwed (2019) – a precipitation variability in Poland. However, more often the temperature and precipitation trends are considered concurrently. We can name the works of Ay (2020) for the western Black Sea region and Corobov et al. (2019) — for the Dniester and Prut basins. A basinwide approach in trend analyses is more and more used in conjugate climatic and hydrological research (Aili et al., 2019; Luiz Silva et al., 2019; Mutti et al., 2020; Nikzad Tehrani et al., 2019; Rahimi et al., 2019).

As an example, in *Fig. 8.1* there are shown linear trends of annual air temperature and precipitation in the Prut basin (Corobov et al., 2021a). Here, the slope of trend lines characterizes the direction of change, a digit before **'*x*'** shows the value of temperature and precipitation change per a year; ***p***-value[1] characterizes statistical significance of estimated relationships. As one can see, the practically negligible and statistically *insignificant* trend of annual temperature (*p*-value much more than 0.10 that is permissible in such estimations) in the 1961-1990 period has changed by its sharp increase (about 0.8°C per decade) later. Moreover, this increase has a high level of reliability ($p \ll 0.001$), confirming statistically the undoubted warming of the Prut basin's climate. This conclusion is supported by the sharply increased Coefficient of determination *(R^2)*[2]: in the last three decades the linear trends of mean annual temperature explain 53.5% of its interannual variability, unlike to 0.02% in 1961-1990.

With regard to precipitation, if in 1961-1990 in the Prut basin a slight decrease (about 2 mm/year) of annual precipitation was observed, then due to global warming this negative trend slightly weakened (to less than 1 mm/year). However, in both periods the observed trends are not statistically significant in order to be taken into account ($p > 0.10$).

Statistical comparison of climatic elements change

The statement about the reliability of change in any climatic element values is valid only when this change is confirmed statistically. To compare whether or not the differences between these values in two compared periods are statistically significant, the Sample Comparison procedure that runs a t-test is used (Statgraphics, 2014). Usually, the comparison is carried out for two samples' averages and their standard deviation *(Sd)*. As an example, in *Table 8.1* there are shown results of such analysis for climate change significance in the Prut basin.

[1] *P*-value is the probability of obtaining results at least as extreme as the observed results of a statistical hypothesis test, assuming that the null hypothesis is correct. A smaller p-value means that there is stronger evidence in favor of the alternative hypothesis. *See, e.g., https://www.simplypsychology.org/p-value.html*

[2] The Coefficient of determination *(R^2)* is a statistical measurement that examines how differences in one variable can be explained (in %) by the difference in a second variable, when predicting the outcome of a given event. *See,e.g.:* https://www.investopedia.com/terms/c/coefficient-of-determination.asp

