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ECONOMIC VALUATION IN THE MONITORING OF ECOSYSTEMS SERVICES
METHODICAL GUIDE

Project funded by EUROPEAN UNION

PROJECT BSB165 "*HydroEcoNex*"

"Creating a system of innovating transboundary monitoring of the transformation of the Black Sea rivers ecosystems under impact of hydropower development and climate change"

Economic valuation in the monitoring of ecosystem services Methodical guide

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ABBREVIATIONS AND ACRONYMS

CBD	Convention on Biological Diversity
CF	Coefficient of fragmentation
CRIDA	Climate Risk Informed Decision Analysis
DHPC	Dniester hydropower complex
ES	Ecosystem service
ETS	Emissions Trading System
EV	Economic valuation
HPP	Hydropower plant
GEF	Global Environment Facility
GG	Greenhouse gas
IHA	International Hydropower Association
INBO	International Network of Basin Organizations:
IBI	Index Biotic Integrity
IRBM	Integrated River Basin Management
IWRM	Integrated Water Resources Management
IUCN	International Union for Conservation of Nature
MSY	Maximum Sustainable Yield
MDL	*Moldovan leu*
MEA	Millennium Ecosystem Assessment
SDG	Sustainable Development Goals
TDS	Transboundary Diagnostic Analysis
TEEB	*The Economics of Ecosystems and Biodiversity*
TEV	*Total Economic Value*
WFD	EU *Water Framework Directive*
WSS	Water supply and sanitation services

FOREWORD

This document was prepared in the framework of Project BSB165 "HYDROECONEX" and presents a methodology of economic valuation of the hydropower and climate change impacts on the aquatic and water-related ecosystem services. The proposed methodology combines the intensive latest international and Moldova's national experience in this field, which was triggered by general recognition that ecosystems loss and fragmentation are the greatest worldwide threat to nature biodiversity and a primary cause of species extinction. Based on this recognition, a concept of economic valuation of ecosystem services and their principal provisions have been developed.

The discussed methodology depends both on the type of ecosystems service and spatial scale of valuation. In particular, there are separately considered the provisioning, regulating, habitat and cultural ecosystems services on two spatial scales – a catchment and water body. The catchment/basin scale offers an area for application of the ecosystem services concept in river basins management; it is also an appropriate scale to observe and quantify processes related to a water cycle or to implement monitoring and management plans. In this study, as such scale the Dniester River basin below the Dniester Hydropower Complex was selected. At a water body scale, the economic valuation is focused on the analysis of those ecosystem functions, which support specific ecosystem services, as well as on the study of their alteration under specific impacts and their different combinations. At a water body scale, the proposed methodology is demonstrated on the Ramsar site "Lower Dniester" due to an especial place that is given to wetlands. These ecosystems drastically reduced in their number and areas in many world regions because of climate warming, water recourses decrease and intensive human use, and now they are considered as 'hot-spots'. The carried out assessments were based on results of the relevant monitoring of physical and biochemical transformations in ecosystems that were identified in the framework of the HYDROECONEX project's other activities.

On the whole, considering an economic valuation as a prerequisite for making the optimal choices regarding the protection and conservation of ecosystems and their services, the presented study describes some approaches and provides a set of tools and recommendations for making informed decisions on this kind of problems.

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 (Rosenberg et al., 2000) and the fragmentation of river channels caused by dams and reservoirs, can profoundly affect biological populations over a substantial area.

Already in 1998, Postel S.L. (Postel 1998, p. 636) noted: "Large dams and river diversions have proven to be primary destroyers of aquatic habitat, contributing substantially to the destruction of fisheries, the extinction of species, and the overall loss of the ecosystem services on which the human economy depends. Their social and economic costs have also risen markedly over the past two decades". This statement is not surprising in view of the extent of hydrological development today. The environmental implications of the human appropriation of huge amounts of water are profound: decreasing amounts of fresh water are available to maintain ecological values and related ecosystem services. The conspicuous impacts of large–scale hydrological alteration, summarized in Rosenberg et al. (2000), include: the habitat fragmentation within dammed rivers, downstream habitat effects caused by altered flows, such as loss of floodplains, riparian zones and adjacent wetlands, and deterioration and loss of river deltas; the deterioration of irrigated terrestrial environments and associated surface waters and dewatering of rivers, leading to reduced water quality because of dilution problems; the genetic isolation as a result of habitat fragmentation and changes in ecosystem–level processes such as nutrient cycling and primary productivity; the impacts on biodiversity and contamination of food webs and greenhouse gas (GG) emissions from reservoirs.

Different conservation organizations, governments and donor agencies make intensive efforts to save life on earth. The accomplishment of this urgent task is consistent with another challenging mission — conservation of biodiversity. The third edition of the Global Biodiversity Outlook (SCBD, 2010) insisted that urgent actions must be taken during this and next decades to reduce biodiversity loss and prevent reaching the tipping points. At the same time, despite numerous actions, the biodiversity continues to be lost, ecosystems are degrading and a consequent decline in ecosystem services threatens to undermine human well-being. Such conclusion was supported by the latest assessment of The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, 2018, p. 2), which states: "… nature's contributions to people are critically important for a good quality of life, but are not evenly experienced by people and communities, and are under threat due to the strong ongoing decline of biodiversity".

In this context, in October 2010, the tenth Conference of the Parties to the Convention on Biological Diversity (COP-10), which was held in Aichi-Nagoya, Japan, adopted the Strategic Plan for Biodiversity 2011-2020, including 20 "Aichi Biodiversity Targets".[1] A key to implement this strategic plan is the establishment of corresponding national targets and their integration into updated national biodiversity strategies and action plans (NBSAPs). In particular, according to Target 11 of Strategic Goal C: "Improving the status of biodiversity by safeguarding ecosystems, species and genetic diversity", by 2020 at least 17% of terrestrial and inland waters, especially of extraordinary importance for biodiversity and ecosystem services (ES), should be conserved through ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures.

The freshwater ecosystems hold a special place in this activity. On the one hand, while the earth's rivers, lakes, and wetlands contain a mere 0.01% of the world's water resources, their ecosystems occupy a disproportionately large fraction of the Earth's biodiversity. On the other hand, worldwide freshwater biodiversity is more threatened than terrestrial, since it is being

[1] Available at: *https://www.cbd.int/sp/targets/*

subjected to an array of threats operating over a range of scales (Abell et al., 2002). The excessive exploitation of ecosystem services can turn into a pressure for an ecosystem and its biodiversity.

In recent decades, a new dimension of impacts on biodiversity has been introduced by climate change and its consequences (Elmhagen et al., 2015). The possible biodiversity losses due to this factor may modify the structure and function of ecosystems, thus affecting the delivery of ecosystem services. In the biotic environment, species can respond to change either through evolution, adapting to new conditions, or by tracking suitable conditions through their dispersal. Furthermore, the impact of climate change on biodiversity differs, depending on the status of certain species in an ecosystem. To meet such challenges, additional researches are needed at different spatial scales – water body, catchment, European ones – because the response strategies rely on the quality of available information and the capacity to make informed decisions (Grizzetti et al, 2015). Ecosystem loss and fragmentation are considered as the greatest worldwide threat to biodiversity and the primary cause of species extinction. Moreover, these processes are as much as an issue for biodiversity in aquatic environments as they are for terrestrial ones (Laverty and Gibbs, 2007). For example, river systems and wetlands are being fragmented by natural forces such as bottom topography, river flows, floods, as well as by human activities such as drainage, groundwater extraction, dams, sedimentation, etc (DSU, 2017; Zaimes et al., 2019). For example, wetlands are drastically reduced in area and number in many regions of the world due to their intensive drainage and human use. Thus, according to Laverty and Gibbs (2007), in the continental United States, where wetlands' study has been more extensive, they have declined by more than half (from 89 to 42 million ha) between 1780 and 1980, and the rate of loss is speeding up. In Europe and Central Asia, the extent of wetlands has declined by 50% since 1970 (IPBES, 2018).

However, while the current state of knowledge in this sphere is quite enough for physical disruptions of river discharge and biogeochemical alterations, the economic valuation of the observed effects is making its first steps. At the same time, the ecosystem services that relate to freshwater resources encompass the benefits to people that can be estimated in economic terms (Grizzetti et al., 2015; Reya et al., 2018). Likewise, any damage to ecosystems and their biodiversity should also be evaluated economically. Thus, the idea of a special study dedicated to the economic valuation of ecosystems and biodiversity is driven by the numerous reasons.

Generally, the *HydroEcoNex* project aims to analyze the effects of hydropower and climate change 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).

1. Economic valuation of ecosystem services as a concept

The economic valuation of ecosystem services is a prerequisite to make optimal choices regarding their protection, conservation and sustainable use; EV also provides a set of tools for informed decision making. Through EV the ecosystem goods and services can be comparable with other investments in economic activity, and as such they allow including properly the natural values in economic calculations. Highlighting how many ecosystems contribute to society, a valuation study helps to understand benefits and costs of any intervention for their modification, while the lack of prices for such services leads to economic insecurity. Prices, which don’t take into account the environmental component, give distorted signals about the importance of ecosystem services for society (GEF, 2018; DEFRA, 2007).

The idea and concept of ecosystem services was developed and described in different publications and reports, starting in the late 1990s, for example, by Costanza et al. (1997) and Daily et al. (2000). However globally, the concept was developed in considerable detail by the United Nations´“Millennium Ecosystem Assessment” (MEA 2005) and in the increasing number of subsequent publications (e.g., DEFRA, 2007). The last definition of this concept was proposed by GEF (2018): *“Ecosystem services* are the many and varied benefits that humans obtain from the natural environment and from properly-functioning ecosystems for free”. The report of the Economics of ecosystems and biodiversity ecological and economic foundation (TEEB, 2010) categorized services, representing different benefits and goods that ecosystems provide, into four broad categories or types: *provisioning, regulating, habitat and cultural* services.

According to the *Total Economic Value (TEV)* concept the ecosystem services were also divided into those providing the so-called *using values* and those providing *non-using values.* The TEV is a common approach from the field of environmental economics (Plottu and Plottu, 2007) to create a single monetary metric, which combines all activities within an area, and to express the levels of each activity in units of a common monetary measure. This is a useful tool for exploring what types of values each ecosystem service provides and helps in determining the valuation methods required to capture these values (DEFRA 2007). Before *TEV* concept was introduced, the economic values of ecosystems were defined as "benefits", being simply attributed only to raw materials and physical products that ecosystems generate for human production and consumption. However, such understanding of economic values represents only a small part of the total value of ecosystem services that generate economic benefits far in excess of just physical or marketed products.

In today's interpretation, TEV include (GEF, 2018):

- ***Use values,*** which additionally are divided in *direct use values*, or using a resource either in a consumptive way (e.g., fishing) or in a non-consumptive way (e.g., water transport), and *indirect use values* when a benefit from ecosystem services is supported by a resource rather than its actual use (e.g., flood mitigation through watershed protection);
- ***Non-Use Values*** are those associated with benefits derived simply from knowledge that the natural environment exists and is maintained. These values can be split into three basic components: *altruistic value* that means a user can enjoy goods and services the natural environment provides; *bequest value*, associated with knowledge the natural environment will be passed on to future generations; and *existence value*, derived from a simple satisfaction of knowing the ecosystems continue to exist, regardless of their use now or in future, thus associating with *intrinsic value*.

With such a paradigm, *economic valuation* is a tool for valuing ecosystems and their services in monetary terms. It quantifies the benefits provided by ecosystems and the impact of ecosystem changes on wellbeing of people. Although ecosystem services are crucial for the well-being, their contribution to economic systems is difficult to quantify in monetary terms. Since some of them are not quantified (not traded in commercial markets), they are often given too little (or no weight at all) in decision making, e.g. in the development of big infrastructure projects. In such situations, final decisions may favor outcomes, which have a commercial value, and thus turning unsustainable use of ecosystems more profitable in a short term, while having considerable economic long term costs (GEF, 2018).

According to the Millennium Ecosystem Assessment (MEA, 2005), in the concept of 'ecosystem services valuation' a *value* is understood as the contribution of an action or object to user-specified goals, objectives, or condition, while *valuation* is the process of attributing the value. Any decision involving trade-offs of ecosystem service implies valuation, where the value of ecosystem services is the relative contribution of ecosystem to the goal of supporting sustainable human wellbeing (Costanza et al., 2014). There are multiple values and multiple valuation metrics. The values that are captured by the ecosystem service concept depend on how they are implemented, or what approaches and methodologies are used. Moreover, different stakeholders have different value systems and perspectives.

The notion of value should not be restricted to a merely monetary value but also embrace their larger range. As Keeler et al. (2012) noted, restricting the value of ecosystem services to economic value only, creates a risk to fail accounting all value dimensions and environmental components (trade-offs) of policy decision, and other, including non-monetary, valuation methods should be also adopted (Jax et al. 2013). Such idea means, in essence, a so-called 'value pluralism'. However, the recognition of the importance of integrating different dimensions of the value concept, challenges difficulties in integrating different metrics of its valuation.

EV can be focused on a single ecosystem type of special interest and an ecosystem services it provides, but similarly it can be dedicated to one specific ecosystem service of relevance in the area of interest. In certain cases it can consider an important singular pressure or impact resulting from this pressure, and the resulting losses in ecosystem services. In particular, as such pressure there are widely considered global warming, changing hydrometeorological conditions in the river basin, or hydropower developments, cardinally transforming the river streamflow and thus impacting the aquatic and riverine ecosystems and their services (e.g., Christie et al., 2006,2007; De Groot et al., 2012; Georgiou et al., 2006; TEEB, 2010).

And at last, the economic valuation as the analysis of impacts on ecosystems and their services, with in-depth assessment of economic costs and benefits in a specific area, could have as its principal objective to demonstrate the economic values at risk or economic values that can be maintained/increased by a specific activity under an analysis, with the aim to influence on corresponding policy decisions.

2. METHODOLOGY

2.1 General provisions: Scoping study

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 (e.g., GEF, 2018; Secretariat…, 2007; TEEB, 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 (e.g., 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 (GEF, 2018) recommends to use a *"benefit transfer"* method based on transferring available information from the studies already completed in another location and context. 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. 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 (them?), 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.

2.2 Setting the Scene

2.2.1 *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.

At the *water body* scale as a main focus of this process should be the analysis of specific functions of ecosystems, which support certain ecosystem services, and the study of their alteration under specific impacts and their different combinations. The *catchment scale* offers the relevant areas for the application of ecosystem services concepts in river basins management. Within a catchment, the aquatic and riparian ecosystems and their services can be further mapped and studded at a water body scale or by subcatchments, depending on data availability and desired resolution for the assessment. The catchment is also an appropriate scale to observe and quantify processes related to a water cycle or to implement monitoring and management plans.

As an example (Fig. 2.1), 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.

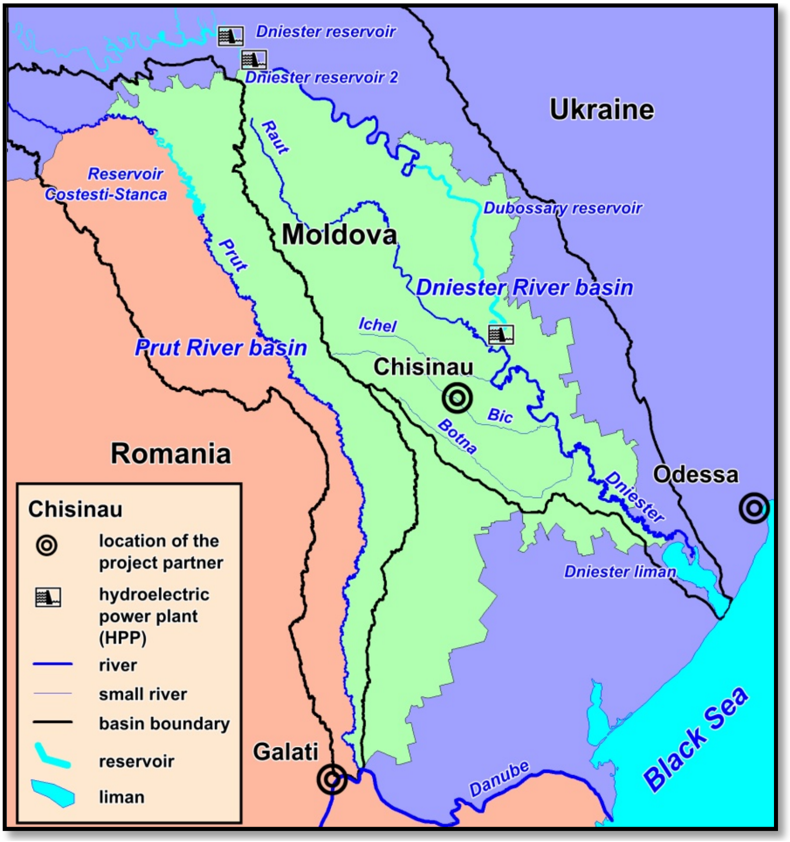


Fig. 2.1: Setting the Scene in the *HydroEcoNex* project

2.2.2 Wetlands as "hot-spots" in economic valuation

In the Setting the Scene for economical valuation, an especial place should be given to wetlands. As it was mentioned in the Introduction, such conclusion is driven by evidences that these ecosystems drastically reduced in their number and areas in many regions of the world due to intensive climate warming, water drainage and human use (Laverty and Gibbs, 2007; IPBES, 2018).

In this study, there was used a common definition of wetland: the transitional lands between terrestrial and aquatic systems, where the water table is usually at or near the surface, or the land is covered by shallow water. Some wetlands are linked to rivers because in floodplains, due the dynamic nature of ecosystems, it is sometimes very difficult to distinguish between terrestrial and aquatic species and habitats. The ecosystem services, which relate to wetlands with their freshwater and natural resources, encompass benefits to people that can be estimated in economic terms (Reya et al., 2018). Likewise, damage to wetland ecosystems and their biodiversity should also be evaluated economically.

The idea to include wetlands as an individual object of economic valuation was driven by the following principal scientific and practical reasons:

(1) There is an urgent need to conserve water ecosystems that are among the world's most productive environments with a wide array of benefits. Wetlands are cradles of biological diversity, providing with water and primary productivity, upon which countless species of plants and animals, including wildlife resources, depend on survival, being also important storehouses of plant genetic material. Many of the wetlands are 'biodiversity hotspots' and the numerous threats they face, along with the many ecosystem services they offer, have led to their protection status by the Ramsar Convention (Ramsar, 2009) and the Natura 2000 Network (European Commission, 2007); their conservation or re-establishment, especially in human modified environments, has become a worldwide priority (Abell et al., 2002).

(2) Wetlands not only have zoogeographic relevance, but also serve as the most appropriate units for the conservation of freshwater biodiversity. The quality of wetlands habitat at any location is a function of all upstream and upland activities, and sometimes downstream activities too. Many of the threats to wetlands systems are the result of land-use practices or hydropower development (Vejnovic, 2017), which occur within their surroundings, and thus must be addressed (Abell et al., 2002).

