Assessment of Water Pollutant Sources and Hydrodynamics of Pollution Spreading in Rivers

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Abstract Water is a necessary component of the human environment, as well as all vegetal and animal ecosystems. Unfortunately, water quality not just in Slovakia but also in other countries of the world, worsened in the course of the twentieth century, and this trend has not been stopped even at present. Current legislation evaluating the quality of water bodies in Slovakia is based on the implementation of the Water Framework Directive (2000/60/ES). The Directive requires eco-morphological monitoring of water bodies, which is based on an evaluation of the rate of anthropogenic impact. This does not refer only to river beds but also the state of the environs of each

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stream. While in the past point sources of pollution were considered as the most significant source of pollution in surface streams, after the installation of treatment plants for urban and industrial wastewater, non-point sources of pollution emerged as the critical sources of pollution in river basins. This contribution deals with the distribution and quantity assessment of pollutant sources in Slovakia during the period 2006–2015. The primary point sources evaluated are the ones representing higher values than the 90 percentile of the empirical distribution of total mass and also the mass of applied manures and fertilisers as non-point pollutant sources.

The development of computer technologies enables us to solve ecological problems in water management practice very efficiently. Mathematical and numerical modelling allows us to evaluate various situations of spreading of contaminants in rivers without immediate destructive impact on the environment. However, the reliability of models is closely connected with the availability and validity of input data. Hydrodynamic models simulating pollutant transport in open channels require large amounts of input data and computational time, but on the other hand, these kinds of models simulate dispersion in surface water in more detail. As input data, they require digitisation of the hydro-morphology of a stream, velocity profiles along the simulated part of the stream, calculation of the dispersion coefficients and also the locations of pollutant sources and their quantity. The highest extent of uncertainty is linked with the determination of dispersion coefficient values. These coefficients can be accurately obtained by way of field measurements, directly reflecting conditions in the existing part of an open channel. It is not always possible to obtain these coefficients in the field, however, because of financial or time constraints. The other aim of this contribution is to describe the methodology of this coefficient calculation and to present the value range obtained. The results and obtained knowledge about values of longitudinal dispersion coefficients and dispersion processes can be applied in numerical simulations of pollutant spreading in a natural stream.

1   Introduction

Water is an essential component of the human environment, as well as all vegetal and animal ecosystems. The development of industry, transportation and agriculture, increase in living standards, an extension of urban areas and subsequent increase in storm water volumes transported by sewer systems all significantly influence the environment. Pollutants from point and area sources worsen the quality not only of water but also of soil and the atmosphere.

In the classification process of water sources, it is insufficient to classify the capacity (quantity) alone because the water quality is a determining factor for many applications (water supply, food industry, the pharmaceutical industry, irrigation). Water quality is defined as a representative dataset which defines the physical, chemical and biological

water attributes from the possibilities of water use for different purposes (drinking water, recreational, industrial, agriculture, power generation, transport).

In the field of surface and groundwater protection, the situation in the Slovak Republic has significantly changed due to the accession of the country to the EU. Since that accession process, practically all the Slovakian legislation concerning water has been changed due to the acceptance of the principles of the EU Water Framework Directive (WFD 2000/60/EC), including the basic water management activity: protection of the quality of water sources. The Water Framework Directive requires as an obligatory goal to achieve and maintain "good water quality" status within the defined period (for Slovakia this period was set up to the year 2015). For surface water, the main criterion is the level of ecological and chemical quality.

In connection with the adoption of measures for improving the surface water quality, numerical simulation models are very useful tools which can simulate the consequences of the adopted measures, i.e. their suitability and efficiency, or otherwise to show the inefficiency or unsuitability of proposed measures.

2 Water Quality

Water quality is construed as affording the possibility of using water for the required purpose. However, the purpose itself is neither precisely defined nor essential. In practice, this means that it is not the chemical purity but its desirable properties that determine the quality of water. For instance, distilled water can be used for filling accumulator batteries, but it is not suitable for drinking purposes. Conversely, drinking water is not suitable as accumulator filler. Hence, what is an inappropriate component of water (e.g. minerals) in one case is precisely the desirable component in another.

It is necessary to bear in mind that achieving or maintaining good water status is the purpose of water use (see the definition in WFD 2000/60/EC, for instance). It is also necessary to take into account the degree of toxicity for waterborne organisms, or organisms bound to aquatic ecosystems, as well as the degree of toxicity for the environment in general.

It is also necessary to realise that the notion of water quality is a relative one, i.e. it changes in time and space.

Water protection is considered to be a basic water management activity towards which most of the activities performed in water management are directed. The integrated protection of water resources, which currently constitutes one of the limits to the development of human society, is the goal of this activity.

Politics and economic interests may also play a negative role in the area of water protection. Economic forecasters predict that, just as there are currently wars for oil, in the future there will be wars for water, which is becoming a restricting factor on the development of society in some locations, due to the depletion or deterioration of water resources.

The statement that life is not possible without water sounds like a platitude, but it nevertheless remains valid. However, it should be added that life is not possible without good-quality water, i.e. water not meeting the requirements for its use in terms of its

quality: either as drinking, utility, irrigation or other water uses. Thus, water quantity and quality become the basic parameters of the utility value of water; this can be expressed as the resultant product of these two parameters. Hence, if one of these two parameters (water quantity or quality) is zero, the total utility value of the water is also zero (there is more water in a small pure spring than in a dirty river).

The role of water managers is not to maintain water in nature in an absolutely pure condition; after all, that is probably not even possible (except areas with strict nature and landscape protection). Their role in terms of sustainable development is rather to maintain water quality at an adequate level. It means to maintain water quality at such a level as to ensure the exploitation of water resources for the required purpose, or to ensure universal water protection, including aquatic ecosystems and ecosystems dependent on water. Generally, it means the improvement of water status and the effective and economical utilisation of waters. It is necessary to recognise that the requirement of "returning to the original state" is no longer feasible today, not to mention that it is not possible to define the "original state" of waters.

The development of human society down the centuries has also led to pressure on water quality protection, not only to ensure basic human requirements (drinking water) but also to utilise water in other spheres of human activity (e.g. industry, recreation, urban sanitation). At the turn of the nineteenth and twentieth centuries, water managers virtually became some of the earliest protectors of nature and also users of biotechnologies (wastewater treatment processes). The traditional philosophy was based on the principle of protecting people from nature. The increased sensitivity of the population to essential nature protection and the popularisation of environmentally friendly perspectives have also been reflected in the ambit of water protection, hence in the introduction of a new concept of nature protection. This means that new, opposing opinions on environmental protection prevail today, as well as the related requirements of protection of nature against people.

3   EU Legislation in the Area of Water Protection and Its Basic Principles

One of the first documents adopted jointly by the European Community was the European Water Charter [1]. It was prepared by the European Committee for the Conservation of Nature and Natural Resources of the Council of Europe and adopted on 6th May 1968 in Strasbourg. The Water Charter defines the basic principles of water protection and management which were later reflected in the overall EU policy.

1. There is no life without water. It is a treasure indispensable to all human activity.
2. Freshwater resources are not inexhaustible. It is essential to conserve, control and, wherever possible, increase them.
3. To pollute water is to harm humans and other living creatures which are dependent on water.
4. The quality of water must be maintained at levels suitable for the use to be made of it and, in particular, must meet appropriate public health standards.

5. When used water is returned to a common source, it must not impair the further uses, both public and private, to which the common source will be put.
6. The maintenance of adequate vegetation cover, preferably forest land, is imperative for the conservation of water resources.
7. Water resources must be assessed.
8. The wise husbandry of water resources must be planned by the appropriate authorities.
9. Conservation of water calls for intensified scientific research, training of specialists and public information services.
10. Water is our common heritage, the value of which must be recognised by all. Everyone must use water carefully and economically.
11. The management of water resources should be based on their natural basins rather than on political and administrative boundaries.
12. Water knows no frontiers: as a common resource, it demands international cooperation.

