Nutrient Management in the Danube Basin and its Impact on the Black Sea
daNUbs EVK1-CT-2000-00051

Deliverable 5.11. Revised regional emission model

by
Horst Behrendt,
Dieter Opitz
and Heide Schreiber

Leibniz Institute of Freshwater Ecology and Inland Fisheries,
Department of Shallow Lakes and Lowland Rivers Berlin
Müggelseeedamm 310, 12587 Berlin

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Contents

I. Changes of the database and model approaches

Horst Behrendt[1] and Dieter Opitz[1]

I.1 Changes of the Database

Within the Deliverable 5.4 of the project and Schreiber et al. (2003) a harmonised and GIS supported database was established for the total Danube basin. This database was raw for certain parameters as especially Soil, Hydrogeology, Erosion and waste water treatment plants. Within the work of Deliverable 5.11 the resolution of some of these digitised maps could be increased by involvement of a new geological map for the Danube according to *Regional Co-operation of the Danube Countries (Regionale Zusammenarbeit der Donauländer)*, 1986, in "Die Donau und ihr Einzugsgebiet" part 3 (see Figure I.1). This geological map could be used especially for a better distinguish between consolidated and unconsolidated rocks as well as between the unconsolidated rocks with shallow and deep groundwater layers. The use of the new geological map has improved the model results for some of the subcatchments as the Wulka.

Within the revision of the database and the model it was also planed to establish a new map for the soil erosion based on the USLE. For this task the new dataset of the digital elevation model was prepared with a resolution of 90 m according to the SRTM data of the NASA (ftp://e0mss21u.ecs.nasa.gov/srtm/Eurasia/). This map is show in Figure I.2. From this map a slope map was also derived. The establishing of a new soil loss map could not be finished, because the European soil map in the scale 1:1,000,000 was not available for the project partner. Further the co-worker could not continue this work, because of a complicated pregnancy.

A database for all Austrian WWTP's larger than 2000 population equivalents was provided by the Austrian Environmental Agency (UBA Vienna) for the use within the daNUbs project. By this WWTP data the available information of the point source inventory was increasing from 3453 WWTP datasets to 3957 (see Figure I.3).

Further enlargements of the database were due to the modelling of the historical changes of the nutrient loads into the Danube river system and the scenario calculations according to Deliverable 5.12. These specific enlargements are presented in chapter III of this Deliverable as well as in 5.12 (van Gils et al., 2005).

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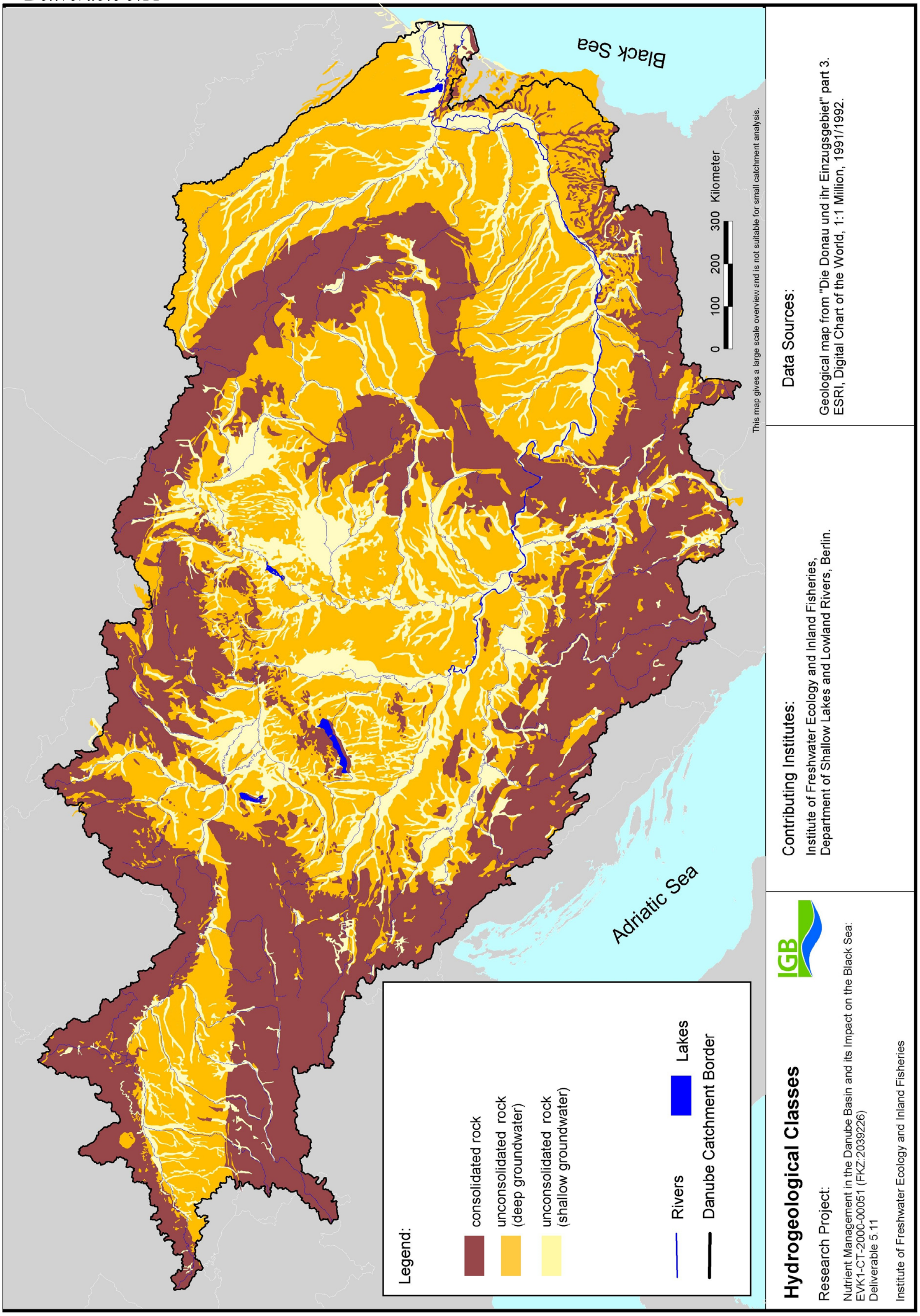

Figure I.1 Hydrogeological classes used by MONERIS according to *Regional Co-operation of the Danube Countries (Regionale Zusammenarbeit der Donauländer)*, 1986, in "Die Donau und ihr Einzugsgebiet" part 3

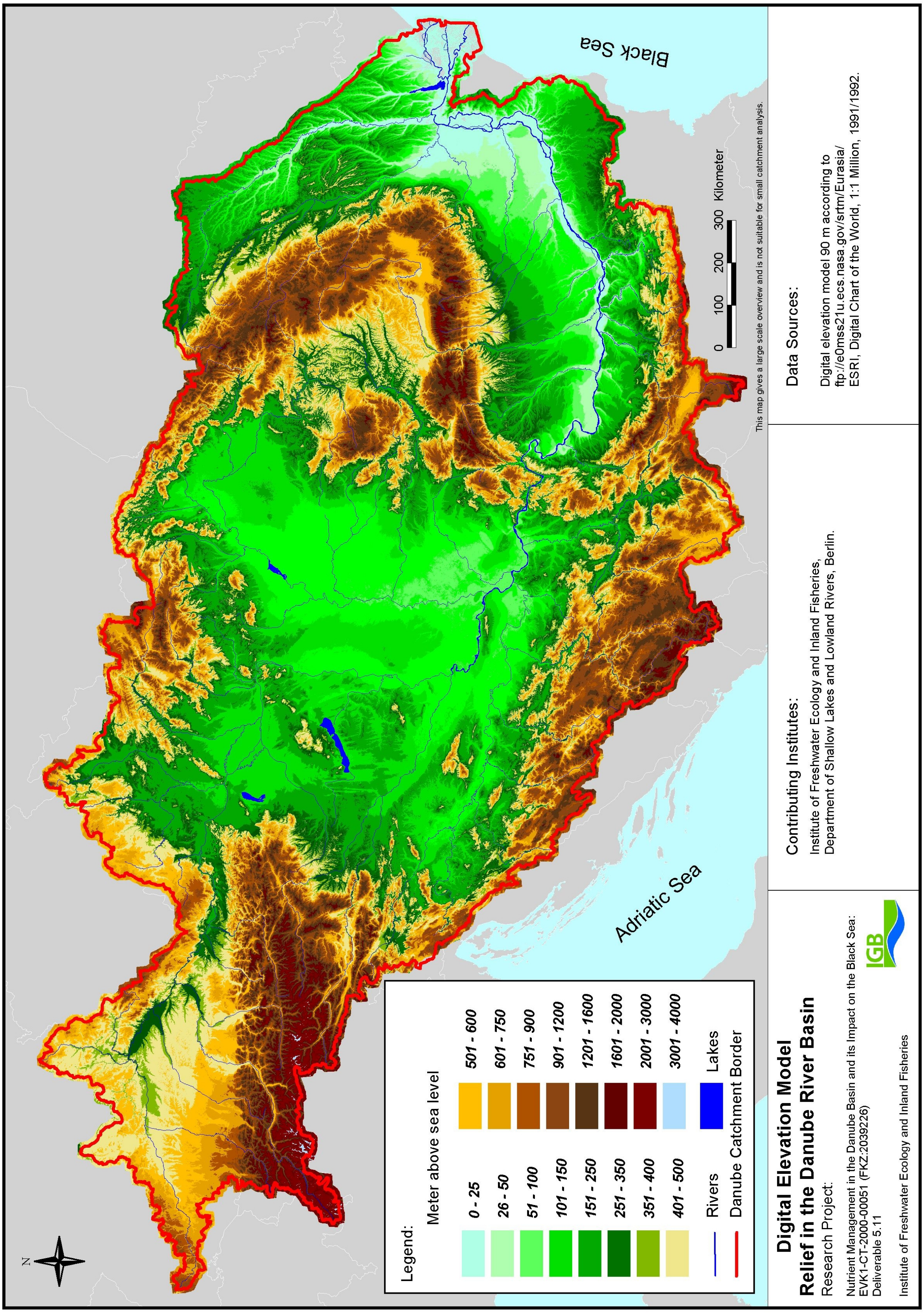

Figure I.2 Digital elevation model for the Danube with 90m resolution according to ftp://e0mss21u.ecs.nasa.gov/srtm/Eurasia/

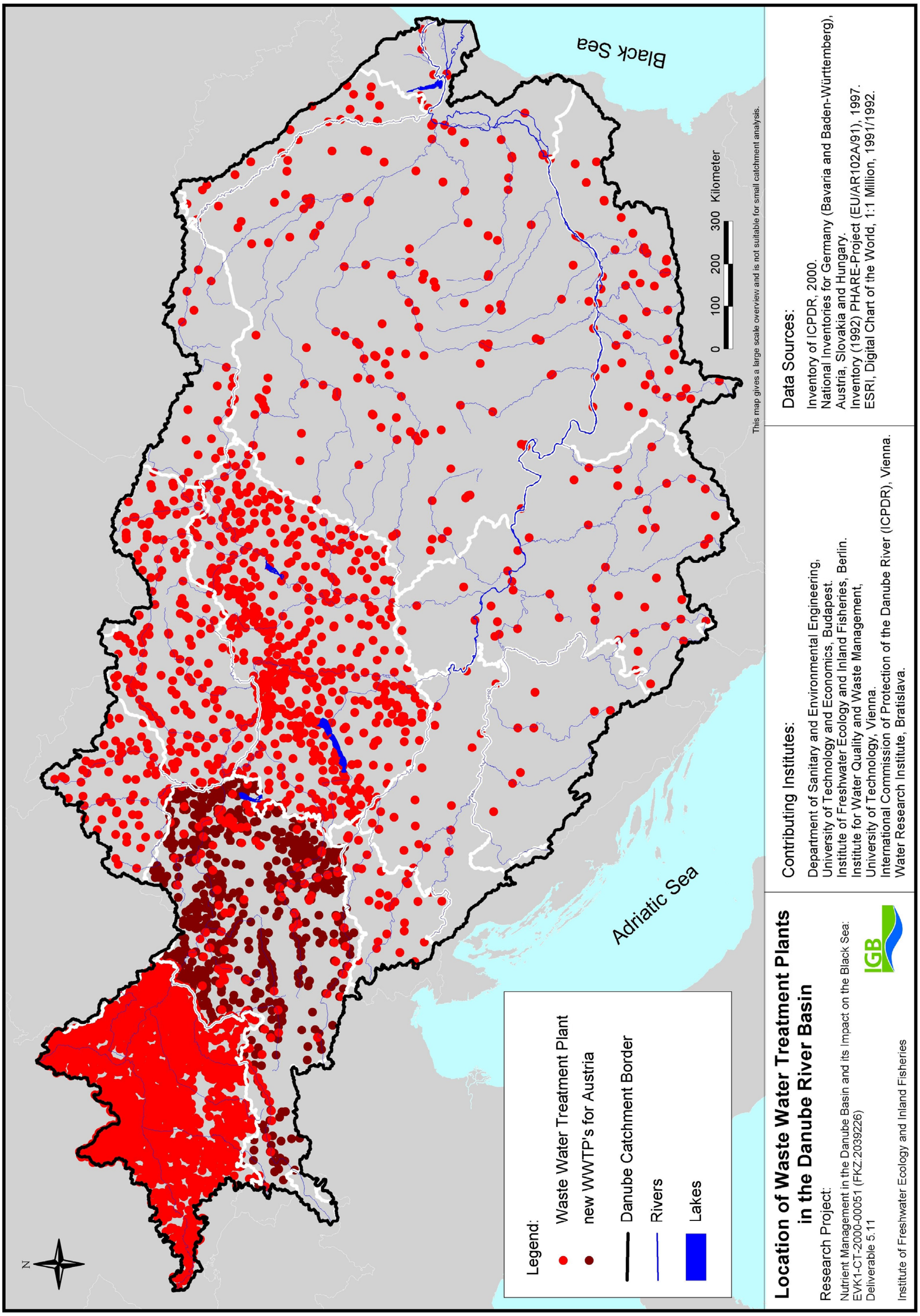

Figure I.3 Map of waste water treatment plants in the Danube basin according to ICPDR point source inventory and national inventories for Austria, Germany, Hungary and Slovakia.

I.2 Changes of the model MONERIS

For the modelling of the historical changes and the scenarios the model structure was enlarged by an additional file, which includes all data of the Danube countries and the main sub-basins of the Danube which are changing over time. Temporal changes of the sub-national level (e.g. districts) were not considered due to missing data for the most of the countries.

The changes of the applied model MONERIS are limited only to the modelling of the emissions dissolved phosphorus by surface runoff. The changes were necessary, because the assumptions on the dissolved P-concentrations for surface runoff of agricultural land were only based on German conditions and could not be transferred to other countries of the Danube, where the P-accumulation in the top soils was substantial lower than in Germany.

In the old version of the model the dissolved P-concentrations of surface runoff from arable land and pasture were set as constant (see Figure I.4).

The new approach takes into account that P-concentrations given in Figure I.4 will be reached for the German part of the Danube.

For the other countries these concentrations are calculated depending on the following

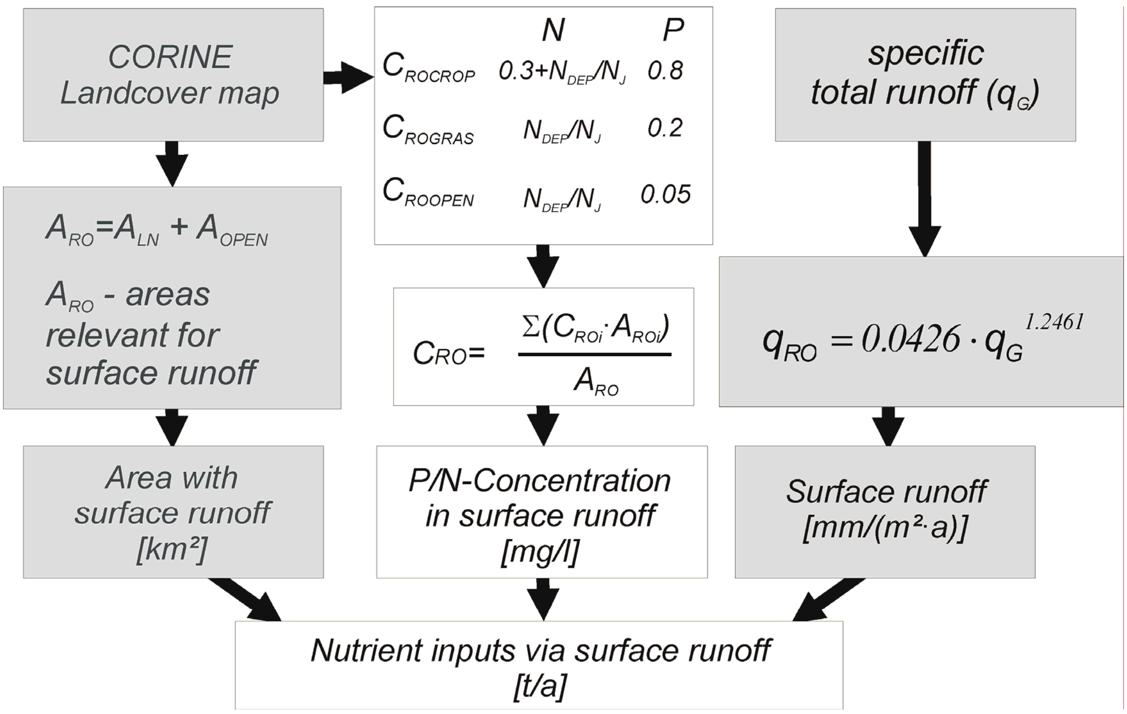

Figure I.4 Diagram for the estimation of the nutrient emissions by surface runoff into a river system according to SCHREIBER et al. (2003).

equations:

$$C_{ROCROP} = a + b \cdot \exp(\frac{P_{ACCX}}{P_{ACCDE}} \cdot c_{SATCROP}) \quad (1)$$

$$C_{ROGRAS} = a + b \cdot \exp(\frac{P_{ACCX}}{P_{ACCDE}} \cdot c_{SATGRAS}) \quad (2)$$

with $C_{ROCROP_{N,P}}$ = nutrient concentration in surface runoff from arable land (mg/l),
$C_{ROGRAS_{N,P}}$ = nutrient concentration in surface runoff from grassland (mg/l),
P_{ACCX} = P accumulation in the agriculture of the country X (kg/ha),
P_{ACCDE} = P accumulation in the agriculture of Germany = 880 kg/ha,
a = constant (0.035),
b = constant (0.000000618),
$C_{SATCROP}$ = constant (14 or 90% of saturation level) and
$C_{SATGRAS}$ = constant (12.4 or 80% of saturation level).

The mean concentration of P in the surface runoff can be calculated by using equation (1) and (2):

$$C_{RO_{N,P}} = \frac{C_{ROCROP_{N,P}} \cdot A_{AR} + C_{ROGRAS_{N,P}} \cdot A_{GRAS} + C_{ROFOR_{N,P}} \cdot A_{FOR} + C_{ROOP_{N,P}} \cdot A_{OP}}{A_{AR} + A_{GRAS} + A_{FOR} + A_{OP}} \quad (3)$$

with $C_{RO_{N,P}}$ = nutrient concentration in surface runoff (mg/l),
A_{AR} = area of arable land (km²),
A_{GRAS} = grassland area (km²),
A_{FOR} = area of forest(km²),
A_{OP} = open area (km²),
$C_{ROFOR_{N,P}}$ = nutrient concentration in surface runoff from forest (mg/l),
$C_{ROOP_{N,P}}$ = nutrient concentration in surface runoff from open land (mg/l).

For forest and open land the level of P-saturation of the soils was assumed to be below 50%. Than the P-concentration for both landuse categories is nearly the constant a in equation (1) and (2) or 0.035 mg/l P.

Using this new approach the model calculates for the past and also for the present state of the Eastern European countries a substantial lower P-concentration for the surface runoff. All other approaches used for the modelling are in the revised version of MONERIS the same as given by Schreiber et. al (2003) in Deliverable 5.4.

In the following chapters the results of the model calculations with the revised database and model approaches are given.

II. Present state of significant human drivers in the countries of the Danube basin

Horst Behrendt[2]

II.1 Introduction

Whereas the load of substances from point discharges can be measured or calculated from measured concentrations and flows, the emissions of substances from diffuse sources cannot be measured. For small watersheds the loads can be estimated but for medium and large river catchments the estimation of the diffuse source pollution is only possible by mathematical modelling. This is done using land use, hydrological, soil and hydrogeological data collected in a Geographical Information System (GIS) as well as statistical information for different administrative levels.

The definition of significant sources of pollution for the diffuse emissions is a very complex theme. This is especially the case for large transboundary river basins such as the Danube. The main problem is to distinguish between areas with low and high levels of diffuse pollution. These levels are not only dependent on anthropogenic factors such as land use and land use intensities, but also on natural factors such as climate, flow conditions and soil properties. These factors influences, the pathways of the diffuse nutrient emissions and the retention and losses on the way from the origin to the inputs into the river system. Absolute values of the significant diffuse source of pollution are also difficult to define. This is because the level of the intensity of land use as the main indicator for the diffuse emissions into the river is also dependent on the population density in the catchment area.

Criteria for estimating the significant diffuse sources, which ignore the natural and basic anthropogenic conditions are not reliable for distinguishing between significant and insignificant levels. Furthermore, a number of uncertainties need to be taken into account when analysing the data.

II.2 Present state of the nutrient point discharges

The total nutrient point discharges into the Danube was about 134.4 kt/a nitrogen and 22.7 kt/a phosphorus in the year 2000 (see part II).

Figure II.1 and Figure II.2 shows the difference in the present state of the specific nutrient point source discharges within the Danube countries. For these figures the estimated point discharges of nutrients for the individual countries were divided by the population in the countries which is connected to Waste water treatment plants (WWTP). For ni-

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trogen it is shown that the lowest point N discharges of 4 g/(Inh.·d) per connected inhabitant is realised by Germany followed by Austria and Czech Republic. The Figure represents the differences of the level of waste water treatment between the countries. For some countries the specific N discharges are higher than the assumed N emission per inhabitant of 12 g/(Inh.·d). This is due to the fact that the point source database includes industrial discharges emitted into the river indirectly (via sewer system) and directly (industrial point sources) and the low level of nitrogen removal in most of the WWTPs of these countries (dominance of mechanical treatment).