(3) During the past century, many wetlands have been lost and degraded. Sometimes, labeled as wastelands and treated as 'dustbins' for wastewaters and solid wastes, they receive no worthy attention in the development plans (Gopal, 2015). Therefore, protecting wetlands biodiversity, their specific biophysical characteristics and benefits (ecosystem goods and services) requires a major change in national policies. The multiple roles and value of wetland ecosystems have been increasingly understood and documented, resulting in large expenditures to restore their lost or degraded hydrological and biological functions, including in Moldova (Andreev, 2017; Andreev et al., 2013; Rubel, 2007, 2009). But this is not enough, and there is a

need to improve practices on different scales in the attempt to cope with the accelerating water crisis.

(4) Wetlands degradation and their loss are more rapid than those of other ecosystems and are continuing at an alarming rate (Jiménez Cisneros et al., 2014), primarily due to infrastructure development, land conversion, water withdrawal, eutrophication and pollution, the introduction of invasive alien species, etc. (MEA, 2005). Somewhere, the occupation of wetlands and adjacent floodplain areas for the intensive urban and agricultural land-use has led many of them to functional disconnection with their rivers. For instance, because of water shortage and mismanagement, in the last 50 years, half of the Mediterranean wetlands have disappeared. Pollution from cities and agriculture, especially nutrient loading, results in declines in water quality and the loss of essential ecosystem services (Settele et al., 2014), including the species groups from the IUCN Red List (http://www.iucnredlist.org/). It is very likely that these stressors for wetlands ecosystems will continue to dominate as the human demand for water resources grows, accompanied by increased urbanization, ongoing hydropower construction on rivers (Smith et al, 2007) and expansion of irrigated agriculture.

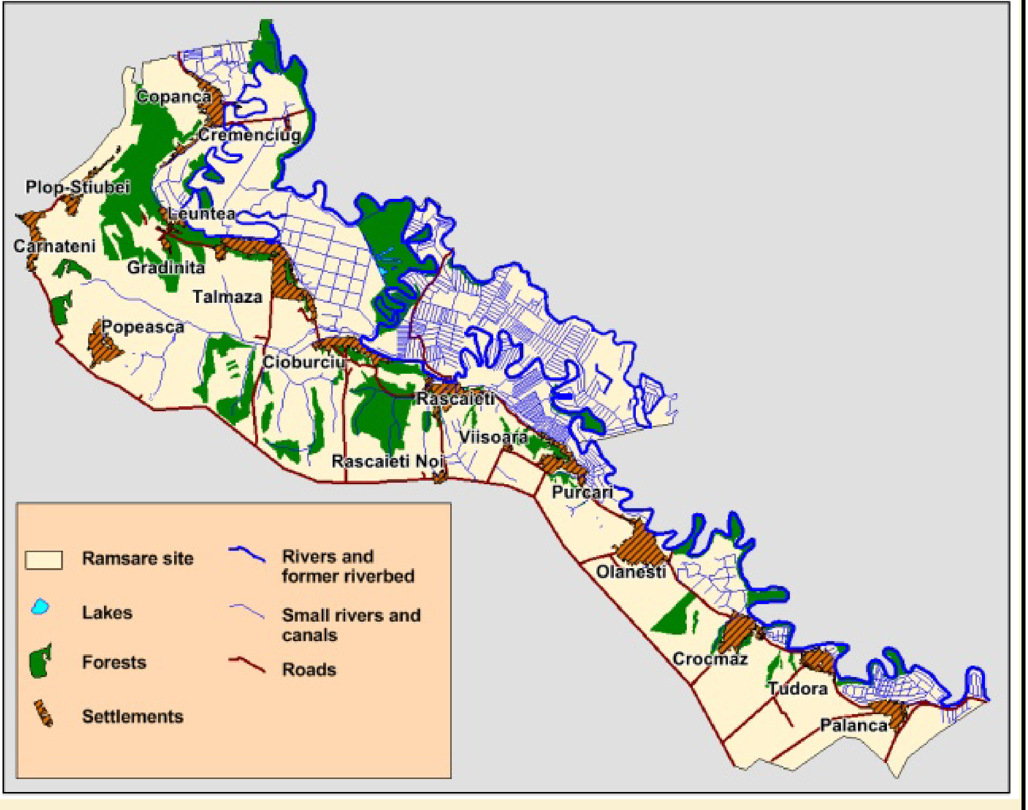
Considering an economic valuation as a prerequisite for making optimal choices regarding the protection and conservation of wetlands biodiversity, this Guidance aims, mainly on the example of one wetland used as a case study (see Box 2.1), to demonstrate some approaches and provide a set of tools for making an informed decision on this kind of problem.

2.2.3 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 Grizzetti et al. (2015) and GEF (2018). In particular, the MARS (**M**anaging **A**quatic ecosystems and water **R**esources under multiples **S**tress) research project, described by Grizzetti et al. (2015), was focused on: (1) the

Box 2.1: **Ramsar site "Lower Dniester" as a case study**



This wetland in the Dniester River basin, selected for this study (Figure), occupies currently about 60.64 ha and includes 18 natural complexes. Due to the international natural and ecological importance, in 2003 this territory was designated to be under the Ramsar Convention (Ramsar, 2009) and received the official status of the international zone Nr. 1316 (3MD003): Ramsar Site "Lower Dniester" (hereafter, sometimes, Lower Dniester wetla**nd**). For a long time, this territory was exposed to an intensive anthropogenic pressure that has led to its transformation, fragmentation of their natural complexes, and reduction of biological diversity and ecological stability. Therefore, in order to support the natural functional organization, the conservation of this wetland' natural systems from further anthropogenic loading is a very practical problem for its survival.

ecosystem services delivered by the aquatic ecosystems, which can be linked to the water body status, and (2) the hydrological ecosystem services relevant for river basin management and including processes related to the interaction of water and land in different ecosystems, such as forest, agriculture, riparian areas, wetlands, and water bodies. MARS's vision of the List of ecosystem services relevant for water systems can be found in this project Report (Grizzetti et al., 2015, p. 83).

The GEF Guidance (GEF, 2018) considers ecosystems/habitats and ecosystem services selected according to the MAES typology (European Commission, 2013), distinguishing them between rivers and lakes. MAES (**M**apping and **A**ssessment of **E**cosystems and their **S**ervices[2]) approach developed a system of ecosystem classification in the sense that an ecosystem is defined as a complex of flora and fauna in relationship with the abiotic environment.

Beside the open water bodies themselves, there are also considered closely linked *riparian ecosystems* (e.g., *riparian wetlands*) and groundwater dependent ecosystems, listed as "other inland wetlands", which can be partly vegetated. However, only ecosystems functionally linked to rivers and/or their tributaries in terms of flows are to be considered. For instance, forests or other significant ecosystems for water-related services like water storage also presenting in the watershed are excluded from this type analysis; groundwater bodies are included as part of groundwater-dependent ecosystems, e.g. wetlands.

After comparing these two approaches, in this study the preference was given to the GEF Guidance (Table 2.1).

[2] https://biodiversity.europa.eu/maes

Table 2.1 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 ecosystems
Provisioning Servises	**Food**: • ***Fish*** • Aquaculture • Other product • Genetic and medical resources	***Direct use***	Rivers Lakes Inland wetlands
	Foresty:fiber, timber, fuel		Inland wetlands
	Water: drinking water, irrigation, cooling		Rivers, lakes
Regulating Services	***Air quality regulation*** ***Climate regulation (Carbon sequestration***) Moderation of extreme events (e.g. floods) ***Water treatment*** Erosion prevention	***Indirect use***	Inland wetlands
	Nutrient cycling and maintenance of soil fer-tility		Rivers, lakes, inland wetlands
Habitat Services	***Maintenance of life cycles of migratory*** Rivers, lakes, inland wetlands ***species (nur-sery service for fish species)*** ***Maintenance of biodi-versity***		
Cultural Services	***Opportunities for tourism/recreation***	***Direct use***	
	Aesthetic inspiration, Spiritual experience, Education	***Non-use***	

2.2.4 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 studded 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. 2.2 and Table 2.2. 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.

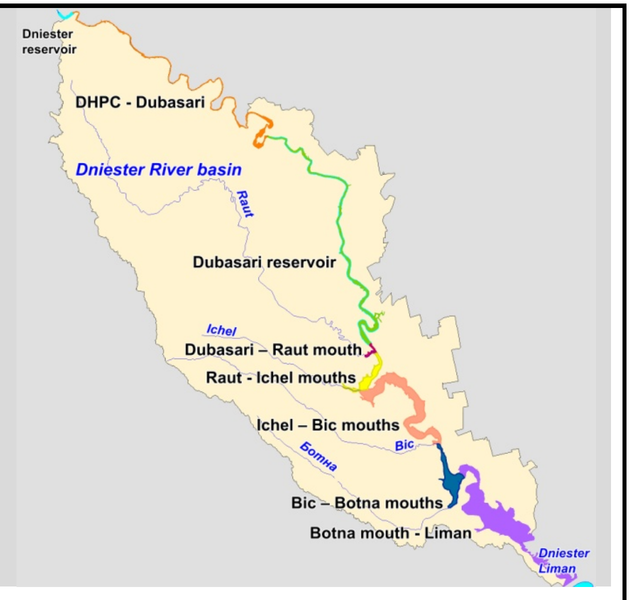


Fig. 2.2 Breakdown of the Dniester floodplain into clusters to study the ecosystems and their services

2.2.5 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.

Table 2.2 Area of ecosystem types *(km^2)* in the Moldavian part of the Dniester river basins[1]

Ecosystem	Clusters							
	DHPC -Dubasari	*Dubasari reservoir*	*Dubasari – Raut mouth*	*Raut mouth - Ichel mouth*	*Ichel moth – 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

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 example of such assessment of ecosystems distribution is given in Box 2.2.

Box 2.2 Distribution of natural ecosystems in the Ramsar site "Lower Dniester"

At the moment, the territory of the Lower Dniester includes 18 natural complexes, the largest among which are Copanca-Leuntea (2.40 ha), Lunca Talmaza (1.60 ha), Zaozernoe–Nucari (1.54 ha), Cioburciu–Răscăeți (1.23 ha) and Olănești–Crocmaz (1.48 ha). Morphometric characteristics of this wetland's natural ecosystems are presented in Table 2.3.

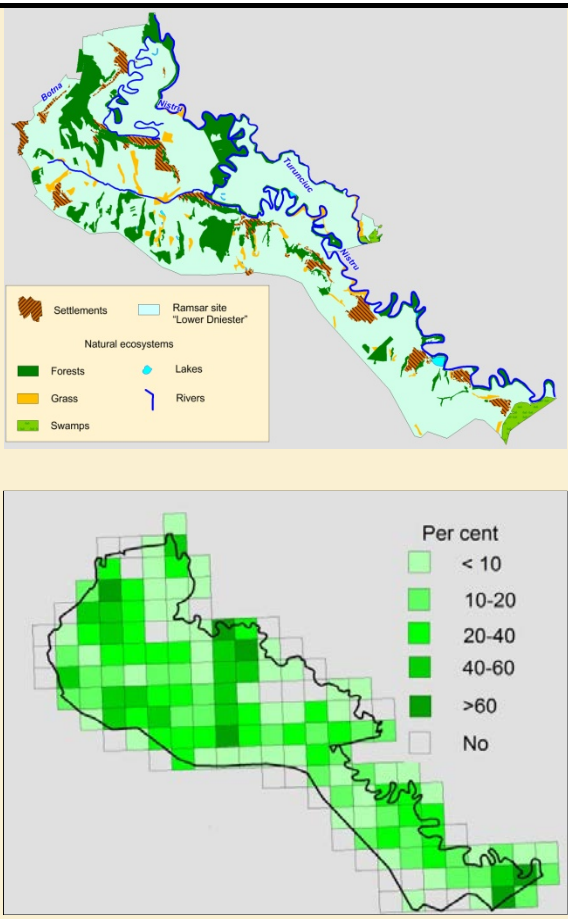


***Fig.* 2.3** Natural complexes of the Lower Dniester *(above)* and their spatial distribution in percentage *(below)*

A main peculiarity of the ecosystems territorial distribution within these natural complexes is its unevenness. So, large forest ecosystems are confined to slopes and partly to the Dniester River floodplain areas; the largest lakes are located in this wetland's southeastern part. At the same time, it should be noted that all complexes are also characterized by a combination of ecosystems, most clearly expressed on the Talmaza overflow lands. To assess the unevenness of ecosystems distribution, as a specific indicator the number of mapping units with different shares of individual ecosystems was used (Fig. 2.3; *below*). At almost half of the territory, the natural ecosystems are either absent or occupy less than 10%, and only on 4% of the territory their share exceeds 60%.

Table 2.3 Morphometric characteristics of natural ecosystems of the Ramsar Site "Lower Dniester"

Natural ecosystems	Number	Mean area, km^2	Mean perimeter, km
Forests	40	2.296	11.493
Grass plots	78	0.544	3.647
Water objects	25	0.251	2.248
Swamps	24	0.506	3.491

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…, 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…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 (Коробов и др., 2014).

However, assessing the fragmentation is not only the assessment of the ecosystems 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.

As a case study, in Box 2.3 there is shown an assessment of the fragmentation of forest ecosystems in the Lower Dniester wetlands.

Box 2.3:* Fragmentation of forest ecosystems in the Ramsar site *'Low Dniester'

The total area of forests in the Lower Dniester wetland is about 9,200 ha, with a forest coverage rate of its territory of 15.3% and a total number of woodlands of 40. The average area of woodlands is 2.3 km^2 (from 0.05 to 25.3 km^2) and the average perimeter is 11.5 km (Fig. 2.5.1). Based on these values, here the forests average CF equals 5.1, but significantly changing (from 2.67 to 68.54). Such a range indicates a high degree of forest ecosystems fragmentation and its territorial differentiation across the wetland. Moreover, such CF value is high for the Ramsar sites, thus requiring a system of measures to reduce it.

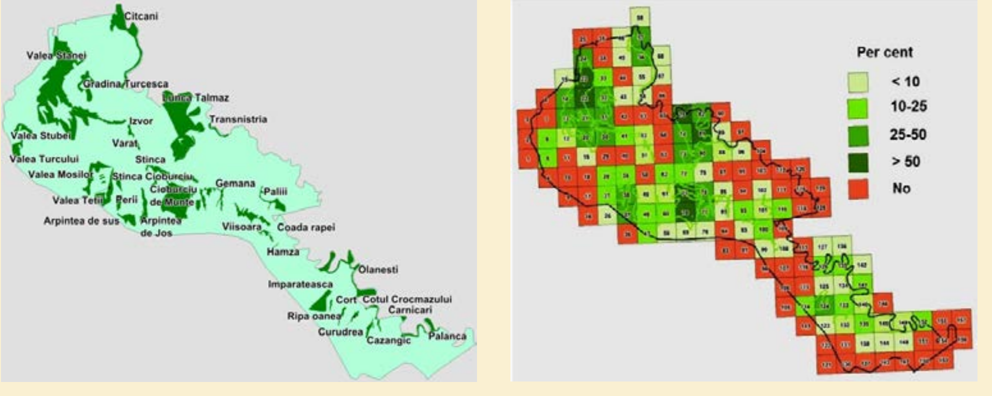


Fig. 2.3 Distribution of forest ecosystems *(left)* and percentage of forests covers in mapping units *(right)* in the Lower Dniester wetland

The studies on the dependence of higher plant species richness in this area (Andreev et al., 2017) indicate a gentle trend in the number of species in small areas, but a steep rise approximately in areas <1,200 ha. Therefore, in the Lower Dniester the analysis of forest areas distribution was carried out only for the two more this value (Table 2.4). We see that CFs of large woodlands is significantly lower than their average (5.1) for this wetland.

Table 2.4 Morphometric characteristics of large (>1200 ha) forest areas

Forest natural complex	Mean area, km^2	Mean perimeter, km	Fragmentation coefficient
Lunca Talmaza	16.93	43.38	2.56
Valea Stanei	25.30	91.20	3.60

3. ECONOMIC VALUATION OF ECOSYSTEM SERVICES AND THEIR LOSSES

3.1 Selection of methodology

Economic Valuation as a common approach, taken from the field of environmental economics (Plottu and 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 (e.g., GEF, 2018; Secretariat..., 2007; TEEB, 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 3.1).

Table *3.1* Methodologies that can be used for economic valuation *(EV)* of freshwater ecosystems services

Type of ES	**Ecosystem service**	**Category of use**	**Methodology for EV**
Provisioning Services	Fish Aquaculture Other products Timber, fuel Water (drinking, irrigation)	Direct use	Market prices
Regulating Services	Carbon sequestration Moderation of extreme events Water/Sewage treatment Erosion prevention	Indirect use	Benefit transfer
Habitat Services	Nursery service Maintenance of life cycles of migratory species Maintenance of genetic diversity		
Cultural Services	Tourism Recreation	Direct use	Market Prices, Benefit Transfer
	Aesthetic informtion, Spiritual experience, Education	Non-use	Benefit Transfer

Source: Adapted from GEF (2018)

3.2 Economic valuation of provisioning services

3.2.1 Water

The theory and practice of pricing in the field of environmental management indicate a need to use different methods of finding prices for natural resources. In the context of adopting the principles of sustainable development, along with market pricing, there is important a method of normative pricing. This method guarantees an economic interest in the reproduction (replacement) of a natural resource as of an owner's object and a source to satisfy needs (Vovere and Buģina, nd; Неверов и др. 2017).This fully applies to water resources, the principles of which management are enshrined in the so-called Dublin Principles of Integrated Water Resources Management (IWRM), among which the most important principle is consideration of water as an economic commodity (GWP, 2010; GWP and INBO, 2009).

In particular, Principle IV of IWRM states: "*Water has a value as an economic good"* (GWP, 2000; p.18). Many past failures in water resources management were attributable to a fact that water has been viewed as a free good or at least that its full value has not been recognized. In a situation of competition for scarce water resources such a notion leads to considering water as a low-value use; this provides no incentives to treat water as a limited asset. Therefore, in order to extract the maximum benefits from the available water resources there is a need to change such perceptions about water values.

First of all, it is necessary to distinguish between *valuing* and *charging* for water. The *value* of water in these two alternative uses is important for its rational allocation, whether by regulatory or economic means, as a scarce resource, based on the "opportunity cost" concept. *Charging* for water is applying an economic instrument to affect users' behavior towards conservation and efficient water usage, to provide incentives for demand management, to ensure cost recovery and to signal consumers' willingness to pay for additional investments in water services.