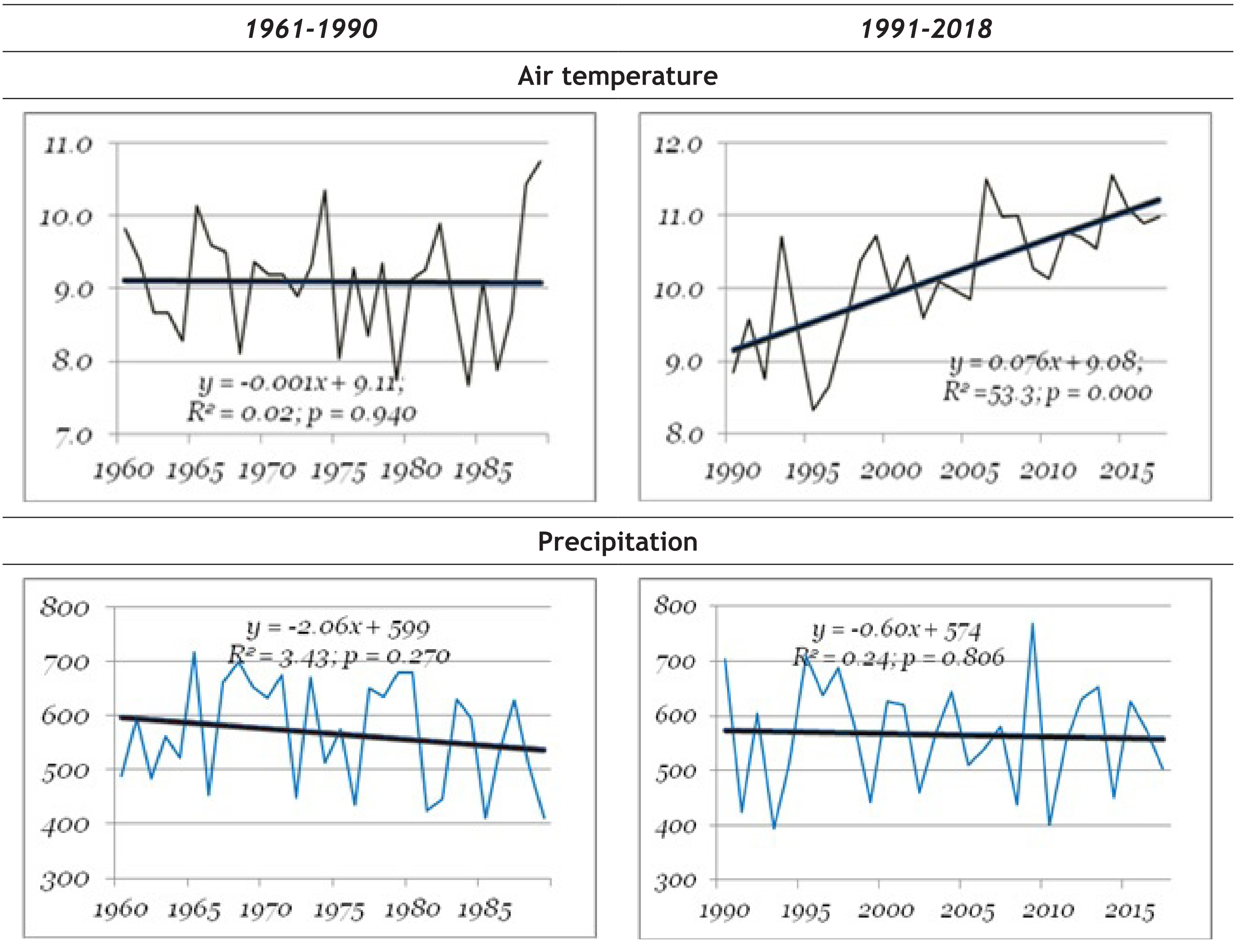

Figure 8.1. Linear trends of annual mean temperature and precipitation totals in the Prut River basins in two climatic periods

Table 8.1. Statistical comparison of mean temperature and precipitation in the Prut basin

Season	Average, °C				Standard deviation, °C			
	1961-1990	1991-2018	Difference	p-value	1961-1990	1991-2018	Difference	p-value
Mean annual temperature								
Winter	-2.34	-1.35	*0.99*	***0.034***	*1.91*	*1.52*	*-0.39*	*0.235*
Spring	9.27	10.41	*1.14*	***0.002***	*1.46*	*1.12*	*-0.34*	*0.164*
Summer	19.59	21.30	*1.71*	***0.000***	*0.79*	*1.05*	*0.26*	*0.143*
Autumn	9.85	10.34	*0.49*	***0.091***	*1.04*	*1.13*	*0.09*	*0.672*
Year	9.09	10.20	*1.11*	***0.001***	*0.79*	*0.85*	*0.06*	*0.697*
Precipitation								
Winter	108.5	93.4	*-15.1*	*0.200*	47.8	39.9	*-7.9*	*0.348*
Spring	135.8	133.3	*-2.5*	*0.830*	42.1	45.7	*3.6*	*0.662*
Summer	211.7	202.5	*-9.2*	*0.512*	47.0	58.5	*11.5*	*0.248*
Autumn	111.7	133.6	*21.9*	*0.166*	55.8	63.1	*-8.1*	*0.520*
Year	567.3	565.9	*-1.4*	*0.957*	98.2	100.8	*2.6*	*0.889*

Note: In bold italic there are shown statistically significant changes

As we can see, in 1991-2018 the annual averages of Tmean have increased, relative to the previous thirty years, by 1.11°C. In absolute terms, the maximal absolute increase of temperature was observed in summer, the minimal increase — in autumn. All increases are statistically significant with p-values <0.05 (except autumn) which means the observed increase of air temperature in 1991-2018, in comparison with 1961-1990, is reliable at 95.0% and higher confidence level. Nevertheless, the performed analysis does not give grounds to assert about statistically significant changes in temperature variability: all *p*-values in the Sd comparison are greater than 0.05.