The picture for phosphorus presented in Figure II.2 is similar to that for nitrogen (Figure II.1), but the differences between the countries are much larger. This is due to the fact that the specific P point discharges reflect, not only the state of the P elimination in waste water treatment plants, but also the existing use of phosphorus in detergents, and discharges from direct industrial sources. This is the reason that the specific P emissions are above 1 g/(Inh.·d) for Slovenia, Croatia, Serbia and Montenegro as well as Bulgaria. The medium level P emissions between 0.5 and 1 g/(Inh.·d) were found for Czech Republic, Slovak Republic, Hungary, Bosnia Herzegovina, Romania and Moldova. Beside Germany and Austria, the specific point P discharges are also below 0.5 g/(Inh.·d) for Ukraine. These relative low specific P emission for Ukraine is probably due to an underestimation of the P point source discharges in the point source inventory of ICPDR or an overestimation of people connected to WWTP.

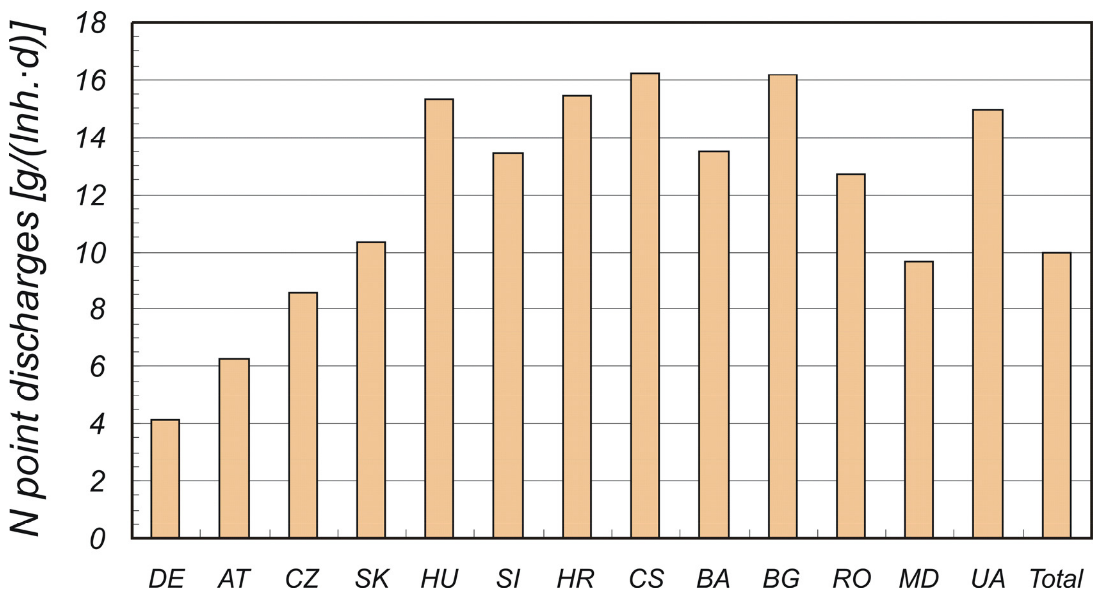

Figure II.1 Inhabitant specific N discharges from point sources in the Danube countries for the period 1998 to 2000 according to SCHREIBER et al. (2003).

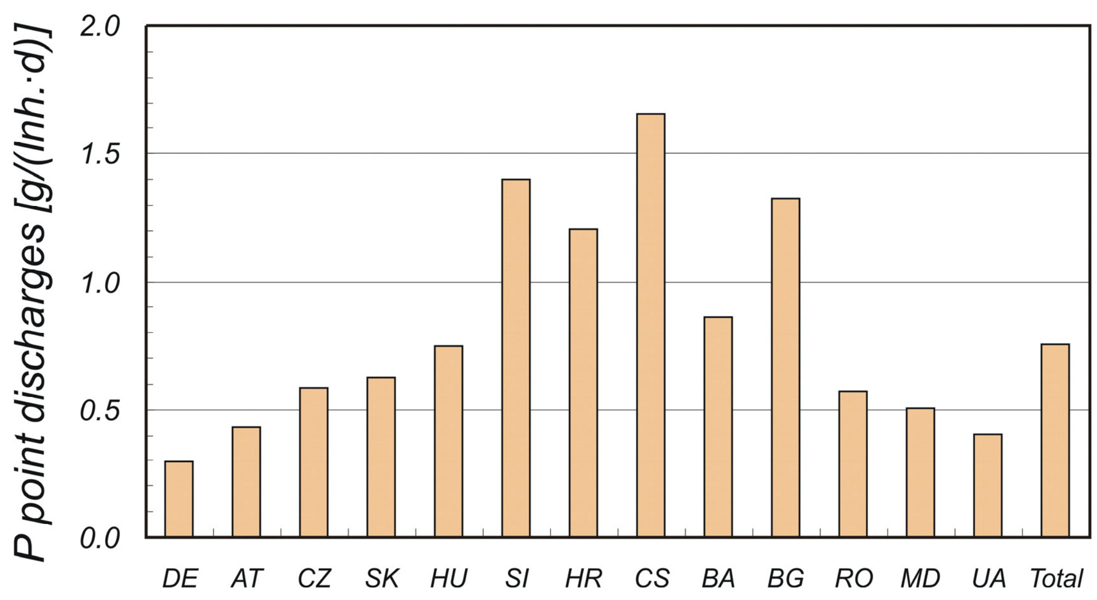

Figure II.2: Inhabitant specific P discharges from point sources in the Danube countries for the period 1998 to 2000 according to Schreiber et al. (2003).

II.3 Land use patterns and agricultural indicators

The Danube basin is characterized by large gradients of anthropogenic and natural indicators, which are important for affecting nutrient inputs into the river system. One indicator for the level of the diffuse emissions of substances can be the land use within the basin and its regional distribution.

Figure II.3 gives an overview of the portion of differing land uses, arable land, grassland and pasture, forest and other kinds of land use, related to the total area of the Danube countries. The use of these country averages does not allow a calculation of an average for the total Danube river basin. The figure shows an increase of the share of arable land, and a decrease of forest, from the upper to the lower part of the Danube. Because most countries (Hungary is the exception) have only a portion of their territory in the Danube catchment, the estimation of a Danube average for the land use pattern is not possible using data on the country level. In addition, it must be considered that the average land use for the countries can deviate from the status within the parts of the countries only in the Danube basins. This is due to the inhomogenous distribution of land use within the countries.

Another source for information on the land use patterns in the Danube is the available CORINE land cover map. This data is not yet available for Croatia, Serbia and Montenegro, Ukraine and Moldova. Schreiber et al. (2003) tried to fill this gap by transferring of the USGS land cover map into the classes of CORINE. A similar procedure was ap-

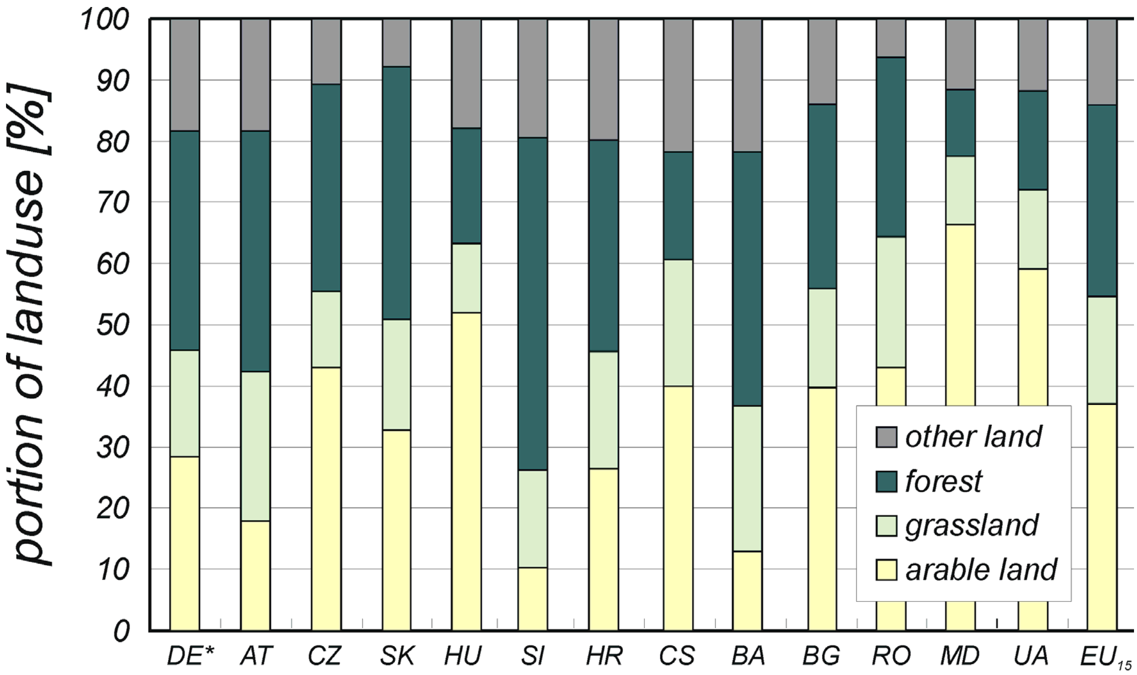

Figure II.3 Portion of land use types in the total area of the Danube countries for the period 1998 to 2000 (data source FAO the exception is Germany - DE* represents the land use for Baden-Württemberg and Bavaria according to the German Federal Statistical Office for the same period).

plied to establish the land cover map 6 for the total Danube River basin. As shown by information from Schreiber et al. (2005) for Bosnia and Herzegovina, such a transfer can lead to substantial deviations for the land use pattern. The advantage of using the land use patterns according CORINE is that it contains the higher segmentation for the land use classes, and the possibility to estimate the land use for the river basin, as well as the sub-catchments.

Figure II.4 shows, the land use patterns for those parts of the countries within the Danube basin, and the average for the whole Danube. If both figures are compared, it is obvious that the estimated portions of the arable land are higher based on CORINE data.

These differences are due to the different classification systems being used. The national statistics represent the actual uses of land whereas the CORINE data reflects the cover of the land according to the classification of satellite images. Because the resolution for the classification of CORINE is 25 ha, this procedure leads further to an overestimation of the dominant land cover (arable land and forest) and an underestimation of the other classes. For the total Danube the share of the land use is: arable land 47.4 %, grassland and pasture 6.2 %, forest 33.5 %, urban areas 3.9 %, surface water area 0.9 % and other areas including open land, wetlands and glaciers 8.0 %.

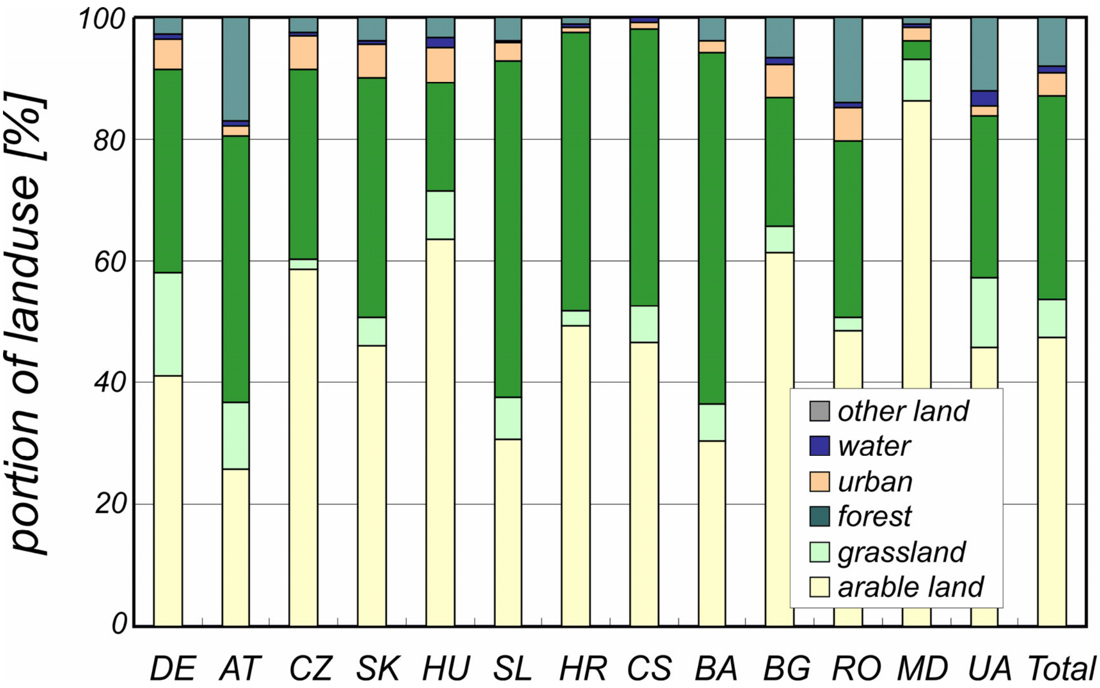

Figure II.4 Portion of land use types at the parts of countries within the Danube basin and the average for the total Danube according to CORINE land cover map and transferred USGS land cover map (source Schreiber et al., 2003).

Besides being influenced by the land use it self, the level of the emissions into the surface waters of a river system is also dependent on the intensity of the land use. Because agriculture activities are a main source for the diffuse nutrient emissions into the river system, it is important to show differences in intensity of use on a unique database. Statistical data for the countries is the best way to do this. Figure II.5 shows the consumption of nitrogen fertilizer used in agriculture of the Danube countries. The source of the data is the FAO agricultural statistics for the individual countries for the years 1998 to 2000 (http://apps.fao.org/cgi-bin/nph-db.pl?subset=agriculture). For Germany the information is not national but is from the federal states of Baden-Württemberg and Bavaria where the data of Behrendt et al. (2003) were used based on the German Statistical Yearbook (1999 to 2001). The figure includes also the average value for the 15 countries of the EU (before May 2004), and the maximum value reached within the set of countries. Further the area weighted average for the Danube basin is given.

From Figure II.5 three groups of countries can be distinguished. Germany, Slovenia and Czech Republic are the countries with a consumption of mineral nitrogen fertilizer of more than 50 kg/(ha·a) N, although there is a large difference between the amount of use in the three countries.

In the second group of countries (Austria, Slovak Republic, Croatia and Hungary) the use of mineral fertilizers in agriculture is low to moderate, between 25 and 50 kg/(ha·a) N. In all other countries the level of mineral fertilizer consumption is significantly below 25 kg/(ha·a) N. The area weighted average of consumption of N fertilizer was estimated as 31.4 kg/(ha·a) N for the Danube basin. Comparison with the average of the EU15 countries shows that the level of fertilizer consumption in the Danube basin is less than half this amount. The maximum of N fertilizer consumption reached in the EU15 countries is 5 times higher than in the Danube basin.

If N fertilizer consumption is calculated per inhabitant living in the countries an another picture emerges (see Figure II.6). The deviation between the countries, with exception of Bosnia i Herzegovina and Ukraine, is lower. The Danube average is 16.6 kg/(inh.·a). This is only 64 % compared to the average of the EU 15. The EU 15 maximum is also 4.5 times higher than the average of the Danube basins.

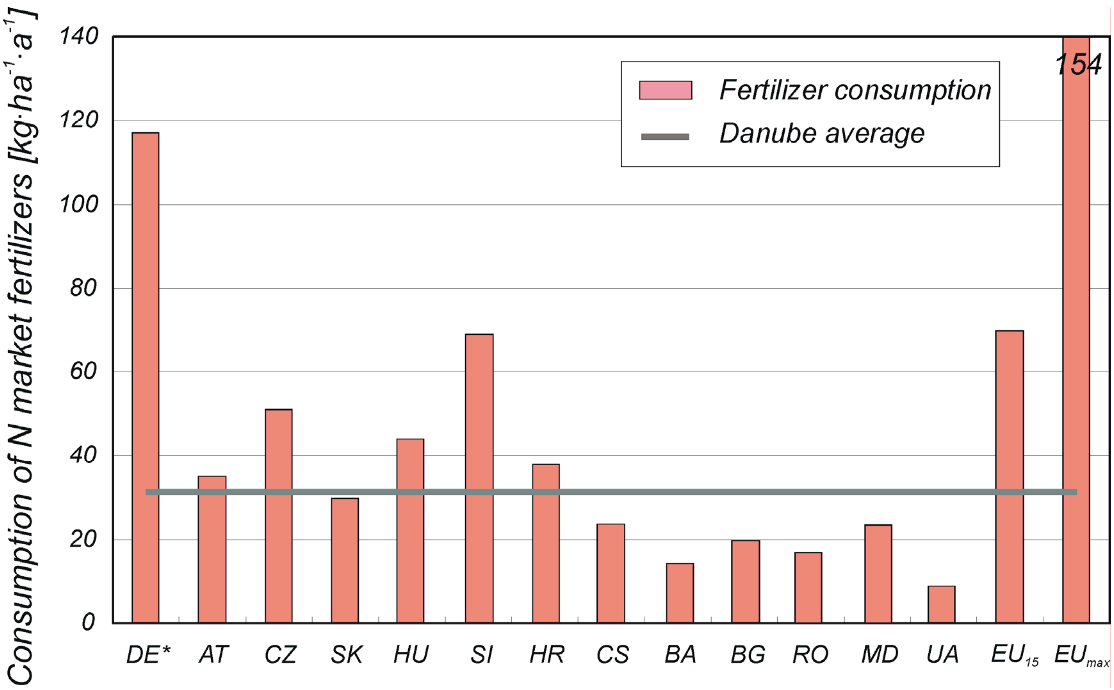

Figure II.5 Consumption of nitrogen fertilizers in the Danube countries, within the EU 15 countries, and EU maximum value in the period 1998 to 2000 (*The bars represent the consumption of nitrogen fertilizers per agricultural area of the Danube countries. The data given for DE* represents the average N fertilizer consumption of the German states Baden-Württemberg and Bavaria. The database is national statistics published by the statistical offices of the countries or by FAO).*

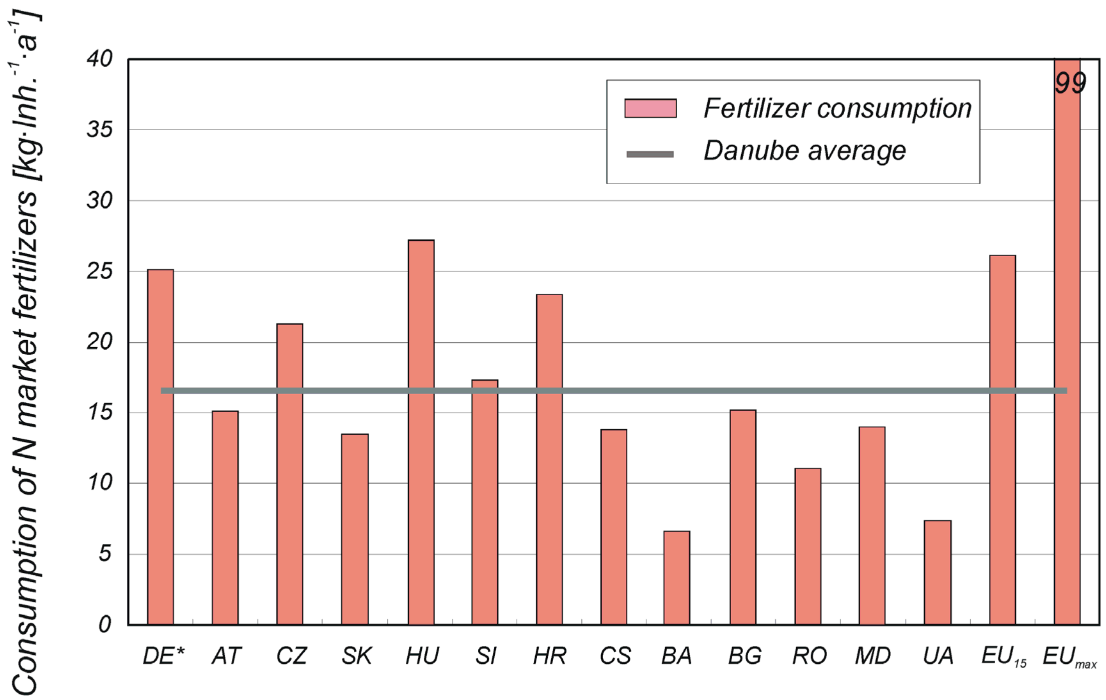

Figure II.6 Consumption of nitrogen fertilizers per inhabitant in the Danube countries, the EU 15 countries, and EU maximum value in the period 1998 to 2000 *(Data sources see Figure 5).*

In addition to the application of mineral fertilizer, the number of livestock is an indicator for determining land use intensities that affect diffuse nutrient inputs. Figure II.7 shows the livestock density as animal units per hectare agricultural area for the Danube countries. The animal unit (AU) corresponds to a live weight of 500 kg. Coefficients used for the conversion of animals of various types into the animal unit differ from state to state. In the Figures II.7 and II.8 the coefficients common in the Czech Republic and Germany, respectively are applied for the purpose of comparison. A systematic deviation is found when using the different equivalents. The animal unit number calculated with the German equivalents is on average only 79 % of the number found for the Czech equivalents. Figure II.7 includes the average for the total Danube basin for both kinds of animal units, as well as this indicator for the average of the EU 15 countries, and the maximum of these countries.

The countries with a density of 1 or 0.8 animal units per hectare and more are Germany, Austria and Slovenia. All other countries have a livestock density lower than 0.5 animal units. The reason for these low densities is that in most countries of eastern Europe there has been a strong reduction of livestock numbers after the changes of socio-economic conditions around 1990. The average density of animal units in the Danube basin is only 55 % of the EU 15 average. The maximum of the EU 15 countries is more than 7 times higher than the average of the Danube basin.

The deviation between the countries for the livestock density is much lower if this indicator is calculated as animal units per inhabitants living in the countries (see Figure 8).

