Ministry of Education and Science of Ukraine

Odessa State Environmental University

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ASSESSMENT OF MODERN STATE AND MANAGEMENT OF AQUATIC ECOSYSTEMS

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The tutorial allowence to outline the teaching of the various issues and processes impacting on water systems must assume that the student has a background up to third year university science with a speciality in a natural science. Even though the course would be of benefit to social sciences and related disciplines a simple knowledge of science is required. The course is directed at fourth year university students and, with an appropriate research dissertation, could be used as the basis for a higher-level degree, e.g. Honours, Diploma or Masters. Network Environmet Assesment and Remidiation Curriculum in Natural Environmental Science,Terre et Environnement, 2005, 2010 Vol. 50,88, materials were used for the preparation of this lecture course.

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INTRODUCTION
(Common information about nature using)

Science graduates are traditionally not trained to interact with the general public or with the media, despite a large and increasing public appetite for information on science and technology. This is particularly true for environmental concerns where public interest and involvement have a major driving effect on political decision making. The scientist thus has a responsibility to communicate scientific findings to the public. In democratic political systems, the scientist has the right to seek public support for scientific programmes essential to the sustainable management of ecological resources, including the sustainable management of water. Under conditions where the scientist is prohibited from public interaction, or is punished or discriminated against for attempting such interaction, the political commitment to democratic freedom must be suspect.

The explosive growth of the human population during the past 50 years, added with the ever -increasing water demand per capita has resulted in a dramatic increase of human pressures on natural water supplies. These fast growing pressures led to general mismanagement of water resources, affecting all links of the water abstraction-distribution-use-collection-treatment-discharge management chain (even though in most cases worldwide this theoretical chain is never complete) (Baron *et dl.*, 2002). Therefore, nowadays, water has become a very scarce natural resource as a result of human activities. Moreover, once impacted by human activities, the ability of water resources to recovery from pollution varies from days to years in rivers and lakes, and to centuries in groundwater.

Protection of freshwater resources is paramount and the role of the public in this task, both directly and indirectly, is essential. Public participation can only be achieved by public awareness. Once impacted, water must be treated to protect its most sensitive use, nominally for drinking water purposes. It is in the public interest to ensure the best quality for natural waters and to ensure the best treatment is available to meet the standards required for drinking water, or for any other use that may be required, e.g. irrigation. Both protection and treatment require.

The nature using (NU) is sphere of social and productive activities for human needs using the natural resources. According N. Reymers (Reymers, 1990) there is different definition of the term: 1) the aggregate of all forms of exploitation of the natural resources potential; 2) the totality of the productive forces and relations of production, management and economical forms and institutions using the natural objects for human needs; 3) using the natural resources in a process of social production for material and cultural needs; 4) the combined impact of humanity on the geographical envelope of the Earth; 5) complex scientific discipline for investigation of general principles of rational using of environment.

The main goal of the Management of nature is to construct the general principles for anthropogenic activities linked with using of natural resources or its impact. The final result of this paper has to provide the single way for investigation of the nature (*Kurazhkvsky*, 1969). The nature using consists of the elements of natural, social and technical sciences. At the same time the main theoretical basis is geography and ecology so the Management of nature is the natural science (*Reymers*, 1990).

The nature using linked with ecology (the science about negotiation of living organisms between them self and environment); environmentology (the science about quality of environment and environment protection); natural resources investigation (the science about integrated resource and its components); econology (the science about harmonization of economy and ecology); economy of natural management (the science investigated the processes and results of society and nature interaction by economical methods).

The object of the nature using as a science is the complex of negotiations between the natural resources, the natural conditions of social life and its socio-economic development.

The subject of the nature using is the process of optimization its negotiations for the safety and reproduction in habitat. The process of optimization is the process of choosing the best options from possible ones, or the process of bringing the system in the best (optimal) condition.

According N. Reymers (*Reymers*, 1990) this term means:

1) receive a maximum of possibilities with a minimum of effort (economic costs during the short period);

2) the process of aspiration to a state closest to the dynamic equilibrium (quasi-steady state); optimal density of animals and plants, without undermine the food base or the depletion of soil. There is optimization of environment, optimization of territorial and ecological, optimization of the economic, ecological and economic etc. (*Enders*, 1955).

The optimal (rational) using of the natural resources is maximizing the efficient use of natural resources while reducing the anthropogenic impacts on the environment (*Sahaev*, *Shevchuk*, 1995). The object of the nature using is natural system as a synonym f terms ecosystem, geosystem, landscape etc. Ecosystem is any community of living organisms and its environment as a functional unite and based on interaction links between separate ecological components (*Reymers*, 1994). Natural System is combination of systems and subsystems in functional components on the high levels of hierarchical organization (biogeocoenosis, biome, biosphere, etc.). For the model of the ecosystem is characterized by the direction of the connection factors "environment" (object), model of natural system is typical recognition of the equality of all links (*Protection of landscapes*,1982).

From the position of the nature using may be of interest, both biotic and abiotic components of the natural system, but the optimization of the nature using provides for the continuation of favorable conditions for the existence and development of living organisms and, firstly - the human population. For example, if the water object is the ecosystem, then the main object is the community of hydrobionts (biocenosis). If the water object is natural system then biotic and abiotic components are equal. The interest may be as biological resources and also as mineral resources, priority is determined by the needs of the economy.

The optimization of the natural using provides biological and mineral using that does not violate the ecological balance of aquatic objects and safe the good conditions for hydrobionts and human. Often this is not observed. For instant, in acute shortage of hydrocarbons, there are searching, exploration and exploitation of oil and gas fields in the shelf of the seas, at the same time the problem of biodiversity challenge sidelined.

In a process of optimizing the nature using have to use a certain part of their term, that is, the natural-resource potential. According Reymers (*Reymers*, 1990, 1994):

- the natural-resource potential is the ability of natural systems without damage to itself (and, consequently, for the people) to give the necessary for humanity products or produce useful work (for mineral resource limitation can be contamination of the earth's surface, changing the natural seismicity, etc.);

- the natural-resource potential is the Earth's natural resources and the nearest space, it can be really involved in the economic activity of these technical and socio-economic opportunities to the society for the preservation of the habitat of humanity, the cost of natural resource potential is part of the national wealth;

- the natural-resource potential is a system of natural resources, environment, phenomena and processes, that, from the one side, is a territorial and resource base of society, and from the other - is opposed to it as an object of human impact;

- theoretically, the natural-resource potential is the maximum amount of natural resources that can be used by humankind without undermining the conditions that can exist and develop man as a species and social organism determined by the level of the ecological balance of the biosphere and its major units constituting the limits of the existence and development.

Also, it is important the concept of ecological and economic potential. The concept of "ecological and economic potential" is the potential ability of the biosphere and its parts to maintain the properties of self-reproduction in the human load. It includes only the conditions and resources that can potentially be used in the process of social reproduction and acquire economic significance and are considered from the standpoint of territorial restrictions (*Environmental Encyclopedia*, 2008).

Global ecological and economic potential means maximum permissible anthropogenic load on the entire set of self-organized natural systems of the Earth that does not lead to irreversible destruction of the structure of this set. Regional ecological and economic potential means human pressure on the territory (water area), may not lead to an increase in reverse the negative impact of disturbed natural systems on economic development. System-specific collection of all kinds of natural resources - material, energy, information - as factors of society, combined with the material and human resources called integrated resource. This collection is characterized by a qualitative and quantitative change in one of the resources (factors) will inevitably lead to more or less marked changes in the quantity or quality of other resources (eg, reduction of water content changes of energy and other indicators of terrain conditions for the creation and preservation of material resources and reproduction manpower). Natural resources are the natural objects and phenomena used in the present, past and future direct and indirect consumption, contributing to the creation of material wealth, the reproduction of the labor force, keeping the human condition and quality of life (*Reymers*, 1995). Natural conditions are the set of living organisms, bodies and natural phenomena, in addition to the existing human activities that affect to other living organisms, bodies and phenomena. There are many classifications natural resources (*Reymer*s, 1990, 1994):

1) natural resources are differentiated by source and location: energy resources, atmospheric gas resources, water resources, soil and geological resources, biological (resources of producers, consumers and decomposers), a comprehensive resource group (climatic, recreational, cognitive information, etc.);

2) natural resources are differentiated by the speed of exhaustion: fast-exhaustible (for example, the resources of valuable fish species), slow-exhaustible (eg, resources, NaCl and other salts of the oceans);

3) natural resources are differentiated by possible self-healing and culture: renewable and nonrenewable - respectively capable or not capable of self-healing for periods commensurate with the terms of their consumption (eg, vegetation, river water, ground water - renewable; mineral resources - non-renewable);

4) natural resources are differentiated according to the pace of economic replenishment (due to the search for new sources or new technologies withdrawal;

5) natural resources are differentiated by the possibility of replacing some other resources: interchangeable resources (for example, metals - plastic or ceramic and essential resources (optimum composition of the air, drinking naturally balanced composition of natural waters, etc.).

The theoretical basis of nature using comprises more than 250 different laws, regulations, principles and axioms. As an example, the law of limited of the natural resources - all natural resources (and conditions) of the Earth are finite. Consequently, the concept of "inexhaustible natural resources" is incorrect.

Regulation of the nature using is organization of relations between some components of the natural system, which leads to the intended results.

Rigid management of natural systems is, as a rule, technological impact and interference in natural processes, a radical transformation of the very mechanisms and systems of nature (continuous logging, development of virgin lands, hydraulic engineering, creation of irrigation systems, etc.). Rigid management of natural systems gives the highest economic effect in the local and regional scale, unless there is a sharp decrease in the natural resource potential, but only during the relatively short time interval. Later, the anthropogenic impact to the natural systems leads to their degradation; therefore there is a need for mechanisms for the soft management of natural systems. Soft management of the natural systems is predominantly mediated, indirect impact, usually through the use of natural mechanisms of self-regulation (forest melioration, small hydropower stations, organic farming, selective logging, etc.) For a long time period, only "soft" management of natural systems can be effective.

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«WATER AS A RESOURCE. CHARACTERISTICS OF AQUATIC SYSTEMS»

1. Eutrophication of the Black Sea

This case study illustrates how eutrophication has developed in the Black Sea and examines the contribution of riverine, coastal, atmospheric and sediment inputs to the nutrient budget of the Northwestern shelf area. The impacts of the eutrophication on biodiversity are also discussed.

Characteristics of the Black Sea

The Black Sea is one of the biggest landlocked seas of the World. An important aspect of the sea is the small and very limited water exchange through the Bosporus with the Mediterranean. The Black Sea's geographical position and morphometric features make it a classic example of an "ecological target" which has been influenced by human activities. The surface area is 42,300 km^2, and the drainage area is more than 2.3 x 10^6 km^2. There are 26 European countries and 300 large and small rivers in the catchment area. The main rivers are the Danube, the Dnieper and the Dniester. The combined discharge of these rivers is 256 km^3 a^{-1} contributing about 80 per cent of the freshwater runoff to the Black Sea. The Danube (204 km^3 a^{-1}) is the main riverine input to the Black Sea, providing up to 60 per cent of the total freshwater input; thus the influence of the Danube on the Black Sea is very strong. This influence is particularly marked along the Romanian and Bulgarian shelf and sometimes spreads up to the Bosporus. In some years the influence of the Danube can occupy 70 per cent of the Northwestern shelf of the Black Sea; in other years it is only 20-30 per cent. The total area of the Black Sea influenced by the Danube, according to the area of freshwater phytoplankton species recorded, is not less than 10^5 km^2.

The flows of the Danube, Dnieper and Dniester affect the structure and functioning of the coastal aquatic ecosystem. Freshwaters significantly reduce the salinity, temperature and transparency of the shallow marine water. There is a vertical gradient of salinity and temperature (in warm periods) from riverine at the surface to marine water at depth. These gradients limit vertical mixing between the layers. For example, the drop in density can be 3-5 kg m^{-3} per meter depth, which corresponds to the same change as for the open ocean over a depth of about a thousand meters. Salinity at the mouth of the rivers is the lowest at 5-10 $^{o}/_{oo}$. Surface water salinity in the central region is the highest at 18.2 $^{o}/_{oo}$. The maximum horizontal gradient of salinity is at a distance of 2-8 km from the river mouth. The range of seawater temperatures at the surface is 2 °C in January to 22 °C in August in the Northwestern shelf. Ice occurs about every five years. The highest temperature recorded in the summer was 27 °C. The daily fluctuation in temperature under a moderate wind can reach 6 °C. The vertical temperature structure changes seasonally (*Bolshakov*, 1970).

Variability in the hydrochemistry depends mainly on river runoff, precipitation, seasonal temperature distribution, hydrobiological activity, especially phytoplankton, and anthropogenic factors such as pollution from industry and agriculture. Natural cycling of river runoff and regulated flow in the rivers is also very important.

Intensive economic development and exhaustive environmental management has led to considerable ecological pressure on the Black Sea ecosystem. Development of the eutrophication process in the Black Sea occurred as a result of increasing amounts of nutrients in the river runoff associated with nutrient-enriched water coming from fertilized fields.

World-wide anthropogenic eutrophication began in the 1970s resulting from global economic activities at the end of 1960s, i.e. the so called "green revolution" aiming for increased food production.

The development of near-bottom hypoxia (i.e. a concentration of dissolved oxygen less than $2 mgl^{-1}$) was noted at the beginning of the 1970s for the Danube coastal zone and for the whole Northwestern shelf of the Black Sea. Near-bottom hypoxia starts to develop in shallow water (8-15 m deep) in June under the seasonal pycnocline. Later in July, following thermocline development, the near bottom layer with its oxygen deficit falls to depths of 20-25 m, but only rarely to depths of 30-50 m up to the border of the Northwestern shelf. At this time oxygen is replenished in shallow waters because of turbulent diffusion. Deep-water hypoxia, and sometimes anoxia, continues until November when it is destroyed during vertical mixing in the winter caused by storm activity (*Berlinsky*, 1989).

In the winter period, the bottom layer is saturated with oxygen. In summer, disturbances of the oxygen regime can occur in the shallow water due to upwelling processes. Under the influence of westerly winds, surface water moves out towards the open sea and water from the bottom layer with a deficit of oxygen moves up to the surface near the shore – the so called "upwelling phenomenon". On average this happens five times per season. This effect had been marked in the coastal zone between the Danube and Dnieper in the Ukraine and near the shores of Romania and Bulgaria. As a result, heavy mortalities of fish and benthic marine organisms have been recorded during the last 30 years. From 1973 to 1990, in the Northwestern shelf of the Black Sea, the zone of hypoxia occupied 3,500-40,000 km^2 (Fig. 1.1; 1.2). It has caused the mass mortality of $60 \times 10^6$1 of bottom animal's and $5 \times 10^3$1 of fish, especially young fish (*Zaitsev and Mamaev, 1997*). It is important to note that in the marine environment, the areas affected by eutrophication are extremely large and therefore difficult to control because they are open spaces without precise limits.

Fig. 1.1. The development of near-bottom hypoxia in the Northwestern shelf of the Black

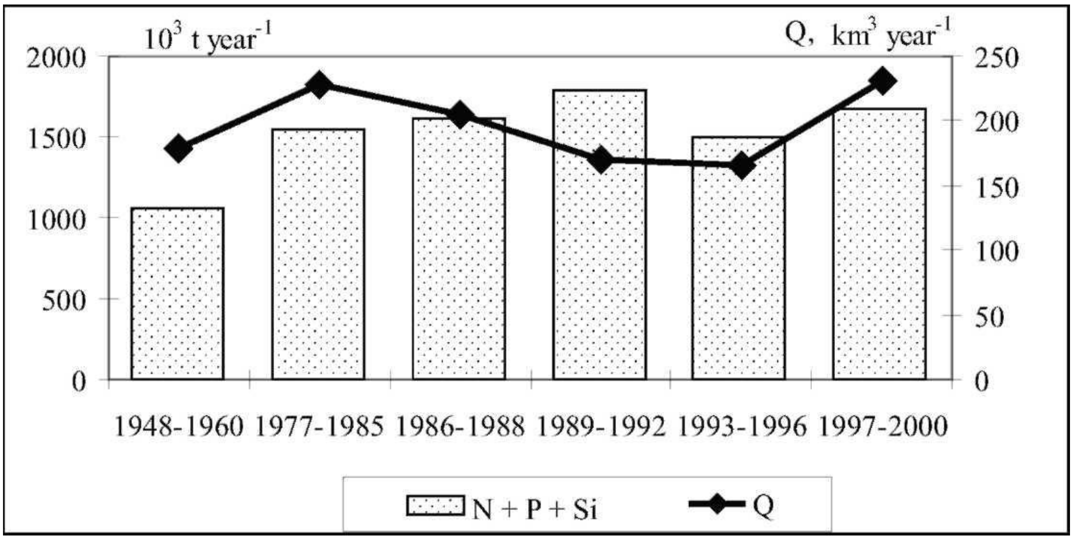

Fig. 1.2. Trends in discharge, Q (km^3 a^{-1}), and combined nutrient, total N+P+Si (10^3 t a^{-1}), runoff for the Danube, 1948-2000 (Data from Bogatova and Garkavaya)

Sources of nutrients

At present, it is hard to define the natural and anthropogenic sources ofnutrients, because even natural sources of nutrients (river runoff, atmospheric deposition, bottom sediments) are influenced by anthropogenic activities. These sources can, however, be divided into point and non-point sources. Point sources (municipal and industrial wastewater discharges) are discharges of contaminants that come from a stationary or fixed facility, for example, from a pipe, ditch or drain. Point sources are regulated by laws that place limits on the types and amounts of contaminants released to water. Non-point source discharges are river runoff, atmospheric deposition, stormwater runoff, groundwater and bottom sediments.

The main sources of nutrients into the Black Sea, since the commencement of eutrophication, are river runoff, atmospheric deposition, municipal and industrial wastewater discharges and bottom sediments.

River runoff

The Danube with its average runoff of 204 km^3 a^{-1} is the main source of nutrients into the Black Sea. During the last 50 years nutrients in river runoff have changed significantly (Table 1.1).

Table 1.1. Changes in nutrient concentrations in the Ukrainian part of the Danube during 1948-2000

Nutrient (mg l^{-1})	Period					
	1948-1960*	**1977-1985****	**1986-1988*****	**1989-1992******	**1993-1996******	**1997-2000******
NH_4^+	0.248	0.620	0.575	0.441	0.125	0.042
NO_2-	0.012	0.044	0.160	0.118	0.074	0.015
NO_3-	0.530	1.000	1.126	1.626	1.184	0.580
N_{min}	0.790	1.664	1.861	2.185	1.383	0.637
N_{org}	0.630	0.900	3.072	5.069	3.739	4.397
N_{total}	1.420	2.564	4.933	7.254	5.122	5.034
PO_4^{3-}	0.071	0.165	0.281	0.233	0.091	0.079
P_{org}	0.031	0.073	0.100	0.113	0.096	0.038
P_{total}	0.102	0.238	0.381	0.336	0.187	0.117
SiO_3^{2-}	4.375	3.980	2.571	2.979	2.356	2.120
River runoff (km^3 a^{-1})	179.4	227.7	204.7	169.7	195.1	230.9

Sources: **Almazov and Maistrenko*, 1961; **Enaki, 1987; ****Enaki and Zhuravleva*, 1993; Garkavaya *et al.* 1997; *****Garkavaya* et al. 1998; *Bogatova*, 2002

Nutrient values in the Danube delta runoff for the period 1948-2000 can be defined in the following periods:

Period I (1948-1960) - before regulated stream and eutrophication rocesses,

Period II (1977-1985) - start of eutrophication processes, and

Period III (1986-2000) - development of eutrophication processes.

In the 1950s and 1960s the hydrochemical regime of the Danube was stable - high turbidity and current velocity hindered the process of photosynthesis. The dynamics of the nutrients were related with river runoff only (Hydrologia Estuary of the Danube, 1963). Values for the period 1948-1960 can be used as a natural background and for analysis of the following variability. Starting from the 1970s the hydrological and hydrochemical regime changed. Construction of reservoirs, dams and hydropower stations in the middle section of the river (Gerdap 1 and Gerdap 2 (Iron Gate), Gapchikovo, Nadiamarosh, etc.) caused a decrease in current velocity and increased the water transparency (because water storage allowed sedimentation of suspended matter). The increase in water transparency allowed increased photosynthesis and the mass development of phytoplankton. At the same time, increases in anthropogenic pressures on the river ecosystems were marked: more intensive mineral fertilizers were used in agriculture and urban wastewater inputs increased. Sharp increases in nutrient levels in water can stimulate the eutrophication process, causing the growth of phytoplankton and very high concentrations can cause phytoplankton "overgrowth" known as "algal blooms".

In 1977-1985 (Table 1.1) ammonia nitrogen in the water of the delta increased by 2.5 times, nitrites by 4 times, nitrates by 5 times and phosphate doubled when compared with the period of 1958-1960. During the 1977-1985 period, mineral forms of nitrogen comprised 65 per cent of the total nitrogen in the water. The development of "algal blooms" (mainly diatomaceous phytoplankton) in reservoirs in the delta (*Ivanov*, 1993) led to increases in organic compounds but silicon concentrations decreased. Thus, in the Danube delta, concentrations of organic nitrogen increased by 1.5 times, phosphorus doubled and silicon decreased by 1.1 times. During this period, suspended mater and pollutants accumulated in the bottom sediments of reservoirs.

When eutrophication due to river runoff was most intensive (1986-2000), long-term seasonal monitoring in the Danube delta (Kiliya branch) was carried out by the Odessa Branch of the Institute of Biology of Southern Seas, National Academy of Sciences of Ukraine (OB IBSS). Some trends in the quantitative characteristics of nutrient runoff were observed. A general tendency of increasing proportions of organic nitrogen in the total nitrogen compounds of the river runoff, and decreasing silicon concentrations, were observed in spite of variations in the mean levels of Danube runoff (1986-1988 and 1993-1996 - medium water runoff; 1989-1992 - low water runoff; and 1997-2000 - high water runoff). Thus, at present, concentrations of organic nitrogen are much higher than the sum of the mineral forms (ammonia nitrogen, nitrite and nitrate) and contribute 90 per cent of the total nitrogen in the Danube delta. Over nearly 50 years silicon concentrations have decreased three-fold from 4.375 mg l^{-1} in 1958-1960 to 1.438 mg l^{-1} in 1997-2000. This correlates with photosynthetic activity and silica use by diatomaceous phytoplankton species for cell construction. Prior to this, algal blooms occurred mainly in reservoirs but they now occur permanently in the delta during summer. Labile organic matter has been increased as a result of phytoplankton decay.

At present, in the Danube delta, the level of mineral compounds of nitrogen and phosphorus have decreased and returned to the averages for 1958-1960, i.e. to the period before the beginning of eutrophication. Organic compounds of nitrogen and phosphorus are the potential sources of nutrients entering the Black Sea where they are incorporated into biological cycles, increasing further the level of eutrophication. Total nitrogen and phosphorus compounds (mineral and organic forms) and silica entering with the waters of the Danube during 1948-2000 increased by 1.5 times (Fig. 1.2).

Similar processes for nutrient runoff from the Dnieper (water runoff 45 km^3 a^{-1}) and Dniester (water runoff 8 km^3 a^{-1}) have also been observed. Construction of reservoirs has stimulated decreasing mineral compounds and increasing organic compounds in the water runoff of these rivers. In addition, these rivers flow through highly populated areas that extract water for industrial and irrigation purposes. The water runoff of these rivers is significantly less than that of the Danube and the proportion of nutrient input they contribute to the Northwestern shelf of the Black Sea is about 10 per cent (Fig. 1.2).

Atmospheric precipitation

Atmospheric precipitation (snow, rain) plays an important role in nutrient enrichment of the upper layer of the Black Sea. According to different authors, precipitation contributes 119-300 km^3 a^{-1} in the Black Sea water balance, with 25-30 km^3 a^{-1} in the Northwestern shelf (*Zaitsev and Mamaev,* 1997). The atmosphere is polluted with industrial discharges, exhaust gases, etc. and this has led to an increase in the atmospheric precipitation of certain chemicals, including nitrogen and phosphorus compounds. Nutrient concentrations in atmospheric precipitation to the Northwestern shelf (Table 1.2) were used to calculate nutrient inputs to the sea. With 20-25 km^3 a^{-1} of precipitation in the Northwestern shelf, the estimated inputs were 65 x $10^3$1 and 23 x 10^3 t of mineral and organic nitrogen respectively, and more the3x10^3t and about 1.4x$10^3$1 of mineral and organic phosphorus respectively. This is comparable with the quantities of these compounds entering the Black Sea with the Dniester and Dnieper runoff (*Garkavaya and Bogatova*, 2001).

Table 1.2. Nutrients in atmospheric precipitation (mg l^{-1})

Region	PO_4^{3}-	P_{org}.	N_{min}	N_{org}	$SÌO_3^{2-}$
Back Sea coast*	-	0.072	1.41	-	0.251
Danube basin**	0.100	0.040	3.22	2.82	1.56
NW Black sea shelf**	0.161	0.055	2.542	0.245	1.60

Sources: * *Rozhdestvinsky* 1979; ** *Garkavaya and Bogatova* 2001; Data supplied by *Bogatova and Garkavay*

Bottom sediments

Eutrophication of the Northwestern shelf has led to an accumulation of allochthonous and autochthonous organic matter in the bottom sediments of the shelf. Concentrations of mineral and organic nitrogen compounds, phosphorus and silica in sediment pore water exceed concentrations in the water column.

Due to diffusion, the activity of benthic organisms and resuspension during storms, nutrients from the bottom sediments are released to the sea water in the near-bottom layer. The rate of nutrient release to the water column increases significantly during hypoxic conditions in the near-bottom layers of the sea. Observations have shown that if hypoxia is maintained for one month, an additional 50-80 x 10^3 t of ammonium nitrogen, 10-17 x 10^3 t of phosphates and 40-90 x 10 t of silica are added to the bottom waters of the Northwestern shelf.

Considering that there are many regions of the Northwestern shelf where reducing conditions last for a couple of months, the amount of nutrients released could be much greater (*Garkavaya and Bogatova*, 2001). Studies by *Friedrich* et al. (2002) have shown that release of nutrients from bottom sediments also occurs in the estuary regions of the Danube and Dniester. In the shelf areas, up to depths of 50 m the quantity released is determined by oxygen conditions in the near-bottom layer and the amount of organic matter in the bottom sediments.

It is possible to estimate the amount of nutrients stored in the bottom sediments of the Northwestern shelf of the Black Sea during recent eutrophication from the amount of organic matter in the water column, which is related to the total biomass of phytoplankton. It is known that the biomass of phytoplankton during eutrophication of the Northwestern shelf has increased significantly; in the 1990s the maximum values reached were 1.6 kg m . The average values of total phytoplankton biomass in the Northwestern shelf in 1996-2000 were 1,000-2,000 mg m or 1,000-2,000 t km (*Nesterova*, 2001). The total storage of organic matter in the Ukrainian part of the Northwestern shelf only (water volume 1,150 km^3) is 1,150-2,300 x $10^3$1, with an average value of 1,725 x 10 t. This quantity is significantly underestimated because, in the impacted regions of the Northwestern shelf (e.g. Odessa Gulf, ports, estuarine zones), the biomass of phytoplankton exceeds the Northwestern shelf could have an additional 1,700 x 10 t of organic matter annually. Oxidation of the organic matter in the photic zone during the warm period of the year (i.e. the period of maximum diffusion of nutrients from the bottom sediments) can contribute up to 24 x 10^3 t of phosphates and up to 240 x10^3 for nitrogen.

Coastal sources

Coastal sources comprise the industrial and municipal discharges of cities, which enter the coastal zone of the sea. An estimated 110.4 x 10 m of sewage effluents from Odessa, Nikolaev and Kherson region entered the Northwestern shelf in the year 2000 (*Anon*., 2002).

It is hard to estimate the influence of sewage effluents entering the Northwestern shelf without knowing their origin and composition. Calculations of phosphorus inputs based on population data for cities, such as Odessa (1.3 x 10^6 people) suggest more then 40 x 10^3 $t a^{-1}$ (*Edelstein*, 1997). This is comparable with the input of phosphorus from the Danube runoff.

Local sources of nutrients enriching the coastal zone can also include drainage waters. In the Odessa area there are 11 culverts, with a combined average annual runoff of 0.012 km. All year-round monitoring during 1990-1995 showed that the drainage waters were enriched with mineral and organic compounds of nitrogen and phosphorus, which would contribute to the eutrophication process in the coastal zone. It was calculated that drainage waters input 0 t of nitrates and up to 600 t of organic nitrogen to the coastal zone each year. These values are comparable with the Dniester input of these compounds to the Black Sea each month. These additional nutrient inputs to the coastal zone are the cause of intensive growths of phytoplankton and algae, as well as saprophytic bacteria which facilitate the destruction of organic matter. The intensification of biological processes in the coastal zone has contributed to a worsening of the water quality (*Alexandrov* et al., 2000). The present breakdown of the relative contributions of non-point and point sources of nutrients to the Northwestern shelf is given in Fig. 1.3.

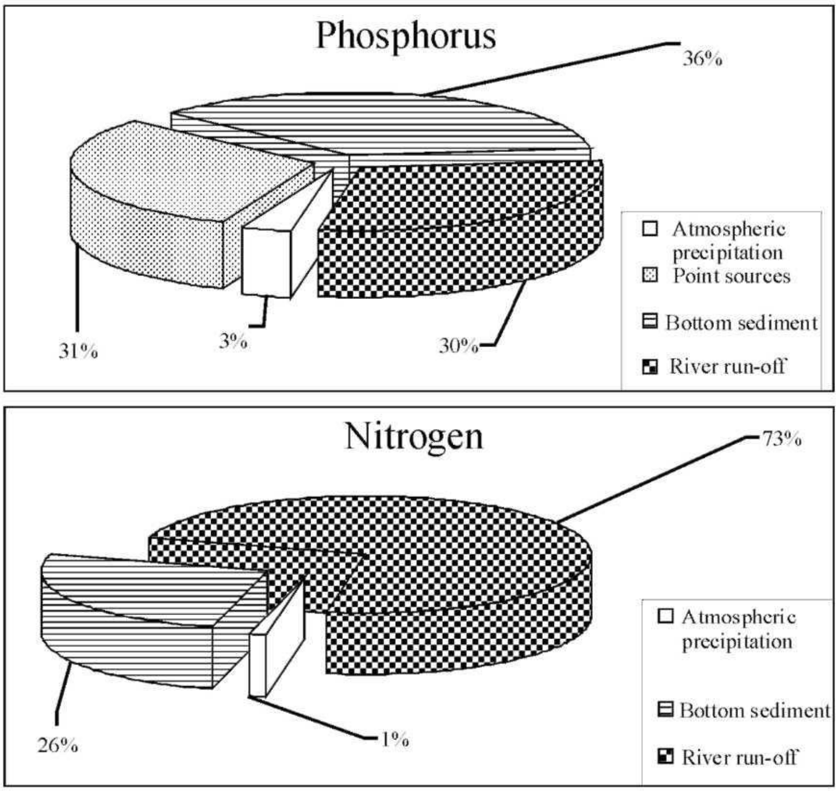

Fig. 1.3. Relative contributions of point and non-point sources to nitrogen and phosphorus inputs to the Northwestern shelf of the Black Sea in recent years (Data from *Bogatova and Garkavaya*)

Trends in nutrient concentrations

Human impacts on the Black Sea ecosystem have led to violation of the carbon cycle of the ecosystem, i.e. to the synthesis and destruction of organic matter. Nutrient concentrations in the surface and euphotic layers of the sea have been significantly increased. Maximum nutrient concentrations have been noted in the estuarine zones and particularly rapid increases have been recorded in the zone of influence of the Danube waters. It has been established by *Berlinsky* (1989) and *Berlinsky and Dikhanov* (1991) that nutrient concentrations in surface waters, algal blooms and near bottom hypoxia in the Northwestern shelf are directly correlated with the Danube runoff and the onset of the spring flood. If the peak of the flood is in April, the main mass of the Danube waters extends beyond the Northwestern shelf under the prevailing northeasterly wind direction that occurs in springtime. In this situation, near bottom hypoxia does not develop. If the peak flood occurs in May, the main mass of the Danube waters remains in the Northwestern shelf because the prevailing wind direction is southerly. In this situation the marine water masses, with their nutrients, form topographic vortexes in which algal blooms have been observed during the summer period.

Fig. 1.4. Zones of moderate (a) and intensive (b) summer phytoplankton blooms in the Black Sea and Azov Sea (After *Zaitsev and Mamaev*, 1997)

Ecosystem response at different trophic levels

The Northwestern shelf has now become the largest hypertrophic area in the whole Mediterranean Basin. The result of eutrophication of this region has been the decline of biological diversity at all systematic levels in all biotopes of the Northwestern shelf.

The initial consequence of the eutrophication is an increase in biomass of phytoplankton, especially dinoflagellates, euglenoids, and different species of picoplankton. The most dramatic examples of such phytoplankton blooms occur on the Northwestern shelf, where in the 1950s the share of peridinians in the total phytoplankton biomass was 18.8 per cent, but in the 1980s it reached 54.4 per cent. Red tides caused by peridinians have become common phenomena on the Northwestern shelf (Fig. 1.5).

Another consequence of phytoplankton enrichment has been an increase in the number of some herbivorous and detritivorous Zooplankton species. The most typical forms are the flagellate *Noctiluca scintillans,* the infusorian *Mesodinium rubrum,* the jellyfish *Aurelia aurita*, some rotatorians, the cladoceran *Pleopis polyphemoides* and the copepod *Acartia clausi.*

The populations of relatively large pelagic crustaceans, including carnivorous, mixotrophic and even herbivorous species has began to decline; e.g. populations of the neustonic Pontellidae *Centropages kroyeri*, the zoeae and megalopae of crabs, etc. This effect could be explained by the concentration of toxicants in the surface microlayer. The decrease in crab populations can be assumed to be due to hypoxic conditions on the shelf (Fig. 1.5).

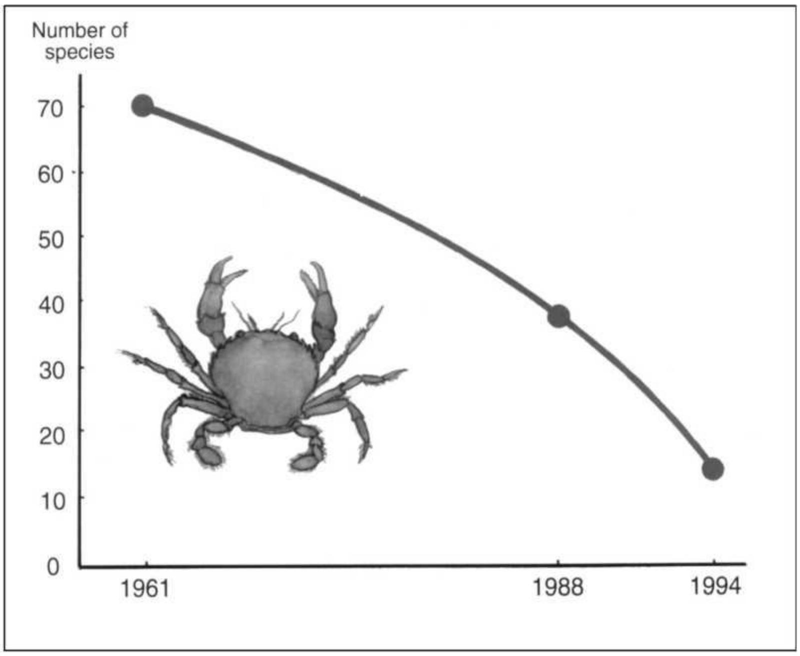

Fig. 1.5. Decline in the number of bottom macrofauna species occurring on the Romanian shelf of the Black Sea (After *Zaitsev and Mamaev*, 1997)

A decline in water transparency has reduced the photosynthetic capacity of bottom algae due to the low light penetration and a reduction in the shelf macrophytocoenoses, except for some forms growing in the very shallow areas at the edge of the sea. The change has been typified by the degradation of Zernov's *Phyllophora* field (Fig. 1.6). The red *Phyllophora* has practically disappeared and the Northwestern shelf has, as a result, lost an important habit for marine invertebrates and fish and a rich source of oxygen.

Sedimentation on the shelf of large amounts of dead phytoplankton is the result of cultural eutrophication. The decomposition of this organic material causes hypoxia and even anoxia in the near bottom layers of water. The formation of large hypoxic areas is a new phenomenon in the Black Sea ecosystem, which was first recorded in August-September 1973 on the North-western shelf. In more recent years, cases of mass mortality of benthic organisms from hypoxia have become commonplace. The magnitude of the mortalities depends on the meteorological, hydrological, hydrochemical and biological peculiarities of each summer season.

As the result of hypoxia, silting and other factors, the standing stock of the Black Sea organisms has decreased markedly. In the early 1960s, the total mussel biomass on the Northwestern shelf was about 10 x 10^{6}1. In the 1980s, the mussel biomass fell to a little over 3 x 10^{6}1, with a high proportion of juveniles. The oyster *Ostrea edulis* is very sensitive to silting and has now almost

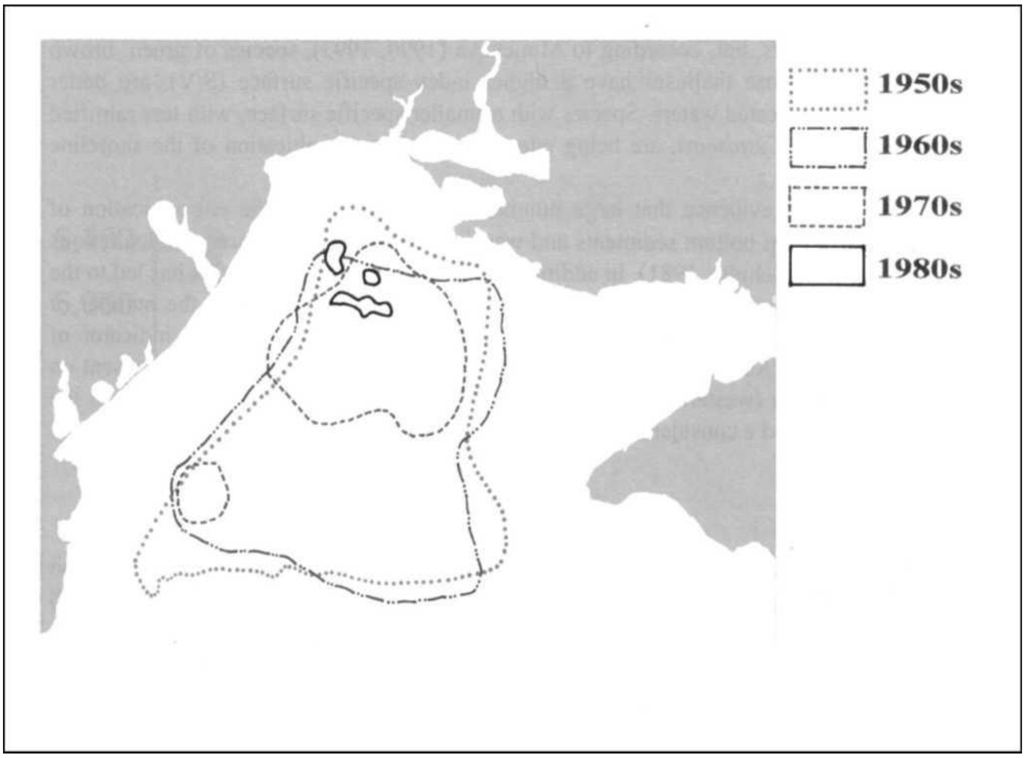

Fig. 1.6. Progressive reduction in "Zernov's *Phyllophora* field" in the Northwestern shelf of the Black Sea between the 1950s and the 1980s (*After Zaitsev and Mamaev*, 1997)

completely disappeared from the Northwestern shelf where, in the late 1950s, there were once more than 50 x 10^{6} individuals. The populations of crabs, hermit crabs, ghost shrimps, and bottom fish such as turbot, which were also very common on the Northwestern shelf until the 1960s, have also practically disappeared (*Anon*, 1998).

Summary

The Northwestern shelf of the Black Sea is now showing a strong imbalance in the carbon cycle. Recent eutrophication is characterized by decreasing nutrients but increasing organic compounds in river runoff and in marine water, because of regulation of all the rivers entering the Northwestern shelf of the Black Sea and reduced use of mineral fertilizers in agriculture. In the summer, mineralization of organic compounds is rapid producing nutrients that provoke phytoplankton development, i.e. eutrophication.

At present bottom sediments are also an important source ofnutrients. The storage ofor-ganic matter in bottom sediments increases annually because of the mass development of phytoplankton blooms. Upwelling is common in the Northwestern shelf of the Black Sea, occurring five to seven times in summer and lasting for up to a week. These upwellings contribute nitrogen and phosphorus compounds from bottom sediments to biological cycles. The quantities of nutrients contributed by river runoff and bottom sediments are currently comparable.

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2. Contaminations in the aquqtic enbvironment, monitoring of toxicants in aquatic ecosystems

Use of a battery of bioassays to classify hazardous wastes and evaluate their impact in the aquatic environment. In today's world, generation, storage, treatment, transport, recovery, transboundary movement, and disposal of wastes pose formidable problems for society and represent a serious threat for human health and the environment. Great concern exists for the future if this issue is not properly addressed (*Rummel-Bulska*, 1993). Consequently, it is advisable to manage waste as well as possible, which implies an adaptation of the legislation and innovations from the technological and scientific point of view.

Waste management is now moving from a "ways of dealing-based approach" (disposal, incineration and/or treatment of wastes) to an "objectives-based approach" (reduction, re-use, valorisation, stabilization/solidification, vitrification, risk assessment, ecocompatibility...). The framework for required action is based on a hierarchy of objectives and focused on the four following major waste-related program areas:

- minimizing wastes;
- maximizing environmentally sound waste reuse and recycling;
- promoting environmentally sound waste disposal and treatment;
- extending waste service coverage.

One of the challenges to achieving such an "objectives-based approach" is improvement of waste classification methodologies. Managers need a sound regulatory framework and to dispose of the adapted technological methods that have been used in prospective studies for establishing the optimum approach to waste management. In this context, the present case study gives an example of how ecotoxicological assessment of wastes contributes to hazardous waste classification under regulatory requirements. Moreover, an example of environmental hazardous assessment of waste deposits illustrates the potential of such a waste management procedure.

Hazardous waste legislation

In the European Union (EU), the overall structure for the framework for waste management is set out in a series of directives, decisions, regulations and resolutions on waste and hazardous waste (i.e. http://europa.eu.int/eur-lex/en/lif/reg/en_register_15103030.html). Among the existing rules, Council Directive 91/156/EEC (CD, 1991a), amending the Council Directive 75/442/EEC (CD, 1975) often referred to as the Framework Directive on Waste (FDW), constitutes the legal framework for the avoidance, environmentally sound management and disposal of all wastes (Slide 3). This directive also defines wastes as *"any substance or object in the categories set out in Annex I which the holder discards or intends or is required to discard"*. In other words everything that belongs to any of the 16 categories of waste outlined in Annex I of this directive is regarded as waste.

In connection with the FDW, Council Directive 91/689/EEC (CD, 1991b), referred to as the Hazardous Waste Directive (HWD), aims to identify which waste must be regarded as hazardous and to ensure the correct management and regulation of such waste. For the purposes of the directive, hazardous wastes are defined in essence as:

Wastes displaying one or more of the 14 hazardous properties listed under H1 to H14 in Annex III of the directive (see Table 1, Slide 4); these 14 criteria are distributed among 4 types: H1 to H3 = physical hazard; H4 to H12 = hazard for human health; H13 = hazard following elimination of waste; H14 = environmental hazard.

Wastes containing any constituents listed in Annex II of the directive and having one or more hazardous properties. The list goes from C1 to C51. For example, C25 is asbestos.

The hazardous character of a waste is thus defined by reference to a list of properties (physico-chemical, toxicological and ecotoxicological), or by reference to its composition. As a result, a waste list, the so-called European Waste Catalogue (EWC) was established in Decision 94/3/EC pursuant to the FDW and a subset of this, the Hazardous Waste List (HWL), was introduced under Decision 94/904/EC pursuant to the HWD. However, since 1 January 2002, the two decisions have been replaced by Decision 2000/532/EC which envisaged the amalgamation of the EWC and the HWL into a single list by indicating on the EWC/HWL ifa waste is hazardous.

Determination of the hazardous characteristics corresponding to the various H criteria in Table 1 is based on test methods. While some of the methods developed for risk assessment of chemicals may be directly applied, there is no specific method immediately lending itself to application to criterion H14 (i.e. "Ecotoxic"). To fill the gap, the French Ministry of Environment (1998) proposed a working document under the title *"Criteria and methods for the assessment of the ecotoxicity ofwastes"*. This methodology was developed with the aim of it being accepted by the EU as a technical support procedure for waste classification under criterion H14.

The procedure described in the French proposal follows a general scheme as summarized in the Slide 5. Globally, the ecotoxicity of any waste can be assessed through either its chemical composition or its ecotoxicological characteristics by applying a battery of bioassays. Both approaches can be used on raw waste and on its leachate prepared in well-defined conditions. The chemical composition is used as a positive criterion, i.e. the presence of at least one pollutant in a concentration higher than the limits fixed in the proposal allows classification as ecotoxic and, consequently, as hazardous under the terms of the HWD. If the chemical characterization is inconclusive, ecotoxicological characterization is needed. The proposal assumes that the ecotoxicological characterization can be used as a positive or a negative criterion. The positive criterion means that if at least one bioassay shows a toxicity value (i.e. an effective concentration expressed in percentage of dilution of either liquid extract or solid waste) inferior to a limit value the waste is classified as hazardous. On the contrary, the negative criterion presumes that the waste can be classified as non-ecotoxic if the bioassay toxicity values are higher than the fixed limits.