In the following part, the basic principles of EU legislation related to water protection are explained.

With regard to the scope of the individual legal documents, we may divide EU legislation into two basic groups. The first is called horizontal legislation which covers the entire environmental area (e.g. EIA, nature and landscape protection regulations), while the second is called vertical (specific) legislation which is focused more on the individual components of the environment (e.g. water, soil, air quality protection).

In the area of water protection, the EU legal system uses the following three forms of legislative documents:

• Directive
• Regulation
• Decision

An EU Directive expresses an endeavour to introduce universal legal norms whereby, however, it is possible to maintain traditional practice and adapt to the degree of development in the given countries. A Directive is a legal document which does not take precedence over the legislation of a Member State. However, Member States are required to "indirectly" apply a Directive, i.e. to apply the principles of the Directive which have to be absorbed into the legislation of the given Member State. However, the principle is that a Member State may adopt measures going "beyond the framework" of the respective Directive, but it must not adopt less strict criteria than those stipulated by the given Directive. Currently, within the EU, only Directives are being applied in water management. Absorbing the principles of EU Directives into the national legislation is referred to as transposition of the law; implementing the adopted measures is referred to as an implementation of the law.

An EU Regulation is a generally valid legislative document directly applicable within the territory of all Member States. From the legal perspective, a Regulation takes precedence over national law.

A Decision is a highly specific legal document, directly binding only for those for which it is intended. It is usually issued only when one of the Member States violates the provisions of EU legislation; it can be compared to a court decision.

It is evident that harmonisation of the legislative requirements of EU Member States is quite a demanding task, due not solely to differences of opinion on environmental protection but mainly due to the varying levels of protection in the individual Member States, which are related to their levels of economic and social development. For this reason, the EU bodies have adopted the principle of the so-called lowest common denominator, i.e. the primary determination of the minimal environmental protection requirements which are common and acceptable to all Member States. Following approximation and implementation, these minimal requirements will be increased incrementally until they achieve the target status (protection level) standard for all Member States.

EU environmental legislation recognises the following universal principles:

- Environmental protection must not encroach upon the protection of the EU internal market, nor constrain competition within the EU.
- Prevention is emphasised.
- Greening of the economy and social policy (ultimately of all activities).
- "Polluter pays" principle (PPP).
- Harmonisation and unification of Member States' legislation.
- Right of citizens to information on the status of the environment.

In addition to the above principles, specific principles also apply to water management and water protection. We list at least some of these principles here:

- Payment of all costs incurred by activities in the area of water management (WM) must be self-fundable.
- WM activities to be pursued based on natural river basins.
- Achievement (or maintenance) of the so-called good status of water bodies within the EU.

4   Legislation in Slovakia

4.1   Water Framework Directive (WFD, 2000/60/EC)

The Water Framework Directive (WFD, 2000/60/EC) is the primary legislative document for water quality (but also for the entire EU water management policy).

The WFD introduces a new approach to water management based on river basins, or natural geographical and hydrological units; it imposes specific deadlines on the EU Member States to develop river basin management plans including programmes of measures. The new approach to water protection makes it possible to create a unified system for water evaluation within the EU Member States, affording reliable and comparable results of the condition of water bodies in any European region. The

next asset is the application of same procedure for the determination of objectives and implementation of all necessary measures for the protection and improvement of water status, as well. The WFD deals with surface waters (rivers, lakes); transitional, coastal waters; groundwaters; and, under certain specific conditions, also terrestrial ecosystems dependent on water and wetlands. The WFD introduces several innovative approaches to water management, such as public participation in planning and integration of economic approaches to the planning and integration of water management with other economic sectors.

The main objective of the WFD is to achieve the so-called good status of waters in the Member States, which will ensure the protection and improvement of the state of aquatic ecosystems and sustainable, balanced and equitable water use. This status should have been achieved by 2015 or must be achieved by 2027.

The European Commission developed a basic document for the EU Member States: the WFD Common Implementation Strategy adopted by the Member States in May 2001. This Strategy is regularly updated at 2-year intervals for the subsequent period.

4.2 Council Directive 91/271/EEC Concerning Urban Wastewater Treatment

The main objective of this Directive concerning urban wastewater treatment is the protection of aquatic ecosystems from the adverse effects of discharges of untreated or insufficiently treated urban wastewater.

The requirements of this Directive can be characterised as follows:

- The requirement to build a public sewage system and two-stage wastewater treatment plant in agglomerations of over 2,000 population equivalents (p.e.).
- Each discharge of wastewater must be permitted by the relevant authority.
- More stringent criteria in agglomerations of over 10,000 p.e., in the food industry and in sensitive areas (elimination of nutrients, nitrogen (N) and phosphorus (P)).
- Permits for wastewater discharges are subject to review.
- Emphasis on the reduction or disposal of pollution at the point of origin and reuse of treated water.
- Sludge must not be disposed of in surface waters, and it should be recycled.

The emission requirements of Directive 91/271/EEC on urban wastewater treatment are complemented by qualitative immission water protection requirements which are formulated in the related directives, mainly [2]:

- Directive 76/160/EEC concerning the quality of bathing water
- Directive 75/440/EEC concerning the quality required of surface water intended for the abstraction of drinking water
- Directive 78/659/EEC on the quality of freshwaters requiring protection or improvement in order to support fish life

Based on the requirements of this Directive, it is quite evident that the implementation of these requirements demands major measures and costs. There are mainly investments in the construction of new sewage systems and wastewater treatment plants (WWTP) and in the renovation of the existing systems and reconstruction of existing WWTPs (alteration of technologies to extended disposal of bionutrients).

5   Water Pollution Sources

Sources of pollution are considered to be any activity or phenomenon resulting in a deterioration of water quality. Based on the geographical form, each water pollution source can be categorised as follows:

- Point sources (e.g. sewage system outflow, oil spillage)
- Line sources (e.g. transport structures or pipelines)
- Diffuse sources (e.g. numerous leaking cesspits in a village)
- Areal sources (e.g. agricultural pollution such as fertilisers, pesticides, herbicides; exhaust gases, precipitation)

Significant water pollution sources are usually included in tabular or map form in the basic water management land-use planning documents.

A point source of water pollution is a pollution source with a concentrated input of pollution into waters which is limited to a relatively small area or almost confined to a single geographical point. These pollution sources are usually precisely quantifiable, so it is usually easy to monitor them, and the impact of every individual source can be accurately determined. As a result of diffusion and transport of the pollutant, linear or areal contamination of groundwater or surface waters can occur.

Line sources of pollution usually consist of leaks of pollutants along transport and traffic structures such as highways or railways, or along with other transport facilities such as oil pipelines or large sewage collectors.

In the literature, diffuse sources of pollution are usually understood as several point sources of pollution together, whereby it is not possible to determine the impact or effect of the individual (point) source. A village with leaking cesspits or septic tanks which, in combination, act almost as an areal source of pollution but where, in this case, there are several point sources of pollution is a typical example.

Areal sources of pollution are those where the pollutant is input over a large area. It is usually not possible to quantify the pollutant nor to accurately demarcate the point of penetration of the pollutant. In these cases, this is primarily groundwater pollution. Agricultural activities are a typical example, e.g. areal application of fertilisers and pesticides.

5.1   Water Quality and Pollution Source Deployment in Slovakia

Surface water quality at all monitored sites complied in each year with the limits for selected general indicators and the radioactivity indicators. Exceeded limit values were recorded mainly for synthetic and non-synthetic substances, hydro-biological and microbiological indicators and nitrite nitrogen. Until 2007, surface water quality was assessed according to STN 75 221 in five quality categories and eight indicator groups. In the years 1995–2007, 40–60% of abstraction sites showed the fourth and fifth quality categories for the groups of F (micro-pollutants) and E (biological and microbiological indicators) [3].