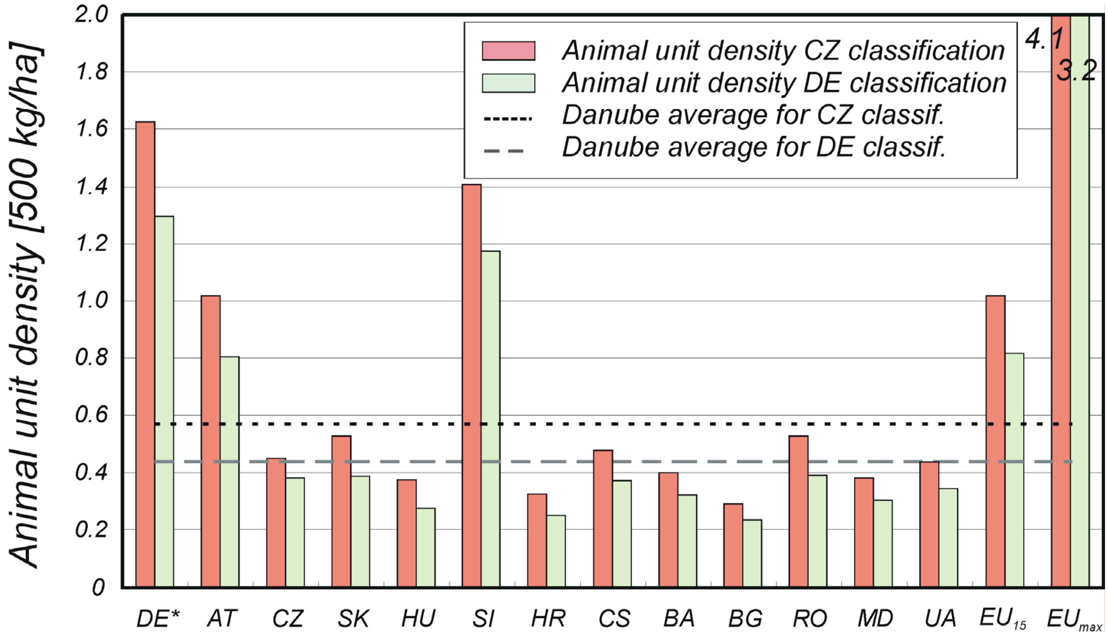

Figure II.8 Animal units per agricultural area in the Danube countries for the period 1998 to 2000. (*The bars represent the animal units per agricultural area in the Danube countries. The data given for DE* represents the animal unit density of the German states Baden-Württemberg and Bavaria. The database is national statistics published by the statistical offices of the countries or by FAO, equivalents for Czech Republic and Germany were used).*

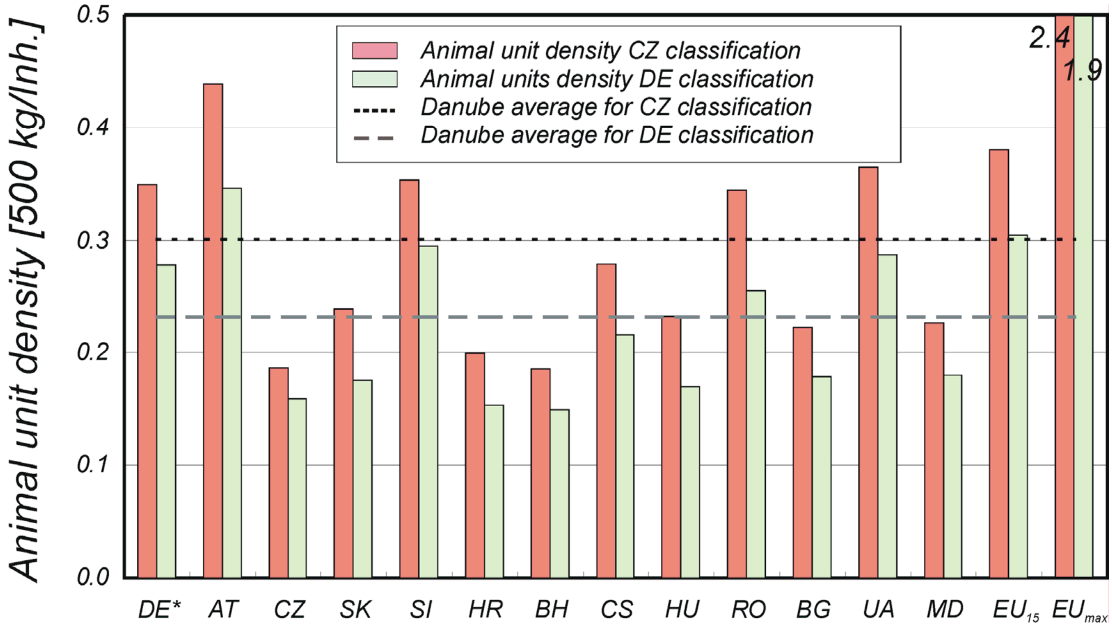

Figure II.7 Animal units per inhabitant in the Danube countries for the period 1998 to 2000 *(Data sources see Figure 7)*

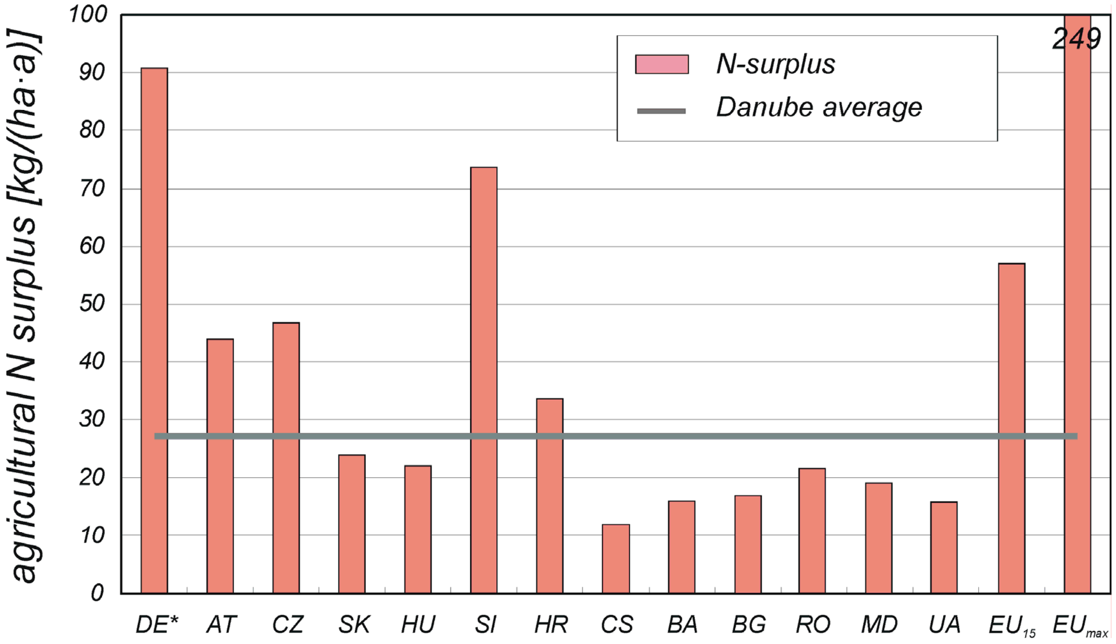

Figure II.9 Nitrogen surplus per agricultural area in the Danube countries for the period 1998 to 2000 *(Data sources Schreiber et al. 2003, based on data of FAO and national statistics for German federal states)*.

The group of countries with a value above the Danube average includes Romania and Ukraine. The average of the Danube is between 75 and 80 % of the livestock density of the EU15 but about 8 time lower than the EU15 maximum. This is due to the lower population density within the Danube than in the EU15 countries.

Consumption of mineral fertilizer and livestock density are the major sources of information on nutrient inputs from agriculture. If the inputs by atmospheric deposition, seeds and for nitrogen N-fixation, and the outputs by harvested crops are taken into account, then the nutrient surplus on agricultural area can be calculated. The procedure for this calculation can differ from country to country and within the countries. The results presented in Figure II.9 are for all Danube countries using the OECD procedure. The coefficients used for the transfer of the different livestock excreta and crops into nitrogen and phosphorus was the coefficients for the Czech Republic. As shown by Schreiber et al. (2003), the N surplus can differ within a minor range if the coefficients or procedures of other countries were applied. It should be pointed out that the application of different sets of coefficients for the individual countries would lead to systematic errors and consequently to incompatibilities of the data.

The high animal density, and the large consumption of mineral nitrogen fertilizer, is the reason that Germany and Slovenia are also the countries with the highest nitrogen surplus per hectare agricultural area (see Figure II.9). The level of the N-surplus was 91 and 74 kg/(ha·a) N respectively for the period 1998 to 2000.

From Figure II.5 and Figure II.7 a higher difference in the N-surplus between Germany and Slovenia could be expected, but higher specific nitrogen outputs by harvested crops partly compensate for the larger fertilizer consumption and higher animal density in Germany. For the second group of countries (Austria, Czech Republic and Croatia) the estimated N-surplus is moderate, between 30 and 50 kg/(ha·a) N. The level of the N-surplus of all other countries is below 25 kg/(ha·a) N. Figure II.9 presents the wide variation in nitrogen surplus between country and indicates that the potential for nitrogen inputs into the surface waters of the Danube from countries also varies widely.

The area weighted average of the N surplus within the Danube basin was estimated as 27 kg/(ha·a). In comparison to the EU15 countries this level of N surplus is only about 47%. The maximum of the EU15 countries is more than 9 times higher than the average of the Danube basin.

Because the annual phosphorus surplus on agricultural area is large part accumulated in the soil, one main indicator for diffuse P emissions into the river system is the longterm P accumulation on the agricultural area. This indicator provides a basis for determining the P emissions by erosion and surface runoff into the river system. Figure II.10 shows the estimated P accumulation on the agricultural area of the Danube countries.

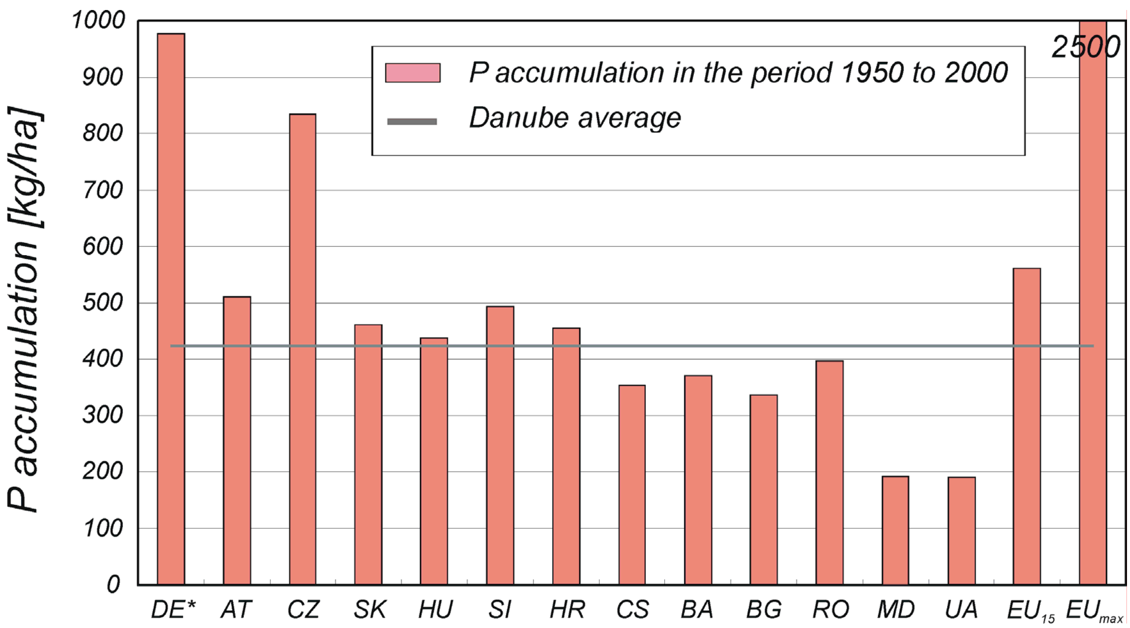

Figure II.2 Phosphorus accumulation on agricultural area in the Danube countries for the period 1950 to 2000 *(Data sources Schreiber et al. 2003, based on data of FAO and national statistics for German federal states)*.

According to Figure II.10 the highest P accumulation was estimated for Germany and Czech Republic. For these countries, the P-accumulation of agricultural soils is about the double of the value for the most of the other countries. Moldova and Ukraine have an estimated P-accumulation, which is half that of most countries.

The nitrogen surplus on the agricultural area, as well as the long term P accumulation on this area, reflects the differences of the intensity of land use. The interpretation of the consequences of these differences between countries involves more than examining the agricultural sector. The level of agricultural intensities in the countries is also dependent on the people living in the region. If consideration is given to this factor then the results will change.

Figure II.11 shows the agricultural area per inhabitant living in the countries. The figure shows that the agricultural area per inhabitant is the lowest in Germany and Slovenia, where only a little more than 0.2 ha per inhabitant are used for agriculture. A second group of countries has an inhabitant specific agricultural area of about 0.6 ha/inh. or more (Croatia, Serbia and Montenegro, Hungary, Romania, Bulgaria, Ukraine and Moldova). This is at least three times higher than for Germany and Slovenia. The Danube average is 0.54. This is about 50 % higher than the EU15 average and more than 4 times higher than the minimum reached within the EU15 countries.

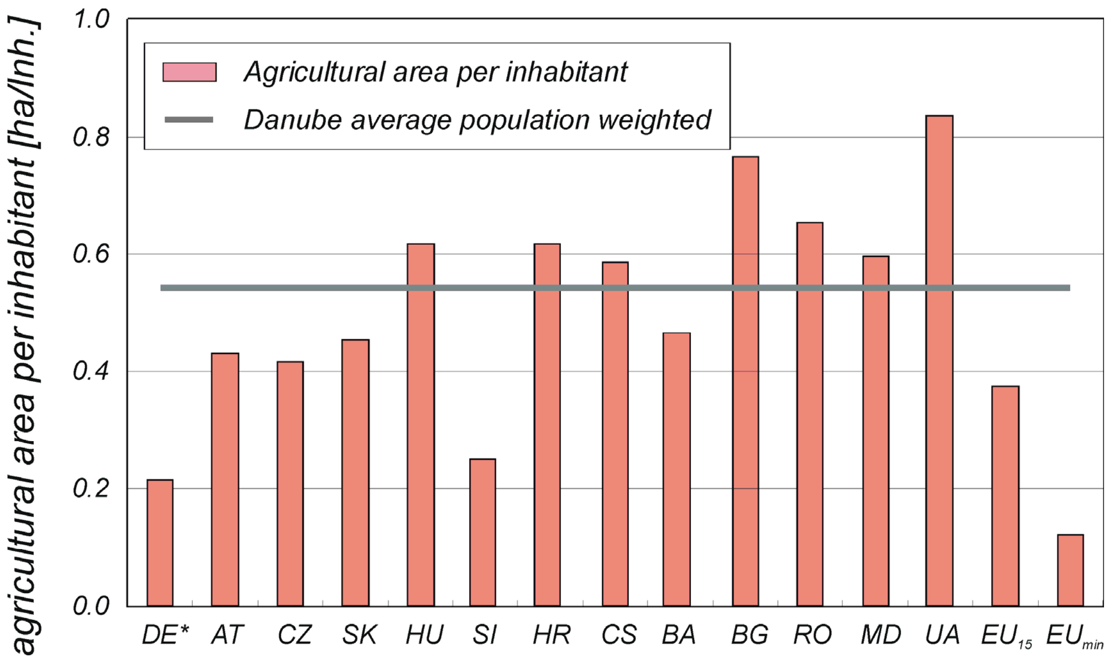

Figure II.3 Agricultural area per inhabitant living in the Danube countries, the EU 15 countries, the minimum in the EU 15 countries, as well as the population weighted average for the Danube basin for the period 1998 to 2000 *(Data sources see* **Figure 9***).*

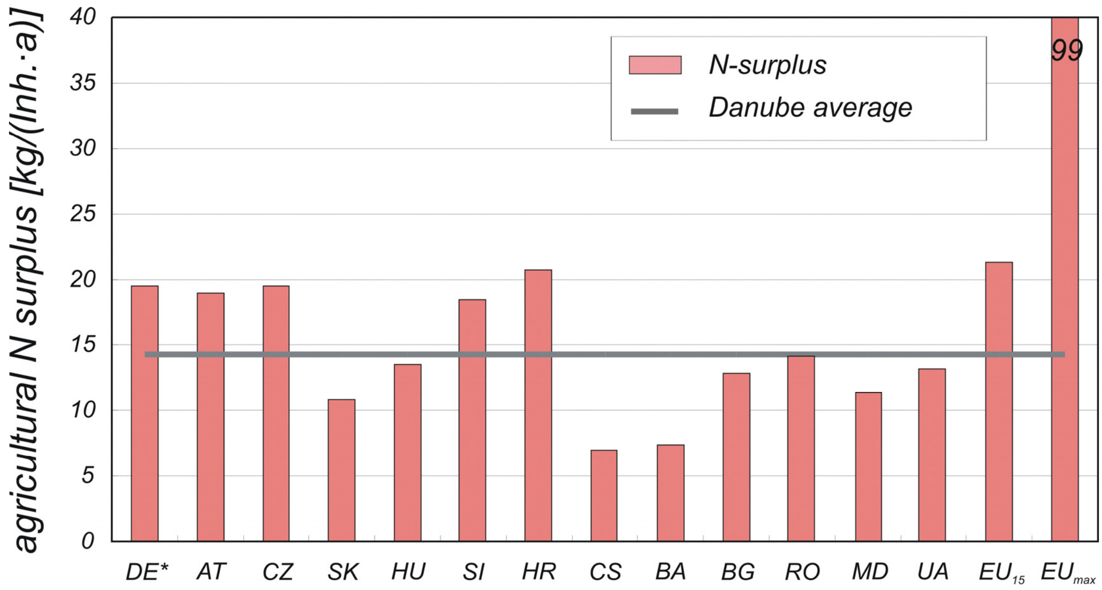

Figure II.4 Nitrogen surplus per inhabitant in the Danube countries for the period 1998 to 2000 *(Data sources see* **Figure 9***)*.

From this, the nutrient surplus per inhabitant can be calculated (see Figure II.11), which shows that the behaviour regarding nutrients is much more similar in the countries than may be deducted from the previous graphs.

Figure II.12 shows that one reason for the very high N surplus in Germany, in comparison to the most of the other Danube countries, is the highest population density. If this is taken into account, the variation of the agricultural intensities is much lower. The N surplus per inhabitant is very similar in Germany Austria, Czech Republic, Slovenia and Croatia (about 20 kg/inh.). The second group of countries (Slovak Republic, Hungary, Romania, Bulgaria, Ukraine and Moldova) possesses an N surplus per inhabitant between 10 and 15 kg/inh. Only for Bosnia and Herzegovina as well as Serbia and Montenegro the N surplus per inhabitant is below 10 kg/inh. The average of the N surplus per inhabitant within the Danube basin was estimated as 14.7 kg/(ha·a) N. This value corresponds to 67 % of the EU15 average and is about 7 times lower than the EU15 maximum.

II.4 Acknowledgements

The presented results originates from the EU research project “Nutrient Emissions in the Danube Basin and its Impact on the Black Sea” (daNUbs) supported under contract number EVK1-CT-2000-00051 by the Energy, Environment and Sustainable Development (EESD) Programme of the EU 5th Framework Programme. Details on the project can be found on the project homepage: http//:danubs.tuwien.ac.at. For data supply, support and helpful discussions we thank especially the members of the EMIS expert group of the ICPDR.

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III. Results of the revised model.

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III.1 Introduction

Increased and unbalanced riverine delivery of nutrients (N, P and Silicone (Si)) is known to cause coastal eutrophication, undesirable algal blooms and modifications of marine foodwebs (CONLEY et al., 1993, BILLEN & GARNIER, 1997, LANCELOT et al. 2002). The ecological functioning of the northwestern coastal area of the Black Sea has been strongly affected by coastal eutrophication during the last few decades (MEE, 1992, MEE & TOPPING, 1999; MEE, 2001, LANCELOT et al., 2002). High nutrient loads and their consequences are recognised as one of the most severe problems of the river Danube catchment area. The Danube, the second largest river system in Europe, with a catchment area of about 802,900 km² represents about 34 % of the total catchment area of the Black Sea (about 2,400,000 km² according to JAOSHVILI, 2002 and BEHRENDT, pers. comm.). The population living in the Danube catchment is about 43 % of the total population in the Black Sea basin (BEHRENDT, pers. comm.). 57.5 % of the freshwater (JAOSHVILI, 2002) and major parts of the sediment and nutrient loads entering the Black Sea originates from the Danube (GLOBAL ENVIRONMENTAL FACILITY BLACK SEA ENVIRONMENTAL PROGRAMME, 1997; MAKSIMOVIC & MAKROPOULUS, 2002). These facts illustrate that the state of the northwestern Black Sea is strongly dependent on the Danube. If the ecological state of the Black Sea is to be improved, the information about emission quantities, their sources, and their regional distribution is necessary. Further, the analysis of the present state of nutrient emissions and loads is a prerequisite for the establishment of management plans according to the European Water Framework Direc-

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tive (WFD). The nutrient state of a river system depends on natural characteristics and the intensities of anthropogenic activities, especially in the water and agricultural management.

Because 18 countries are placed within the Danube catchment, (13 countries with more than 1% of the total catchment area) the Danube is the most transboundary river system in the world. Due to the different social and economic situations in the Danube countries the analysis of the nutrient situation is a very complex task.

Former studies estimated the nutrient emissions in the Danube River Basin at a national level using different databases and methods (EU/AR102A91, 1997; HASKONING, 1994, SOMLOYDY et al., 1999). These investigations do not include all Danube countries due to the political situation in some Danube countries. The results of these studies were published by ZESSNER & GILS (1999). On the other hand the existing transnational emission study of GARNIER ET AL. (2002) is very raw in relation to the diffuse nutrient emissions and weak in relation to the data base of the population living in the Upper Danube (see below).