Implementation of methodology

Within a period of 4 days after arrival at the laboratory, the samples of BA were submitted to a crushing procedure with the aim of obtaining fragmented material with a particle size lower than 4 mm, as required by the leaching procedure. Because the samples of 2SL corresponded to a powder in which the particle size was lower than 4 mm, no crushing treatment was applied. For both wastes, the moisture content was determined by drying a small portion of each samples at 105 ± 5 °C, until constant weight was reached. The values obtained were then taken into account for the adjustment of the liquid to solid ratio (L/S) in the leaching procedure. All samples of BA and 2SL were stored at ambient temperature inside tightly sealed containers to prevent contact with the atmosphere prior their use for experiments.

Batch leaching procedure

The pre-treated samples were submitted to the leaching methodology described in the draft European standard EN 12457 - 2 (2002) using a liquid to solid ratio (L/S) of 10:1. Briefly, sub-samples prepared from each sample were brought into contact with deionised water in the defined L/S ratio for 24 hours with agitation at constant temperature of 20 ± 2 °C. The different mixtures were placed in capped one-litre polyethylene bottles and the extraction process was performed in a roller-rotating device working at 100 rpm. After 24 hours of agitation, each mixture was settled for 15 minutes and centrifuged for 10 minutes at 3,500 rpm in order to remove the suspended matter from the leachates. The ecotoxic potential of the obtained supernatants was then assessed immediately without filtration and pH adjustment.

Ecotoxicological characterization of leachates

The ecotoxicological parameters analyzed in the leachates were obtained from acute bioassays (i.e. Microtox™ test and *Daphnia magna* immobilization test) and from chronic bioassays (i.e. algal growth inhibition test and *Ceriodaphnia dubia* reproduction inhibition test). In addition, the direct ecotoxicity of the pre-treated wastes was assessed using the acute plant growth inhibition test.

Microtox™ test

The Microtox™ test measures the decrease in bioluminescence induced by depletion in the cell metabolism due to a toxic effect ofthe tested leachate. Each test was performed according to the procedure described in the standard AFNOR T90-320-3 (1999). Briefly, the test system used the special light emitting bacterial reagent *(Vibriofischeri)* supplied in freeze-dried form by Azur Environmental (Carlsbad, CA, USA), culture medium (NaCl 2 per cent) and the Microtox™ temperature-controlled photometer (Microtox™ Model 5000). In each test, different dilutions of tested leachate were compared with one control. The freeze-dried bacterial culture was initially reconstituted with deionised water (Milli-Q water purification system) at 4 °C. In each of cuvettes, the light produced by 0.1 ml of a bacterial suspension prepared by mixing 10 ul of the bacterial culture and 90 ul of NaCl 2 per cent solution and equilibrated to 15 °C was measured. Then, 0.9 ml of the test leachate dilutions adjusted to 2 per cent NaCl and 0.9 ml of 2 per cent NaCl control solution were added to the respective cuvettes. Light output from each cuvette was measured after 30 minutes. Results were corrected by time-dependent change in light emission under test conditions without any toxic influence and the percent difference between initial light output and final light output was quantified.

Algal growth inhibition test

The algal growth inhibition test is based on the measurement of growth inhibition of the algae *Selenastrum capricornutum* (strain ATCC 22662, Rockville, MD, USA), renamed *Pseudokirchneriella subcapitata,* during 72 hours of exposure. Each test was performed according to the adapted procedure proposed in the standard AFNOR T90-375 (1998) using sterile 96-well microplates as described earlier by Blaise *etal.* (1986). Briefly, inoculafrom cultures in the mid-exponential phase were adjusted to 40,000 cells per ml in the standard growth medium at double strength, and 125 ul was micropipetted into each of the 60 internal microplate wells. Then, 125 ul of leachate solution at twice the desired concentration was added (water was used for controls). Each microplate was sealed with its cover to minimize evaporation during the exposure period. After 72 h of incubation under continuous illumination of 3,000 lux at 24±1°C, the algal growth was followed by measuring the fluorescence using a microplate fluorimeter. The fluorimeter was set with the excitation filter at 440 nm and emission filter at 640 nm.

Daphnia magna immobilization test

The *Daphnia magna* immobilization test is based on the measurement of mobility inhibition of this cladoceran during two days of exposure. It was conducted according to the standard AFNOR T90-301 (1996). Each assay was carried out in 28 glass cups, which allowed testing of six concentrations and one control, with four replicates. Ten millilitres ofthe tested leachate dilution prepared in the standard dilution medium (dilution medium alone acted as the control) were introduced in each cup. No food was added. The test was initiated by transferring five daphnids which were less than 24 hours old into each cup. Next, all cups were covered to reduce evaporation and then incubated for 48 h at 20 ± 1 °C in a temperature controlled chamber in darkness. After 24 h and 48 h of incubation, the numbers of motile and/or immobilized daphnids were counted.

Ceriodaphnia dubia reproduction test

The *Ceriodaphnia dubia* reproduction test is based on the measurement of reproduction inhibition of this cladoceran after seven days of exposure. It was conducted according to the standard EPA 600/4-91/002 (1994). Test animals were exposed for seven days in a daily static-renewal system to different test solutions (one control and five concentrations, with ten replicates) at 25 ± 1 °C in a temperature-controlled chamber with 16h:8hlight:darkphotoperiod. Illumination ranged from 300 to 500 lux. Prior to the start or the renewal of the test, a stock of500 ml for each of the five test solutions was prepared by mixing the appropriate volume of leachate to obtain the desired dilution, 10.5 x 10^7 cells of green algae *Pseudokirchneriella subcapitata,* 1.15 ml of a 5 g l^{-1} fish food suspension (Tetramin®), 1.15 ml of a 5 g l^{-1} dried cereal leaves suspension (Cerophyll®) and growth medium to adjust to the final volume. At the same time, a stock of 500 ml of control solution was prepared by mixing only the nutritive mixture and the growth medium. The growth medium corresponded to a mixture of two French commercial natural waters, Evian and Volvic respectively with a 1:4 v/v proportion. Then, test and control solutions were divided up to give 20 ml each test cup. At the start of each test, organisms less than 24 h old and all those within6hofthe same age were individually placed in the test cups. For each daily renewal of the test and control solutions, the organisms were transferred from the old cups into new cups containing test and control solutions freshly prepared. During the seven days of exposure, mortality and reproduction of each animal were recorded at each daily renewal of medium.

Plant growth inhibition test

The phytotoxicity test is based on the measurement of growth inhibition of the lettuce *(Lactuca sativa)* after 14 days of exposure. It was conducted according to the standard OECD 208 (1984). The reference soil was a natural loamy soil, and samples were taken from the 0-25 cm surface layer (organic matter 3 per cent, pH 6.6). Each sample of pre-treated waste was incorporated into the soil at different concentrations prior to potting. Three pots (disposable plastic, 7 cm diameter, 6 cm height) were used for each test concentration and 15 lettuce seeds were sown in each pot containing 100 g of soil or a mixture of soil and solid waste. Water evaporation was determined by daily weighing of pots and any loss was compensated by addition of distilled water. Plants were grown at 24 ± 1°C in a temperature-controlled chamber with lighting of 1,600 lux under a 16h:8h light:dark photoperiod. Fourteen days after planting, the total number of germinated plants was recorded, and then lettuces were cut at soil level in order to determine the wet and dry mass of the plant material.

Statistical analyses

For each test, results were expressed in terms of effective concentrations EC_{50} or EC_{20} (i.e. concentrations that cause respectively 50 per cent and 20 per cent of effect on the assessment endpoint). For the algal, ceriodaphnid and plant tests, EC_{20} or EC_{50} were determined by regression using the ICp method based on a linear interpolation of means. Daphnid EC_{50} and Microtox™ EC50 were determined by regression using a Probit model and a log-linear model respectively. Calculations were performed using the commercial software package TOXSTAT which can be purchased from Western Ecosystem Technology Inc. (address via http://www.west-inc.com).

Classification of the tested solid wastes according to the French methodology

In Table 2.1 summarizes the ecotoxicological results obtained for each waste.

The range ofreported EC_{50} endpoints obtained from BA leachates (Slides 10 and 11) varied from 26.30 per cent (Microtox™ test) down to 2.12 per cent (algal test), whereas it varied from 0.43 per cent (algal test) down to 0.07 per cent (Microtox™ test) for the 2SL leachates. Biomass and germination results in the lettuce growth tests clearly demonstrated that some effects were also detected with the solid-phase procedure and that the 2SL displayed a higher toxicity than the BA. Such results obtained by direct and indirect measures of toxicity point out that the 2SL possesses a higher level of hazard potential than the BA.

Although the various bioassay measurement endpoints clearly do not have the same ecotoxicological significance (e.g. reproduction EC_{50} vs mortality EC_{50}), they nevertheless allow ranking each waste as a function of their sensitivity. For the BA leachate, the sequence in decreasing order of sensitivity was as follows: algal test > daphnid test > Microtox™ test > ceriodaphnid test.

This information clearly identifies the algal test as a good candidate to assess BA toxicity leachate fluxes. The sensitivity of algae for this type of waste had previously been shown by *Lambolez et al.* (1994), *Ferrari et al.* (1999) and *Lapa et al.* (2002). In contrast, the decreasing sensitivity sequence for the 2SL leachates was as follows: Microtox™ test > ceriodaphnid test > algal test ~ daphnid test. This indicates that the Microtox™ test should be a good indicator of the 2SL toxicity leachate fluxes. However, because a recent literature review demonstrated a complete lack of ecotoxicological data for this type of slag, no comparison could be made with other results.

Table 2.1. Direct and indirect ecotoxicity of a municipal solid waste incinerator bottom ash (BA) and a slag from a second smelting of lead (2SL): comparison with limits defined in the French proposal for the H14 criterion assessment

	Measurement endpoints		**2SL**	**2SL**
INDIRECT TEST		Leachate (L/S=10): Results in % of leachate		
Microtox™ (30 min)	EC_{50}	26.30	0.07*	10
D. magna (48h)	EC_{50}	3.70*	0.40*	10
P. subcapitata (72h)	EC_{20}	0.88	0.19	0.1
	EC_{50}	2.12	0.43	/
C. dubia (7d)	EC_{20}	20.40	0.20	0.1
	EC_{50}	22.70	0.28	/
DIRECT TEST		Solid waste: Results in % of dry waste equivalent		
Lactuca sativa (14d)	EC_{50} (germination)	40.00	1.20*	10
	EC_{50} (germination)	40.00	1.20*	10
	EC_{50} (dry biomass)	31.25	1.70*	10

* - Effective concentration (EC) inferior to the minimum limit value authorized in the methodology : in this case, the waste is considered as hazardous. Source: *French Ministry of Environment*, 1998

Considering the limit values defined in the French proposal for the H14 criterion assessment, it is clear that the 2SL can be classified as hazardous waste because three of the five applied bioassays showed toxicity values inferior to their corresponding fixed limits. Such a classification is in accordance with the current EWC/HWL (cf. §2) which identifies this kind of residue as hazardous. For the BA, although only the acute daphnid test displayed a toxicity value lower than the recommended limit, it is sufficient to classify this waste as hazardous (Slide 14). However, in contrast with the 2SL classification, the current EWC/HWL does not clearly identify this kind of waste as hazardous. Even if some other studies have also pointed out the hazardous character of BA (*Lapa et al.*, 2002), the difficulty of classifying them as such in the EWC/HWL may be linked to the large variability in the responses to the test methods because of their variability in composition, in time and location (*Radetski et al.*, 2004). Such results show also that the most sensitive test (i.e. algal test for the BA) may not be the test that enables classifying a waste as hazardous according the French methodology, which underlines the importance of using a battery of bioassays.

Using the H14 criterion assessment for environmental waste management

Waste classification under the criterion H14 may be taken further and used to highlight the potential hazardous impact of waste leachates on aquatic biota and thus, to ensure that unacceptable adverse effects would not arise from storage, treatment, re-use or disposal of the waste. In other words, this approach may be used as a prerequisite step to select the most suitable way for managing a waste. For example, because the ecotoxic hazard potential of the BA and the 2SL is indicated, these two wastes need to be either evaluated in a higher tier of risk assessment or treated before their valorisation, or stored under specific conditions (e.g. in a special landfill). The importance of this for aquatic biota is evident when considering that deposition of these two wastes directly in the environment, without the precaution of storage, has generated ecotoxic percolates. Indeed, two large field-scale leaching tests were built on an experimental site to simulate real conditions of a waste deposit receiving rain or run-off water and located near a river receiving effluents after percolation through the waste (*Perrodin et al.,* 2002). The first field leaching test consisted of a large tank where 39 tons of BA received water leading to the production of 2 m^3 of percolates per ton of dry BA every four months. The second one consisted of a smaller tank than the previous where 0.45 tons of 2SL received water leading to the production of 7.5 m^3 of percolates per ton of dry 2SL every four months. On the whole, three fractions (P) from each tank were recovered *in situ.* These fractions were defined as follows:

P0.5, P1 and P2 corresponded to the cumulated quantities of percolates, according to the L/S ratios of 0.5, 1 and 2 for the BA (expressed in the cumulated volume of the leachate obtained at the exit of the tank by the dry-weight of waste),

P2, P2.5 and P7.5 corresponded to cumulated quantities of percolates, according to the L/S ratios of 2.5, 5 and 7.5 for the 2SL.

When received at the laboratory (48 h after being sent from the field), the samples were settled for 15 minutes and centrifuged for 10 minutes at 3,500 rpm. Then, the ecotoxicity of the different fractions was assayed using the same battery of aquatic bioassays presented in Table 1 (i.e. Microtox™, algal, daphnid and ceriodaphnid tests).

Whatever the waste and the bioassays, the reported EC_{50} indicated: i) that BA percolates (Slide 17) appeared to be less hazardous than 2SL percolates ii) that the first fraction is the most ecotoxic, and iii) that the ecotoxicity of the percolates was reduced as a function of the L/S ratio reached. However, at the end of the experiments, no threshold "without apparent ecotoxic effects" was reached for both wastes because residual ecotoxicity was observed in the last fractions (between 10 and 100 per cent for BA and 1 and 10 per cent for 2SL). Consequently, the long-term ecotoxic potential hazard of these waste percolates for aquatic biota must be assumed.

This conclusion is reinforced by the ecological approach presented by *Perrodin et al.* (2002). Briefly, outdoor artificial streams (5 m, 440 litres, 1 control + 3 concentrations) colonized by aquatic invertebrate communities were supplied continuously by water which had received the same effluents produced by the field-scale leaching tests but having previously percolated through permeable subsoil. In these conditions, it was shown that a 10 per cent concentration of BA percolates was sufficient to produce significant effects on the abundance, richness and emergence of the organisms whereas only a 1 per cent concentration of 2SL percolates was needed (except for emergence, on which no effect was observed because of the season) (Slides 18 and 19).

Conclusion

The contribution of the ecotoxicological approach to hazardous waste classification under the regulatory requirement has been shown. It is evident that the use of bioassays to evaluate the toxicity of wastes is strongly recommended in order to have a more direct and integrated estimate of their environmental toxicity (*Lambolez et al*., 1994; *Ferrari et al*., 1999). In this sense, a preliminary assessment of the potential hazard of waste, as part of the potential impact studies, may be used as a prerequisite step to select the most suitable way for managing a waste in order to avoid possible surface and groundwater contamination. Moreover, coupling ecotoxicological and ecological approaches will provide even greater understanding during prospective or retrospective waste impact studies on the aquatic environment.

References

AFNOR 1996 Water quality - Determination of the inhibition of the mobility of *Daphnia magna* Straus (Cladocera, Crustacea) - Acute toxicity test. N°T90-301, *Association Française de Normalisation*, Paris.

AFNOR 1998 Water quality - Determination of water chronic toxicity by growth inhibition of the fresh water algae *Pseudokirchneriella subcapitata (Selenastrum capricornutum).* N°T90-375, Association Française de Normalisation, Paris. *AFNOR 1999* Water quality - Determination of the inhibitory effect of water samples on the light emission of *Vibrio fischeri* (Luminescent bacteria test) - Part 3: Method using freeze-dried bacteria. N°T90-320-3, Association Française de Normalisation, Paris.

Blaise, C., Legault, R., Bermingham, N., Van Coillie, R. and Vasseur, P. 1986 A simple microplate assay technique for aquatic toxicity assessment. *Toxicity Assessment*, 1**,** 261-281.

CD 1975 Council Directive 75/442/EEC of 15 July 1975 on waste. *Official Journal* L 194,25/07/1975, Brussels, Belgium, 39-41.

CD 1991a Council Directive 91/156/EEC of 18 March 1991 amending Directive 75/442/EEC on waste. *Official Journal* L 078, 26/03/1991, Brussels, 32-7.

CD 1991b Council Directive 91/689/EEC of 12 December 1991 on hazardous waste. *Official Journal* L 377, 31/12/1991, Brussels, 20-7.

EN 12457-2 2002 Characterization of waste - Leaching - Compliance test for leaching of granular waste materials and sludges - Part 2: One-stage batch test at a liquid to solids ratio of 10 l/kg for materials with a particle size below 4 mm (with or without size reduction). CEN/TC292/WG2, European Committee for Standardization, Brussels.

EPA 1994 Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms. EPA 600/4-91/002, Environmental Systems Laboratory, Cincinnati, OH.

Ferrari, B., Radetski, C.M., Veber, A.M. and Férard, J.F. 1999 Ecotoxicological assessment of solid wastes: a combined liquid- and solid-phase testing approach using a battery ofbioassays and biomarkers. *Environmental Toxicology and Chemistry,* **18,** 1195-1202.

French Ministry of Environment 1998 Criteria and methods for the assessment of the ecotoxicity of wastes. French Ministry of Environment/Directorate for Prevention Pollution and Risk Control, January, Paris.

Lambolez, L., Vasseur, P., Férard, J.F. and Giesbert, T. 1994 The environmental risks of industrial waste disposal: an experimatal approach including acute and chronic toxicities studies. *Ecotoxicology and Environmental Safety,* **28,** 317-328.

Lapa, N., Barbosa, R., Morais, J., Mendes, B., Méhu, J. and Santos Oliveira, J.F. 2002 Ecotoxicological assessment of leachates from MSWI bottom ashes. *Waste Management,* **22,** 583-593.

OECD 1984 Guidelines for testing of chemicals: Terrestrial plants, growth test. Document 208, Organisation for Economic Co-operation and Development, Paris.

Perrodin, Y., Gobbey, A., Grelier-Volatier, L., Canivet, V., Fruget, J.F., Gibert, J., Texier, C., Cluzeau, D., Gros, R., Poly, F. and Jocteur-Monrozier, L. 2002 Waste ecocompatibility in storage and reuse scenarios: global methodology and detailed presentation of the impact study on the recipient environments. *Waste Management,* **22,** 215-228.

Radetski, C.M., Ferrari, B., Cotelle, S. Masfaraud, J.F. and Férard, J.F. 2004 Evaluation of the genotoxic, mutagenic and oxidant stress potentials municipal solid waste incinerator bottom ash leachates. *Science of the Total Environment,* **333,** 209-216.

Rummel-Bulska, I. 1993 The Basel Convention : A global approach for the management of hazardous wastes. In Proceedings of *Hazardous Waste Conference,* 3-6 may 1993, Atlanta.

3. Antropogenic influence to aquatic system

Sources and Transport of Nutrients

a) Natural sources of nitrogen and phosphorus (including atmospheric sources) together with global cycles. Global cycle of carbon should also be discussed.

■ Background levels of nitrogen and phosphorus in lakes, rivers and coastal seas

b) Anthropogenic sources to be defined and discussed as follows:

- agriculture
- municipal
- atmospheric

■ relative contributions of different sources in different catchments to lakes and ocean

- inter-basin transfers of nutrients including inter-basin import and export of nutrients as fertilizers, crops and related products

c) Dissolved and particulate forms of nutrients and availability

d) Agricultural practices and losses of nitrogen and phosphorus

e) Pathways of nutrients in the environment (atmosphere, surface and groundwater) river transport: flux measurements

f) Nutrient budgets and loadings in water bodies

- external loadings
- internal loadings

Measurement and Monitoring of Eutrophication

a) Objectives and rationale

b) Indicators (basic e.g. pH, dissolved oxygen, transparency, cell counts, chlorophyll a; advanced e.g. nitrogen and phosphorus concentrations, soluble and particulate, primary productivity)

c) Relationships with hydrology; (concentrations, loadings, flux)

d) Frequency of monitoring, selection of methods and organisation of programme (site, time, depth, etc.) for lakes, rivers, estuaries and coastal zones

e) Interpretation of results in relation to water use

f) Precision and accuracy of measurements, quality assurance of monitoring programmes.

Management of Eutrophications

a) Prevention measures (forward planning) e.g. timing and quantity of fertilizer application, banning of phosphorus in washing powders

b) Conservation - halting of progress of by control measures (tilling practices, soil loss control etc.)

c) Control of pollution sources, e.g. agricultural and municipal (animal husbandry, waste management, treatment methods, etc.)

d) Prediction and modelling

■ overview of models available for prediction and management, e.g.Vollenweider, Jorgensen

■ coastal and estuarine modelling including examples

e) Remediation and restoration, e.g. biomanipulation, physical and chemical manipulation

f) Integrated watershed (catchment) management

Nutrient Turnover in Lakes and Marine Coastal Areas

a) External loading

■ Seasonality of discharge

■ Seasonality of particulate matter

■ Seasonality of nutrient content

■ Bioavailability as a function of the forms of P and N and environmental conditions (more detailed than given in 2c) above)

b) Internal loading

▪ Sediment and bottom characteristics

▪ Compartments in the sediments and the water column

■ Forms of phosphorus in particulate matter

■ Einsele - Mortimer theory

▪ Redox potential and oxygen conditions

▪ Mineralization pH

■ Transport processes:

* diffusion
* bioturbation
* wind-induced processes
* resuspension
* benthic recruitment

- Differences between shallow and deep lakes
- Differences between marine and lake systems
- Internal physical processes:
- mixing
- upwelling
- entrainment
-

Phytoplankton development

a) Algal blooms

■ Species composition

■ Biomass, density

■ Chlorophyll content

■ Ratios between different groups and species

▪ Conditions

- * light
- * temperature
- * nutrients - ratios and availability of nutrients
- * salinity
- * pH
- * turbulence
- * stability of the water column

■ Examples from different areas

Case studies

■ Specific characteristic of the basin as an area stressed by the Danube, Dniester and Dnieper river discharges

■ Sources of nutrients. Natural and anthropogenic sources of nutrients

■ Impact of bottom fishing, dragging and sand extraction

Recommended reading

Annonymous 1999 Planning and management of lakes and reservoirs: an integrated approach toeutrophication. UNEP Technical Publication Series 11, UNEP-IETC, Osaka/Shinga, 375pp. Gomoiu, M.-T. 1992 Marine eutrophication in the north-western part of the Black Sea. *In:* Marine Coastal Eutrophication, Proceedings of an International Conference, 21-24 March 1990.

Bologna, Elsevier, Amsterdam, 683-692. Livingstone, R.J. 2001 Eutrophication processes in coastal systems. CRC Press, Boca Raton, 327pp. Nehring, D. 1992 Eutrophication in the Baltic Sea. In: Marine Coastal Eutrophication.

Proceedings of an International Conference, 21-24 March 1990, Bologna, Elsevier, Amsterdam, 673-682.

Vollenweider, R.A. 1992 Coastal marine eutrophication: principles and control. In: Marine Coasta Eutrophication, Proceedings ofan International Conference, 21-24 March 1990, Bologna, Elsevier, Amsterdam, 1-20.

Vollenweider, *R.A*., Rinaldi, A. and Montari, G. 1992 Eutrophication, structure and dynamics of a marine coastal system: reulsts of ten-year monitoring along the Emilia-Romgna coast (Northwest Adriatic Sea). In: *Marine Coastal Eutrophication, Proceedings of an International Conference, 21-24 March 1990, Bologna*, Elsevier, Amsterdam, 63-106.

Wetzel, R.G. 2001 *Limnology*. Academic Press, San Diego,1006 pp.

Contamination in aquatic environment

Sources and transport mechanisms associated with refining and processing, including examples

■ Examples of emissions associated with pipelines (point source) and transport of oil by ships

▪ Emissions from nuclear power plants (point and diffuse sources)

■ Risks and long-term implications of accidents at power plants, oil and gas drilling rigs, oil tanker spills, etc.

Waste disposal

* Types of waste disposal: municipal and hazardous waste; landfill, dumping at sea, incineration
* Case studies of water pollution arising from landfill leachates

▪ Agricultural activities

* Importance of agriculture for human populations.
* Types of agriculture: intensive, organic, aquaculture, etc.
* Use and growth in use of pesticides
* Diffuse agricultural sources: application of pesticides, surface water run-off
* Potential point sources from storage of pesticides
* Farm drainage and disposal of animal wastes

a) Transport processes.

▪ Macroscale processes (large temporal and spatial scale)

* Inputs
* Atmospheric exchange
* River transport
* Advection and mixing
* Sedimentation
* Resuspension
* Diffusion
* Bioturbation
* Burial

▪ Microscale processes (molecular scale, short time scales)

* Physical (Adsorption, Desorption, Precipitation, Aggregation and Coagulation)
* Chemical and biochemical transformation (Complexation, Redox reactions)
* Ligand exchange
* Metabolic transformations (A very brief introduction to such mechanisms as Biomagnification)

b) Partitioning of toxic substances in aquatic ecosystems (30 minutes) - Provides an overview of the environmental compartments in which pollutants are transported and which collectively constitute the aquatic ecosystem

▪ Environmental compartments

* Water and suspended matter (including colloids and origin of particles)
* Sediments and pore water
* Biota

d) Dynamic of toxic substances in aquatic ecosystems - the processes involved in controlling the distribution of pollutants in the different compartments of the aquatic ecosystem. Exchanges between compartmentsat both the micro and macro scales

■ Phase exchange (real field effects of microscale processes)

* Water column cycling

- Dissolved (soluble + complexed)
- Colloids
- Particulate

* Removal processes (sedimentation, transport and accumulation)

▪ Dynamics at the watershed scale

* Rivers (short turnover time, intense turbulence, and flooding)

* Reservoirs (water levels fluctuations, hydrological regime, sediment trapping)

* Lakes (long turnover time, highly vulnerable)

Recommended reading

Clark, R.B. 1992. Marine Pollution. Clarendon, Oxford, 172 pp.

Donze, M. (Ed.) 1990. Aquatic Pollution and Dredging in the European Community. Delwel, Hague, 184 pp.

Hakanson, L. 1999. Water pollution - methods and criteria to rank, model and remediate chemical threats to aquatic ecosystems.

Backhuys Publishers, Leiden, 299 p. Stumm, W. (Ed.) 1987 *Aquatic Surface Chemistry*. Wiley, New York, 520 pp.

Monitoring of toxicants in aquatic ecosystems

The discussion of monitoring is designed to provide a clear account of the sampling and analytical techniques required in understanding levels and impacts of toxic pollutants. It should discuss how these relate to guidelines, objectives and other instruments of management. Recognition should be made of nutrient monitoring given in Modules I and II and, even though a brief refreshment should be provided, the focus must remain on toxic substances which clearly requires many differing monitoring strategies both in sampling, selection of analyses and interpretative requirements.

▪ Definitions

* Monitoring: The actual collection of information at set locations and at regular intervals in order to provide the data which may be used to define current conditions, trends, etc.

* Assessment: The overall process of evaluation of the physical, chemical and biological nature of the environment in relation to natural quality, human effects and intended uses, particularly uses which may affect human health and the "health" of the environment itself

■ Types of monitoring

* Monitoring: Long-term standardized measurement and observations of the environment in order to define status and trends.

* Survey: A finite duration, intensive programme to measure and observe the quality of the environment for a specific purpose

* Surveillance: Continuous, specific measurement and observation for the purpose of environmental quality management and operational activities

■ Examples of objectives

* Determine the extent, duration and nature of impact of a toxic discharge on biota

* Rapid assessment of effect of accident or spill

* Early warning of toxic impact on, for example, drinking water source

Monitoring design - Provides an overview of the criteria on which monitoring should be designed, what information is required, what sampling design is needed and what should be measured amidst the options available, all placed within the context of specific needs and objectives

▪ Site selection

* Single site, e.g. as defined by water use and criteria for use, or effluent samples

* Necessity for background or "control" sites

■ Criteria for determination of frequency of sampling

* Compliance with guidelines or licence conditions

* Single event sampling for surveys

▪ Hydrological information

* Residence times in lakes and reservoirs

* Current speed and direction in open waters, such as coastal area

* Stratification in lakes, reservoirs and estuaries

▪ Possible media for sampling or analysis

* Particulate matter (suspension and sediments)

* Biota (filter feeder, predator, sedentary, mobile)

▪ Selection of monitoring media

* Is the toxic substance soluble in water?

* Is the toxic substance lipid soluble?

* Is there a risk of bioaccumulation?

▪ Monitoring options

* Biological methods

* In site tests

* Continuous monitoring

▪ Selection of monitoring approach

* Are precise concentrations required?

* Are effects on the environment important?

* Is there a risk to human health?

* Is there a risk that the toxic substance may bioaccumulate?

c) Biological monitoring approaches - Describes advanced methods for biological monitoring, their rationale, selection of the most appropriate to meet monitoring objectives

- Bioassays and toxicity tests

* Carried out under defined laboratory conditions, in controlled environments, or *in situ*
* Reveal or confirm the presence of toxic substances in water samples
* Evaluate persistence of toxic combinations in effluents and in water bodies
* Monitor dispersion or locate sources of toxic substances
* Provide rapid evaluation of toxicity from accidents and spills

Recommended reading

Bartram, J. and Ballance, R. (Eds) 1996 Water Quality Monitoring: A Practical Guide to the Design and Implementation of Fresh Water Quality Studies and Monitoring Programmes. E & FN Spon, London.

Chapman, D. (Ed.) 1996 Water quality assessment: A guide to the use of biota, sediments and water in environmental monitoring, 2nd edition,E&FN Spon, London, 626 pp. Helmer, H. and Hespanol, I. 1997 *Water Pollution Control.E&FN* Spon, London, 510pp. International Organization for *Standardizatio*n (ISO) - www.iso.ch

Wells, P.E., Lee, K. and Blaise, C. 1998 Microscale testing in aquatic toxicology. Advances, Techniques and Practice. CRC Press.

Management of toxic substances, remediation/rehabilitation of water systems impacted by toxic substances

This theme is designed to answer the question of what should be done if the ecosystem has been polluted by toxic chemicals. It discusses differing methods of control and remediation but provides a focus on the integration of remedial measures at the ecosystem level within the catchment/watershed boundary. It will provide the student with the understanding that watershed systems are interactive and that remediation of one pollutant source cannot be done in isolation of other sources. It demonstrates the need for a planned approach to control and remediation using a battery of techniques from isolation and technological solutions to the use of natural ecosystem processes.

a) Provides a quick reminder of sources and transport as a
b) background for remediation. Discusses remediation and rehabilitation of degraded
c) ecosystems and the objectives of embarking on an ecosystem remediation programme

- Refreshing basic notions

* Typical contaminated situations Sources and transport processes of toxic substances (point and diffuse sources, transport dissolved in water, attached to coarse and fine particles). The main problems are linked to sediments (persistent contamination, chemical time bombs)

▪ Basic principles of remediation

* Overall aims of remediation and rehabilitation: Reduce the environmental impact of the contaminated site to a minimum within a reasonable financial frame, using if possible soft methods (cf. *Chap. 4, Fôrstner*, 1998). Guarantee human and ecosystem health.

* Presentation of the two fundamental possibilities in remediation: complete isolation or long-term dispersion and dilution. The best solution lies in the middle: try to use if possible natural phenomena and barriers.

* Various approaches to risk assessment: legal aspects in various countries (guide line values and international trans-boundary conventions, cf. *Fergusson and Kasamas*, 1999).

b) Management concepts to handle toxic materials and situations - Presents a management model to ensure the fullest support and commitment to restoration of degraded ecosystems

▪ Definitions (assessment, management, conceptual site model, financial and perceived risk)

▪ Science and non/science factors (criteria when to intervene versus regulations)

■ Working in the triangle authorities/decision makers - Science - Public (a sound knowledge of legal aspects and environmental technologies, as well as teaching capabilities to explain complex processes in a simple way, will be necessary!)

c) Treatment technologies for water and sediments - Provides the student with an overview of treatment technologies and systems to treat toxic wastes. Sufficient general process mechanisms are given for the student to understand the selection of the most appropriate process in any specific situation

■ Overview of environmental technologies (end-of-pipe, production- and product-integrating, avoiding toxic substances)

■ Wastewater treatment processes (precipitation, oxidation, reduction, neutralization, adsorption, ion exchange, flotation, centrifuging, extraction etc.)

■ Soil- and sediment treatment processes (solidification, stabilization, encapsulation, excavating/dredging and re-deposition in a waste repository, in situ treatment, capping, chemical extraction etc.)

d) Catchment concept of remediation and management of toxic substances - Discusses the most current philosophies for integrated catchment toxic substance control and remediation strategies.

■ It is a new integrated approach, still in development, considering the problem on a large scale (head water areas, flood plains, estuaries, deltas, coastal areas), often a trans-boundary problem

▪ Identification of all sensitive areas (mines, industries, existing waste repositories which could leak or flood, flood plains containing groundwater used for drinking water or irrigation etc.)

■ Before resorting to classical technologies, consider all possible "soft" technologies ("ecological engineering", "geochemical engineering"), which possibly need large areas of a catchment, such as phyto-remediation, bacterial bioremediation, use of wetlands, natural barriers etc.

▪ In order to prevent further degradation of the catchment area studied, combined actions of reducing emissions (point and diffuse sources) and adequate remediation of already polluted areas will be necessary

▪ The often-cited "natural attenuation" should not be used as an argument to leave toxic situations as they are. One should at least monitor these situations and possibly find ways to enhance the naturally occurring fixing or degradation processes and maintain the hydrological integrity of the catchment.

e) Case study - Provides background data on thewatershed characteristics of the Danube River and requests the students, as an exercise, to develop a watershed management strategy for toxic substances control and remediation in the Danube watershed.

The Danube catchment and its various known contaminated areas (and point sources): Exercise to collect the student's detailed propositions to remediate the different situations (mines, industries, dam sediments, delta areas, coastal problems).

Recommended reading

Aplitz, S.E. and Power, E.A. 2002 *J. Soils & Sed.* **2,** 61-66.

Burden, F. R., McKelvie, I., Forstner, U. and Guenther, A. (Eds) 2002 Environmental Monitoring Handbook. McGraw Hill, New York.

Ferguson, C., Darendrail, D., Freier, K., Jensen, B.K., Jensen, J., Kasamas, H., Urzelai, A. and Vegter, J. (Eds) 1998 Risk Assessment for Contaminated Sites in Europe. Volume 1, Scientific Basis. LQM Press, Nottingham. Ferguson, C. and Kasamas, H. (Eds) 1999 Risk Assessment for Contaminated Sites in Europe.

Volume 2, Policy Frameworks. LQM Press, Nottingham. Forstner, U. 1998 *Integrated Pollution Control.* Springer, Berlin.

Forstner, U. 2002 Geochemical Techniques on Contaminated Sediments - River Basin View.

Reports of the Institute for Technical Chemistry, Research Center Karlsruhe, Geo- and Water-Technology, **1** (3), 151-170. Also see recommended reading at end of Section 3.

4. Watersheds

The objectives are to provide the students with an insight into watershed characteristics, land uses, material usage and transfer processes on the land surface to the river. River transport and water quality are discussed, along with the concepts of loadings, fluxes and impacts on receiving waters and coastal marine processes. The curriculum addresses those issues and processes related to the watershed.

General Introduction

a) Discuss why watersheds and river transport of materials are important and how they are related to the issue of fresh water and coastal water quality

b) Reiterate and review the hydrological cycle and demonstrate surface water drainage, emphasising the connection of ground water to surface water

c) Review the classification of river basins and discuss river order within basins of differing sizes.

d) Review the process of infiltration, showing the relationship of infiltration to land surface characteristics, and discuss the changes to infiltration brought about by human activities such as deforestation, agriculture and urbanization

e) Briefly discuss the processes of erosion (rill, gully and sheet erosion) and relate them to land use. Include a brief description of stream bank erosion.

Hydrology

a) Describe the hydrograph and discuss the means of measuring water flow and discharge

b) Discuss river height (stage) and define relationship to flow velocity and volume discharge

c) Show how the hydrograph varies with changes in infiltration and discuss the general problems of river flooding

d) Discuss sediment transport in general and river carrying capacity, while providing a focus on suspended material

e) Show the relationship of suspended sediment to the hydrograph and note the role of suspensions in the transport of pollutants

f) Show the changes in sediment concentration with the curve of the hydrograph and discuss the reasons behind the changes. Discuss sediment exhaustion, effects of sediment storage capacity and the delivery ratios of sediment as a function of watershed size and characteristics.

Water Quality

a) Define nutrients, major elements, trace elements, anions and xenobiotic compounds. Provide some indication of the role of these elements and compounds in pollution and in the characterization of the water, e.g. pH, dissolved oxygen, specific conductance and temperature. Relate to productivity and toxicity. Include ground water, surface water and rainwater in this discussion

b) Discuss sources of nutrients, xenobiotics and acidifying substances in some detail and, where necessary, relate to elemental cycling in the environment, e.g. exchanges between airshed, watershed, groundshed, biological production and decay

c) Define diffuse sources and point sources and list land and human uses relative to these two categories. Tabulate elements and compounds originating or enhanced by these sources

Transport

a) Transport processes in depth always relating back to erosion, run-off to solutes and sediment binding

b) Describe analytical methods for all categories ofpollutants and give a short overview of monitoring. Again focus on the hydrograph, soluble and particulate compartments

c) Discuss relationships of discharge curves of pollutant categories relative to the hydrograph. Show how these relationships can be used to provide a clear understanding of the sources of the pollutants

d) Describe monitoring of river systems, methods of calculating loads and fluxes. Discuss the needs for these activities relative to downstream river waters and use in watershed management

e) Discuss sediment transport and deposition in rivers. Show how major changes in the watershed, such as dams and deforestation, may dramatically change infiltration, erosion, stream bed morphology and associated hydrology giving rise to amplified flooding as degradation of downstream water quantity, qualify and ecology. Use examples such as the Ganges and Danube Rivers.

River Basin Management

Land use and human intervention are the prime cause of water quality problems in rivers and in materials delivery to lakes, reservoirs and coastal seas. Management of the quality of receiving waters is based on watershed management or management of man's use of the land surface.

Since it is the change to land use practices, together with changing land use, that forms the major component of basin management, then involvement of the land user and public is necessary. The curriculum must thus incorporate:

a) The development of public awareness and public participation in decision-making and land use planning

b) Methods of social awareness surveys to determine and recognize public opinion

c) Inventories of land types, land uses and transportation and forestry, etc, in GIS format

d) Relating pollutants to specific land uses and sources

e) Discussion of techniques both social and engineering

f) Description of what constitutes a sound watershed management plan

g) Description and use of watershed models

h) Discussion on means, social and legislative, of implementing a plan and the monitoring requirements to observe results and to make any necessary adjustments.

Recommended reading

Allan, J.D. 1995 Stream Ecology. Kluver, Dordrecht, 388 pp.

Ward, A.D. and Elliot, W.J. 1995 Environmental Hydrology.

5. Global issues

General Introduction

a) The significance of globalization, commerce, and public travel in bringing about the rapid integration of global ecosystems with both benefit and ecosystem degradation

b) Transboundary movement of waters (fluvial, limnic, and marine), and stress the juxtaposition of airsheds and watersheds in the context of Transboundary flow and materials transport

c) In general terms the impacts of global issues on society, emphasizing the role of water, both quantity and quality

d) The four issues that are to be discussed in the module with brief reference to other issues, particularly to the issues relating to Transboundary pollution and international management of waters in multi-jurisdictional systems.

Impact of Global Climate Change on Water Resources

What is the meaning of climate warming and climate change. Define recent geological changes and the natural warming and cooling of the global climate. Where do we stand currently relative to historical cycles and how is the position being affected by human activities.

Greenhouse gases. Greenhouse gases and impact on retaining heat in the atmosphere

The implicated gases. Their sources both natural and anthropogenic. Show the global cycling of the gases but focus on CO_2 and its historical and current increase. Here one should note briefly the role of sulphur and briefly allude to acidification as a Transboundary problem which has not been eliminated Global CO_2 models and show the predictions from these models for temperature increases both spatial and temporal. Relate the predictions to the recent historical trends in temperature increase and CO_2 build-up.

Impacts on Water Systems. The potential impacts of global warming on local climate change, in geo-climatic regions, desertification and other effects such as changes in surface water flows, groundwater re-charge, lengthened growing seasons, etc.

Glacier retreat and melting of polar ice. The impact of rising seawater levels. The marine circulation pumps in the North Atlantic and discuss the impact of cessation, slowing or geographical change in ocean currents and potential implication of rapid cooling in the northern hemisphere.

Management. current effort to manage gaseous emissions and the implementation ofthe Kyoto Agreement.

Technological and societal changes may be adopted to stop or reverse climate warming.

Water Related Environmental Risk of Global Importance

a) Review the major issues affecting water resources focussing on global significance of eutrophication, disease vectors and environmental contamination by toxic substances. Show how many of these are inter-related and how many have been overcome by strict controls and effective management

b) Eutrophication. Eutrophication is a global problem because of its response to local conditions of land use and sewage waste disposal. Discuss in the context of population and waste disposal practices. Give examples of remediation such as the Black Sea, the Great Lakes of North America, the Swiss lakes, etc.

c) Disease Vectors. Define the major diseases carried in water particularly cholera and typhus (not insect vectors). Show how these relate to waste disposal and hence are closely linked to eutrophication.

Non-indigenous species in the Black Sea littoral zone

Introduction

Against the background of eutrophication in the Black Sea, there has been another important ecological event - the introduction of foreign species that has occurred since 1970-1980. The most spectacular appearance of an alien species along the Romanian coast was that of the bivalve mollusc *Mya arenaria.* The presence of this species has resulted in multiple impacts on the structure and functioning of the ecosystems it populates.

Most of the Romanian research into non-indigenous species (NIS) has been concerned with *M. arenaria* (*Gomoiu*, 1981a, 1981b, 1981c, 1983a, 1983b; *Gomoiu and Petran*, 1973; *Petran and Gomoiu*, 1972). The great beach deposits formed by this immigrant species after each storm attract and maintain abundant populations of seagulls and are also used by the local populations to feed poultry. The spectacular appearance of this species triggered research that has lasted for over 10 years. The aim of the research was to understand the development of the populations, their bioproductive and biochemical potential and the possibilities for using this new resource in aquaculture or for the extraction of active biological substances (e.g. medicinal products).

M. arenaria is not found in the Mediterranean Sea, but has a Boreo-Atlantic origin and a circumpolar distribution: North Atlantic - all the littoral shallow water zones; some of the northern seas; then off the Pacific coast to the San Francisco Gulf and from the Kamchatka Peninsula to the Southern region of Japanese islands. The introduction and rapid colonisation of the Black Sea by *M. arenaria* has created many problems and poses a real challenge for scientific study. It was introduced with ships ballast water and, after only a very short pioneering stage in the Black Sea, it out competed some indigenous species. Despite its quantitative fluctuations from year to year it has become the dominant element of sedimentary bottoms down to 30-40 m depth. The Romanian continental platform seems to be one of the most favourable zones in the Black Sea for the development of the new immigrant.

The establishment of Mya arenaria populations

In the Black Sea, *M. arenaria* was first reported in 1966 near Odessa, and then found in the zones in front of the Dnieper and the Bug River (Beshevly and Kolyagyn, 1967). One year later, some small specimens of the new NIS were found along the Romanian shore in some sheltered areas (*Gomoiu and Porumb*, 1969).

An important monitoring research programme on *M. arenaria* in the Romanian Black Sea littoral zone started in 1970 and continued until 1982 (*Gomoiu and Petran*, 1973; *Petran and Gomoiu*, 1972). In the first two years of research (1970-1971), the qualitative and quantitative spread of *M. arenaria* populations along 245 km of coastline was investigated with a network of 105 stations on the sedimentary bottom down to 10 m depth. The recorded density and biomass show that, within 4-5 years of its colonisation, *M. arenaria* had settled perfectly along the Romanian coast in the fine sediment zones (sand, muddy-sand and sandy-mud) and in some places reached values of over 8,000 specimens per m^2 and 16,000 g m^{-2}. The *M. arenaria* populations were found in a nearly continuous strip between the Danube mouth and Constantza (according to the distribution of fine sediments). To the south of Constantza the bivalve had quite a patchy distribution, limited to areas of fine sediment accumulated between rocky platforms that harbour another NIS, *Rapana venosa,* which had colonised earlier.

At the beginning of the 1970s *M. arenaria* populations had an uneven distribution, being found in small agglomerations in the sheltered zones of harbours; these small agglomerations were even found in waters of only a few centimetres depth. *M. arenaria* avoids high energy zones and, therefore, at the open beaches along the entire coast populations they were found on the bottom in depths of 2-3 m down to 10 m. By sampling down to 32 m, it was shown that the spreading of *M. arenaria* populations was still going on and that the species was reaching new areas.

The size structure of populations from different zones was quite heterogeneous. Comparative analysis of the average length of shell, in 1970 and 1971, gave the following values:

34-49 mm at 0 m depth,
19-28 mm at 3 m depth,
22-23 mm at 4 m depth,
14-19 mm at 9 m depth.