The statistical comparison of precipitation's statistics supported conclusions of its trends analysis: the differences between precipitation averages and standard deviations in two periods are not statistically significant for all seasons, and their annual totals differ by only about 1.4 mm. The ony thing that deserves attention is an obvious increase in autumn precipitation with a decrease in other seasons.

Assessment of changes in the annual course of climatic elements

Global warming leads not only to a change in the temperature and precipitation climatological normals, but also to change in their annual course. The simplest way to assess of these kind changes is to plot the appropriate diagrams. Continuing to use as a case study the Prut basin (Corobov et al., 2021a), in *Fig. 8.2* the examples of such diagrams are shown.

In particular, though the Prut basin air temperature is changing, its annual course in two compared periods is preserved, with minimal values in December—February and maximal ones – in July-August (Fig. 8.2). At the same time, a temperature increase is visually observed practically in all months. On the other hand, although the annual precipitation totals remain almost unchanged, their certain redistribution by months is evident. So, the monthly precipitation maximum (82 mm), which was in 1961-1990 in June, now has decreased to 75 mm and is observed in July. The previous monthly precipitation minimum in October (27 mm) in last decades has disappeared as such, and the new minimum has shifted to February (28 mm). The other, although not so significant changes in precipitation patterns, are also observed in the rest months.

Climate change simulation

At present, the basinwide projections on likely future climate are usually based on the high-resolution (12.5 km) climate change scenarios, established for Europe within the EURO-CORDEX initiative (Jacobs et al., 2013). The EURO-CORDEX scenario simulations used a new approach to the identification of future greenhouse gas *(GG)* emissions — the so-called *Representative Concentration Pathways (RCPs)*. RCP scenarios assume some pathways to achieve certain radiative forcing on the climate system that can yield in the resulting changes of global climate according to the radiative forcing different scenarios. In *Fig. 8.3* there is shown disposition of the Dniester basin in the CORDEX greed. To improve the accuracy of modeling the expected climate change, the Dniester basin in its Moldavian part was divided into three parts (Corobov et al., 2014). The results of climate change modeling are shown in *Table 8.2* and *8.3*.