The full value of water (Fig. 3.2.1, *left*) consists of its use, or economic, value and its intrinsic value. The *economic value* of water depends on a user and the way it is used, and includes: (1) direct water value for users, (2) net benefits from water that is lost through evapotranspiration or other sinks, and (3) the contribution of water towards the attainment of social objectives. The *intrinsic value* includes non-use values such as bequest or existence values.

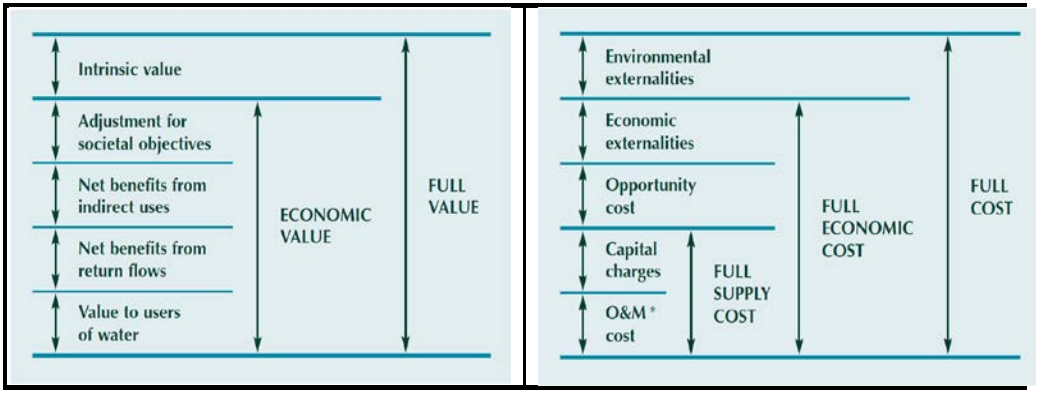


Fig. 3.2.1 General principles for valuing *(left)* and costing *(right)* of water

* O&M – *Operation and Maintenance*
Source: GWP, 2000

According to *water cost concepts*, the full cost of providing water (Fig. 3.2.1, *right*) includes the full economic cost and the environmental externalities, associated with public health and ecosystem maintenance. Thus, the full economic cost of water consists from the full supply cost due to resource management, operating and maintenance expenditures and capital charges, the opportunity costs from alternative water uses, and the economic externalities arising from changes in economic activities of indirectly affected sectors. Estimation of full water cost and especially her losses may be very difficult. In situations involving conflicts over water, including the trans-boundary ones, the attempts should be made to at least estimate the full economic cost as a basis for its allocation.

With adopting the concept of estimating the full cost of water (including environmental costs), the calculation of a current annual value *(Rw)* of aquatic ecosystems is carried out according to the following equation (ТКП, 2013):

$$R_w = \frac{P * K_R}{1+p+K_R} * K_e * K_d * V * q_o \qquad (1)$$

Where:

$\boldsymbol{P}$ – market price of fresh drinking water that is determined, taking into account the average market selling price of 1 m^3 of still drinking water; $\boldsymbol{p}$ = 0.3 – the coefficient of efficiency (profitability) of water extraction;

$\boldsymbol{K_R}$ = 0.30 – coefficient of a resource reproduction efficiency (fresh water)[1];

K_e = 0.95 – output coefficient of the final product (fresh water), taking into account technological losses during its extraction, transportation and refining;
K_d – differentiation coefficient of a drinking water value, based on a class of its quality determined by the totality of hydrobiological indicators[2];
V – stock of water resources, m^3 (average annual river flow, volume of water accumulated in lakes, ponds, etc.) per unit of the water area;
q_o – coefficient, which value is inversely proportional to the reproduction period of a consumed natural substance, which forms the basis of a natural ecological system[3].

Note: [1]K_R is determined by the average achieved levels of efficiency in the relative branch of production (IRFS, nd);
[2]K_d is evaluated by four classes of water purity: ***0.8*** – 1-2 class; ***0.6*** – 3 class; ***0.4*** – 4 class; ***0.2*** – 5 class;
[3]q_0 for water ecosystems with the reproduction period of 43 years, this coefficient equals 0.02 (ТКП, 2013).

Thus, the specific current (annual) assessment is valuation of an economic effect obtained annually as a result of exploitation (reproduction) of fresh water within an aquatic ecosystem per ha of its area. To estimate the total cost of ecosystem services *(Pw)*, it is necessary to multiply *Rw* by the area it occupies (*Sw*):

$$P_w = R_w * S_w$$

Where:

P_w – estimation of the total cost of an aquatic ecosystem;
R_w – current (annual) value of an aquatic ecosystem services per ha;
S_w – area of aquatic ecosystem, ha.

The most difficult element in the economic valuation of water services is determination of its market price. Using for these purposes a water tariff for the population is not satisfactory because this tariff carries a significant social burden associated with ensuring the financial accessibility of water supply services for users. Therefore, it does not reflect not only the full cost of water, but even its full economic value. As an example, in Table 3.2.1 there are shown tariffs for water in Moldova. For example, in 2017 with an average national tariff of 14.96 MDL (0.85 USD) for cubic meter, it was 12.32 MDL (0.7 USD) for the population and 31.99 MDL (1.83 USD) – for other consumers, varying from 8.2 to 16.2 MDL for cubic meter for the population and from 12.7 to 51.8 MDL for cubic meter – for other consumers. Moreover, the tariff depends not only on consumers, but also on local conditions of water supply. As can be seen from Table 3.2.1, the lowest water tariffs are set in the central regions of Moldova, where the capital of Moldova is located, and the water infrastructure is more developed. The highest rates are set in the southern most arid regions of the country.

Table 3.2.1 Tariff for water in Moldova by operators and regions in 2017 *(MDL/m* ^{3}*)*

Water operator	Mean tariff	Population	Other
S.A. “Apă-Canal Chisinau”	8.86	8.06	12.7
Î.M. “Apă-Canal” Basarabeasca	9.7	9.0	36.0
Î.M. “Apă-Canal” Cahul	11.25	12.0	27.97
Î.M. “Apă-Canal” Anenii Noi	13.53	13.5	37.4
I.M. Regia “Apă-Canal" Balti”	15.05	11.08	23.64
S.A. “Regia Apă-Canal” Soroca	17.85	15.26	35.2
S.A. “Apă-Termo” Ceadîr-Lunga	18.76	16.2	40.0
Î.M. “Apă-Canal” Edineț	21.35	12.5	25.05
Î.M. “SCL” Rezina	21.5	12.8	51.8
Region	**Mean tariff**	**Population**	**Other**
Nord	15.60	12.11	32.67
Centru	13.54	11.69	28.56
Sud	15.65	13.29	34.72
Average tariff for Moldova	***14.96***	***12.32***	***31.99***

Substantiating the tariff, the Special Working Group on Green Actions (OECD, 2003) recommends adhering to such criterion of water accessibility when the total annual cost per capita for water supply and sanitation services (WSS) must not exceed 3.5-4% of the average annual disposable income per capita in the area served by an existing WSS system. A need to adhere to 4% threshold for accessibility to total household expenses is also indicated in a number of other studies (e.g., Pienaru et al., 2014; OECD, 2016), but it does not reflect the economic aspects of operators' activities. Therefore, in a number of countries, including Moldova (Eptisa, 2012), most households spend on paying for these services more than 5% of their income.

The concept of a common tariff for the entire service area, commonly implemented, for example, in Romania, was not introduced in all regions of Moldova. A main obstacle for such decision is the resistance of local public authorities to accept a common tariff because it is usually higher than that with a decentralized solution. This is mostly explained by a fact that decentralized solutions provide unsustainable low-quality service, which requires lower operating costs.

In this regard, it seems more justified to use the market price of drinking water, which is determined taking into account the average sale price for 1 m^3 of drinking still water. Although retail prices for bottled water vary greatly depending on many factors (transportation, packaging, place of sale, etc.), in general they vary from $0.05 to $6.0 per liter, which is equivalent to $50-6,000 per m^3 (see: https://en.wikipedia.org/wiki/Waterpricing). In Moldova, for example, this price is lower and amounts to $0.025/liter or $25/m^3.

Economic valuation of river water as an ecosystem service allows assessing correctly the corresponding losses caused by the influence of different factors. The example of such assessment is given in Box 3.2.1.

Box 3.2.1: **Economic valuation of losses of the Dniester River water provisioning services due to hydropower impact**

The assessment of changes in the Dniester water streamflow due to the Dniester hydropower complex *(DHPC)* operation was based on comparing the streamflow volume *(Q)* at hydrological post Zalishchyky, located upstream of DHPC, and Mohyliv-Podilskyi and Bender – located downstream, in periods before (1951-1980) and after (1991-2015) this complex construction. Results of the comparison are shown in Table. The decrease of *Q* downstream the DHPC in 1991-2015, compared with its increase upstream, indicates an undoubted effect of this complex.

The Dniester annual runoff *(km^3)* upstream and downstream of DHPC before and after its construction

Hydrological 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

Source: Corobov et al. (in press)

The economic valuation of losses due to DHPC impact was carried out using above shown Eq. 1. Assuming that all terms in that equation, except ***V***, remained unchanged after the complex construction and considering ***V*** as the decrease in Dniester streamflow, we have get annual economic losses of $30 million in Mohyliv-Podilskyi and above twice more – in Bender (at a water price of $25/m^3).

3.2.2 Fishery

Some general points

Regarding ***fishery***, according to GEF recommendation (GEF, 2018), as a basis for its valuation, as one of ecosystem services, the sustainable *annual output/yield* should be taken rather than total value of all available fish stocks or the revenues generated from any fish harvesting activities, which result in the depletion of the natural capital stock (for example, in a situation of an overfished stock).

Freshwater ecosystems incorporate fish/fishery and aquaculture products. Information on these products is generally available in two forms: either as an absolute value (in monetary terms) or as a relative value. The *absolute value* is presented as a “total value” (e.g., total value of all fish catches in an area per year). The *relative value* is presented by a number relative to a unit of measurement (e.g., “value per ton caught” or “value per m³ harvested”). In the first case, the absolute value is related to a single hectare or square kilometer; hectare is recommended as most economic values: “value per hectare”. In the second case, there is a need to calculate the absolute value by multiplying a value per kg/ton/m³ with the overall amount produced or harvested. One example of calculation is given by the GEF Guidance (GEF, 2018; p. 35).

However, fisheries should be included in the economic valuation only as long as it is provided on a sustainable basis, i.e. the Maximum Sustainable Yield (MSY) should be taken as the basis for the valuation as ecosystem service, rather than the total value of all available fish stocks. This means that if you have information on the annual catches, and at the same time you know they are not sustainable (i.e. above the MSY), then you need to reduce the amount/value down to MSY.

In addition, when assessing the loss of ecosystem services provided by fishing, we should take into account not only a decrease in fish productivity resulting from negative impacts, e.g. of hydropower or global warming, but also change in the water bodies and water ecosystem areas, particularly of fresh waters. Thus, if the EV shows the area of river ecosystems in a studied area has decreased, this, accordingly, leads to an additional loss of ecosystems capacity to provide fishing.

The example of economic valuation of fishery as an ecosystem service is given in Case study 3.2.1.

Case study 3.2.1: Dniester's fishery losses due to hydropower and anthropogenic impacts

The long-term dynamics of the volumes of commercial fishery on the Dniester River (Fig. 3.2.2) indicates their significant reduction, undoubtedly associated with HPPs construction. In particular, the first sharp reduction took place in the 1950s and was caused by the Dubasari HPP construction; the second, equally obvious decline, which occurred in the 1990s, was due to the commissioning of the Dniester hydropower complex. The total decrease in the stocks of commercial fish catches in the territorial boundaries of Moldova amounted to about 90-95 tons per year.

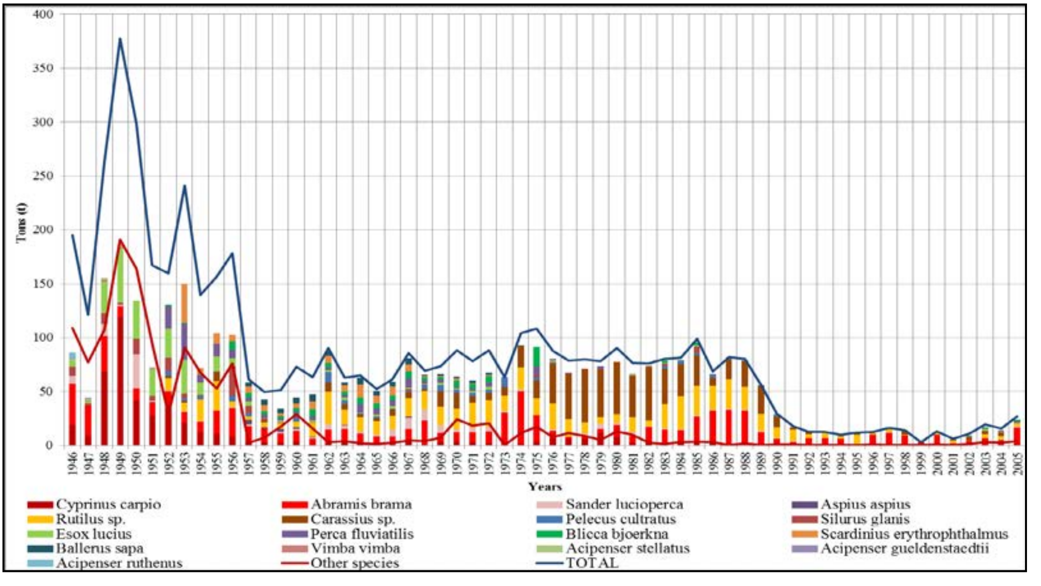


Fig. 3.2.2 Dynamics of the commercial catch of fisheries on the Dniester River by its volume *(tons)* and species. *Source:* Institute of Zoology of Moldova, analyzed by Bulat (2017)

For a more detailed study of the dynamics of fish catches' fall in the Dniester, an appropriate regression analysis was carried out (Fig. 3.2.3). The second-degree polynomial regressions, which most reliably describe the process under study, were constructed both for total fish species and for their categories with different value. In such categories there were selected: *high value (*Starry sturgeon, Sturgeon Starlet); *mean value* (Carp, Pike-perch, Weal, Pike, Vimba); *low value* (Bream, Asp, Roach, Crucial carp, Sablefish, Perch, White bream, Rudd) and *Other value*. However, the regression relationships for high valuable fish were not built, because their very small catch was recorded only from 1946 to 1949, with a sequential decrease from 6 tons to 1.7 tons (only 11.8 tons for this period).

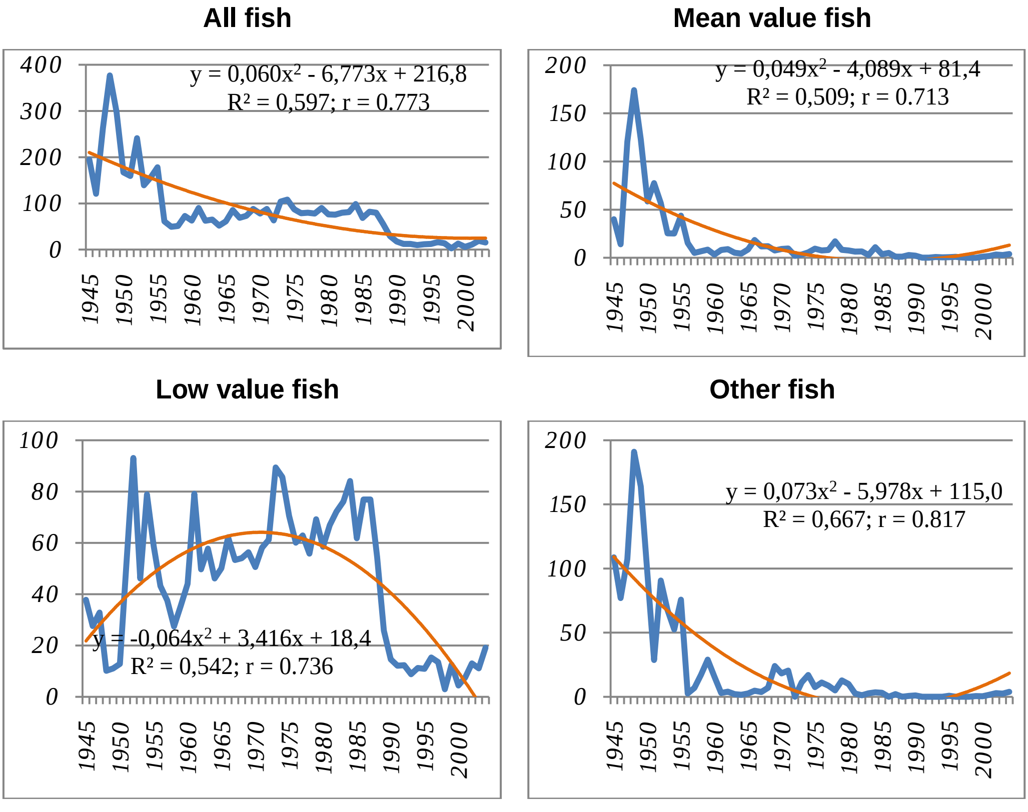


Fig. 3.2.3 The second-degree polynomial trends of fish catches of different commercial value

A purely visual analysis of the obtained dependencies allows drawing the two main conclusions. First, a high statistical significance is observed for all regressions, and the correlation ratio ***r***, which is more than 0.7 in all cases, characterizes a strong relationship. Secondly, the gradual replacement of high- and medium-valuable fish species with less valuable ones. This is clearly seen when to compare the relevant trends. Thus, along with the general decrease of fish stocks, the stock of commercially valuable species has decreased especially significantly. In quantitative terms, this conclusion is well confirmed by the data in Table 3.2.2.

***Table* 3.2.2** Annual catches *(tons)* of different values fish in the Dniester River in three time periods

Time period	Stati-stics	Fish value			Total
		Mean value	Low value	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

Here, to assess the impact of hydropower plants on these processes, the volumes of catches were divided into three time periods: before damming the Dniester for Dubasari HPP construction (1946-1953); between this damming and the second damming for DHPC contraction (1954-1983), and the subsequent years (1984-2005). So, after the first damming, the average annual catches of mean-valuables fish decreased by almost eight times, and after the river second overlap — by another five times, decreasing for sixty years from about 83 to 2.1 tons. At the same time, catches of low-value fish at the same time periods increased initially from 34.8 tons to 58 tons, and at the beginning of the current century they decreased by only 6.4 tons per year compared to the pre-damming period. On the whole, the economic value of fishery as the Dniester River's ecosystem service falls threateningly.

One more example, demonstrates this situation. If prior to HPPs construction in the Naslavcea–Camenka part of Dniester, the main commercial fish species were sterlet *Acipenser ruthenus*, European carp *Cyprinus carpio*, vimba *Vimba vimba*, sheatfish *Silurus glanis*, nase Chondrostoma nasus, barbel *Barbus* (Ярошенко, 1957), then today the commercial fish are largely superseded by low-value, short-cyclic and invasive species, where three-spined smelt *Gasterosteus aculeatus*, bitterling *Rhodeus sericeus* and bleak *Alburnus alburnus* dominate (Bulat, 2017).