The longest shells belonged to specimens from the beach deposits. The size structure over the two years of the preliminary study can be summarised as follows (Slide 2). In 1970, in 10 of the 17 investigated beaches, small individuals from +10 mm class were most abundant at 27%. These were not found in 1971.

- In 1970 the classes with the highest frequencies were +20 mm (25%) and +30 mm (31%), and in 1971 they were +40 mm (24%), +50 mm (21%) and +60 mm (20%).
- In 1970 individuals of the +90 mm class were missing, while in 1971 they were found, but only as a small proportion (<1%).

After 1971 the monitoring research ofthe *M. arenaria* populations along the Romanian littoral zone were extended to 30-35 m depth through a network of 55 basic stations spread over 11 transects (*Gomoiu and Petran*, 1973) and an area of almost 1,800 km^2 .

The active settling of *M. arenaria* detected in 1970-1971 (Petran and Gomoiu, 1972) continued with marked intensity in 1972, when significant changes in the size and distribution of the populations, as well as in their size structure, were recorded. At shallow depths *M. arenaria* populations settled much better than in previous years, in some places a maximum density of about 4,300 specimens per m^2 and a biomass of 900 g m^{-2} were achieved with small sized individuals less than 20 mm in length. All the results obtained from the 1970-1982 studies showed that *M. arenaria*, after only 10 years of existence in the Black Sea, had developed from a poor and randomly distributed population covering a reduced area between the 4 m and 8 m depth contours, to become a common element on all alluvial bottoms to depths of 30 m in Romanian waters.

The populations of *M. arenaria* showed some important fluctuations; they took over as the dominant species from *Corbula mediterranea* - a species that characterised one of the most eutrophic biocoenoses of the Black Sea. The results of long-term research show how *M. arenaria* out competed *C. mediterranea* (Fig. 5.1).

The effects of eutrophication/pollution (*Gomoiu*, 1981d) were reflected in the *M. arenaria* populations. For example, in 1974, before the great algal bloom of 1975 (*Bodeanu et al.*, 1998) the *M. arenaria* populations were present on all the sedimentary bottoms north of Constantza, forming biomasses of more than 1 kg m^{-2}; in 1982, the *M. arenaria* populations and the species associated with them were absent from large areas of the bottom, especially in shallow the sedimentary bottoms north of Constantza, forming biomasses of more than 1 kg m^{-2}; in 1982, the *M. arenaria* populations and the species associated with them were absent from large areas of the bottom, especially in shallow water zones. This was a consequence of the algal blooms that were followed by mass mortalities of benthic organisms (Slide 3). More details on the ecological state of the *M. arenaria* populations and their associated species two years after the dense red-tide of 1975, with its consequences, can be found in a special monitoring report from 1977.

By the beginning of the 1980s, *M. arenaria* populations had stabilised on the Romanian continental shelf up to 35 m depth (Fig. 5.2). Together with *M. arenaria,* 13 other species of autochthonous molluscs had also appeared in samples. The species with the highest occurrence in the 84 stations of the monitoring network, were *M. arenaria* (70%), *Cardium edule lamarcki* (49%), *Mytilus galloprovincialis* (30%), *Corbula mediterranea* (20%) and *Spisula*

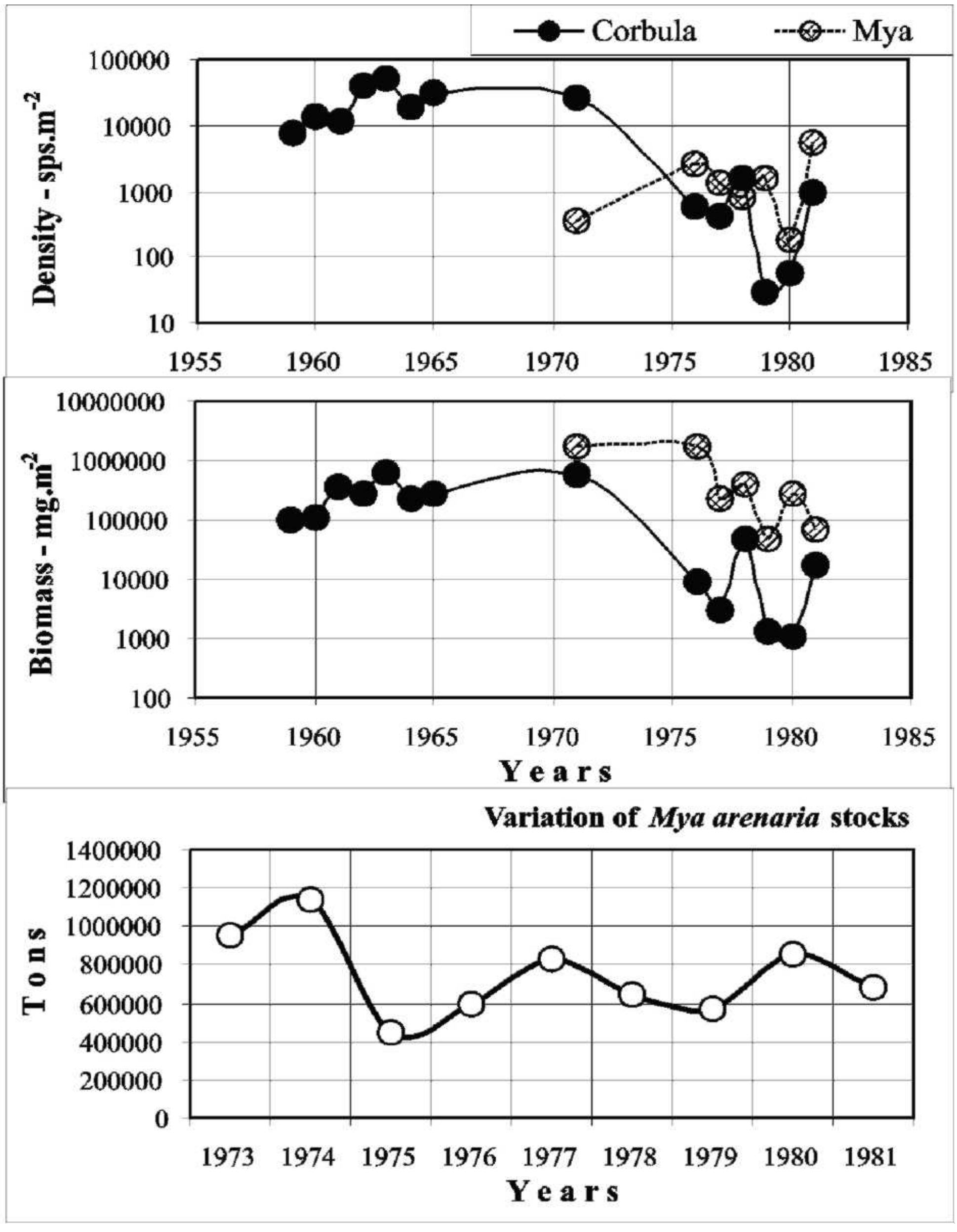

Fig. 5.1. Variations in *Corbula mediterranea* and *Mya arenaria* populations from the sedimentarv areas of the Black Sea between the mouth of the Danube and ConMva

subtruncata (20%). The frequency of the other molluscs was reduced, but within narrow bathymetric limits the situation was different. However, in nearly all bathymetric zones, the populations of the newcomer *M. arenaria* were the most widespread.

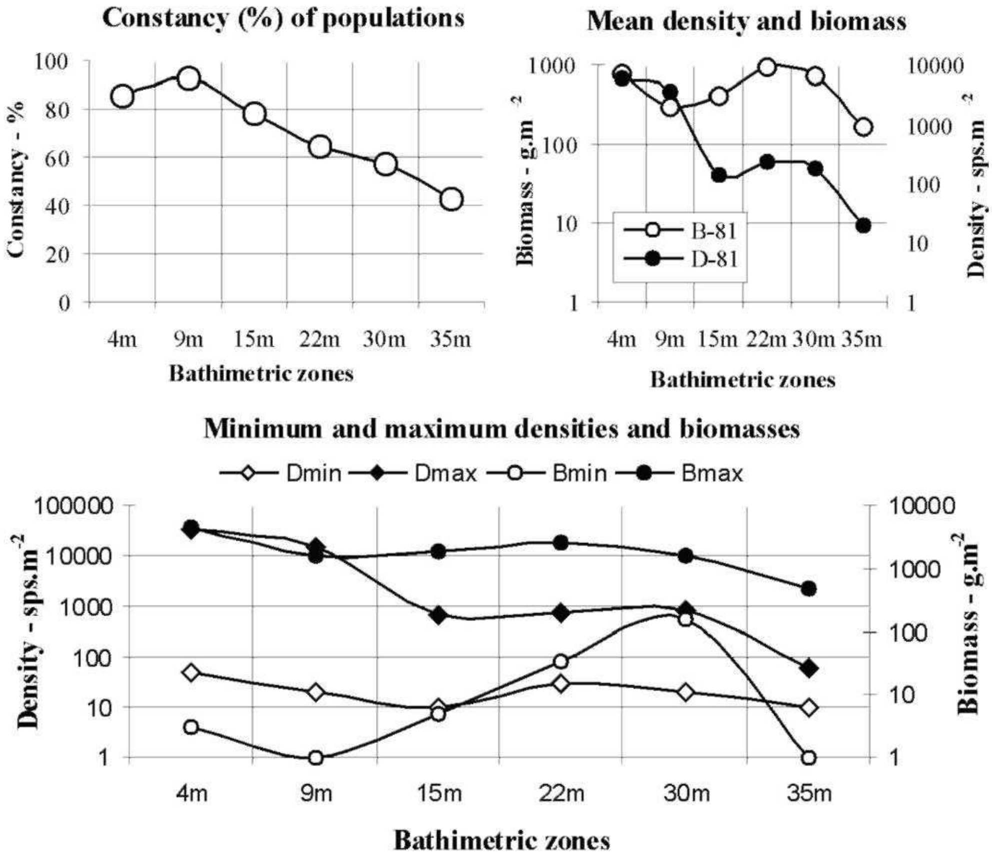

Figure 2 Synoptic analysis of *Mya arenaria* populations in 1981

Fig. 5.2 Characteristics of Mya erearia (by *Gomoiu*, 1983)

A few main nuclei of species associations and characteristic biocoenoses could be delimited at various depths, based on the analysis of mollusc frequency. In comparison with thesituation in the previous decades, i.e. the period before *M. arenaria* invasion (*Bacescu et ah,* 1971), the mollusc associations were scarcer. After the *M. arenaria* invasion, a simplification took place of the benthic community structure at the macrobenthos level represented by molluscs.

The graphic representation of the presence of different molluscs at various depths of the Romanian littoral zones clearly highlights the development of a *M. arenaria* continuum from the shallow water zones to the offshore zones at 35-40 m depth. The former continuum of the small bivalve *Corbula mediterranea* (the species with which *M. arenaria* competed from the beginning) has been gradually reduced and the *Mytilus* continuum was also affected by reduced numbers.

The end of the consolidation period of *M. arenaria* populations in the new biotope was characterised by the reduced presence and frequency of other species of molluscs (56% of those recorded in the reference zone in the preceding period). The great number of associations and their random mosaic distribution in the Romanian littoral zone are evidence of the disorder and fragmentation of the major biocoenoses identified in the past decades.

Within the monitoring programme, each year since 1972, mortalities of benthic organisms have occurred to a greater or lesser extent in the Romanian littoral (*Gomoiu*, 1977a, 1981d, 1983b). Out of 550 stations analysed between 1972 and 1982, mortalities were registered in 284 (51.6%), and this completely or partially affected the benthic populations, especially *M. arenaria*, as follows:

In 59 stations (10.7%) mass mortalities occurred up to the abiotic level followed by H_2S appearance on the bottom,

In79 stations (14.2%) mass mortalities were registered, but new generations of *M. arenaria* juveniles immediately repopulated the zone (post-mortality repopulation), and In 146 stations (26.5%) partial mortalities occurred.

All the mortalities were characterised by spatial and temporal discontinuity.

In the period 1973-1982, on the sedimentary bottoms up to 30 m depth, between the Danube mouth and Constantza, the total stock of *M. arenaria* lost by mortality was estimated at $4.1 x 10^6$t (48.6% of this was mass mortality).

The most important losses (21% of total stock) occurred in 1975, 1978 (10.4%), 1981 (14.8%) and 1982 (13.5%). When compared with the stocks estimated in the Romanian littoral zone in the year before the mortalities, the losses were greater at 70% in 1975, 54% in 1977 and 1978, 64% in 1979, 66% in 1981, and 77% in 1982. On average, almost half of the investigated area (1,765.11 km^2) was affected annually by mortalities. The most affected zones were the coastal zones and, as depths increased, the affected areas decreased.

Along the Romanian littoral zone mortalities generally increased from the Danube mouth to Constantza. Therefore, the average intensity of mortality decreased, as a rule, from the coast to the offshore zone and from the north to the south (Slide 8).

Mass mortality of the *M. arenaria* populations clearly revealed the presence of very unsuitable conditions in the shallow bottom of the Black Sea. These unfavourable conditions have spread to deeper zones usually populated by mussels, and which had represented a strong biofiltration belt. At the beginning of the *M. arenaria* population explosion on the sedimentary bottom of the Black Sea Romanian littoral zone, no essential changes could be observed in the qualitative and quantitative structure of the associations of autochthonous molluscs (*Gomoiu and Petran*, 1973). *However*, in 1975, when *M. arenaria* had already undergone massive population development, a thorough study carried out in the colonised zones revealed the scarcity of native species.

The monitoring programme was completed in 1981 and produced the following conclusions (Gomoiu 1981a, 1981b, 1981c):

- The populations of *Corbula*, which formerly (before *M. arenaria* entered the Black Sea) had an average of 25,000 specimens per m and 170 g m , had became very rare.+

- The stock of *Corbula* and the area occupied by this species in the Romanian littoral zone, formerly estimated at 112,000 t over an area of 650 km , had decreased irregularly from year to year (470-4500 t over an area of about 475 km) during the period of consolidation of *M. arenaria* populations.

The decrease in the populations of *Corbula* and the areas they occupied in the Romanian littoral zone was initially a consequence of the competition between this species and the new bivalve; *M. arenaria* out-competed the autochthonous *Corbula.* It is also likely that the tiny bivalve *Corbula* declined because the more robust *M. arenaria,* which burrows into the sediment and filters all the particles from the water, also filtered the larvae of *Corbula.*

In the late 1980s when *M. arenaria* was well established in the Romanian littoral zone, the associations of benthic organisms on the shallow sea bed (*Tiganus and Dumitrache*, 1991-1992) were greatly reduced in comparison with the 1960s, prior to *M. arenaria* introduction (*Bacescu et al.,* 1965). Out of 48 species recorded in the 1960s, 32 species were no longer found in the 1980s even though some had occurred in quite a high precentage of the samples taken in the 1960s (e.g. *Podocoryne cornea* - 15%, *Nereis diversicolor* - 21%, *Heteromastus filiformis* - 12%, *Abra nitida milaschewichi* - 18%, *Spisula subtruncata* - 18%, *Nassarius reticulatus* - 18%, *Cyclope neritea* - 55%, *Pseudoparamysis pontica* - 15%, *Bathyporeia guilliamsoniana* - 15%, *Cardiophylus baeri* - 18%, *Crangon crangon* - 73%, *Diogenes pugilator* - 12% etc.); instead, 16 new species appeared *(Phyllodoce maculata* - 12%, *Polydora limicola* - 44%, *Capitella capitata* - 16%, *Scapharca inaequivalvis* - 19%, *M. arenaria* - 76% etc.). The changes in the mollusc populations are illustrated in (Slide 9).

The cause of these changes is difficult to establish: is the explosion of *M. arenaria* the cause or could the changes in the benthic populations be the consequence of eutrophication and pollution? Both of these have been dominant phenomena in the Black Sea in the last decade of the 20th century. Whichever is the cause, *M. arenaria* consolidates its populations, while *Corbula mediterranea* is on the decline (Gomoiu, 1981c).

The need to control *Mya arenaria* populations in the sedimentary zones down to 30 m depth highlighted the tendency in 1977 for the species to reflourish and consolidate. The populations surviving the consequences of algal blooms in 1975 recovered at an average rate of 33% per year. Within the associations of molluscs on the sedimentary beds, *Mya arenaria* had become dominant in terms of frequency and biomass. Since 1977 its populations have demonstrated the following features:

- Uneven spatial distribution in the Romanian littoral zone.
- Quantitative changes in the distribution of populations, with the highest densities being dominant in shallow zones and highest biomasses occurring in zones 10-30 m in depth.
- Small-sized specimens dominant in the populations, which is an indicator of better recruitment (over 50% of the populations consist of specimens 20 mm in length).
- Relatively reduced survival of the generation in 1976. Spatial distribution showing a tendency to aggregate.

In 1977 populations of other specimens associated with *Mya arenaria* in various communities on the sedimentary beds showed a decline, which was marked by the following modifications:

- A reduction of the specific density by 9-42% in different bathymetric zones due to the

absence of more sensible species in the associations *(Abra ovata, Plagiocardium simile*, *Lucinella divaricata*, *Retusa truncatella* etc.).

- Reduction in the frequency of diverse species over the whole area of study and the various bathymetric zones.

General reduction in densities and biomasses, or extinction of populations, in some zones.

Serious decrease in the number of most species compared with 1976 *(Cardium edule lamarcki* by 22%, *Corbula mediterranea* by 59%, *Mytilus galloprovincialis* by 87%, *Abra nitida milachewichi* by 49% etc.).

The causes of the regression of the mollusc populations are difficult to establish, especially as they had not been studied prior to 1977. However this decline is considered to represent, at least partially, the result of the interspecific relationship between the native fauna and the new immigrant *Mya arenaria*, which develops rapidly in the Black Sea. With its bigger size and biofiltration "pumps" (nonselective seston filtering) *Mya* out-competes other species completely.

In the sedimentary zones situated between the Danube mouths (Sulina - I) and Constantza (Mamaia - XI) up to 30 m depth, a stock of almost 970,000 t of sedimentophilous molluscs was estimated. This stock consisted mostly of *Mya arenaria* populations (82%) with the contribution of the other molluscs to this stock being insignificant *(Cardium edule lamarcki* - 7%, *Mytilus galloprovincialis* - 4%, *Corbula mediteraneea* - 0,2%, *Abra nitida milachewichi* - 0.2% etc.).

The total stock of *Mya arenaria* estimated in 1977 (860,000 t) had the following bathymetric distribution: 5% in the shallow water zone up to 10 m deep, 45% between 10 m and 20 m depth, 42% between 20 m and 30 m, 7% between 30 m and 40 m and nearly 1% in the 40-50 m depth zone. Although the stock of *Mya arenaria* was high in some places it did not present any possibility for commercial exploitation because it was distributed over a large area (almost 1,800 km^2) and consisted of very young, small-sized populations that permanently registered important quantitative changes.

The important quantitative fluctuations in time and space of the densities and biomasses of *Mya arenaria* in the zones where it settled, then its introduction into new deeper zones, as well as the domination of the structure of the populations by juveniles, showed that this species which was recently introduced into the Black Sea, developed rapidly in the first 10 years but its populations had not reached the ecological stability and dynamic equilibrium necessary in the ecosystem to make them exploitable.

The considerable quantities of *Mya arenaria* recorded from the Romanian littoral zone in 1977 proved that the biological productivity of the sedimentary bottoms up to 30 m depth had increased. At the same time, biological productivity had simplified due to the development of one species, Mya, which had become the dominant species in the macrobenthic sedimentophilous associations, with a special role in the littoral ecosystem:

- It represented a trophic basis for benthophagous fishes.
- It released a great quantity of larval plankton, which provided food for feeding planktivorous fishes.
- It formed a strong biofiltration belt that successfully consumed large quantities of primary phytoplanktonic production (which is in excess when eutrophication occurs), in seston and even in zooplankton, thus contributing to the purification of littoral waters.
- It represented a source of material for shell sands and beach deposits.

Considering the importance of *Mya arenaria* populations in the Romanian littoral zone and the fact that they were the subject of a large-scale natural experiment in self-acclimatisation, it is absolutely necessary to study the qualitative and quantitative state and the evolutionary tendencies of this population in the future. To understand the ecological aspects is essential for finding solutions to the scientific and practical problems raised by the presence of *Mya arenaria* (e.g. commercial use as food or in pharmaceutical industries for bioactive substances, cultivation).

Rapana venosa

The gastropod *Rapana venosa* (Val.), a native species of the Sea of Japan, was reported for the first time in the Black Sea as *Rapana thomasiana* Crosse in Novorossysky Bay in 1947, although it was considered to have settled in the Pontic basin somewhat earlier in 1930-1940. This large voracious predator rapidly colonised the seabed of the shallow water coastal area, practically around the whole Black Sea. In 1949 *R. venosa* was found in the oyster bank of Gudautsk, in 1954 at Yalta and Sevastopol and finally in 1963 along the Romanian shores (*Bacescu*, 1963; *Chukchin*, 1961c; *Drapkin*, 1953; *Eberzin*, 1951; *Stark*, 1957).

The rapid advance and explosive development of dense populations of the new immigrant along the Caucasus shore, where it appeared for the first time in the Black Sea, and then to the Anatolian, Krimean, Romanian and Bulgarian shores, might be explained initially by its trophic requirements, i.e. it spread to where food was plentiful. *R. venosa* is very fertile and does not have serious competitors and it can also adapt to low salinity and tolerate water pollution and hypoxic conditions. Thus the newcomer has succeeded in forming dense populations, with a total biomass in the order of thousands of tons in some areas (*Zolotarev*, 1996).

R. venosa, which mainly feeds on oysters and mussels *(Ostrea edulis, Pecten ponticus* and *Mytilus galloprovincialis pravailing),* completely destroyed within a rather short period the entire stock of these molluscs in the Gudautsk bank, totally changing the pattern of the benthic communities (*Stark*, 1957). Similar situations are well known in the history of the Black Sea; the disappearance of certain bivalve species can be clearly followed in the geological deposits (e.g. *Pseudomussion denudatum* Reuss, *Ostrea cochlear* Poli, *Ostrea griphoides* Schloth. var. *sarmatica* Fuchs) and may be attributed to the rapacity of such gastropods as *Murex subclavatus* Basterot, or species of the *Nautica and Massa* genera. Having destroyed its preferred food, i.e. the larger species *(Ostrea edule taurica* Siemaschko, *Mytilus galloprovincialis* Lam., *Modiolus adriaticus* Lam.), *R. venosa* was forced to attack smaller species *(Chione gallina* L., *Pitar rudis* Poli, *Paphia rugata* (B.D.D), *Spisula subtruncata* (Renier) etc.), migrating east and westwards along the coastline from Gudautsk rapidly populating new zones in its search for oysters and mussels (Slide 10).

At the Romanian coast the species first appeared at the mouth of the Danube River and rapidly spread southwards, becoming a common element in shallow waters, both on sandy and rocky bottoms. For a long time *R. venosa* was reported from the Romanian coast of the Black Sea on the basis of empty shells found on the beaches. However, the large number of specimens deposited by the storms in the 1960s provided proof of the invasion of this gastropod and of the potential danger this rapacious snail represented for the littoral malacofauna (*Grossu*, 1970; *Grossu and Lupu*, 1964; *Iliescu and Radulescu*, 1968; *Popescu-Marinescu and Paladian*, 1971).

On the Romanian coast at the beginning of the 1970s, the average size of *R. venosa* was 63 mm, with larger specimens occurring in the rocky bottom population than in the sedimentary bottom population (Slide 11). The maximum height of shells reached 112.9 mm on rocky bottoms and 72.0-87.8 mm on sedimentary bottoms. It is likely that on the sedimentary beds *R. venosa* have to expend more energy digging for pray than on rocky beds, where their food is readily available with little effort. The mean height of *R. venosa* reached 67.36 mm where the populations living on rocky bottoms had access to mussels, which are their preferred food organism, thereby supporting optimum growth rates.

When their preferred food organism is missing, or limited, *R. venosa* has a very low growth rate and individuals are much smaller (*Chukchin*, 1961a, b).

The results of underwater diving observations made at the beginning of 1970s on the Romanian littoral zone can be summarised as follows:

- *R. venosa* were found along the shore up to 10 m depth, on sandy and rocky bottoms, as indicated by the great number of living and empty shells washed up on the beaches along the whole coastline. Specimens of *R. venosa* were also found on shelly bottoms, up to 30 m depth.

- On the sandy bottoms, in the *Corbula mediterranea* association, *R. venosa* was found at depths of 3-8 m, with an abundance of one or two specimens per 100 m^2. The snails were very clearly discerned against the monotonous uniformity of the underwater sandy desert seascape, moving upon the substrate, usually in pairs. *R. venosa* was found only rarely in very protected areas of the Romanian coast, on the bottoms covered with eel-grass, where they occurred as very small specimens. It is only in the larger *Zostera* meadows, in the vicinity of hard substrata covered by *Mytilus galloprovincialis,* that *R. venosa* populations were more abundant.

- *R. venosa* occurred most abundantly on rocky bottoms and was frequently encountered in the zone of 4-10 m depth, exhibiting a maximum density at 8-10 m, with up to 10-12 specimens per m^2. In sheltered and deeper places, *R. venosa* usually lived under overhanging stones, or on the rocky thresholds. They were never observed on denuded stone platforms under strong sunlight. They preferred the shadowy places and were very sensitive, so that when a diver approached they reacted immediately, drawing back into their shells. As a rule, *R. venosa* lived at the edge of the mussel colonies upon which they were feeding; in areas with low mussel density they were more abundant (2-4 specimens per m^2) than in the areas with compact mussel colonies (1-2 specimens per m^2). The highest densities were observed along the rocky infra-littoral strip, usually towards the end of summer (August-September) when the species appeared to cluster together, towards the 8-10 m depth contour, which was the lower limit of the stone platforms. In autumn, 10-12 specimens per m^2 were found within an area of about 2 ha, located at the stone edge. Therefore it appeared that *R. venosa* performed a true seasonal migration due to temperature changes and storm intensification.

- During the rapid growth of the populations, the egg-capsules of *R. venosa* were recorded along the Romanian coast after mid June, being encountered all through the summer until September. They were placed on the largest specimens of living mussels or mussel shells or, in the sandy areas, on bigger shells, usually on *M. arenaria* shells. The colour of the egg capsules after hatching was milky-white and after 7-14 days they turned yellowish-brown, and then bluish-violet. 5 Divers observed the gastropod feeding at the edge of mussel colonies. It envelops a mussel with the sole of its foot as if to "suffocate" the bivalve, and then pushes in between the valves and eats up the entire contents. On the sandy bottoms *R. venosa* consumed larger psammobiotic bivalves (one snail was seen holding three specimens of *Chione gallina* under the sole of its foot while it fed upon them). In experimental conditions, *R. venosa* began to feed actively after three days starvation, consuming one mussel every 2-3 days (about2gof fresh meat, which represents 14-15% of the snails own body weight).*R. venosa* is very active in searching for its food. In several experiments carried out under laboratory conditions batches of 10 snails, from +60 - +70 mm height class, were kept without food in a tank connected to another tank by an 8 cm opening situated at the upper part of the tank wall, at about 30 cm from the bottom. *M. arenaria* specimens were put in the second tank; some of them were buried in the sand and others were placed on the sediment surface. After three days *R. venosa* began to climb the vertical glass wall of the tank and passed into the second tank containing *M. arenaria* where they began attacking the bivalves laying on the sand and feeding on them. A few days later the snail began to dig up the *M. arenaria* specimens dwelling in the sediment, removing the upper sand layer and devouring the soft-shelled clams. *R. venosa* attacks and feeds on *M. arenaria* and other shellfish in the same manner as it feeds on *Mytilus*, but it takes less time. With such feeding habits *R. venosa* succeeded in actively influencing the biocoenoses of the soft sedimentary bottoms and the hard rocky bottoms, contributing to the reduction of many bivalve populations (e.g. *Chion gallina, Tellina tenuis, Spisula subtruncata, Pitar rudis, Paphya rugata, Modiolus adriaticus* etc.).

The present composition of beach deposits suggests that *R. venosa* populations are in decline along the Romanian coast; trophy shells of this rapacious gastropod, caught in important quantities for consumption at the beginning of 1990, have become very rare. The same situation seems to be occurring in the entire Black Sea. On the Anatolian coast there are no more *R. venosa* reserves to be exported to Japan as in the 1970s, when more than 8001 of snail meat were exported per year. In the north-western part of the sea the *R. venosa* populations are poor and their influence on the marine ecosystems seems to be fairly weak. The cause of these low densities is unknown, although food resources seem to be abundant enough and levels of water contamination had been reduced (Zolotarev, 1996). It is likely that overfishing of *R. venosa* has led to the population decline. Without proper studies and continuous monitoring, it is difficult to give a plausible explanation for the *R. venosa* population decrease in the Black Sea. Information on the ecology of *R. venosa* from the Romanian Black Sea should be considered as preliminary new research is necessary to assess the state of *R. venosa* populations in the Black Sea and to elucidate the species ethology and ecology in the present ecological conditions.

At the beginning of its colonisation phase, *R. venosa* was considered to be a deleterious species in the Black Sea; the wide-ranging dispersal of the new predator stimulated research on use and control of the undesirable rapacious gastropod. The highly adaptive capacity of *R. venosa* was found to be due to the biochemical peculiarities of its haemolymph, namely the high concentration of haemocyanin and protein (up to 16.9%) that provided buffering properties. Further biochemical investigations showed that it could be a real source of some biologically active substances (Serban and Rosoiu, 1992), such as some natural inhibitors of pepsin, tripsin and chemotripsin. However, all the hopes for new drugs that might have been extracted from *R. venosa* are now lost, because of the decline of the populations.

Other NIS in the Black Sea

Ctenophora Lobata

Mnemiposis leidyi was introduced into the Black Sea at the beginning ofthe 1980s and was considered to have major ecological significance (*Vinogradov et al.,* 1989; *Zaika and Sergeeva*, 1990). The introduction of this species had strong consequences for the populations and the trophic structures of the plankton community (*Vinogradov et al.,* 1995, *Vinogradov and Shushkina*, 1992; *Shushkina and Vinogradov,* 1991; *Vinogradov et al.,* 1989), causing severe disturbances in the ecosystems, which had already been disturbed by eutrophication and pollution. This species from the Atlantic coasts of the USA, first mentioned in 1982 in the north-eastern sector of the Black Sea at Novorosiisk, rapidly spread into the whole pontic basin, including the Azov Sea. By the summer and autumn of 1988, it had become abundant throughout the Black Sea, including its offshore regions and probably the Romanian coasts. The population of this voracious predator developed explosively and outcompeted the autochthonous forms, eliminating most of them.

One of the first quantitative studies concerning the distribution of *Mnemiopsis* populations showed that in August 1995 there were approximately 12-16 x 10^6 t of fresh comb-jelly mass in the waters covering the Romanian continental shelf (*Gomoiu and Skolka*, 1997). The presence of *Mnemiopsis leidyi* represents a challenge which should lead to a programme of research and monitoring of this species in the Romanian waters (at least as much as it has already been done for other sectors of the Black Sea).

Ctenophores from the *Beroe* genus were recorded for the first time in the Black Sea in the summer of 1997, near the southern, western and north-western coasts. In 1998 *Beroe ovata* appeared again during the warm season, with a population explosion in August and September 1999, and thousands of specimens were observed daily in the near-shore waters of the Romanian littoral. *Beroe* ctenophores are predators specialised in feeding on other ctenophores, especially the lobate forms of the *Bolinopsis* and *Mnemiopsis* genera. The presence of Beroidea in the pontic basin was closely connected with the development of *Mnemiopsis leidyi* populations, another species that had recently migrated into the Black Sea.

The way *Beroe* invaded the Black Sea is still unclear, with two possible explanations: immigration from the Atlantic area in the ballast water of ships, together with *Mnemiopsis leidyi* (as *Beroe* occurs in the same waters as the latter species) or immigration from the Mediterranean, through the Bosphorus strait. In the latter scenario, the Beroidea could not have colonised the Black Sea earlier because their preferred food, the lobate ctenophores, were not present. It is also possible that the Beroidea entered the Black Sea from the Atlantic, as well as from the Mediterranean. In the future, it is likely that the Beroidea will develop in great numbers, because the lobate ctenophores from the *Mnemiopsis* genus on which they feed are present in the pontic basin in enormous numbers.

Mollusca Gastropoda

Potamopyrgus jenkinsii is a fresh-brackish water species from the South Pacific, particularly from the waters of New Zealand, which was first mentioned in Europe in 1882 in the Thames estuary. Later it adapted to fresh and brackish water lakes, and was found in the Baltic Sea in 1927, at Marseilles in 1959 and on the Spanish coasts in 1953 (*Grossu*, 1986). It was first mentioned in the Romanian littoral zone in 1952 in the lagoon complex Razim-Sinoe where it developed major populations. By using an empty trophic niche, *Potamopyrgus jenkinsii* did not cause major disturbances in the structure of the biocoenoses where it settled, and it is considered a useful species as an intermediate link in the food chains.

Doridella obscura comes from the North America Atlantic littoral zone, where it occurs along all the littoral zone of the USA: Florida, Texas, the Gulf of Mexico and the Caribbean Sea. It feeds on bryozoan colonies, especially *Electra crustulenta,* and both species are often found together. *Doridella obscura* is resistant to variations in salinity, and occurs in waters with a salinity of 2 psu (in some river estuaries) as well as waters with a salinity of 23 psu. This new immigrant probably invaded the Black Sea early in 1980-1985, at approximately the same time as another North-Atlantic species, the comb-jelly *Mnemiopsis leidyi* Agassiz. The low salinity of the Black Sea and the temperatures recorded there were not an obstacle to the spreading of this gastropod. The species was recorded for the first time in the north-western area of the Black Sea in 1986, then south of Crimea where it was collected from the shells of farmed mussels, and also in association with *Electra crustulenta* (*Roginskaya and Grintsov*, 1990, 1995, 1997). It was most probably brought into the Black Sea with the fouling communities of ships hulls as adults or as egg-capsules. The theory that the larvae were transported in ballast water is less probable.

In the Black Sea, *Doridella obscura* takes over an empty ecological niche and develops stable populations that are well adapted to their new environment. Being a predator with no competitors, this mollusc has unlimited food resources (in the Black Sea there are no predators specialising on Bryozoans exclusively). In its Pontic habitat, the Bryozoans such as *Electra*, *Membranipora* and *Conopeum* species, which are its preferred prey, are widely distributed, covering mussels and rocky bottoms between the surface and 10-15 m depth. In the future, it is likely that this species will become common on the rocky bottoms in the biocoenoses with *Mytilus galloprovincialis* on the Romanian Coast. The presence of more than 10 young specimens in a single sample is a sign that this species is no longer rare.

Mollusca Bivalvia

Scapharca inaequivalvis is a native of the Indo-Pacific region, and was accidentally introduced into the Mediterranean sometime around the 1960s. The species developed large populations rapidly, especially on the western coasts of the Adriatic Sea, and also spread into the lagoons near the Po River estuary. In the Black Sea it was recorded for the first time on the beaches north of Constantza (*Gomoiu*, 1984). Comparative studies showed that the bivalve *Anadara* sp., observed along the Bulgarian coast by *Marinov et al.* (1983) was also *Scapharca.* The observations and biometric studies performed on this bivalve's populations suggested that they had invaded the pontic basin in 1980-1981, because the specimens collected were 42-50 mm in size and 16-18 g in weight, which corresponded to four years of age (Gomoiu, 1984).

The populations of this bivalve spread rapidly on the sandy and muddy bottoms and they became common. The populations of this immigrant thrive even under the difficult conditions found in the benthic ecosystems, which are influenced by more and more frequent algal blooms and by periodic hypoxia and anoxia. However, the ecological tolerance of the species enables survival in waters with variable salinity, as well as in conditions of oxygen deficit. It is considered to be a marker species for organic pollution. Since 1989, the populations of this species have become integrated into the sandy infralittoral biocoenoses. The numbers of *Scapharca* decrease with the increase in *Cardium edule* populations.

Anodonta woodiana is a freshwater bivalve from the Amur River basin that was first recorded in Europe in 1978 in Romania. It was accidentally introduced with juvenile Chinese carp *(Ctenopharyngodon idella)* taken to Romania for acclimatisation. This species is very resistant to pollutants and gradually dominates the associations of molluscs in the main rivers of the West Romanian plain and in the Lower Danube.

Corbicula fluminea is an eastern Asian species introduced into the USA in 1920. It appeared in South America in the 1960s and in Central Europe in the middle of the 1980s and rapidly became the dominant species in the Rhine basin. It was known in Europe as a fossil species because shells were discovered in Quaternary deposits. In the winter of 1997, young living specimens (2-4 mm) of *Corbicula fluminea* were identified in samples collected from the Romanian sector of the Danube (the Iron Gates). This species probably entered the Danube via the Rhine-Danube canal and it is likely that it will soon be observed in other zones of the Danube as well.

Crustacea Decapoda

Rhithropanopaeus harrisii tridentatus originated fromthe coastal zones ofthe IndianOceanand was brought to Europe at the beginning of the 20th century. During 1932-1935 the "Dutch crab" was described from the mouths of the Dniester and the Bug, and then in 1954 it was observed along the Bulgarian coasts. A little later it was found in the Romanian littoral zone in the brackish water lagoons of the Razim-Sinoe Complex (Fig. 6, Slide 10). The presence of this brackish water species was later described in the shallow water marine zones and in some fresh water lakes. *Rhithropanopaeus harrisii* is a eurytopic organism occupying a great variety of ben-thic environments and waters with less than 15 psu because it is very resistant to low salinity and low temperatures. This crab is currently consumed by predatory fishes, such as the pike perch. Other introduced decapoda that are less frequently reported include *Callinectes sapidus* and *Eriocheir sinensis.*

Entoprocta Urnatelidae

Urnatella gracilis is one of the freshwater species that migrated into Europe at the beginning of the 20th century, but its distribution is still not well known. The species is a native of the eastern USA, where it occurs in the inland waters of some eastern coast states from the Great Lakes area, as well as in Texas, Ohio and Indiana.

In the Black Sea basin the species is mentioned by Bacescu (1954), from the Danube and from the northern coast of the Black Sea (Taganrog). Established populations do not usually reach significant biomasses, although sometimes the population numbers are high. *Urnatella gracilis* can survive in brackish waters with a salinity of 3 psu. The species has also been recorded in the lower Dniestr under the name *Urnatella dniestriensis* (Zambriborschs, 1958) and in Hungarian territory (Tisza and Danube).

Recently, in the autumn of 1998, colonies of this species were identified for the first time in the Southern Dobrudja, in the Mangalia Lake. The colonies were covering empty shells of *Anodonta cygnaea*, and sometimes the stolons were almost completely covering the shells. In addition to the *Urnatella gracilis* stolons, numerous bryozoan statoblasts have been observed on the same shells.

Conclusions

This summary of NIS in the Romanian Black Sea emphasises the possibility that the gateway to the Black Sea has been permanently opened for exotic organisms. When the Black Sea was in a healthy ecological state, all the niches were occupied and the chances for newcomers to develop sustainable populations were minimal. However, once ecological conditions had been seriously damaged by pollution and eutrophication processes, the empty niches were ready to absorb new populations.

The lessons learned from the arrival of exotic species along the Romanian Black Sea coast can be summarised as follows:

- the process of species introduction is ongoing and it is necessary to pay more attention to the toxic and harmful species;

- the impact of alien species is complex and unpredictable. Species biodiversity monitoring is absolutely necessary with special attention to the proper monitoring of microflora and microfauna;

- legal measures and regulations are necessary to limit the invasion of new species.

Training human resources in biological taxonomy and systematics is a high priority. There are fewer of the older generation of marine botanists and zoologists remaining who understand the Black Sea ecosystems and the measures necessary for their protection. Greater understanding of the biota can only be achieved by educating specialist marine biologists.

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6. Design and interpretation of toxic chemical distribution and stress (Detroit river case study)

Toxic Contaminants in the Aquatic Environment highlight the importance of a stratified random sampling design in providing statistically sound estimates of spatial and temporal patterns of toxic chemicals within the overall aquatic ecosystem.

Introduction

The Detroit River is well documented as being the primary source of contaminants to Lake Erie (*Haffner and Koslowski,* 2000; *Carter and Hites,* 1992). The influence of the river as a source of chemical contamination is such that concentration gradients have been established within the lake (*Stevens and Nielson*, 1989; Wesloh et al., 1992), and along the length of the river (*Leadley* et al. ,1998; *Lovett-Doust* et al., 1997; *Metcalfe* et al., 2000; *Platford* et al., 1985). Although the river dominates environmental quality issues in the Western Basin of Lake Erie, there have been few studies resolving the relative importance between current and historic loadings of contaminants along the reach of the river (*Hamdy and Post,* 1985; *Farara and Burt,* 1993). This lack of information has been further compounded by a failure to integrate past monitoring and research programmes in a manner that would yield information as to the overall quality of the river itself. Previous studies of the sediments of the Detroit River have been limited to areas of known or suspected degradation (*Besser* et al., 1996; *Zarull* et al., 2001), and do not represent the overall health of the river ecosystem. The approach to just sampling "problem areas" has resulted in a historically large, but very biased, data set representing contaminated sediments in the river (*Thornley and Hamdy*, 1984). Furthermore, both water and sediment quality studies have been implemented in a sporadic manner, thus there are limited data sets available to quantify either temporal or spatial changes in the environmental quality of the river. Based on these biased historic data, the Detroit River is often referred to as one of the most polluted rivers in North America, a reputation that has hindered the economic development of the communities along the shores of the river.

The Detroit River extends 51 km from Lake St. Clair to Lake Erie, and gradually widens from 600 m to 6.1 km width. Therefore the flow in the river is quite variable, and it is estimated that the river discharges 453 x 10^6 m^3 per day to Lake Erie. The water entering the river is primarily from Lake Huron (99%), taking just over 24 h to reach the head of the river and, in less than 20 h later, the water is discharged to Lake Erie. Such high volume replacement, combined with complex flow channels makes direct water sampling a questionable approach for monitoring environmental quality. The morphological complexity of the river combined with the lack of coordinated monitoring has hindered loading estimates and has resulted in a tendency to rely on sediment quality as a means of assessing the health of the river (*Chapman*, 1996; *Power and Chapman*, 1992).

Significant clean up efforts have been made since the Upper Great Lakes Connecting Channel Study, and gradually a debate about the relative importance of in-place pollutants versus that of current discharges began to emerge. During the implementation of the Remedial Action Program for the river, it was discovered that historic data were not sufficient to quantify either loadings to the river or the overall quality of the river. Although the Detroit River can boast one of the longest records of environmental quality monitoring in the Great Lakes, simple questions as to the state of the river and whether the river was improving or not could not be answered.

In order to develop a baseline for the status of the sediment quality in the Detroit River, a river wide sediment monitoring programme was implemented in 1999. The aim of the study was to establish a benchmark for the relative importance of heavy metal pollution in the river. The sampling strategy was designed to determine the relative importance of upstream inputs associated with sediment transport and sedimentation processes in the river compared with local inputs along the Canadian and United States shorelines that would need to be addressed during the remedial action process.

Methods. Sampling design

Sampling stations were selected according to a stratified random design in order to provide a representative description of sediment quality in the river. Stations were assigned to three reaches of the river representative of the headwaters, mid section and the lower outflow area (Slide 2). Sampling stations were evenly divided between the United States ofAmerica (USA) and Canadian portions of the river. The upper reach contained 30 stations, the mid-reach contained 30 stations and 90 stations represented the lower reach. Shipping channels were de-emphasized such that two-thirds of the stations were less than the median depth of the river within each respective reach. To reduce clustering of the sampling stations randomly situated within each of the reaches, the minimum distance between stations was set at 300 m.

Samples were collected using a Ponar grab sampler. As sediment characteristics varied from site to site, the number of grabs required at each site varied. To standardize this spatial complexity, sampling continued at each site until 2 litres of sediment was collected. In the laboratory, the sediments were sieved to ensure a grain size of less than 2 mm and then frozen until submitted for analysis.

Total recoverable mercury analysis

Each 3.5 g wet sediment sample (1.7 g dry equivalent) was placed into a 125 ml Erlenmeyer flask. Fifteen ml of a 2:1 sulphuric:nitric acid (trace/ACS grade) mixture was then added, while samples were cooling in an ice bath, then 2 ml of hydrochloric acid was added. A vigorous foaming reaction occurred for some samples: these samples were cooled in an ice bath. Sample flasks were then placed in a water bath, and potassium permanganate was added in 5 ml increments. Foaming was allowed at 60 ^{o}C for 2 hours. Following the cooling of the sample flasks to 20 ^{o}C, 15 ml of 5% potassium permanganate was added in 5 ml increments. Samples were cooled at this time, to control the exothermic reaction caused by the permanganate additions. Samples were then allowed to sit for 30 minutes, after which time 5 ml of 5% potassium persulphate (ACS grade) were added and mixed. Samples were then left to stand overnight at room temperature in a fume hood. The following day, if a purple colour had not persisted, additional potassium permanganate was added until the colour persisted for 15 minutes. Then 2 ml of a 6% hydroxylamine hydrochloride-sodium chloride (ACS grade) were added, while shaking every 10 minutes. Samples were then transferred into dry, pre-weighed 125 ml low density polyethylene (LDPE) Nalgene bottles. Particles were then allowed to settle for 2 hours prior to measuring. The final solution was made up to 100 (±0.01) g by weight. Samples were shaken and allowed to settle for 2 hours prior to measurement.