Within the this study a harmonised database was established for the time period 1998-2000 and a model for estimation of nutrient inputs was applied at catchment level. This model allows the identification of regional distribution of nutrient influx in relation to input pathways and the assessment of potential management options to change the nutrient loads. In the following, a short overview is given concerning the database and the used method. Subsequently, the main results of the analysis of nutrient inputs within the Danube river basin are presented and discussed. A more detailed presentation of the database, the applied methods and the results is given by SCHREIBER et al. (2003) and BEHRENDT et al. (2000, 2003).

III.2 Database and method

In order to apply a unified methodology for the analysis of nutrient inputs from point and diffuse sources in the river system of the Danube and their regional distribution, the compilation of a harmonised database was necessary. In addition the database had to be integrated into a Geographic Information System (GIS). The database comprises :

- The digital river net from “Digital Charts of the World” (ESRI, 1:1 Million, 1991/1992).
- The digital elevation model (DEM) Hydro 1K from the United States Geological Survey (USGS) with a spatial resolution of 30 arcsec. The DEM and the digital river net were used for the derivation of the borders of the total Danube Basins and the sub-catchments areas,.

- The sub-catchment borders of the river basin upstream of the stations from the Trans National Monitoring Network (TNMN) from the International Commission for the Protection of the Danube River (ICPDR, 2000a) and selected stations from the national networks of the Danube countries. Considering the different countries and their shares at the sub-basin level the whole river system of the Danube was divided into 388 sub-catchments with a size between 40 km² to 16,000 km² .
- The land use after CORINE Land Cover (CLC) with a resolution of 100x100 m (EEA, 2002) resp. 250x250 m for the part of Switzerland (BUNDESAMT FÜR LANDESTOPOGRAFIE). Because such a database is not yet available for Croatia, Serbia and Montenegro, Moldova and Ukraine, the land cover database from USGS with a spatial resolution of 1000x1000 m (USGS 1997) was used for these countries after adaptation to land cover classes of CLC (SCHREIBER & BEHRENDT, 2004).
- The digital FAO soil map (FAO, 1997) with a scale of 1:5 000 000 was used for the characterisation of physico-chemical properties of the soils, e.g. texture and nitrogen content in the topsoil.
- For the separation of the shares of consolidated and unconsolidated rock areas within the sub-basins a digital map was made available from the Netherlands National Institute of Public Health and Environmental Protection (RIVM).
- For the calculations at sub-catchment level, hydrometeorological input data with a resolution of one degree from the Global Precipitation Climatology Centre (GPCC) of the German Weather Service (DWD) were used for the period 1998-2000. In addition maps of longterm precipitation and runoff were digitised from the Danube Atlas published by REGIONALE ZUSAMMENARBEIT DER DONAULÄNDER (1986).
- For the estimation of the atmospheric nitrogen deposition the results of the EMEP-Program (Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe) from the Norwegian Institute for Air Research (NILU) were used for the years 1998,1999 and 2000. The spatial resolution of these data was 50 km per grid cell.
- For the calculation of the nutrient inputs by erosion the digital map of sediment yield from the RIVM (KLEPPER ET AL., 1995) was used.
- The administrative boundaries considered at different levels (countries, regions, districts and municipals) are based on the data from MACON (1999) for the year 1999 with the scale of 1:1 Million.
- For the estimation of the population, the respective national official statistics at municipal level were used.
- For the identification of the location and size of point discharges from municipal and industrial waste water treatment plants, the Waste Water Treatment Plant

(WWTP) inventory of ICPDR (2000b) was taken into account. This database includes only 75 % of the point dischargers and was therefore completed by data og national inventories for Germany (LfW Bavaria, Munich and LfU Baden-Wuerttemberg, Karlsruhe), Austria (Federal Environmental Agency, Vienna), Slovakia (by the Water Research Institute, Bratislava), Hungary (by the Department of Sanitary and Environmental Engineering, Budapest University of Technology and Economics, BUTE) and the inventory of the PHARE-project (EU/AR102A/91, 1997) which was supplied by the Institute for Water Quality and Waste Management, University of Technology Vienna.

- For the calculation of the discharges in the sub-catchments of the Danube and the verification of the model results based on observed loads, the database of the TNMN from the ICPDR for the period 1998-2000 was used. This inventory comprises information on discharges and nutrient concentrations from monitoring stations in the Danube river system. Considering the results from national monitoring programmes, the ICPDR inventory was extended so that a total of 93 stations were considered regarding loads of dissolved inorganic nitrogen (DIN) and 65 stations for total phosphorus loads (TP).

The agricultural data required for the calculation of nutrient inputs from diffuse sources in the sub-catchments were compiled based on statistical informations on country level (Slovenia, Croatia, Bosnia-Herzegovina, Serbia & Montenegro, Bulgaria, Moldova and Ukraine) or on smaller administrative levels (Germany, Austria, Romania, Czech Republic, Slovakia, Hungary and Romania). The OECD method (OECD, 1997) was uniformly used for the calculation of the nutrient balances. Data for the estimation of the share of tile drained areas at the agricultural area were not available for all countries.

The MONERIS (MOdelling Nutrient Emissions into RIver Systems) model was applied to estimate the nutrient inputs by point and diffuse sources into the Danube river system. MONERIS is a model designed to quantify annual average total nitrogen and total phosphorus emissions from catchments to surface waters. It is based on a combination of several conceptual model approaches. Basic input information entering the model are data on river flow and water quality, extensive statistical information for different administrative levels and scales, such as number of inhabitants, crop statistics, livestock number, etc. as well as information from digital maps integrated in a geographical information system (GIS). Whereas inputs from municipal wastewater treatment plants and industrial discharges enter the river system directly, diffuse nutrient inputs into the surface waters arise from a number of different pathways realised by several runoff components (see Figure III.1) as surface runoff, base flow and interflow. Distinction among the inputs from the different runoff components is necessary because the concentrations of nitrogen (N) and phosphorus (P) within the runoff components and the processes within these are very different. Consequently, MONERIS takes seven pathways into account:

- point sources (as municipal waste water treatment plants (WWTP), direct industrial discharges and discharges from people connected to sewers but not to WWTP's) (E_{POINT}),
- atmospheric deposition directly on the

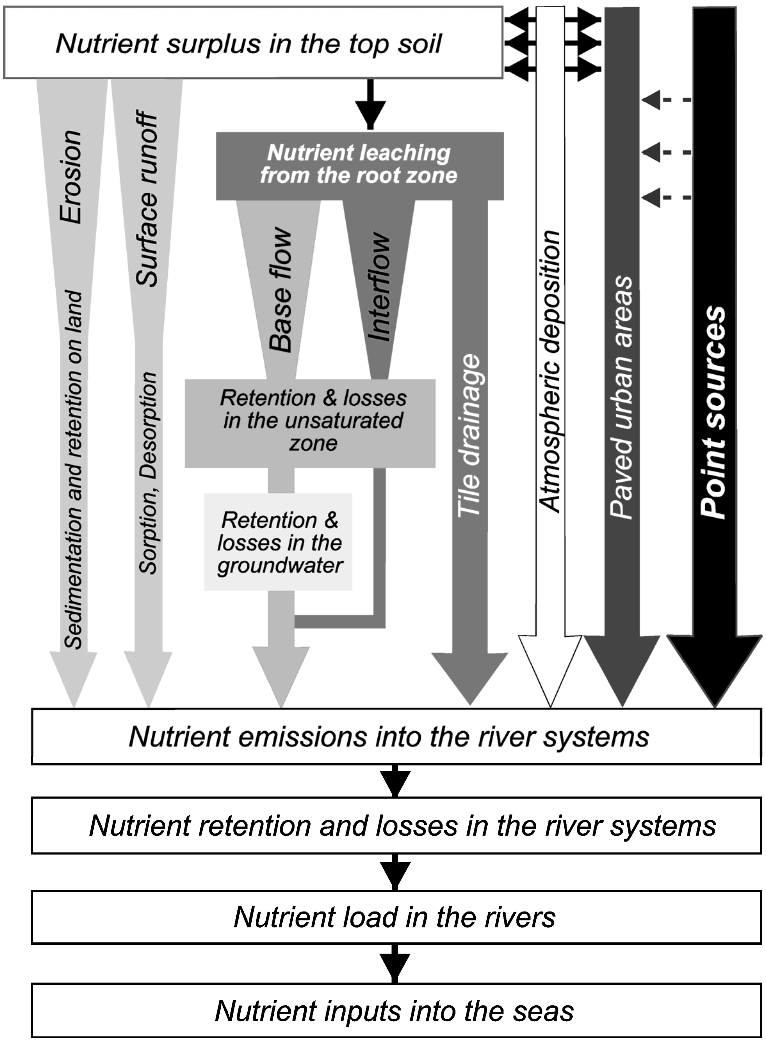

Figure III.1: Pathways and processes in MONERIS.

water surface (E_{DEP}),

- erosion (only particulate nutrients) (E_{ERO}),
- surface runoff (only dissolved nutrients) (E_{RO}),
- groundwater (including the inputs coming from natural interflow) (E_{GW}),
- tile drainage (E_{DR}) and
- paved urban areas (inputs from combined sewer overflows, from areas connected to separate sewer systems, from areas connected to sewers without treatment and areas not connected to sewer systems) (E_{URB}).

Along the pathway from the source to the emission into the river, substances are governed by manifold processes of transformation, retention and loss dependent on the hydrological and morphological conditions within the river system. To quantify and forecast the nutrient inputs into river systems in relation to their source requires knowledge of these transformation and retention processes. Because the current state of knowledge and existing databases are limited for large-scale river basins, this is not yet possible in terms of detailed dynamic process models. Therefore, existing approaches of macro-scale modeling are needed to be complemented and modified. An important step in the development of the individual sub-models in MONERIS was to validate these models by comparing the results with independent data sets. For example, the groundwater sub-model was validated with measured groundwater concentrations.

The ascertained nutrient emissions by MONERIS are validated by comparing the observed river load with a calculated load. In MONERIS the processes of retention and loss within a river system, e.g. sedimentation, denitrification and plant uptake, are taken into account by an empirical retention module (BEHRENDT & OPITZ, 1999). It is also possible to estimate the loads based on the nutrient inputs and to compare the model results with the loads calculated from measured nutrient concentrations and discharges. A complete description of MONERIS was published by BEHRENDT et al. (2000) and BEHRENDT et al. (2002). In this paper only major modifications for the Danube River Basin are described. A detailed description is given by SCHREIBER et al. (2003).

Even though the measured discharges are entered as input data for the modeling, the shares of the runoff components at the total runoff within the sub-catchments must be estimated. Thus the surface runoff, the runoff from tile drained areas and paved urban areas, as well as the runoff caused by snow melt were estimated by empirical equations. For the application in the Danube River Basin, a modification of the approach for the calculation of the surface runoff (SCHREIBER et al., 2003) was necessary.

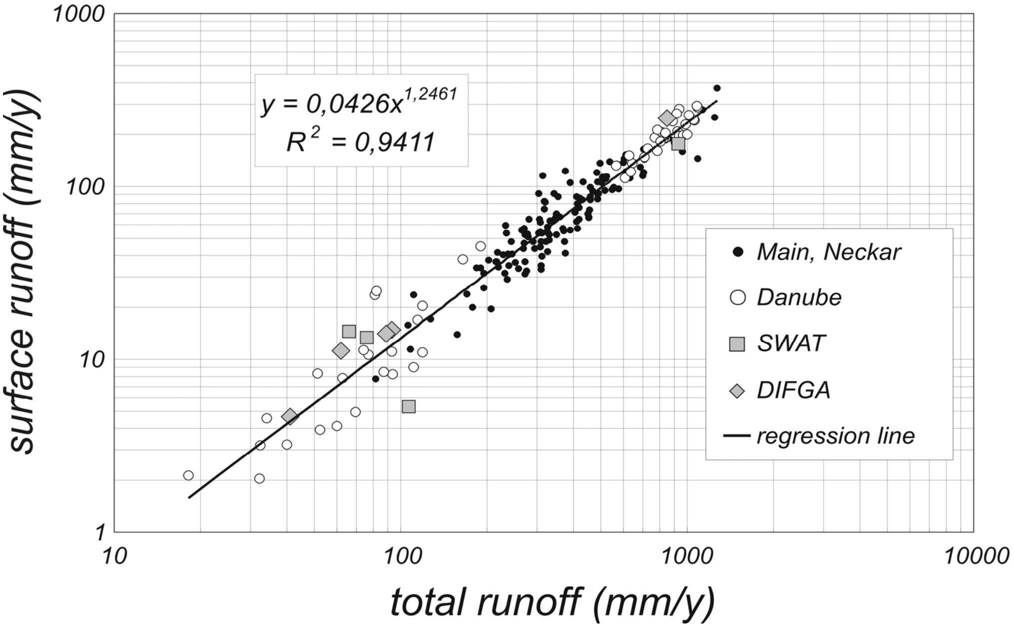

Figure III.2: Dependency of the mean annual surface runoff on the total runoff for different river catchments in Germany and for the case study catchments of the EU-project daNUbs.

This approach describes the relation between the annual average of surface and total runoff by a power function (see Figure III.2). Figure III.2 demonstrates that the results of this approach are comparable with the results of the hydrological models SWAT and DIFGA applied for case study catchments within the daNUbs-project (EVK1-CT-2001-00051, SCHILLING et al, 2003).

With the MONERIS model, the individual components of the water balance (except the base flow) were calculated based on empirical equations and longterm total runoff from digital maps (see database). The total runoff values for the individual sub-catchments were corrected by a factor to minimize the difference between the mean of the measured discharge values for the period 1998-2000 at the outlet of the catchments (resp. monitoring stations) and the correspondingly total longterm runoff values. The calculation of nutrient inputs based on the mean of a three year period enhances the identification of changes in nutrient inputs caused by anthropogenic activities and improves the comparison of different states of nutrient inputs. This should also be appropriate to exclude errors caused by shorter periods in hydrological and meteorological time series.

The model allows not only the quantification of the present state of nutrient inputs by pathways, but also the estimation of background conditions based on scenario calculations (BEHRENDT et al., 2003). In addition, an analysis of the nutrient input state related to different anthropogenic activities like agriculture, water and waste water management was undertaken.

In the present study, the estimations were performed with the same methodology for all 388 catchments within the whole Danube River Basin for the time period 1998 to 2000. The results of calculations are aggregated for the major sub-basins grouped according to the ICPDR (Table III.1, Figure III.4). With exception of the changed approach for the calculation of the surface runoff and the adaptation of the model with regard to the different digital map for the soil losses by water erosion the model was not calibrated for the application for the Danube Basin.

Table III.1: Major features of sub-basins and calculated specific phosphorus (P) and nitrogen (N) emissions into surface waters, the relative share of total N and P emissions by diffuse sources within the Danube and its tributaries in the period 1998-2000.

Subbasins[1]		Population density	Specific runoff	Catchment area	Share agric. area[2]	Specific nutrient inputs		Share of diffuse sources	
						TP	TN	TP	TN
		Inh./km²	$l \cdot km^{-2} \cdot s^{-1}$	km²	%	$kg \cdot ha^{-1} \cdot y^{-1}$	$kg\ ha^{-1} \cdot y^{-1}$	%	%
1	Upper Danube	170	15.2	49,940	55.7	821	19.5	76.0	87.5
2	Inn	90	29.5	26,070	32.1	980	15.2	81.6	92.3
3	Austrian Danube	97	16.8	26,240	46.3	648	10.6	83.0	88.6
4	Morava	117	3.4	26,650	63.2	879	10.4	62.8	75.6
5	Vah, Hron, Ipel	107	5.4	29,840	51.3	782	13.2	72.3	86.1
6	Pannonian Danube	148	2.8	60,370	67.1	1071	8.3	33.5	47.8
7	Drava	80	14.1	40,310	40.6	753	9.2	64.8	83.8
8	Sava	90	13.8	95,890	39.8	989	10.0	45.3	79.0
9	Tisza	89	7	151,780	57.8	672	7.8	56.3	78.1
10	Banat-East. Serbia	83	4.5	28,940	60.6	1277	8.7	37.4	59.5
11	Velika Morava	105	5	37,630	33	809	5.7	36.4	69.7
12	Mizia-Dobrudscha	70	5.4	54,060	55.2	964	7.9	45.4	68.7
13	Muntenia	121	6.3	82,250	63.8	874	8.9	43.3	70.4
14	Prut-Siret	95	5.3	73,470	56.6	716	7.00	69.9	78.3
15	Delta-Liman	62	0.5	19,450	71.3	451	4.5	50.9	69.9
Danube total		**102**	**8.6**	**802,890**	**53.7**	**841**	**9.4**	**53.4**	**76.9**

[1]Sub-basins according to ICPDR (2000); [2]based on CORINE and USGS-Land Cover-map

III.3 Results and Discussion

III.3.1 Characteristics of the river system and nutrient balances

Based on the harmonised database for the Danube River Basin, Table III.1 shows the estimated areas for the sub-basins, the share of agricultural area, the mean population density, and the mean specific runoff in the sub-basins and for the whole Danube catchment.

In the current study, a total catchment area of 802,900 km² was estimated for the Danube. This differs from the previously published area of 817,000 km^2 (UNDP/GEF, 1999; DOMOKOS, 2001). The shares of the different country catchment area differ for the two references. Especially the figures given from UNDP/GEF (1999) for the countries with small shares at the Danube catchment area (Switzerland, Italy and Poland together 14770 km²) are contradictory to the results of our study (3300 km²) and the results of DOMOKOS (2001) (3100 km²). On the other hand the catchment area of the main Danube countries estimated by the GIS analysis of this study (799,600 km²) is very similar to the results of UNDP/GEF (1999) (802200 km²) but different to the area of 814300 km² given by DOMOKOS (2001).

The calculated sub-basin areas presented in Table III.1 are based on the digital elevation model and the rivernet by using the projection Lambert Azimuthal Equal-Area with the central meridian at 20° and the reference latitude of 55°. The total population of 82.2 million people living in the Danube in the period 1998 to 2000 corresponds to a mean population density of 102 inhabitants/km². The population density varies in the Danube sub-basins from 62 to 170 Inh/km². The highest population density was identified within the subcatchment of the upper Danube (8.5 million people). This is in large contrast to the population assumed by GARNIER et al. (2002) for this subcatchment (1.5 million people).

The hydrological conditions in the Danube Basin differ also in a wide range. The specific runoff varies in the main sub basins of the Danube between 0.5 and 29.5 $l \cdot km^{-2} \cdot s^{-1}$. Agricultural areas contribute to total sub-basin area between 32% and 71%. According to the differences of these main indicators one can also expect large differences between the nutrient inputs into the river systems of the sub-basins. The different intensities in the use of agricultural areas multiply these ranges of potential nutrient inputs.

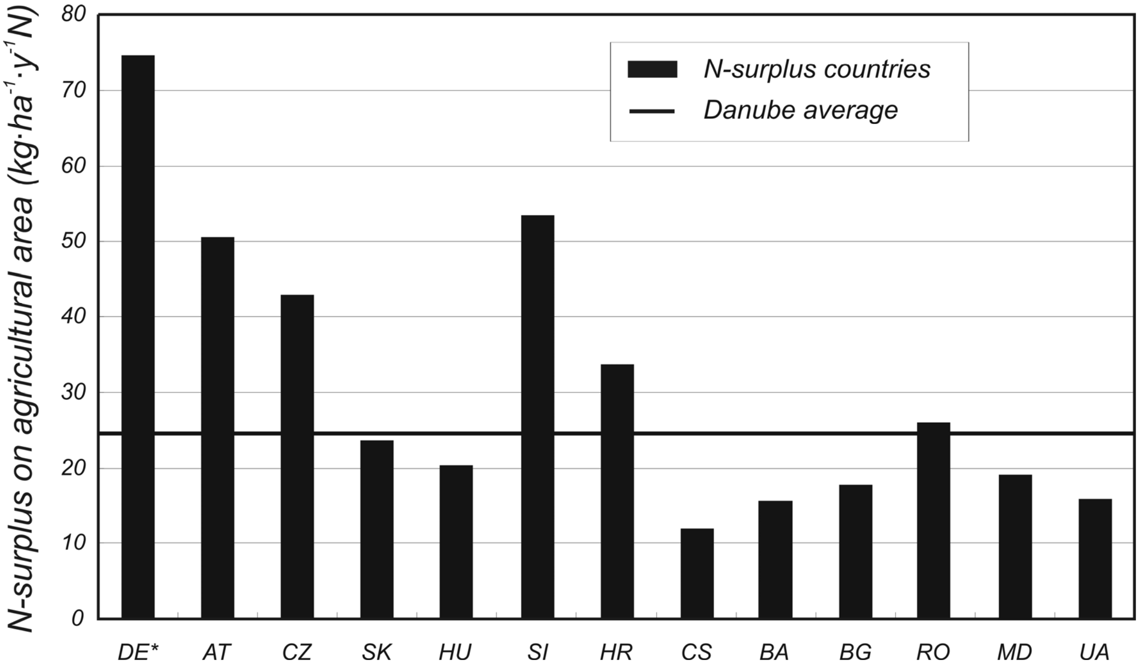

Figure III.3: N-surplus normalised to agricultural area of the Danube countries according to CORINE land cover in the period 1998-2000 (DE* = only Bavaria and Baden-Wuerttemberg).

Figure III.3 gives an overview of the nitrogen surplus calculated for the Danube countries for the period 1998 to 2000 by the application of the OECD-method (OECD, 1997). According to this method, the soil surface balance is calculated as the difference between the total quantity of nutrient inputs entering the soil and the quantity of nutrient outputs leaving the soil annually. The estimate of the annual quantity of nutrient inputs for the soil surface nitrogen balance includes the mineral fertilisers, livestock manure nutrient production, atmospheric deposition of nutrients, biological nitrogen fixation and nutrients contained in seeds and planting material. The estimate of the annual quantity of nutrient outputs for the soil surface nutrient balance includes the harvested and forage crops. The calculations performed for the different countries are based on the agricultural statistics and nutrient equivalents for livestock and crops (SCHREIBER ET AL., 2003). In order to estimate the nutrient surplus within the regions of the Danube catchment area, agricultural area was calculated by CORINE land cover and not the published national statistics.