A negative tendency in commercial fishing, expressed in the significant decrease in recorded catches and change in their structure, is also observed in the Dubasari reservoir (Fig. 3.2.4), despite the great efforts on fish stock maintaining. So, in 1998-2010 about 94 tons of fish tries were released into the Dubasari reservoir for this aim (Usatîi et al., 2016).

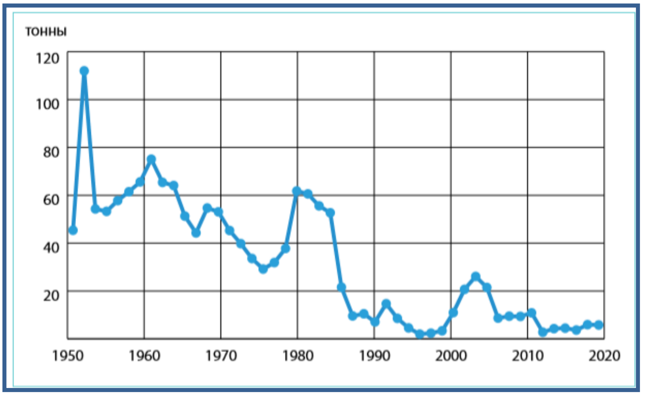


Fig. *3.2.4* Commercial catch of fish in Dubasari reservoir, tons. *Source:* Usatîi et al., 2016

The hydropower impact on fish stock in the Dniester River is also strengthened by general ecological situation in the basin. Its permanent deterioration also plays certain role, negatively affecting the ecological state of main aquatic ecosystems. Their quality status is presented in Table 3.2.3 where categories of quality were defined according to the Water Framework Directive (WFD, 2000). This Directive, based on the structural-functional status of a fish fauna, has highlighted five quality categories of aquatic ecosystems.

Table 3.2.3 Attribution of the ecological quality classes to aquatic ecosystems in the Dniester River based on Index Biotic Integrity values (IBI 9) by WFD (2000)

Ecosystem type	**Ecosystem**	**Biotic Integrity Class**	**Quality category in accordance with the Water Framework Directive (2000/60 EC)**	
Lotic	Dniester River	Poor	IV	Weak
Slow	Dubossary reservoir	Poor	IV	Weak

Source: Bulat, 2017

To valuate economically the fisheries losses, three approaches have been used:

1. *Cost of direct losses.* Before the beginning of hydro construction on the Dniester River, fish productivity in its part from Rybnitsa to Palanca was 6-7 kg/ha (Ярошенко, 1957). Based on the area of river and lake ecosystems located here (143.41 km^2), the fish stock was 93.2 tons; approximately the same amount was a real annual catch before the construction of the DHPC.

Currently, the catch amounts to about 20 tons, and the resulting difference (about 73 tons) represents the loss in the fishery's *provisioning* ecosystem service. Based on the price of freshwater fish, established by GLOBEFISH[3] (FAO, 2020), which in 2019 was $2.35 for kg, the observed losses were more than $172 thousand per year.

2. Similarly, it is possible to estimate the loss of annual fishing catches in the Dubasari reservoir that decreased from 60 tons in the 1980s to 2-3 tons at present (about $135 thousand per year), despite the measures taken for its artificial stocking.

3. *The costs of maintaining the habitat services.* If to consider the cost of ecosystem conservation and maintaining as a value of loss of its ecosystem services, then the cost of maintaining the fish spawning grounds *(nursery habitat)* can be also considered as certain equivalent of the damage done to this ecosystem. So, the cost of 150.15 tons of fries of various fish species, launched for example in 1998-2018 in Dubasari reservoir for maintaining its fish stock, amounted to 6.3 million MDL (360,4 US dollars)[4]. Undoubtedly, this figure is also one of components of economic valuation of HPPs' caused damage to the Dniester ecosystem services as a whole.

4. *The cost of losses in cultural services* was indirectly estimated by the scale of amateur fishing. Currently 15,000 fishermen are registered in Moldova, and for amateur fishing on the Dniester it is necessary to purchase a fishing ticket. Revenues from sport fishing are estimated at 2.5-4.5 million MDL, or about 145-260 thousand USD per year. Thus, an increase or decrease in the number of amateur fishermen is a reliable indicator of the ichthyofauna conditions in the river basin.

[3] GLOBEFISH is a multi-donor funded project within the FAO Fisheries and Aquaculture Department responsible for providing up-to-date trade and market on fish and fishery products.
[4] According to Fish Farming Service of Moldova. See: https://ru.sputnik.md/society/20180425/18782986/dnestr-ryba.html

3.2.3 Forestry

The calculation of the current (annual) economic value *(Ri)* for forest ecosystems is carried out according to the following equation (ТКП, 2013):

$$R_i = \frac{P * K_R}{1 + p + K_R} * K_{ev} * K_{bp} * K_e * V \qquad (1)$$

Where: $\boldsymbol{P}$ – average market price of the main product of forest use (timber);

- $\boldsymbol{p}$ – coefficient of efficiency (profitability) of timber production as a result of its exploitation;
- $\boldsymbol{K_R}$ – coefficient of efficiency of forests reproduction;
- $\boldsymbol{K_{ev}}$ – coefficient of economic value of the main tree species on an evaluated area;
- $\boldsymbol{K_{bp}}$ – coefficient reflecting the cost of by-products;
- $\boldsymbol{K_e}$ – coefficient of ecological significance of forest types;
- $\boldsymbol{V}$ – annual forest productivity per 1 ha of area.

The case study below demonstrates the economic valuation of forest ecosystems in the Lower Dniester.

Case study 3.2.2 Economic valuation of forest ecosystems in the Ramsar site 'Lower Dniester'

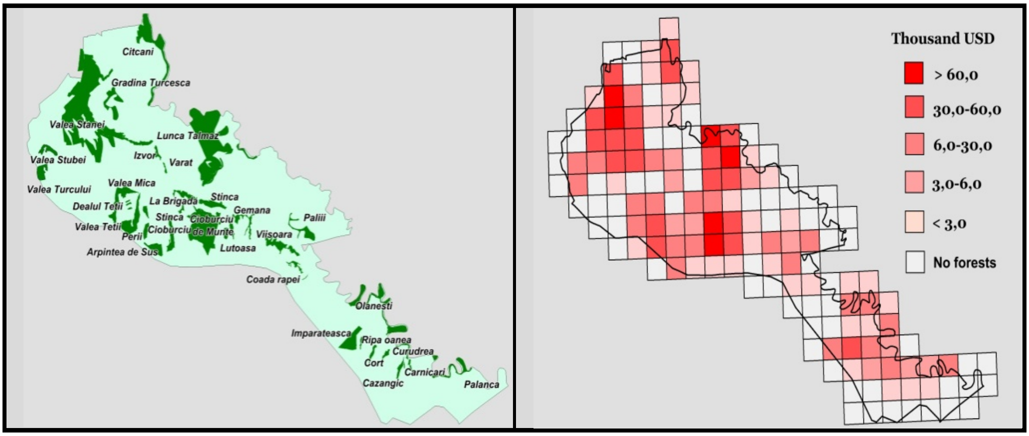


Fig. 3.2.5 General distribution of forest ecosystems *(left)* and economic values of their provisioning services in mapping units (*right*) in the Lower Dniester

Calculation of the economic value of provisioning services of the Lower Dniester's forest ecosystems was carried out, using above described Eq. 1 where, by analogy with TKP's (2013) recommendations, the values of its term were taken equal to: $\boldsymbol{p}$ = 0.3; $\boldsymbol{K_R}$ = 0.3; $\boldsymbol{K_{ev}}$ **=** 2.5 for oak and 0.5 – for others species; $\boldsymbol{K_{bp}} = 1.25$; $\boldsymbol{K_e}$ = 1.3 for oak and 1.0 for other species. Annual forest productivity was determined according to the total average woodland growth for the forests' particular type and growth class in m^3/ha/year: ***V =*** 3.0 for oak, 7.9 for poplar, 3.9 for acacia, and 3.3 for others. As a result, the average price of 1 m^3 of timber ***(P)*** equals 480 MDL (Andreev et al., 2017).

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. The economic value of forest ecosystem services, presented through mapping units (Fig. 3.2.5, *right*), varies from 0.3 to more than 60 thousand USD.

Forests of Copanca-Leuntea, Cioburciu-Răscăeţi and Lunca Talmaza (Fig. 3.2.5) have the highest value of production services; here oak forests of different species (in the first two) and poplar forests (in the last one) dominate in their composition.

3.2.4 Integrated valuation of provisioning services

3.2.4.1 Introduction

The methodology for **in**tegrated assessing and valuing ecosystem services and their losses has to be able to capture all multiple stressors effect on the services delivery, as well as to consider relationships between aquatic ecosystem status and services.

In particular, Maes et al. (2014) proposed an approach based on the assumption that a delivery of ecosystem services depends both on the spatial accessibility of ecosystems and on their conditions. Following this assumption, the working structure of the integrated assessment includes four steps:

- spatial mapping of ecosystems included in the assessment;
- assessment of these ecosystems conditions;
- quantification of the ecosystem services;
- integrated assessment of these components through combining the range of ecosystems and ecosystems services and their spatio-temporal relationships.

In the case of aquatic ecosystems, this working structure analyzes on the one hand – the ecological status of water bodies, and on the other hand – the ecosystem services delivery. Multiple pressures and their changes can result in the alteration of both. A main challenge here is to discover the complex relationships between stressors, status of ecosystems and their services, as well as to distinguish correctly the indicators of their conditions. This approach, proposed and realized in the MARS project[5], was also used in this work.

Depending on objectives of the economic valuation of ecosystem services and the scope of targeted results, two methodological approaches are usually used: (a) an approach based on the *integrated valuation* of ecosystem services, and (b) an approach based on the *element-wise valuation* of ecosystem services.

The methodology of *integrated valuation* of ecosystem services is based on the theory of environmental rent and the mechanism of its expression – the so-called opportunity cost; simultaneously, should take into account the efficiency of its reproduction in economic and environmental spheres. In the present work this approach was used in evaluating the provisioning services.

[5] *MARS* is the abbreviation of the ***M***anaging ***A***quatic ecosystems and water ***Re***sources under multiples ***S***tress research project, funded by the Seventh Research Framework Programme (FP7) of the European Commission (Grizzetti et al., 2015)

The *element-wise valuation* of ecosystem services is based, for example, in the assessment of carbon dioxide deposition by forests, the assimilation potential of forest ecosystems, the biodiversity conservation services, etc. In this work, through this approach the regulating services were evaluated.

3.2.4.2 Integrated economic valuation of natural ecosystems service

Calculation of the integrated value of natural ecosystem services (C_{ev}) is usually carried out for their three main types (forest, grass and water) according to the following equation:

$$C_{ev} = \sum R_{esi} * S_i \qquad (1)$$

Where: $\boldsymbol{R_{esi}}$ – current (annual) economic value of the *i-th* type ecosystem's service per ha;

$\boldsymbol{S_i}$ – area of *i-th* type ecosystem, ha.

In turn, $\boldsymbol{R_{esi}}$ is determined according to the equation:

$$R_{esi} = \left(R_i \frac{q_e}{q_{d_i}} - R_i\right) = R_i \left(\frac{q_e}{q_{d_i}} - 1\right) \qquad (2)$$

Where:

Ri – specific annual value *(differential rent)* of the *i-th* type ecosystem per ha;

$\boldsymbol{q_e}$ – capitalizer of an economic sphere, which is conditionally taken based on the specifics of this economic sphere (% of its annual growth); e.g., in (ТПК, 2013) it was accepted at the level of 0.05;

$\boldsymbol{q_{d_i}}$ – discount factor, which value is inversely proportional to the reproduction period of a consumed natural resource that forms the basis of *i-th* type natural ecosystem.

The *discount factors* q_{d_i} are equal (ТПК, 2013):

- for forest and grass ecosystems of national parks and reserves – 0.005
- for aquatic ecosystems (1/43 years) – 0.02
- for swamp ecosystems (1/1000 years) – 0.001.

Thus, according to *Eq. 2* current values of ecosystem services *(R_{esi})* equal:

- for forest and grass ecosystems = 9 R_i
- for aquatic ecosystems = 1.5 R_i
- for swamp ecosystems = 49 R_i.

Case study 3.2.3: Economic valuation of the current value of natural ecosystems service of the Lower Dniester wetlands

As was mentioned above, the integrated value of natural ecosystems services is usually carried out according to their three main types: forest, grass and aquatic. Since the economic valuation of forest ecosystems of the Ramsar site "Lower Dniester" has been examined in detail in Section 3.2.3, here EV only of two other types will be demonstrated.

Economic valuation of grass ecosystems. On the whole, in the Lower Dniester wetlands there are 78 grass plots. Their average area is about 0.54 km^2, significantly differing for individual plots (from 0.06 to 3.81 km^2); their mean perimeter is 3.65 km, and the resulting fragmentation coefficient CF equals 6.7, far exceeding the CF of forest ecosystems. The grass ecosystems fragmentation expressed through mapping units of the regular network is shown in Fig. 3.2.6.

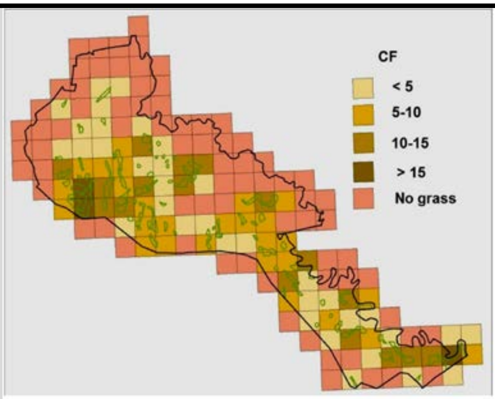


Fig. 3.2.6 Coefficient of fragmentation (*CF*) of the Lower Dniester grass ecosystems, %

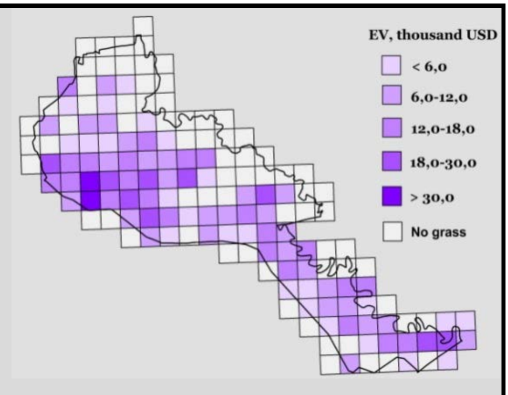


Fig. 3.2.7 Economic value of the Lower Dniester grass ecosystems providing service

Calculation of the current value for grass ecosystems *(R_i)* is carried out according to the following equation (ТКП, 2013):

$$R_i = \frac{P * K_R}{1 + p + K_R} * K_{out} * V$$

Where: $\boldsymbol{P}$ – market price of the main resource grass product *(hay)*; in our case, its average selling price is 700 MDL;

$\boldsymbol{p}$ = 0.3 – coefficient of profitability of the main grass product;

$\boldsymbol{K_R}$ = 0.3 – coefficient of efficiency of reproduction of the grass product;

$\boldsymbol{K_{out}}$= 0.95 – coefficient of the main grass product output, taking into account technological losses during its drying and transportation;

$\boldsymbol{V}$ – annual productivity of grass product per 1 ha (in our case, 3.5 t/km^2).

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 (Fig. 3.2.7) they vary spatially from six to more than 30 thousand USD. The grass ecosystems with the highest provisioning services value are located in the northwestern and south parts of this area, primarily due to the significant plots of high quality grass communities still surviving here.

Economic valuation of aquatic and swamp ecosystems. EV of water as an ecosystem itself was shown in Section 3.2.1. Therefore, this sub-chapter concerns only some ecosystems that are dependent on water.

The total area of water bodies in the Lower Dniester wetland accounts for about 0.63 thousand hectares (~6.3 km^2), or 1.1% of its territory; their average area is about 0.251 km^2, significantly differing for individual objects – from 0.01 to 1.93 km^2. The total area of swamp ecosystems is about 1.2 thousand ha (~12 km^2), or 2% of the whole territory; their average area is about 0.506 km^2, also significantly spatially differing (from 0.003 to 6.87 km^2). The morphometric characteristics of the water and swamp ecosystems, required for the assessment of their fragmentation, are given in Table 3.2.4.

Table 3.2.4 Morphometric characteristics of the water and swamp ecosystems in the Ramsar site "Lower Dniester"

Ecosystem	Number of objects	Mean area, km^2	Mean perimeter, km	Coefficient fragmentation
Water	25	0.251	2.248	8.95
Swamp	24	0.506	3.491	6.90

As can be seen from this table, the aquatic ecosystems are more fragmented than those of the swamps. Therefore, as a result of the comprehensive review of all ecosystem types in the Lower Dniester wetland, its total fragmentation decreases compared to the fragmentation of individual ecosystems, primarily due to a cumulative effect, as well as to their spatial discrepancy in the distribution over the territory of this site. Moreover, 20% of the territory is not provided with the considered types of natural ecosystems, and the level of provision with the natural environment stabilizing complexes can be considered satisfactory only for one third of the territory. The highest diversity is characteristic of the natural Copanca-Leuntea, Lunca Talmaza and Tudora-Palanca complexes, mainly due to their swamps.

According to corresponding calculations, the cost of the provisioning services of aquatic ecosystems (Fig. 3.2.8) in the Ramsar site "Lower Dniester" is about 11.4 million MDL, which is approximately equivalent to 0.7 million USD or 320 USD per ha of water surface. Such significant cost of aquatic ecosystem services here is associated, first of all, with significance of water for this Ramsar site, which already experiences obvious climate change and aridization with a tendency to their further increase (Corobov and Trombitsky, 2019).