In line with the requirements of WFD 2000/60/EC, water quality is expressed in terms of the ecological and chemical balance of surface water bodies. Adverse and critically adverse ecological situations are recorded in approx. 4–8% of water bodies, and approx. 3–10% do not reach good chemical balance.

Monitoring for groundwater chemical balance is carried out as part of basic monitoring (171 stations) and operational monitoring (295 stations). Both types of monitoring show exceeded values for set contamination limits. In 1995–2006, groundwater quality was assessed according to STN 75 7111 in 26 water management significant areas.

Major sources of contamination of water bodies include residential agglomerations, industry and agriculture. The main point sources of surface water pollution comprise industrial plants and wastewater treatment plant outlets. Applications of fertilisers in agriculture represent an areal source of pollution [4, 5].

The contamination of surface water bodies is characterised in general by chemical oxygen demand by dichromate (COD_{Cr}), biochemical oxygen demand (BOD), total nitrogen amount and insoluble substances (IS). Distribution of the main producers by these parameters in Slovakia is shown in Figs. 1, 2, 3 and 4.

Fertilisers applied in agriculture are divided into two groups: industrial fertilisers based on chemicals (N-P-K) (nitrogen, phosphorus, potassium) and organic fertilisers. Their consumption is summarised by the district in each year. The total amounts of applied N-P-K and organic fertilisers in each district during the 10-year period (2006–2015) are shown in Figs. 5 and 6. The consumption of these fertilisers can also be monitored in kilogrammes per hectare of agricultural soil, as shown in Figs. 7 and 8. As can be seen from these figures, the distribution of applied amounts of fertiliser is different in partial districts in this case. So, from the point of view of water contamination evaluation, it is important to pick out suitable and comparable parameters and units.

The total annual consumption of specific kinds of fertiliser was different in each year. It turns out that the minimum amount of industrial fertilisers was applied in 2009 and 2010 (Fig. 9). From these years onwards, the N-P-K fertiliser consumption has increased in each year. In contrast, organic fertiliser application has slightly decreased.
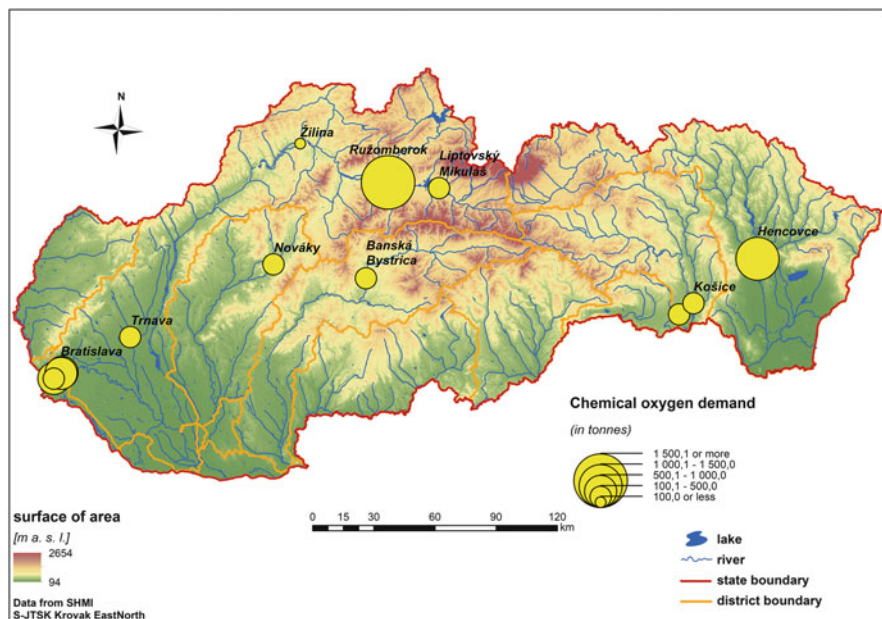
Fig. 1 Deployment of the main producers of surface water contamination based on the average discharge amount of COD_{Cr} (in the period 2006–2015)

6   Hydrodynamic Numerical Models of Pollutant Transport

The development of computer technologies enables us to solve ecological problems in water management practice very efficiently. Mathematical and numerical modelling allows us to evaluate various situations of contaminant spreading in rivers without immediate destructive impact to the environment.

A lot of mathematical and numerical models have been developed to simulate water quality (e.g. WQMCAL, AGNSP, CORMIX, QUAL2E, SWMM, P-ROUTE, MIKE1, ZNEC, MODI, HSPF, SIRENIE). These models are based on various approaches, including hydrodynamic, statistical and balanced (reviewing past similar events). These models can simulate the real situation in streams. However, the range of reliability and accuracy of the results is vast [4, 6–12].

Problems of dependability and correctness of dispersion numerical models are wide-ranging, and it is not possible to cover them only in one chapter or study. For this reason, part of this chapter is focused on simulation models based on the hydrodynamic approach, i.e. models based on numerical solution of the advection-dispersion equation and determination of one of the leading characteristics of mixing processes in streams: the longitudinal dispersion coefficient.

Simulation models can describe the transport of contaminants even three-dimensionally. On the other hand, the input is labour intensive, and the development
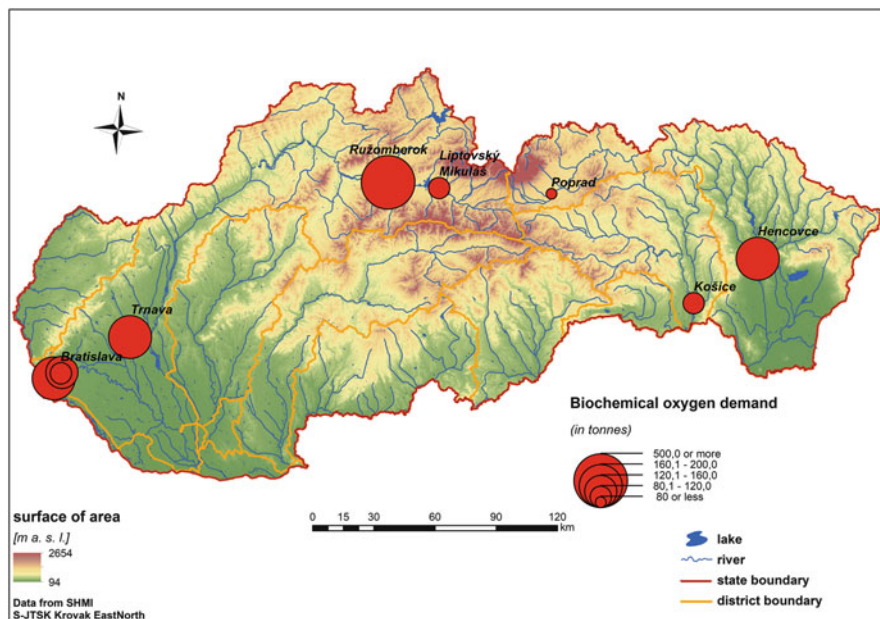
Fig. 2 Deployment of the main producers of surface water contamination based on the average discharge amount of BOD (in the period 2006–2015)

of the model structure is time-consuming as well. However, this lengthy approach is important for deep reservoirs with thermal stratification. Two-dimensional modelling of the transport processes is reasonably accurate if applied to shallow reservoirs without strong thermal stratification. Otherwise, it can be used in detailed studies of the movement of pollutants into the surface water before complete mixing of transported substances across the section of flow occurs (the so-called mixing length). After this moment it is sufficient to apply a one-dimensional model of water quality.

The reliability of models is influenced by the fact that numerical simulations always mean some simplification of the complicated natural conditions. Finally, the rate of reliability is closely connected with the level of input availability and validity [13].