Total Hg was measured using an atomic absorption spectrophotometer (AAS-300; Varian) equipped with a single element hollow cathode lamp (*Hg* in *Ar*; Varian, Australia) and a vapour generation accessory unit (VGA-76; Varian). The first sample (containing pure water) was pumped into the reaction vessel at a flow rate of 6-7 ml per minute, along with the reducing agent (25% stannous chloride in 20% hydrochloric acid) from a separate feed line at 1-1.5 ml per minute for 5 minutes in order to establish a stable baseline signal. Individual calibration standards were subsequently pumped into the reaction vessel as described above. A calibration blank and a standard check (every 9-11 samples) were interspersed among individual sample analyses. The reported detection limit for total mercury was 0.10 ug g^{-1} dry weight.

Extractable metals **(**total recoverable *Al, As, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb, V, Zn*)

Each 3.0 g wet sediment sample was placed in a 50 ml glass beaker with 5 ml 1:3 (nitric:hydro-chloric acid). This mixture was heated to 100 ^{o}C for 5 hours, and filtered with a No. 4 filter paper. The supernatant was transferred to a preweighed 125 ml LDPE bottle (Nalgene via Fisher Sci., Toronto, ON, Canada) and made up to 100 g by weight with purified water.

Sediment samples were analyzed using an Inductively Coupled Plasma Optical Emission Spectrophotometer (IRIS #701776, Thermo Jarrell Ash Corporation). Liquid samples were introduced into the instrument via a

Meinhard concentric glass nebulizer (TK-30-K2, JE Meinhard Associates Inc., California, USA) combined with a cyclonic spray chamber. The aerosol was introduced into a radial orientation argon plasma, resulting in characteristic emission lines that were simultaneously resolved using argon purged echelle optics and a thermostatted charge injection device detector.

Other analyses. Total organic carbon was measured by treating samples with HCl to dissolve carbonates and then measured using a TOC analyzer (*Hakanson*, 1992).

At each site the benthic community was sampled and organisms identified to the lowest possible taxonomic level (order, family, genus) and enumerated.

Statistical considerations

Principal component analysis (PCA) was performed on dry weight concentration data using a correlation matrix (autoscaled) using SYSTAT software. As PCA requires a complete matrix with no missing data, a random number generation process was used to replace "non-detects".

Analysis of variance (ANOVA) was used to quantify significant differences among reaches of the river overall, as well as to assess for chemical gradients along the USA and Canadian sections of the river.

As each site was determined using a Global Positioning System, a GIS format was used to present the data to represent spatial patterns. The data were geostatistically interpolated with an inverse-squared distance weighting function (ISWD) to develop contour maps (Kravchenko, 1999). ISWD assumes that values within close proximity to one and other are more alike than those further apart. When predicting values for unmeasured locations, the measured values closest to the prediction location will have more influence than those further away.

Results

Principal components analysis revealed that metals were distributed in the river ecosystem in a very complex manner because no component explained a significant proportion of the total variance. This is not unexpected in river ecosystems that represent a variety of habitats (erosional areas, depositional areas) with seasonal flow patterns, and provides a critical example of why the stratified, random approach is essential when monitoring and assessing river ecosystems. This complexity of river ecosystems is represented in the benthic studies, which revealed three dominant communities ofbenthic organisms (Slide 3). These three communities tended to reflect the hydrological patterns of the river with two depositonal communities (communities 1 and 2) being observed and one community (community 3) representative of fast flowing, high energy conditions (Slide 4). All three communities were distributed throughout the entire river range (Slide 5), although the high energy community was most frequently observed in the narrower, fast flowing upper channel.

Metal concentrations in the sediment samples often exceeded established low effect levels (LEL) or severe effect levels (SEL) as established by the Ontario Ministry of the Environment (*Persaud* et al., 1992). Sites that exceeded either SEL or LEL guidelines, for at least one metal, were found throughout the entire river (Slide 6). A quick comparison of Slide 6 reveals there is no simple correlation between the benthic community type observed and the concentration of heavy metals. This lack of a direct influence of metal toxicity on the benthic community there was no significant correlation between benthic community type and either LEL or SEL sediment guideline exceedences (Chi Square $p > 0.05$).

Analysis of variance results (Table 3), however, revealed that some metals (*Al, Cr, Hg, K, Ni* and *Zn*) were accumulating in down-river depositional sites, whereas others (*Pb, As, Fe* and *Mn*) were accumulating at sites throughout the river, with metals such as As and Pb not showing any significant spatial pattern even though they frequently exceeded sediment guidelines.

As noted by the PCA analysis, the metals in the Detroit River are being transported and distributed by an array of complex factors. Past monitoring of contaminated sediments has focused on "potential problem areas", and therefore has not been able to identify metals of priority concern nor quantify the magnitude of the toxic stress associated with contaminated sediments. Historically, Hg had been singled out in the Detroit River as being the top concern for environmental remediation and Pb and *As* were not really considered as threats to environmental health. Only by developing system level monitoring and assessment strategies can the relative importance of multiple stressors be quantified.

By using the stratified/random approach to quantifying environmental health, not only is a base line data set obtained from which temporal trends can be monitored, but there is also the ability to determine the relative importance of processes that regulate the transport and fate of the different contaminants in the system. In the case of the Detroit River, the sediment sampling regime was coupled transport in the river was being regulated by the offset of Lake Erie water levels during strong West wind events. During such events, flows in the Detroit River would increase and pockets of contaminated sediment would be washed out into the lake. Using the GIS system approach with the stratified/random sampling design, it is now possible to see which areas were scoured during such events and estimate contaminant loads to Lake Erie. Furthermore, the fact that the sediments in these areas were replaced with more contaminated sediments provided important evidence that current loadings are still very much a concern. Management of the river system needs to address these current loadings before trying to correct the issues associated with in-place pollutants.

By combining the metal sediment sampling with a study of the benthic community, it is possible to start understanding the relative importance of toxic stress associated with the sediments. When simply compared with environmental guidelines, such as Lowest and Severe Effect Levels, the conclusion would be that contaminated sediments must be having a major effect on the overall health of the Detroit River. The benthic community data, however, indicated that despite the elevated concentrations observed in the sediments, these high concentration do not necessarily mean that critical toxic exposures are occurring. In the case of the Detroit River it appears that future studies need to focus on metal bioavailability in the sediments before the real hazard of metal toxicity can be quantified.

Usually when assessing sediment health, a triad approach is recommended that requires information on chemical concentrations, benthic community and toxicity. The first two components of the triad were implemented in this study, and now using these data, future sampling for toxicity stress (life cycle studies, accumulation studies) can be performed in the more critical areas (see *Michallet-Ferrier et al.,* 2004). To perform toxicity tests at all sites would put the cost of monitoring and assessment beyond the funding capabilities of the responsible agencies. By staging the toxicity tests as part of the stratified/random monitoring design, the critical areas requiring toxicity tests can be identified for later study at considerable cost savings to monitoring agencies.

Another advantage of the stratified/random sampling design is that the same approach can be used within a strata to obtain fine scale information on sediment hazard, and again using a GIS framework it will then be possible to target specific areas for remediation. Once the remediation has been implemented, the area can be resampled to confirm that the environmental hazard has been ameliorated.

Finally, another important aspect of the stratified/random approach is that it allows identification of "outliers" that either represent a spurious data point or an area of potential high concern. For example, in the Detroit River, the distribution of Hg concentrations basically fit a normal curve, and therefore the highest concentrations observed throughout the river are not anomalous. This has important implications for remedial action plans, in that dredging a small section of the river to address the areas with higher concentrations will not have much effect on reducing the overall hazard of Hg in this ecosystem.

The lesson to be learned from the Detroit River case study is that monitoring and assessment must be designed and implemented at a systems level. Restricting sampling to possible areas of concern or chemicals of concern does not yield the information required to assess the overall hazard, and can often lead to wasted efforts on environmental management and remediation. Too often, remedial efforts have been implemented and no positive results observed after spending large sums of money. This is not to say improvements did not occur in some circumstances, but that the data base required to quantify improvement was not developed as part of the original management programme.

Conclusions

An issue by issue approach to monitoring and assessment can yield false interpretations of overall environmental quality and result in expensive and ineffective remedial actions. The complexity of aquatic ecosystems requires a sampling strategy that will yield statistically sound estimates of spatial and temporal patterns within the overall ecosystem. As noted in the Detroit River, the stratified/random approach to sampling design provides a statistically rigorous estimate of both biological and chemical properties of these systems. With the development of GIS models, this approach is strongly recommended for long, complex river systems where multiple stresses are imposed on the aquatic ecosystem.

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7. Monitoring of toxicants in the surface water

Monitoring the aquatic environment in Poland. Monitoring of the aquatic environment in Poland is the responsibility of the Polish State Monitoring System (Fig. 7.1). For implementation of this system the State Inspectorate for Environmental Protection was set up in 1991. Since then systematic, standardised measurements have been carried out for various compartments of the environment.

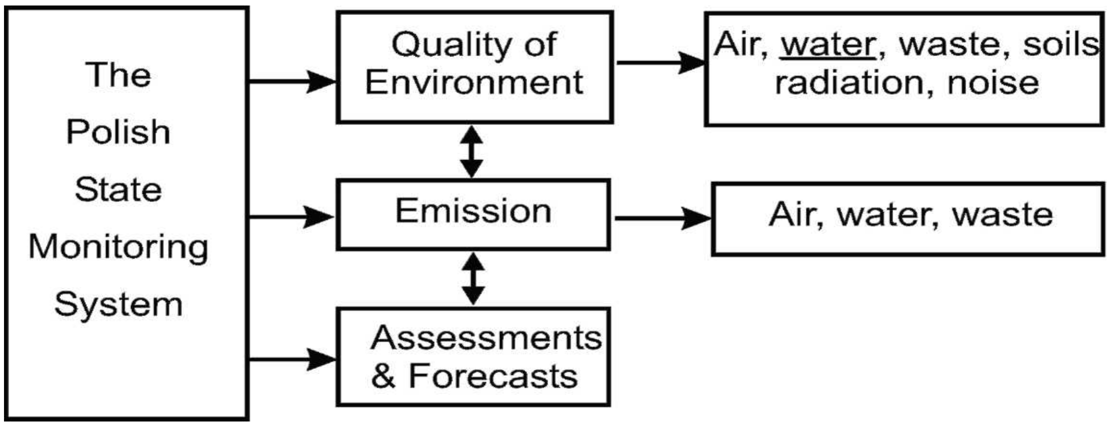

Fig. 7.1. The Polish State Monitoring System

Networks

Monitoring of the aquatic environment in Poland includes surface, ground and sea water. Surface water monitoring measurements are carried out on rivers, lakes, and reservoirs. The type and extent of monitoring activities for surface waters depend on the use of the water and the pollution risk determined by regional Boards of Water Management (monitoring of drinking water is not a responsibility of the State Monitoring System). Water quality measurements are made for the following purposes:

- diagnostic monitoring (including particularly harmful substances for the aquatic environment);
- operational monitoring;
- research monitoring;
- monitoring of water sensitive to eutrophication;
- monitoring of water quality for the protection of fish, crustaceans and molluscs.

The monitoring network for the quality of running water includes about 400 measurement and sampling sites, located uniformly on major rivers (i.e. Vistula, Oder) and their main tributaries (also at the boundary cross-sections) and on the Pomeranian rivers flowing directly into the Baltic Sea (Slide 6). The frequency of measurements depends on monitoring needs, water quality and the type of pollutants present. Sampling sites selected for diagnostic monitoring and monitoring of water sensitive to eutrophication are investigated at regular intervals, 12 times a year, and the measurements cover a range of specific indices. The scope and frequency of diagnostic monitoring can be broadened and increased for the occurrence of specific harmful substances. If adapted to local needs the programme is designated as operational monitoring and identification of accidental pollution it is called research monitoring. The frequency of water monitoring for fishery protection is limited (i.e. four times a year) if the preliminary results of investigations meet standards.

Surveys of sediment quality are conducted along major rivers (longer than 60 km) and rivers flowing in and out of the territory of Poland. The total number of sampling sites is about 300 and the frequency of sampling varies from once a year for the benchmark sites (usually in a summer) to once in three years for the others sites.

Monitoring of lakes is carried out within the regional networks for lakes larger than 100 ha, as well as for lakes that are important from the economic and natural points of view. The assumption is that each designated lake will be studied once in five years during the periods of spring water circulation and summer stratification. Surveys of lake sediment quality are conducted at the same time.

Monitoring of reservoirs of capacity exceeding 40 x 10^6 m^3 is obligatory and it is also carried out for other reservoirs recognised as important in local water registers. Sampling is done usually four times a year, usually as fishery protection monitoring.

Water quality legislation

Significant changes have occurred in Polish legislation relating to water quality and water monitoring between 2002 and 2004. This was necessary to bring legal regulations closer to the those of the European Union and the process will continue over the next two years. Despite the changes, the system is still heterogeneous and distinguishes separate standards for:

- surface and ground water quality (Min. of Environ., 2004),
- inland waters for fish habitats (Min. of Environ., 2002a),
- surface water used for drinking water supply (Min. of Environ., 2002b).

Water quality is evaluated by comparing measured concentrations with the permissible (limiting) values fixed for each quality class. Legislation distinguishes five surface water quality classes (Class I, II, III, IV, V) based on ecological classification but also considering suitability for drinking water supply (Slide 7):

Class I -very good quality (no anthropogenic impact, suitable for drinking water supply after simple treatment);

- Class II - good quality (slight anthropogenic impact, suitable for drinking water supply after standard treatment);
- Class III - satisfactory quality (moderate anthropogenic impact, suitable for drinking water supply after standard treatment);
- Class IV - unsatisfactory quality (distinct anthropogenic impact, suitable for drinking water supply after advanced treatment);

- Class V - poor quality (strong anthropogenic impact, not suitable for drinking water supply).

Toxicants in water legislation

In the surface water quality monitoring system (i.e. diagnostic monitoring) 52 parameters are monitored but with different frequency, depending on the kind of pollutant. Water quality is assessed with respect to:

- physical indicators (temperature, odour, colour, suspended solids, pH) - 12 times a year,
- oxygen indicators (O_2, BOD_5, *CODq-*, $CODm_n$, TOC) - 12 times a year,
- nutrients (nitrogen and phosphorus compounds) - 12 times a year,
- salinity indicators (conductivity, dissolved solids, sulphates, chlorides, calcium, magnesium, fluoride) - 12 times a year,
- metals and metalloids (*As, Ba, B,* Cr_{tot},$Cr+^{6}$, *Zn, Al, Cd, Mn, Cu, Ni, Pb, Hg, Se, Fe*) -4 times a year
- industrial pollution indicators (cyanide, phenols, pesticide, mineral oils, anion-active detergents, policyclic aromatic hydrocarbons) - once a year,
- hydrobiological indices (plant and animal indicators, i.e phytoplankton, periphyton and benthos) - 4 times a year,
- sanitary state (pathogenic bacteria) - 12 times a year.

Toxicants from the following groups of indicators are investigated with different frequencies: inorganic substances (cyanide, fluoride); metals and metalloids; and organic contaminants (phenols, pesticides, mineral oils, anion-active detergents, polycyclic hydrocarbons).

Toxicants from the following groups of indicators are investigated with different frequencies: inorganic substances (cyanide, fluoride); metals and metalloids; and organic contaminants (phenols, pesticides, mineral oils, anion-active detergents, polycyclic hydrocarbons).

Only 14 indicators are monitored in inland waters for fish habitat quality, with a frequency of 4 to 12 times a year. Monitoring requirements are defined separately for salmonidae and cyprinidae. Only four toxicants are investigated: phenol compounds, oil hydrocarbons, zinc and copper. Apart from the narrow range of toxicants monitored for*fishery protection* it is interesting to note how the assigned values and recommended methods of determination for phenol compounds and oil hydrocarbons are defined. In both cases the "taste" method is recommended and for oil hydrocarbons a visual method is also recommended.

Assigned permissible values seem to be very high and do not correspond to the values of ecological classification.

Toxicants in monitoring practise

According to the diagnostic monitoring schedule fluoride is one of the most frequently monitored toxicants. Listed among "salinity indicators", fluoride is measured 12 times a year in selected cross-sections (benchmark and boundary cross-sections). In these same cross-sections metals and metalloids are monitored four times a year and the rest of the toxicants only once a year. The possibility of increasing the frequency and number of sampling points in operational and research monitoring is not considered, except for incidental pollution events, due to insufficient funds.

The frequency of monitoring activities for *fishery protection* is usually reduced to four times a year, if existing results meet the standards or are described as "very low" (close to detection limit). Unfortunately "very low" concentrations are reported not only when there is actually a very low content of a particular compound but also sometimes because of a lack of adequate analytical equipment and methods.

Equipment available in the laboratories of the Voivodship Inspectorate for Environmental Protection (where almost all measurements are made) allows the determination of all required parameters but does not allow for the use of more sophisticated techniques. Such techniques are necessary especially in the case of some toxicants; e.g. differentiation between the state of oxidation for arsenic or chromium or determination of metalo-organic complexes. Also detection of non-volatile phenols (usually present in greater quantities than volatile phenols) is not performed. There is also no distinction between forms of compounds in different monitoring media (i.e. filtered water, suspended solids).

Determination of loads of pollutants is difficult because pollutant concentrations and flows are measured by two separate institutions, at different cross-sections and on different days. These problems can cause inaccuracies and divergences in the evaluation of results of monitoring and sometimes lead to incorrect conclusions. An example of such a situation is the case of monitoring in the upper Dunajec watershed (southern Poland) described below.

The upper Dunajec watershed as an example of surface water monitoring
Watershed description

The upper Dunajec watershed is located in the Carpathian Mountains (Southern Poland) (Fig. 7.2). This region is widely renowned for its unique landscape and valued natural features, with only modest industrialisation and urbanisation. Agriculture, fruit-growing, sheep-breeding and traditional crafts such a tanning and furriery, provides most employment. Natural resources are protected by the regulations for the national parks established in that area (Tatra National Park, Pieniny National Park and Gorce National Park).

The Dunajec river originates from the confluence of two main streams: the Biafy Dunajec (5.3 m^3s^{-1}) and the Czarny Dunajec (9.0 m^3s^{-1}) (Bogdanowicz *et ah*, 1995). The Kowaniec cross-section, close to the Nowy Targ village, is the first gauge with the mean annual flow of 14.3 m^3s^{-1} (Punzet, 1991). In 1997 the Czorsztyn reservoir (16.8 x 10^6 m^3) was constructed on the Dunajec river 173.3 km upstream from its confluence with the Vistula river. The mean annual flow of the Dunajec river in the present dam cross-section (Czorsztyn) was 23.8

m^{3}s^{-1}. The aim of the dam construction was attenuation of the seasonal flooding of the Dunajec river which has periodically caused damage to crops, livestock and property.

Fig. 7.2. Map of the upper Dunajec watershed *[www.zzw-niedzica.com.pl] with*

Despite the existing infrastructure (the municipal wastewater treatment plant and 11 local wastewater treatment plants) wastewater management in the upper Dunajec catchment is not adequate. Disordered wastewater discharges and surface run-off deteriorate the quality of the surface water used for drinking purposes. The greatest risk is the abundance of faecal bacteria but tannery wastewater management in that region is also unsatisfactory. A significant number of small tanneries located near the reservoir discharge pre-treated or raw wastewater into local sewers or directly into streams (Szalinska, 2001). The tanning technology is based on chromium compounds which periodically contaminate surface water, especially in the autumn-winter season when tanned leather production is at its maximum.

The monitoring network

Monitoring activities in the upper Dunajec watershed started in 1977. Since then the location of sampling sites, sampling frequency and the range ofmeasured indicators has changed repeatedly. The location of sampling sites used recently in the Czorsztyn reservoir area is presented in Fig. 7.2.

According to reports published by the Voivodship Inspectorate for Environmental Protection, water quality in the upper Dunajec watershed has improved in the last 10 years [*Szalinska*, 2001]. Water quality in sampling sites 1 and 2, which was initially described as beyond classification (only three water quality classes existed in the former decree) is now determined as "good quality" (Class II) (WIOS, 2003). This improvement has been attributed to the installation of wastewater treatment plants prior to the filling of the Czorsztyn reservoir.

The precise determination of interannual trends or fluctuations of particular elements or compound concentrations is difficult because water quality data collected by the Inspectorate are discontinuous for all cross-sections. In addition the frequency of river sampling varied from 4 to 24 times per year depending on the cross-section and the period. The location of the sampling point and timing of sampling can be relevant in the interpretation of the data as illustrated in Boxes 1 and 2. Determination of inter-annual trends or fluctuations of particular elements or compounds concentrations is based on annual means and their respective confidence intervals. If the differences in the annual means are larger than their confidence interval the trends or fluctuations can be detected and considered as statistically significant. The problem is more complicated if the number and periods of discrete sampling vary from one year to the next. In the Dunajec river watershed the number of discrete sampling events varied from $n = 4$ to $n = 24$per year. Comparison between annual means is dubious if the hypothesis that the means are derived from the same population can be rejected. Such a situation can occurs if, for example, two of the total four samples were taken by chance during two flood events when concentrations of an element were exceptionally low due to dilution effects.

Chromium as an example of toxicant monitoring

A specific toxic pollutant in the upper Dunajec watershed is chromium. About 300 small tanneries are located in this region, using chromium technology in their process. The tanneries discharge pre-treated or raw wastewater into sewers or streams. Discharged wastewater volumes and chromium concentrations are seasonal and intermittent in character with the greatest discharges occurring between November and February when tanned skin production is greatest. The main chromium inputs are located upstream to cross-section 1 (Fig. 7.2, Slide 10).

Chromium is included in the State monitoring activities and is monitored four times a year (January, April, July and October) in water samples filtered with a non-calibrated paper filter using the FAAS technique. Despite the legally required differentiation between Crtot and Cr(VI), only the total chromium is determined**.**

Conclusions

Monitoring of toxicants in Poland needs:

- detailed identification of water pollution local problems;
- adaptation of the monitoring programmes to local needs;
- evaluation of the effectiveness of management actions;
- updating the list of toxicants to be monitored (especially organic micro-pollutants).

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8. Possible catchment scale solutions to contaminanted sediments in the Elbe river Introduction

Many sources contribute to sediment contamination in a river catchment area: wet and dry fallout from air emissions, agricultural runoff from farms, solid and dissolved inputs from mines, discharges from landfills, industrial plants and sewage treatment plants and direct dumps into rivers, lakes and coastal seas. At several areas of a river basin the water current is usually low enough to allow the polluted suspended solids to sediment out. The biggest "sediment trap" is found at the end of a river where the annual deposition of fresh water sediment, for example in large ports, can be enormous (e.g. 10 x 10^6 m^3 a year at Rotterdam). Nevertheless, substantial amounts of polluted sediments are also trapped upstream in flood plains, locks, dams and smaller ports, and in channels connected to the river.

Remediation techniques for contaminated sediments generally are much more limited than for other solid waste materials, except for mine wastes. The diverse contamination sources in larger catchment areas usually produce a mixture of pollutants, which is more difficult to treat than an industrial waste. For most sediments arising from maintenance dredging, there are more arguments in favour of disposal rather than treatment. Mechanical separation of the less contaminated fractions could be a useful step prior to the final storage of the residues.

In the future, remediation methodology will be included in the context of sustainable sediment management. Recent developments in "soft" (geochemical and biological) techniques for contaminated soils and sediments, with respect to policy aspects as well as technical developments, have led to an increase in **in-situ** remediation options. "Geochemical engineering" (*Salomons and Forstner*, 1988) applies principles such as stabilization, solidification, and other forms of long-term, self-containing barriers to control the mobilization and biological availability of critical pollutants. There are three principal approaches as discussed below.

In the upper and middle course of river systems, sediments are predominantly affected by contamination sources like wastewater, mine water from flooded mines and atmospheric deposition. Measures applied to the source of contamination are particularly important and may include improvement of traditional wastewater purification, but also approaches to the in-situ treatment of highly contaminated effluents such as introducing active barriers (fly ash, red mud, tree bark, etc.) into ore mines to prevent heavy metal dispersion during flooding.

During floods, sediment-bound pollutants can undergo large-scale dispersion in floodplains, dike foreshores and polder areas. The complex mixtures of toxic compounds and the scale of the pollution often preclude technical measures like chemical extraction or solidification of contaminated soil and sediment material. Instead, alternative measures have to be taken that consider different local factors such as soil, sediment and water quality, flow velocity, and the dynamics of the water level. The measures implemented should be flexible and easy to adjust to changing conditions. Any problem solution strategy for such sites, therefore, has to consider both the chemical stabilization, e.g. by processes of (enhanced) natural attenuation, and an increase of mechanical stability (reduced erodibility).

Under anoxic, strongly reducing conditions much of the metal content in contaminated sediments is present as practically insoluble (compared to carbonates, oxides and phosphates) sulphides. Such conditions can be provided by a permanent covering of water, whereby diffusion of oxygen into the sediment is inhibited. Sub-aquatic depots are an emerging science-based technology for the final storage of sediments. The EU Landfill Directive does not refer to waste disposal below the groundwater level, but the two most promising prerequisites for a sediment depot can be found there: (i) permanent anoxic conditions to guarantee extremely low solubility of heavy metals, and (ii) base layers of compacted fine-grained sediments, which avoid advective transport of contaminants to the groundwater.

These three geochemical approaches are demonstrated here with examples actually planned for the Elbe River catchment area.

Issues in the Elbe River catchment

The substances of concern along the Elbe basin include arsenic and the heavy metals cadmium, mercury, lead and zinc, which cause parts of the Elbe to be assigned to the highest contamination class IV under the LAWA-system ("Länder Expert Group on Water"). With the exception of lead, which is of least concern for the Elbe and only shows increased levels in parts of the Mulde and the Saale, all heavy metals and organic substances like PCB, HCH, DDT and HCB show their highest concentrations between Magdeburg (sometimes even further upstream) and Hamburg.

Historical contamination in the Elbe Basin was mostly derived from former extensive mining activities and the intensive chemical industry in the former GDR and Czechoslovakia (Slide 3). The largest brown coal mining area was the "Mitteldeutsches Revier" between the two mountainous regions "Harz" and "Erzgebirge". A dense concentration of chemical industries settled in the area along the Saale and the Mulde tributaries of the Elbe, i.e. close to the energy sources. During the last 20 years, most of the industrial plants have closed down and mining activities have stopped. Production and use of fertilizers, application of manures and intensive farming also decreased in the area, while the efficiency of wastewater treatment has improved. As a consequence, heavy metal emissions from points sources (industry and wastewater treatment plants) to the Elbe catchment area decreased between 1985 and 1999 by80-90 per cent. Emissions from diffuse sources, although less extreme, also declined significantly: 40-80 per cent in urban areas and 50-80 per cent in rural areas (*Vink*, 2002).

In-situ treatment of mine effluents with reactive materials

The main cause of heavy metal contamination is the flooding of ore mines in several mining areas, such as the Freiberger Revier in the Erzgebirge. After the mine closure in 1969 it was decided to discontinue the lowering of groundwater and the mine was flooded. Mine water reached the Reiche Zeche shaft mouth, which is the main shaft in the Freiberger Revier, and since 1971 it has been discharged through the Rothschoenberger upper mine level to the River Triebisch, and via Freiberger Mulde and Mulde into the Elbe river.

The drawdown of the groundwater table led to an oxidation of ores in the mine. Consequently, the water was acidified and heavy metals were mobilized. This "first phase" led to major contamination of the river Triebisch. Current remediations techniques usually treat acidic mine waters of the "first phase" by neutralization. However, during the treatment of acidic solutions with lime under oxic conditions, iron oxide coatings are formed and slow the reaction. This problem can be partly avoided by an anoxic limestone drain.

Year by year, the heavy metal content decreased, but eventually reached a relatively high "equilibrium level" (second phase). At the same time acid production decreased and the pH increased from 3.8 to 6.2. However, because of huge mine water flows (>5 m^3/min), the heavy metal loads still present a significant environmental problem. The objective of the study was, above all, to reduce the long-term heavy metal discharges from the pits ("second phase") to the adjacent surface waters with economical in-situ measures. In the case of the Freiberger pit a method was needed which would ensure the removal of heavy metals from mine water in the day level, or better, in the shaft. Because any engineered system would have high capital and high running costs, materials were tested that could be filled into the shafts and that could remediate mine waters by sorbing or precipitating heavy metals, and by binding them on a long-term basis.

Stabilizing materials

During stabilization or demobilization, different approaches can be considered and which can also be combined:

Sufficient long-term buffering capacity, according to environmental conditions Formation of new "reservoir minerals" that can incorporate pollutants in their internal structure.

Reduction of permeability for dissolved contaminants by secondary mineral precipitation or by injecting soft gels into the pore space of the waste body.

Possible stabilization materials are bark and scale arrears (steel industry) as sorption barriers, and red mud (aluminium industry) and fly ash (brown coal combustion) as sorption and precipitation barriers (for references see *Zoumis et al*., 2000). In particular, the use of fly ash has been intensively studied. Fly ash from a lignite combustion power plant can remove *Cr* (VI), *Cu, Pb,* and *Cd* effectively from waste water. The pH is the determining factor for the leachability of heavy metals from fly ash. Some researchers have studied the adsorbent properties of red mud and have also suggested its possible use for wastewater treatment.

The materials mentioned were comparatively characterized (specific surface area determined after BET, acid neutralizing capacity, sorption capacity

(Slide 4) and the results indicated a good correlation to the sorption results (*Zoumis et al*., 2000). The low specific surface area obtained is due to the fact that the sorption sites in the micropores could not be determined with the absorption gas (*N_2*). The other values showed clearly that the two factors are responsible for heavy metal removal from solution. Both red mud and fly ash have very good sorption and acid neutralization capacities (ANC), but ANC seems to be the determining factor.

Initial conclusions suggested that an application of red mud as an active barrier material for neutral to weakly acidic mine water is possible (*Zoumis et al*., 2000). Red mud incorporates already a certain amount of heavy metals; however, experiments did not show any leaching of the studied metals at pH 6 or higher. The used and contaminated materials would then have to be treated or stored safely and whether storage in the pit would be possible should be investigated. The best approach would be to develop a material that could remain in the mine.

Floodplain soils and sediments - natural attenuation approach

Contamination of river sediments is mostly discussed in relation to harbour sediments (Rotterdam, Hamburg) and to the impact of pollution on coastal ecosystems. Remediation and storage of contaminated dredged materials is a key issue at harbour sites. However, sources of contamination such as wastewater, mine water from flooded mines and atmospheric deposition can affect the middle and upper courses of river systems to a considerable extent. Sediments are intermittently mobilized and deposited. During floods, sediment bound pollutants can undergo large-scale dispersion in flood plains, dike foreshores and polder areas (*Fattorelli et al*., 1999). Apart from seasonal flooding of polder areas and flood plains (*Miehlich*, 1987; *Schuster and Miehlic*h, 1989; *Friese et al*., 2000) there have been catastrophic floodings in the recent past due to extreme rainfall and the failure of dams (e.g. flood of the Oder in 1997: *Müller and Wessels,* 1999; *Wolska et al*., 1999). Events like the braking of tailing dams in highly contaminated areas such as mining districts, e.g. Aznalcollar, Spain 1999 (*Grimalt*, 1999) and Romania 1999/2000 (*Anon*, 2000) have caused an enormous environmental hazard. The result is long-lasting poisoning of river sediments and flooded soils.

In most cases, apart from analyses of pollution loads and some restrictions on the use of such sites, real remediation measures have not been taken. In addition to the costly removal of sediments from harbours, navigation channels, locks and river stretches, there are even bigger challenges for economy, technology and policy to provide effective precautionary measures for reducing the most serious consequences from such events.

In the so-called Chemistry Triangle of the upper Elbe River system (Slide 5), for more than 100 years, the production plant areas of the former chemical enterprises and also the groundwater in the region Bitterfeld-Wolfen were polluted by harmful substances (e.g. HCN isomers, hexachloroethane, DDT and other chloro-organic sludges, benzyl chloride residues, distillation residues, lyes and salts). Chemical analysis and multivariate statistical methods have given significant indication that inorganic processes such as magnesium production, could be the main source of dioxin pollution, especially for the sediments from Hamburg harbour, approximately 300 km downstream from the production site at Bitterfeld-Wolfen (*Gotz et al*., 1996).

Part of the land surrounding Bitterfeld-Wolfen, such as approximately 60 km^2 of the large lowland area "Spittelwasser", is affected by pollutants. Floods transform the lowland area into a large lake landscape of approximately 10-30 km^2. The sediments are deposited and rearranged along the river course with the flowing water but during higher flow rates the pollutants bound to suspended particles are flushed out of the Spittelwasser. Thus, the polluted sediments of the Spittelwasser represent a risk for the areas downstream, i.e. for the flood sediments of Mulde/Elbe rivers. For the Port of Hamburg this particularly represents an economic risk, due to the increased treatment costs, and for the North Sea it presents an ecological risk.

Interdisciplinary approach

The Spittelwasser area was used by a German group to propose a stepwise approach combining different monitoring techniques and remediation measures (Slide 6) for international comparison (*Anon*, 2000).

In the first step, the possible ecotoxicological effects of the sediments and soils over time, particularly with respect to the potential threat to groundwater, should be investigated and determined with the aid of a battery of biological tests. Appropriate biomarkers should also be determined. The battery of tests should be used on a quarterly basis, particularly after flood events, to determine the progress of natural processes over time and of the measures undertaken.

In the second step, measures such as the installation of efficient sediment traps, withdrawal of sediments rich in pollutants at specific points, as well as use of "natural attenuation" processes in the floodplain area and promotion of plant growth, may be investigated. Methods for determining their efficiency should also be developed. A statement of the final outcome for the discharge and treatment of sediments, according to the legal targets of soil protection and waste law, should also be prepared.

Natural attenuation processes

At the Spittelwasser site, the German group mainly planned investigations on the effects of plant growth and of "natural attenuation" processes of organic and inorganic contaminants in floodplain sediments and soils (*Forstner et al.*, 2000). Of the natural attenuation processes, non-destructive, "intrinsic" bonding mechanisms and their temporal development have so far been given less recognition than destructive processes such as biological degradation (*Forstner* *and Gerth*, 2001). Nevertheless, these so-called "diagenetic" effects, which apart from chemical processes involve an enhanced mechanical consolidation of soil and sediment components by compaction, loss of water and mineral precipitation in the pore space, may induce an essential reduction of the reactivity of solid matrices.

Initial findings from soil studies show that as the residence time of compounds, such as phenanthrene, in soil increases they become increasingly unavailable to micro-organisms and resistant to mild extraction (*Hatzinger and Alexander*, 1995). Part of these effects may be related to specific geosorbents such as combustion residue particulate carbon (e.g. chars, soot and ashes) that exhibit typical non-linear, hysteric sorption behaviour for organic and inorganic substances (*Luthy et al*., 1997). Preferential sorption of planar contaminants, such as chlorobenzenes and PAH on soot-like material has been found in sediments from Lake Ketelmeer, The Netherlands (*Jonker and Smedes*, 2000). Investigations by Karapanagioti and Sabatini (2000) on organic matter from samples taken from different depths and locations in an alluvial aquifer demonstrated that opaque organic matter fractions dominate the sorption process and that quantifying this fraction alone can virtually predict the sample KOC value.

For inorganic pollutants, mainly heavy metals and arsenic, the effect of ageing mainly comprises enhanced retention via processes such as sorption, precipitation, co-precipitation, occlusion, and incorporation in reservoir minerals (*Salomons*, 1980; *Forstner and Schoer*, 1984). (*Forstner and Gerth*, 2001).

Inclusion of these "aging" processes will provide a more realistic estimation of risks and may therefore constitute a significant factor for saving remediation costs. This has been exemplified by *Chen et al*. (2000) from their study on five chlorinated benzenes and four natural sediments, where the sediment quality criteria (SQC) have so far been derived from water quality data on the basis of a linear equilibrium model. Using 1,4-dichlorobenzene as an example suggests that the SQC would be nearly two orders of magnitude less strict when the process of irreversible adsorption on the resistant fraction in sediment (as determined by laboratory experiments) is taken into account.

Large-scale sediment remediation - organisational aspects

Unlike problems relating to conventional polluted sites, the problems in floodplains are primarily connected with the erosion and mobilisation of highly contaminated soil and sediment material, and the transport and deposition of contaminated solids in downstream river and harbour sediments. The handling of such problems is a complex task which cannot be tackled by science and engineering alone. It deserves thorough consideration of legal and socio-economic aspects, including public relations (*Anon*, 2000; *Forstner et al.*, 2000).

The choice of remediation measures has to be sufficiently flexible to allow adaptation to changing basic conditions. For this reason results obtained from partial measures and investigations have to be checked continuously by the project management team in order to recognize requirements for adaptation of the measures in time and to counteract any adverse developments. Thus, close coordination between the project management team, planners, technicians and local authorities is crucial. Incorporating the project into large regional remediation projects for polluted sites would be an advantage.

The necessary expertise is gathered by forming an interdisciplinary project team covering law, planning, engineering, control and public relations. The Governmental Environmental Agency Dessau-Wittenberg, as a technical monitoring authority, and the Government Board Dessau, as a supervisory authority, will be included in the preparation of the Elbe project. The Federal Environmental Agency of Saxony-Anhalt and the competent biosphere reserve administration "Central Elbe, Dessau" should also be involved in further technical discussions and evaluation.

To achieve acceptance and credibility, and to counteract potential reservations about the implementation of the measures, the general public will be informed of the changes in pollution conditions over time, including the results of the assessment of exposition and use-related hazards, and of the project targets. For this purpose, the establishment of a citizens' advice bureau would seem to be suitable and appropriate. Interested property owners will also be integrated into the planning process.

Subaqueous depots and capping of dredged material

Since the late 1970s there has been a controversy over the various containment strategies. Some experts have argued that upland containment (e.g. on heap-like deposits) could provide a more controlled management than, for example, containment in the marine environment. Others have suggested that contaminants released either gradually from an imperfect impermeable barrier (also to groundwater) or catastrophically from failure of the barrier could produce substantial damage.

In an early review of various marine disposal options, *Kester et al*. (1983) suggested that the best strategy for disposing of contaminated sediments is to isolate them in a permanently reducing environment. Disposal in capped mound deposits above the prevailing sea-floor, disposal in sub-aqueous depressions, and capping deposits in depressions are examples of such procedures for contaminated sediment (*Bokuniewicz*, 1983). In some instances it may be worthwhile to excavate a depression for the disposal site of contaminated sediment which can then be capped with clean sediment. This type of waste deposition under stable anoxic conditions, where large masses of polluted materials are covered with inert sediment, have become known as "sub-aquatic deposits". Under such conditions the solubility of metal sulphides is particularly low, compared with the respective carbonate, phosphate and oxide compounds. There are indications that the degradation of highly toxic chlorinated hydrocarbons is enhanced in a sulphidic environment relative to oxic conditions (*Kersten*, 1988).

It should be noted, that similar conditions occur in natural or man-made depressions in the course of a river and its tributaries. A typical example is the Mulde reservoir (~ 6 km^2 area, ~ 120 million m^3 water) in the Elbe River system, which was created in 1975 when a 10 km section of the river was displaced in order to get access to a lignite coal area. Approximately 50 per cent of the sediment-bound cadmium discharge of the Elbe River is retained and it has been predicted that this type of sediment trap could last for 500 to 1,000 years (*Zerling et al*., 2001).

Contaminants may disperse in two different ways: (i) by advective transport, e.g., dispersion of dissolved or particle-bound pollutants in the groundwater or surface water flow; and (ii) by diffuse transport, e.g. by transport of pollutants from a zone of high concentration into a zone of low concentration ("tea bag in warm water"). According to initial estimates on sediment depots such as the Dutch De Slufter, the major transfer route of pollutants into groundwater was expected to arise from the "squeezing out" of porewater at the bottom of the depot during compaction and consolidation. However, measurements of water and soil tension in the De Slufter depot have shown that the permeability of the base layer of already consolidated dredged material is so low, that the compaction water cannot be squeezed out. In addition, the measurements at the De Slufter depot indicate that the hydraulic resistance of the whole depot, 10 years after the start of filling, has increased to 1,000,000 days. This means that the potential porewater release and transfer into the groundwater, as derived from model calculations, has been overestimated (*Anon*, 1998; *Kamerling*, 1999; *Anon*, 2002).

Two types of subaqueous depots can be distinguished: excavation (pit) type of depot and dike (ring wall) type of depot. Both are characterized by reducing conditions, but from the Dutch experience one of the major advantages of the former type, at least for smaller depots in flat areas is that they are no longer visible after the filling period (*Anon*, 2002).

Sediment capping techniques

The cost-efficiency, togetherwiththe operational safety, ofsub-aquatic depots should be a convincing argumentforthe applicationofthese containment technologies at upstreamsites as well as in coastal areas. However, some additional precautionary measures should be considered, e.g. an armouring layer that provides erosion protection. Sediment capping is a typical exmple of a collaborative project involving strategic research, applied research and development, and technology sharing (*Azcue et al*., 1998). Major steps are:

- characterization of sediment materials (reactivity, mobility of pollutants),

- suitability of capping techniques (currents, steep gradients, groundwater seepage),
- provision of capping material (sand, granular materials, geotextile, additives; logistics; soft sediment/coarse, dense cover; impermeable materials, water flow),
- thickness of capping material,
- reactive additives,
- monitoring of the sediment/cap system and early warning systems.

Recent developments relate to reactive cap additives to reduce pollutant transfer from sediment through pore water into the open water. Cap additives have to meet a number of pre-requisites such as good retention potential, chemical and physical properties suited for an underwater application, low contamination, and low cost. Some of the properties may be altered by appropriate treatment of the material. For example, surfaces of clays and zeolites can be modified for an enhanced sorption of organic and anionic contaminants. Fine-grained materials, such as clays or red mud, which would rather form a hydraulic barrier than a reactive, permeable one, may be granulated. However, this pretreatment may raise the capital costs. Fortunately, natural microporous materials, and in particular natural zeolites, show highly favourable chemical and physical properties with respect to their application in subaqueous capping projects, along with a worldwide availability at relatively low cost.

Combination of subaqueous depot and active capping technologies

The application of a combined sub-aquatic depot and active capping technology can be considered for small yachting harbours. For the Hitzacker/Elbe harbour site, draft approval has been granted which involves the excavation of approximately 10,000 m^3 of fine grained, polluted sediments from the harbour area and their sub-aquatic deposition close to the site, in a communication channel between the Elbe River and the harbour. For this purpose, the channel is closed on both sides, using sheet piling to the Elbe river and a dam on the harbour side (Fig. 8.1).

Transfer of contaminated harbour sediment into the new containment will be performed for 1:1 sediment extraction and redeposition using pumping and conveying equipment that avoids resuspension and dispersion of contaminated particles in the surface water. Active capping of the sediment depot will include natural zeolite additives and monitoring of the site will be performed using a dialysis sampler and diffusional gradient technique probes3 (*Jacobs*, 2003).

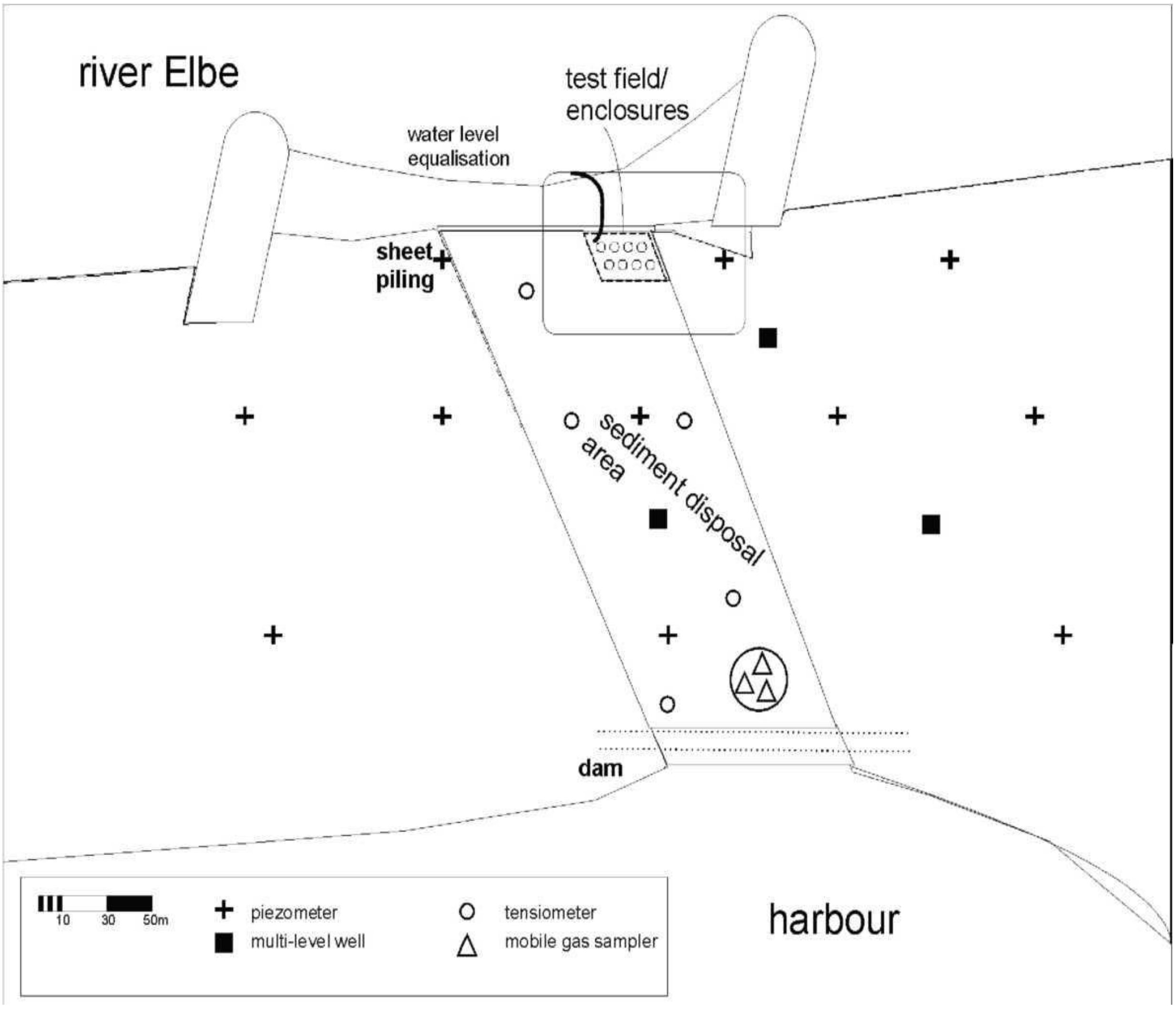

Fig.8.1. Subaqueous depot and active barrier system: Demonstration project Hitzacker/Elbe (I) (Design: *Josef Möbius*, Hamburg)

Summary

Polluted sediments from river catchments are often deposited in the estuarine zone, which for many larger rivers includes the port or harbour areas. Nevertheless, polluted sediments may also be trapped in other upstream sections, such as lochs and dams. Disposal of such contaminated sediments is difficult and the possibility of remediation techniques may be more attractive in terms of sustainable development. Geochemical approaches to remediation, including in-situ methods, are currently being explored in a number of locations, including the river Elbe in Germany.