The figure shows three groups of countries with a similar level of nitrogen surplus on the agricultural area. In the German catchment area of the Danube the nitrogen surplus related to the agricultural area after CORINE land cover is about 75 kg ha^{-1} a^{-1} and about 50 % higher than for the second group of countries (Croatia, Austria, Slovenia and Czech Republic) with N-surpluses between 35 and 55 kg ha ^{1} a $^{-1}$ for the period 1998-

2000. For all other countries the N-surplus ranges between 12 to 26 kg ha^{-1} a^{-1} and is below or close to the average N-surplus of the agricultural areas in the Danube basin.

For a number of countries it was possible to estimate also the long term changes in nitrogen surplus on the agricultural area, based on the statistical information of the countries. The knowledge about the long term changes in N-surplus is necessary for the estimation of nutrient inputs by groundwater. Due to the fact that the residence time of the water in the unsaturated zone and in the groundwater varies in very large ranges (years to centuries, KUNKEL & WENDLAND, 1997) one can expect that the present N-concentrations are influenced by anthropogenic activities in the past. BEHRENDT et al. (this volume, Figure III.2) shows this longterm trend for Germany, Austria, Czech Republic, Hungary, Romania and Bulgaria. In all countries there is an increase of the nitrogen surplus until the beginning of the 1980s. Since the 1990s one can observe a great decrease of nitrogen surplus for the Eastern European countries, remaining at a nearly constant level since 1995. The main reason is the remarkable decrease of mineral fertilizer application and of livestock numbers in the agriculture of these countries. Since the mid of 1980s Germany and Austria also show a moderate to low decrease of nitrogen surplus.

III.3.2 Nutrient input pathways and sources

The total P-emissions into the Danube River Basin were about 68 kt/y in the period 1998-2000 (Table III.2). A total of 44 % of the P-emissions originated from municipal waste water treatment plants (WWTP), industrial waste water discharges and inhabitants which are connected to sewerage systems but not to WWTP`s. Erosion (38 %) was the main diffuse source of P in the Danube River Basin. P-emissions into the surface waters by urban areas, groundwater and natural interflow as well as surface runoff contributed 7 % and 6 %, respectively, to the total P-emissions. Other sources are of minor importance for the P-emissions into the river system of the Danube.

The total amount and the sources of P-emissions vary between the tributaries (Ta-

Table III.2: Nutrient emissions by point and diffuse pathways into the Danube River Basin in the period 1998-2000.

		E_{DEP}	E_{RO}	E_{DR}	E_{ERO}	E_{GW}	E_{URB}	E_{POINT}	E_{tot}[1]
Phosphorus	kt/y	0.9	2.2	0.4	23.3	4.6	31.5	4.6	67.5
	%	1.3	3.2	0.6	34.6	6.9	46.6	6.8	100.0
Nitrogen	kt/y	32.0	76.6	67.0	30.0	333.7	174.3	42.3	755.8
	%	4.2	10.1	8.9	4.0	44.2	23.1	5.6	100.0

[1] E_{tot} = total emissions of all pathways

ble III.1). The average of specific P-emission for the whole Danube River Basin is of 0.84 kg ha^{-1} a^{-1}, whereas the highest specific P-inputs were in Banat-Eastern Serbia (1.28 kg ha^{-1} a^{-1}) and Pannonian Danube sub-basin (1.07 kg ha^{-1} a^{-1}). The reason for this high specific P-inputs is the high portion of point sources is for these both catchments (67% Pannonian Danube; 63% Banat-Eastern Serbia), due to low waste water treatment and moderate population densities.

For N the situation concerning the emissions into the Danube River system is different due to the geo-chemical conditions in the subcatchments. 23 % of the total N-emissions are caused by point sources, 77 % by diffuse sources. Groundwater is the main source of the N-emissions in the Danube River Basin containing 44 %. The contribution of surface runoff and tile drained areas were 10 % and 9 %, respectively. Other sources (atmospheric deposition, erosion and urban areas) contribute only 4-6 % to the total N-emissions and are of minor importance for the whole Danube catchment. Nevertheless the contribution by these pathways can be important for individual catchments (SCHREIBER et al., 2003).

As with P, N-emissions and sources differ considerably between regions. The mean specific N-emission for the whole Danube River Basin is of 9.4 kg ha^{-1} a^{-1}. The highest specific N-emissions come from the Upper Danube (19.5 kg ha^{-1} a^{-1}) and the Inn (15.3 kg ha^{-1} a^{-1}). Specific N-emissions larger than 10 kg ha^{-1} a^{-1} were also found in the Austrian Danube, the Morava, the Sava and the Vah-Hron-Ipel sub-basin. The high specific N-emissions in these sub-basins resulted from the high N-surplus in the upper part of the Danube River Basin e.g. in the German and Czech part of the Danube, and the high specific flow conditions in the German, Austrian, Slovakian and Slovenian regions. The high N-emissions in the Czech part are due to the high portion of tile drained areas. The N-emissions further in the other subbasins are below 10.0 kg ha^{-1} a^{-1} due to the low percolation rate of water and higher N-retention in the unsaturated zone and in the groundwater. This highest N-retention in the subsoil and groundwater was found in the Pannonian lowlands and in the Delta-Liman sub-basin. Together with low N-surplus and low point source discharges the consequence is that the Delta-Liman sub-basin is the subcatchment of the Danube with the lowest N-emissions (4.5 kg ha^{-1} a^{-1}). If the analysis is performed with smaller catchment areas, larger differences in both total N-emissions and relative contributions of specific pathways are observed (Figure III.4).

The diffuse portion of N-inputs, was the highest in the Inn sub-basin (92 %) and is well above the average for the total Danube River Basin (77 %). The lowest portion of diffuse N-emissions (48 %) was found in the Pannonian Danube.

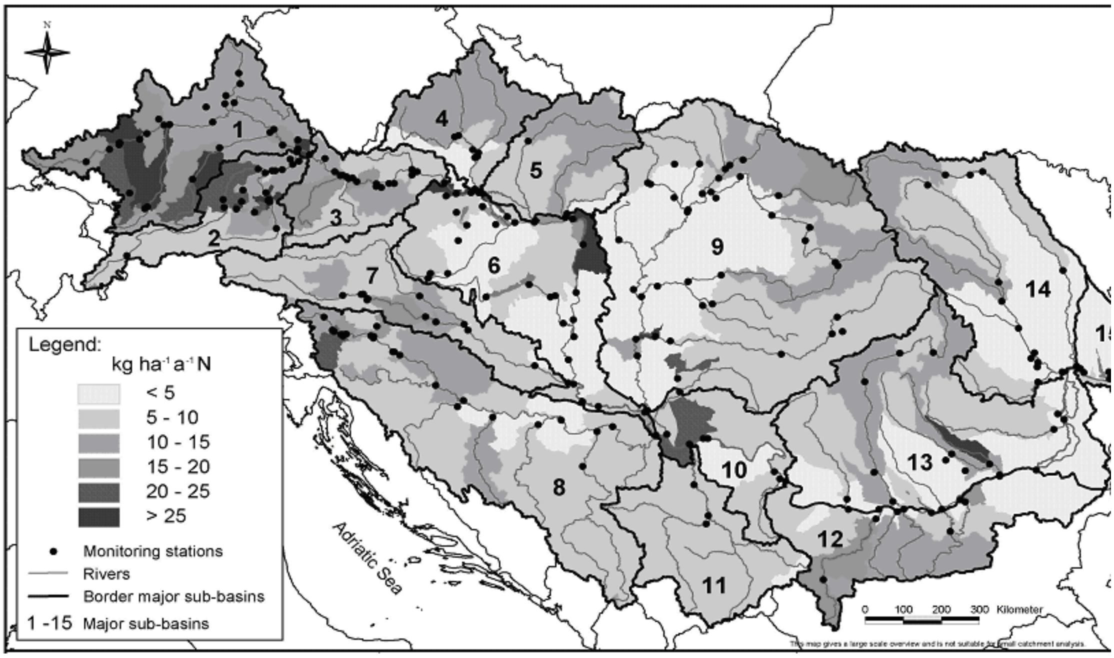

Figure III.4: Regional distribution of mean specific nitrogen emissions into surface water in the Danube River Basin calculated with MONERIS for the period 1998-2000.

Figure III.4 presents the regional distribution of specific total nitrogen emissions into surface water in the different Danube River catchments for the period 1998-2000. The highest specific N-emissions originate from the catchments in the upper part of the Danube, the Morava, the upper Sava, the tributaries of the Tisza, the upper Siret and the upper Prut catchments exceeding 15 kg ha^{-1} a^{-1} N. High N-emissions are also found in those sub-catchments of the Danube containing major cities (Vienna, Bratislava, Budapest, Sofia, Bucharest).

For evaluation of the present state of nutrient emissions in the Danube river system it is necessary to distinguish between emissions caused by natural conditions and those caused by human activities. This is only possible if the emissions under background conditions (independent of human influence) are estimated. Knowledge on the background conditions is also necessary for the definition of the water quality targets in the European Water Framework Directive. For this reason the background emissions where estimated with the MONERIS model supported by a background scenario (SCHREIBER et al., 2003). The scenario is based on the mean annual discharge conditions from 1998-2000, and the following assumptions that define the background conditions for the Danube River Basin:

- Nutrient inputs from point sources, urban areas and drainage are assumed to be non-existent.
- Agricultural and urban areas are considered to be either forest or permanent grassland.
- Soil input through erosion except from areas subject to natural erosion (alpine and foothills) is ignored.
- For the background conditions a constant nitrogen deposition is assumed for all regions (see BEHRENDT et al., 2003). The amount of the potential N-deposition at background condition was estimated by a comparison of the estimated TN-concentrations in the rivers of the Danube catchments with background concentrations published by SMITH ET AL. (2003) for streams and rivers in the United states. These authors found that the background concentration for nitrogen varies between less than 0.02 to more than 0.5 mg L^{-1}. If we assumed for the background scenario a N-deposition of 1 kg ha^{-1} y^{-1}, the estimated TN concentrations within the Danube subcatchments were in a range of 0.03 to 0,49 mg L^{-1} and agree well with the possible range given by SMITH ET AL. (2003).
- The P-concentrations in groundwater is assumed to be identical in all wetlands.
- The ratio of total to dissolved phosphorus concentration under anaerobic groundwater conditions is assumed to be 1.5 instead of 2.5.

The modelling results in background emissions of approximately 6.5 kt/y P and 61 kt/y N for the Danube river system. This corresponds to specific background emissions of about 0.08 kg ha^{-1} a^{-1} P and 0.7 kg ha^{-1} a^{-1} N. Catchments in mountainous regions have higher background emissions due to the occurrence of natural erosion and high precipitation and high surface runoff. Anthropogenic activities increased the phosphorous and nitrogen emissions between 940 % and 1150 % of the background levels. Based on the nutrient emissions for the diffuse pathways and for the background conditions it is possible to calculate the proportion of emissions related to agricultural activities. For this purpose it was assumed that the emissions caused by agricultural activities are all emissions from surface runoff, erosion, groundwater and tile drained area less the background emissions. For nitrogen it was additionally assumed that the atmospheric deposition of ammonium is caused by agricultural activities.

Figure III.5 shows the contribution of human activities to the total phosphorus and nitrogen emissions for the sub-basins of the Danube River. Background emissions contribute between 1 % and 41 % to the P-emissions and 4 % and 16 % to the N-emissions. This contribution from background emissions is remarkably higher in catchments with alpine or mountainous areas like in the catchments of Inn, Austrian Danube, Drava or Sava.

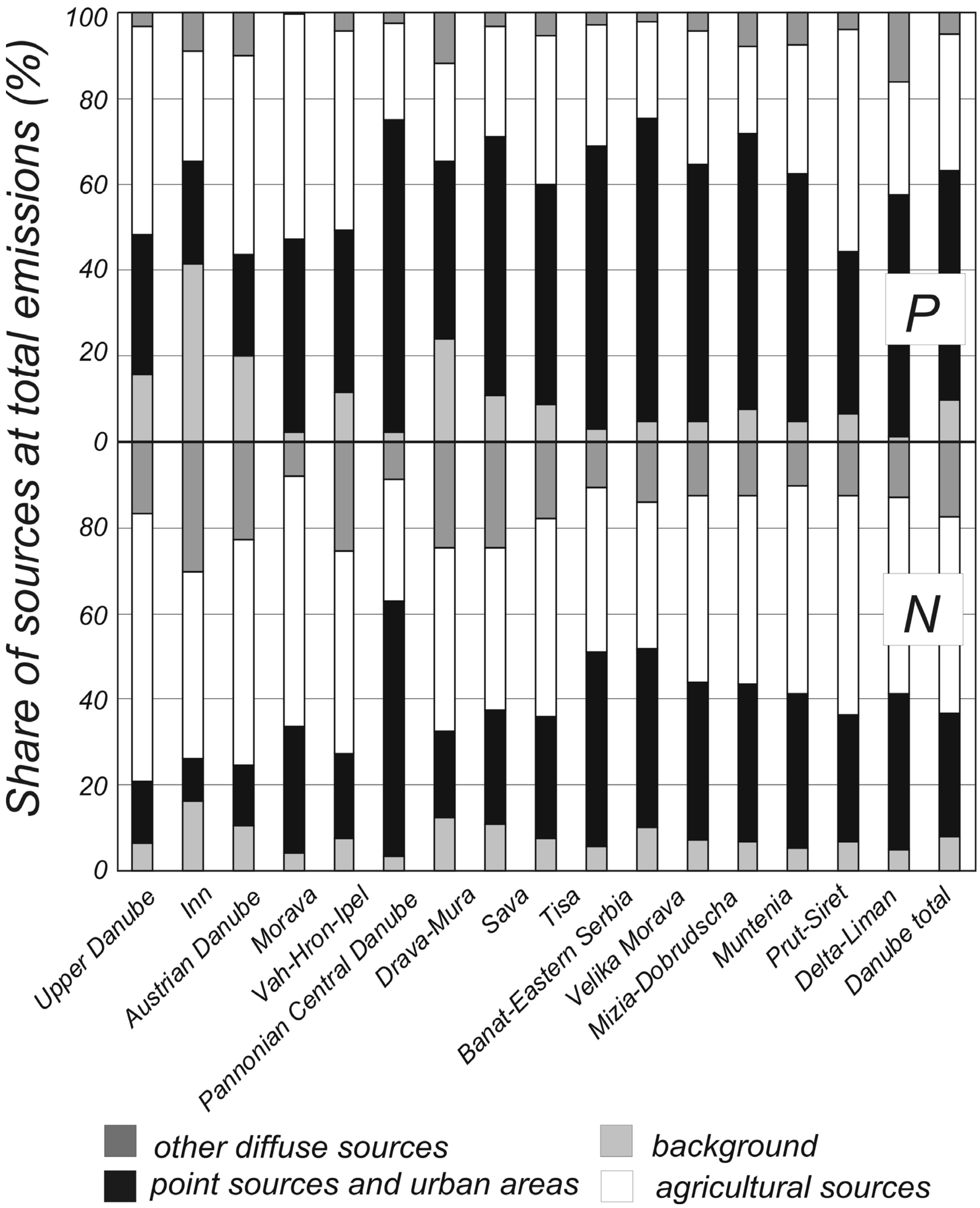

Figure III.5: Shares of anthropogenic activities at the nutrient emissions for the Danube River sub-basins in the period 1998-2000.

The mean contribution from agricultural P-emissions is approx. 30 % for the total Danube River Basin and varies in the sub-basins from 20 % to 53 %.

The largest contributions to P-emissions are found in the sub-basins Prut-Siret, Vah-Hron-Ipel,Morava, Austrian and Upper Danube. Regarding the N-emissions into the river system of the Danube the agricultural sources are dominant for all sub-basins with exception of the Pannonian Danube, Banat-Eastern Serbia and Velika Morava. The mean contribution to N-emissions related to agricultural activities is 46 % for the total Danube River Basin. The highest portion of N-emissions by agricultural sources was identified for the the Upper Danube (63 %) and the Morava sub-basin (58 %).

The point and diffuse P-emissions from urban settlements are in most of the sub-basins between 23 % and 73 % of the total P-emissions. The highest values were found in the Pannonian Danube (73 %) and Velika Morava (71 %). The reason for these high P-emissions is the low level of P-elimination in the existing WWTP's and the persisting use of phosphorus in detergents in some Danube countries (see SCHREIBER et al., 2003).

III.3.3 Comparison with measured data and model uncertainties

Based on river parameters as hydraulic load and specific runoff (BEHRENDT & OPITZ, 1999), the loads for the Danube River sub-basins could be calculated from the emissions, by application of the retention functions of MONERIS for nitrogen and phosphorus. Additionally it was assumed that the Iron Gate reservoir can retain nutrients to a certain amount. The percentage of this retention was estimated by the application of the Danube Water Quality Model (DWQM). The simulations of the DWQM (DANUBS, 2004) show that the Iron Gate retains about 32% of Phosphorus and 6% of nitrogen. The calculated nutrient loads could be compared with the observed loads calculated from the measured nutrient concentrations and discharges at the monitoring stations. For dissolved inorganic nitrogen (DIN, loads at 93 monitoring stations) the mean deviation is 21.2 % and for total nitrogen (TN, loads at 16 monitoring stations) 20.5 % (Figure III.6). The figures for deviation between calculated and observed loads within the Danube river system are comparable with the findings for other river systems (BEHRENDT et al., 2003, BEHRENDT et al., 2004a). The highest deviation in observed nitrogen loads were found within the Hungarian part of the Danube and some Romanian catchments. The comparison shows a low tendency that the calculated loads are systematically lower than the observed loads of DIN. The reason for the large deviations could be an underestimated annual observed DIN-load as well as an overestimated retention or emission in these river systems. It is assumed that one reason for the scarce large deviations of the DIN-load especially in the Pannonian Danube is an overestimation of the surface water area which was derived for German rivers, where the plains are often wet and artificially drained by ditches. For the dry areas of the Pannonian plains the surface water area could probably be overestimated with the applied equation. In this case the retention would be overestimated.

For total phosphorus (TP, loads at 65 monitoring stations) the mean deviation between the calculated and observed loads (Figure III.6) within the river system of the Danube was 32 %. This is about 10 % higher than for nitrogen.

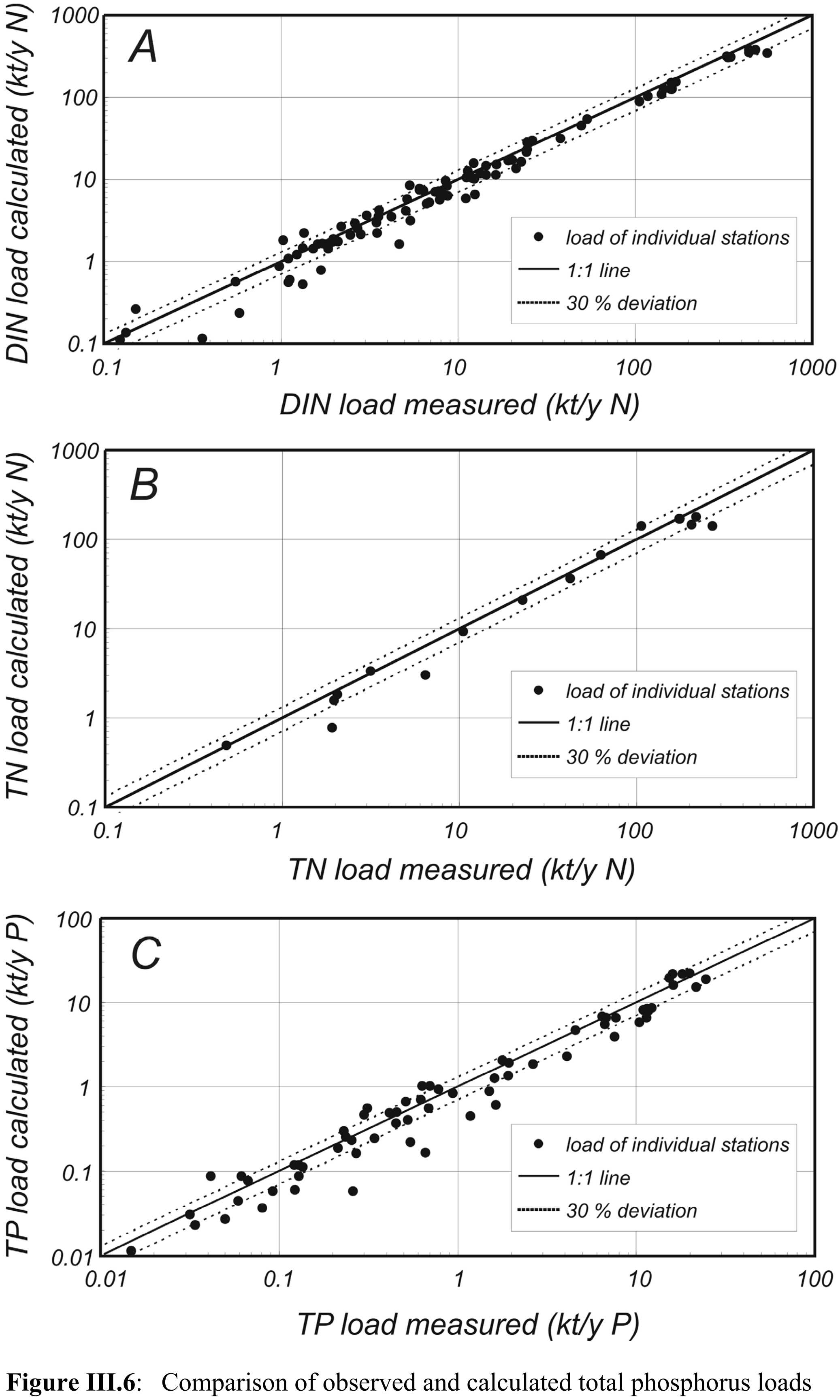

Figure III.6: Comparison of observed and calculated total phosphorus loads (TP, bottom) and nitrogen loads (DIN, top, TN, middle) in the Danube and its sub catchments for the period 1998-2000.