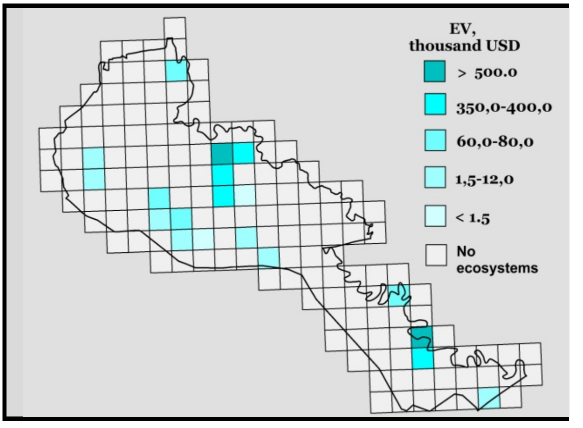


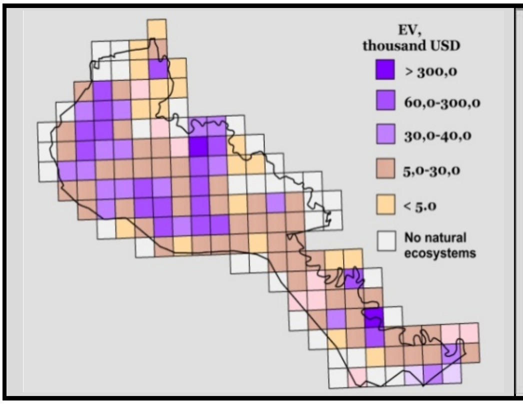
Fig. 3.2.8 Economic value of provisioning services of the Lower Dniester aquatic ecosystems

The integrated value of provisioning services of the Lower Dniester natural ecosystems. Assessment of the integrated value of ecosystem services is carried out through summing the corresponding results for all ecosystems under study. The area of each *i-th* ecosystem type's zone *(Si,* ha*)* within the Ramsar site was determined, using cartographic materials.

As a result of the calculations, the obtained total value of provisioning services of the Lower Dniester ecosystems amounts to about 3.2 million USD or 203 USD per ha (Table 3.2.5). At the same time, significant territorial differences were observed caused by the uneven distribution of various ecosystems throughout this Ramsar site and their individual qualitative characteristics. The spatial distribution of the integrated values expressed in mapping units is shown in Fig. 3.2.9; it varies from 5 thousand to more than 300 million MDL.

Table 3.2.5 Integrated value of the Low Dniester natural ecosystems' provisioning services

Ecosystem	**Area, ha**	**Economic value, USD**	
		Per ha	*Total*
Forest	9,005	162	1,46
Water	2,090	320	0,67
Grassland	4,569	231	1,06
Total	***15,664***		***3,18***



Fi. 3.2.9 Total economic value of natural ecosystems service in the Lower Dniester

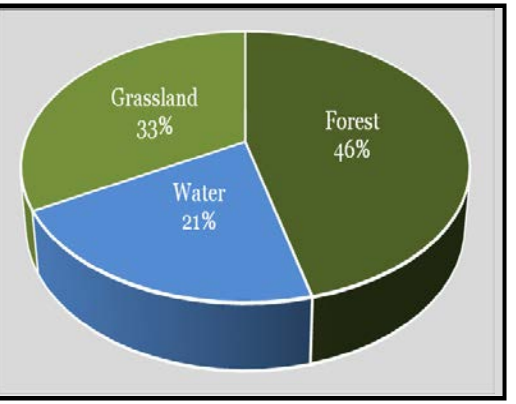


Fig. 3.2.10 Contribution of individual ecosystems to the total value of their provisioning services in the Lower Dniester

The highest integrated value of provisioning services is associated with combinations of forest, grass and water ecosystems, which together provide high productivity of their main natural resource products (in our case – wood, hay and fresh water). In turn, in the structure of these services, the highest share falls on forest ecosystems (46%); grass and water ecosystems account for 33% and 21%, respectively (Fig. 3.2.10).

3.3 Economic valuation of regulating ecosystems services

3.3.1 Economic valuation of carbon deposit services

3.3.1.1 Carbon deposit by forest ecosystems

Valuation of the annual carbon dioxide absorption by a forest ecosystem is calculated by the following equation:

$$R_{CO_2} = P_{CO_2} * A \qquad (1)$$

Where: $\boldsymbol{P_{CO_2}}$ – average world price of one ton of CO_2 absorption
$\boldsymbol{A}$ – absorption of CO_2 by a forest ecosystem, ton/year.

In turn, term ***A*** is calculated by Eq. 2:

$$A = \sum K_{i_{CO_2}} * S_i \qquad (2)$$

Where:

$\boldsymbol{K_{i_{CO_2}}}$ – specific indicator of CO_2 absorption by *i-th* forest-forming species, t/ha/year
$\boldsymbol{S_i}$ – area of *i-th* forest-forming species.

Box 3.3.1 Carbon deposit service of the Lower Dniester forest ecosystems

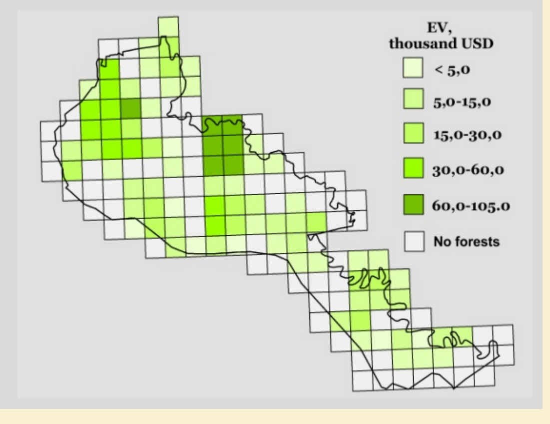


Fig.3.3.1 Spatial distribution of the economic value of an annual CO_2 deposit service by the Lower Dniester forest ecosystems

For this valuation there were used data from the Forest Research and Management Institute of Moldova. According to its estimates, CO_2 accumulation for the main forest-forming species in Moldova is (in ton/ha/year): oak – 7.7, poplar – 10.7, white acacia – 8.4, other species – 4.1. Based on the data on species composition and area that each species occupies the total accumulation of CO_2 by the Lower Dniester's forest ecosystems has been determined (Andreev et al., 2017). The resulting current economic value of annual carbon deposit service of these ecosystems is 1.53 million USD, or on average – 168 USD per ha. Spatial distribution of this service in the Lower Dniester is shown in Fig. 3.3.1, varying across the territory by its mapping units from less than 5 to 105 thousand USD.

The absorption price of one ton of CO_2 is accepted as the average price of the CO_2 emission quota in the framework of the European Union Emissions Trading System (EU ETS; see: https://ec.europa.eu/clima/policies/ets_en). For example, in March 2020 the average price of CO_2 allowance was 24.1 EUR, or 31.1 USD, increasing, for example, from 5.8 EUR/ton in 2017.

Realization of this method is demonstrated in Box 3.3.1.

3.3.1.2. 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 CO_2 by these ecosystems equals 0.705 ton/year (TKP, 2011).

Economic value of CO_2 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.

3.3.2 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 *(E_{ap})* of water-related forest ecosystems is calculated as the sum of corresponding estimates for individual pollutants (fluorine compounds, sulfur dioxide, nitrogen oxides, hydrocarbons, etc.) by the following equation:

$$E_{ap} = \sum_{ijn} \frac{1}{T_{ij}} * O_{ijn} * T_n \qquad (3)$$

Where:

$\boldsymbol{O_{ijn}}$ – ultimate load of *n-th* pollutant on *i-th* tree species stand of *j-th* type of forest, expressed in natural terms, ton;

$\boldsymbol{T_{ij}}$ – actual age of the *i-th* tree species stand of *j-th* type of forest, years;

$\boldsymbol{T_n}$ – pollutant emission charge taking into account the hazard class of *n-th* pollutant.

In this equation, the maximum load of pollutants on tree species is determined by the equation:

$$O_{ijn} = H * Y * V * K_{ok} * S_{ij} \qquad (4)$$

Where:

$\boldsymbol{H}$ – Maximum possible content of *n-th* pollutant in pine needles as a species that is most sensitive to gaseous toxicants, t/t. The maximum load of main pollutants on pine phytocenoses is: for sulfur *(S)* – 0.0013 t/t; for nitrogen *(N)* – 0.02844 t/t; for fluorine *(F)* – 0.00012 t/t. When the content of harmful substances is higher than the specified value of ***H***, then a toxic effect of these substances on pine forests is taken (TPK, 2013);

$\boldsymbol{Y}$ – Coefficient of forest phytocenoses resistance to influence of *n-th* pollutant for: pine – 1.0; spruce – 1.29; small-leaved species – 1.86; broad-leaved species – 2.14 (Белоусова, 2001);

$\boldsymbol{V}$ – Average stock of stands, m^3/ha;

$\boldsymbol{K_{ok}}$ – Volume-conversion coefficients for converting the volume stock (stock change) of stem wood (m^3/ha) to the mass of individual phytomass fractions (t/ha);

$\boldsymbol{S_{ij}}$ – area of the estimated stands of *i-th* species of *j-th* forest type, ha.

The maximum load ***(H)*** on forest ecosystems of other toxic compounds, due to insufficient knowledge on their harmful effects, can be determined by introducing into calculation the hazard coefficient of *i-th* pollutant. The calculation is based on pine, and the above-mentioned coefficients for other species are introduced. Table 3.3.1 gives *H*-estimate of sulfur, nitrogen and fluorine on 1 ha of different tree species.

Table 3.3.1 Calculation of the maximum load of sulfur *(S)*, nitrogen *(N)* and fluorine *(F)* on different forest species, ton/ha.

Sp	T_{ij}	H	Y	V	K_{ok}	O_{ijn}
Oak						
S	53	0.0013	2.14	150	1.038	0.433
N	53	0.02844	2.14	150	1.038	9.476
F	53	0.00012	2.14	150	1.038	0.040
Poplar						
S	27	0.0013	2.14	186	0.834	0.432
N	27	0.02844	2.14	186	0.834	9.441
F	27	0.00012	2.14	186	0.834	0.040
Acacia						
S	18	0.0013	2.14	69	0.677	0.130
N	18	0.02844	2.14	69	0.677	2.843
F	18	0.00012	2.14	69	0.677	0.012
Other						
S	40	0.0013	1.86	124	0.677	0.203
N	40	0.02844	1.86	124	0.677	4.441
F	40	0.00012	1.86	124	0.677	0.019

Note: Names of symbols in the table heading as in Eq. 3 & 4.

However, it should be noted that insufficient knowledge on the nature of pollutants harmful effect leads to significant differences in the values of coefficients of hazard (aggressiveness) in regulatory documents of different countries. So, the comparison of hazard coefficients in the regulatory documents of Moldova and Belarus shows their significant discrepancy. For example, some coefficients in these countries respectively equal: for nitrogen – 25 and 41.1, for sulfur – 22 and 16.5, for fluorine – 200 and 980. This is one of reasons for the discrepancy in cost estimates of the assimilation potential for forest ecosystems.

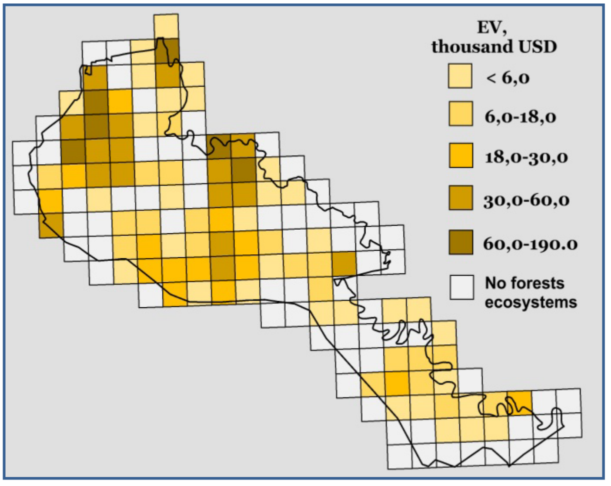
Case study 3.3.1: The assimilation potential of the Lower Dniester's forest ecosystems

The assimilation potential of forest ecosystems can be considered as a kind of an equivalent to reducing the load of pollutants on the environment. From this viewpoint, any cost of preventing or reducing the contaminating emissions by an ecosystem can be considered as an economic value of its regulatory service. In turn, to evaluate the assimilation potential of forest ecosystems, the national standards and procedures for charging pollutant emissions are usually used. For example, from 1.01.2020 in Moldova the standard payment for pollutant emissions from stationary sources is 21.6 lei for a ton in large cities and 17.3 lei – in the country's regions. The standard charge for an emergency emission is increased by 50 times (see: http://base.spinform.ru/show_doc.fwx?rgn=118653). The actual mass of a pollutant is converted into conventional tons through its multiplying by the coefficient of this pollutant aggressivity (Table 3.3.2).

Table 3.3.2 The coefficient of aggressivity *(CAg)* for some pollutants released into the air in Moldova

Pollutant	CAg	Pollutant	CAg
Nitrogen dioxide	5.0	Soluble fluorides	100
Nitrogen oxcide	20.0	Hydrogen fluoride	200
Sulfurous gas	22.0	Fluorine compounds	200
Hydrogen sulfide	54.8	Carbon monoxide	1.0

Using the data in Table 3.3.2, the corresponding calculations were carried out, and 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. As can be seen from Fig. 3.3.2, the economic value of the forest



ecosystems assimilation potential, expressed in mapping units, varies here from 6 to about 90 thousand USD per ha.

Fig. 3.3.2 Spatial distribution of the economic value *(EV)* of the Lower Dniester forests' assimilation potential

3.3.3 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. Such an indirect valuation of the wetland ecosystem services for natural water treatment *(Ow)* is determined by the equation:

$$O_w = O_i * \sum \frac{S * \lambda_n}{\lambda_i} \qquad (5)$$

Where:

- $\boldsymbol{O_i}$ – present value of an industrial treatment plant (conventionally assumed to be 1,000 USD);
- $\boldsymbol{S}$ – area of the corresponding type of a swamp, ha;
- $\boldsymbol{\lambda_n}$ – filtering ability of this swamp, m^3/day/ha;
- $\boldsymbol{\lambda_i}$ – filtering capacity of a selected industrial treatment plant, m^3/day.

The example of economical valuation of a sorption function is presented in Box 3.3.2.

Box 3.3.2 EV of a sorption function of the Lower Dniester swamps

The economic valuation of the water purification function of swamp ecosystems of the Ramsar site "Lower Dniester" was made according to Eq. 5 above. The values of corresponding terms in this formula were accepted as: filtering ability (λ_n) – 137 m^3/day per ha of lowland swamps[1]; filtering capacity of treatment plant (λ_i) – 1,500 m^3/day[2]. Based on the swamps area of this Ramsar site, the economic value of their absorption services is about 107 USD or 91 USD per ha on average. However, this value, expressed in mapping units, varies from 1,000 to more than 30,000 USD (Fig. 3.3.3).

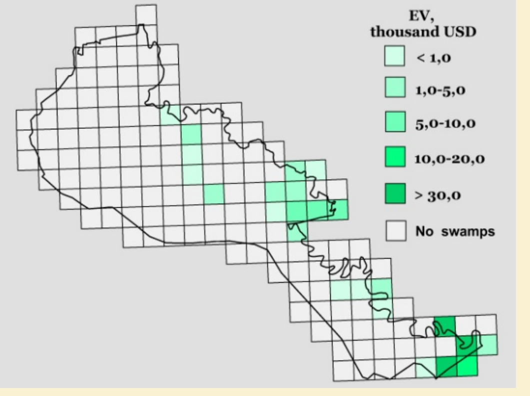


Fig. 3.3.3 Spatial distribution of the economic value of swamps water-cleaning function

Source:

[1]Бобылев С.Н., Медведева О.Е., Сидоренко В.Н. и др. 1999: *Экономическая оценка биоразнообразия*. М., 70 с.;

[2]Шимова О.С., Лопачук О.Н., 2007: Методические аспекты экономической оценки водно-болотных экосистем. *Природные ресурсы* **4**:115-121.

3.3.4 Water protection and water regulation services

Water regulation services, firstly of the forests, consist in equalizing seasonal fluctuations in an annual surface runoff, preventing its sharp reductions and thus reducing the intensity of floods. Forests ability to equalize intra-annual runoff fluctuations is expressed by the ratio of its surface and underground components. Redirecting the surface runoff into subsoil and ground, the forests contribute to an increase in an underground runoff, thus decreasing the surface runoff and, consequently, reducing soil erosion. Forests and forest litter promote an increase in soils water permeability through slow down of snow melting and mitigating soils freezing. So, forests in the forest-steppe and steppe zones are able to almost completely transfer surface runoff to underground.

Thus, the forest is a buffer that very efficiently absorbs superficial water flow. In the period of snowmelt, the soil under forests absorbs 10-15 times more moisture than the field plots with under-winter ploughing. The total amount of water absorption by a field soil ranges from 40 to 50 mm, while by the forest soil – from 450 to 500 mm and even more (Молчанов 1973). According to Ануфриев (2013), the forested catchments make it possible to transfer up to 95% of the surface runoff to an underground part, thanks to what evenly providing the territory with water resources in the future.

However, existing studies on the role of forests in water conservation do not yet allow coming to unambiguous estimates of an optimal forest cover, and there are different concepts, both linear and nonlinear, of the dependence of runoff characteristics on the degree of afforestation. Due to a large number of regional factors affecting the forest functions in water regulation, it is a rather difficult task to reveal the relationship between a forest cover and runoff characteristics. The diversity of estimates has led to forming an idea of the uncertain or unstable hydrological function of the forests (see e.g. Онучин, 2013).

In a general case, the water-regulating function, which depends on the increase in underground flow, is estimated by the following equation (Неклюдов, 2011):

$$\Delta S = X * \alpha * K_1 * \mu * (C_1 * K_2 * K_3 * K_4 - C_2) \qquad (6)$$

Where:

- ΔS – annual increase in underground flow, mm
- X – average annual precipitation, mm
- α – coefficient of river flow
- μ – percentage of summer precipitation
- K_1 – swamp coefficient
- C_1 and C_2 – coefficients of underground runoff of forested and treeless territories
- K_2 – coefficient characterizing the plantation age
- K_3 – coefficient characterizing the plantation class
- K_4 – coefficient depending on the plantation density.

The value of the river flow coefficient α depends on a natural zone of vegetation and relief (e.g., mountain or plain). Coefficients of underground runoff (C_1 and C_2) depend on the level of forest cover, type of plantations and soil texture. The swamp coefficients (K_1) are inversely related to the wetland itself: the more severe a wetland, the lower this coefficient's value. The values of coefficients characterizing the plantations age *(K_2)* and density *(K_4)* depend directly on these characteristics: the older a plantation and the greater its density, the higher this coefficients' value. As to the class of a plantation *(K_3)*, there is an inverse dependence: the higher its class, the smaller this coefficient value.

To determine an economic effect of the forests water-regulatory function, the utility payments for water use and water tariffs for industrial enterprises can be used as the financial equivalent.

A case study of practical realization on the discussed methodic is demonstrated in Box 3.3.3

Box 3.3.3 **Economic value of the water-regulation services of the Lower Dniester ecosystems**

The areas with slopes >5°, where forest ecosystems contribute most of all to the surface-underground redistribution of surface runoff, occupy more than 20% of the Lower Dniester territory (Fig. 3.3.4). This factor provides a significant increment in ecosystem water-regulating functions.