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9. Water quality assessment based on watershed characteristics

Introduction

Water quality assessment is the evaluation of the physical, chemical, and biological condition of a water resource in relation to intended uses. It encompasses monitoring, data evaluation, reporting, and dissemination of the condition of the aquatic environment. Water quality assessments have a variety of purposes. According to the World Bank Technical Note, Water Resources and Environment, they can be used to:

Describe water quality at regional or national scales, including a determination of trends in time and space;

Determine, whether or not, water quality meets previously defined objectives for designated uses, including public health;

Deal with specific pollution management issues, including post-audit functions;

Determine investment options based on potential benefits from proposed or alternative remediation options;

Provide a comprehensive assessment of river or lake basins and aquifers, especially to determine the relative importance of point versus non-point source pollution;

Support regional or river and lake basin planning, and groundwater planning, including the development and implementation of national and regional policies.

For this case study the general strategy of watershed assessments was applied with the intention of using and developing new methods of data collection and analysis, such as

Geographic Information Systems (GIS), and interdisciplinary methods in data processing, (GIS and mathematical modelling). The assessment strategy took the following form: defining the assessment goal, data compilation, fieldwork, data analysis.

The assessment is not exhaustive; it is focussed mainly on identifying the aspects of land use that generate pollutants and the balance of pollutants at the watershed scale. The analysis methodology and the assessment results may be refined further by interdisciplinary teams. Furthermore, the assessment process is a dynamic one, because periodically new information brings additional knowledge and understanding of the processes that define water quality. The integrated assessment represents an important step in designing integrated watershed management plans, which use a good understanding of natural and social processes. The social aspects were not covered in detail in this study, but the results can constitute a valuable source ofinformation, especially for designing alternative development scenarios.

Watershed characteristics. Physical geographical features

Somesul Mic watershed is located in the North West part of Romania. The dimensions of the catchment are: length (178 km), average width (63 km), maximum height (1,640 m), average altitude (594 m), minimum altitude (232 m), average slope (approximately 8 m km^{-1}), and coefficient of sinuosity (1.68). Two sectors, which are general ecoregions of Romania, can be differentiated by their features. The map of the Somesul Mic watershed relief is shown in Slide 3.

The Mountain sector

The Mountain sector is a part of the Apuseni Mountains ecoregion and encompasses parts of Bihor, Gilau-Muntele Mare Mountains, occupying 761 km^2 (19 per cent) of the total watershed surface. The altitude exceeds 800 m. Average precipitation is relatively high (700-1,000 mm a^{-1}) and, due to reduced evapotranspiration, the region contributes to water resource generation. The area is highly tectonic and has a rich network of rivers, with two major headwater branches of the Somesul Mac: Somesul Cald and Somesul Rece. The average runoff is 750-850 mm and the slopes are steep (15-100 m km^{-1}). The river network density reaches values of 0.7-0.9 km km^{-2}. These conditions have enabled the river to be used for hydroelectricity generation and water supply through a cascade of four reservoirs built specifically for these purposes.

Generally, the geological substrate is composed of metamorphic and old magmatic rocks with younger granite intrusions. Palaeocene formations are present at the edge of the mountain area, but are not in contact with the reservoirs. The headwater area of the Somesul Cald has Mesozoic limestone with extended karst. As a result of the geological substrate, the underground water resources were classified in two main domains:

- Eruptive rocks and crystalline schist, which are well-defined in the Somesul Rece and Somesului Cald regions, are characterised by their reduced extent in the phreatic layers due tocompaction of the rocks. Groundwater is present only temporarily in rubble and slide rocks,deluvial deposits and alluvial cones or alluvial deposits, and in other areas it only circulates through the cracks of the rocks. Ground water appears as linear seepage at the base of the soil crust and as springs, depending on the precipitation regime. This water has reduced mineralisation.

- Limestone and limestone deposits in the area of Somesul Rece headwaters. Groundwater circulates in a well-defined underground karstic network. The chemical composition is strongly in-fluenced by the presence of limestone.

The vertical distribution of vegetation in the mountain region can be classified as follows:

Alpine pastures and shrubs on approximately 467 km (12.38 per cent of the total catchment surface) consisting of alpine herbs and dwarf shrubs (Festuca rubra, Agrostis tenuis, Nardus stricta, Calamagrostis sp.). This vegetation is subjected to increasing pressure from intense grazing and sometimes the resulting vegetation has a low economic value.

Sub alpine meadows occupying an area of 155 km (Cinosorus cristatus, Arrhenatherium elatius, Festuca pratensis and shrubs of Pinus montana, Alnus viridis)/

Forests: Up to 800 m altitude the forests are mainly of oak and beech and between 800 and 1,300 m they are mostly beech. Over 1,300 m conifer forests occur. There is a general pattern of forest decline, with implications for water quantity and quality.

Agricultural land comprising pastures and arable land located below the forest belt, but sometimes rising to 900 m.

Riparian vegetation, which is generally abundant.

The biocoenosis of the mountain region has been influenced by human presence. Generally the distribution of fauna is associated with the altitude but the presence of people has encouraged opportunist species. Notably, the presence of large mammals such as wolf, stag, wild boar and roebuck, as well as small mammals such as rabbits, etc. The construction ofreser-voirs has promoted the presence of waterfowl.

Part of the mountain sector of Somesul Mic watershed is included in the Apuseni Mountains National Park and the rest of the area needs extended protection measures to secure drinking water resources for approximately 500,000 inhabitants.

The Hilly sector

The Hilly sector consists of the landforms of 400/600 m altitude, particularly the major formations of the Clujului and Dejului Hills and

Transylvania Plain (Campia Transilvaniei), and part of the major Transylvanian Tableland ecological region. Its total surface is 2,925 km^2 (77.4 per cent of the watershed) and comprises the middle and lower course of the Somesul Mic River. Precipitation patterns show a decrease from west to east from 700 mm to barely 600 mm a^{-1}, with increased evapotranspiration. The main source of water is the river network, which is characterized by low runoff (50-100 mm a^{-1}); some of the rivers have an intermittent flow. There are also some artificial ponds and to a lesser extent some natural lakes. Clujului and Dejului Hills and Campia Transilvaniei have a tabular or monocline relief structure that determines the flow directions of the Somesul Mic tributaries. With a few exceptions (Nadas-Capus area where Eocene limestone is present) the geology consists of Miocene deposits: sandstone, shale, salt and dacit tuffs with superficial deposits of Sarmatian rubble.

Groundwater in the hilly region is characterised by low discharge, due to the climatic conditions, and the chemical properties are determined by the presence of sedimentary rocks. The Tortoniene deposits enclose horizons of sulphitic, calcium and salt waters due to the presence of salt deposits sometimes as outcrops. Salt springs appear downstream of Cluj-Napoca in the localities of Dezmir, Cojocna, Gadalin, Sic, Gherla, and Ocna Dej, etc.

The flora and fauna are greatly affected by human activities (agriculture and animal breeding). The natural biocoenoses that remain consist of xerophytic forest and protected areas of grasses. Aquatic biocoenoses are present in the vicinity of the lakes (natural or man-made) and the Fizes valley is being subjected to a wetland restoration programme. Riparian vegetation along the Somesul Mic valley has been affected by river regularisation activities. Somes valley represents an important river corridor extending for 80 km, and 1-3 km wide in the river flat and 4-5 km wide at the upper terrace level. Two narrow stretches are present in the areas of Cluj and Gherla cities. The river is bounded by a flood plain along almost all the course of the river Somes in this region, but the terrain is more hilly on the main tributaries, accounting for 5 per cent of the total sector surface.

Climate, hydrology and soil cover

The climate of the watershed is temperate continental, being influenced by the presence of the mountains. Meteorological stations are located at Vladeasa in the Mountain region and in the hilly region, Cluj in the central part of the catchment and Dej in the eastern part.

The Somesul Mic River gathers its water from tributaries in the Apuseni Mountains, as well as from other tributaries with reduced runoff from the hilly region - although two thirds of the catchment surface covers this area. There are many lakes in the Somesul Mic watershed, including reservoirs in the upper catchment, ponds in the lower basin, anthroposaline lakes (in the area of former salt exploitation) and natural lakes in the lower basin.

Population and land use

According to the census of 2003 the total population of the catchment was 490,187 inhabitants. The density of population is greatest in the urban agglomerations and river corridor Fig. 9. 1Industrial and agricultural activities are also pollution sources (point and diffuse).

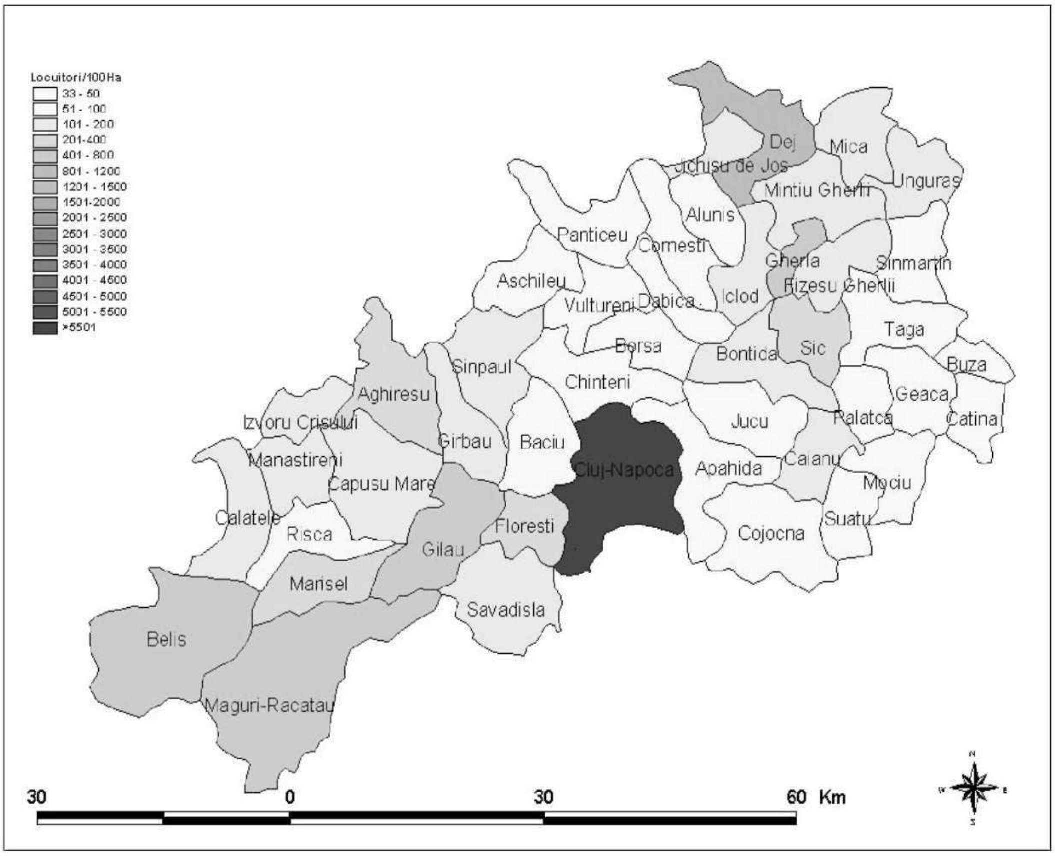

Fig. 9.1. Population density by administrative district in the Somesul Mic watershed

Industrial activities are concentrated in the urban areas of Cluj-Napoca and Gherla and in the mining areas of Capus and Aghiresu. Agricultural activities vary from animal breeding in the mountain and hilly area to intensive rearing of cattle pigs and hens in the corridor areas. The population distribution largely reflects the economic activities, which in turn govern the pollutant potential. Land use has a major impact on water resources, disturbing the natural water balance (especially evapotranspiration and infiltration) and affecting water quality. Among the stress factors which contribute to the alteration of ecosystem functioning are:

Massive deforestation in the mountain area Overgrazing Irrational agricultural practices leading to increased erosion (especially in the hilly region) Destruction of wetlands, etc.

An important aspect to achieving integrated watershed management is rational exploitation of the land. Analysis of land use for the watersheds of the tributaries using the CORINE database (CORINE, 1992) shows that since 1967 the forested areas decreased simultaneously with the increase in agricultural uses and pastures. These modifications indicate a decrease in the self-regulation capacity of the entire watershed.

Pollution sources

The main point pollution sources are industrial activities and wastewater treatment plants. Industrial pollution sources are distributed mainly in the lower part of the catchment and characterized by an almost constant discharge during the whole year.

Diffuse water pollution sources have only received the attention of specialists since the 1980s and have been analysed in conjunction with potential sources or transport vectors. Generally, diffuse pollution sources are classified as: local sources (generated by the activities in the watershed and correlated with hydrological processes) and regional and transboundary sources (correlated with the deposition ofatmospheric pollutants). Diffuse pollution sources show great variability in time, being directly influenced by the precipitation regime and the timing of agricultural activities.

Computing the contribution of pollution sources

Mathematical modelling provides a complementary approach in the calculation ofthe contributions of diffuse sources. The most appropriate model, with respect to complexity and data availability, was selected. This was the Generalized Watershed Loading Functions model (GWLF) (*Haith et al.*, 1992). The soil erosion risk assessment was prepared using the Universal Soil Loss Equation model (USLE) (*Wischmeier and Smith*, 1978), applied to a GIS base. The data generated by the model were also used in the diffuse pollution calculation.

The complexity of the loading functions model falls between that of detailed, process-based simulation models and simple export coefficient models which do not represent temporal variability.

The GWLF model provides a mechanistic, but simplified simulation of precipitation-driven runoff and sediment delivery, but is intended to be applicable without calibration. Solids load, runoff, and groundwater seepage can then be used to estimate particulate and dis-solved-phase pollutant delivery to a stream, based on pollutant concentrations in soil, runoff, and groundwater.

The GWLF model simulates runoff and stream flow by a water-balance method, based on measurements of daily precipitation and average temperature. Precipitation is partitioned into direct runoff and infiltration using a form of the Natural Resources Conservation Service's (NRCS) Curve i.e. Number method (SCS, 1986). The Curve Number determines the amount of precipitation that runs off directly, adjusted for antecedent soil moisture based on total precipitation in the preceding five days. A separate Curve Number is specified for each land use by hydrologic soil grouping. Infiltrated water is first assigned to unsaturated zone storage where it may be lost through evapotranspiration. When storage in the unsaturated zone exceeds soil water capacity, the excess percolates to the shallow saturated zone. This zone is treated as a linear reservoir that discharges to the stream or loses moisture to deep seepage, at a rate described by the product of the zone's moisture storage and a constant rate coefficient. Flow in streams may derive from surface runoff during precipitation events or from groundwater pathways. The amount of water available to the shallow groundwater zone is strongly affected by evapotranspiration, which GWLF estimates from available moisture in the unsaturated zone, potential evapotranspiration, and a cover coefficient. Potential evapotranspiration is estimated from a relationship to mean daily temperature and the number of daylight hours. The user of the GWLF model must divide land uses into "rural" and "urban" categories, which determines how the model calculates loading of sediment and nutrients. For the purposes of modelling, "rural" land uses are those with predominantly permeable surfaces, while "urban" land uses are those with predominantly impervious surfaces. Monthly sediment delivery from each "rural" land use is computed from erosion and the transport capacity of runoff, whereas total erosion is based on the universal soil loss equation (USLE) (Wischmeier and Smith, 1978), with a modified rainfall erosivity coefficient that accounts for the precipitation energy available to detach soil particles (*Haith and Merrill*, 1987). Erosion can occur when there is precipitation, but not necessarily any surface runoff to the stream. Delivery of sediment to a stream however, depends on the surface runoff volume. Sediment available for delivery is accumulated over a year, although excess sediment supply is not assumed to carry over from one year to the next. Nutrient loads from rural land uses may be dissolved (in runoff) or solid-phase (attached sediment loading as calculated by the USLE).

The application of the model on the study area

Input data were obtained from the GIS analyses ofthe previous results ofthis study. Because of the diversity of the geographic factors (climate, soil, land uses, etc.) it was necessary to divide the watershed into five sub-watersheds, i.e. into units compatible with the model procedure. Determination of homogenous regions from the point of view of modelling characteristics is important because it can influence the model outcome. For the purpose of integrated watershed management, cumulative criteria must be identified that can be used in the watershed regionalisation, having taken into account all the aspects related to the forming, use and management of water resources. For the Somesul Mic watershed the following major sub-regions can be identified (Slide 17):

- Somesul Mic upper basin;
- Nadasul basin;
- Feleacului Hills;
- Somesul Mic basin - downstream Cluj-Napoca;
- Somesul Mic basin - downstream Gherla.

The main criteria determining the major sub regions were the homogeneity (relief, precipitation, pollution sources, water quality objectives, possible risks, etc.) and the compatibility with the model requirements and data availability.

The land use and soil characteristics were used to calculate the nutrient quantities introduced into the water bodies of the watershed by leaching and sediment generation and transport. Values for nutrient inputs from dissolution processes were approximated using the specific values for each land use given by the GWLF. The number ofinhabitants and types ofseptic systems were obtained from the environmental studies and population census. Application of GWLF requires information on land use, land cover, soil, and parameters that govern runoff, erosion, and nutrient load generation. In order to determine loads (which are required to calibrate a water quality model), flow measurements (or estimates) are needed at the time that water quality samples are collected. Stream flow data was obtained from existing Romanian Water Authority gauging stations. Nutrient loading caused by sediments is approximated as a fraction of the total eroded soil quantity and by taking into account the nutrient content of the superficial soil layer (0-30 cm). For Somesul Mic watershed the average content of the superficial soil layer was estimated to be: Nitrogen: 3,000 mgN/kg soil Phosphorus: 450 mgP/kg soil.

Watershed model results

By studying the results of the model it became obvious that diffuse pollution sources were more important in the months with significant precipitation. Also the contribution of agricultural sources was highlighted by the increased contribution of arable land to the total balance of nutrients and sediments.

The modelled GWLF results are presented in Slide 24 and compared with monitoring data from the "Apele Romane" National Company (CompaniaNationala, 1997-2001). A good agreement can be seen, especially for nitrogen. The higher phosphorus value obtained by the model does not take into account watercourse self-purification processes.

This preliminary study has shown good agreement between measured and modelled data, but these results still need to be refined before substantial policy decisions can be based on the model. Further measurements of nutrient loads into the watershed are required to verify the concentration and loading assumptions. Pilot studies are also required on small sub-watersheds in order to improve the calibration of the model. The study area can be extended easily to watersheds with similar characteristics.

Conclusions

The integrated watershed assessment highlighted the interaction of the anthropogenic and natural factors in defining the state and quality of the water resources.

The analysis revealed water quality problems generated by the cumulative impact of anthropogenic activities.

The self-purification capacity of the Somesul Mic hydro ecosystem was exceeded in the reach down stream Cluj-Napoca.

Diffuse pollution sources account for at least 50 per cent of the total nutrient loads to surface waters, but atmospheric contributions should also be considered.

Mathematical modelling proved to be a useful tool in support of the watershed assessment providing supplementary data and facilitating the analysis of various development scenarios within the frame of an integrated watershed management plan.

Design and implementation of measures to reduce diffuse pollution sources has become a major part of watershed plans.

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Quastions for self control

Water as a Resource. Characteristics of Aquatic Systems

The Hydrological Cycle: Describe the characteristics of watersheds, ground-sheds and air-sheds. Show their interconnectedness and how the water flows between them and the marine system.

1. What is the important information on global distribution of water and relate to the fraction available for human use?
2. Why the uneven distribution and stress the areas of water shortage and the implications take place?
3. Describe the variety of water use and how this is impacted by global changes and changing demographics.
4. Limitations for Use: Discuss limitations in use from the point of view of quantity and quality. Why degradation of water both fresh and marine take place, and how this impacts on use and availability of water? In particular, stress ecological change and human health.
5. Economic Value of Water: Describe the necessity of water to mankind, its essential role in providing the infrastructure of society and how it relates to population growth and determines the limits to such growth. Discribe the basics of water pricing and economic loss from water degradation.

Types of Water Bodies.

Describe the general characteristics and phys-ical/geomorphological classification of:

6. Freshwaters - rivers, lakes and reservoirs
7. Estuaries and deltas
8. Marine coast zones
9. Ground water aquifers
10. Provide a clear indication of the rate of water flow or residence time (renewal rate) in these systems.

Physical Aspects

For rivers:

11. Describe discharge and its measurement relative to basin dimensions and show how discharge relates to basin dimensions.
12. Define stream order and describe morphological characteristics of different river sections and types.

For lakes and reservoirs:

13. Describe circulation, stratification and residence time mixing and show how this process can be used for classification and how lakes relate to both climate and altitude. Point out the differences in the dynamics of deep and shallow lakes and the specific features of reservoirs.

For marine systems:

14. Describe tidal influences and river/sea interaction in estuaries, deltas and the open sea.

Present an overview of particulate matter, its origins, characteristics, transport mechanisms and sedimentation.

Chemical and Biological Aspects: Describe:

15. Trophic levels and community structure both benthic and pelagic.

16. Zonation of organisms in rivers, lakes and seas.

17. Describe primary production and seasonal cycling.

18. Describe predation and food web dynamics and how these may influence both water chemistry and transparency.

19. Describe bio-manipulation as a water management tool.

20. Ensure that biological changes with stratification are well described.

21. Describe how biological production and diversity relate to physical and chemical characteristics and how aquatic systems function as integrated ecosystems.

22. Describe redox processes related to carbon/oxygen cycle resulting from production-respiration cycles and physical mixing.

Seasonality of phytoplankton

23. Conditions - temperature, light, turbulence, stability of stratification

24. Primary production

25. Climatic effects - ice cover, circulation, stratification temperature, wind, precipitation

26. A typical lake example on seasonality of phytoplankton

27. A typical coastal area example on seasonality of phytoplankton

Species Invasions and Biodiversity

Define the problem of species invasions in the context of biodiversity and changes in ecosystems due to accidental introduction of exotic or foreign species into stable aquatic systems. This section is best taught by the use of well-known examples though it must be stressed that these ecosystems are continuously changing and being threatened. Examples to be used are numerous but should include the Black Sea and the Great Lakes.

28.Causes should be described in some detail focussing on ballast water and canal construction. Control of species movements should be discussed.

Continental Fluxes to the Coastal Oceans

Weathering

29.Briefly review physical and chemical weathering and the generation of solutes and particles.

30. Describe iscuss geological salinization of the oceans.

31.Describe the carrying capacity of water and sediment, and how pollutants are delivered to the oceans.

32.Present examples of fluxes from individual basins, e.g. Danube, and show impact on receiving waters as for example, the Black Sea.

33.Show how understanding the origin, sources transport and impact are the basis for the long-term management and conservation of the world's oceans.

What is Eutrophication?

34.Definition of Eutrophication: natural and cultural Eutrophication

35.History and background

36.Concept of limiting nutrients

37.Nutrient enrichment and productivity

38.Trophic state and classification system

39.Comparison of eutrophication in freshwater including tropical lakes and marine waters

40.Impacts of Eutrophication: taste and odour, oxygen levels, changes in fish communities, fish mortalities, algal blooms, toxic blooms, increased need for water treatment and related economic losses

Recommended reading

Berner, E.K. and Berner, R.A. 1996 Global environment. Water, Air and Geochemical Cycles.

Prentice Hall, New Jersey, 376pp.

Butcher, S.S., Charlson, R.J., Orians, G.H. and Wolfe, G.V. (Eds) 1992 Global Biogeochemical Cycles. Academic Press, London, 377pp.

«WATER MANAGEMENT. ESTABLISHING OF COASTAL ZONE AND WETLAND PROTECTED AREAS»

10.Legal and regulatory instruments for the control of water pollution

In the course of economic, industrial and social development countries have developed a variety of legal and regulatory instruments for the control of water pollution. Early industrialization has been accompanied by, often dramatic, deterioration in the quality of water resources that are vital for drinking water supply, manufacturing and agriculture. As a consequence there has been a promulgation of laws and regulations to reduce or prevent such deterioration. Although effective instruments for the control of municipal or industrial wastewaters exist, non-point source pollution such as surface runoff from urban or agricultural areas presents scientific and technical problems for the regulator.

There are a large number of alternative approaches to pollution control through regulation available to the responsible authorities, and it is for the policy makers to evaluate the specific national or local situation when choosing the most suitable legislative framework (Helmer and Hespanhol, 1997). Regulatory instruments should, in principle, be applicable to all natural waters, including inland surface freshwaters, ground waters, estuaries and coastal waters. Experience has shown that no single system of control meets all requirements and all situations within a given area or country. In practice, it is essential to use a combination of the available mechanisms, including legal, regulatory and financial regimes to achieve the desired quality improvement in national water resources. In addition, consideration needs to be given to the amount of investment and operational costs required to meet any new regulations that are imposed by the legislator. Lack of funding, and often a shortage of specialized scientists and of skilled technical and administrative staff, hampers the enforcement of laws and regulations, thus limiting their practical usefulness and environmental effectiveness.

This chapter attempts to describe and evaluate the various legal approaches and regulatory instruments according to guiding policy principles, and with regard to their application to different water pollution problems and situations. Finally, some examples of existing international legislation and standard setting approaches are presented.

Guiding policy principles and implementation strategies

Development of a legal framework for water quality protection inevitably has to be preceded by a policy making process that provides a coherent and acceptable basis within the overall legal system of the country or region. Guiding principles are needed to put the political intentions into a more practical context that supports the overall environmental policy objectives of the country (Larsen *et al.*, 1997). For the sound management of water resources and their protection from pollution, the following principles have been widely recognised:

Prevention of pollution rather than treating symptoms.

Use of the precautionary principle.

The polluter pays principle.

Promulgation of realistic standards and regulations.

Balancing regulatory instruments with economic incentives/disincentives.

Water pollution control measures applied at the lowest appropriate level.

Cross-sectoral integration of water-related regulations.

Participation of all relevant stakeholders.

The formulation and application of a successful water pollution control strategy requires three key elements: (i) an enabling environment providing a framework of national policies, legislation and regulations, (i) an institutional framework that allows for effective interaction of services involved, vertically as well as horizontally, and (iii) enforcement capabilities with available resources and the necessary social and economic support structure. Scientists of different disciplines are called upon to provide the necessary substantive basis, to stimulate the strategic debate, and to provide support to the national and local enforcement services and authorities.

Regulatory instruments

There are alternative principle mechanisms in use in different countries that have been applied to point sources and non-point sources of pollution, to surface or ground waters, and also in various combinations (Chave, 1997). The most common starting point is to produce an inventory for pollution control. These may take the form of substances inventories, environmental risk assessments, and pollution discharge inventories. Once sources of pollution have been identified and quantified, standards have to be derived which may take different forms, such as water quality objectives (WQOs), environmental quality standards (EQSs), or limit value or uniform emission standards.

Regulation of point sources

The common characteristics of point source discharges are that they are identifiable and that they can be monitored. Providing suitable legislation has been put in place, they can usually be controlled effectively. Most developed countries have had legislative provisions in place for many years that enable the authorizing or licensing of potentially polluting operations. The key instruments are:

Discharge permits,

End-of-pipe controls,

Toxicity-based controls,

Process-based controls,

Waste minimization and cleaner technology,
Voluntary schemes,
Enforcement mechanisms, and Compliance assessment.

Non-point source pollution

It is more difficult to control non-point source pollution, such as from diffuse pollution arising from agricultural activities, such as spreading fertilizer, or runoff from urban areas. These pollutants are difficult to identify and even more difficult to quantify. Key approaches include:

Catchment inventories,
Nutrient control from agricultural sources,
Runoff from roads,
Separate urban drainage systems,
Catchment management planning, and
Laws and regulations on land use.

Groundwater protection

The need to prevent groundwater pollution is particularly important where aquifers are used for potable water supply. Between 8 per cent and 99 per cent of potable water is taken from groundwater in different European countries (EU, 2007). Regulations may be aimed at general protection of underground water resources or the protection of zones from which drinking water is abstracted (Helmer *et al.*, 1997). Typical problems are: controlling abstraction rates, disturbance of groundwater flow, and impacts from waste disposal on land, including leachates from landfill. Urban and industrial developments frequently affect groundwater quality through, for example, leaking sewers and inadequate industrial waste storage facilities. Some land uses present a high pollution risk for groundwaters, especially in vulnerable geological conditions, such as karst areas. Regulatory instruments that protect groundwater include licensing of abstraction rates, planning regulations for the location of septic tanks and farmyard waste storage facilities, and guidelines for the design and maintenance of landfills.

The Danube River Protection Convention

The Convention on Co-operation for the Protection and Sustainable Use of the River Danube (known as the Danube River Protection Convention) is the legal instrument for co-operation and transboundary water management in the Danube River Basin. It was signed on 29 June 1994 in Sofia, Bulgaria by eleven of the Danube Riparian States (Austria, Bulgaria, Croatia, the Czech Republic, Germany, Hungary, Moldova, Romania, Slovakia, Slovenia and Ukraine) and by the European Community. It came into force in October 1998.

The main objective of the Convention is to ensure sustainable and equitable management and use of surface waters and groundwater within the Danube River Basin. This involves:

the conservation, improvement and rational use of surface waters and groundwater preventative measures to control hazards originating from accidents involving floods, ice or hazardous substances, and measures to reduce the pollution loads entering the Black Sea from sources in the Danube River Basin.

The signatories have agreed to co-operate on fundamental water management issues by taking "all appropriate legal, administrative and technical measures to at least maintain and where possible improve the current water quality and environmental conditions of the Danube river and of the waters in its catchment area, and to prevent and reduce as far as possible adverse impacts and changes occurring or likely to be caused." Source: ICPDR, 2010.

Transboundary pollution

Many large rivers flow through, or border, more than one country, such as the Rhine, Danube, Oder and Elbe. Estuaries or enclosed seas also have coastal zones that span several countries, such as the Baltic and the Black Sea. More than 100 Conventions, Treaties and other arrangements have been concluded amongst European countries to strengthen co-operation on transboundary waters at bilateral, multilateral and pan-European levels. Their aim is to prevent the deterioration of water quality in transboundary waters. Examples are:

International Commission for the Protection of the Rhine against Pollution (ICPR, 2010).

Convention on cooperation for the protection and sustainable use of the Danube River (Danube River Protection Convention) (see Box 1).

Convention on the Protection and Use of Transboundary Watercourses and International Waters (UNECE, 1992).

Conclusions

There are a large number of potential legal and regulatory instruments which are available for pollution prevention and control, and examples can be found in operation in many industrialized countries. Developing countries need to examine these in the context of their capability to estab-lishand to maintainthe necessary infrastructures. Finance is oftenone ofthe mainlimitations, in both the public and private sector. The availability of natural science experts and of legal experts trained in water quality issues, as well as in the judicial, managerial and organizational aspects of comprehensive water pollution control, is equally critical. The design of enabling legislation, regulatory mechanisms, and effective enforcement structures needs to be based realistically on the material and human resources made available by the political decision makers.

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ICPR 010 International Commission for the Protection of the Rhine.

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11. Socio-economic development and the need for sustainable management and protection of water resources

Freshwater is essential for all life on Earth because most living cells, including cells in the human body, are comprised largely of water as the basic solvent. It is estimated that, in order to prevent becoming dehydrated, people need to take into their bodies between 2 and 5.5 litres of water a day depending on gender, activity and location (Howard and Bartram, 2003). In addition to their intake of water in food and drinks, people also need water for many other activities, such as basic hygiene, cooking, growing crops, sustaining livestock and industrial processes. In practice, therefore, the daily use of water is much greater, varying from about 50 litres per capita per day in the developing world (whatever can be carried by members of the family) to about 300 litres per capita per day in wealthy countries where most households have piped domestic supplies and high water-consumption appliances such as washing machines, dishwashers and power-aided showers. Although water can be considered a renewable resource through the hy-drological cycle, there are many regions in the world where there is insufficient freshwater for human needs either because there is not enough water available, or because the quality of the water is inadequate. Careful management of freshwater resources is becoming increasingly important and new and innovative approaches are starting to emerge that combine the need for water as part of an ecosystem (including all the benefits that ecosystems provide) with the needs of human settlements.

The global freshwater resource

Freshwater resources, whether surface or ground waters, depend on rainfall to replenish them. Rainfall is not evenly distributed around the world, leading to regions where freshwater is scarce and other regions where it is plentiful. Annual rainfall and run-off therefore provide the basic freshwater resource, but human needs for water vary depending on the level ofeconomic development and the population density (Revenga, 2000). In order to determine how much freshwater is available for human uses, water availability is usually expressed in terms of quantity per capita per year. Where population densities are high and rainfall is low, water resources are scarce on a per capita basis, such as in North Africa, the Middle East and parts of Europe. Each country receives rainfall that contributes to its own internal freshwater resources and many countries receive additional inflows (rivers and groundwaters) from neighbouring countries that contribute to their total available renewable resources.

On a global scale, agriculture (mainly through irrigation) accounts for the largest proportion of water abstraction and use, especially in the less developed regions, but in Europe agriculture accounts for only 24 per cent of abstraction, with energy production (largely in the form of cooling water) accounting for 44 per cent (Collins *et al*, 2009). Experts believe that if a country or region exploits more than 20 per cent of its total renewable freshwater resources it is becoming water stressed. About 1.7 billion people live in countries that are water stressed and this figure is projected to rise to 5 billion if there are no major changes to global water management strategies (PEP, 2006). Abstracting more than 40 per cent of available water suggests severe stress and serious competition for water resources (Raskin *et al.*, 1997), including competition with the needs of the freshwater ecosystems themselves (Smakhtin *et al.*, 2003). Hungary, Belgium and the Netherlands are using more than 40 per cent of their internal water resources. The Netherlands is using more than 100 per cent of its internal resources and is therefore dependent on inflows from other countries (OECD, 2004). According to the European Commission, at least 11 per cent of the population of Europe has been affected by water scarcity (EC 2007 cited in Collins etal., 2009). Without adequate freshwater resources, socio-economic development may be affected through a lack of essential services, such as water supplies and sanitation, inadequate food and nutrition, and ultimately poor health. When countries are dependent on water resources also shared with other countries there is a particular need for careful management to ensure adequate quantity and quality of the resource for all users.

Freshwater for social and economic development

Poor water quality can lead to the spread of human disease through transmission of waterborne pathogens and toxins. Many of the pathogens in freshwaters come from contamination of waterbodies with animal or human faucal material. Many of the pathogens that are ingested with water result in diarrhoeal diseases such as cholera and cryptosporidiosis. About 1.8 million people die every year from diarrhoeal disease and most of these are children under the age of five years in developing countries (WHO, 2004). Inadequate sanitation facilities and unsafe water supplies are responsible for 88 per cent of all diarrhoeal disease and in 2002 there were 2.6 billion people in the world without improved sanitation facilities and 1.2 billion without an improved water supply (WHO, 2004). Scarce or poorly managed water resources therefore lead to poor health, which in turn leads to restricted ability to work and resultant poverty.

Sustainable water resources management

With so many demands on freshwater resources it is essential that they are managed equitably and in a sustainable way so that "downstream" and future users are not disadvantaged by insufficient availability or inadequate quality, both of which are essential to sustain existing economies and to facilitate socio-economic development. The quality of the water required is defined by the type of use and, where the quality is not adequate, expensive treatment may be necessary. Drinking water has the highest quality criteria in order to protect human health (*WHO, 2008*) but in many wealthy countries much of the water that is treated to potable standard is used for other domestic, industrial and agricultural activities, and much of that is wasted! In order to reduce wastage of water, it needs to be "valued" by users and by society (see below). Water use inevitably leads to the production of wastewater, which is usually discharged back to freshwater bodies, or to the sea, and although it is returned to the hydrological cycle it is often of a much poorer quality (e.g. it contains organic matter or chemicals and other contaminants) than the natural source water. On a national or even a global scale this is often leading to an overall deterioration in the quality of the available resource (UNEP, 2007). Where water resources are shared, it can also lead to problems where the activities of one water user can lead to the degradation of another users' water supply, as in the case of transboundary rivers, lakes and groundwaters.

Increasing water demand, and the resultant discharge of wastewater, is leading to a gradual decline in surface water quality in most world regions. Not only does this deterioration in quality affect whether it can be used by other people, but it also damages the aquatic ecosystem which provides many other benefits to communities, either directly (such as fish for food) or indirectly (such as providing habitats for other organisms). Until recent decades, the indirect benefits of aquatic ecosystems have been ignored and the traditional approach to managing water resources has tended to simplify the hydrological cycle and consider the water body in isolation of its surrounding catchment (*Jewitt, 2002*). Hence, in the past, whenever more water was needed, rivers would be diverted or impounded (and groundwaters would be extracted) without much consideration of the downstream or long-term consequences. In trying to meet the needs of their growing population centres, many developed countries have considered any residual flow in the water system that could not be abstracted to be "wasted" water. When water resources were inadequate, such as during droughts or as a result of over-abstraction, the solution was to seek another source of water, build a dam, or divert water from elsewhere. Some schemes have transported water thousands of kilometres, such as the Great Man-Made River Project in Libya.

When managing abstraction and wastewater discharge, a typical approach would be to abstract the maximum amount of water and to leave only the minimum that would ensure sufficient flow to carry away any wastewater discharges. Attempts to control impacts on water quality were often limited to ensuring that the quality was adequate to sustain only one or two key aquatic species, without consideration of the impact on other species that could play important roles in the ecosystem. Full and effective water resource management should ensure that the quantity and quality of water in the resource sustains the aquatic ecosystem as well as human needs (*Baron* et al., *2002*). This can be achieved through various tools, such as water quality or aquatic ecosystem objectives, source directed controls such as abstraction and discharge licences, demand management initiatives and monitoring of quantity and quality.

Integrated Water Resources Management

Effective management of water resources should take an integrated approach, involving all stakeholders. Competition between water users should be reduced by co-ordinated planning and management of the water resources within a catchment (i.e. surface and groundwaters), including the land-water interactions (such as diffuse run-off of nutrients from agricultural activities). The requirements for successful water resources management have been identified by Baron *et al.* (2002) as:

Link water quantity with water quality.

Incorporate the needs of ecosystems, e.g. through recognition of importance of flow regimes.

Define resources according to watersheds, river basins, or aquifers.

Recognise that human societies need naturally fun ctioning ecosystems.

Use ecological principles in resource restoration.

Increase communication between engineers, hydrologists, economists and ecologists. A handbook for improving the governance of freshwater resources through the integrated water resources management (IWRM) approach has been produced by the Global Water Partnership (GWP) and the International Network of Basin Organizations (INBO) (GWP/INBO, 2009).

Aquatic ecosystems provide many benefits in addition to the water itself. Wetlands are an integral part of many aquatic ecosystems and they play an important role in flood prevention and the provision of other goods, such as food, nutrients, wildlife conservation, etc. (see Module VII). Some aquatic organisms, especially fish, are an important human food source and other aquatic organisms, especially microbes, contribute to the ability of freshwaters to purify organic waste. These organisms are part of the aquatic ecosystem and therefore it has been argued by some authors that an ecosystem approach is an essential component of any attempt at integrated water resources management (especially if the full range of benefits is to be available to all stakeholders) (Jewitt, 2002).

In order to ensure that water is used wisely and is not wasted, the principle of charging for water use is now widely accepted. Such water charges help support the infrastructure that distributes water and, where necessary, ensures that it is of adequate quality through water treatment. There are many approaches to charging for water use, such as a rate per unit of water supplied or used, but such simple approaches do not take account of the less obvious benefits of the water resource such as its value to wildlife conservation, recreation uses, etc. The application of valuation techniques to water resources management have been reviewed by Birol *et al.* (2006) who argue that it is necessary to establish the full value of environmental resources so that they can be incorporated into private and public decision-making processes. Two examples of the application of ecosystem valuation (WRI, 1998) to decisions on water development projects are given in Box 2.

As a foundation to an integrated approach to water resources management, the amount of water abstracted should be kept to the minimum required and it should be ensured that it is used efficiently. Demand management is one approach that attempts to make better use of the water in the system, rather than relying on the possibility of abstracting more (see Turton, 1999 for a case study). Approaches to demand management include:

reducing leakage and losses in distributions systems,
metering use and charging according to the amount used,
encouraging the use of water-efficient devices,
water conservation techniques, and
forecasting and managing supply.

Such approaches are particularly important in arid regions where irrigation is essential for agriculture and groundwater resources are diminishing (Baroudy *et al.*, 2005).

Awareness and appreciation of the value of a more integrated approach to water resources management led to the development of the European Union Water Framework Directive (EC, 2000), where resources are defined and managed according to river basins. The Directive emphasises better protection for groundwaters and aquatic ecosystems and also requires international collaboration where river basins cross national boundaries. It includes consultation with stakeholders and the public in the development of management plans and anticipates achievement of a "clean and healthy water environment". The specific objectives of the Water Framework Directive are given in Box 3. This ambitious Directive is presenting many challenges to European countries but is an example of an attempt to take account of environmental, economic and social considerations in the sustainable management of water resources.

Box 1. Libya is one of the driest regions of the world with an annual rainfall of 10-500 mm per year. As a result groundwater meets 96 per cent of demand for water. There are vast underground reserves of water in the desert regions but most of the population is concentrated in the coastal areas of the North. A feasibility study showed that the most economical option was to transport available groundwater from the desert regions to of conveying over 6.0 $* 10^6 m^3$ of water every day from well fields deep in the Sahara desert to the population centres on the northern coastal strip. Currently almost 4,000 km (2,485 miles) of mainly 4-m diameter prestressed concrete cylinder pipe transports the water from 1,116 production wells. The distribution system also includes reservoirs for drinking water, flow regulation and agricultural supplies. The project was designed on the assumption that the aquifer system would last 50 years, but it has been estimated that it could last as little as 20 years or perhaps as long as 200 years (Murakami, 1995).

Box 2 Examples of the use of ecosystem valuation in water development projects New York City Rather than build expensive water treatment units to treat water for drinking water supplies in the USA, attempts are being made to protect natural watersheds so that they deliver adequate quantities of source water of good quality. New York City, for example, found it could avoid building new water treatment plants at a cost of US$6-8 billion by protecting its watershed. The natural watershed effectively treats the water for free and, therefore, the city spent US$1.5 billion to buy land around its reservoirs. By implementing other protective measures on this land, the city saved money on keeping its water resources pure and was also able to enhance recreation, wildlife habitats, and achieve other ecological benefits.

Hadejia-Jama'are flood lain, Nigeria

More than a half of the wetlands in the Hadejia-Jama'are flood plain, Nigeria have been lost due to drought and the building of upstream dams. The costs and benefits of proposals to divert more water away for irrigated agriculture were examined using an ecosystem evaluation approach. The net benefits of the water diversion were valued at US$29 per hectare. However, the flood plain was already providing US$167 per hectare of benefits to local people (farming, fishing, grazing livestock, and gathering fuelwood and other wild products). All of those benefits would have been greatly diminished or lost following the irrigation project. Consequently, the wetland was shown to be more valuable to more of the local people if left untouched than if water was diverted away for irrigation. In addition to this, there were other benefits arising from the preservation of the wetland habitat for wildlife.

Conclusions

Human society needs water for health and for social and economic development. Past approaches to exploitation of water resources have been unsustainable, leading to over-exploitation and degradation of surface and groundwaters. New initiatives (i) in placing a value on the water itself, as well as on aquatic ecosystems, (ii) in managing demand and reducing wasteful use of the resources, and (iii) in recognizing the importance of the resource to many different stakeholders, are gradually resulting in a more integrated approach to water resources management. Within Europe, the Water Framework Directive is making an initial attempt to take such an approach.

Box 3 The EU Water Framework Directive the EU Water Framework Directive was introduced in response to a need for collaboration in certain international river basins and to bring together existing, but fragmented, legislation. It also places emphasis, for the first time at EU level, on the protection of aquatic ecosystems. The main aim of the Directive is to achieve "good ecological status" for all water by 2015.

The objectives of the Directive are:

Protect and enhance the status of aquatic ecosystems (and terrestrial ecosystems and wetlands directly dependent on aquatic ecosystems).

Promote sustainable water use by the long-term protection of available water resources.

Provide for sufficient supply of good quality surface and groundwater for sustainable, balanced and equitable water use.

Provide for enhanced protection and improvement of the aquatic environment by reducing or phasing out discharges, emissions and losses of priority substances.

Contribute to mitigating the effects of floods and droughts.

Protect territorial and marine waters.

Establish a register of "protected areas", e.g. areas designated for protection of habitats or species.