The analysis of the loads shows also a small tendency that the calculated P-loads with the MONERIS model are below the observed loads. A more detailed analysis of the catchments with high underestimation of P-loads shows that most of these catchments are also located in the the Pannonian Danube. Also for phosphorus it can be assumed that the deviations are caused due to overestimation of the P-retention (overestimation of the water surface area) or the underestimation of total P-emissions with the model (SCHREIBER et al., 2003) and underestimation of the observed TP-loads. The comparison of the point source discharges estimated on the Hungarian point source inventory (this study) and more detailed investigations for two Hungarian subcatchments (KOVACS, 2004) indicates that the point source discharges of the inventory could be in general to low. But more reasonable for the larger deviation between the observed and calculated TP loads seems to be an underestimation of the observed loads, especially for smaller catchments with low (monthly) frequency of measurements. Different authors (e.g. KLAVERS, H. & DE VRIES, A., 1993 cited in DE WIT, 1999) have shown that a bias of the nutrient load exists for such conditions, which is substantial higher for substances with a high portion of the particulate fraction (phosphorus) as for substances mostly transported in dissolved forms (nitrogen). The reason is probably the high dynamic of the input as well as the transport processes for the suspended solids, which is not recorded at monthly sampling.

Within the Danube the deviation between calculated and observed nutrient loads is lower in the sub-catchments of the upper parts (19 % for nitrogen and 29% for phosphorus. For the most of these sub-catchments the spatial resolution of the input data is better than for the subcatchments of the lower Danube, where the deviations were 23 % (N) and 35 % (P). Furthermore the mean deviation between calculated and observed loads was only 14 % (N) and 20 % (P) for the 22 stations along the main stream of the Danube (see also BEHRENDT et al., 2004b, this volume). Both findings indicate that the model results are also to a large extent dependently on the resolution of input data. Input data with higher spatial resolution especially for the Danube part below Hungary can substantially improve the model results.

The estimated nutrient loads for the Danube upstream the station Reni were 22.5 kt/y for total phosphorus (TP), 396 kt/y for dissolved inorganic nitrogen (DIN) and 468 kt/y for total nitrogen (TN) for the period 1998 to 2000. Van Gils et al. (2004) estimated for the Danube at Reni (station direct upstream the Danube delta) a load of about 17.8 kt/y for TP and 462 kt/y for DIN for the same time period. Compared to these the results the calculated loads deviate from the observed loads at 26,5 % (TP) and 14 % (DIN). The mean deviation for the eight main stations analysed by VAN GILS et al. (2004) was 26 and 10 % for TP and DIN, respectively.

A measure for the quality of the model is according to MAYER & BUTLER (1993) the model efficiency. The model efficiency (EF) can be calculated by the following equation:

$$EF = 1 - \frac{\sum (y_i - y_{ci})^2}{\sum (y_i - y_m)^2}$$

where y_i represent observed values, y_{ci} calculated values and y_m the average of all observed values.

The highest possible value for the model efficiency is 1. A value of 0 will be reached for the very simple model, which is equal to the average of all observed data.

If the model efficiency is calculated for the nutrient loads of all stations the results is 0.93 and 0.95 for total phosphorus and dissolved inorganic nitrogen. This high model efficiency for the nutrient loads is mostly determined by the stations with the largest loads. Therefore the model efficiency was also calculated for the specific nutrient loads (load divided by the catchment area upstream the station). For this case the model efficiency is 0.41 for TP and 0.75 for DIN. Whereas the model efficiency can be evaluated as high for nitrogen for original nitrogen loads and the specific load, the efficiency of the model is relatively low for the specific phosphorus loads. This indicates that a main task for the development of the model is the improvement of database and the process description regarding the pathways dominating the P-emissions as erosion. But this needs also the improvement of the observations especially measurement with higher frequency or dependent on discharge.

III.4 Conclusions

The analysis has shown that the MONERIS model is suitable for modelling of nutrient emissions and loads of a large transboundary river basin. Amongst the individual subbasins and the whole Danube River Basin the nutrient emissions and the share in the various emission pathways vary to a relatively large extent. The results of calculations allow the identification of regional hot spots, and the derivation of specific regional measures to reduce the emissions into the Danube and consequently into the Black Sea. The mean deviation between the measured and calculated nitrogen loads is in the same range as found for other river catchments. The deviation for P-loads is about 5% higher. Problems and uncertainties in the results are mainly caused by insufficient spatial and temporal resolution of data and maps. Estimated nutrient emissions are highly dependent on information of nitrogen surplus in topsoils and the proportion of different soil types. Thus a better resolution of information on land use for Croatia, Serbia and Montenegro, Bosnia-Herzegovina and the Ukraine as well as statistical data would increase the spatial resolution for the estimation of nutrient surplus of agricultural areas for the district or regional level for these parts in the Danube River Basin. A better information on soil properties (basin wide) could also improve the reliability of the calculations. The nitrogen emissions via groundwater are sensitive to the proportion of different types of aquifers which influences the travelling time, hence a better spatial resolution on hydrogeology is also required to improve the model results. Also the enlargement of point source inventory including more information on population connected to sewer systems and WWTP's is a further need to enhance the model calculations. An improvement of the results could be expected by further adaptation of the model approaches, especially for erosion. The next step of model development should focus on the implementation of the results from the case study regions addressed in particular to the water balance, erosion, groundwater and retention (Schilling et al., 2004). The Comparison of results from different model approaches can help to identify sources of uncertainties of conceptual models like MONERIS. Here, the comparison of down- and up-scaling approaches, if applicable in combination with process based models, could give interesting results for further error quantifications.

For derivation of effective management measures the future research should concentrate on scenario calculations. The aimed targets for scenario calculations should also take into account the European Water Framework Directive and the European Waste Water Directive to ensure the close connection of research results with policy and management stemming from the Danube River Protection Convention (DRCP).

III.5 Acknowledgements

The presented results originates from the research project "Harmonised Inventory of Point and Diffuse Emissions of Nitrogen and Phosphorus in the Danube Basin" founded and supported by the German Federal Environmental Agency, Berlin, and the project "Nutrient Emissions in the Danube Basin and its Impact on the Black Sea" (daNUbs) supported under contract number EVK1-CT-2000-00051 by the Energy, Environment and Sustainable Development (EESD) Programme of the EU 5th Framework Programme. Details on the project can be found on the project homepage: http//:danubs.tuwien.ac.at. For data supply, support and helpful discussions we thank especially the members of the EMIS and MLIM expert group of the ICPDR. We address our particular thanks to the national consultants, Research Institutes and States Agencies for providing the basic data for the estimation of the nutrient emissions and loads into the river system of the Danube.

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IV. Long term changes

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IV.1 Introduction

The Danube river system drains an area of 802,900 km² which is approximately one third of the Black Sea basin. 82.2 Mio. people live within the Danube catchment representing about 51 % of the total population of the Black Sea basin (SCHREIBER et al., 2004; this volume). According JAOSHVILI (2002) the share of the Danube to total discharge into the Black Sea is about 54 %. These numbers illustrate the importance of the Danube river system for the Black Sea ecosystem. The status of this ecosystem changed over the last decades and eutrophication is one of the most sensitive indicators for these changes (LANCELOT et al., 2002; Mee, 2001; KROISS et al., 2004). Beside the fact that newest observations show a improvement of the situation (HORSTMANN et al., 2003) and the loads of nutrients from the Danube to the Black Sea were reduced in the last decade (VAN GILS et al., 2004), it is important to clarify what the sources of the nutrient fluxes within this river system are and what the causes are for these changes. Furthermore it is important to clarify the influence of the construction of dams and barrages, like the Iron Gate, on the nutrient transport of the Danube to the Black Sea. Former studies on this subject where focused mostly on the silica transport (Humborg et al., 1997) or partly speculatively (Lancelot et al., 2002), because the analysis did not include time periods before the establishment of dams.

Previous studies on nutrient entries in the Danube River (HASKONING, 1994; EU/AR102A/91, 1997; SOMLODY et al., 1999; ZESSNER & VAN GILS, 2002) have already tried to identify the most important sources, pathways, stocks and sinks of nutrients within the Danube catchment, but these studies were limited regarding the resolution in space (only country level) and the use of different data sources especially for the estimation of the diffuse nutrient inputs into the river system. Furthermore, they did not cover the whole basin of the Danube, because information on the countries of former Yugoslavia was only available for Slovenia and partial Croatia.

GARNIER et al. (2002) tried for the first time to model the nutrient loads for the whole Danube basin, but the database was incomplete and partly weak. Furthermore, the applied sub-model for the emissions into the river system was limited in space (only 12 subcatchments) and estimated the nutrient inputs from export coefficients and retention

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factors, based on empirical knowledge from other catchments. The analysis was done for two recent time periods (88-91 and 94-96) and thus only reflects the decrease of the nutrient loads in the Danube River.

All former analysis did not cover the 1950's an 1960's. But this time period is very interesting, because during this time the ecological status of the Black Sea was not characterised by the problems of the later decades (Horstmann et al., 2003).

Information on long term changes of the nutrient loads from rivers to the ocean is important so that ecological changes in the sea can be clarified. Because measurements of nutrient concentrations in the Danube are often only available for the last two decades (VAN GILS et al., 2004), loads of the past can only be quantified by modelling.

SCHREIBER et al. (2004, this volume) present a harmonized database and applied a unified model approach for the total area of the Danube Basin and 388 subcatchments. The nutrient emissions from point and diffuse sources into the whole river system of the Danube were estimated for the most recent time period of 1998 to 2000. With help of this established database and the model MONERIS (BEHRENDT et al., 2000) an attempt for the reconstruction of the nutrient emissions into the river system of the Danube for the time period from the 1950s to the mid of 1990s is presented here. The aim was to compare the model results with the published data of the former studies and to identify the causes for the changes of the nutrient loads of the Danube into the Black Sea in the last decades. Furthermore, it should be tested whether it is possible to reconstruct the nutrient loads and sources of nutrient emissions to the river system of the past. This is important for the calculation of scenarios with potential future load changes, because a model, which does not reflect the historical changes, can not be applied for the simulation of possible changes in the future.

IV.2 Database and Methods

- The applied model

The model MONERIS (MOdelling Nutrient Emissions into RIver Systems) developed to estimate nutrient emissions in German river systems (BEHRENDT et al., 2000) was recently applied to the Danube basin (SCHREIBER et al., 2004; this volume). Besides the inputs of municipal waste water treatment plants and of direct industrial discharges the model includes six diffuse or non-point pathways of emissions into the river system (atmospheric deposition, erosion, surface runoff, groundwater, tile drainage and paved urban areas; see SCHREIBER et al., 2004, this volume, Figure IV.1) and a sub-model to estimate the retention and losses within the surface waters of the river systems (BEHRENDT

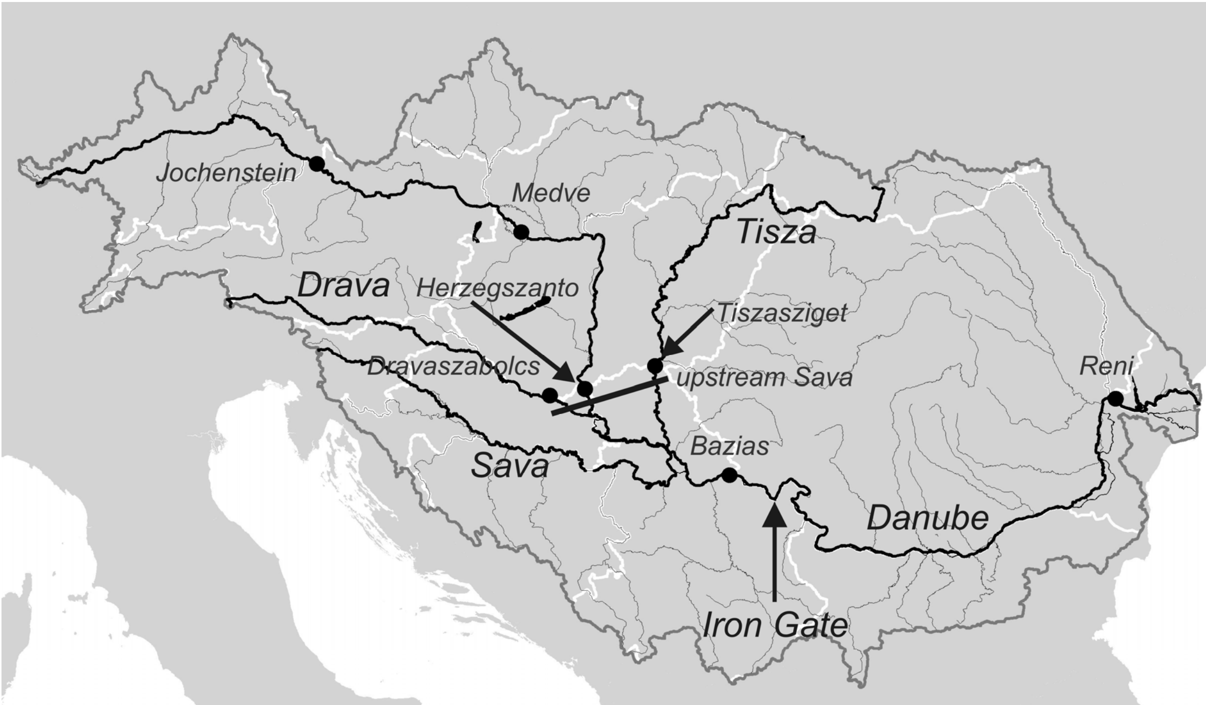

Figure IV.1: Changes of the nitrogen surplus in the agriculture of the Danube countries from 1950 to 2000.

& OPITZ,1999). In the study presented here the methodology and model version of SCHREIBER et al. (2004, this volume) are used.

- Input data on long term changes

To model changes of the nutrient emissions and loads the database was used for the state in 2000 as described by SCHREIBER et al. (2004, this volume). Changes of the main human effects on the nutrient emissions are taken into account as time series for the Danube countries as shown in Table IV.1 to IV.6. Official statistics of the individual countries and the FAO (FAOSTAT, 2004) were used for the description of the development of population (Table IV.1) and agricultural area (Table IV.2) as well as fertilizer application, livestock numbers and harvested crops. The nutrient surplus on agricultural area (Figure IV.1) was estimated according to the OECD method (OECD, 1997).

Table IV.3 shows the database for the connection of people to sewer systems and waste water treatment plants (WWTP's), derived by different sources and additional assumptions for missing data. Because data on the connection of people to sewerage and WWTP's was not available for the 1950's, the trend from the 1960's and 1970's was extended to the 1950's.

Table IV.4 represents the database for the development of waste water treatment for the individual countries. In general, it is assumed for the 1955's that existing WWTP's possessed only mechanical treatment. The change of the nutrient elimination of WWTP's for the individual countries was estimated by the multiplication of the elimination rates given in Table IV.5 with the portion of the WWTP's with primary, secondary and tertiary treatment given in Table IV.4.

Table IV.6 represents the development of the specific P emissions of people and industrial indirect discharges into the sewer systems due to the changing use of phosphorus in detergents. The development of this indicator for Germany and Austria is based on the published data of KLOTTER & NEUSSEL (1959), BUCKSTEEG (1966), BERNHARDT et al. (1978), LÖFFLER (1980), FIRK & GEGENMANTEL (1986) and SCHMOLL (1999). For the other countries it is assumed that the change of the specific P emission follows from the 1950s to 1990 the trend found for the former GDR (BEHRENDT, 1994). For the last period data of the ICPDR on the use of phosphours in detergents within the different countries were taken into account (SCHREIBER et al., 2003). Between 1990 and 2000 a linear interpolation was done. As shown by BEHRENDT (1994) the specific N emission per capita is not changing in time and a constant value of 12 $g \cdot Inh.^{-1} \cdot d^{-1}$ was assumed for all

Table IV.1: Changes of the population in the Danube basin from 1955 to 2000 (data according to FAOSTAT data (2004), DE represents the data of Bavaria and Baden Württemberg according to Genesis (2004) and Statistisches Landesamt Baden-Württemberg (2004a), all country data weighted by the portion of population in the Danube basin according to Schreiber et al. (2003) for the period 1998-2000.

country	**1955**	**1965**	**1975**	**1985**	**1990**	**1995**	**2000**
	1000 Inh.	1000 Inh.	1000 Inh.	1000 Inh.	1000 Inh.	1000 Inh.	1000 Inh.
DE	7148	7850	8833	9339	9538	9736	9717
AT	6604	6912	7390	7589	7703	7672	7702
CZ	2323	2497	2522	2568	2639	2734	2767
SK	4160	4471	4566	4649	4778	4949	5009
HU	10739	11098	10503	10693	10895	11076	10944
SI	1355	1478	1493	1543	1610	1683	1738
HR	2350	2563	2585	2672	2787	2913	3010
BH	2108	2299	2701	2791	2912	3043	3144
CS	7106	7751	7835	8097	8447	8828	9121
RO	16213	17647	17456	18576	19486	20363	20844
BG	3938	4309	3895	4030	4140	4207	4254
MD	609	715	743	781	819	854	892
UA	2061	2420	2457	2584	2708	2824	2951
total	**66716**	**72009**	**72979**	**75913**	**78462**	**80883**	**82092**

time periods. Additionally to the nutrient emissions of people the emissions of industrial discharges into the sewer systems have to taken into account. According to BEHRENDT (1994) and LINDTNER & ZESSNER (2003) these specific N and P emissions per capita were assumed to be 6 $g \cdot Inh.^{-1} \cdot d^{-1}$ N and 1 $g \cdot Inh.^{-1} \cdot d^{-1}$ P for 1980 to 1995 for all countries. For 1955 we assumed that these emissions were zero and increased linearly to the indicated value until 1980.

For the direct industrial nutrient input it is assumed that these discharges for the periods before 1995 were the same as estimated by EU/AR102A91 (1997) and that the variation is proportional to the change of the N discharges of WWTP's. Beside the point sources from industrial and municipal discharges the amount of nutrient discharges caused by the large pig production units in Romania (three units of 1 Mio pigs) have to be taken into account (EU/AR102A/91, 1997).

Table IV.2: Relative changes of the agricultural area in the Danube countries from 1955 to 2000 (data according to FAOSTAT data (2004), DE represents the data of Bavaria and Baden-Württemberg according to Statistisches Landesamt Baden-Württemberg (2004b) and Genesis (2004)

country	**1955**	**1965**	**1975**	**1985**	**1990**	**1995**	**2000**
	[%]	[%]	[%]	[%]	[%]	[%]	[%]
DE	122.7	119.0	111.8	102.1	100.8	100.1	100.0
AT	115.9	114.0	108.5	100.5	100.2	101.0	100.0
CZ	108.2	106.1	104.1	101.7	100.4	100.0	100.0
SK	108.2	106.1	104.1	101.7	100.4	100.0	100.0
HU	121.0	118.8	115.6	111.7	110.6	105.6	100.0
SI	105.5	104.8	104.1	103.2	102.1	103.9	100.0
HR	105.5	104.8	104.1	103.2	102.1	91.0	100.0
BH	105.5	104.8	104.1	103.2	102.1	100.0	100.0
CS	105.5	104.8	104.1	103.2	102.1	100.4	100.0
RO	98.5	99.8	100.8	101.3	99.6	100.1	100.0
BG	91.6	93.5	96.1	99.6	99.4	100.1	100.0
MD	105.5	104.8	104.1	103.2	102.1	101.2	100.0
UA	105.5	104.8	104.1	103.2	102.1	101.2	100.0

Table IV.3: Changes of the connection of people to sewers, P and N elimination in waste water treatment plants of the Danube countries from 1955 to 2000 (References see below the table; numbers without index interpolated from the development of comparable countries)

	1955	1965	1975	1985	1990	1995	2000
	[%]	[%]	[%]	[%]	[%]	[%]	[%]
Population connected to sewers							
DE	56	66*	78*	86*	88*	91*	95*
AT	40[+]	47[+]	54[+]	65[+]	71[+]	79**	86**
CZ	30	43	60**	70**	73**	73**	75**
SK	20	29	37	46[+++]	48[+++]	48[+++]	54**
HU	21	30	35**	48**	50**	44**	57**
SI	27	39	51	63	66	66	74**
HR	16	23	30	37	38	39	43[++]
BH	16	23	30	37	39	39	44[#]
CS	18	26	34	42	44	50	56[++]
RO	17	24	32	39	41	41[x]	46[++]
BG	25	36	47	58	61	67**	68**
MD	21	30	39	48	50	50	56[##]
UA	20	28	36	45	47	47	53[##]
Population connected to WWTP							
DE	26	37*	68*	82*	86*	89*	94*
AT	30[+]	39[+]	48[+]	61[+]	70[+]	77**	86**
CZ	12	22	39**	48**	50**	56**	65**
SK	2	5	9	17[+++]	21[+++]	32[+++]	49**
HU	3	6	13**	25**	20**	21**	35**
SI	4	8	16	26	29	37	51**
HR	3	6	11	18	20	25	35[++]
BH	2	3	6	10	11	14	19[#]
CS	3	6	12	20	22	32	44[++]
RO	2	5	9	14	16	20[x]	28[++]
BG	3	6	12	19	21	35**	38**
MD	2	4	8	13	14	18	25[##]
UA	1	2	4	6	7	8	12[##]

* Statistisches Landesamt Baden-Württemberg (2004a) and Bayerisches Landesamt für Statistik und Datenverarbeitung (2003); [+] Fleckseder H. (1994); ** EUROSTAT (2004); [++] UN/ECE (2004); [#] LSMS Bosnia (2004); [##] EECCA (2004); [+++] OECD (2002); [x] EU/AR102A91 (1997)

Table IV.4: Changes of the connection of people to different waste water treatment plants for the Danube countries from 1955 to 2000 (References see below and Table 3; numbers without index interpolated from the development of comparable countries - percentages in relation to the total population connected to WWTP's)

	1955	**1965**	**1975**	**1985**	**1990**	**1995**	**2000**
	[%]	[%]	[%]	[%]	[%]	[%]	[%]
Population connected to WWTP with mechanical (primary) treatment							
DE	100	79*	36*	6*	3*	2*	1*
AT	100	85[+]	52[+]	11[+]	7[+]	1[+]	1[+]
CZ	100	90	69	48	25	1	1[xx]
SK	100	88	63	39	27	16	5[xx]
HU	100	100	62	32**	30**	14**	11**
SI	100	100	73	45	31	18	4
HR	100	100	93	86	83	81	60
BH	100	100	92	84	80	88	95
CS	100	100	80	61	51	41	31
RO	100	100	73	46	32	19	16
BG	100	100	68	35	19	3**	3**
MD	100	100	67	33	17	0	0
UA	100	100	71	41	27	12	12
Population connected to WWTP with biological (secondary) treatment							
DE	0	21*	64*	77*	29*	17*	8*
AT	0	15[+]	48[+]	82[+]	83[+]	35[+]	21[+]
CZ	0	10	31	52	62	73	50[xx]
SK	0	12	37	61	73	86	46[xx]
HU	0	17**	38**	68**	65**	81**	77**
SI	0	0	27	55	69	82	96[xx]
HR	0	0	7	14	17	19	40[xx]
BH	0	0	8	16	20	13	5[xx]
CS	0	0	20	39	49	59	69[xx]
RO	0	0	27	54	68	81[x]	84[xx]
BG	0	0	32	65	81	97**	97**
MD	0	0	33	67	83	100[x]	100
UA	0	0	29	59	73	88[x]	88
Population connected to WWTP with additional nutrient elimination (tertiary)							
DE	0	0	0	17	68	81	91
AT	0	0	0	8	10	64	78
CZ	0	0	0	0	13	26	49
SK	0	0	0	0	0	2	49
HU	0	0	0	0	5	5	12
SI	0	0	0	0	0	0	0
HR	0	0	0	0	0	0	0
BH	0	0	0	0	0	0	0
CS	0	0	0	0	0	0	0
RO	0	0	0	0	0	0	0
BG	0	0	0	0	0	0	0
MD	0	0	0	0	0	0	0
UA	0	0	0	0	0	0	0

[xx] ICPDR (2000b)

Table IV.5: Elimination rates of the different treatments in WWTP's estimated from point source inventories of the Danube countries according to Schreiber et al. (2003)

Kind of treatment	Phosphorus %	Nitrogen %
Mechanical / primary	15	10
Biological / secondary	35	30
Biological with additional nutrient elimination / tertiary	87	80

from industrial and municipal discharges the amount of nutrient discharges caused by the large pig production units in Romania (three units of 1 Mio pigs) have to be taken into account (EU/AR102A/91, 1997).