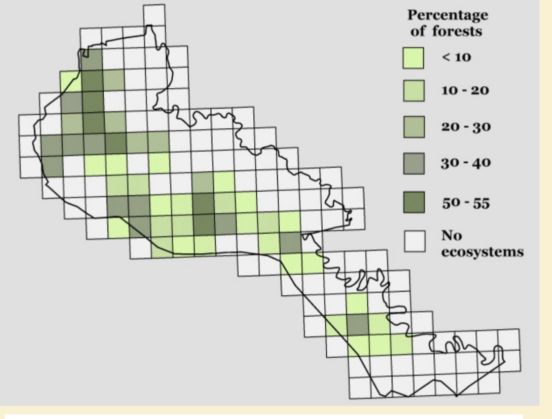


Fig. 3.3.4 Distribution of sloping forest ecosystems in the Lower Dniester

To assess the water-regulating effect of forest ecosystems on the underground here, the following parameters were laid down in Eq. 6: annual precipitation – 525 mm; river flow coefficient α – 0.5; percentage of summer precipitation – 70%; coefficients of underground runoff C_1 for a forested area (20%) – 0.3 and for a treeless area C_2 – 0.12; coefficient of plantations age K_2 – 1.0; coefficient of plantations class K_3 – 0.6; coefficient of plantation density K_4 – 0.7. The swamp coefficient was not used in this study. As a result, the increase in underground runoff due to the water-regulation function of forests amounts:

$$\Delta S = 525 * 0.5 * 0.7 * [0.3 * 1.0 * 0.6 * 0.7 - 0.12] = 16.54 \text{ mm, or } 165.4 \text{ m}^3/\text{ha}$$

Based on the area of sloping forest ecosystems, the volume of water accumulation is ~485,000 m^3. If to use the average payment for water of industrial enterprises as of 31.99 MDL/m^3, the total economic effect of this accumulation is 11.87 million MDL.

3.4 Economic valuation of habitat services

3.4.1 Background to research

Habitats provide everything that an individual plant or animal needs to survive: food, water, shelter. In this framework each ecosystem provides different habitats that can be essential for a species' lifecycle, while *habitat services* highlight the importance of ecosystems to provide such habitats both for local and migratory species. In the second case this is especially important for migratory species, which, for example birds and fish, depend upon different ecosystems during their migrations. Along with these tasks, the habitats promote to maintenance of bio- and genetic diversity within species populations. Some habitats have an exceptionally high number of species, which makes them more biologically diverse than others, being known as 'biodiversity hotspots'.

In turn, the biodiversity of natural resources provide a range of goods and services that have fundamental importance for human wellbeing, health, livelihood, and even for survival. First of all, this concerns the poorest nations that have the greatest immediate dependency on natural biological resources, such as food, fuel, building material or natural medicines. Moreover, much of the world's biological resources are located in developing countries, being under the greatest threat from human driven pressures (both inside and outside the developing world). These threats include, along with population and economic growth, new challenges, including climate change and hydropower development (MEA, 2005).

For example, water regulation run by the Dniester hydropower complex, significantly changes the seasonal natural rhythms of the Dniester River runoff that often leads to drainage of its delta in spring and summer. This regulation negatively affects not only the wellbeing of a local population, but also all living organisms: fish, amphibians, birds. In the drying Dniester's delta, according to prognosis of Щеголев at all (2016), in the next 20 years, with continued water-accumulating operation of the DHPC, the final degradation of freshwater ecosystems and the inevitable extinction of main wetland birds species can occur.

Therefore, understanding the role of biodiversity in securing livelihoods and wellbeing of people, and its economic valuation is important since it provides a useful vehicle to highlight and quantify the range of delivered benefits. To place the monetary values of biodiversity and its ecosystem services will bring them into a common currency for use in decision-making, allowing direct comparing their benefits with other trajectories of sustainable development (Christie et al., 2008).

In the last decades the considerable researches to examine how people value biodiversity have been undertaken. The majority of these works have been conducted in the developed world (e.g., Christie et al., 2006, 2007; DEFRA, 2007), but with limited application in developing countries (e.g., Christie et al., 2008; Georgiou et al., 2006). Such situation is caused by low knowledge of foreign literature, the lack of local research capacity and insufficient capacity to build awareness of biodiversity importance, by a high cultural diversity, etc. (Christie et al., 2008). The use of non-economic techniques, such as questionnaires, focus groups, participatory appraisal approaches and other tools to assess the importance of biodiversity has been suggested as a possible way to address some of these issues (Grizzetti et al., 2015; Maes et al., 2013).

There are also other types of works. In particular, one of the latest global reviews of global estimates of the value of ecosystems and their services (De Groot et al., 2012) was conducted for 10 biomes and 22 types of ecosystem services. From 320 publications with 1,350 estimates, these authors selected 650 works that are comparable for an analysis, and their average, minimum and maximum values were calculated. However, these estimates significantly fluctuated for different ecosystems services, sometimes by several orders of magnitude due to a fact that, as was shown e.g. by Costanza et al. (1997), the value of ecosystem services was not valued directly by the market. Sometimes, the importance and corresponding values of ecosystems are driven mainly by an attitude towards them. In particular, the great attention paid to wetlands has led to a kind of paradox when the total value of their services was estimated at 25,682 USD/ha per year, far exceeding the services of tropical and moderate forest biomes (respectively 5,264 and 3,013 USD/ha per year), or grasslands – 2,871 USD/ha per year.

Sometimes it is not clear how the proposed techniques can best complement economic approaches to elicit values and provide meaningful results that can inform policies at national and international levels.

3.4.2 Economic valuation of biodiversity

Usually, economic valuation of biodiversity is carried out, using the method of replacement cost of main representatives of the animal world. According to this method, the total cost of a biological species restoration ***Ci*** is determined as follows (ТКП, 2013):

$$C_i = V_i * N_i \tag{1}$$

Where: $\boldsymbol{V_i}$ — replacement cost of one individual of *i*-type species;

$\boldsymbol{N_i}$ — total number of individuals of *i*-th species living within the study area.

When calculating the replacement cost of one individual *(V_i)*, there are used a resource value *(k_r)* and increasing *(k_m)* coefficients indicating that species belongs to those included in the Red Book or subjected to international treaties, including Convention on the International Trade in Endangered Species of Wild Fauna and Flora (CITES Convention; available at: https://www.cites.org/sites/default/files/eng/cop/08/E-Resolutions.pdf):

$$V_i = k_r * k_m \tag{2}$$

Where: $\boldsymbol{k_r}$ — resource value coefficient taking into account the resource value of wildlife objects; it is taken as a fine size for the destruction of one specimen species in accordance with the law;

k_m — increasing coefficients

For example, according to (ТКП, 2013), k_m equal:

2 — for species covered by international treaties

3 — for wild animals belonging to species included in the Red Book

A case study of this approach use is given in Box 3.4.1.

Source: Blog of I. Vanstein. http://www.winstein.org/publ/1-1-0-682

Box 3.4.1: **Economic valuation of bird losses due hydropower impacts**

As was shown above, the water-regulating Dniester hydropower complex had changed significantly the volume and seasonal distribution of the Dniester streamflow, often causing draining of its delta. Such destructive influence of the complex on main representatives of the Dniester delta's natural ecosystems has manifested in a catastrophic reduction in the number of populations (by 70-99%) of almost 80% of its fauna. In particular, in the last decades about 160 species of birds have disappeared here, and currently a number of their species that form a background for the Dniester delta's biodiversity are on the verge of extinction (Щеголев et al., 2016). Among them such rare species as glossy ibis (*Plegadis falcinellus*), yellow heron (Ardeola ralloides), bittern (*Botaurus stellaris*), gray goose (*Anser anser*), gray-toad grebe (*Podiceps griseigena*), etc. can be called. In particular, a glossy ibis – an endangered species of sacred ibis birds – is listed in the Red Book of Moldova and Ukraine. Before the beginning of DHPC construction, it was the most widespread bird in the Dniester delta where 2,500-3,000 of its adult individuals nest steadily in 1970-1982. In 1988-2002 the number of breeding loafs decreased here by 8-11–14-25 times, ranging from 100 to 350 adults individuals. In 2003-2009 the decrease was continuing further – to 40-150 adults, and in subsequent years (2010-2015) this bird has disappeared almost completely from the Dniester delta as a breeding species.

According to the Ukrainian legislation, the penalty for the destruction of one glossy ibis is 12,063 hryvnas (~ 434 US dollars*), that taking into account the increasing coefficient k_m (Eq. 2) equal to ~1,302 USD. Considering the amount of fines as a kind of compensation for one of the lost environmental services, the total losses from reducing the number of glossy ibises as a result of hydropower negative impacts on the environmental conditions in the Dniester delta is about 3.9 million USD, even without taking into account the coefficient of reproducibility. In reality, this figure is much larger.

* At the official rate 28.14 hryvnas on 01.01.2020 (*https://bank.gov.ua/)*

3.4.3 Benefit transfer in economic valuating the habitat services

In some cases, the valuation of biodiversity, considered as an ecological resource, is accepted as a capitalized value of the current cost of services for its conservation. In particular, the estimation of costs of the Lower Dniester wetland's biodiversity conservation (Cazanteva et al., 2019) was based on the approach using a so called "reference value". The application of such unit of measure is caused by a requirement to ensure that uniformity of any economic valuation should be provided through the identity of units used for its measurement. Such reference value is obtained based on available information on the cost of maintenance/conservation of biodiversity services, by calculating the average cost per unit area. According to its idea, this approach is close to the *"benefit transfer"* method that is used in situations where significant local expert knowledge and resources cannot be provided. In situations of this kind, the economic valuation, as it has been already noted above, is conducted by transferring information available from the studies completed in another location and context (GEF, 2018). Recall also that within the scope of the cited Guidance, this approach is referred as a *"tier 1"* project.

Some details of this approach use are demonstrated in Cazanteva et al. (2019) and Case study 3.4.1.

Case study 3.4.1: The economic valuation of the Lower Dniester ecosystems service

In the presented case study, the value of biodiversity, accepted as a "reference value", was found by averaging the literature information on the assessed values of particularly rich in relation to their biodiversity territories. As a result, two indicators: the average minimum (3,520 USD) and the average maximum ($6,705 USD), both per hectare, were identified as the reference values for economic valuation of ecosystems service in "key territories" of the Lower Dniester wetland (Fig. 3.4.1).

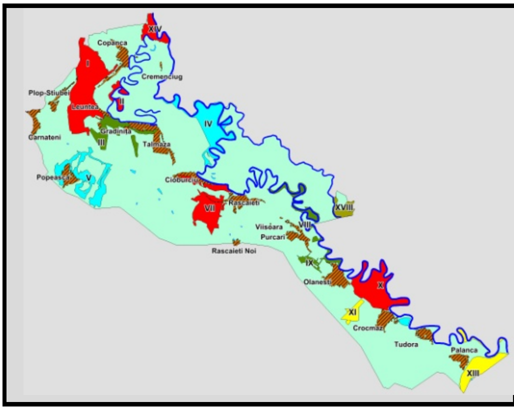


Fig. 3.4.1 Key territories of the Ramsar site "Lower Dniester" by their importance: *red* – international; *blue* – national; *yellow* – local

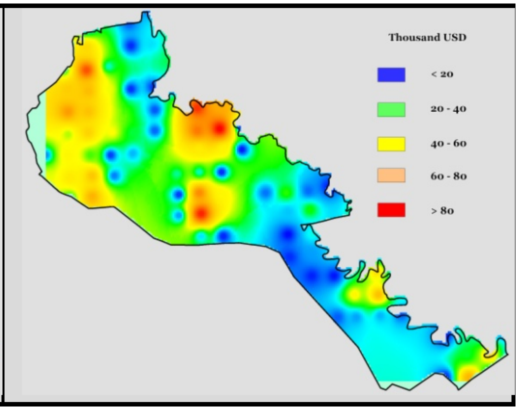


Fig. 3.4.2 Economic value of biodiversity in the Ramsar site "Lower Dniester"

Additionally, to take into account the quality and productivity of ecosystems, an additional coefficient was introduced into the reference values. This so-named *estimating coefficient* was based on the before made estimation of the Ramsar site Lower Dniester fragmentation (see Section 2.2).

Some sufficiently large areas of the site, which are mainly on balance of local authorities and are occupied by low quality tree vegetation, have also been taken into account in the conducted economic evaluations, but with a reduction factor of 0.1.

The results of the economic valuation of the biological diversity in the key territories of the Lower Dniester are shown in Table 3.4.1. These areas also serve as territory-cores of the national ecological network of Moldova (Andreev et al., 2012). The spatial interpolation of biodiversity values in the mapping units (Fig. 3.4.2) demonstrates pronounced territorial differences in the economic value of this wetland's ecosystems.

Table 3.4.1 Economic values of key territories *(KT)* biodiversity in the Lower Dniester

KT Code	Key territory	Area, ha	Estimating coefficient	Value, millions of USD	
				by minimum reference value	by maximum reference value
I, III	*Copanca-Leuntea, Tufa-Talmaza*	3306.5	3	34.9	66.5
II	*Grădina Turcească*	251.0	3	2.7	5.0
IV	*Lunca Talmaza (Bălţile Talmaziene)*	1686.5	5	29.7	56.5
V	*Popeasca*	1188.0	4	16.7	31.9
VII	*Cioburciu-Răscăeţi*	1192.1	4	16.8	32.0
VIII	*Răscăeţi -Olăneşti*	883.7	2	6.2	11.9
IX	*Purcari*	115.0	1	0.4	0.8
X	*Olăneşti -Crocmaz*	1614.2	2	11.4	21.6
XI	*Impărăteasa*	266.6	1	0.9	1.8
XIV	*Pădurea Chitcani*	398.5	2	2.8	5.3
XVIII	*Diculi-Cuţa*	266.0	3	2.8	5.4
Forests under local authorities		2557.9	0.1	0.9	1.7
	Total			***135.6***	***258.4***

3.5 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" (MEA, 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). These authors also proposed a framework of cultural ecosystem services and distinguished their four types with regards to ecosystem services on the whole, namely: playing and exercising; creating and expressing; producing and caring; gathering and consuming. Somewhat another list of cultural ecosystem services is proposed by GEF (2018); as an example, it was shown in Table 2.1.

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. In particular, in the case of bringing the losses of past years to the present time the following formula is applied: $Yb = U_t\ (1 + E)^t$; to bring the losses to the future period – the formula: $Yb = Y_t(1 + E)^{-t}$, where Yb is a discounted loss, Y_t is an actual loss in t-year, E is a discount rate. To assess the total losses for a certain period of time, the actual losses of each year, reduced to one time point, are added. As a discount rate in case of an anthropogenic impact with specific environmental consequences, experts recommend using the rate of 0.03-0.05 (Бруна & Зубарев, 1998). In particular, for the case analogous to that under discussing here, these authors selected the rate of 0.05 (Box 3.5.1).

Box 3.5.1 Economic valuation of the recreation service in the Dniester Riverside trip

Any change in a river flow regime and temperature-humidity conditions in its basin, caused either by climate warming or hydropower plant functioning, as has been shown in the previous sections, leads often to negative impacts on aquatic and riparian ecosystems. This inevitably affects the attractiveness of the recreational areas and, consequently, yields in the loss of possible income of the inhabitants of a riverside strip.

So, as of the 1990s, in a 1 km-wide strip along the Dniester floodplain downstream the Dniester reservoirs’ dam, there were lived approximately 23 thousand people, or about six thousand families. In those years, according to statistical survey data on the structure of rural households’ income, every 10th family supposedly took summer residents. The potential annual income of these families was about 5,000 MDL, and thus the total economic value of current losing this cultural ecosystem service could be 26,000 MDL per year, or 5,600 USD at the 1997 exchange rate. Taking into account the gradual accumulation of losses from the moment when DHPC was commissioned, the economic value of past losses could amount to ~32,000 USD.

Source: Based on Брума & Зубарев (1998) estimations

4. ECONOMIC VALUATION OF CHANGE AND LOSSES IN ECOSYSTEM SERVICES

Schematically, Grizzetti et al. (2015) express the logical sequence of economic valuation of changes in ecosystems services, caused by a new status of the environment, as follows:

Change in Pressures → Change in Ecosystem Status → Change in Ecosystem Services → Change in Value

Since from these four steps the assessment of changes in ecosystems status and ecosystem services was considered in Chapter 3, this Chapter is dedicated mainly to the description of impacts and valuation of corresponding losses in ecosystem services. Based on the purpose of this document, its main emphasis is placed on valuating the economic consequences of impacts on ecosystems due to functioning of hydropower under climate change.

4.1 Change in pressures on ecosystems

4.1.1 Changes in the regional climate

As a low carbon technology, the hydropower makes a significant contribution to achieving the targets of the Paris Climate Agreement and the Sustainable Development Goals. At the same time, any failure to consider adequately new risks due to HPP operation may lead to shortcomings in its technical and financial performance, water security aspects, and environmental functions. If not designed and managed appropriately, the hydropower projects could exacerbate the impacts of climate change on local communities and the environment. Furthermore, without assessing the climate change-related opportunities, the investment decisions may not adequately recognize the role of hydropower infrastructure in providing climate-related services. This also includes a hydropower's role in supporting the greater use of less flexible forms of low-carbon electricity generation (IHA, 2019).

Because most rivers are within watersheds, which already are stressed by human activities, the climate change will add to or will magnify present stresses through its potential to alter precipitation, temperature or runoff patterns, and, correspondingly, to disrupt biological communities and their ecological linkages. Many communities will face shrinks in their water supplies due to temperatures rise; shifts in precipitation pattern can result in dramatic impacts through drying the environment, threatening public health, weakening economies and decreasing quality of life.

Usually, as two key climatic indicators to analyze potential impacts of observed changes in climate on regional ecosystems and their services, the air temperature and precipitation are chosen. Temperature is a basic physical factor that affects many natural processes through altering precipitation and runoff patterns, affecting availability of freshwater supplies and thus leading to a wide range of adverse consequences for ecosystems. Precipitation, both in the form of rain and snow, is a primary source of water supply.

To identify evidences of changes in climate, it is necessary to compare different time periods. The statistical analysis of climate differences in these periods includes usually two items:

(1) the comparison of trends in main climatic variables to indicate observed tendencies in climate dynamic

(2) the comparison of seasonal and annual averages of these variables to indicate change in climate conditions that form a runoff in river watersheds.

As a case study, in this subsection the changes in regional climate of Moldavian part of the Dniester basin will be considered. The initial material included historical observation of temperature and precipitation at weather stations located here (11 stations). Additionally, this part of the basin was divided into two sub-basins: from the DHPC's dam to the Dubossary HPP and from the later to the river mouth. Hereafter, these sub-basins will be conditionally named as the Dniester's Upper and Lower parts. As two climatic periods there were compared 1961-1990 and 1991-1918 respectively reflecting a relatively "normal" historical regional climate and the climate of intensive global warming that started in the 1990s.

The simplest and most frequently encountered method for describing a change in a climatic variable over time is a linear trend, where in its equation *($y = a_o + a_i t$)* the coefficient a_i characterizes the average rate of a measured variable dependence on a time unit.