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12. Sustainable management of wetlands

Classification and physical attributes of wetlands and their interaction with pollutant inputs.

Wetlands are areas of land where the water table is usually close to the surface, or the land is intermittently or permanently covered by water. Wetlands are an attractive landscape feature and they provide a habitat for native wildlife as well as a potential source of reusable water. There may also be the potential for sporting and recreational facilities, and agriculture and aquaculture farming within artificially managed wetland systems. These factors lead to wetlands being considered as some of the most precious environments on Earth.

Wetlands are classified according to their depth, period of inundation and salinity. The role of wetlands as biological filters to remove or convert contaminants from the open sea or from the river input is of prime importance.

Figure 12.1 gives an overview of the physical processes in a typical marine wetland.

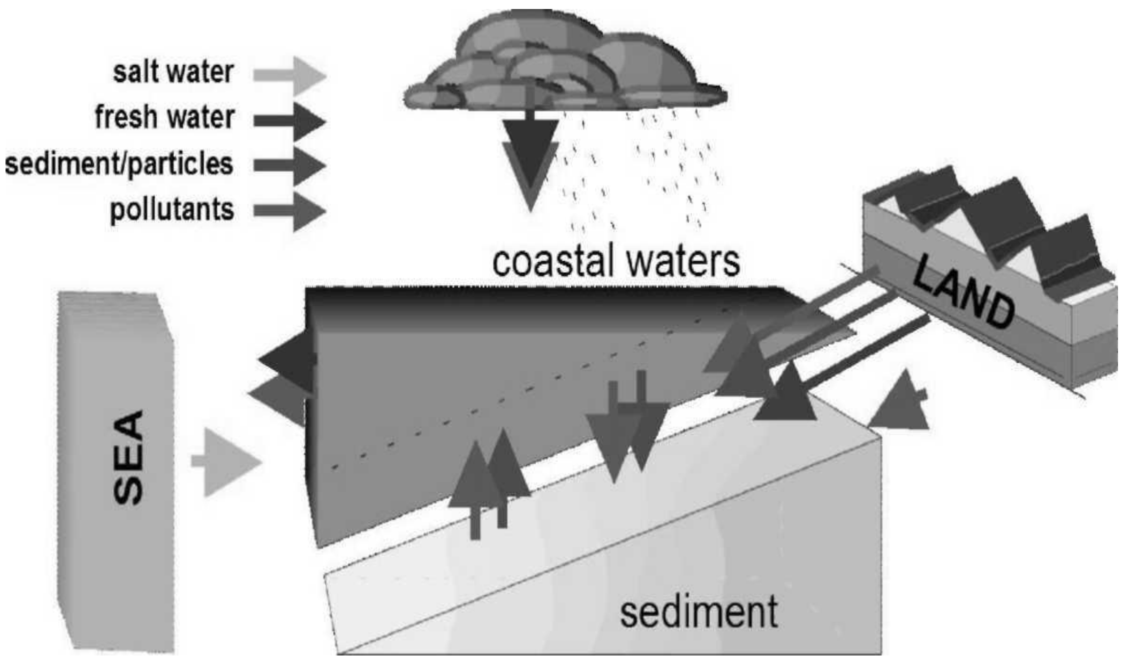

Fig. 12.1. Processes in a typical marine wetland

This chapter tries to define in more detail what a wetland is and how it can be classified. Many possible classifications exist in the literature. Some of them can be summarized as:

The Ramsar Convention,

Functional Classification,

Salinity Stratification,

Flow rates,

Topographic Classification,

Lagoons and water exchange, and

Residence time.

"For the purpose of this Convention wetlands are areas ofmarsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres. "Wetlands " may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands" (Ramsar Convention, 1971).

Under the Ramsar Convention wetlands can be classified in three major groups: marine and coastal, inland and man-made wetlands.

Functional classification

In the functional classification the following divisions can be made:

Inlet zone: This is a transitional zone between the catchment area and the wetland. In this zone the velocity of the inflowing water is reduced which enables the larger particles to settle and sink. Aquatic plants may grow at the edges of this area.

Macrophyte zone: Emergent and submerged aquatic plants occupy this area. The plants, the micro-organisms and the sediments in which these plants grow, take up or convert the nutrients and thus assist in the treatment of the water. Water depths from 10 to 40 cm provide the most effective way to filter and uptake pollutants like suspended solids, nutrients, heavy metals and organic material. Decomposition of organic matter may occur in this zone.

Open water zone: This is a deeper area with lower velocities that allow finer particles to settle and sink to the bed. Due to sunlight and nutrient availability, periodic algal growth may occur in this zone. This process will trap dissolved nutrients and allow them to enter the food chain or settle to the bottom.

Classification through salinity distribution in estuaries

This classification tends to distinguish water bodies through the vertical salinity distribution found in estuaries. To apply this classification it is necessary to define estuaries. Two major definitions exist. The first is by Cameron and Pritchard (1963):

"An estuary is a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage ".

Estuaries described by this definition are known as positive estuaries. Another definition can be derived that does not have this limitation. The new definition includes situations where intermitent closure of the estuary to the sea can happen and where evaporation exceeds the fresh water supply from rivers and from local rain. These are called inverse estuaries and have been classified by Tomczak (2002) as:

> *"An estuary is a narrow, semi-enclosed coastal body of water which has a free connection with the open sea at least intermittently and within which the salinity ofthe water is measurably different from the salinity in the open ocean."*

Using these definitions it is possible to classify estuaries with the help of the vertical salinity distribution found (see Fig.12 2):

Highly stratified estuaries (high flow, low tides)

Slightly stratified estuaries (moderate flow and tides)

Vertically mixed estuaries (low flow, high tides)

Inverse estuaries (hypersaline lagoons)

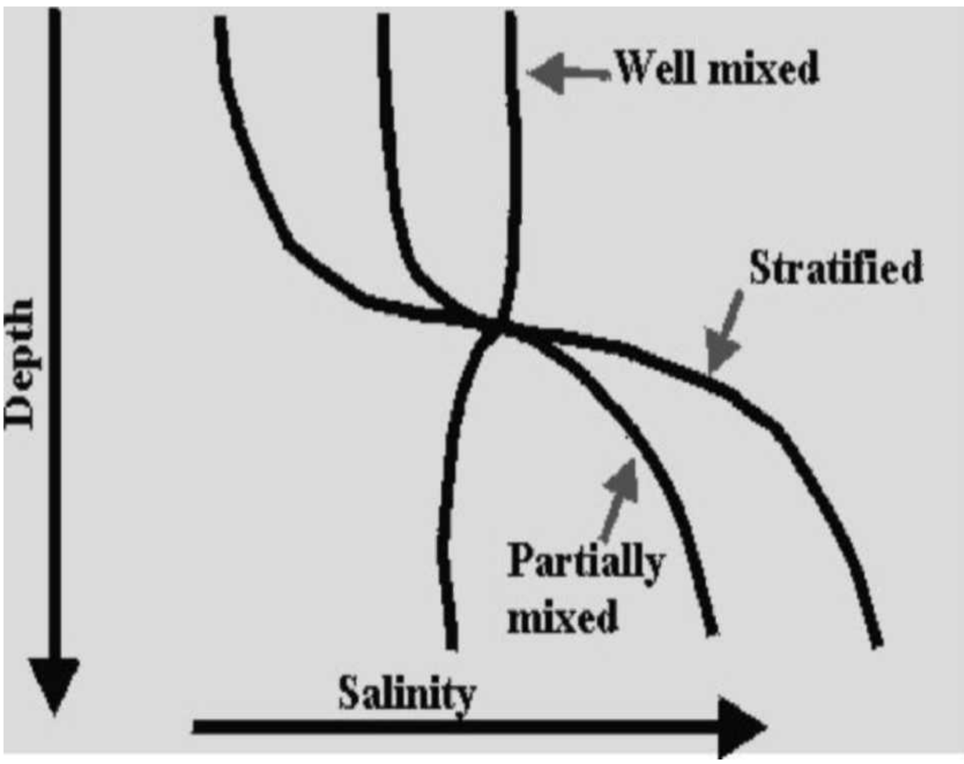

Fig. 12.2 Different types of mixing in an estuary

Highly stratified: A highly stratified, salt-wedge type estuary is one in which the outgoing lighter freshwater overrides a dense incoming salt layer. The dense salt wedge will advance along the bottom until the freshwater flow forces can no longer be overcome.

Partially mixed, slightly stratified: A partially mixed estuary is one in which tidal energy is dissipated by bottom friction produced turbulence. These turbulent eddies mix salt water upward and fresh water downward with a net upward flow of saline water. As the salinity of the surface water is increased, the outgoing surface flow is correspondingly increased to maintain river flow plus the additional upward-mixed saline water. This causes a compensating incoming flow along the bottom. This welldefined, two-layer flow is typical of partially mixed estuaries.

Well-mixed: In estuaries where tidal flow is much larger than river flow and bottom friction large enough to mix the entire water column, a vertically homogeneous (well-mixed) estuary results. If the estuary is wide, Coriolis force may form a horizontal flow separation; and in the northern hemisphere, the seaward flow would occur on the right side (looking down-stream), while the compensating landward flow would be on the left.

Classification based on flow rates and circulation

Estuarine circulation was a dominant consideration in an earlier classification of the Chesapeake Bay by Flemer *et al.* (1983). The flow ratio of estuaries (the ratio of the volume of freshwater entering the estuary during a tidal cycle to its tidal prism) is a useful index of the mixing type and the stratification that will develop (Fig. 12.3). If this ratio is approximately 1.0 or greater, the estuary is normally highly stratified; for values near 0.25 the estuary is normally partially mixed; and for ratios substantially less than 0.1, it is normally well mixed (Biggs and Cronin, 1981).

Hansen and Rattray (1966) developed a two-parameter classification scheme based on circulation and stratification of estuaries. Circulation is described through a non-dimensional parameter U_s / $U\boldsymbol{f}$**,** where U_s is the net (time-averaged) longitudinal surface current and $U\boldsymbol{f}$ is the cross-sectional average longitudinal velocity. Stratification is represented by the non-dimensional parameter AS/ S_0, where AS is the top-to-bottom difference in salinity and S_0 is the mean salinity. Other possible classification schemes are based on the ratio of tidal amplitude to mean depth and other parameters, and have been reviewed by Jay *et al.* (2000). They stress the predictive ability of these simple parameter schemes with regard to salt transport needed to maintain salt balance in modelling.

Dronkers (1988) proposed an estuarine classification that uses water exchange processes (e.g. river flow, tides and waves) affecting mixing and fluxes of energy and material. Hedistinguishes various types of estuarine ecosystems. This classification suggests that river flow variation relative to hydrodynamic residence time can be important for the salinity properties and observed patterns inside estuaries (Cifuentes *et al.*, 1990).

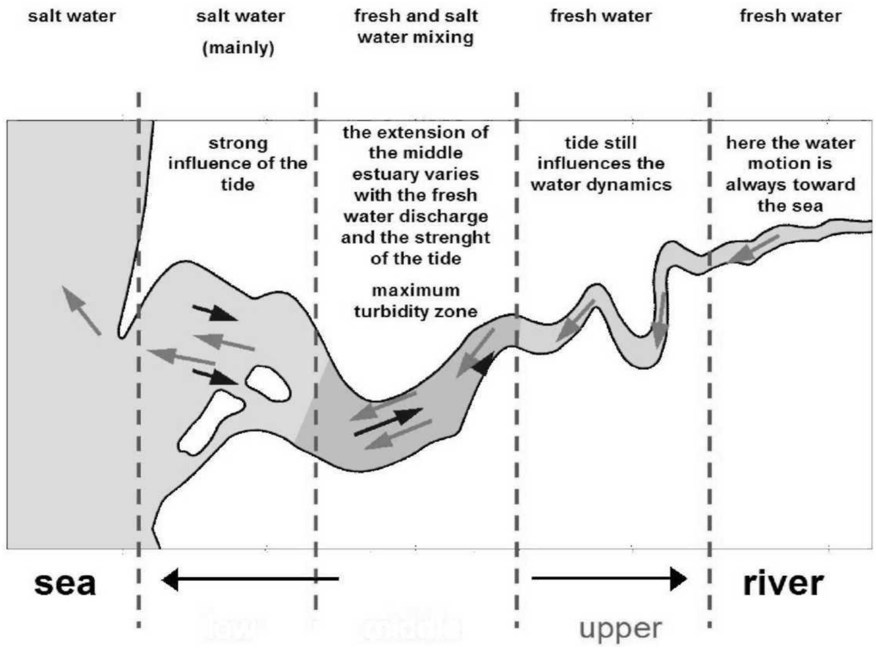

Figure 12.3 Typical zonation in an estuary

Topographic classification

Coastal plain estuaries: The coastal plain estuaries (classical and salt marshes) are characterized by well-developed longitudinal salinity gradients that influence the development of biological communities. Vertically stratified systems with relatively long residence times (e.g., Chesapeake Bay) tend to demonstrate phenomena such as hypoxia formation.

Salt marsh estuaries: The salt marsh estuary lacks a major river source and is characterized by a well-defined tidal drainage network (Day *et al.* 1989). Exchange with the ocean occurs through narrow tidal inlets, which are subject to closure and migration following major storms. Consequently, salt marsh estuarine circulation is dominated mainly by ground-water inflow and tides. The drainage channels usually constitute less than 20 per cent of the estuary, while most of the estuary consists of intertidal salt marshes.

Fjords: Classical fjords are typically several hundred meters deep. Most of them have a sill at their mouth that greatly impedes flushing. Hypoxia or anoxia is often present in the system as a natural feature but anthropogenic nutrient loading can worsen the problem. Fjords were formed by glacial scouring of the coast.

Bar-built structures: Lagoons are characterized by narrow tidal inlets and are uniformly shallow (i.e. less than 2 m deep) open-water areas. The shallow nature enhances sediment-water nutrient cycling. Residence times are usually high. Most lagoon-like estuaries are primarily wind-dominated and have a drainage channel network slightly less developed than the salt marsh estuary.

Some lagoon-type systems have relatively strong vertical stratifications near the freshwater river mouth and may be subject to hypoxia formation.

Other: Tectonically caused estuaries were created by various natural phenomena, such as faulting, graben formation, landslide, or volcanic eruption. Their appearance is highly variable and they may resemble coastal plain estuaries and lagoons.

Lagoons and water exchange

Kjerfve (1986) classified lagoons into three main types:

Leaky lagoons: these have wide tidal channels, fast currents and unimpaired exchange of water with the ocean.

Choked lagoons: these occur along high-energy coastlines and have one or more long narrow channels, which restrict water exchange with the ocean. Circulation within this type of lagoon is dominated by wind.

Restricted lagoons: these have multiple channels, well-defined exchange with the ocean, and tend to show a net seaward transport of water. Wind patterns in restricted lagoons can also cause surface currents to develop, thus helping to transport large volumes of water downwind.

A dominant physical factor in lagoons and estuaries is the tidal amplitude, which provides a means to classify these water bodies relative to their sensitivity to nutrient supplies. Monbet (1992) analyzed phytoplankton biomass in 40 estuaries and concluded that macrotidal estuaries (mean tidal range greater than 2 m) generally exhibit a tolerance to nitrogen pollution despite high loadings originating from freshwater outflows. These systems generally exhibit lower concentrations of chlorophyll *a* than do systems with lower tidal energy, even when they have comparable concentrations of nitrogen compounds. Estuaries with mean annual tidal ranges of less than 2 m seem more sensitive to dissolved nitrogen, although some overlap occurs with macrotidal estuaries.

Classification based on residence time

Water residence time, the average length of time that a particle of water remains in an estuary, influences a wide range of biological responses to nutrient loading, and therefore also the appearance of lagoons and estuaries. It also directly affects the residence time of nutrients in estuaries, and therefore the nutrient concentration for a given loading rate, the amount of nutrients lost to internal processes (e.g. burial in sediments and denitrification), and the amount exported to downstream receiving waters (Nixon *et al.*, 1996). Residence times shorter than the doubling time of algae will inhibit bloom formation because algal blooms are exported from the system before growing to significant numbers. Residence time can also influence the degree ofrecruit-ment of species reproducing within the estuary (Jay *et al.*, 2000). There are a number of definitions of water residence time (Zimmerman, 1976; Takeoka, 1984; Miller and McPherson, 1991; Hagy *et al.*, 2000), some of them in apparent disagreement.

Estuaries that flush rapidly (i.e. have a short residence time) export nutrients more rapidly than those that flush more slowly, resulting in lower nutrient concentrations in the estuary. By contrast, highly stratified systems are more prone to hypoxia than are vertically mixed systems. Stratification limits downward transport of oxygen from atmospheric re-aeration, and retains nutrients in the upper layers, making them more available to phytoplankton. In stratified systems, it may be more appropriate to estimate the dilution potential of the estuary using the volume above the pycnocline rather than the entire volume of the estuary.

Pollutants in coastal wetlands

When the freshwater of continental origin meets the sea water (mixing zone), a series of physical and physico-chemical processes occur (mixing processes), due to the different characteristics of the two waters, i.e. density, temperature, ionic field, suspended particles concentration, etc (Burton, 1976; Liss, 1976). In particular, finer suspended particles (and organic matter) transported by the freshwater flow tend to increase their weight by aggregating (flocculation, precipitation), thus increasing their tendency to sink to the bottom. By contrast, turbulence occurring in the water column (and particularly in the lower layers close to the bottom) can lead to the re-suspension of particles. The overall result is quite often the presence of (maximum) turbidity zones.

Finer suspended particles (and organic matter) are the preferential carriers of pollutants from the land to the sea. As a consequence, high levels of pollutants are found in the zones where finer particles accumulate, i.e. slack dynamics zones such as shallow lateral areas of estuaries, lagoons and bays. Moreover, these locations where polluted sediments accumulate could be also subjected to the presence of other pollutant sources, such as cities, industrial districts and wastewater treatment plants, etc. Sediments can, in turn, behave as secondary sources of pollutants for the water column. As a consequence, the monitoring of sediment condition is important to describe and control the ecological status of the coastal zone.

Investigations in the estuarine environment offer fundamental information on the trends in pollutant processes, for maintaining biodiversity, and for the control of the impact of human activities on aquatic living organisms. By studying water circulation and matter transport in these environments basic information for the management of the freshwater bodies and for the planning of coastal protection actions can be obtained.

Coastal zones are essential for the equilibrium of the whole marine ecosystem, because the large amount of organic matter that is produced in their waters is fundamental for the mainteance of the food chain. Despite accounting for no more than 15 per cent of the surface and 0.5 per cent of the total volume of the ocean (Goldberg, 1976), about 90 per cent of marine living resources comes from the coastal waters. In addition, more than 60 per cent of the world's population lives within 60 km of the sea, and human activities are changing rapidly the fluxes of matter and related pollutants from the continent to the coastal zones (Murray *et al,* 2001). These systems are therefore subjected to very strong and increasing pressures, which induce degradation phenomena, including the loss of habitats for living organisms and biodiversity, water and sediment pollution, eutrophication and landscape deterioration. European coasts, in particular, are most affected, with some 80 per cent being at risk.

Global awareness of the need for comprehensive protection of the marine environment and coastal resources has grown stronger since the 1972 United Nations Conference on Human Environment held in Stockholm. International agreements were established and different policy actions were taken towards sustainable and integrated coastal management.

Nevertheless, scientific investigations are still needed to acquire an adequate understanding of coastal area conditions and related environmental processes, to counteract the effects of stress factors, and to permit the forecasting of consequences of management options. Freshwater discharged by rivers, together with municipal and industrial effluents, are generally the main source of pollutants for coastal zones, because most of the pollution load to the oceans derives from land-based activities, including releases into the atmosphere. This issue places the quality of freshwater delivered to the sea as a topic of world-wide importance.

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13.Understanding coastal morphology and sedimentology

Beaches are sedimentary bodies situated at the land-sea interface and usually made up of non-cohesive particles of mainly sand size, although coarser sediments (such as shingles) are dominant in some specific settings (Bird, 1996). With regard to mineralogy and petrography, coastal sands consist mostly of quartz grains, but may also comprise various lithic fragments, mica, heavy minerals, oolites and others. Shells, shell fragments, or non-biotic carbonaceous fragments such as eroded limestone clasts, volcanic material, pebble and boulder-size fragments from nearby cliffs may also be present and even predominate under specific circumstances. Beaches show a high spatial and temporal variability in terms of morphology, sedimentary balance, source of sediments and erosional/depositional character. Each beach comprises several specific emerged and submerged sub-domains (or zones) with distinct particle behaviour and related morphology.

Defining the limits of beach zones has been a matter of discussion over past decades, but still remains a difficult task (Friedman *et al.*, 1992). The beach landward limit is considered to be the maximum reach of storm waves, in front of coastal dunes. For the seaward limit, four options have long been debated: the "low water line", the external limit of oscillatory waves, the outer limit of the breaker zone and a fixed selected water depth of 10 m. Each of these outer limits has its weak points (see Friedman *et al.*, 1992 for a detailed discussion). Recently, most researchers seem to have accepted the external limit of oscillatory waves, where the interaction between waves and sediments begins, as the seaward limit of a beach. The depth of such a limit will vary in response to changing wave characteristics and catastrophic events, such as tsunamis or extremely strong storms, would shift the limit further seaward to deeper waters. From a geomorphologic point of view, Friedman *et al.* (1992) classify beaches as: mainland beaches - placed at the boundary between continent and sea;

strand plains - prograded sedimentary bodies, deposited by waves and currents at a certain distance from the coast; and barrier beaches - situated at the sea-side edge of barrier islands and spits.

Because coastal wetlands and lagoons are generally separated from the sea by beaches, the genesis and evolution of the latter are essential for the overall dynamics of the entire coastal wetland or lagoon system. The specific morphology of each beach is the "mirror" of the way in which coastal sediments react to the local wind and wave regime. Knowing the sedimentology and geomorphology of the beaches bordering coastal lagoons and wetlands, some of the possible interactions between the marine coastal waters and the lagoon or wetland waters can be understood. Indeed, the predominant sedimentary processes (erosion, stability or accumulation) controlling a beach which borders a lagoon or a wetland also influence the overall dynamics and migration of the lagoon or wetland. Case studies for some of the theories of coastal morphology and sedimentology presented in this chapter can be found in Dominik, Stanica and Thomas (this volume). Also note that the theories presented here are mainly characteristic of microtidal coastal environments.

Beach morphology

Starting from the mainland and moving seawards, a typical beach transverse profile includes the backshore, the foreshore and the nearshore (Figure 13.1). The backshore, i.e. the emerged part ofa beach, is limited in landward direction by the dune line, which separates it from the coastal dune zone. Although they do not evolve under the direct influence of the sea, the dunes play a major role in the sedimentary budget of beaches because they act as sand reservoirs when the wind direction is towards the sea. The coastal dunes (Nordstrom *etal.*, 1990) are areas of active aeolian sedimentation in case of prograding beaches, or when the aeolian sedimentary balance is positive. During storms, coastal dunes play a major protective role for the beach system, acting as a barrier against storm waves and related sea surge. The backshore is under the alternate influence of the aeolian and wave transport, depending on the wind and wave regime.

The seaward limit of the backshore is the active berm crest (see Fig. 13.1).

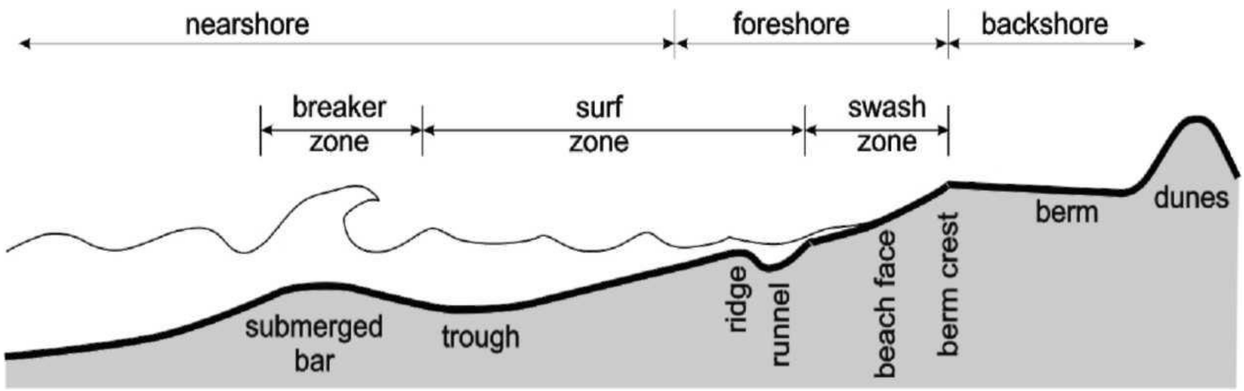

Fig. 13.1 Transverse view of the three main zones of a beach (nearshore, foreshore, and backshore) and of their most relevant morphological characteristics. Breaker, surf and swash zone identify the relevant regions for wave dynamics and particle transport on the beach (see text for details).

The whole structure of the backshore actually consists of fine sediment laminae (gently dipping landwards), representing a succession of stacked fossil berms. These berms record the history of the variations of recent wave energy and the related advance of the beach on land. The berm crest records a change in the dip of the sediments' laminae to seaward direction. Berms are formed through repeated deposition of particles rolled onto the swash zone (Fig. 13.1) by incoming waves, sometimes advancing over the berm crest. The relationship existing between the wave energy and the shape, dip angle and width of the swash zone explains why several berms, corresponding to various energetic conditions, are often met along the same transverse profile.

The foreshore is the seaward dipping part of the beach and is separated from the backshore by the berm crest. The landward part of the foreshore is a gently dipping surface (5-8°) known as the beach face, where the waves break (see also "swash zone" (Figure 13.2). The shape of the beach face depends on the wave height (a proxy of wave energy) and on the beach sediment grain-size. The beach face is in a state of almost continuous change, with cyclic and seasonal patterns depending on the wave energy. The beach face continues seawards with a shallow ditch called a "runnel" and followed by an asymmetric sand bar known as a "ridge". A small morphologic step, where the coarsest sediments of the beach concentrate, is present between them. Beach progradation takes place by sediment deposition on the ridge until the water surface level is reached and the runnel is filled up with the particles which are washed over the ridge crest. The ridge then becomes a new berm. The time necessary for this process is variable, but it can be as short as a few days in the case of calm sea with constructive waves.

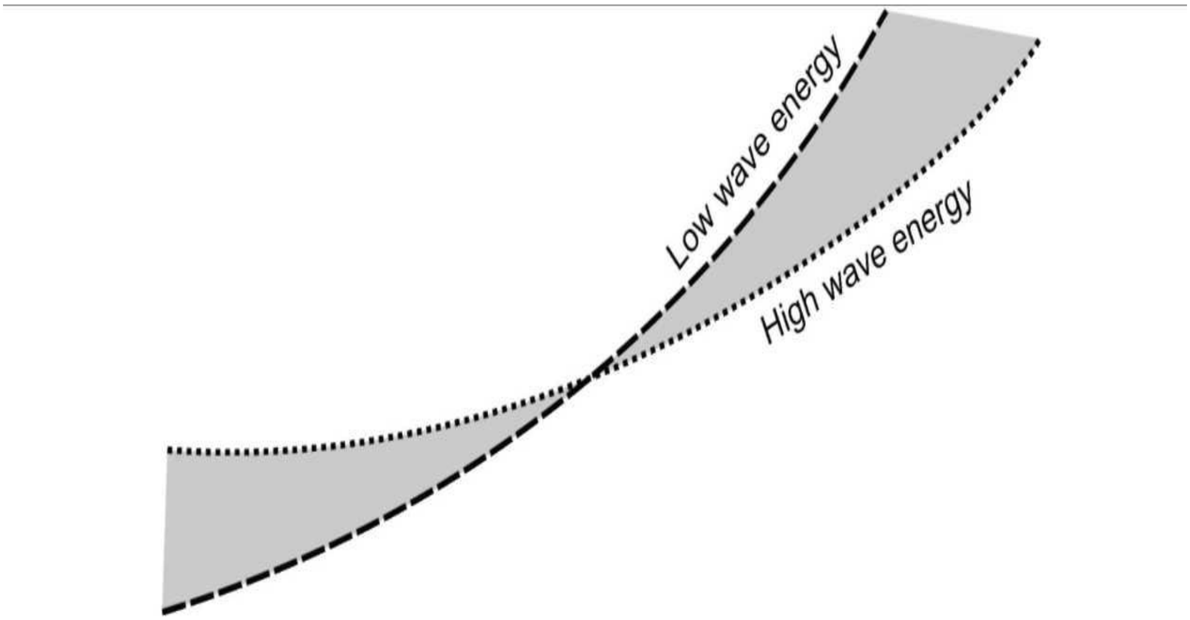

Fig. 13.2 Relationship between wave energy and swash dimensions and angles. The higher swash slope corresponds to a low wave energy which in turn marks the predominance of sediment accumulation processes. The lower swash slope corresponds to high wave energy, when the sediments are generally mobilised from the swash offshore.

The nearshore is characterized by the presence of one or several bars separated by troughs (channel-like depressions). The mechanism of bar formation is the response of the beach to the action of waves breaking over the seabed.

Shorelines are characterized by other specific and significant longitudinal beach morphological features, such as the beach cusps (a succession of small scale bays and promontories), which usually develop along regressive shorelines. Cusp amplitude and wavelength depend mainly on wave energy and shoreface slope. Beach scarps (nearly vertical slope along the beach, immediately under the berm crest, generated mainly by waves acting vertically on the shoreline) are also a significant longitudinal morphological feature, and may form when the erosive tendency is more important.

Dynamics of beach sediments

Littoral sedimentation is mainly controlled by marine hydrodynamics (e.g. Davis, 1992). Waves and currents are the transport agents which play the most significant role in controlling sediment migration and deposition. Both waves and currents are mainly generated by, and dependent on, wind conditions.

Heavier and coarser sediment particles are transported as bedload by rolling and dragging; while the lighter and finer particles are transported in suspension. Intermediate grains, too heavy to remain permanently suspended, are taken in suspension by the approaching wave and then resettled. Each particle is reworked by successive waves, hence the good sorting and roundness of sand grains on beaches.

Sediments are transported in a direction which is either approximately transverse to the shoreline (either landwards or seawards) or alongshore (e.g. U.S. Army Corps of Engineers, 1984). Sand starts to be transported by waves towards the coastline as soon as the water depth reaches one quarter of the wavelength. Above the nearshore submerged bars, the reduced water depth causes the wave steepness to reach a threshold so that waves break and maximum turbulence is created. As a consequence, the finer material is taken into suspension and transported landwards by currents and only the heavier particles remain on the bar crest (Fig. 13.3).

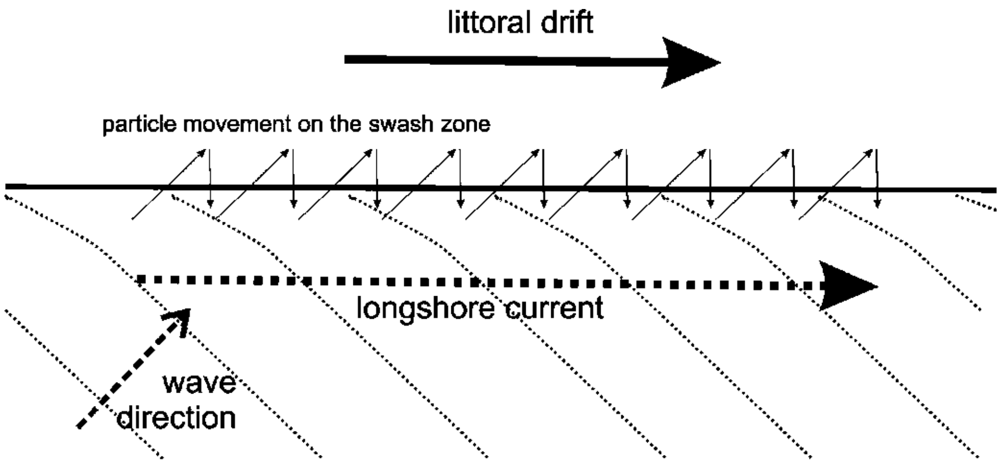

Figure 13.3 The motion of sediment particles in the coastal zone. Formation of longshore littoral drift and longshore current.

Further landwards of the breaker zone, mainly because of the very shallow water depth, the predominant motion of water and sediments is no longer oscillatory but a landwards translation. This part of the beach is known as the surf zone (see Figure 13.4).

The final water and particle movement takes place in the swash zone. The upward movement on the swash is called "uprush" and carries all the material in a carpet-like transport in the direction of the incoming wave. Under certain conditions, some of the material is transported beyond the berm crest and contributes to the formation of the backshore. The water returns seaward along the shortest direction ("backwash") and carries back only the finer sediments (Friedman *et al.*, 1992). Longitudinal transport is partially a consequence of this water movement on the swash. If the incoming waves approach the coast with an incident angle, the combination of uprush and backwash creates a saw-like particle movement: the beach drift. Another component of the longshore sediment transport is the general, wave-induced, longshore littoral drift. The longshore drift may change direction with varying conditions (wind, sediment grain-size, sea bed slope, etc). Every short-time constant movement of the sediment is known as a migration and can be described as a vector. The sum of all migrations in a year represents the littoral transfer or littoral drift of the sediments.

The general wave movement causes an excess of water to accumulate near the shoreline. The water balance is equilibrated by the formation of rip currents (Fig.13. 4). The space between two rip currents delimits a littoral cell. Rip currents (Davis, 1992) play a significant role in the seaward transport of sediments. They remove large amounts of sand from the Komar, 1998, Ungureanu and Stanica, 2000). Actually, they block the natural longshore current causing starvation of downdrift beach sectors.

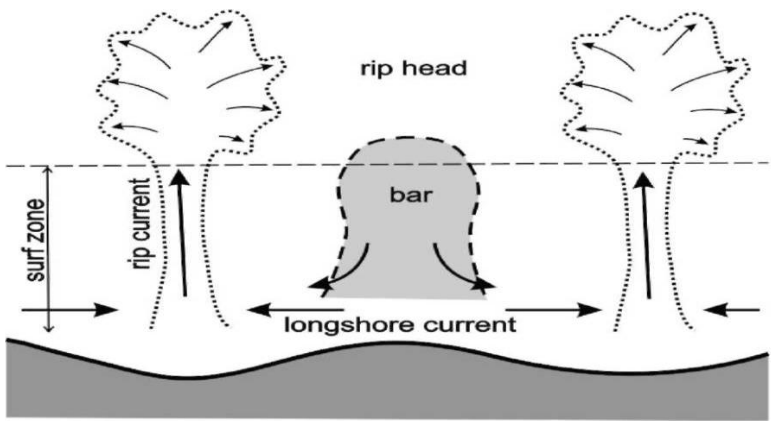

Fig. 13.4 Schematic view of a littoral cell. Cycle of water movement - mainly landwards in the surf zone and, after the uprush-backwash cycle, moves mainly alongshore.

Foreshorezone and depositit back when their velocity decrease sand the flow expands sea wardof the surf zone, in what is known as a rip head.

The overall sum of the components of sediment transport defines the sedimentary budget (also known as sedimentary balance), which is the main parameter that characterizes the dynamics of the littoral zone. It is computed for each littoral sector by taking into account all sediment inputs into, and outputs from, the chosen region (Van Rijn, 1993). Inputs of sediments are: rivers that flow into the sea, littoral longshore transport from neighbouring beach sectors, onshore transport, eroding cliffs or coastal dunes, or human related activities such as artificial sand nourishment or planned sediment by-passing of certain obstacles. Sediment outputs include transport to neighbouring sectors of the coast by the longshore current or towards the offshore direction. Sediment can also be removed from the dynamic part of the beach system by deposition on the coastal dunes or on prograding coastlines.

The human factor

Human activities have significantly interfered with natural conditions and processes, mainly during the last two centuries. With regard to beach morphology and sedimentology, the main consequences have been important disturbances of either the sediment budget or the sediment transport patterns, or both. Significant anthropogenic impacts on the littoral evolution are documented worldwide (e.g. U.S. Army Corps of Engineers, 1984; Bird, 1996; Komar, 1998; Ungureanu and Stanica, 2000; Poulos and Collins, 2002). The most common interventions are dredging of river mouths and canals for navigation purposes, damming of rivers, blocking of littoral longshore currents by hard coastal engineering interventions (groynes, jetties, breakwaters, etc.), removal of beach sand for economic reasons, and beach by-passing and nourishment.

Dredging river mouths and canals for navigation removes large quantities of sediments from the coastal sedimentary budget because the dredged material is usually dropped into the open sea. Damming (Poulos and Collins, 2002) and other hydrotechnical works along rivers affect the coastal sedimentology by severely decreasing the river sediment load. The result is a significant deficit of sand brought by the rivers to the adjacent littoral zones.

Coastal engineering works are probably the in situ structures which have most influenced coastal zone evolution. The construction of harbours, regardless of whether they are large commercial harbours or small-size marinas, is associated with hard protection structures, such as groynes and dykes. Harbours and associated structures always modify the distribution oflittoral currents (e.g.

The same effect has been noticed in the case of hard defence works built to protect tourist beaches worldwide. In general, the construction of hard coastal engineering structures (such as seawalls, jetties, groynes, breakwaters) simply moves the erosional phenomenon elsewhere (or even increases it locally). Recent decades have proved that the most effective coastal engineering work, with an overall positive effect, is artificial beach nourishment. This "soft" coastal engineering method, if performed properly, mitigates the sediment budget disequilibrium generated by humans by supplying the missing volumes of materials to the sediment-starved beaches.

Conclusions

This chapter describes some basic theories regarding beach morphology and sedimentary processes which occur in low-lying coastal areas. In the case of sandy beaches bordering coastal lagoons or wetlands, knowing the beach dynamics enables the factors which can affect the overall conditions of the lagoon or wetland system to be understood. For example, erosion may cause both an overall inland motion of the beach bordering a lagoon or wetland (that is a change in land characteristics within the lagoon or wetland territory) and temporary or permanent breaches in a littoral sand bar (beach), with significant consequences on the marine-lagoon water exchange processes.

Humans have probably become the most significant disturbing factor in the evolution of beaches. Human interventions can be grouped in two main categories: direct interventions along the coast, or works in neighbouring river catchment areas which usually reduce the volumes of sediment supplied to the coast. Recent decades seem to have brought an increased application of more environmental friendly management options to mitigate the negative effects of human interventions on the morphology and dynamics of beaches.

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14. Habitats and species diversity: relationships in the Danube Delta

The Danube Delta is located between the northern parallels 46°42' and 44°24' and the eastern meridians 28°44' and 29°46'. It is delimited by the Bugeac plateau in Ukraine to the north, by the Dobrudja horst in Romania to the west and by the Black Sea to the east and south-east (Banu and Rudescu, 1965). The Danube Delta territory is a recently formed, low-relief unit resulting from the silting of a marine gulf. The process was prolonged over several thousand years, undergoing several stages of silting, and is continuing today. The Delta is divided into two geographic subregions: the proper delta, situated between the Danube branches, and the lagoon complex Razelm-Sinoe (Diaconu and Nichiforov, 1963; Vespremeanu, 2004).

The Danube Delta has a continental-temperate climate with large fluctuations in tempera-tures and an annual average air temperature of 11° C. Annual precipitation events are generally scarce, averaging a total of 450 mm, decreasing from west to east. By contrast, air humidity increases towards the east due to the intense evaporation occurring in this part of the Delta, creating a suitable climate for the development of a mosaic of vegetation (Banu and Rudescu, 1965; Paltineanu et al., 2000). In fact, the Danube Delta is a region with a remarkable variety of substrata and climatic conditions, which is favourable for the development of different habitats and for colonisation by various species. This contribution briefly presents the habitat and species di¬versity of the Danube Delta, their spatial and temporal dynamics, and the major driving forces behind such dynamics. A glossary of terms used in this chapter is given in Box 1.

Box 1 Glossary of terms

Alluvia, deposit of sand and mud ofriverine origin.

Benthos, the assemblage of organisms living on or in the aquatic ecosystem's bottom. Biocoenose, all organisms living together and interacting in a specific place/biotope. Biome, assemblage of different and strongly interlinked biocoenoses situated in an area with defined climate and geographical conditions. Biotope, area presenting uniform environmental conditions suitable for a specific assemblage of plants and animals. Canal, artificial waterway (or one made by altering a natural river channel). Channel, natural passage along which water flows.

Cosmopolitan, in biogeography, an organism that is distributed throughout the world in suitable habitats.

Ecosystem, natural unit definited by all plants (primary producers), animals (consumers and parasites) and microorganisms (decomposers) living and functioning together with all of the physical factors of the environment.

Endemism (endemic), restriction of a species to a single place.

Eurasiatic, in biogeography, an organism that is distributed in Europe and in Asia.

Floating vegetation, usually indicates macrophytes, but sometime includes also filamentous alga developed at the water surface.

Geostrophic current, marine current resulting from the balance between gravitational forces and the Coriolis effect.

Habitat, the place were an organism lives/can be found.

Horst, elongated, upthrown block of crust bounded by systems of faults on each side. Isobaths, lines joining points of equal depth. Levee, river embankment against floods.

Pan-European, in biogeography, an organism that is distributed in whole Europe.

Phenology, study of periodic plant and animal life cycle events and how these are influ¬enced by seasonal and inter annual variations in climate.

Plateau, an extensive elevated continental surface underlain by essentially horizontal rock layers.

Primary production, the synthesis of organic matter from inorganic components (through photosynthesis or chemosynthesis) Sandbank, deposit of sand along a river or the sea shore.

Secondary production, production by heterotrophic organisms; i.e., the production of primary consumers. Silting, process consisting in covering of a surface with fine sand and/or mud. Species diversity, in ecology, a numerical measure combining the number of species in an area with their relative abundance. Zooplankton, planktonic animals (primary consumers, suspension-feeders and carnivores).

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Benjamin Cummings, San Francisco

Genesis of the Danube Delta

The genesis of the Danube Delta is a long, complex and extremely dynamic process and the reader seeking detailed information on the subject is referred to the comprehensive works of Panin (1996; 1998; 1999), Panin and Jipa (2002) and also to the more recent theory of Giosan et al. (2006). This section presents only a simplified overview of the factors contributing to the determination of habitat (and hence species) diversity.

The Danube Delta resulted from the interaction between the river and the sea as a result of several favourable factors: the large catchment area of the Danube river (and consequently the huge quantity of alluvia transported to the Black Sea), the typical microtidal marine environment of the Black Sea and the presence of a large continental shelf. The shelf is gently in¬clined offshore (9.80 per mile on the eastern meridian 31°), reaches a depth of about 200 m, and has the greatest width in the north-western part of the Black Sea, just in front of the mouth of the Danube (Onciu et al., 2006). The general water circulation, which in the north-western part of the Black Sea is from north to south and parallel to the Danube Delta region (Vespremeanu, 2004), is also an important factor in shaping the Delta and controlling its evolution.

With regard to sedimentary processes, the north-western shelf of the Black Sea includes both an internal shelf fed by the Danube's sediment and an external, sediment-starved shelf (Panin et al., 2002). For the purposes of this chapter, it is however better to distinguish between the fluvial delta to the west and the fluvial-maritime delta to the east (Panin, 1989). These two regions reflect the relative roles of fluvial sediment sources (i.e. the Danube itself, as well as the Ukrainian rivers which discharge north of the Danube mouth and from which the sediments are transported by the longshore currents) and of the general circulation patterns of the north-western Black Sea (Panin, 1989; Panin and Jipa, 2002) in determining the particular ecological characteristics of the various habitats and, consequently, their ability to host different flora and fauna. In the fluvial Delta, the three branches of the Danube (Chilia, Sulina and Sfantu Gheorghe) through which the river flows into the sea, divide the Delta into three large areas improperly called islands: Letea (bordered by the Chilia and Sulina), Sf. Gheorghe (located between Sulina and Sf. Gheorghe) and Dranov (situated between the Sf. Gheorghe branch and the Razelm Lagoon complex).

The evolution of the Danube Delta has been greatly affected by the construction of many hydrotechnical works. In particular, the Iron Gate I and II dams greatly altered the sediment fluxes carried to the lower reaches of the river and to the Delta. Panin et al. (2002) estimated that the construction of the Iron Gates dams decreased the annual sediment discharge from 61.5 x 106 t a-1 to 25-35 x 106 t a-1. Some of the most destructive human activities in the fluvial Delta started under the "Program for the remodelling and the integral use of the natural resources in the Danube Delta", developed during the last two decades of the Communist period. Embankments were built along about 17 per cent of the Delta channels and many natural habitats were either transformed into fishponds or drained and transformed into polders for agricultural purposes. Consequently, water inundated representative habitats, such as reed beds or grassland on river levees, creating new pond ecosystems with a permanently low productivity. On the other hand, lakes with reduced exchange of water and partially covered with floating vegetation, or other flooded reed beds, were drained for creating polders which became saline in a few years (Gastescu and Stiuca, 2008).

3. Main forces determining habitat and species diversity

The Danube Delta is a complex assemblage of strongly interlinked aquatic and terrestrial habi¬tats that can be defined as a biome. The major driving force of the Delta is the hydrological regime of the Danube River and its periodic floods which shift the ratio between terrestrial and aquatic habitats, thus affecting all the components of the biome (Botnariuc and Vadineanu, 1988; Torica, 2006; Tudorancea and Tudorancea, 2006).

The main factors controlling the habitat and species diversity in the whole Danube Delta biome and, consequently, in its various ecosystems include:

The inundation pattern of the ground with respect to the zero level of the Black Sea. The periodic flooding determines a specific cycle for the life of the whole biome and affects the structure of vegetation and fauna and their relationships, as well as the phenology of the individual species.

The flood duration and intensity. The period of high water and the water levels reached in the Delta strongly influence the biological production and the species patterns contributing to it.