Air temperature	Precipitation

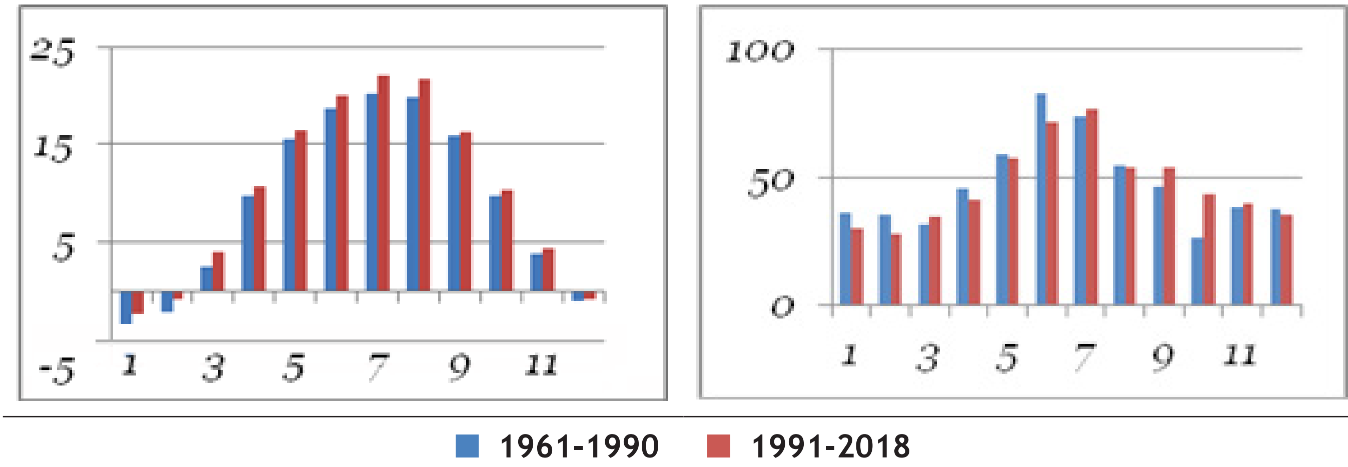

Figure 8.2. Average monthly air temperatures (°C) and precipitation in the Prut basin in two climatic periods

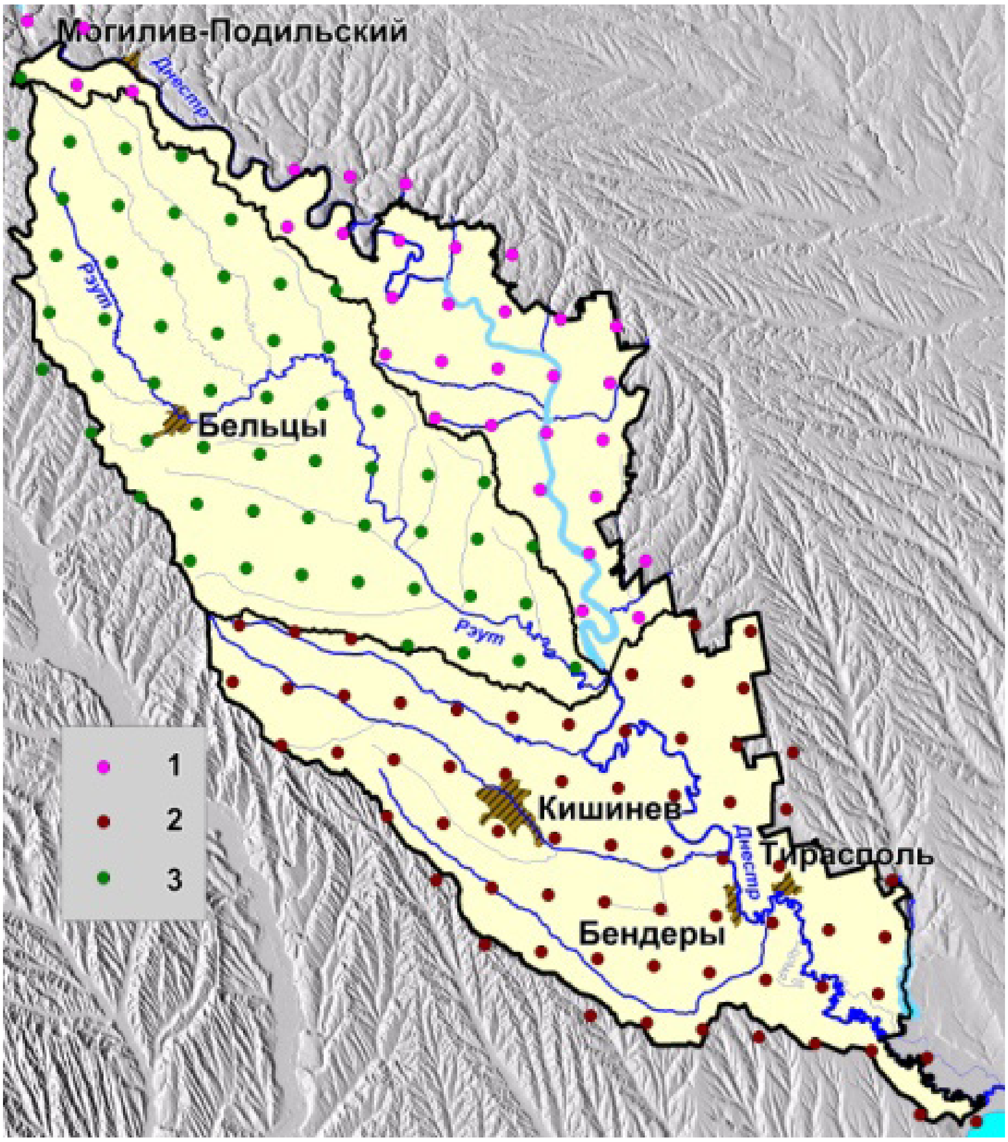

Figure 8.3. The Dniester basin in the CORDEX greed:
1 — Middle Dniester; 2 — Lower Dniester; 3 — Reut River basin (Corobov et al., 2014)