Fig. 4.1.1 shows plots of annual temperature trends in areas under study in two periods. It is clearly visible that both trends are identical in many ways. In particular, the negligible (0.02°C/decade) increase of mean annual temperature in the Dniester's Upper part and even some decrease in its Lower part (-0.08°C/decade), which were observed in 1961-1990, have been changed by a more intensive increase to about 0.6°C/decade in 1991-2018. A more detail comparison of trends for other temperatures parameters is shown in Table 4.1.1 where '*slope*' means a temperature change per year, while *p-value* characterizes statistical significance of estimated relationships.

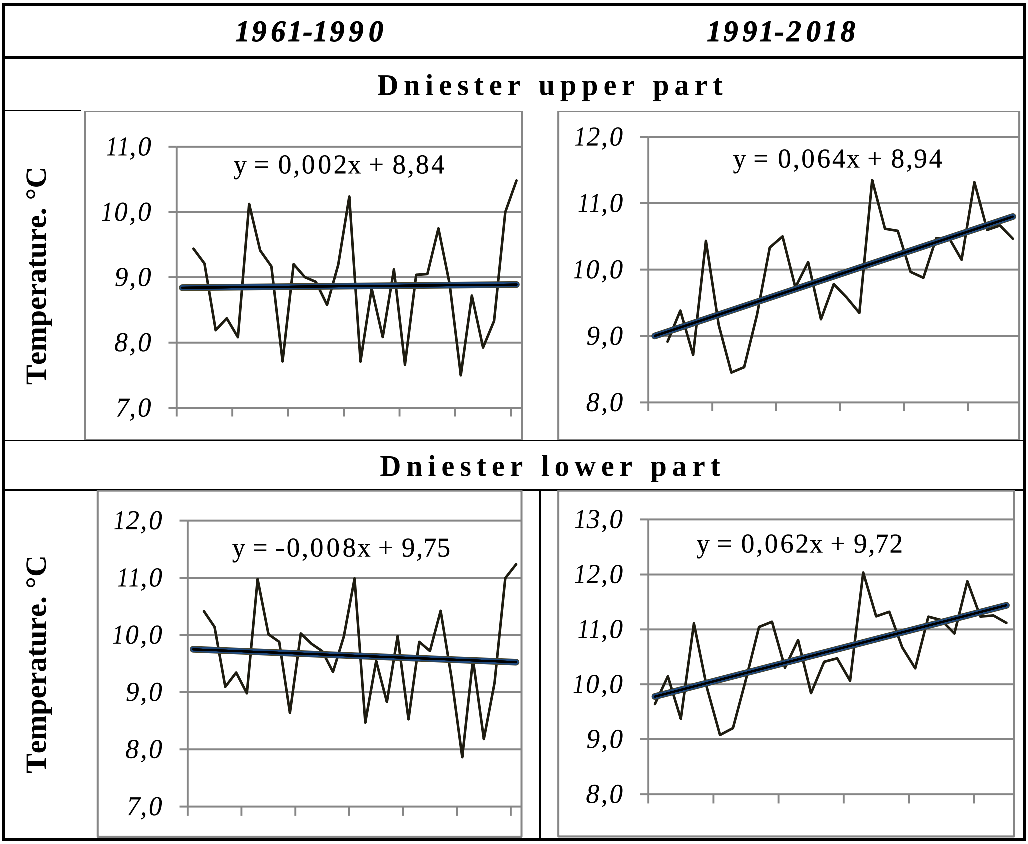


Fig. 4.1.1 Linear trends of annual mean temperature in the Dniester basin in two climatic periods

Table 4.1.1 Statistics of annual trends of air temperature parameters in two climatic periods

Years	Study area	Air temperature					
		Minimal		*Mean*		*Maximal*	
		slope	p-value	slope	p-value	slope	p-value
1961-1990	Upper Dniester	0.007	0.678	0.002	0.924	0.016	0.332
	Lower Dniester	-0.001	0.938	-0.001	0.938	-0.018	0.646
1991-2018	Upper Dniester	0.040	0.022	0.064	0.000	0.080	0.000
	Lower Dniester	0.041	0.034	0.062	0.000	0.094	0.006

Analysis of Table 4.1.1 leads to the following conclusions:

- in both observation periods, the trends of mean (T*mean*), maximum (T*max*) and minimum (T*min*) temperatures are almost identical in their direction;
- in *1961-1990*, both an increase (positive slope) and decrease (negative slope) of temperatures were extremely small in their absolute values and statistically insignificant: *p*-value is much more than 0.01, which is a permissible value in such estimations;
- in *1991-2018,* a sharp temperature increase took place, which amounted to about 0.4-0.6°C, 0.6-0.8°C and 0.8-0.9°C per decade for T*min*, T*mean* and T*max*, respectively; with a few exceptions, these trends were highly significant (in most cases $p < 0.001$).

Results of the ***precipitation trends*** comparison are shown in Fig. 4.1.2 and Table 4.1.2.

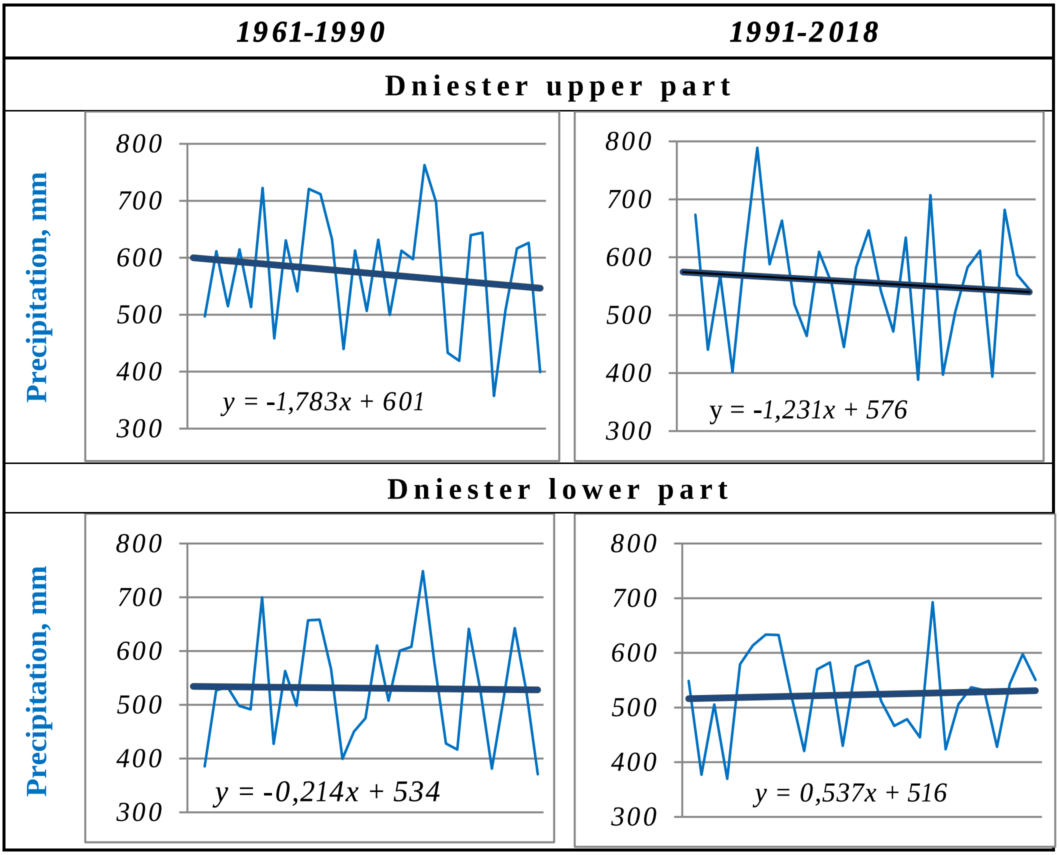


Fig. 4.1.2 Linear trends of annual precipitation in the Dniester basins in two climatic periods

In 1961-1990 in both basins a slight decrease of annual precipitation was observed: about -2 mm per year in the Upper Dniester and about -0.2 mm per year – in its lower part. With the global warming increase, the negative trend of precipitation slightly weakened in the Upper Dniester, but increased in the Lower Dniester (up to 0.5 mm per year). However, observed changes are too small: in all models *p*-values are much higher than permissible limits of confidence (p >> 0.05).

Table 4.1.2 Comparison of annual precipitation trends in two climatic periods

Years	**Study area**	**Slope**	**p-value**
1961-1990	Upper Dniester	-1.789	0.433
	Lower Dniester	-0.220	0.919
1991-2018	Upper Dniester	-1.224	0.624
	Lower Dniester	0.530	0.787

And, at last, in Table 4.1.3 there are shown results of the seasonal average temperatures and precipitations comparison in the Dniester basin.

Table 4.1.3 Seasonal temperature and precipitation averages in two climatic periods in the Dniester basin

Season	**Years**	**Study area**	***Air temperature, ºC*** Tmin	Tmean	Tmax	***Precipitation, mm***
Winter	*1961-1990*	Upper Dniester	-5.30	-2.45	2.52	105
		Lower Dniester	-8.00	-1.74	4.74	110
	1991-2018	Upper Dniester	-4.41	-1.52	5.47	94
		Lower Dniester	-6.28	-0.85	7.79	98
Spring	*1961-1990*	Upper Dniester	4.43	9.26	14.68	133
		Lower Dniester	5.02	9.75	15.25	122
	1991-2018	Upper Dniester	4.96	10.29	16.20	129
		Lower Dniester	5.36	10.72	16.69	118
Summer	*1961-*1990	Upper Dniester	13.60	19.34	25.40	221
		Lower Dniester	14.76	20.33	26.32	192
	1991-2018	Upper Dniester	14.85	21.01	27. 53	200
		Lower Dniester	15.92	21.97	28.45	178
Autumn	*1961-*1990	Upper Dniester	4.91	9.41	14.59	113
		Lower Dniester	5.80	10.20	15.21	108
	1991-2018	Upper Dniester	5.53	9.89	15.00	113
		Lower Dniester	6.21	10.63	15.80	133
Year	*1961-*1990	Upper Dniester	4.42	8.89	14.32	572
		Lower Dniester	5.24	9.67	15.38	532
	1991-2018	Upper Dniester	5.23	9.92	16.05	555
		Lower Dniester	5.90	10.67	17.18	*526*

The main conclusions from the analysis of Table 4.1.3 can be summarized as follows:

- the direction and magnitude of seasonal changes in all variables is in good agreement with the findings of an annual trends analysis;

- the reasonable proximity of principal results obtained for two sub-basins indicates the reliability and quality of a chosen assessment method;
- against the background of positive trends in air temperature, the warmer and more arid climate is clearly seen in the Lower Dniester;
- the observed realities require their consideration in all economic valuation on changes in aquatic ecosystems service.

More details of the observed study can be found in Corobov et al. (2019a).

4.1.2 Hydropower impacts on the rivers streamflow

Hydropower generation is an element of water infrastructure. However water, additionally to serving this infrastructure, needs to be able to support the complex interactions of all ecosystems and their components in the basin and to address different social and economic purposes upstream and downstream of a hydropower dam. Thus, the benefits derived from hydropower generation should be included in any analysis only as long as they are provided on a sustainable basis, without severe impacts on nature, ecosystems and their services. Coordination efforts between water, energy and environment sectors are especially challenging under the ongoing changes in climate (IHA, 2019). The complexity of coordination increases substantially in transboundary river basins where the impacts spread from one country to another, and trade-offs and externalities may cause frictions between riparian countries. Moreover, in spite of some common features, every river basin has significant differences requiring their careful thorough study and accounting for in the process of transboundary monitoring a river's flow.

Undoubtedly, among many negative effects of damming a riverbed for hydropower needs, the changes in its flow volume and regime are most important. Namely these changes entail all others consequences. Therefore, any economic valuation of hydropower operation impacts on water related ecosystems should begin by the assessment of changes in the river flow. Statistical approaches to solving this problem are almost identical to those used in a study of any processes that is changing over time. As minimum, they include the comparison of trends, as indicators of observed tendencies in a streamflow dynamic, and the comparison of its seasonal and annual averages, as indicators of water discharge that maintains the ecosystems state.

In this Guide again, as a case study of the hydropower impacts assessment, the changes in the Dniester streamflow are considered. As the source of impacts the DHPC operation was selected.

The presented assessment of DHPC impacts was based on a comparative analysis of the Dniester water discharge *(Q)* in periods before and after this complex construction (1951-1980 and 1991-2015, respectively). Usually, tasks of this kind are solved by measuring water discharge at hydrological posts located at the HPP's reservoir entrance and downstream of its dam. Therefore, in this study there were used historical observations of water discharge at three posts: Zalishchyky (56 km upstream of the Dniester reservoir) and Mohyliv-Podilskyi (hereafter sometimes Mohyliv) and Bender, located respectively in above 40 km and 450 km downstream of its dams.

Since the methodology of statistical comparison was described in Subsection 4.1.1, in this analysis only the obtained results will be presented.

Water discharge trends. In 1951-1980, before the filling of DNPC reservoirs, in all parts of the Dniester basin and during all seasons the *positive trends* of *Q* were approximately the same by their shape. The differences in absolute values of trend slopes at three posts are explained by a natural increase of water flow downstream: from the smallest values in Zalishchyky (~4.15 m^3/s per year) to the largest – in Bender (~7.51m^3/s per year).This increase was statistically significant with a high level of confidence for annual and seasonal Q in almost all cases, except for the winter-spring period in Zalishchyky (p = 0.401).

A completely different picture was observed in 1991-2015. The previous statistically significant increase in the average annual water discharge was replaced by its ubiquitous annual decrease, although small (~1-2 m^3/s per year) and statistically insignificant for all seasons (except autumn when *p-value* < 0.05). The change in trends direction of the Dniester streamflow was undoubtedly associated with an air temperature increase and precipitation decrease in the Dniester catchment caused by global warming, the intensification of which has been distinctly manifested since the 1990s (Corobov et al., 2019a). However, negative trends in Q can be fully explained by global warming only in Zalishchyky, where the impact of the Dniester reservoir is completely excluded. The trends observed here could serve only as indicators of climate change impact on the Dniester runoff that is formed in its catchment upstream the reservoir. Reasons for the trends change downstream should be sought in *DHPC creation and operation*. As confirmation of this conclusion, unlike the water discharge in Zalishchyky, which in 1991-2015 decreased in all seasons, in Mohyliv-Podilskyi its slight increase was observed in the spring and summer, and in Bender – in winter.

Hydropower impact on water discharge. The comparison of Q upstream and downstream of the DHPC dam has shown:

- In all seasons a gradual increase in annual Q was clearly visible from the Dniester's source to its mouth: from 142.5 m^3/s in Zalishchyky to 320.1 m^3/s in Bender in 1951-1980 and, respectively, from 160.6 m^3/s to 283.6 m^3/s in the subsequent years;
- Approximately 13% increase in the last three decades of winter streamflow in Zalishchyky indicates an earlier onset of snowmelt caused by climate warming, just like above 27% increase in autumn streamflow resulting from an autumn precipitation increase (Corobov et al., 2019a). However, against the not disturbed by HHPs increase in winter-spring streamflow in Zalishchyky, the Q decrease at two other posts clearly indicates winter water accumulation in the Dniester reservoir. The maximal and statistically significant Q decrease took place in spring that is an evident manifestation of the DHPC's *negative impact,* because at this time the Lower Dniester's ecosystems, for example, ichthyofauna and its spawning grounds, especially require sufficient water volumes for their well-being. In autumn, an additional inflow from Dniester tributaries, more abundant precipitation and some decrease in water requirements strengthen Q increase in the Lower Dniester.
- Thus, a fact that in 1991-2015 in Zalishchyky, despite the above shown trends towards a decrease in annual Q, its some increase (by about 4%) compared to the previous period has preserved, while in Mohyliv-Podilskyi and Bender it has decreased (respectively by 6.3% and 11.4%), should undoubtedly be attributed to influence of the DHPC complex.

***Changes in the streamflow annual regime*.** Damming of riverbeds and HPPs operation affect not only a total streamflow, but also transform the river annual regime. As to the Dniester River, while the monthly distribution of an undisturbed water discharge in Zalishchyky in both periods has not essentially changed, it has changed drastically downstream of DHPC after its construction. There is an accumulation of spring streamflow in the Dniester reservoir (March-April), expressed as a decrease of differences between Q in two compared periods. An analogous difference is not observed in Zalishchyky, indicating the spring accumulation of water in the Dniester reservoir.

Hydropower impacts on volumes of river flow. A *river runoff (W)* is the volume of water passing any location for a certain period of time. Thus, this parameter, expressed in km^3, is a more obvious indicator for any impact comparison, both in temporal and spatial dimensions. For example, In 1991-2015, with a slight increase in annual *W* (by 0.25 km^3) in Zalishchyky, it has decreased by about 0.6 km^3 in Mohyliv-Podilskyi and 1.1 km^3 in Bender, thereby confirming the DHPC impact on the Dniester downstream flow. In the same years, in all seasons there is observed a certain decrease in the maximum *W* along with an increase in its minimum, which also can be explained by a regulatory function of Dniester reservoirs. As a result, the range of average interannual flow volumes fluctuation decreased by 2.4 km^3 in Zalishchyky, but already by 3.4 km^3 downstream the DHPC.

These considerations are illustrated in Table 4.1.4 where the DHPC's impact is estimated through contribution of individual sections of the Dniester catchment to the total flow volume at Bender hydrological post, conditionally taken as 100%. So, the results for 1951-1980 confirmed well-established estimates that approximately 2/3 of the Dniester annual flow is formed in its basin's upper part (68.9% in Zalishchyky); in Mohylev this share increased to 87.2%. After DHPC construction, a share of runoff in Zalishchyky increased by 10.7%, but in Mohylev – only by 3.8%. Thus, now the Ukrainian Carpathians generates about 4/5 (79.6%) of Dniester annual runoff! Another 11% is formed due to the lateral tributaries in the river sub-catchment from Zalishchyky to Mohylev-Podiskyi and only 9% – in the rest part of the catchment.

Table 4.1.4 The Dniester River absolute *(km^3)* and relative *(%)* streamflow volume upstream and downstream the DHPC as compared to its value at the Bender hydrological post, considered as 100%

	Post	Winter		Spring		Summer		Autumn		Year	
		km^3	%	km^3	%	km^3	%	km^3	%	km^3	%
1951-1980	*Zalishchyky*	1.10	65.9	2.62	66.8	2.13	74.2	1.18	66.7	7.03	68.9
	Mohyliv	1.48	88.6	3.33	85.0	2.54	88.5	1.54	87.0	8.89	87.2
	Bender	1.67	100	3.92	100	2.87	100	1.77	100	10.22	100
1991-2015	*Zalishchyky*	1.25	69.1	2.63	89.2	2.00	79.7	1.39	73.5	7.28	79.6
	Mohyliv	1.45	80.1	2.62	88.8	2.53	100	1.73	91.5	8.33	91.0
	Bender	1.81	100	2.95	100	2.51	100	1.89	100	9.15	100

Source: Corobov et al. (in press)

4.2 Economic valuation of changes in natural ecosystem services

In this section, the demonstration of economic valuation of possible losses in natural ecosystem services is carried out on the example of Moldavian part of the Dniester floodplain. For a more detailed valuation, it was divided into separate sub-areas *(clusters),* already used in Chapter 2 (see Section 2.2).