Hydrodynamic models simulating pollution transport in open channels require a good deal of input data and computation time, but on the other hand, these kinds of models simulate dispersion in surface water in more detail. As input data, they require digitisation of the hydro-morphology of the stream bed, velocity profiles along the simulated part of a stream, calculation of the dispersion coefficients and also the positions of pollutant sources and their massiveness. Calculation of dispersion coefficient values has the highest extent of uncertainty. These coefficients can be exactly obtained by way of field measurements, directly reflecting conditions in an existing part of an open channel. It is not always possible to obtain these coefficients in the field though, because of financial or time reasons. Several authors [8, 11, 12, 14–22] have
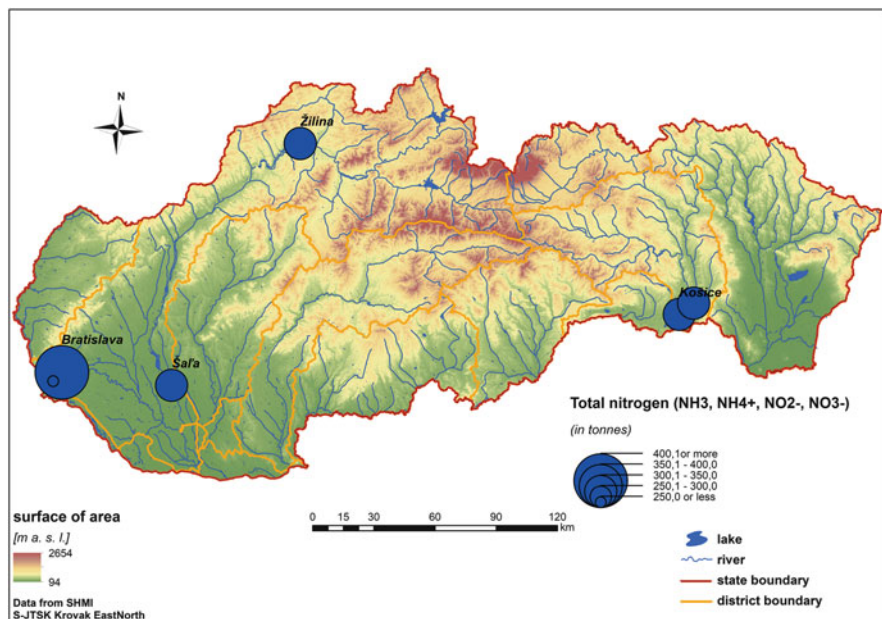
Fig. 3 Deployment of the main producers of surface water contamination based on the average discharge amount of total nitrogen (in the period 2006–2015)

tried to get the empirical relations, especially for the longitudinal dispersion coefficient. Their studies could be used to estimate an approximate value of the dispersion coefficient. The results of studies [11, 23–26] on the conditions of Slovakian rivers show that the formulae derived in this way are often not applicable for those rivers. The reasons behind this are the longitudinal slopes used in the formulae are very flat, or the precondition of stream channel roughness is not suitable or the shape of the cross-section profile is inappropriate. So, for this reason, we need to try to obtain the applicable range of this dispersion coefficient values.

7 Longitudinal Dispersion

7.1 Basic Theoretical Terms

Dispersion, from the hydrodynamic point of view, is the spreading of mass from highly concentrated areas to less concentrated areas in flowing fluid. Mass in flowing water is not transported only in the reach of the stream line, but it also gradually spreads outside that line as a consequence of velocity pulsations and mass concentration differences. Mass dispersion with advection is the basic motion mechanics of particles transported in water. Reductions in maximum concentrations are the result
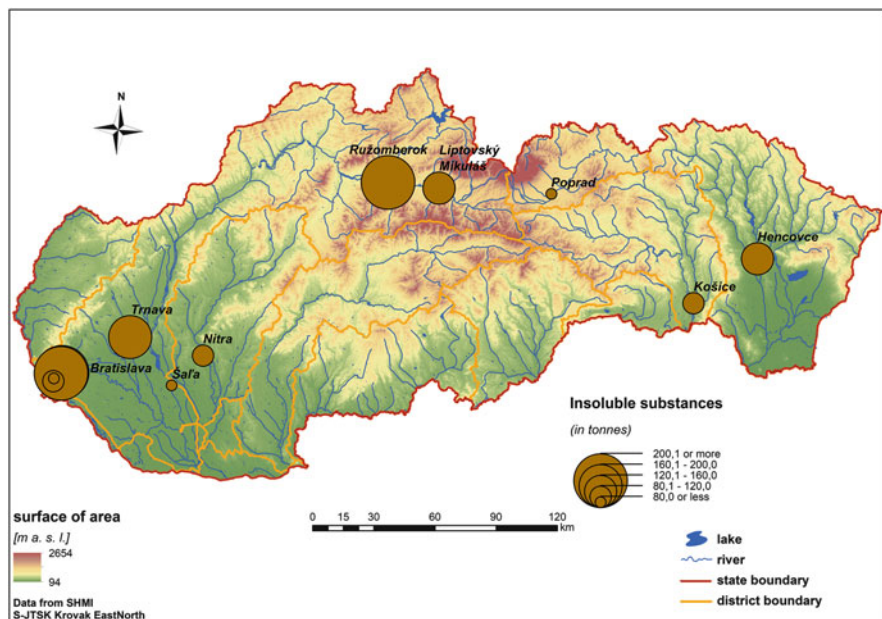
Fig. 4 Deployment of the main producers of surface water contamination based on the average discharge amount of IS (in the period 2006–2015)
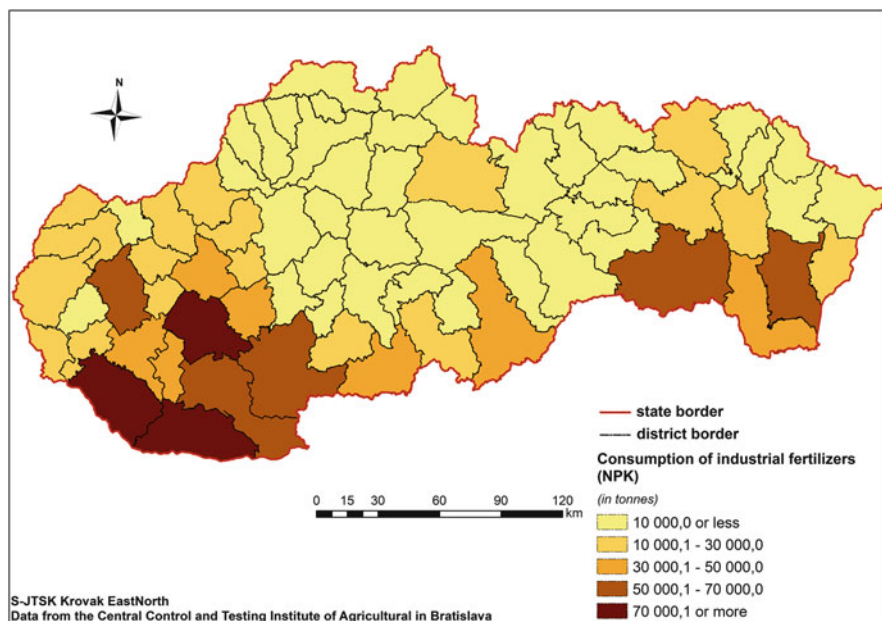