The hydrology-driven processes. Floods promote an intense circulation of mineral substances and organisms in all ecosystems of the Delta and disperse the water layers rich in H_2S, thus replenishing the oxygen content of the water in the area. Even the primary and secondary production of terrestrial ecosystems greatly depend on aquatic ecosystems, and many terrestrial vertebrates also feed on aquatic plants. Finally, floods carry a substantial nutrients load which determines the enrichment of water in the levees.

Water transparency. The importance of this feature is the main consequence of the interplay between the large surface area and the small water depth in the Delta. Light penetration is a decisive factor for plant development and aquatic macrophytes do not develop when water transparency is lower than 0.2 m.

Habitat diversity

Reed beds, covering over 50 per cent of the Danube Delta area, are the dominant habitat type. However, the Delta is characterized by a high diversity of habitats which can be classified according to different criteria such as human impact, hydrology (e.g. terrestrial and aquatic habi¬tats depending on water level) and vegetation cover (which acts as an indicator of environmental conditions).

Specialists consider that each ecosystem is characterized by specific habitats. Within each ecosystem of the delta, specific habitats determine the development of a specific local biocoenoses or associations. The basic component of all these biocoenoses is the primary pro-ducers, among which macrophytes have the dominant weight and, due to the diversity of habitats, have a mosaic-like distribution (Donita et al, 2005; Sarbu et ah, 2005). The plethora of habitat types that can be identified in the Delta also explains the high species diversity that can be observed for all taxonomic groups.

Species diversity

The Danube Delta is part of the Black Sea and steppe biogeographical provinces that provide habitats for 50 per cent of the 3,800 plant species recorded in Romania. The water habitats of the Delta itself and the Razelm-Sinoe lagoon support 779 species belonging to Eurasian, continental Asiatic and cosmopolitan groups.

Plants

In the Danube Delta there are two endemic plants (Centaurea jankae and C. pontica) (Gomoiu, 1998; Donita et al., 2005; Sarbu et al., 2005), but euroasiatic and cosmopolitan plants dominate the aquatic, swamp and marsh communities. Among the latter, the reed (Phragmites australis) which has a worldwide distribution, the flag iris, the bulrush and the purple and yellow loosestrife are noteworthy. Many other submergent plants have a pan-European distribution and include the white water lily (one of the symbols of the Danube Delta Biosphere Reserve) and the water chestnut (Trapa natans) which, uncommon or absent in Western Europe, is well represented in the Delta. A variety of terrestrial plants occur along the edges of rivers and canals and on marginal sandbanks. Salix spp. and Populus spp. dominate forests developed on marginal sandbanks; while mixed oak forests characterize the marine levees. Euphorbia spp., Secale spp. and the interesting Ephedra distachya are common on the sand dunes. The Mediterranean vine Periploca graeca is widespread in Letea Forest (Sarbu et al., 2005).

Invertebrates

Invertebrates are highly diversified within the Delta with over 190 species of copepods and cladocerans as well as 418 species of rotifers that are found in the zooplankton of running waters and, especially of standing fresh and brackish waters. Nematode worms, oligochaeta and three species of polychaeta are common in the benthic communities of sandy or muddy bottoms. Eighty-five species of molluscs (65 gastropods and 20 bivalves) have been identified in the Danube Delta Biosphere Reserve. Insects are well represented with 196 species listed as endan-gered and endemic species, such as the bush-cricket Isophya dobrogica on the Popina Island. Insect larvae also represent an important food item in aquatic ecosystems (Skolka et al., 2005; Zinevici and Parpala, 2007).

Fish

The fish fauna has a high diversity (more than 75 species) and, given its great economic value, is one of the most studied groups of organisms. Starting with the classical monograph of Antipa (1909) and continuing with the recent papers of Nalbant (2003) and Otel (2007), the literature on ichthyofauna grows continuously. Three main fish habitats are usually considered: the three Danube River branches (harboring the richest ichthyofauna including strictly running-water species), the deltaic lakes (whose fish populations are less diversified, but of great economic im-portance) and the lagoonal complex Razelm-Sinoe where the diverse fish fauna has experienced important changes in the last decades. The progressive blockage of the connections between lakes of the Lagoon complex and the Black Sea (Bretcan et al., 2009), and the subsequent relative increase in fresh water inflow from the Danube, has led to marked and abrupt salinity changes, a decrease in transparency, changes in the depth of lakes and, coverage of the hard bottom of the lakes with alluvia from the Danube. Consequently, the qualitative and quantitative structure of phytoplankton, zooplankton, bottom macroflora and zoobenthos has changed, thereby altering the available food resources for fishes. Populations of sander (one of the most important fish species from an economic point of view, especially in the Sinoe Lake) have decreased as aresult of areduction in goby populations. Goby fed onDreissena sp., but in the new conditions in the lakes larvae of zebra mussels have not found the rocky substrata on which to settle (Otel, 2007).

Amphibians and reptiles

There are 11 species of amphibians within the Delta (Triturus (Lissotriton) vulgaris, T. dobrogicus, B. bombina, Hyla arborea, Bufo viridis, B. bufo, Pelobates fuscus, P. syriacus, Rana (Pelophylax) esculenta, R. (Pelophylax) ridibunda, R. (Pelophylax) lessonae) and 11 species of reptiles (Vipera ursinii, Natrix natrix, N. tessellata, Coronella austriaca, Coluber (Dolicophis) caspius, Emys orbicularis, Eremias arguta, Lacerta agilis, L. trilineata, Podarcis taurica) (Cogalniceanu et al., 2000; Skolka et al., 2005).

Birds

There is no other place in Europe where such a great diversity of land and water birds can be found. Three hundred and seventy-five bird species are recorded in Romania and 325 of them live in the Delta or migrate into it during the summer or winter. One hundred and sixty-six spe¬cies nest in the Biosphere Reserve and most of them are summer migrants that spend the winter in Africa or in the Mediterranean, such as the common and Dalmatian pelicans, white storks, herons, egrets, spoonbills and swallows. The Delta provides a habitat for about 60 per cent ofthe world population of pigmy cormorant, and a significant proportion of the world's population of red-breasted geese overwinters in the Dobrudja horst. Two hundred and twenty-four species of the Delta's avifauna are currently given strictly protected status (Radu, 1979; Ciochia, 2002).

Mammals

Among the 110 mammal species that are native in Romania, 40 have been recorded in the Danube Delta. Mammals can find good shelter in the high grassy vegetation, the willow woods, and in the forests on the most important sand banks (Murariu, 1996).

Conclusions

The Delta consists of a great variety of habitats including the largest area in the world (about 2,800 km2) covered with Phragmites australis. The alternating lotic and lentic water bodies is also remarkable. Natural river branches and man-made canals reach a total length of 3,496 km which is roughly equally split between the two categories. Delta lakes currently number 479 and cover about 8 per cent of the total Delta area, although their number and surface cover have shrunk from the respective values of 668 and 9.3 per cent measured at the beginning of the 1980s.

Such a high habitat diversity is reflected in the high species diversity. The Delta practically represents a supra biocoenotic level of organization of the living matter, where species and habitat diversity are linked via the dynamic processes occurring in the Delta itself. Although the Delta is a disturbance-dominated system, human interventions, especially those altering the characteristics (occurrence, length and extent) of spring and autumn floods, can drastically modify the delicate balance between abiotic and biotic factors.

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15. Biological diversity and habitat diversity: a matter of Science and perception

Any measure of diversity depends on the approach used to identify and classify the objects for which an evaluation of diversity is required. This is the main reason why the search for the most "natural" taxonomic system in natural history and biological sciences has been a hot topic since Linnaeus, if not since Aristotle. In ecology (and more recently in spatial and landscape ecology, as well as in environmental science and environmental management), landscape heterogeneity and habitat diversity have become of paramount importance for measuring and evaluating chorological relationships adequately; that is the study of the spatial distribution of organisms. For example, environmental quality evaluation procedures, such as bioassessment and biocriteria (US-EPA 1998, 2000), are based on the concept of biological and habitat diversity. While a robust taxonomy for organisms (i.e. biological diversity) has been developed and continuously updated for centuries, there are still many habitat classification schemes and they are not yet harmonised among countries and disciplines. For wetlands, reaching a consensus about habitat classification is further complicated by the multifaceted nature ofthe coastal transitional ecosystems with which wetlands are associated.

This chapter presents a few concepts regarding biological diversity and habitat diversity. Although the focus is on wetland environments and ecology, the ideas presented here are generally applicable to other environments and disciplines.

The universal idea of diversity is based on the two basic concepts of *objects* and *classes*. Objects belong to, or rather, are assigned to, different classes on the basis of some common attribute(s). Diversity evaluates and measures how many classes are present and how objects are distributed among them. Specifically, a high diversity is obtained when objects are evenly distributed among a high number of classes. For organisms, classes can be taxonomic levels such as species, genera, and families or other kind of groupings such as functional or trophic guilds. Objects are usually quantified by counts (i.e. number of individuals), units of biomass, or percentage area coverage. In this way, species diversity is often defined as an integrated measure of the number of species (classes) present in an assemblage (i.e. the specific definition for species richness) and of the way in which individuals (objects) are distributed among the species (i.e. the definition for species evenness). An underlying assumption is that the classes are comparable which, in this particular case, means that all species represent equal levels of biological differentiation.

Methods and indices to measure diversity are widely available in the scientific literature (e.g. Washington, 1984; Magurran, 1988; Farina, 1998), but the more complex the object of the "diversity assessment", the more accurate the definitions of classes and objects need to be. The identification of objects and classes is quite straightforward for the biological compartment at the level of species, but it becomes increasingly difficult at higher levels of ecological organisation such as ecosystems, habitats and landscapes. At these levels, the identification of classes and objects is usually process- and scale-dependent, thus becoming a very complex task involving a high degree of discretion by the scientist (note, however, that species diversity is also affected by the spatial scale of analysis because rare species show a highly patchy distribution of individuals over a given territory, so that the number of observed species in a given study depends on the size of the area considered by the study). Examples of habitat classes include habitat types derived from classification schemes (see Section 4 below) or types of land use. The corresponding relevant objects can be the areas of each habitat type (e.g. square kilometres of saltmarsh).

These few basic concepts demonstrate that the exact identification of classes and objects through operational definitions, and of their spatial and temporal domain of application is a prerequisite to any measure of diversity. Using the words of Valiela (2001): *"Any thing or idea that is not operationally definable is not accessible to study by empirical science. All terms used in descriptions or explanations must be specifically and realistically defined. This makes it possible for someone else to understand explicitly what the facts are, and perhaps carry outfurther tests of the facts or explanations. Operational definitions make ideas testable."*

The development of definitions and terminology should start from a set of attributes which experts consider fundamental for identifying the main features of relevant objects and classes. Where possible and appropriate, attributes should be hierarchically ranked and linked among them within conceptual schemes. Definitions have to be informative (i.e. capture the main features of the items they aim to describe) and concise (i.e. using the minimum number of attributes to identify any item). They should also be etymologically correct and unambiguous (e.g. avoid using terms with a variety of meanings within or across disciplines). Semantically correct terms can sometimes be rejected by the scientific community when they are associated with not-yet fully-accepted theories. A large consensus, and not necessarily only among scientists, is obviously one of the keys for the success of terms and definitions and it can be reached only through large and worldwide participation in the definitional processes (see Tagliapietra *et al.* (2009) for examples relevant to estuaries, lagoons and associated environments).

Defining "biological diversity"

Although biological diversity (or biodiversity) and species diversity are often considered synonyms, the former is much more inclusive of ecological, spatial and temporal relationships among organisms and their habitats (Hamilton, 2005). The International Convention on Biological Diversity (UNCED, 1992) provided the following definition: " *'Biological diversity' [biodiversity] means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems".* This definition emphasizes that biodiversity is a multi-scale concept applicable from the lower levels of biological complexity (e.g. organisms or even sub-organism levels such as genes) to the highest (e.g. habitat or ecosystem diversity). The more popular concept of biodiversity as amea-sure of diversity obtained from a given set of living organisms, at a given site and at a given moment, is included in the general definition. In stricter terms, this type ofmeasurements define a biological community or assemblage. However, the concept of biological community can itself be defined in different ways (Box 1) which, in turn, will determine what "diversity" is being measured. When trying to measure biodiversity, it is therefore critical to state what is actually being measured and how spatial and temporal contexts are defined.

Defining "habitat diversity"

As for the "biological community", the term "habitat" is not always clearly and univocally defined. Definitions of habitat were originally established for a specific organism or population, but were then extended to entire biocoenoses (see Box 2). Mitchell (2005) published a critique on the concept of habitat to illustrate the three main concepts that scientists try to convey with it: where an organism lives, the environment in its physical and chemical aspects, and a concept of community. Mitchell's work is a good example of the problems that can be encountered when trying to define suitable *attributes* for objects and classes.

The Water Framework Directive (2000/60/EC) uses the term habitat just with reference to special areas of conservation according to the Habitat Directive (92/43/EEC). The related "Interpretation Manual of European Union Habitats" (EC, 2007) gives definitions of habitat types in a pragmatic, descriptive way. However, the Habitat Directive (and its official interpretations) do not use a hierarchical, nested system. For example habitat types, such as coastal lagoons, tidal mudflats and *Salicornia* meadows, are presented at the same level, whereas in the real world they are regularly nested. This and other analogous issues in defining an appropriate scale to study habitat diversity are discussed more in depth in the next section.

Box 1 The most widely known definition of biological community and biocoenosis (listed in order of appearance in the scientific literature)

"Every oyster-bed is thus, to a certain degree, a community of living beings, a collection of species, and a massing of individuals, which find here everything necessary for their growth and continuance.f...] I propose produces changes in other factors" (Möbius, 1877).

"The *collection of organisms or species inhabiting an area. This term usually implies particular known or hypothesized interactions between the organisms (as opposed to Assemblage)"* (Cooke, 1984).

"A *group of population of plants and animals in a given place'* (Krebs, 1985).

"A tightly structured arrangement consisting of many types of organisms at different trophic levels" (Underwood, 1986).

"An *organized body of individuals in a specific location'* (Southwood, 1988).

"An *assemblage of species populations which occur in space and time" (Begon et al., 1990).*

"A *tightly structured, organised group of organisms at different trophic levels, co-occurring in space and time. Contrary to the term Assemblage, a community implies the existence of known or assumed relations between the organisms"* (U.N. Atlas of the Oceans, 2009).

Habitat and habitat scales

Section 3.2 described how the concept of habitat is very difficult to define univocally. Even when there is an agreement on what is meant by habitat (e.g. "Plant and animal communities as the characterising elements of the biotic environment, together with abiotic factors" - see Box 2), the identification of the spatial scale of analysis must also be considered. A habitat can be a lagoon as well as a saltmarsh, which is contained in a lagoon. However, homogeneous land units identified at the lagoon scale and at the saltmarsh scale are different; meaning that the resulting measures of habitat diversity cannot be compared. For this reason, many habitat classification systems report hierarchical, nested schemes of units which can be recognized at different spatial scales.

The general structure summarized below (with indicative size ranges for each level) is derived from a series of habitat classification systems conceived for coastal, wetland and transitional ecosystems (Allee *et al.,* 2000; Roff and

Taylor, 2000; Madley *et al.*, 2002; Roff et al., 2003; Connor *et al.*, 2004; Madden *et al.*, 2005):

Biogeographical districts: determined by bioclimatic factors and with surface areas generally ranging from 10^4 to 10° km^2.

Physiographic types: identified on the basis of geomorphology and general hydrology and with an area range of about 10 to 10^3 km^2. Environments ranging from

Box 2 Selected definitions of habitat (listed in order of appearance in the scientific literature)

"abstraction of the essential physical factors and of the essential co-inhabitant biota, in a locality where individuals of that population regularly live and reproduce' (Udvardy, 1959).

"the *place where an organism lives, or the place where one would go to find if* (Odum, 1971).

"environment in its physical and chemical aspects" that it is "usually conceived as the range of environments or communities over which a species occurs" (Whittaker *et* al., 1973).

"Plant and animal communities as the characterising elements of the biotic environment, together with abiotic factors (soil, climate, water availability and quality, and others), operating together at a particular scale" (EUNIS, 2009).

Note that the last definition is more general and puts less emphasis on a given species, population or community compared with the other, more "classical" definitions.

wave-dominated to tide-dominated estuaries and coastal lagoons (identifiable on the basis of wave, tide, and river energy) are defined at this level.

Hydrogeological zones (or geomorphic zones, Roy *et al.*, 2001): identified on the basis of local hydrodynamic patterns such as residence time, salinity and substrate type. Their approximate surface range is about 1 to 10^2 km^2. Examples of hydrogeological zones are marine tidal deltas, central mud basins, fluvial deltas, and bayhead estuaries.

Hydrogeological facies: can be delineated inside the hydrogeological zones and include structures such as salt marshes and mudflats. Their approximate surface range is 10^2 to 10 km^2. Hydrogeological facies provide the primary substrate for biocoenoses.

Bioformed habitats (vegetation and bioconstructors): often structured by vegetation that shapes the three-dimensional architecture of the local substrate and the general ecological conditions. These habitats can also be structured by animals (the so-called bioconstructors, such as oyster reefs and polychaete reefs) able to build hard substrates that serve for other living organisms. The spatial extension of bioconstructors can be very limited (in the order of tens of square metres in wetlands), but their role is very important. The extension of bioformed habitats typically varies from 10 ^{3} to 1 km^2, although it can be much larger in marine environments (e.g. coral reefs).

Biotopes: physical environment surrounding specific communities or characteristic taxa. The overall biological communities are often termed "biocoenoses" and the physical surroundings "biotope"; while, more specifically, animal assemblages are termed "zoocoenoses" and the physical environment "zootope".

This hierarchical structure shows that biological features become more and more relevant in habitat classification as the spatial scale becomes finer. The first three levels are developed mostly on physical features, whereas the last two levels are developed on the basis ofbiotic variables. Costello (200°), commenting on the EUNIS system (Davies *et al.*, 2004), noted that higher levels of the classification are more stable than the lower two because higher levels of organisation are less dynamic and more definable on the basis of the physical driving forces.

Relationships between biological diversity and habitat diversity

There are many theories about which forces create and maintain biological diversity. Palmer (1994) listed 120 published hypotheses formulated to explain coexistence or variation in species richness, many of them involving physical relationships with the habitat. Some of these hypotheses are of particular interest in understanding the relationships between biological diversity and habitat diversity.

The "habitat heterogeneity hypothesis" (e.g. Simpson, 1949; MacArthur and MacArthur, 1961; MacArthur and Wilson, 1967; Connor and McCoy, 1979) states that an increase in the number of habitats and/or, at a different scale, an increase in their structural complexity leads to an increased species diversity. A larger number of (micro)habitats practically means a larger number of niches exploitable by different species (Tews *et al.*, 2004). Differentiation of habitats would lead to an increase in p (beta) diversity, which is a measure of biological diversity between habitats, enhancing, in turn, the y (gamma) diversity at a regional level (Whittaker, 1972; Magurran, 1988; Mumby, 2001). Once again, the actual measure of diversity depends on the spatial scale adopted during a given study. Different taxa perceive differently scaled characters of the environment for every functional trait and life stage, so that there is a need for community studies to include several scales of analysis (Gonzales-Megias *et al.*, 2007). The Habitat Heterogeneity hypothesis can partly explain biological diversity in wetlands. However, it is a mechanism needing environmental equilibrium and stability to show all its strength; two conditions which are rarely encountered in transitional environments.

An alternate mechanism, the "intermediate disturbance hypothesis" (Connell, 1978) postulates that an intermediate disturbance leads to a higher diversity. While highly-disturbed habitats can support only short-lived opportunistic species, a few well-adapted species can outcompete all the others in habitats with low levels ofdisturbance (optimal intermediate condition). Given that environments characterised by fast evolution such estuaries and lagoons often show high diversity compared with other habitats, the intermediate disturbance hypothesis is probably more suitable to explain the relationship between biological and habitat diversity in these cases.

Conclusions

Definitions and scales are very important in any computation of diversity. This short review has illustrated how the measure of diversity depends on the way objects are defined and assigned to classes. Definitions, in turn, depend on personal perception and knowledge of the world. Fundamental ecological units, such as biological communities, habitat and ecosystems, can be described in different ways according to the purpose of the definition and to the background of the definers. Spatial and temporal attributes are of particular importance in the definition of high-rank ecological units. In general, definitions tend to become less precise as the complexity of the subject increases and the associated spatial and temporal scales widen. The concepts illustrated in this review apply to all ecosystems, but are particularly relevant for wetlands and transitional environments which, by their very nature, are highly prone to changes and can be studied in many ways.

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16. Conservation, restoration, and effects of climate change on wetlands

Wetlands are unique ecosystems that exhibit permanent or regular inundation or saturation with fresh or saline water. Marginal wetlands are created at the edges of rivers, lakes and the sea and are characterised by rooted vegetation that is permanently, or occasionally, partially submerged. Other wetlands, such as bogs, swamps and marshes, exist without direct connection to a large water body and depend on groundwater and precipitation. All types of wetland are usually dominated by vegetation that has adapted to survive with its roots in saturated soil conditions.

Wetlands serve many important hydrological and biological functions in the natural environment and provide many essential services to human communities and yet they have been degraded and even destroyed by human activities for decades. The importance of wetlands was recognised by the Ramsar Convention on Wetlands signed in Iran in 1971 (UNESCO 1994). This intergovernmental treaty provides a framework for wise use and conservation of all types of wetlands. The Convention currently has 159 contracting parties and 1,886 designated wetlands (Ramsar Convention Secretariat, 2010).

All wetlands are intricately linked with the hydrological cycle and thus they may be vulnerable to some of the potential impacts of global climate change, such as increased flood and drought events or storm surges. In addition, wetland vegetation can make an important contribution to the sequestration of carbon, thereby contributing to the mitigation of climate change. The need to consider conservation and restoration activities will be, therefore, even more important in the future.

Benefits from wetlands

Wetlands are an essential component of the natural environment helping to maintain a natural balance in hydrological systems and their associated habitats. In addition, they provide many additional benefits to human communities, such as food (e.g. fisheries) and fibre (e.g. from reeds).

Other benefits include habitats for rare and endangered species, natural water purification, overwintering habitats for waterfowl, flood control and many more (Barbier *et al.*, 1997). An initial attempt at placing a monetary value on the functions of natural ecosystems, including wetlands, was carried out by Constanza *et al.* (1997). As an example, the potential benefits from swamps and floodplains were estimated to have a value of US$ 19,580 per hectare per year and the value of the goods and services arising from coastal areas and inland wetland ecosystems combined was approximately 4°% of all ecosystems worldwide. It is mainly as a result of appreciating the role and potential value of wetlands that many restoration and conservation programmes have been initiated.

Being able to place a value on the benefits from a particular wetland often strengthens the case for conservation and even restoration and makes the need for such actions more apparent to managers and the public. There is a need therefore to assign values to a wetland before any degradation occurs. It can be particularly difficult to assign monetary values to services such as aesthetic beauty or recreational use and the present lack of scientific understanding of wetlands is tending to lead to undervaluation of many benefits. However, the role of wetlands in maintaining fisheries is fairly well understood. As an example, IWWR (2003) argue that approximately 75 per cent of commercially harvested fish and shellfish in the USA are dependent on estuaries and their associated wetlands. In the year 2000, commercial fisheries were worth US$3.5 billion and thus any loss of wetlands could result in a substantial economic loss to the fishing industry. In 199°, recreational anglers in the USA spent US$38 billion fishing for species that are dependent to some degree on wetlands, often for spawning (IWWR, 2003).

Climate change and other pressures on wetlands

Wetlands are dynamic ecosystems that would normally adapt slowly to changing environmental conditions such as precipitation and sea level. Over time many wetlands would undergo a natural succession where some areas would become terrestrial systems, while other new wetland areas would be created. However, over the last few centuries vast areas of wetland have been lost as a result of human activities, particularly drainage and reclamation to create agricultural land. It has been estimated that more than 54 per cent of the original 215 million acres of wetlands in the USA were lost over the 200 year period 1780s to 1980s (Dahl 1990, cited in IWWR, 2003) and between 1637 and 1954 an estimated 3,380 km^2 of wetland in the UK East Anglian fens was reduced to about 10 km^2 (Wade and Lopez-Gunn, 1999).

Wetlands all over the world are still under threat from a range of human activities and are being degraded or lost altogether (Moser *et al,* 199°). Direct threats include draining and filling for land reclamation and indirect threats include impacts from water regulation activities, such as groundwater extraction, impoundment of rivers, and coastal development. In addition, wetlands are vulnerable to invasive species and both point and diffuse sources of pollution.

It is likely that the vulnerability of wetlands from past and present human activities will be compounded by the impacts of global climate change. Anticipated changes in rainfall duration and intensity will affect local hydrological regimes which, in turn, will affect both inland and coastal wetlands. The timing and duration of inundation with freshwaters may change and saline regimes may be altered. These, together with future temperature changes, will affect species distributions. Coastal wetlands may also be at additional risk from sea level rise and from the impact of more intense storms and tidal surges. Nicholls *et al.* (1999) predict that the effects of sea-level rise combined with other human-related pressures could result in the loss of 70 per cent of coastal wetlands by the 2080s. The impacts of the combined pressures of climate change and human activities are likely to vary regionally and even locally (Ramsar Convention, 2002a). However, on a global scale, the actual impacts of climate change on water resources would probbly be rather small in the short term (i.e. less than 20 years) relative to the impacts of other human activities (Arnell and Chunzen, 2001, cited in Ramsar Convention, 2002a).

The principal effects on inland and coastal wetlands that may be associated with climate change are summarised in Table 1. Within Europe the likely impacts of climate change will differ from region to region, with some regions suffering from increased flooding in winter and others experiencing a greater difference between summer and winter flows. There are also expected to be differences in water temperature. Where water temperature increases are coupled with decreased summer flows there could be deterioration in water quality, especially as a result of reduced dissolved oxygen concentrations. This will have impacts on the fauna and flora of wetland systems, such as a gradual change from salmonid communities to cyprinid and percid communities in the north and west of Europe. Migratory birds may also be affected by the possible loss, or change in ecosystem structure, of the isolated inland wetlands of central and southern Europe on which they depend for food during their migration (Ramsar Convention 2002a).

European coastal deltas, estuaries and salt marshes are likely to be threatened by sea level rise, particularly in the Mediterranean, Black and Baltic seas where the tidal range is low (< 1 m). The intensive development along much of these coast lines has reduced the capacity of wetlands to adapt naturally to any changes (Ramsar Convention, 2002a). Evidence also exists for changes in the distribution of estuarine birds in the UK as a result of changing patterns of minimum temperatures along the coasts (Austin and Rehfisch, 2005). There are approximately 3,000 km^2 of salt marshes and more than °,500 km^2 of other unvegetated intertidal habitat within Europe, much of which has been designated under the Ramsar Convention. The loss of these habitats, together with mangrove swamps, has been estimated to reach 31-100 per cent for the Mediterranean, 84-90 per cent for the Baltic and 0-17 per cent for the Atlantic coasts (Ramsar Convention, 2002a). The associated loss of habitats and feeding areas could have significant impacts on fish, wildlife and birds.

Conservation and restoration

Whether as a result of human activities or climate change, impacts on wetlands need to be reduced and preferably prevented. In some cases it may be possible to restore degraded wetlands to a condition that closely resembles the previous habitat. All such activities require financial and human resources, as well as the participation and involvement of local communities and stakeholders.

Conservation is the long-term preservation and protection of the functions and values of wetlands and should involve a combination of activities, including legislation, development of wetland policy, agreements and treaties, stakeholder and landowner involvement and volunteer projects (Canadian Wildlife Service, 2002). Designation of reserves and protected areas form an important part of conservation activities. An example of a wetland conservation plan is Lake Engure in Latvia, where the lakeshore meadow vegetation was being taken over by willows and reeds and damaged by other human activities. The undesirable vegetation succession in this Ramsar site is being controlled by regular cutting of the reeds and grazing the meadow with traditional domestic herbivores such as horses and cattle. Undesirable economic activities are also being prohibited in the area (Racinska, 2004). Approaches to conservation of coastal wetlands could involve removing sea defences so that wetland habitats re-establish themselves naturally landwards to replace areas lost through sea-level rise.

Restoration is the process of returning a degraded wetland (rehabilitation) or former wetland (re-establishment) to its pre-existing condition, or as close to that condition as possible. Restoration often involves activities such as construction work and re-planting in order to restore the hydrological regimes and biological communities. Knowledge of the hydrological regime is often lacking or poorly understood. For a restoration project to succeed it must consider (Canadian Wildlife Service, 2002):

The position of the wetland in its watershed,

Natural sources of seed in the area that will allow recolonisation,

The hydrological connection between the wetland and its surrounding water table,

The wetland sediment, and the water level fluctuations that may be required to support the new wetland vegetation.

There are two approaches to restoration: passive and active. Passive restoration fundamentally involves removing the cause of degradation and allowing the wetland to regenerate naturally. This approach is only likely to work if there are other water and wetland species nearby that will gradually colonise the area being restored. The approach is low cost and the final outcome is a wetland that is likely to resemble the surrounding environment. By contrast, active restoration usually involves considerable design work and the building of structures such as weirs and culverts and intensive planting with native species, combined with substrate creation and control of invasive species (IWWR, 2003). Such methods can be expensive and labour intensive. Examples of methods suitable for different wetland problems are given in Table 2 and further information is available in the many guidebooks available detailing methods and approaches to conservation and restoration in different wetland habitats (Payne, 1992; Galatowitsch and van der Valk, 1994; USEPA 1994, 2000; Zedler, 2000).

If active restoration is likely to involve engineering work, it is recommended that bioengineering or "soft" engineering solutions should be used rather than the traditional "hard" engineering approaches that replace natural ecosystem functions with designed and constructed structures (IWWR, 2003). Bioengineering is more natural and is often more economical. Examples of soft engineering solutions to stream bank erosion are (IWWR, 2003):

Planting native species, such as willows that are fast growing. Using logs to stabilise banks. The logs decompose over time. Using geotextile materials to stabilise banks. These materials can be covered in soil and roots can penetratethrough them.

Conservation or restoration activities can be expensive and may involve the co-operation of many groups of stakeholders. Therefore an important prerequisite to any potential project is an evaluation of the importance of the wetland and of the various benefits and goods that it supplies. This can be achieved through an assessment of the existing extent and condition of the wetland combined with a valuation of its anticipated environmental benefits (Canadian Wildlife Service, 2001). The key questions to be asked as part of an assessment of the feasibility and potential usefulness of any restoration project are (Ramsar Convention, 2002b):

Will there be environmental benefits (e.g. improved water quantity, reduced eutrophication, biodiversity conservation, flood control)?

What is the cost effectiveness of the proposed rehabilitation?

What options, advantages or disadvantages will the restored area provide for local people and the region?

What is the present and possible future ecological status of the project?

What is the status of the area in terms of present land use?

What are the main socio-economic constraints?

What are the main technical constraints?

Assessment of the extent and current status of any wetland prior to commencing a restoration project involves surveying and monitoring techniques, which may be either qualitative or quantitative. Typically assessment would include determination of: hydrological regime, topography and evidence of erosion, vegetation patterns, use by wildlife especially birds, structural developments and adjacent land use. An important aspect ofthe implementation of any restoration project is the continuation of monitoring and assessment activities in order to determine whether actions are achieving the desired objectives. Monitoring should be based on careful selection of appropriate indicators that are related to the specific objectives of the restoration project over relevant spatial or temporal scales, such as changes in breeding bird populaions, changes in areal extent of specific plant species or communities, species diversity, water quality and sediment accretion or erosion. For example, if the objective is to improve the habitat for migratory waterfowl, monitoring should include bird counts, nest sites and the number of juveniles of key species. Success of the project can be measured against a performance standard or target, such as a specified number of breeding pairs of key species that are expected to use the site once restoration is complete. An example of a restoration project with objectives and monitoring programme outline is given in Box 1. A detailed framework for creating inventories, monitoring and assessing wetlands has been produced by the Ramsar Convention (Ramsar Convention Secretariat, 2007).

Box 1 Example of restoration project objectives, targets and associated monitoring: Stevens Creek Tidal Marsh, California

Goal:

Restore deep pit with ponds to salt marsh in order to provide vegetated tidal marsh habitat for rare species

Objectives:

Restore natural tidal influence

Return mudflat to appropriate level for vegetation

Re-establish native tidal salt marsh vegetation

Targets:

Re-establish tidal influence

Develop mudflat on 50 percent of the site within three years, at an elevation available to vegetation

Restore native salt marsh vegetation on 50 per cent of the site within five years

Monitoring and assessment:

Automatic tide gauge measurement, interpreted by a hydrologist ^ Annual quantitative measurements of mudflat elevation by a qualified surveyor ^ Annual vegetation survey along transects on the ground together with aerial photographs interpreted by an ecologist ^ Annual aerial photographs of channel formation, interpreted by a hydrologist ^ Qualitative observations on tidal flow and the presence of non-native species

Source: IWWR (2003)

Conclusions

Wetlands are being degraded and destroyed by many human activities, despite providing a wide range of important and valuable goods and services. The economic value of these habitats has only recently been appreciated and has highlighted the necessity of conserving and restoring them. The impacts of climate change, specifically the changes in hydrological regimes, that are expected to arise from increased rainfall and sea-level rise, may contribute to the degradation and loss of wetlands in some world regions. Conservation and restoration projects are necessary to preserve these valuable habitats but they require financial and technical resources, as well as stakeholder and community participation. With careful planning and implementation, followed by monitoring and assessment, restoration projects can be extremely successful and can lead to increased social and economic benefits for local communities.

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17. Impact and management of reservoirs

At the end of the 20th century, approximately 45,000 large dams (i.e. dams higher than 15 m) with a total reservoir surface of about 500,000 km^2 have been exploited in the world, mainly for irrigation, hydroelectricity and as drinking water reservoirs (Gleick, 1998; WCD, 2000). They provide a major contribution to the economic development of industrial as well as of developing and rural countries, and are often considered as sustainable, e.g. "sustainable hydropower" or "green hydropower" (Truffer *et al.*, 2003, Bratrich *et al.*, 2004).

Before construction of reservoir infrastructures, dam projects are generally evaluated according to the latest developments in engineering standards with respect to floods, seismic hazards and other causes of failure, such as overtopping of the dam, foundation defects, and karst or slope instability (Mouvet etal., 2001; BWG, 2002; ASDSO, 2010). Also, events such as the Sichuan earthquake in May 2008, and the fact that no single dam collapsed during this quake (although many of them have suffered damage), plainly support this practice (Wieland, 2008). However, this should not obscure the fact that other hazards are often not considered adequately, and that dams and reservoirs often do not meet the high standards necessary for environmental sustainability and human health. Truffer *et al.* (2003) comment that "hydropower construction and operation is associated with a number of serious environmental problems: water diversion, interruption of fish migration, hydropeaking, reservoir flushing and inundation of landscapes, and alteration in bio-geochemical cycling (Friedl and Wuest, 2002)." This chapter briefly reviews a large range of potential impacts and hazards linked to the exploitation of dams and reservoirs, such as:

Impact on the hydrological system and river dynamics

Impact on physico-chemical cycles

Impact on soil, surface and groundwater quality and environmental toxicity

Impact on ecosystems

End-of life planning

Although of major concern, issues associated with the socio-economic and landscapes impacts are not considered here.

Key reservoir processes

According to their morphology, capacity and runoff, reservoirs are either close to river or to lake systems. Depending on their use, the water level of reservoirs may be either more or less stable, or it may fluctuate according to a periodicity linked to the water cycle and its exploitation - from full to empty in the extreme case. River dams and reservoirs slow down the runoff, and point-bar sedimentation may lead to a meander-type, mature river morphology. Storage reservoirs are artificial lake systems, characterized by lake processes. River and storage reservoirs are mostly established in a morphological context that differs from natural lake morphologies, with the deepest part of the lake close to the dam. Table 1 summarizes some fundamental reservoir processes that influence the hydrological system. Most of these are lake processes (Wetzel, 2001; Kalff, 2002).

Impact on the hydrological system and river dynamics

Major changes in the river system occur upstream and downstream. River diversion may occur upstream in order to bring more water into the reservoir and increase its capacity. This measure may partially dehydrate the landscapes, as is the case in alpine valleys, and it may also influence the surface and the groundwater systems. As a general rule, groundwater levels decrease in areas with diverted rivers and increase in the areas close to, and downstream of, reservoirs. Rivers may be diverted downstream for hydro-electric purposes (e.g. gain of elevation and thus pressure), for navigation or irrigation. According to the mode of management, irrigation improves agriculture, but may also increase soil salinisation (Cause, 2001). In the case of reservoir contamination, either chemical or microbiological, impacts on human health may be expected.

Downstream of a reservoir, river dynamics may suffer major modifications, particularly a change in the distribution of runoff. This is caused by the water storage and the exploitation of the reservoir volume for irrigation, production of electricity or other purposes. The reduction of the floods of the River Nile is a well known example. In the alpine region, Loizeau and Dominik (2000) have shown the potential impact of reservoir exploitation in the drainage basin of the Rhone River on the oxygenation of Lake Geneva. The downstream effects of flushing a reservoir are inundation, massive sediment transport and damage to the ecological communities. There may also be an increase in the erosive capacity of the river, linked to the decrease in sediment charge as a result of reservoir sedimentation. The erosion may affect the river banks (Palmieri *et al.*, 2001), the river bed itself and any constructions, such as the basement of bridges (Doutriaux, 2006).

Impact on physico-chemical cycles

According to WCD (2000), 46 per cent of the water in the 108 most important rivers of the world is first flowing into a reservoir before it continues its way to a natural lake and/or to the sea. The efficiency of reservoirs at trapping sediment is frequently reported as 70 -90 per cent of the sediment volume delivered from the watershed (Sundborg, 1992; Toniolo and Schultz, 2005) (Table 17.1).

Table 17.1. Main reservoir processes influencing the environment

Reservoir process and parameters	Impact on hydrological system downstream
Water storage and exploitation (main reservoir function	
Formation of an artificial lake or slow-flowing river, sometimes with important variations in water level	Modification of hydrological regime (hydroflushing, flood, low water level, runoff distribution in general) Change of groundwater levels High concentrations of suspended material during flushing events
Detrital (terrigenous) sedimentation	
Trapping of suspended matter and bed load	Decrease of down-river sediment transport
Accumulation and retention of organic matter, adsorbed nutrients and contaminants in the reservoir	Decrease of down-river flux of nutrients and contaminants
Reservoir bed colmatation and reduction of groundwater infiltration	Decrease of groundwater level
Authigenous production, sedimentation and degradation	
Consumption of nutrients and major elements	Decrease of nutrient and major element concentrations, depletion of dissolved oxygen
Increase of primary production	Increase of DOC in downstream water
Oxygen depletion in deep water (associated with eutrophication)	
Temperate cycles, lake stratification and heat budgets	
Change of temperature profile (and temperature linked physical parameters) as a function of meteorology and seasonality	Change of down-river water temperature and viscosity
Impact on biochemical cycles, e.g. oxygen depletion due to ice	Modification of physico-chemical water parameters
Evaporation	
Increase of reservoir water salinity due to evaporation	Increase of river and groundwater salinity
Phosphorus cycle	
Phosphorus cycling between water column, organisms and sediments, mainly linked to the trophic chain. Contribution to organic production(phytoplankton and plant growth)	Reduction of phosphorus, downstream and in the groundwater
Nitrogen cycle	
Nitrogen cycling (mostly nitrification) between atmosphere, water column, organisms(e.g. macrophytes) and sediment. N-loss to atmosphere (NO_2, N_2). Oxygen depletion by nitrification	Improvement of water quality downstream
Other cycles: Iron, manganese, sulphur, silica	
	Changing upstream to downstream water composition

As a result, approximately 30 to 40 per cent of suspended matter transported by the world river network is no longer reaching the coasts of seas, oceans and some major lakes, but is retained in man-made reservoirs, at least for the lifetime of these infrastructures (Vorosmarty *et al.*, 1997). Retention of water, and therefore of dissolved matter, is shorter but also has to be considered. The ability of a reservoir to trap solid matter is a key process, which has many other environmental implications. The main impacts are:

Changes in the sediment budgets of deltas and coast lines. Many river deltas on oceans and lakes suffer a decrease of sediment input, and some deltas have passed from progression of coast lines to regression (e.g. Malini and Rao, 2004). Furthermore, tectonic subsidence and eustatic sea-level rise may no longer be compensated by sedimentation on the delta plain, and the sea is therefore transgressing on the delta, increasing the effects of current sea-level rise due to global warming.

Change in geochemical cycles by storage of contaminants, nutrients and major elements (see below).

No further contribution to flood plain siltation and fertilization.

Change in river dynamics and erosional capacity.

The "trapping efficiency" of reservoirs (Taher-Shami and Tabatabai, 2000; Verstraeten and Poesen, 2000) depends on a wide variety of parameters, and decreases with continuing sedimentation. The capacity of reservoirs is reduced by a mean of 1 per cent per year by sedimentation (WCD, 2000).

Sediment flushing generally allows the export of sediments along erosional channels within the reservoir during lowering of the reservoir level. Flushing efficiency may vary according to the basin topography, compaction of sediments and other parameters and, in some cases, may not prevent the gradual filling of a reservoir with sediment. Therefore, in order to maintain water storage capacity it may be necessary to use other strategies, such as dredging or elevation of the dam (Vischer and Hager, 1998).

After deposition, fine-grained reservoir sediments may be compacted, mainly by burial. Sediment diagenesis may reduce the organic matter content by bacterial degradation linked to methanogenesis, oxidation and CO_2-production, and other processes. Sediment erosion within the reservoir depends on hydrodynamics, i.e. mainly on stream velocity (in river reservoirs) and thus on the river load.

Nutrients, contaminants and major element cycles

Particulate nutrients (organic matter and carbon, phosphorus, etc.), contaminants (metals and organic substances) and major elements such as silica, iron and sulphur are trapped with the sediments (Friedl *et al.*, 2004; Teoduru and Wernli, 2005). Depending on the physico-chemical conditions in the water column and within the sediments, elements may be remobilized and contribute to primary production. The following environmental hazards are of particular concern:

Remobilization of contaminants from sediments and their return to the trophic chain, either by sediment erosion, by uptake by organisms (fish, plants and sediment dwellers), or by infiltration of interstitial water from the sediment to the groundwater (Wildi *et al.,* 2003, 2004).

Reduction of nutrient inputs to coastal areas of oceans and seas, resulting in reduced primary production in offshore areas. This may affect the marine carbon balance (through limitation of production), and therefore climate change (Jaccard, 2008).

Carbon and oxygen cycles

Oxygen depletion by the oxidation of particulate and dissolved organic carbon may lead to deoxygenated or even anoxic conditions in deep water and in the sediment. The main hazards associated with this are:

Eutrophication of deep reservoir water and of water transferred down-stream or infiltrated to the groundwater.

Remobilization of nutrients, metals and organic contaminants from sediments and their availability to organisms (plants, sediment dwellers).

Infiltration of pore water charged with contaminants and dissolved organic carbon to the groundwater (Wildi *et al.,* 2003, 2004).

Impact on soil, surface and groundwater quality and environmental toxicity

There are two main impacts of dams and reservoirs on soil quality. Firstly, salinisation may occur in arid conditions in relation to irrigation, mainly due to the maintenance of a high ground-water level when evaporation and evapotranspiration are strong (Cause, 2001). In addition, contamination of soil in the floodplain by reworked contaminated reservoir sediments during floods may be expected. This mechanism is linked to the accumulation of contaminants in reservoirs (Justrich *et al.,* 2006).

Several reservoir processes listed in Table 1 may influence surface water quality. During sedimentation along the reservoir axis, the coarse fraction (sand and coarse silt) diminishes in suspension, whereas the fine fraction (fine silt and clay) increases as a proportion of the total fraction. As a consequence, and because contaminants are mainly adsorbed onto the fine fraction, suspensions and sediments formed from particle settling at the outlet of a reservoir contain higher concentrations of contaminants and have a higher toxicity than suspensions and sediments at the inlet of the reservoir (Justrich *et al.,* 2006). Eutrophication can be caused mainly by nutrient release from the sediments, DOC-consumption and nitrification (see above). Stratification, mainly thermal stratification, can also occur in deep reservoirs and may result in oxygen depletion. Evaporation can also lead to an increase in the salinity of the water.

A major impact on surface water quality may stem from biological processes such as bacterial contamination due to the release of water from wastewater treatment plants (Pote *et al.*, 2008) and diffuse run-off. Also, in particular conditions, sediment contamination may contribute to the persistence of contamination in the overlying water (Pote *et al.*, 2009). Water turbidity can be increased by high plankton concentrations and there may be a proliferation of mosquitoes and other insects (leading to the risk of spreading malaria), as well as rats and other nuisances.

In the surrounding area and downstream of the reservoir, groundwater levels generally rise due to damming, increased infiltration and rise of the hydraulic base level. During the filling of the reservoir, the increased DOC-content of the infiltrate may lead to depletion of dissolved oxygen, which in turn can lead to an increase in ammonium, and remobilisation of iron and other substances, including contaminants. As a consequence, ground- and drinking water quality may be degraded substantially (AGS, 2009). In the long term, the reservoir bed will be sealed by the deposition of fine sediments, infiltration may decrease and the groundwater level may fall again. Long term impacts on groundwater quality may arise from frequent fluctuations in reservoir level due to exploitation of the water and from disruption of the sediment seal of the reservoir bed by flushing.

Increased environmental toxicity in reservoir environments may be due to sediment contamination by micropollutants, but also to anoxic conditions with high concentrations of sulphur (Pardos, 1996). As shown by Justrich *et al.* (2006), the increase in contaminant concentration caused by the sedimentation of coarse fractions along reservoirs may increase the sediment toxicity for sediment dwelling organisms.