4.2.1 Forest ecosystem services

The forest ecosystems, associated with water, occupy 78.7 km^2, or up to 10% of the Moldavian part of the Dniester floodplain. Ecosystem services of these ecosystems depend to a large extent on their species composition; therefore, the first step in forest ecosystems EV is to quantify main forest species. For the study area such information is presented in Table 4.2.1. As can be seen from this table, poplar and oak predominate among floodplain forests. However, these total values vary across individual clusters that leads to differences in the ecosystem services they provide.

Table 4.2.1 The species composition of forest ecosystems in the Dniester floodplain within Moldova*

Clusters	Area, km^2	Forests area		Structure of forest species				
				Oak	*Acacia*	*Poplar*	*Other*	
		km^2	%	km^2	km^2	km^2	km^2	%
DHPC - Dubasari	56.11	2.78	4.96	0.92	0.29	0.07	1.51	54.1
Dubasari reservoir	93.71	3.83	4.08	0.60	2.05	0.08	1.10	28.7
Dubasari – Raut mouth	5.05	0.33	6.48	0.19	0.03	0.03	0.08	25.2
Raut - Ichel mouths	40.94	2.60	6.34	0.30	0.24	1.09	0.96	37.1
Ichel – Bic mouths	178.8	32.77	18.33	12.99	3.30	7.01	9.48	28.9
Bic – Botna mouths	74.72	7.05	9.43	1.95	0.01	3.31	1.78	25.3
Botna *mouth* - Liman	336.3	29.36	8.73	5.04	0.28	12.37	11. 7	39.7
Total	*785.6*	*78.72*	*10.02*	*22.00*	*6.20*	*23.94*	*26.6*	*33.8*

* *Source:* Adapted from Andreev et al. (2017)

Economic values of the forests' ecosystem services per hectare, calculated according to the methodology proposed in a corresponding subsection (3.2.3), are given in Table 4.2.2. Multiplying these values by the clusters area (Table 4.2.1) gives the total value of corresponding forest ecosystems service on their territory.

Table 4.2.2 Economic value of the forest ecosystems service in the Dniester floodplain

Clusters	**Ecosystem service**				
	Provisioning	Carbon sequestion	Assimilation potential	**Total**	
	USD/ha	USD/ha	USD/ha	USD/ha	USD
DHPC - Dubasari	266.5	183.3	195.7	645.6	2438,6
Dubasari reservoir	187.1	221.5	206.0	614.6	658,7
Dubasari – Raut mouth	382.0	221.9	248.3	852.2	2735,1
Raut mouth - Ichel mouth	215.2	239.0	354.9	809.2	210,0
Ichel mouth – Bic mouth	317.8	229.2	287.5	834.5	27,9
Bic mouth – Botna mouth	293.5	254.9	386.0	934.4	235,2
Botna *mouth* - Liman	240.2	234.7	355.6	830.5	179,8
Average	*275.4*	*231.9*	*316.6*	*823.8*	*6485,4*

The results of these estimations have shown that the largest share in the economic value of the Dniester floodplain forests' ecosystem services belongs to an assimilation potential (39%); the provisioning services account for 33% and carbon sequestration – for 28% (Fig. 4.2.1.). At the same time, in the floodplain's different sections, the contribution of each ecosystem services to their total economic value varies significantly (Table 4.2.2). So, the value of provisioning services ranges from 187.1 to 317.8 USD/km^2, of carbon deposit services – from 183.3 to 254.9 USD/km^2, and of assimilation potential – from 195.7 to 386 USD/km^2. Accordingly, depending on the structure of ecosystems and the area of each cluster, the total value of provided ecosystem services also changes (Fig. 4.2.2).

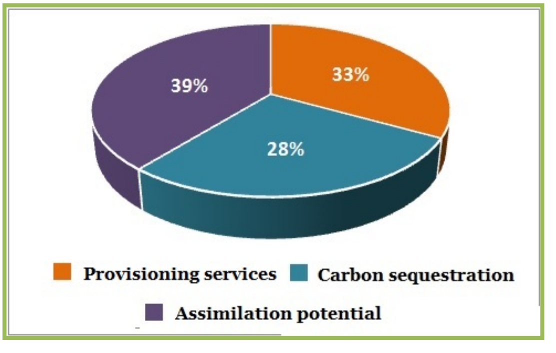


Fig. 4.2.1 The general structure of forest ecosystems services in the Dniester floodplain within Moldova

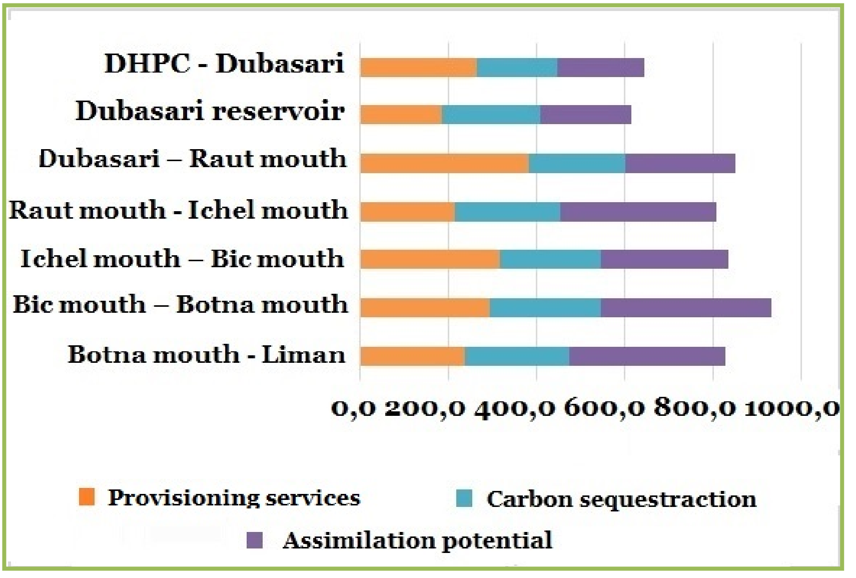


Fig. 4.2.2 The economic value *(thousand USD)* of forest ecosystems service in the Dniester floodplain by its individual clusters

The analysis of the dynamics of changes in the Dniester floodplain's forest ecosystems revealed an increase in their area by 9% in the period from 1979 to 2018. Based on this increase, it was possible to estimate a corresponding increase in the value of their ecosystem services: *provisioning* – by 195 thousand USD; *an assimilation potential* – by 173 thousand USD; *carbon sequestration* – by 224 thousands USD.

4.2.2 Grass ecosystem services

The area of grassland areas in the Dniester floodplain within Moldova is 242.4 km^2, and the estimated cost of their provisioning ecosystem services is 231 USD/ha or about 56 thousand USD for the area on the whole. It can be assumed that a 9% increase in forest area, which took place in 1979-2018, was accompanied by a commensurate 9% decrease in grass ecosystems. Accordingly, the economic value of the decrease in their provisioning ecosystem services amounted to ~ 504 thousand USD.

4.2.3 Aquatic ecosystem and wetland services

The area of aquatic ecosystems in the Dniester floodplain under the study is 5.79 km^2; the economic value of their provisioning ecosystem services equals 320 USD/ha. In 1979-2018, the decrease in the area of aquatic ecosystems here amounted to about 11% that has led to a decrease in the value of their *provisioning services* by about 20 thousand USD.

In addition, in the estimated period of changes, in this area new wetlands were formed, and to-day their area amounts to 11% of the floodplain total area. Considering the wetlands' total area (38.9 km^2) and based on the trend of their expansion, an estimated increase in the economic value of their *water purification* function amounts to 390 thousand USD, with a unit value of 91 USD/ha.

The specific value of *carbon sequestration* by wetland ecosystems is about 22 USD per ha. Based on an increase in their area by 11%, the economic value of this ecosystem service in the Moldavian part of the Dniester floodplain has increased by 9.4 thousand USD.

4.3 Economic valuation of hydropower impacts on the Dniester floodplain's natural ecosystems

The economic valuation techniques, discussed in the previous sections, can be used to identify changes in the value of ecosystems services under influence of a wide range of different factors, including hydropower. Since the economic valuation of changes in ecosystem services is based, along with purely natural and biological factors, on accounting the changes in areas occupied by respective ecosystems, it seems these changes should be one of the variables in such assessments. Let us consider addressing this kind of problem on the following case study.

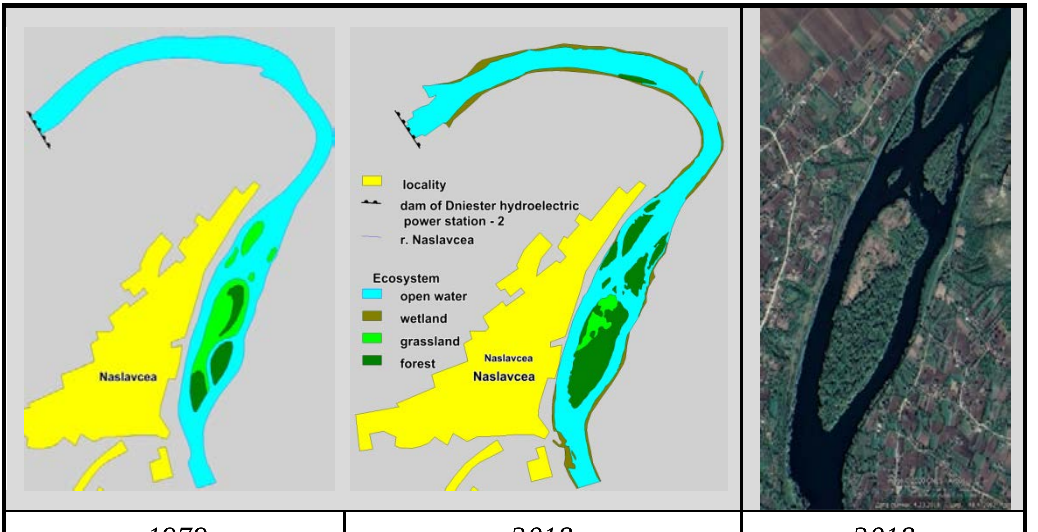
In 20 km downstream of the DHPC, near the village Naslavcea (see photo below), several small islands are located in the Dniester floodplain (Fig. 4.2.3). In this figure, two middle left maps reflect the state of this natural complex in periods before and after of DHPC construction, respectively in 1979 and 2018. Both the change in size and components composition of this natural landscape is clearly seen.

Two trends are evident:

1) decreasing of an open water surface due to its overgrowing and transforming into wetlands;
2) increasing of the forest ecosystems area vs. reducing grasslands (Table 4.2.3).

General view of Naslavcea

Source: https://ru.wikipedia.org/wiki/Naslavcea



1979	*2018*	*2018*

Fig. 4.2.3 Map diagrams of the study area before *(left)* and after *(right)* DHPC construction, and its Google map as of 2018

Table 4.2.3 Change in the case study area in 1979-2018

Ecosystem	1979		2018		Change
	km²	percent	km²	percent	km²
Open water	1.0599	80.20	1.0598	68.64	-0.0001
Wetland	0.000	0.00	0.1756	11.37	0.1756
Forest	0.0996	7.54	0.2669	17.29	0.1673
Grassland	0.1620	12.26	0.0421	2.73	-0.1199
Total	*1.3215*	*100.00*	*1.544*	*100.00*	*0.2225*

In particular, in the area under consideration practically during the period of the DHPC construction and functioning, due to a decrease of the Dniester streamflow rate and in number of floods, its riverside has increased by 0.2225 km^2, or by 16.8%. In the main channel, new small islands have appeared, covered with trees and shrubs, and wetlands were formed as a result of riverside grasslands swamping. The observed increase in areas occupied by forests also was partially due to a herbaceous ecosystems decrease.
Along with changes in ecosystem areas, their structure has also changed (Fig. 4.2.4).

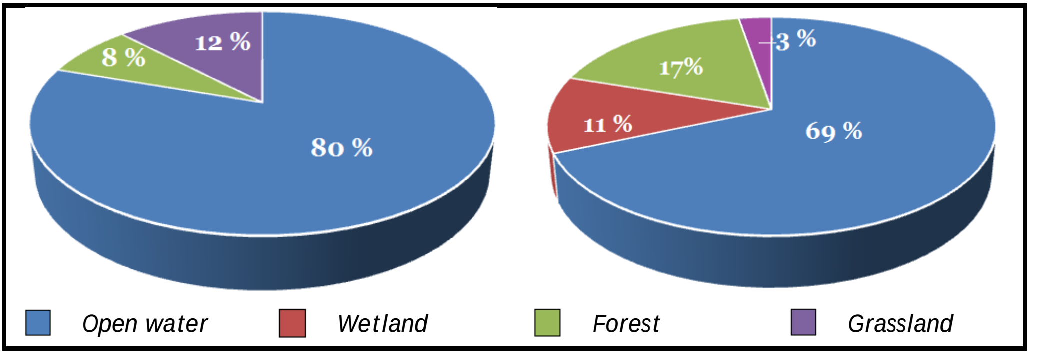


Fig*.4.2.4* Ecosystems structure in the evaluated area in two compared years (1979 – *left;* 2018 –*right*)

Thus, based on the redistribution of ecosystems areas, an assumption can be made about an increase in the provisioning and regulating services of forest ecosystems, as well as the regulating services of wetland (by 9 and 11%, respectively). At the same time, the ecosystem services of grass ecosystems, due to reductions in their area, are correspondingly decreasing.

5. Indicator for water-related ecosystems service assessment and economic valuation

The European Environmental Agency defines an environmental indicator as "…a measure, generally quantitative, that can be used to illustrate and communicate complex environmental phenomena simply, including trends and progress over time — and thus helps provide insight into the state of the environment. Indicators are designed to answer key policy questions and support all phases of environmental policymaking, from designing policy frameworks to setting targets, and from policy monitoring and evaluation to communicating to policy-makers and the public" (https://eniseis.eionet.europa.eu/east/indicators/indicators-search/#b_start=0).

MARS project (Grizzetti et al., 2015), basing on the literature review and according to the type of providing information, organized the potential indicators/proxies for water ecosystem services in three categories: *natural capacity, service flow* and *social benefit*. On the whole, these indicators refer mainly to the ecosystem services delivered by rivers, lakes, groundwater, riparian areas, floodplains, wetlands, transitional and coastal waters. Based on the MERS project's list, Table 5.1 provides indicators that are acceptable for assessing the impact on ecosystems and ecosystem services of hydropower and climate change.

Table 5.1 List of indicators for biophysical assessment of ecosystem services

Ecosystem services	Natural capacity	Service flow	Social benefit
Water for drinking	Surface water availability Renewable accessible water supply Water storage capacity River salinity Nitrate-vulnerable zones	Water consumption for drinking Water abstracted Water exploitation index Consumptive water use by end user	Proportion of population using drinking water from the source under study
Water for non-drinking purposes	Surface water availability Total riverwater resources Water storage capacity	Water abstracted per sector Water exploitation index Area water-logged by irrigation	Total water requirements Cost of water and water delivery

Water purification	Indicators on surface water quality Nutrient concentration Trophic status Ecological status Presence of nitrophilous macroalgae or macrophytes Area occupied by riparian forests Presence of floodplains and wetlands	Nutrient loads Nutrient retention Removal of nutrients by wetlands Amount of waste processed by ecosystems Sedimentation and accumulation of organic matter	Access to safe water Value of ecosystem waste treatment and water purification
Fisheries and aquaculture	Fish population status (species composition, age structure, biomass) Absolute fish abundance Relative fish abundance (catch per unit effort) Condition of fish stocks Number of wild species used for human food	Fish catch Aquaculture production Fish production from sustainable sources (e.g., proportion of fish stocks caught within safe biological limits Wild vegetation used in gastronomy	Number of fishermen Employment in fishing and related sectors Value of fish or value of aquaculture sales Marginal value of a change in fisheries management
Maintaining populations and habitats	Biodiversity (species diversity or abundance, endemics or red list species, spawning areas) Ecological status Coverage, condition and structural complexity of nursery and feeding areas Macrophyte species richness	Habitat suitability Species abundance and richness Habitat change Juvenile density Postlarvae production per hatchery Community perception on the importance of habitat provision Economic value of the annual juvenile fish production based on the price of aquaculture growth	Habitat suitability Species abundance and richness Habitat change Juvenile density Postlarvae production per hatchery Community perception on the importance of habitat provision Economic value of the annual juvenile fish production based on the price of aquaculture growth

Carbon sequestration	Organic carbon stored or carbon stock Above and below ground biomass Carbon in soil or sediments Dissolved organic matter	Carbon sequestration or carbon change Carbon uptake Soil carbon accumulation	Quantity of carbon fixed combined with the marginal damage costs of carbon emissions Market value of carbon
Recreation and tourism	National Parks and Natura 2000 sites Number of beaches Fish and waterfowl abundance Condition of fish stocks Quality of fresh waters for fishing Size of river leisure and recreation hotspots Cover and smell of decomposing algae	Number of visitors to natural places (National Parks, lakes, rivers, protected wetlands) Number of visitors to attractions (e.g. thermal or mineral sources) Number fishing licenses and fishing reserves Number of bathing areas Number of waterfowl hunters, anglers and amateur fishermen	Tourism revenue Traffic census [2] Amount or spending on nature tourism Beach visitors and travel cost Tourists' perception in a marine protected area
Intellectual and aesthetic appreciation	National Parks and Natura 2000 sites Proximity of rivers or lakes to urban areas Monitoring sites by scientists Fish studies as a source of information Seabird populations	Cultural sites and number of annual cultural activities organised Classified sites (e.g. World Heritage) Number of visitors Number of scientific projects, articles, studies, patents Number of educational excursions at a site Number of TV programmes, studies, books etc.	Changes in the number of residents and real estate values Comparative value of real nature estate /cleaner water bodies Price of a hotel room Willingness to pay for improvement in the environment/ improved water quality Taxes and subsidies supporting maintaining open space

			Financial expenditure in research
Spiritual and symbolic appreciation	National species or habitat types Rare species Cultural landscape	Sacred or religious sites Number of sites or species fundamental to performance of rituals Number of visitors Number of (environmental) associations registered	Changes in the number of residents and real estate values Incentives to maintain traditional cultural landscapes

Source: https://natworld.info/raznoe-o-prirode/prirodnye-jekosistemy-vidy-harakteristika-i-foto

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