Because the manure of such large stable units cannot be stored and transported over large distances, it was assumed that all of the excreta of the approximately 3,000,000 pigs were directly discharged to the water bodies. From the development of the number of pigs in Romania (FAOSTAT, 2004) one can conclude that these industrial pig farms existed from about 1980 to 1991. Therefore, these agricultural point sources were only taken into account for the three time periods 1980, 1985 and 1990.

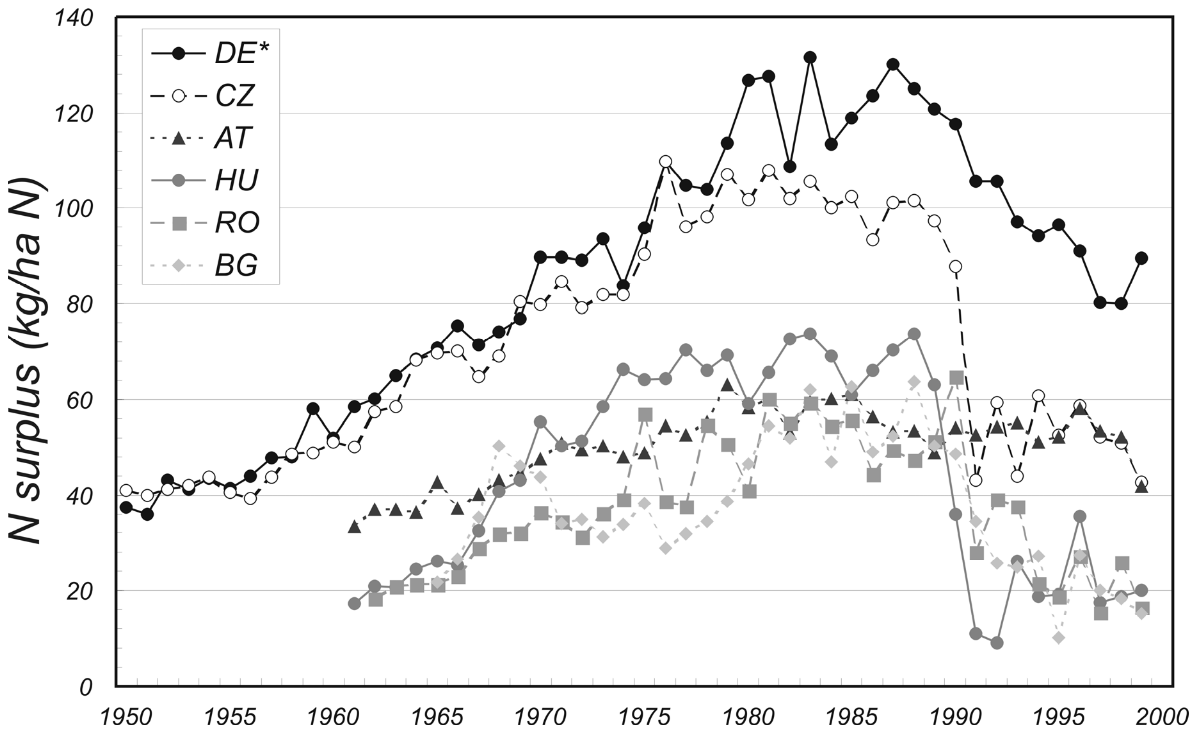

Figure IV.2: The Danube basin and the location of the used monitoring stations within the catchment.

Unfortunately, information on the tile drained area and the soil losses from arable land were not available with exception of the changes of agricultural (Table IV.2) and arable land. Therefore it is assumed that the soil loss rate per arable land was constant over the whole investigation period. For tile drained areas it was taken into account that the drainage of agricultural land was mainly established in all countries in the 1960s to 1980s and that the actual portions of tile drained agricultural areas (SCHREIBER et al., 2003) were reached already in the mid of 1980s.

For the 1950s we assumed that the portion of tile drained agricultural land was only 20% of the present level in all Danube countries. This portion is increasing to 100% for Germany and Austria within a time period of 20 years and for the other countries it is assumed that the present portion of tile drained areas is reached in the mid 1980s.

Table IV.6: Changes of the specific P emissions of the people and industrial indirect dis charges in the Danube countries from 1955 to 2000

country	**1955**	**1965**	**1975**	**1985**	**1990**	**1995****	**2000***
	g/(Inh·d)	g/(Inh·d)	g/(Inh·d)	g/(Inh·d)	g/(Inh·d)	g/(Inh·d)	g/(Inh·d)
DE^{+}	2.5	3.7	5.5	4.3	3.6	2.9	3.0
AT^{+}	2.5	3.7	5.5	4.3	3.6	2.9	3.0
CZ^{++}	2.1	2.9	3.9	5.1	5.6	4.5	3.2
SK^{++}	2.1	2.9	3.9	5.1	5.6	4.4	2.9
HU^{++}	2.1	2.9	3.9	5.1	5.6	4.5	3.1
SI^{++}	2.1	2.9	3.9	5.1	5.6	4.4	2.9
HR^{++}	2.1	2.9	3.9	5.1	5.6	4.9	3.6
BH^{++}	2.1	2.9	3.9	5.1	5.6	4.8	3.5
CS^{++}	2.1	2.9	3.9	5.1	5.6	4.5	2.8
RO^{++}	2.1	2.9	3.9	5.1	5.6	4.3	2.4
BG^{++}	2.1	2.9	3.9	5.1	5.6	4.3	2.5
MD^{++}	2.1	2.9	3.9	5.1	5.6	4.2	2.3
UA^{++}	2.1	2.9	3.9	5.1	5.6	4.2	2.3

$^{+}$ development for westgerman federal states from 1955 to 1990 according to Behrendt et al. (1996)
$^{++}$ development for eastgerman federal states from 1955 to 1990 according to Behrendt et al. (1996)
* data according to Schreiber et al. (2003)
** average for the period 1990 and 2000

- River Nutrient Loads

Calculated nutrient loads were compared with observed data reported by VAN GILS et al. (2004) summarizing the available long term data for the nutrient loads and discharges of seven stations along the Danube, in the lower Drava and the Tisza. For location and characterisation of these stations see Figure IV.2 and Table IV.7. Because long term data on the Danube below the Hungarian border are limited, the available data on the Danube at Hercegszanto, the Drava and the Tisza were summarized to a virtual station

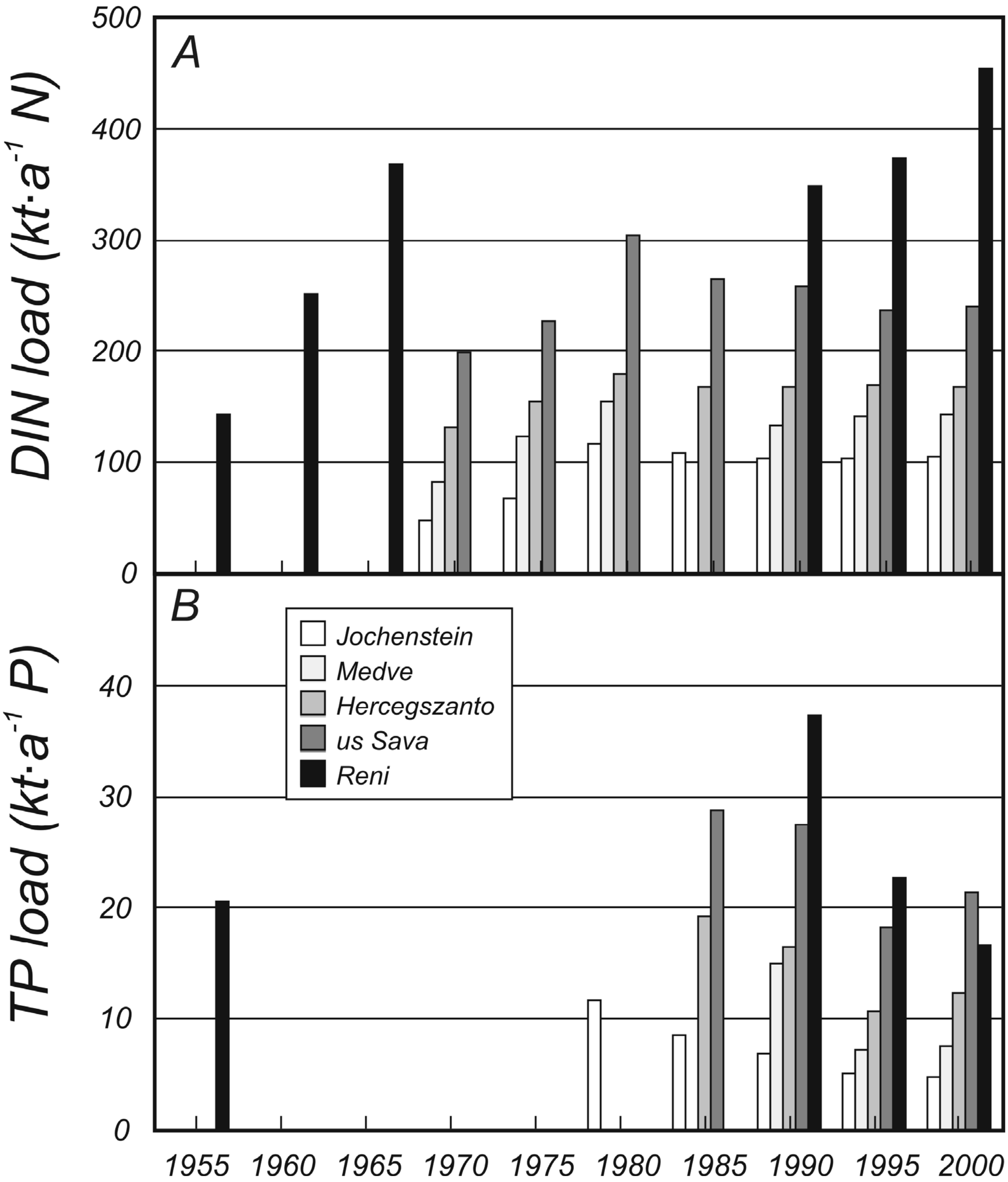

Figure IV.3: Temporal change of the load of dissolved inorganic nitrogen (DIN) and total phosphorus (TP) for the used monitoring stations along the Danube (A – load of dissolved inorganic nitrogen; B – load of total phosphorus).

Table IV.7: Characteristics of the water quality monitoring station used in this paper.

Station	River km	Catchment (km^2)	Altitude (m)
Jochenstein	2204	77,350	290
Medve	1806	132,720	108
Hercegszanto	1435	209,390	79
Dravaszabolcs	78 (Drava)	37,250	92
Tiszasziget	163 (Tisa)	140,000	74
upstream Sava (virtual station)		386,640	
Bazias	1071	565,770	70
Reni	132	788,110	4

(named upstream Sava). Its catchment area is only a little bit smaller than that of the Danube upstream of the confluence of Danube and Sava. Figure IV.3 shows the nutrient loads for the Danube stations over the total period of observations. Additional to the load data given by national authorities and the international monitoring programs, the figure includes the published data of ALMAZOV (1961) and ROJDESTVENSKY (1968), which represent an average of the nutrient load for 1949 to 1959 and for 1960 and 1965 for the Danube at the station Reni or upstream of the Danube Delta. Because these authors published only data on the load of the dissolved nutrient components (Dissolved inorganic nitrogen = DIN = NH_4-N + NO_2-N + NO_3-N; soluble reactive phosphorus = SRP), the data for soluble reactive phosphorus were transformed to total phosphorus (TP) using 1.64 as a mean ratio of TP to SRP for all periods and stations of the Danube. The observed discharges at the stations for the different time periods were used as input data to adjust the model to the flow conditions for the individual time periods. Additionally, all calculations were repeated with the long term discharges (average discharges for the stations over all time periods) as well as the observed lowest and highest discharges of all time periods to distinguish between the effects of the hydrology and of human activities within the catchments.

The barrage of Iron Gate went into operation in 1972 and has been retaining about 32% of the TP-load and 6% of the DIN load entering this barrage (SCHREIBER et al., 2003; VAN GILS et al., 2004). These additional nutrient retentions were taken into account for the periods after 1975. The only effected station of those given in Table IV.7 is Reni (Figure IV.2).

IV.3 Results and Discussion

- *Changes of nutrient emissions from point and diffuse sources*

Figure IV.3 gives a summarised overview on the temporal variation of the nutrient emissions into the Danube river system from individual pathways. For both nutrients the point source emissions increased from 1955 to 1990 by a factor of 4.9 (N) and 5.9 (P), respectively, and then decreased continuously. But the figure does not show that the temporal changes of the point source discharges differed in the upstream part of the basin (stations Jochenstein and Medve, representing mostly the German and Austrian part of the Danube basin) from those located downstream, especially for phosphorus. The faster development of the society in Germany and Austria leads already in the period 1975 to the highest specific P emissions per capita (see Table IV.5) and to a higher share of population connected to sewers and municipal waste water treatment plants in this part of the Danube. The reduction of P in detergents resulted in an initial decrease of the point P discharges during the 1980s followed by a further decline in the 1990s due to improved P elimination in the WWTP's. In the downstream part of the Danube the highest P point source discharges were estimated for the end of 1980's.

The end of the industrial pig farming in Romania, decreasing P content in detergents and to a minor part the improved P elimination efficiency in WWTP's of the lower Danube countries lead to reduced P emissions in the 1990s. The present P point source discharges is about 59% lower as 1990 but still 210 % higher than in 1955.

Changes of the N point sources discharges in the Danube river basin are mainly effected by the construction of sewer systems and the increasing removal efficiency in the recent years. The present N point source discharges decreased by about 32% in the 1990s but still exceed the level of the 1950s by about 195%. For the German and Austrian part of the Danube the present point discharges for P and N are only 54% (P) and 125% (N) compared to the 1950s.

- *Changes of diffuse source emissions*

According to Figure IV.4 the nitrogen emissions from diffuse sources has increased from the 1950s to the mid of 1980s by a factor of 1.6. In the 1990s this pollution decreased by about 13% mainly as the result of a decreasing N-surplus on agricultural areas (Figure IV.1). But the reduction of N-surplus is larger than the reduction of the diffuse nitrogen emissions, especially for the middle and lower part of the Danube. This is due to the longer residence time in the groundwater and the different retention rates for nitrogen in the unsaturated zone and in the groundwater. If the N-surplus will remain on the present level, a further reduction of the diffuse nitrogen emissions can be assumed for these areas in the next years.

Compared to nitrogen the phosphorus emissions from diffuse sources were relatively constant, if the assumption of the model is correct that the soil losses are nearly constant for the investigated time period.

If the uncertainties in the database are taken into account the present level of P emissions into the Danube river system from diffuse sources is probably 1.6 times that of 1950s.

- *Changes of total nutrient emissions into the Danube river system*

The maximum of total emissions occurred in 1985 for nitrogen with 920 kt/y and in 1990 for phosphorus with 110 kt/y (see Figure IV.4). According to Figure IV.4 total

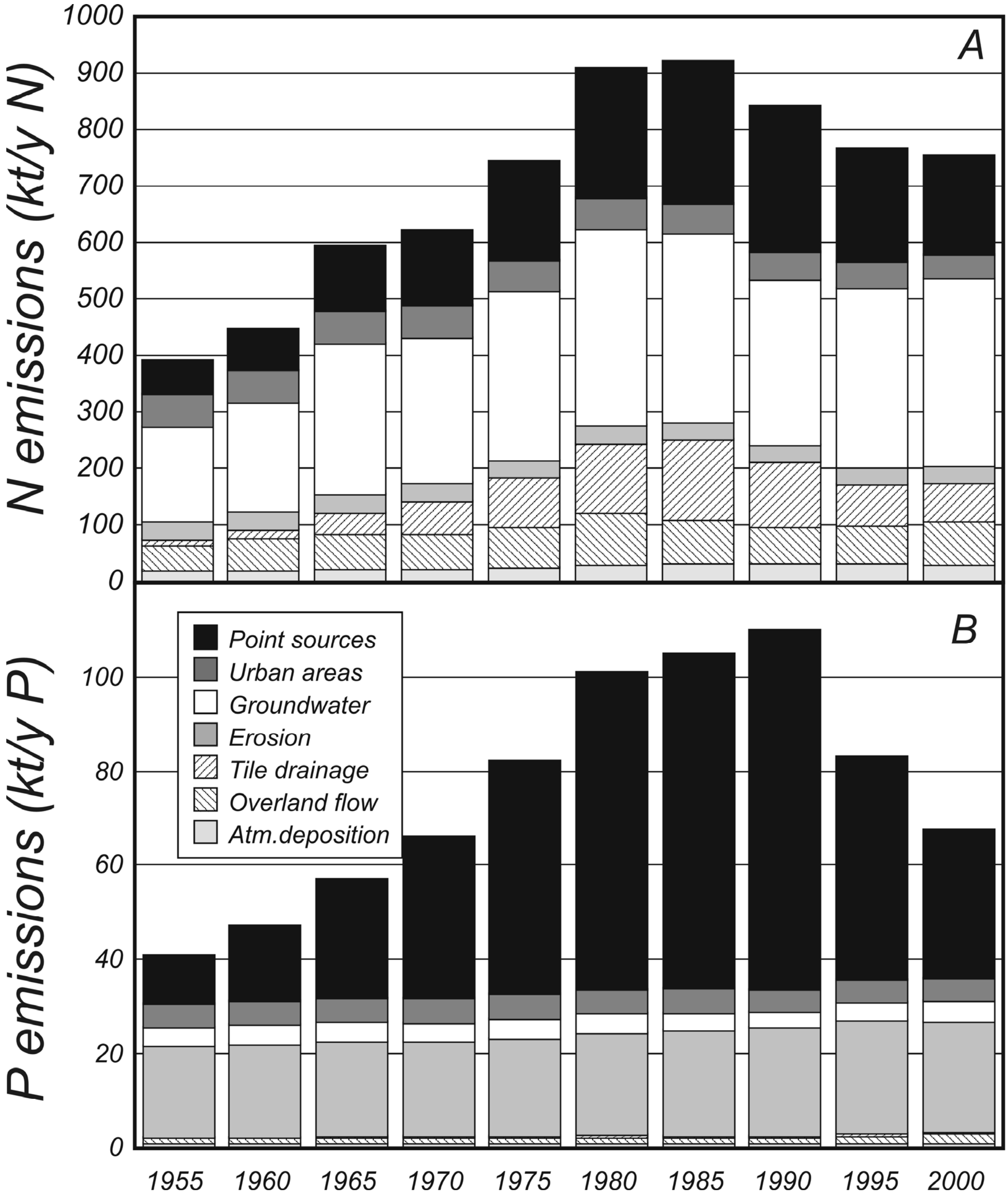

Figure IV.4: Change of the nutrient emissions into the Danube river system for the time period 1955 to 2000 (A – nitrogen, B – phosphorus).

emissions from 1955 to the maximum increased by a factor of 2.6 (P) and 2.4 (N). Compared to the maximum the reduction of the total emissions in the 1990s is 18 %for nitrogen and 39% for phosphorus. The present level of the total nutrient emissions is about 65% (P) and 92% (N) higher than in the 1950s.

The changes of the total nutrient emissions into the Danube river system are the result of large changes of the share of the individual pathways in the total emissions (Figure IV.4).

Within the 1950s the nitrogen emissions from groundwater (43%) were the dominant pathways followed by urban areas (15%) and point sources (15%). For 1990 the point discharges were increased to 27% and the groundwater emissions were reduced to 36%. The emissions from tile drained areas were the third largest pathway (15%). In contrast, the present situation is characterized once more by a share of groundwater emissions of more than 44% followed by point discharges (23%). Tile drained areas and overland flow contribute only about 10% to the total nitrogen emissions. The contribution of N emissions from atmospheric deposition and erosion are of minor importance for all time periods.

For phosphorus the dominant pathway was in the 1950s the erosion (48%) followed by the point discharges (25%) and the emissions from urban areas (13%). At the late 1980s the total P emissions were dominated to 69% from point discharges and the portion of erosion and urban areas was reduced to 21% and 5%, respectively. In the last investigated time period the point sources remained the major pathway (47%) but due to the large reduction of point discharges the portion of erosion has increased to 35%.

Also for phosphorus the contribution of the other pathways (groundwater, tile drainage, atmospheric deposition) is of minor importance for the whole investigated time period.