Fig. 5 Total amount of applied N-P-K fertilisers in districts during the 10-year period (2006–2015)
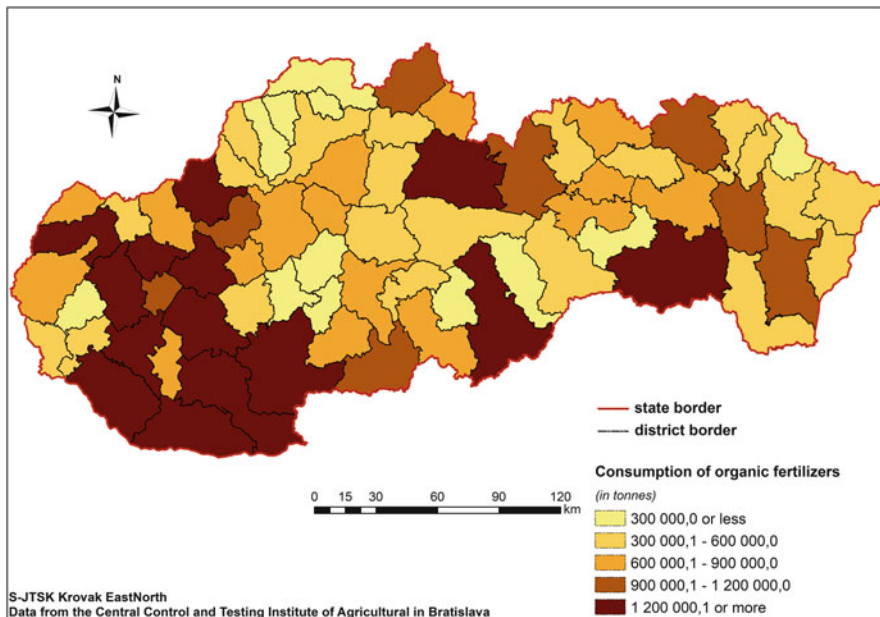
Fig. 6 Total amount of applied organic fertilisers in districts during the 10-year period (2006–2015)
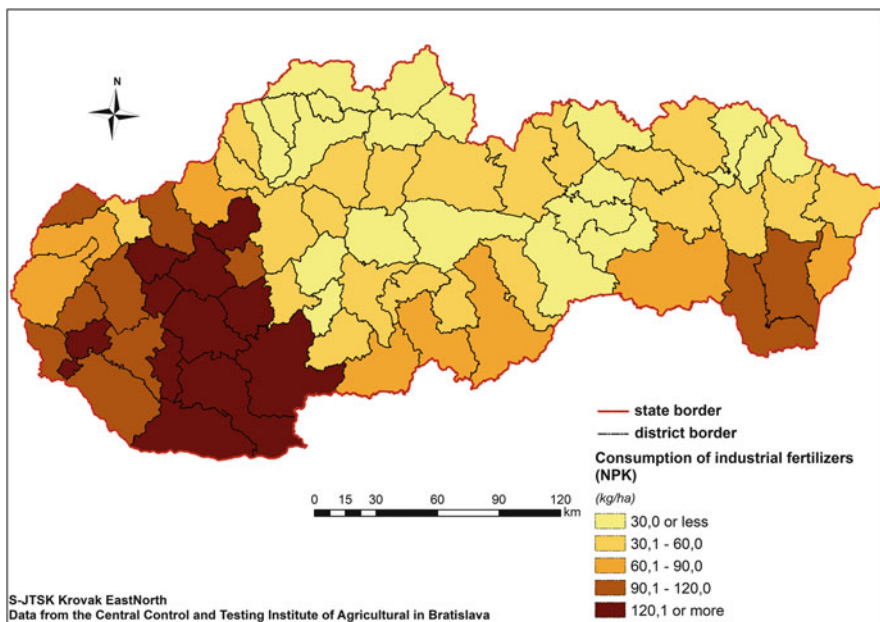

Fig. 7 Average consumption amount of N-P-K fertilisers applied to agricultural soils in the period 2006–2015
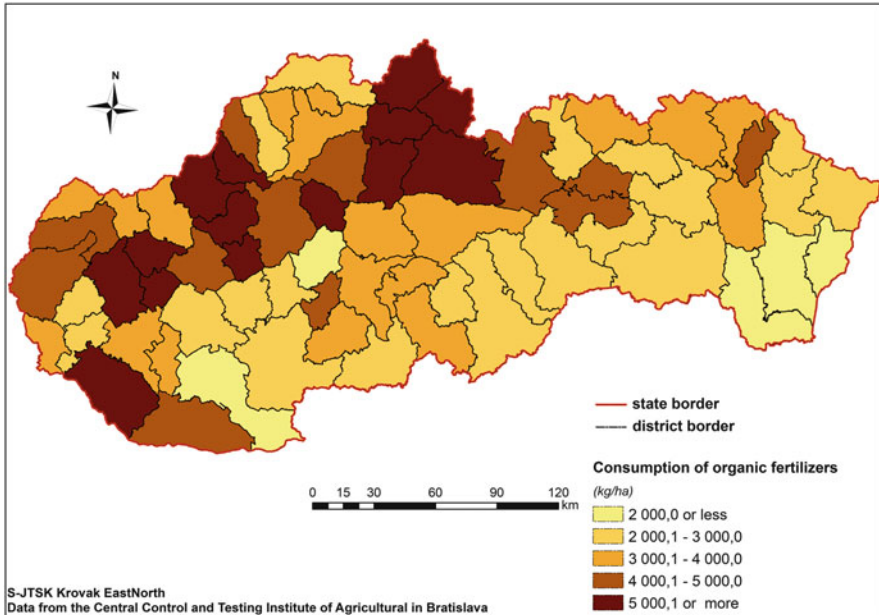
Fig. 8 Average consumption amount of organic fertilisers applied to agricultural soils in the period 2006–2015

Fig. 9 Trend in fertiliser application during the period 2006–2015

of the effects of those mechanics. The main characteristics of dispersion are dispersion coefficients in relevant directions. Identification of these dispersion characteristics is the key task for solving the problem of pollutant transport in streams and for modelling of water quality.

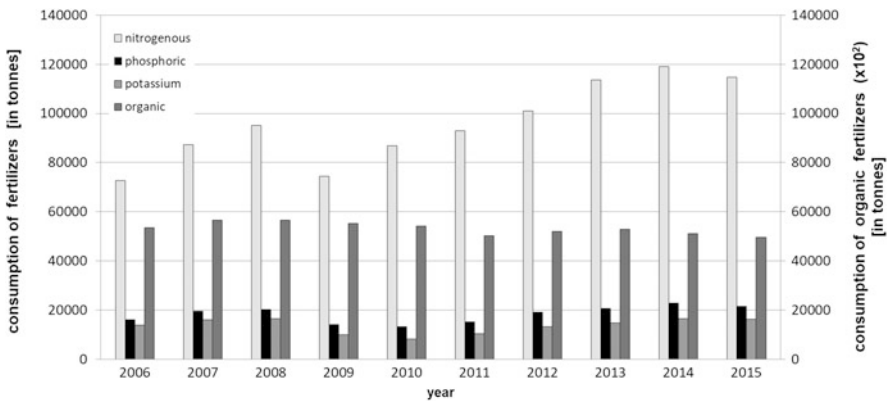The most straightforward description of mass spreading in water is the one-dimensional advection-dispersion equation, which takes the phenomenon in longitudinal direction x (well-proportioned distribution of mass concentration is required along the depth and across the width of a stream). The form of this equation is [25, 27]

$$\frac{\partial A c}{\partial t} + \frac{\partial Q c}{\partial x} - \frac{\partial}{\partial x}\left(A D_L \frac{\partial c}{\partial x}\right) = -A K c + c_s q \qquad (1)$$
$$\underset{\text{I}}{} \quad \underset{\text{II}}{} \quad \underset{\text{III}}{} \quad \underset{\text{IV}}{} \quad \underset{\text{V}}{}$$

where c is the mass concentration (kg m^{-3}), D_L is the longitudinal dispersion coefficient (m^2 s^{-1}), A is the discharge area in the stream cross-section (m^2), Q is the discharge in the stream (m^3 s^{-1}), K represents the rate of growth or decay of contaminant (s^{-1}), c_s is the concentration of the contaminant source, q is the discharge of the source, x is the distance (m) and t is time (s).

Part I in Eq. (1) expresses pollutant concentration change in time, part II represents pollutant transport through the velocity field, part III describes pollutant transport by diffusion and dispersion, part IV means chemical or biological nonconservation of pollutant, and part V represents pollutant sources in the stream.