Impact on ecosystems

Further ecological consequences of damming that have been mentioned in the literature are loss of forests and wildlife habitats, loss of species populations and the degradation of upstream catchment areas due to inundation, loss of aquatic biodiversity, including upstream and downstream fisheries, and loss of the services of downstream floodplains, wetlands, and riverine, estuarine and adjacent marine ecosystems. There can also be cumulative impacts on water quality, natural flooding and species composition and, where a number of dams are sited on the same river, there could be habitat loss (fragmentation, barriers to migration), changes in flow conditions, changes in sediment supply, and decreased nutrient delivery.

End-of life planning

Up to now, considering the time taken for reservoirs to fill with sediments, the mean life-time of dams worldwide would be approximately 100 years. According to Quinn (1999), when considering the situation in the USA

"*The average life expectancy of a dam is 50 years and more than 25% of the dams in the United States have now reached that threshold. In the next 20 years that percentage will rise to more than 80% (American Rivers, 1997). As these structures age and deteriorate, or no longer serve their intended purpose, decisions to repair or remove them are, by necessity, becoming increasingly common. The financial investment to rehabilitate a dam often far exceeds that needed to remove the structure without even factoring in future maintenance and liability insurance costs if the dam is left in place (Born et al. 1998; River Alliance of Wisconsin, 1999). These high repair costs have led many private entities and municipalities, who own over 90% of dams, to consider dam removal as an alternative to structural rehabilitation."* Although this situation is not comparable in all respects with other areas of the world, it shows the general problems of restoration and end-of life planning for dams and reservoirs where conditions have changed compared with those at the time of construction.

In the case of "dam *removal or alternative dam retention alternatives'*, Quinn (1999) mentions *"research components of the environmental assessment"*. According to the Ontario Ministry of Environment and Energy (1993), the key problems to be solved in the case of dam removal are:

Elimination and disposal or stabilization (Kanehl and Lyons, 1997) of sediments, in particular of contaminated sediments and sediments with a high organic matter content.

Sediment management from the former reservoir area.

Revegetation and other aspects of rehabilitation of the river system and river valley in general.

Management of the new ground- and dinking water situation.

In contrast to other major technological projects, such as the construction of nuclear power plants where relatively detailed plans for end-of-life and dismantling have to be presented, this is not the case for dams and reservoirs. Also, sediment management in the reservoir, removal, stabilization or other solutions may be difficult, risky and expensive operations with a significant environmental impact, and should have been already considered during the planning phase.

Sustainable dam and reservoir planning and reservoir management

As argued above, traditional dam and reservoir planning, construction and management may not be sustainable, and may have impacts on the environment in the long term. Thus hydroelectricity production, as well as irrigation or drinking water facilities, may only be available for decades or eventually for a time span of about a century. This is obviously not in agreement with current efforts towards renewable energy sources and sustainable development in general. Concepts should therefore be developed for the planning of facilities, and for their exploitation, that meet the conditions of sustainability. In order to reduce reservoir siltation, Palmieri *et al.* (2001) list the following sediment management methods:

1. Reduction of sediment yield by measures in the catchment area (soil conservation measures, correction of land-slides and accelerated erosion, reforestation, etc.), or by debris dams, which intercept coarse grained sediments in mountainous streams.
2. Sediment routing through off-stream reservoirs, construction of sediment exclusion structures, and by passing through the reservoir (e.g. sluicing).
3. Sediment flushing, whereby the flow velocities in a reservoir are increased to such an extent that deposited sediments are re-mobilized and transported through bottom outlets.
4. Sediment removal by mechanical dredging, or by hydraulic removal.

They also mention: "The results will always be site specific and no standardisation is possible apart from the basic principles on which the solution is based." (Palmieri *et al.,* 2001)

An example of a more sustainable design and reservoir management approach is presented here. The basic conditions to be fulfilled are:

Sediments have to remain in the fluvial cycle. Therefore, as much as possible of the sediment has to transit through the reservoir, instead of being deposited in the reservoir basin.

If sediment is deposited, it has to be in a hydrological and morphological position that facilitates remobilisation by "smooth" flushing operations, preferably during natural floods. In particular, the formation of delta-like sediment deposits at the influx of the river into the basin has to be avoided, because they are generally difficult to remove a posteriori.

A possible design of such a facility could be based on the following elements (see Fig. 17.1):

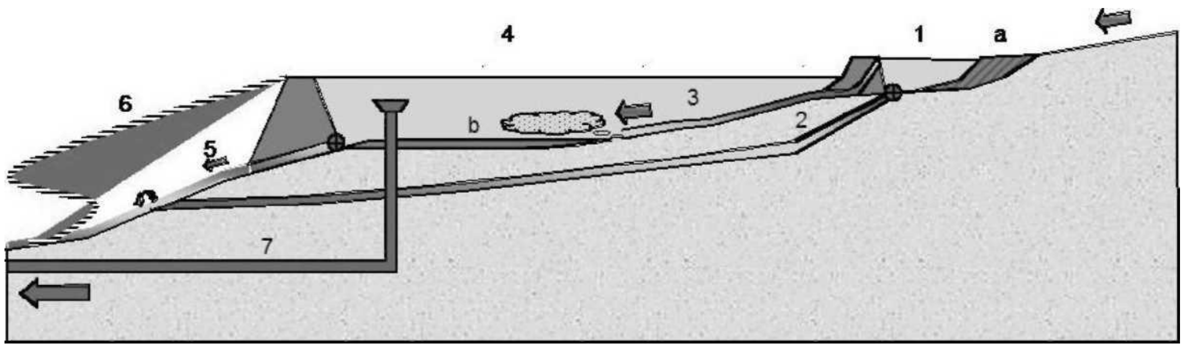

Fig. 17.1 Conceptual dam and reservoir, designed for sustainable sediment management. (1) sediment trap, (a) delta deposit; (2) Residual water and sediment evacuation conduit; (3) Water supply to the main reservoir; (4) Main reservoir, (b) reservoir sediments; (5) Bottom outlet of the dam; (6) Fish ladder; (7) Main conduit for hydroelectric or other

1. Sediment trap. This trap, combined with a "debris catcher", is intended to retain gravel and sand (a: delta body). Its geometry and dimensions have to be planned to ensure that evacuation through (2) is optimal. All the fine sand, silt and shale fractions will not be eliminated by this sediment trap.

2. Residual water and sediment evacuation conduit. This waterway has to provide the residual water downstream of the reservoir. Due to its steeper slope than the former natural valley floor, and because it is lined along its whole length, sediments may be evacuated, mainly during flood situations, without transition through the main reservoir and with a minimum of damage to the riverine ecosystem.

3. Water supply to the main reservoir. The main reservoir is fed by a conduit that starts in the surface waters of the water trap, without any gravel or course and medium sand. This conduit brings the water down into the main reservoir basin (4), in a position sufficiently close to the base outlet of the dam to facilitate re-erosion by "smooth" flushing operations. It may also contribute to the oxygen content of the deep water of the reservoir.

4. Main reservoir. The site of the main reservoir should be selected so that the final deep basin is characterized by simple morphology and a short basin axis, to facilitate flushing of the basin sediments.

5. Bottom outlet of the dam. This outlet is used, if necessary, in combination with other passive and/or active measures (e.g. sluicing) to evacuate fine sediments from the deep basin by flushing operations.

6. Fish ladder

7. Main conduit for hydroelectric or other reservoir use.

Combined with catchment area management and other measures as mentioned by Palmieri *et al.* (2001) (see above), this design should optimize the life expectancy of reservoirs and reduce the environmental impact on the river and stream valley.

For several aspects of sustainable reservoir management more efforts, and in particular research, are needed to improve the current solutions. The costs of reservoir planning, construction and management can also be expected to increase as sustainable solutions are developed.

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18. Potential environmental risks from sediment-bound trace elements: the Ukrainian part of the Danube Delta

Bottom sediments are well known sinks for many environmental contaminants, including trace elements such as As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn whose natural biogeochemical cycles have often been perturbed by anthropogenic activities (Forstner and Wittman, 1979; Salomons and Forstner, 1984). The study of sediment quality therefore provides valuable information on the possible risks originating from anthropogenic contamination of aquatic systems (Chapman, 1996) and several sediment quality guidelines have been proposed to link chemical measures of sediment contamination with expected biological effects (e.g. Burton, 2002; Wenning *et al.*, 2005). Evaluation of sediment quality is particularly advisable when major waterworks involving dredging of sediments are planned or undertaken. Such an evaluation provides a background against which to evaluate possible future changes in contamination patterns and to establish whether contaminated sediments are present in the areas concerned by the planned works. This chapter examines contamination by trace elements in the Ukrainian part of the Danube Delta using samples collected in May 2004. In this period, extensive parts of the river were dredged to allow the reopening of the Bystroe Deep Water Shipping Way.

Trace element contamination issues in the Bystroe Deep Water Shipping Way

Besides the issues discussed regarding the habitat disruption and physical disturbances connected with dredging works, another controversy about the re-opening of the Deep Water Shipping Way (DWSW) in the Ukrainian part of the Danube Delta included the possibility of remobilizing sediment-bound contaminants. Along with the possible transboundary effects of the DWSW, this volume), the ESPOO Commission (named after the Finnish city of Espoo where the convention was adopted) also studied the potential risks of sediment contamination by trace elements based on the Dutch criteria for dumping material at sea (ESPOO, 2006). Such criteria, which have legal value in The Netherlands, can be used to judge whether the material dredged at a certain location can be dumped at sea or if it needs to be disposed of as toxic waste. These types of criteria are primarily concerned with the possibility that, upon dredging and dumping, sediment-bound contaminants may desorb from particles and increase dissolved metal concentrations to levels harmful to biota. In the case of Danube sediments that were dredged in May 2004, the ESPOO expert concluded that the possibility of transboundary effects due to metal remobilization from dredged sediments was unlikely (ESPOO, 2006).

Criteria for dumping dredged material at sea are not appropriate as a means of evaluating whether the level of trace element contamination in sediments could harm the resident biota. The most appropriate way to evaluate this is to compare experimental results with available sediment quality guidelines (SQGs) (Burton, 2002). Sediment quality guidelines do not constitute enforceable environmental standards, but provide an indication of the possible risks associated with sediment contamination for biota and provide information for implementing long-term monitoring and maintenance work. Note, however, that many SQGs have been developed in North America and applying them to watersheds around the world should be done with caution.

Sediment-bound trace elements in the Ukrainian Danube Delta

Samples were collected at various sites in the Chilia branch in May 2004 (see Fig.18.1) and analyzed for selected trace elements (As, Cd, Cr, Cu, Hg, Ni, Pb, Zn). About 0.5 g wet sediment were digested with 5 ml concentrated nitric acid (VWR, Suprapur® quality) followed by analysis by ICP-MS (inductively coupled plasma mass spectroscopy). All samples were analyzed in duplicate. Sediment water content was determined by drying at 105°C for 48 hours and results for trace metal analysis were expressed on a dry weight basis. Total mercury was determined on dried aliquots using an AMA254 (automated advanced mercury analyzer).

Comparison of the measured concentrations with TEC and PEC values suggested that contamination by these two elements was widespread in the Ukrainian part of the Danube Delta. However, Oaie *et al.* (2005) reported that mafic and ultramafic rocks are present in the Danube river basin upstream of the delta. These types of rocks naturally contain high levels of Cr and Ni. Direct comparison with TEC and PEC values (which were derived mainly from data on sediments collected in the North American continent) could therefore be misleading. For samples collected from the continental shelf in the north-western Black Sea (which is influenced by Danube discharge), Secrieru and Secrieru (2002) also concluded that Ni levels in these sediments were essentially natural. These findings illustrate well the necessity of knowing regional background levels when assessing sediment contamination by trace elements (e.g. Matschullat *et al.*, 2000) and the need to integrate various investigations to reach adequate conclusions on sediment quality.

Comparison of measured concentrations with TECs or PECs for individual elements, albeit informative, does not account for the fact that sediments are simultaneously enriched in several trace elements. Use of the Sediment Quality Guideline Quotient (SQG-Qm) represents an effective way to

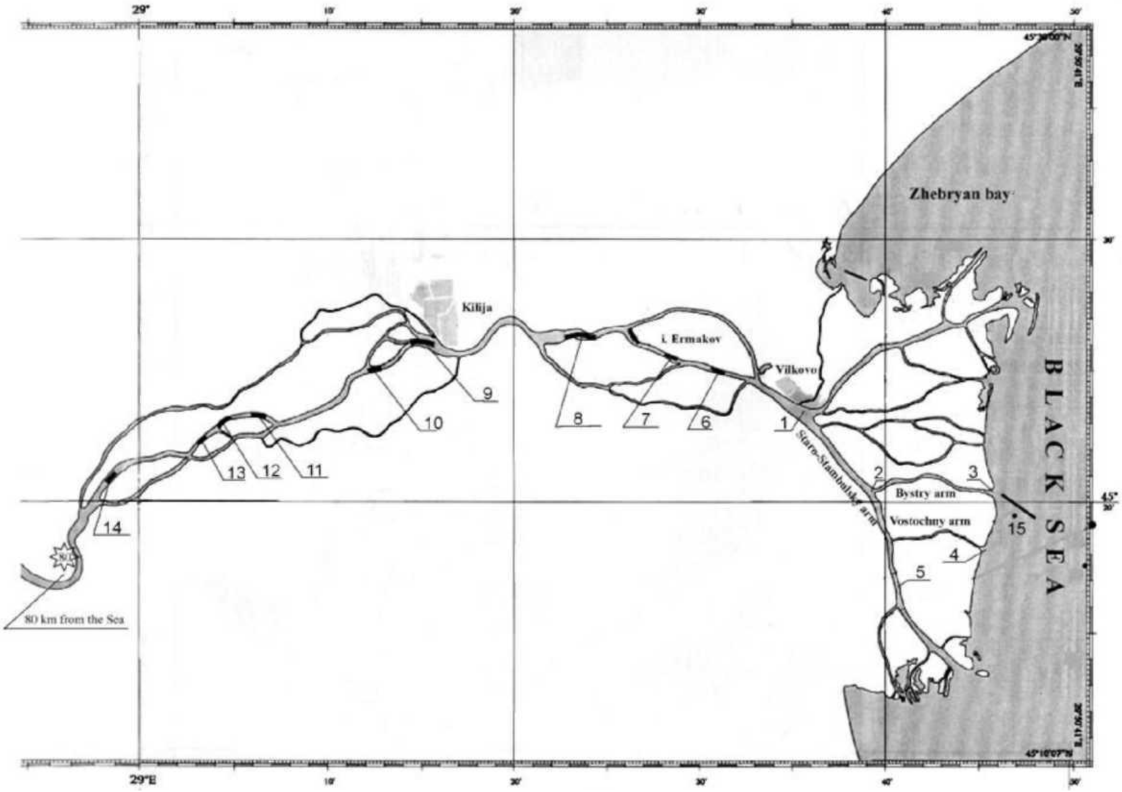

Fig. 18.1. Overview of the Ukrainian part of the Danube Delta (Chilia arm and secondary Staro-Stambulsky delta) and location of the sampling sites. The Chilia and the Staro-Stambulsky arm mark the border with Romania to the South.

account for the possible combined effects of several elements. Calculation of SQG-Qm can be performed using the following formula (Long *et al.,* 2006): Experimentally measured values were compared with relevant Threshold Effect Concentrations (TECs) and Probable Effect Concentrations (PECs) (MacDonald *et ah,* 2000). Threshold Effect Concentrations identify those concentrations below which a given element is not expected to represent a threat for the ecosystem under study, whereas PECs correspond to concentrations above which adverse effects linked to contamination by a given element are likely to happen. Concentrations between TECs and PECs are indicative ofsituations where further studies are needed.For the sediments collected in this study, only the PEC values of As and Ni were exceeded at some locations (Fig. 18.2); while TECs were exceeded on several occasions for various elements and at all sites for Ni. The results indicated that further investigations (e.g. observations of benthic community structure and functioning or ecotoxicological tests) were needed to assess sediment quality in this part of the Danube Delta. Particular caution had to be exercised in the interpretation of data for Cr and Ni.

In Table 18.1 **v**alues of Sediment Quality Guideline Quotient (SQG-Qm) calculated from trace element concentrations (As, Cd, Cr, Cu, Hg, Ni, Pb, and

Zn) measured in sediments collected at various sites in the Ukrainian Danube Delta in May 2004.

Table 18 1. Value of Sediment Quality Quideline (SQG-Qm) calculated from trace element concentrations (As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn) measurement in sediments collected zt various sies in the Ukrainian Danube Delta in Maay 2004

Sample	SQG-Qm	SQG-Qm (no Cr, Ni)	Remarks
1	0.287	0.186	Chilia arm (km 18)
2	0.206	0.101	Bystroe branch head
3	0.226	0.130	Bystroe branch mouth
4	**0.317**	0.178	Vostochny arm mouth
5	0.242	0.138	Starostambulsky branch
6	0.401	0.319	Chilia arm (km 24-25)
7	0.273	0.128	Chilia arm (km 29-30)
8	0.384	0.217	Chilia arm (km 35-38)
9	0.228	0.110	Chilia arm (km 47—49)
10	0.270	0.116	Chilia arm (km 51-52)
11	0.224	0.116	Chilia arm (km 61-63)
12	**0.505**	0.403	Chilia arm (km 64-66)
13	0.288	0.182	Chilia arm (km 67-68)
14	**0.760**	**0.686**	Chilia arm (km 73-74)
15	**0.270**	0.169	Bystroe arm sand bar

SQG-Qm values in bold exceed the threshold given in the literature for toxicity or effect on the total abundance of benthic organisms (see text). Distances are given in km from the Chilia mouth. See text for discussion of differences between SQG-Qm calculated with or without Crand Ni

$$SQG\text{ - }Qm = \frac{\Sigma\, SQG\text{ - }Qi}{n}$$

where :

SQG-Qm is the Sediment Quality Guideline Quotient,

SQG-Qi is the ratio between the concentration measured in sediments and the corresponding PEC value for each individual metal, and *n* is the number of metals considered in the study.

According to Long *et al.* (2006), the probability that sediment contamination will produce a response in standard laboratory toxicity tests for the marine invertebrates *Ampelisca abdita* and *Rhepoxynius abronius* is:

less than 10 per cent for SQG-Qm < 0.1

20 per cent for SQG-Qm in the range 0.1-0.25

30 per cent for SQG-Qm in the range 0.25-0.75

and for freshwater sediments tested with the amphipod *Hyalella azteca* the probability is:

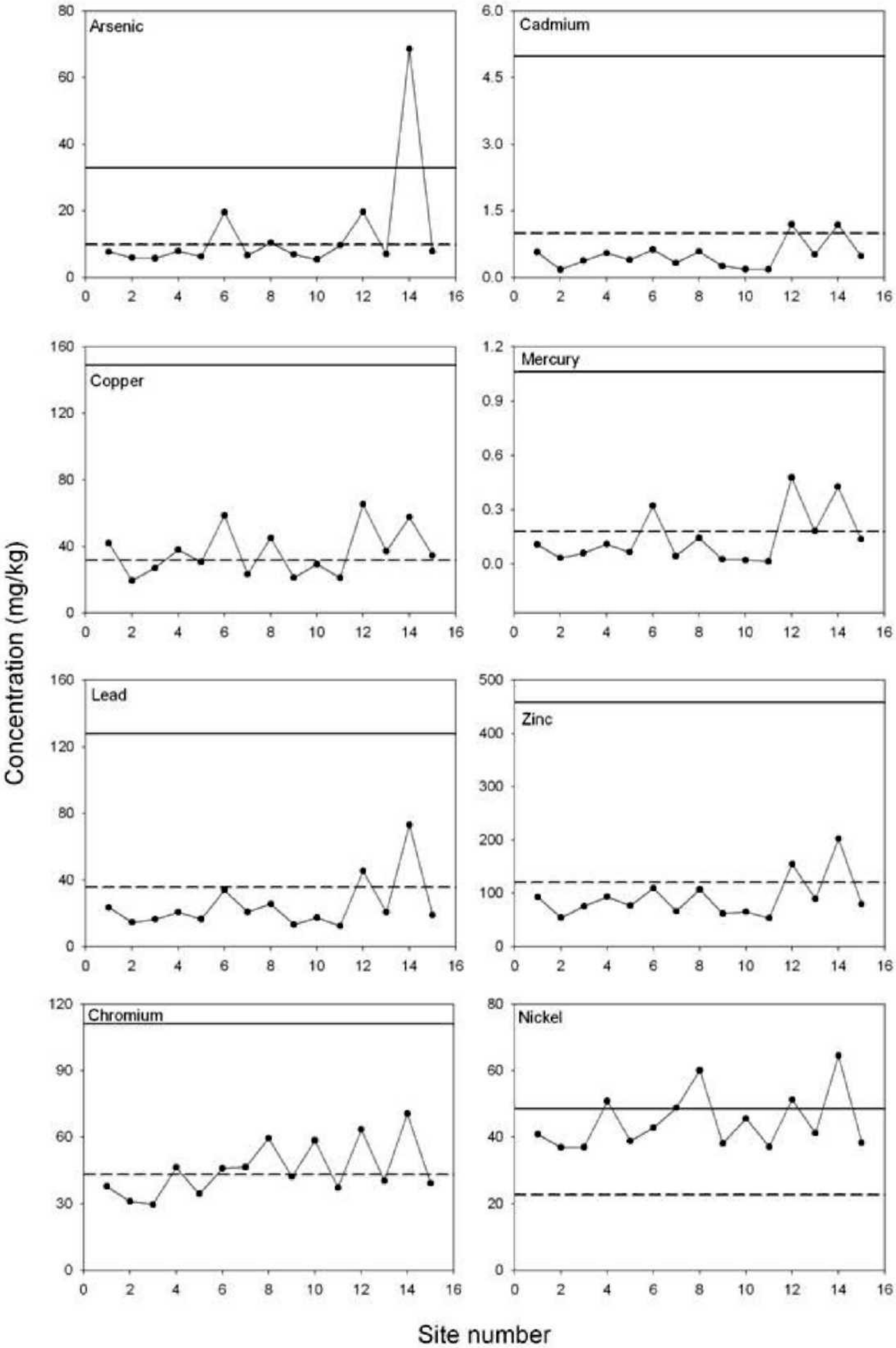

Fig. 18.2. Concentrations of selected trace elements (in mg kg^{-1}) in samples collected in the Ukrainian part of the Danube delta in May 2004 (see Table 1). The corresponding TEC (dashed lines) and PEC (full lines) values are reported for each element. Note the different vertical scales for the various elements.

less than 20 per cent for SQG-Qm < 0.1
less than 20 per cent for SQG-Qm in the range 0.1-0.5
40 per cent for SQG-Qm in the range 0.5-1.0

In this study of the Danube Delta, reference to tests with marine invertebrates had to be made for sites 2 to 5 and 15 because the other sites had predominantly freshwater characteristics. The SQG-Qm calculated for the available samples usually increased if Cr and Ni were included in the calculation, although the probability of toxicity were modified only for sites 4, 12, and 15. Without Cr and Ni, SQG-Qm values indicated that the probability of Danube sediments resulting in toxicity in laboratory tests was below 20 per cent at all sites except at site 14 where the probability of toxicity was 40 per cent. These results confirmed the importance of carefully judging the applicability of available sediment quality guidelines.

It is also important to note that the relationships between SQG-Qm and the probability of toxicity to benthic organisms available from literature were derived taking into account contamination by trace elements and organic pollutants (in general polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs)). In this study, the SQG-Qm calculated for Danube sediments were based solely on six trace elements and may therefore underestimate the actual sediment toxicity. Furthermore, existing sediment quality guidelines were largely based on data for sediments in North American estuaries and coastal regions, which are not necessarily representative of the Danube system and of the Black Sea. Finally, it is possible that effects on community structure and function occur at contamination levels lower than those that elicit a response from organisms in laboratory tests (Hewitt *et al,* 2009). Additional work to establish the risk of sediment toxicity in the region should focus on determining local background concentrations of trace elements, performing adequate toxicity tests, and linking sediment contamination to changes in community structure and functioning.

Conclusions

Based on available sediment quality guidelines, no severe pollution by trace elements was observed for samples collected in the Ukrainian Danube Delta in May 2004. At some sites, however, the combined effects of several elements may be sufficient to elicit toxicity in standard laboratory tests. Furthermore, the potential sediment toxicity calculated in this study did not take into account the presence of organic pollutants and may therefore underestimate the actual risk.

In the same way, endpoints at levels of organization higher than the individual responses in toxicity tests (e.g. community structure and function) may be more sensitive to sediment contamination. Additional extensive work to assess experimentally sediment toxicity (or lack thereof) in the Ukrainian Danube Delta should be considered as a research priority in future studies.

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19. Effects of the Bystroe deep-water shipping way on the water and sediment dynamics in the Danube Delta: current knowledge and future needs

The Danube River has a watershed area of 817,000 km^2 and flows for 2,857 km from west to east across several European countries. In its final section, the Danube forms a delta with an area of about 5,800 km^2; the second largest European delta (after the Volga) and one of the most important European wetlands classified as a United Nations Educational Scientific and Cultural Organization (UNESCO) World Heritage and as a Ramsar site. The Danube Delta consists of three main branches (Chilia, Sulina and Sfantu Gheorghe) interconnected by a multitude of lakes, ponds, channel and streamlets. The Sulina and Sfantu Gheorghe arms are entirely located in Romania, while the Chilia arm (the northernmost one) represents the country border between Romania and the Ukraine (Torica, 2006). The Sulina branch has been subject to extensive engineering works since the second half of the 19th century (Stanica *et al.*, 2007 and references therein) and has been progressively transformed into a shipping channel. It is part of the Pan-European Transport Corridor VII linking the North Sea to the Black Sea via the rivers Rhine and Danube (ICPDR, 2009).

The need for a deep-water shipping way (DWSW) alternative to the Sulina canal between the Black Sea and Danube River is a very delicate matter embracing political, economical, societal, and ecological issues at the local, national, and international scale. In May 2004, under the responsibility of the Ukrainian Ministry of Transport, dredging works were carried out to (re)open a DWSW across the Bystroe channel (see Fig. 19.1). These works led to a major public outcry (Schiermeier, 2004) and a controversy between the Ukrainian and Romanian authorities on their possible transboundary effects. Investigations by the ESPOO Commission (named after the Finnish city of Espoo where the convention on transboundary impacts was adopted) confirmed the likelihood of transboundary effects associated with water redistribution within the secondary Chilia delta (or Starostambulsky delta, see Figure 19.1 and its caption) and increasedturbidity of marine waters during dredging and dumping works (ESPOO, 2006). At the time of writing (February - August 2009), the Ukrainian government was still committed to the maintenance of a DWSW across the Bystroe, although it had acknowledged the conclusions of the ESPOO Commission and ensured that mitigating measures would be taken accordingly.

This chapter illustrates how, in real world situations, conflicting water uses (here represented by navigation *vs.* conservation of valuable natural resources) can be difficult to reconcile and, in the case of large international rivers, can cause serious conflict between neighbouring countries. The reasons behind the choice of the "Bystroe route" among several alternatives are re-examined and original results on the concentrations of suspended particulate matter (SPM) measured in the Danube Delta and in its coastal zone during the

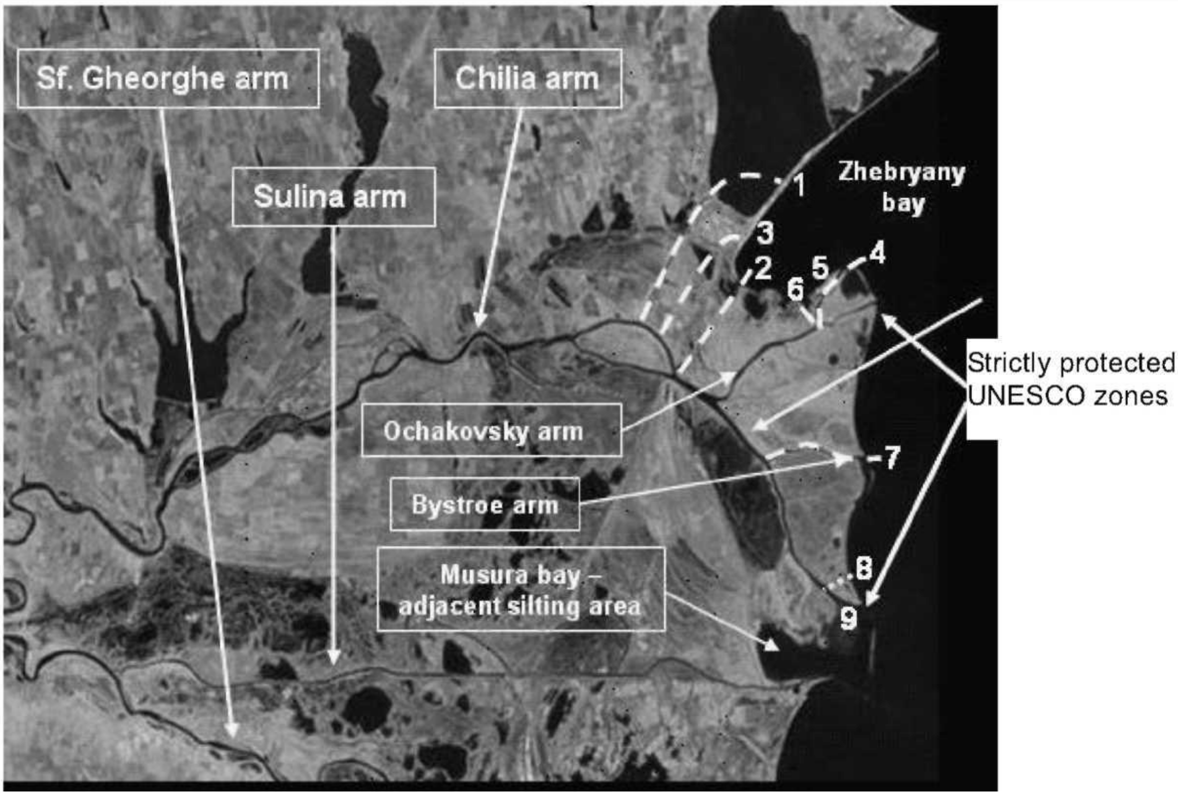

Fig. 19.1 Satellite partial view of the Danube Delta indicating the considered alternatives for the deep water shipping way (DWSW) between the Black Sea and the Danube across the Ukrainian part of the Danube Delta. The secondary Chilia (or Starostambulsky delta) is the area between the Starostambulsky arm to the south and Zhebryany bay to the north. The town of Vilkovo is located on the Starostambulsky arm just before it separates from the Ochakovsky arm. Alternatives for the DWSW are as follows: 1. The Danube - liman Sasyk; 2. Alternative of engineer P.S. Chekhovich (1904); 3. Solomonov branch - Zhebryany bay (engineer V.P. Zizak - 2000); 4. Prorva branch; 5. Connecting canal; 6. Ochakovsky branch - port of Ust-Dunaisk; 7. Bystroe branch (selected alternative); 8. Tsyganka branch; 9. Starostambulsky branch. Modified from Wikimedia commons. Public Domain; created by NASA

dredging works which marked the re-opening of the Bystroe DWSW in May 2004 are presented. These data may not be conclusive, but they provide useful indications on the effects of dredging activities in the region. The ESPOO Commission did indeed conclude that the opening of the Bystroe DWSW was likely to have some impacts on transboundary sediment transport, but acknowledged that the available information was incomplete and often not sufficient to draw definite conclusions.

Possible alternatives for a DWSW in the Ukrainian part of the Danube Delta

All possible alternatives for a DWSW in the Ukrainian part of the Danube Delta crossed the Danube Biosphere Reserve and required dredging works in the shallow waters of the bar zone located at the mouth of the projected navigation channel. Discussions on the best alternative for the DWSW lasted about 10 years and involved Ukrainian specialists, Western experts, NGOs, mass media, etc. Criteria for the selection of the optimum alternative included economic feasibility, ease of use and maintenance of the DWSW, and minimization of anthropogenic pressure on the Danube ecosystem. The relative importance of human influence against the long term evolution of the delta has been a matter of debate along with transboundary impact issues between Ukraine and Romania. A short description of the impacts of each alternative on sediment and water dynamics in the region is given below based on knowledge available at the time of the final decision (Berlinsky and Lonin, 1996; Michaylova, 2004; Berlinsky *et al.*, 2006; Ministry of Environmental Protection of Ukraine, 2006).

Alternatives 1, 2, and 3 (Fig. 19.1) were based on cutting new artificial canals (or deepening existing ones) across the territory of the Danube Biosphere Reserve. Besides a high level of ecological damage, these alternatives would have required the construction of a new port with railway connection in Zhebryany Bay (Fig. 19.1) and of a bridge to prevent the town of Vilkovo (see caption of Figure 19.1) from becoming isolated from the mainland. The hydrological regime in the delta would also have changed causing a deterioration in the general ecological conditions. According to the project documentation, the width of these canals should have been 60 m (at the bottom) with a depth of 8.5 m and with a water runoff of approximately 500 m^3 s^{-1} (this figure is half the natural water runoff of the Bystroe arm, i.e. the alternative finally retained by the authorities for the DWSW). The projected water runoff would have represented about 16 per cent of the Danube runoff at 115 km upstream from the mouth (about 3,000 m^3 s^{-1}). In practice, all these alternatives would have caused a water decrease in the Chilia Delta with a negative impact in the region of Vilkovo. Furthermore, such redistribution of water runoff would have sharply decreased the water entering into the Romanian part of the delta and possibly worsened the silting problems in Musura Bay (Fig. 19.1). In alternative 3 (construction of a new canal with locks), the changes in water runoff would have been less marked (about 2 per cent), but the other problems would have remained. Another problem with these alternatives would have been the shallow water depth (4-10 m, author's unpublished data) in Zhebryany Bay (Fig.19.1). The marine portion of the DWSW canal should have been much longer than in front of Bystroe arm and would have undergone permanent siltation (Berlinsky and Lonin, 1996). This situation implied high maintenance costs in addition to the altered natural conditions in the Ukrainian and Romanian parts of the marine delta area. Salt-wedge penetration into the artificial canal would have had negative consequences for the wetland system and for Zhebryany Bay, which is an important feeding area for Sturgeon.

Alternatives 4, 5, and 6 were linked with the Ochakovsky and Starostambulsky arms (Fig. 19.1). The Ochakovsky system is becoming progressively inactive (dying off from the geological point of view; Panin, 1997). Problems connected with these alternatives arose from silting (which would have required regular, extensive, costly and ecologically damaging dredging works) and from the need for building concrete structures (in the Strict Protection UNESCO zone for alternative 5). Furthermore, these works would have caused a redistribution of water discharge in the inner delta causing a marked modification in the hydrological regime of the Starostambulsky system; which is also in the Strict Protection UNESCO zone of the Biosphere Reserve.

The last alternatives were linked with the Starostambulsky system itself (alternative 9) and the Tsiganka arm (alternative 8). Both the Starostambulsky and Tsiganka arm cross the second Strict Protection UNESCO zone in the Biosphere Reserve and more than 90 per cent of birds included in the "Redbook" inhabit this part of the delta. Furthermore, the dumping area for dredged sediments would have been located close to the Romanian state border, possibly exacerbating transboundary issues.

The Bystroe arm

Although the Bystroe arm passed through the Danube Biosphere Reserve, it was selected by Ukrainian authorities for the construction of the DWSW because there was no need for sediment dredging. This arm is a natural, active branch of the Danube Delta with a depth of 12-14 m. Major dredging works took place in front of the Bystroe mouth, where large volumes of SPM carried by the Danube were deposited, and at some sites along the Chilia arm (see Table 19.1).

Table 19.1 Excavation zones, dumping zones and volumes (10^3 m^3) of material dredged during the re-opening works of the Bystroe Deep Water Shipping Way in 2004

Excavation zone	**Volume**	**Dumping zone**	**Volume**
Marine bars' zone	1,473	Marine dumping	1,544
River bed (upper 20 km)*	973	Shores dumping	687
		River bed dumping	215
Total	2,446	Total	2,446

* River km are counted from river mouth (km 0) The quality of dredged material was evaluated by the experts of the ESPOO Commission who considered the impact of dredging works on water and sediments contamination as "unlikely significant

One important controversy linked with the opening of the Bystroe DWSW was the possibility of water runoff redistribution within the Chilia secondary delta; a redistribution which could lead to damage to the Bilateral Biosphere Reserve. In the absence of adequate monitoring data, opposite conclusions can be reached depending on which model is adopted to estimate the percentage of water redistribution taking place after the dredging works. Based on the figures provided by the Romanian party, the ESPOO Commission (ESPOO, 2006) concluded that no significant impact should be expected in the water distribution among the main Danube Delta branches (i.e. Chilia, Sulina and Sfantu Gheorghe), but that "a *likely significant*

These observations were considered by the ESPOO hydrological experts who stood by their conclusions. Because the DWSW operations following official re-opening were discontinuous, experimental data to confirm either of the modelling results are not yet available. Regardless of any further development (or solution) of the controversy, collection of such data should be actively sought by both parties. One of the main reasons for shifting the choice to the Bystroe arm as a navigation route was the increasing water runoff in this arm (see Fig. 19.2).

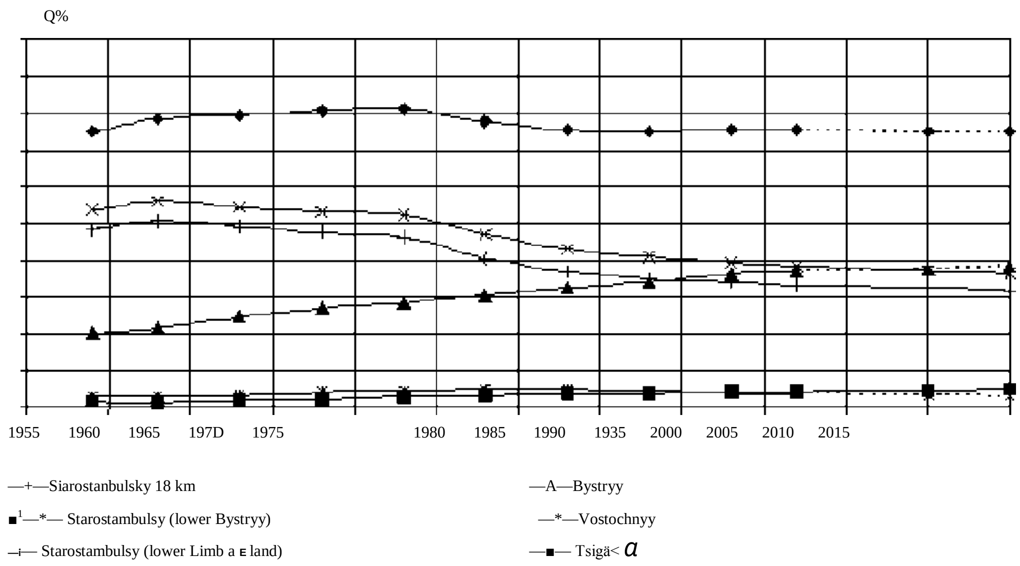

Fig. 19.2. The river runoff in the different branches

The possible transboundary effects of dredging and dumping works on the turbidity of river and coastal waters were the second controversial issue linked to the works in the Bystroe channel. In this case, the ESPOO report (2006) concluded that the impact of dredging works was *"unlikely to be significant"*, but that the impact of offshore dumping would *"likely be significant in view of natural variability"*. According to this report, the duration of the impact would obviously be limited to the period of dumping operations. These conclusions were based on modelling from information made available to the ESPOO experts. Original data on suspended matter concentrations during the period 25-28 May 2004 are presented and discussed below.

Suspended matter concentrations during the dredging works of May 2004

Suspended matter concentrations were determined by filtration at 0.45 um during the period of operation of the dredging ships Josef Möbius and M30 in May 2004. The results of this investigation, which were not available for ESPOO experts, gave the first indication of how dredging works related to the operation of the Bystroe DWSW could affect water turbidity in the coastal zone facing the Chilia (Starostambulski) Delta.

Sampling was carried out at various locations including the dredging and dumping sites. During operations, measurements by marine current meters showed an integral transportation of water masses towards the northwest direction for the water layer between 0 and 2 m. It is likely that the suspended matter transportation followed the same general direction. However, the more frequent coastal drift in this region is from North to South (Torica, 2006) and additional investigations under such conditions must be performed to evaluate properly the possible impact of the Bystroe DWSW. Investigations were also performed at 1 and 15 km from the dumping site to the South and at 18 km to the North. Suspended matter concentrations at the surface and near the bottom layer are shown in Figures 3 and 4. The actual depths for near-bottom sampling varied between approximately 1 m near shore to 25 m in the open sea.

In the zone of sediment dredging, maximum concentrations of suspended matter were measured close to the bottom layer (sampling depth 9.6 m) and were 73.33 mg l^{-1}. At the centre of the dumping site located at 45°19'13" N, 29°51'58" E (i.e. at the centre of the ideal circle formed by the seven adjacent zones selected for offshore infill and delimiting a hypothetical circle with a diameter of 1 marine mile), SPM concentration was 55 mg l^{-1} 10 minutes after the end of dumping operations. However, SPM concentration had decreased to 2 mg l^{-1} 50 minutes later (Figure 19.3). These results show that, locally, dumping works had a very short-term effect on water turbidity.

At the exact point of dumping (some 150 m from the centre of the dumping site), SPM concentrations measured near the bottom (about 22 m) were 61.6 mg l^{-1}; while concentrations at the surface and at 15 m depth were 5.89 and 8.25 mg l^{-1} respectively (data not shown). This difference in SPM concentrations between bottom, intermediate, and surface waters was due to: (i) the working technology used by the German specialists of "Josef Möbius" company (who purposely dumped the dredged material near the bottom layer), and (ii) the presence of the seasonal thermocline and halocline (i.e. a careful choice of dumping location and season) which restricted the suspended matter movement towards the upper layers and concentrated the plume near the bottom. This type of operation probably reduces the impact on the biota in the upper part of the water column, but biological data would be needed to confirm this hypothesis.

At the control station located 16 km south of the dumping site (approximately 9 km opposite the mouth of the Starostambulsky arm and 1 km from the marine border with Romania), the concentration of SPM was 8.71 mg l^{-1} at the surface and 1.85 mg l^{-1} near the bottom (depth 20.5 m). This was typical for SPM in this region because the less dense riverine waters (and their SPM loads) overlay the denser marine waters. At the mouth of the Starostambulsky arm the SPM concentration was 33.51 mg l^{-1} in the surface waters, but no measurements were made near the bottom. In a period without dredging activity (data not shown), concentrations of SPM at the Starostambulsky mouth were 11.05 mg l^{-1} for surface waters and 34.21 mg l^{-1} near the bottom (2 m). Due to the large natural variability of SPM loads in riverine environments, more data are needed to verify the possible influence of dredging activities in this zone.

At 18 km north from the dumping site, at about 3 km opposite to the mouth of Potapovsky arm (a distributary of the Ochakovsky arm, the concentration of SPM was 14.00 mg l^{-1} on the surface and 8.14 mg l^{-1} near the bottom (19 m deep). In the absence of "reference values", the available data do not allow a definite conclusion on whether dredging and dumping works under conditions of northbound transport of water masses cause an increase in SMP concentrations in this zone.

Measurements were also performed at about 100 m from the dredging machine M-30 taking into account the general direction of marine currents (again under stable south-east wind with a speed of 10-12 m s^{-1} and with integral transportation of surface water masses in a northwest direction). In a period without the M-30 dredge in activity, the concentration of SPM was 11.05 mg l^{-1} in surface waters and 34.21 mg l^{-1} near the bottom (approximate depth 2 m; data not shown in figures). During operation of the dredging machine, SPM concentrations near the bottom increased to 316.30 mg l^{-1} and 15.50 mg l^{-1} at the surface. At 450 m from the dredging works, the integrated concentration in the water column (0-1.5 m deep) was 25.61 mg l^{-1}. At a distance of 800 m (opposite to the Ptichy island), SPM concentrations were 28.63 mg l^{-1} in surface waters and 36.86 mg l^{-1} near the bottom (approximate depth 2 m; data not shown in figures). These values are comparable with average long-term concentrations of suspended matter in this estuarine region (25-30 mg l^{-1}) because of the normal influence of the Bystroe arm. Suspended matter concentrations at the Bystroe mouth were 117.35 mg l^{-1} at the surface and 39.00 mg l^{-1} near the bottom. The average long-term SPM concentration at the Bystroe mouth is 90 mg l^{-1}.

With currents in a northerly direction, the influence of dredging works in the shallow coastal waters facing the Bystroe arm were limited to a distance not exceeding 1 km.

Conclusions

Albeit not devoid of ecological consequences and transboundary impacts, the Bystroe DWSW was retained by Ukrainian authorities as the less damaging of all the possible alternatives for a shipping way in the Ukranian part of the Danube Delta.[1] Additional hydrological monitoring is required to establish whether dredging of the sediment bar accumulated at the Bystroe mouth will effectively cause a redistribution of water fluxes in the Danube river system. With respect to sediment transport, direct measurements suggest that the increase in suspended matter concentrations as a result of dredging works and dumping activities is limited to a distance of 800 m and should not reach the Romanian border (20 km from the Bystroe). However, further information should be collected using appropriate sampling plans (possibly jointly established by all interested parties). In particular, a detailed survey of suspended matter concentration in the area needs to be carried out at relevant sites before dredging operations. Monitoring should then be continued for a reasonable time span after the end of the works. Collection of data under south-bound wind conditions would be especially important for a better evaluation of the likelihood and the extent of transboundary impacts at the Ukrainian-Romanian border.

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Навчальне видання

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