In the projections of *annual temperature* change *(Table 8.2)* the numerator shows results obtained by its direct modeling, the denominator — the results obtained by averaging the seasonal projections. Closeness of two estimations characterizes indirectly their representativeness. The simulation results show that in the Dniester basin, depending on the radiative forcing, the annual temperature may increase relatively to 1971-2000, adopted in EURO-CORDEX as the base period, from 0.2°C to 1.7°C by the 2050s, and from 0.3 to 4.4°C — by the end of this century.

With regard to precipitation *(Table 8.3)*, then in the first half of the century they are expected to slightly decrease (practically from 0 to 5%), depending on the radiative load; by the end of the century this decease be replaced by some increase (1-5%).

Table 8.2. Projections of mean air temperature change *(°C)* in comparison with 1971-2000 baseline climate in the Dniester basin according to the EURO-CORDEX scenario simulations

Season	1971-2000	Time horizon					
		2021-2050			***2071-2100***		
		Representative Concentration Pathways *(RCPs)*					
		RCP2.6	***RCP4.5***	***RCP8.5***	***RCP2.6***	***RCP4.5***	***RCP8.5***
Winter	-1.9	0.5	2.1	2.1	0.9	3.1	5.4
Spring	9.4	-0.1	1.3	1.6	0.3	2.5	3.9
Summer	19.6	0.2	1.7	1.5	-0.1	2.8	4.6
Autumn	9.0	0.2	1.1	1.5	0.0	2.1	3.8
Year	***9.0***	***0.2/0.1***	***1.6/1.5***	***1.7/1.6***	***0.3/0.2***	***2.6/2.6***	***4.4/4.4***

Note: The *RCP2.6, RCP4.5* and *RCP8.3* denote, correspondingly, weak, moderate and strong radiative forcings

Table 8.3. Projections of absolute *(Abs, mm)* and relative *(%)* change of precipitation as compared with *1971-2000* baseline climate in the Dniester basin

Season	1971-2000	Time horizon											
		2021-2050						***2071-2100***					
		Representative Concentration Partway *(RCPs)*											
		RCP2.6		*RCP4.5*		*RCP8.5*		*RCP2.6*		*RCP4.5*		*RCP8.5*	
		Abs	*%*	*Abs*	*%*	*Abs*	*%*	*Abs*	*%*	*Abs*	*%*	*Abs*	*%*
Winter	**91**	3	3.3	16	17.6	12	13.2	4	4.4	13	14.3	21	23.1
Spring	**130**	19	14.6	6	4.6	7	5.4	-45	-34.6	12	9.2	14	10.8
Summer	**218**	-44	-20.2	-28	-12.8	-21	-9.6	-6	-2.8	-17	-7.8	-35	-16.1
Autumn	**127**	-8	-6.3	2	1.6	-1	-0.8	11	8.7	21	12.4	5	3.9
Year	***565***	***-30***	***-5.3***	***-4***	***-0.1***	***-4***	***-0.1***	***-36***	***-6.4***	***29***	***5.1***	***5***	***0.9***

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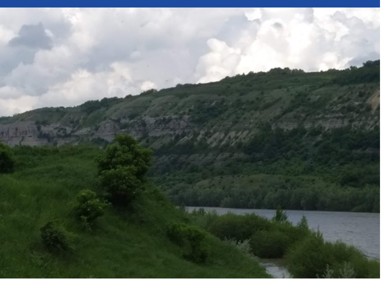

The editor of the material:
Institute of Zoology, Republic of Moldova
Center of Research of Hydrobiocenoses
and Ecotoxicology,
1, Academiei Str., Chisinau, MD 2028
Phone: +373 22 737509
E-mail: izoolasm@yahoo.com
Website: https://zoology.md

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