Equation (1) covers two basic transport mechanisms:

- Advection (or convection) transport by fluid flow
- Dispersion transport by the concentration gradient

This one-dimensional approach is applicable for rivers or streams with comparatively non-wide channels, or for sewers, for example. In this case, the pollutant spreading has markedly one-dimensional character. However, this assumption is not acceptable for reservoirs, where the spreading phenomenon has three-dimensional character, meaning that the hydraulic characteristics and their values vary with the width as well as the depth of the discharge cross-section.

As the dispersion coefficient value is affected by the turbulence intensity in the given stream section, its magnitude depends upon its main hydraulic characteristics: the shape and magnitude of its cross-section profile, its flow velocity and its longitudinal slope. For this reason, the relationships derived by several authors for calculating the coefficient use the same characteristics (see Table 1).

Most of the published relationships used for calculating D_L are based on experimental results from laboratory physical models, or directly from field measurements on the rivers themselves. Such relationships are often expressed in the following form [9, 12, 25, 27]:

$$D_L = p h u_* \qquad (2)$$

where p is the empirical dimensionless coefficient, h is the mean river section depth (m) and u_* is the friction velocity (m s^{-1}).

The empirical dimensionless coefficient p acquires values, according to the authors concerned, in a fairly wide range, depending on the particular local conditions. This can be documented based on the results of Elder [16], Krenkel and Orlob [21] (laboratory conditions), as well as those of Říha et al. [9, 12, 15], Pekárová and

Table 1 Relationships for assignment of D_L

Author	Relationship	D_L (m^2 s^{-1})
Parker [28]	$D_L = 14.28\, R^{3/2} \sqrt{2 \cdot g \cdot i}$	0.345
Elder [16]	$D_L = 5.93\, u_* h$	0.101
Yotsukura and Fiering [29]	$\frac{D_L}{h \cdot u_*} = f\left(\frac{u}{u_*}\right)$	0.222
Krenkel and Orlob [21]	$D_L = 9.1\, u_* h$	0.155
Thackston and Krenkel [30]	$D_L = 7.25\, u_* h \left(\frac{u}{u_*}\right)^{1/4}$	0.113
Fischer [17]	$D_L = 0.011\, \frac{u^2 W^2}{u_* h}$	11.37
McQuivey-Keefer [31]	$D_L = 0.058\, \frac{Q}{i \cdot W}$	1.323
Kosorin [20]	$D_L = \dfrac{0.0696\, W^2 u_{\max}^2}{H\sqrt{gHi}}$	97.78
	$D_L = \frac{0.278\, W^2 u^2}{H\sqrt{gHi}}$	183.96
	$D_L = \frac{0.688\, Q^2}{H^3 \sqrt{gHi}}$	268.78
	$D_L = \frac{0.172\, W^2 C^2 \sqrt{i}}{\sqrt{gH}}$	309.70
Říha et al. [12, 15]	$D_L = \frac{0.001617}{p} \cdot \frac{W^2 C^2 \sqrt{i_e}}{\sqrt{gH}}$	
Kashefipour-Falconer [32]	$D_L = 10.612\, \frac{hu}{u_*}$	
Sahay–Dutta [33]	$D_L = 2hu\left(\frac{W}{h}\right)^{0.96} \left(\frac{u}{u_*}\right)^{1.25}$	

R hydraulic radius, g acceleration of gravity, i longitudinal slope of stream, u flow velocity, W width, Q discharge, H maximum depth in cross-section profile, C coefficient by Chézy

Velísková [11], Brady and Johnson [34], and Glover [35]. The latter results were derived from field experiments on natural streams. The conditions of field measurements are briefly given in Table 2.

The reliability of models is influenced by the fact that the numerical simulations always involve simplification of the complicated natural conditions. Ultimately the rate of reliability is intimately connected with the level of input availability and validity.

The problem of contaminant spreading is current not only in the case of modelling of water quality in natural streams but also in urbanistic structures, i.e. in sewer networks. For that reason, it is necessary to pay attention to this fact. One aspect of our interest is, therefore, the calculation of the longitudinal dispersion coefficient for prismatic channels, in this case for sewers, from field experiments.

7.2   Field Measurements

Our field measurements were done at the experimental hydrological base of the Institute of Hydrology in Liptovský Mikuláš. Part of a sewer network built in

Table 2 Values of dimensionless dispersion coefficient p and longitudinal dispersion coefficient D_L from experimental measurements

Author	Conditions	D_L (m^2 s^{-1})	p
Říha et al. [9, 12]	Svitava river $B = 11.5$ m; $H = 1.1$ m; $Q = (3.4–3.7)$ m^3 s^{-1}; $u = (0.4–0.5)$ ms^{-1}	7.2–8.1	15–23
Říha et al. [9]	Svratka river $B = (20–24)$ m; $H = (0.85–1.9)$ m; $Q = 12$ m^3 s^{-1}; $u = (0.25–0.45)$ ms^{-1}	7.2–9.5	21–32
Brady and Johnson [34]	River Wear $B = (20–27.6)$ m; $h = (0.45–1.85)$ m; $Q = (1.62–3.93)$ m^3 s^{-1}; $u = (0.07–0.15)$ ms^{-1}; $i = (0.004–0.17)\%$	4.4–87.22	94.9–2,200
Velísková and Pekárová [11]	Ondava river (upper part) $B = 12$ m; $H = 0.28$ m; $Q = 0.904$ m^3 s^{-1}; $u = (0.35–0.51)$ ms^{-1}; $i = 0.0033$	0.84–1.36	49.3–80
Glover [35]	South Platte River Meandering river $R = 0.46$ m; $Q = 15$ m^3 s^{-1}; $u = 1.33$ ms^{-1}; $n = 0.028$	15.7	500
Glover [35]	Mohawk River Complicated flow conditions, power station, reservoir, inflows $h = 6$ m; $Q = 30$ m^3 s^{-1}	6.0	800
Fischer [17]	Rectangular laboratory flume	0.0072–0.063	8.7–30
Fischer [17]	Triangular laboratory flume	0.123–0.415	190–640

2004–2005 as part of the ISPA project entitled "Development of the Environment in the Liptov Region" (more specifically the connecting collector between Liptovský Hrádok and Liptovský Mikuláš) was selected for field measurements.

The collector has a profile of DN 500 or 600 mm, and it lies in an area of low slopes (from 2 to 9.5‰), which are near to minimal slopes. After more detailed reconnaissance, two collection parts were selected for measurements: the first is above Podtureň village and the second part is just above the Borová Sihoť campsite, both near Liptovský Hrádok. In the first part (Fig. 10), distributions of tracer concentration in time were measured at various distances in a straight line.

In the second part, there is some curving of the sewer track (30°, 45° and 90° trajectory diversion) in which the distributions of tracer concentration in time were measured in various parts of the sewer (Fig. 11). The aim was to determine the influence of these trajectory diversions on the magnitude of the dispersion coefficient.

Common salt (NaCl) was used as a tracer, and this influenced the variation in wastewater flow conductivity. Fluorescein dye was added to the tracer to monitor the course of the tracer substance along the mensural profile. The dosage of tracer was 5 L, which was discharged into the sewer in a single injection (Fig. 12).

The measurement of conductivity was performed with a portable conductivity metre in a mensural manhole. The conductivity metre probe was situated in the
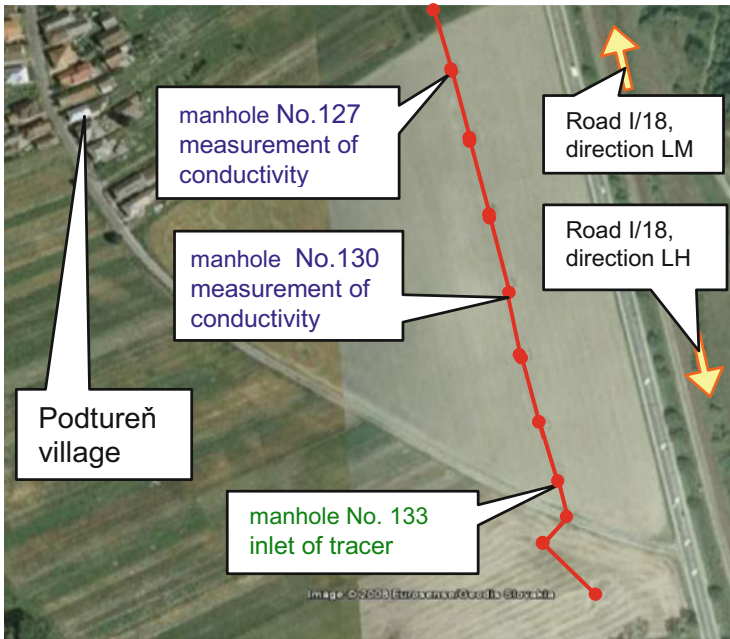
Fig. 10 Sewer collector – straight part (above Podtureň village)

centre of the wastewater flow. The conductivity values were shown on the conductivity metre display in the digital form. A stopwatch was located next to the conductivity metre (see Fig. 13). Fluctuation in the values was recorded by the camcorder, and the record of measurement was manually digitalised subsequently.

The output of measurements is the record of tracer concentration distribution in time in terms of the distribution of wastewater conductivity in the sewer. Examples of graphical expression of this record are given in Figs. 14, 15 and 16.

Evaluation of the experimental results consists in the simulation of the tracer experiment (concentration distribution) for various values of the longitudinal dispersion coefficient.

The basis for this numerical simulation is the analytical solution of Eq. (1) for instantaneous injection of tracer [36]:

$$c\left(x,t\right) = \frac{G}{2A\sqrt{\pi D_L t}} \cdot \exp\left[-\frac{\left(x - \bar{u}t\right)^2}{4D_L t}\right] \tag{3}$$

where $c(x,t)$ is the mass concentration at one place and time, D_L is the longitudinal dispersion coefficient, A is the discharge area in the stream cross-section, G is the mass of the tracer, u is the mean velocity and x is the distance and t is the time.

The difference between the measured and simulated values was evaluated. The minimum difference determined the value of the longitudinal dispersion coefficient

Fig. 11 Sewer collector – incurving of sewer track (near Borová Sihoť campsite)

Fig. 12 Tracer preparation, conductivity measurement
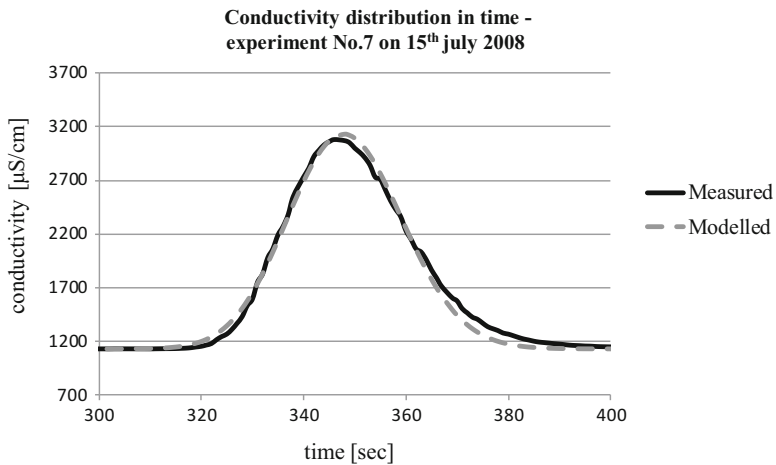
Fig. 13 Measuring device in a manhole

Fig. 14 Behaviour of conductivity in mensural profile (manhole no. 127), experiment no. 7, 15th July 2008 (injection of tracer at manhole no. 133)

for each of the experiments. Although the probe was located in the streamline, it ignored the irregular distribution of concentration (conductivity) across the width of the mensural profile. This was the reason for adding a correction coefficient to the model calculations, which was derived from the ratio of inflow tracer volume and outflow tracer volume.

For the experiment series with lower discharge (and thus also lower water level), the measurement processing revealed that in the given measurement section, there was a stream flow obstacle, creating a kind of "dead zone" in it. Most significantly

**Concentration distribution in time -
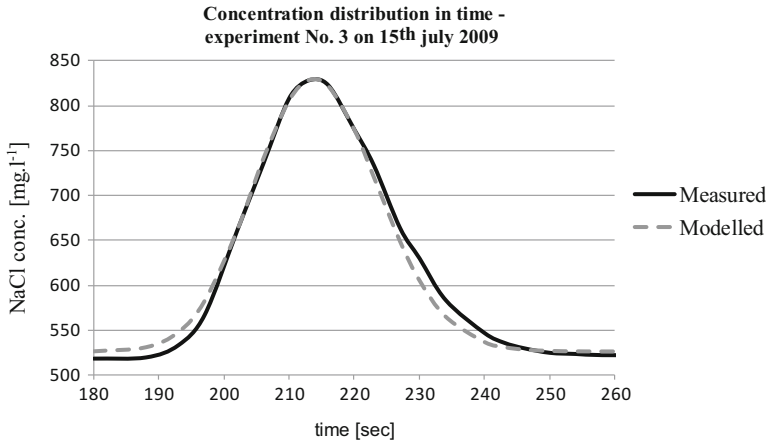experiment No. 3 on 15th july 2009**

Fig. 15 Behaviour of concentration distribution in mensural profile (manhole no. 164), experiment no. 3, 15th July 2009 (injection of tracer at manhole no. 167)
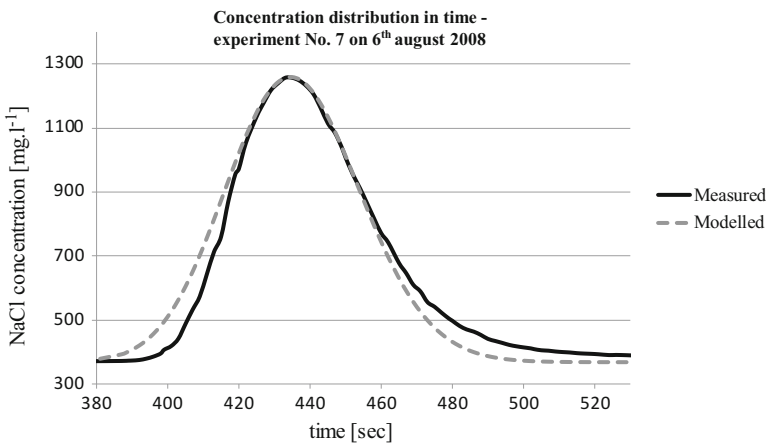
**Concentration distribution in time -
experiment No. 7 on 6th august 2008**

Fig. 16 Behaviour of concentration distribution in mensural profile (manhole no. 127), experiment no. 7, 6th August 2008 (injection of tracer at manhole no. 133)

this phenomenon showed up in the section with the 90° bend (there were also the lowest water depths). Figure 15, therefore, incorporates one of the conductivity examples exactly from this section. The tracer accumulated in this dead zone and was released gradually later. That distorted the conductivity distribution curve, making it asymmetrical and giving it a "tail" because of the later tracer release. This can be seen in the left part of Fig. 16. For this reason, for evaluation, we considered as decisive the concentration rising wave part.

The results of field measurements in the sewer network show the values of longitudinal dispersion coefficient ranging between 0.09 and 0.12 $m^2 s^{-1}$ in the straight

part and between 0.03 and 0.07 $m^2 s^{-1}$ in part with modifications to the sewer track direction.

Field measurements were also performed in various surface water bodies in Slovakia. Two typical cases of stream types in Slovakia were selected. The first one (further marked as "case A") is a typical lowland stream (Malá Nitra stream), where the water velocity and turbulence are very low. For contrast, in the second case ("case B") we picked out a typical mountain stream (upper reach of the river Hron) with a high degree of turbulence.

7.2.1 Case Study A, Malá Nitra Stream

Field measurements were performed along an approximately 400 m section of the stream Mala Nitra, close to the village of Veľký Kýr (Fig. 17). This stream is situated in the south-western part of Slovakia; measurements were taken in Veľký Kýr settlement region (N +48° 10′ 50.02″, E +18° 9′ 19.60″). The regime of the stream discharge is affected by flow regulation in the form of a weir located 15 km upstream at the bifurcation point with the Nitra River. Cross-sections were initially been trapezoidal, but the discharge area along the stream has been slightly changed by natural morphological processes over the years. The longitudinal bed slope was 1.5‰. Measured discharge values during field experiments were within the interval (0.138–0.553) $m^3 s^{-1}$.

7.2.2 Case Study B, River Hron

The mainstream flowing through the town Brezno is the river Hron. This river has a partially alpine runoff regime with maximum flows in April and minimum flows in January. The average annual discharge of the Hron in the Brezno profile, at river km point 243,200, is about 8 $m^3 s^{-1}$.

Fig. 17 Measurements along the Malá Nitra stream (measurements of conductivity in cross-section profiles on the left; comparison and calibration of used conductometers on the right)

Table 3 Values of longitudinal (D_L) and dimensionless (p) dispersion coefficient

		Case study			
		A (Malá Nitra stream)	B (Hron river)	Sewer pipe (prismatic channel bed)	
				Straight part	Part with modifications of track direct
Analytical solution (Eq. 3)	D_L (m² s⁻¹)	0.12–0.18	1.05–1.45	0.09–0.12	0.03–0.07
	p (−)	5–11	21–53	11–25	5–39

The adjusted river bed has a trapezoidal cross-section shape, and the river bed is partially stabilised with stone backfill. Over the years, the shape of the cross-section profile has been partially modified by natural morphological processes (as well as by the stone backfill). The average cross-sectional velocity was 0.64 m s⁻¹, but locally the velocity reached values up to 1 m s⁻¹. Measured discharge values during the fieldwork ranged from 4.2 to 5.3 m³ s⁻¹. The longitudinal bed slope was more than 3‰.

To determine the longitudinal dispersion coefficient, a tracer with a known quantity and concentration was discharged into the geometric centre of the stream width at the beginning of the measured section in a single injection. A solution of common salt (NaCl) was used again as the tracer, causing a change in the flowing water conductivity.

The velocity distribution and discharge were measured at each cross-section for all tracer experiments. Subsequently, the time courses of tracer concentration were monitored at each measured cross-section of the stream. Measured cross-section profiles were distributed evenly along the length of the examined section. Conductivity measurements were completed with portable conductivity metres, located in the centre or evenly across the cross-section width. Measurements were always carried out from the beginning of the increase in conductivity values (front of the tracer wave) until the original (background) conductivity values were restored in each cross-section profile. Each run of the tracer experiment was repeated at least two times.

The same methodology was used here for calculation of D_L values as in the case of measurements in the sewer pipe. Values of the longitudinal and dimensionless dispersion coefficient from all field experiments are summarised in Table 3.

It can be seen that the higher values of longitudinal dispersion coefficient are typical for a natural stream with a higher degree of turbulence. Despite this, D_L values obtained from the lowland stream (case A) are only slightly higher than those from the prismatic channel/sewer pipe measurements. Moreover, the results show that it is necessary to consider the influence of transverse flow in the assignment of longitudinal dispersion values in curved parts of the stream.

8 Conclusions and Recommendations

The issue of water quality is the key to sustainable human development. This chapter gives basic information about terms linked with questions and problems of water quality and contaminant spreading in natural streams in Slovakia. Brief information is also given about legislation in the area of water resource protection valid in

Slovakia nowadays. Despite all the activities concerning water quality protection, several challenges still face us.

By the EU Water Framework Directive, the protection of water resources in use or resources prepared for use to meet a water management need should be construed as the integrated protection of quality and quantity of groundwaters and surface waters. The issue of the sources of water pollution, with either direct or indirect impact on water resources, is the decisive factor in the protection of water resources quality. Water quality protection is based on maintaining the possibility of utilising the water (water resources) for the required purpose. Accordingly, the objective is not to prevent the transport of pollutants into the waters but to maintain their quantity and concentration at such a level that the long-term utilisation of water is rendered possible.

If humans are to manage water resources, it is necessary to know the demands for water from various aspects, and the possibilities, dangers and risks involved, but also the processes of water flow and pollutant transport and spreading. Knowledge of these processes helps us to predict the future status of water resources and design suitable hedges against damage to water resources. These processes in natural conditions are so complex that without using numerical methods and computers, this would be impossible. On the other hand, the outputs of simulation models are only as reliable as their inputs.

Nevertheless, numerical models are useful tools for resolving water quality issues. They need a particular volume of input data, but they also make it possible to evaluate various alternatives of precautions and remedies. One of the crucial parameters or inputs of models simulating contaminant spreading in natural streams is the dispersion coefficient. Its value strongly influences the simulation and calculation results. It is necessary therefore to determine its value as correctly as possible.

One of the ways of determining the dispersion coefficient values is through tracer field measurements. This method covers all typical peculiarities in evaluated conditions or backgrounds in the field. For comparison of different conditions, we performed tracer experiment in a sewer pipe and in two natural streams. The sewer pipe was selected as a model of a prismatic channel with/without modifications to track direction, and in the case of natural streams, we tried to choose different types of streams. The Malá Nitra is a typical lowland stream with low velocities and turbulence, whereas the river Hron is a mountain stream with high velocities and turbulence. Both streams are typical for specific regions in Slovakia.

Results from our sewer pipe experiments show that in comparison with the values gained from the straight line section, it is evident that the values of D_L in the curved part are lower than in the straight, direct line. In contrast, the values of the dimensionless coefficient p show more significant diffusion, and the values specifically in the curved part are higher. These results show and confirm that it is necessary to consider the influence of transverse flow too in the assignment of longitudinal dispersion values in curved sections of a stream. In this case of curved line route, the transverse flow influences the dispersion mechanism in the watercourse. The values of D_L determined using empirical formulas with the inclusion of this assumption also correspond to measured values better than relations which are derived only from longitudinal diffusion process assumptions. Since the shapes of measured curves of distribution of

conductivity show the occurrence of "dead zones" in several parts, the empirical relations which were also used included this fact. The range of values of D_L calculated using these empirical relationships for the hydraulic parameters of each channel and given discharge was nearly identical with measured values.

There were three angles of curvature in the measured route part: 90°, 135° and 105°. The influence of the angle of curvature of longitudinal dispersion coefficient value has so far not been traced down from implemented measurements.

Comparison of measured and calculated longitudinal dispersion coefficient values in curved stream sections confirms that the measured values are near to the values found in laboratory conditions. This fact results from similarity of flow conditions in the selected sewer section to those in a laboratory channel.

Results from tracer experiments in natural stream conditions show that despite careful selection of the investigated stream section, in both cases the hydraulic conditions did not meet ideal flow conditions and that the investigated channel parts were not so prismatic as we supposed. In the case of study A, the reasons were sediments, zones with relatively thick silts or other objects deforming the velocity field and retention in so-called dead zones, which caused deformation of the tracer cloud. In the case of study B, the reason was the irregular distribution of large rocks (boulders) in the river bed, which formed areas with significantly different flow velocities. Such areas with different flow velocities generate a "meandering" stream line with significant deformations. All results show that the spread of tracer was not optimal, preferential flows were established and thus distortions of tracer cloud occurred.

However, as expected, the values of dispersion coefficients were higher in case of study B on the river Hron.

The obtained results and experience can be used for numerical simulation and prediction of water quality and contaminant transport in the investigated streams or similar types by using the values of the dimensionless dispersion coefficient. Using numerical models, it is also possible to design alternative solutions for treated wastewater release back into recipient water bodies without any risks to the water quality and biota.

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