- *Changes of loads within the Danube river system*

Before the results of the model calculations for all time periods are presented and discussed, the uncertainty of these results have do be estimated. Figure IV.5 presents the comparison between the observed and calculated loads of DIN and TP for all stations and all time periods. For nitrogen the deviation between observed and calculated loads

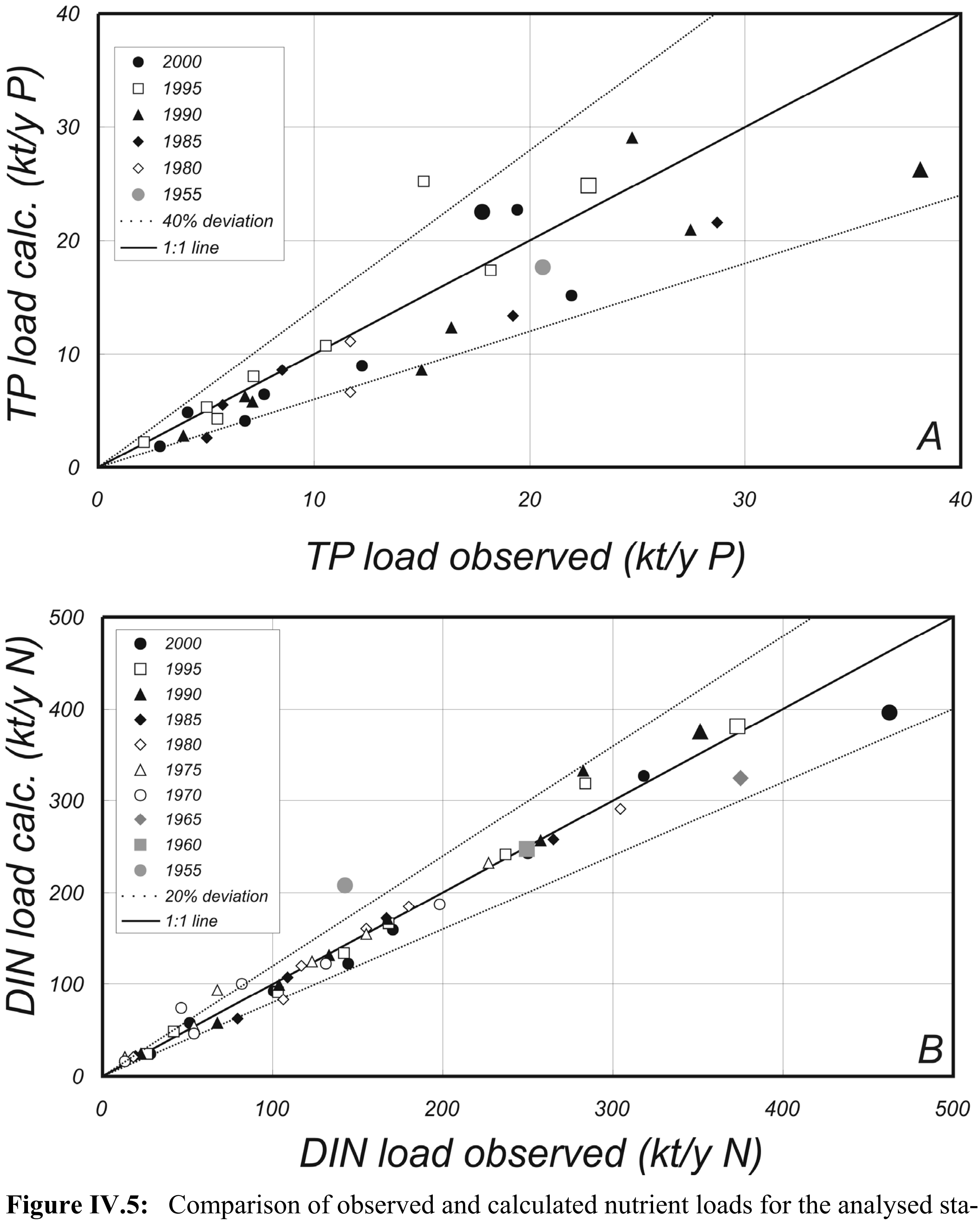

Figure IV.5: Comparison of observed and calculated nutrient loads for the analysed stations in the Danube for the period 1995 to 2000.

was only 13% on average. The deviation increases from the upper to the lower Danube, but decreases with approach to the present time. The model efficiency (EF) is estimated as 0.97 for the total DIN loads over all stations (MAYER & BUTLER, 1993). This value is lower for the area related specific DIN loads (0.9). Comparing the EF of the calculated loads for the temporal changes of the eight main stations (Figure IV.2) with the EF given by SCHREIBER et al. (2004, this volume) for 93 station and the present situation (total DIN load: 0.95; specific DIN load: 0.75), the difference are low for the total DIN load as well as for the area related specific DIN load. That means that the quality of the model results for the 8 larger stations over the whole time period of the last 50 years is similar to the findings for the last period for the set of 93 regionally distributed stations.

The deviation of observed from calculated loads is much higher for total phosphorus (see Figure IV.5). The mean deviation was about 22% for all periods and stations. The EF for the TP load is 0.75 (total load) and 0.60 (area related specific load). The deviations increased (EF decreased) from the upstream to the downstream stations. Whereas the mean deviation of TP loads is only about 19% for all stations upstream of Hungarian border the average of the deviation of the two downstream stations Bazias and Reni is about 44%. According to SCHREIBER et al. (2004, this volume) the EF of the model results for the present state is 0.93 for the total TP loads and 0.41 for the area related specific TP loads (64 stations), which is similar to or lower than our results for the temporal variation.

If the estimated deviations between observed and calculated nutrient loads are taken as a base for the uncertainty of the load calculations the ranges can be defined for the present state as 20% for nitrogen and 30% for phosphorus. For the period 1955 this uncertainty is assumed to 50% for both nutrients. The ranges of uncertainty for the other investigated time periods can be estimated by linear extrapolation of these boundary levels.

Figure IV.6 and IV.7 shows for three of the investigated 8 stations the observed and the calculated DIN and TP loads as well as the possible ranges of uncertainty.

According to Figure IV.6 the increase of the DIN load from 1955 to the maximum in the 1980s was between a factor of 2.2 (Reni) and 2.4 (Jochenstein). The highest DIN load was calculated as 456 kt/y N at Reni for the period of 1980. Surprising is the increase of calculated (1990: 375 kt/y; 2000: 396 kt/y) and observed (1990: 353 kt/y; 2000: 456 kt/y) DIN loads in the 1990s. This is mainly due to the different flow conditions (see Figure IV.6). The periods 1990 and 2000 were the lowest and highest flows in the Danube at Reni of all investigated time periods.

As shown in Figure IV.6 the observed and calculated DIN loads are within the range of uncertainty for all three stations except for Jochenstein in 1970 and Reni in 1955. Because it is not possible to reconstruct the database of the DIN loads given by ALMAZOV (1961) it is difficult to evaluate the large difference between calculated and observed

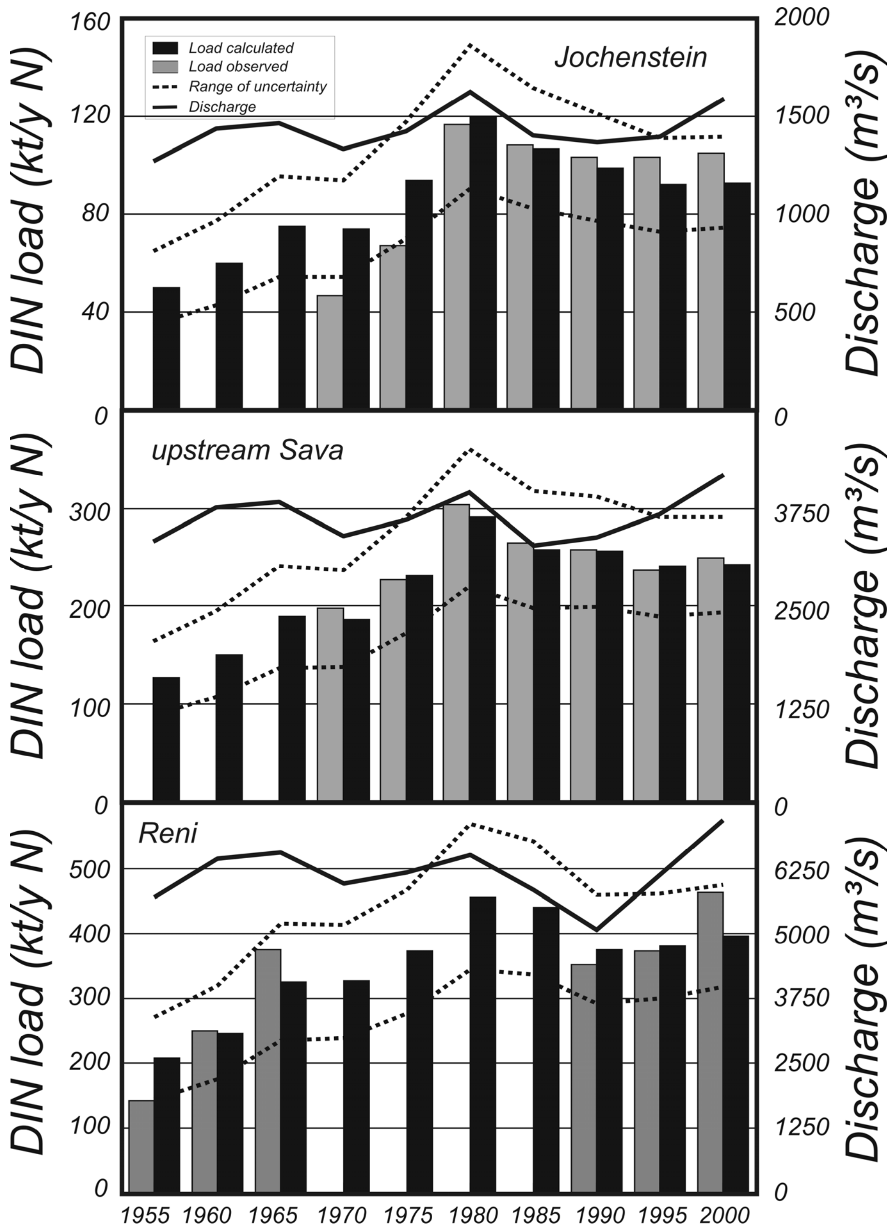

Figure IV.6: Comparison of the temporal change of the calculated and observed DIN load for the stations Jochenstein and Reni as well as the virtual station upstream Sava.

DIN load for 1955, especially against the background of the good agreement with the loads for the later time periods (1960 and 1965) given by ROJDESTVENSKY (1968). One reason can be that the measurements summarized by ALMAZOV (1961) represent mostly the conditions during spring, summer and autumn of the year, where the DIN concentrations are lower due to higher denitrification in the surface waters. But one the other hand some of the basic assumption for the input data of the model can be wrong for this starting period, because some of the data were interpolated backward from the trend of existing data for the 1960s.

The overestimation of the calculated DIN load for 1970 and partly 1975 at the station Jochenstein can be related to the oversimplified description of the processes in the soils within the model. It is assumed that changes in nitrogen storage in the soil are negligible (BEHRENDT et al., 1999), due to missing data on the temporal changes of the N content of different soils. But the nitrogen content of the soil can change because of immobilisation and mineralization, especially for longer time period (ADDISCOTT, 1988). Consequently, the large difference of calculated and observed DIN load at Jochenstein in the 1970s could be an indication that the process of immobilisation has to be taken into account for long term simulations for periods with increasing N surplus. Further reasons for these deviations can be errors in the input data, especially for the tile drained agricultural areas, point sources and atmospheric deposition, as well as higher uncertainty of the observed loads during the initial phase of monitoring.

Besides the DIN loads presented in Figure IV.3 for the eight Danube stations WEILGUNI & HUMPESCH (1999) published also data for the concentrations of nitrate and ammonia for the Danube at Vienna Nußdorf for the time period from 1956 to 1996. The subdivision of the model MONERIS for the Danube allows the estimation of the nutrient loads for this station, too. Average annual nutrient concentration can be calculated by the model after division of the estimated loads by flow. Therefore we can compare the calculated DIN concentrations for the station Vienna Nußdorf also with values given by WEILGUNI & HUMPESCH (1999). The average DIN

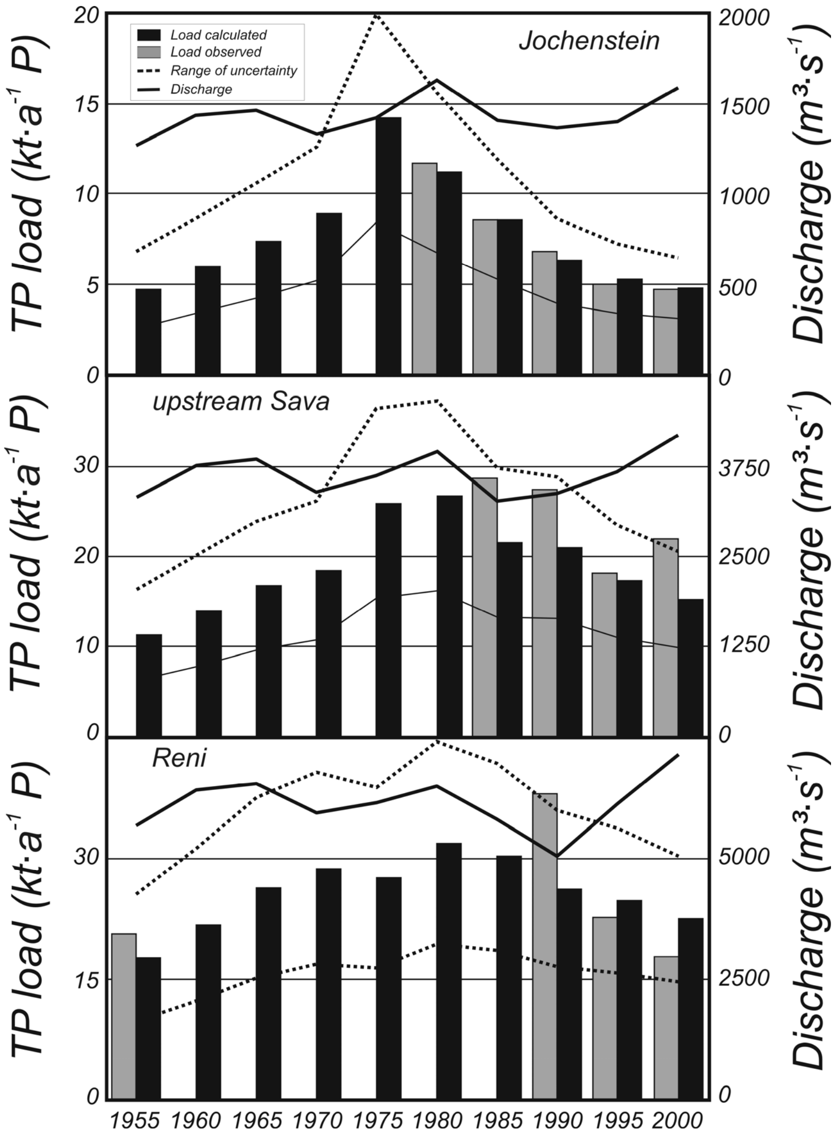

Figure IV.7: Comparison of the temporal change of the calculated and observed TP load for the stations Jochenstein and Reni as well as the virtual station upstream Sava.

concentrations for the time periods 1975 to 1990 and 1995 were estimated as 2.4 and 2.2 mg/l N. This values are very close to the nitrate nitrogen concentrations published by WEILGUNI & HUMPESCH (1999) for the time period 1977-1990 (2.38 mg/l N) and for the year 1995 (2.34 mg/l N). According to these authors, nitrate nitrogen represents more than 90% of dissolved inorganic nitrogen (DIN). Furthermore, the trend of the nitrate nitrogen concentration given by these authors seems to be very similar with the trend of the DIN concentration calculated with the model.

As shown in Figure IV.7, a very good agreement of the temporal trend of the observed and the calculated loads was found for the TP load at Jochenstein. All calculated loads are close to the observed loads. With respect to temporal variability, the highest TP load was about 14.2 kt/y P in the period 1975 for this station. Compared to 1955 (4.7 kt/y P) the TP load in 1975 was about three times higher. The calculated TP load for 2000 was 4.8 kt/y P, which is the same level as in 1955.

The observed and calculated TP loads are also in the range of uncertainty for the sum of the load of Danube, Drava and Tisza at the Hungarian border (virtual station upstream Sava). But the deviations are higher and with exception of the period 1995, the observed load is higher than the calculated load. The highest TP load was estimated as 26.7 kt/y P for the period 1980. The increase from 1955 to 1975 was by a factor of 2.4. In the present time the TP load is about 15.9 kt/y and about 44% lower than in 1980 but 35% above the level of 1955.

For the station Reni the observed and calculated TP loads are not in the range of uncertainty for the period 1990 and much lower than the observed load. The underestimation of the calculated TP load can be explained by missing additional point or diffuse P sources in the lower part of the Danube or errors in the observed TP load.

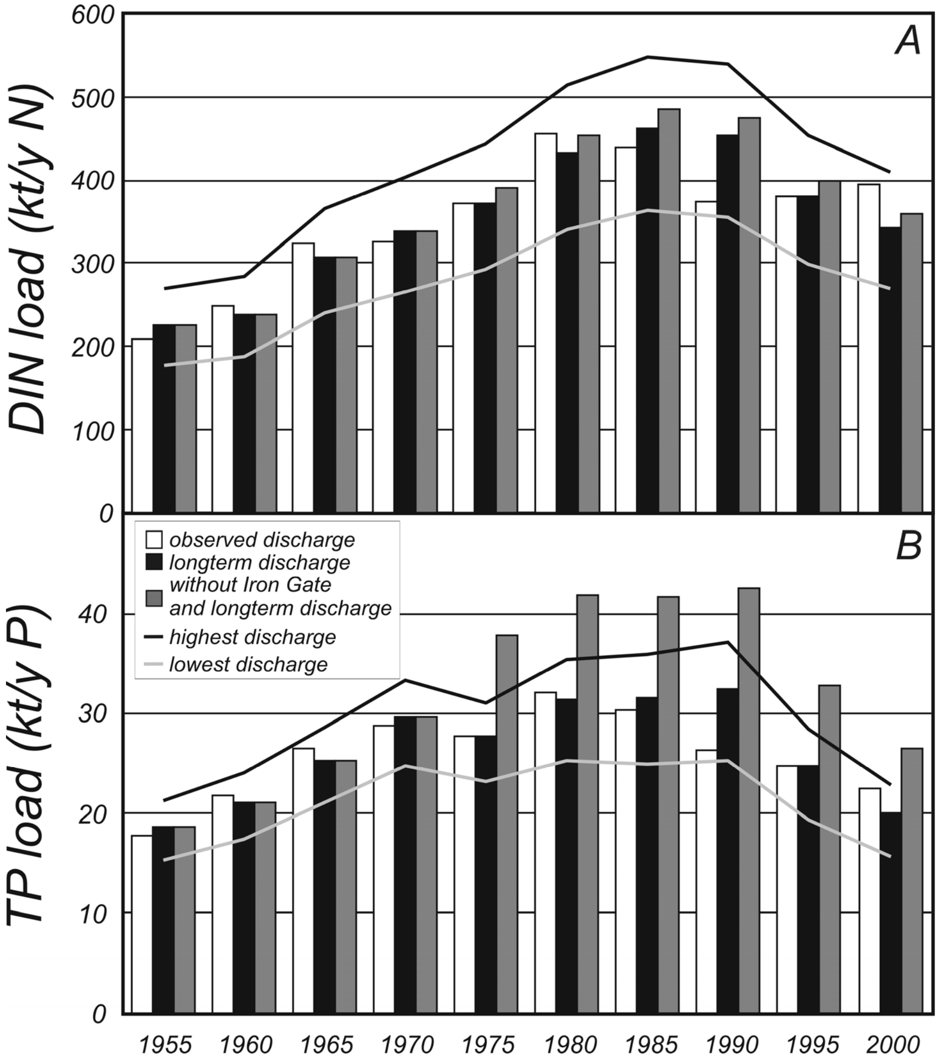

Figure IV.8: Calculated nutrient loads for the Danube at the station Reni for the period 1955 to 2000 for different flow conditions as well as with and without Iron Gate barrage

The calculated TP load shows a maximum for 1980 (33 kt/y P). Between 1970 and 1980 a small reduction is visible that is due to the establishment of the Iron Gate barrage; this period is accompanied by a increase of the retention of phosphorus in the Danube that is larger than the increase of the P emissions (Figure IV.4). Without Iron Gate the TP load at Reni would have reached a maximum of 42 kt/y P in 1990 (see Figure IV.8). The P retention within Iron Gate is the reason that the TP load from 1955 to the maximum in 1980 increased only by a factor of 1.8 and that the present TP load at Reni is only 27% above the TP load in 1955.

While the TP load reduction in the upper part of the Danube is clearly due to the changes of the emissions into the river system, the reduction in the lower part is mostly influenced by the change within the river system. The model results support the results of other studies (HUMBORG et al., 1997; LANZELOT et al., 2002) that the Iron Gate barrage is a large sink for phosphorus and silica and important for the change in the nutrient transport from the Danube into the Black Sea.

Emissions from diffuse sources are strongly dependent on flow conditions (NOVOTNY & CHESTERS, 1981). Therefore the variation of the diffuse nutrient emissions is also influenced by the variation of flow conditions in the different time periods.

To separate the effect of changing flow and human influences on the diffuse emissions all calculations were repeated for all time periods but using average long-term flow conditions in the Danube. The long-term flow was estimated as the average of the observed flow data for the individual stations for all investigated time periods. Additionally, the DIN and TP loads were calculated for the highest and lowest flows observed for each area upstream or between the eight stations within all time periods, to show the possible variance of the loads due to extreme flow conditions. Figure IV.8 presents the results of these scenario calculations for the station at Reni. This figure includes also the estimated load with and without the Iron Gate to illustrate its effect on the nutrient loads.

At long-term flow conditions the variations of both nutrients are a little larger than for the observed flows. The loads increase from the 1950s to the 1980s by a factor of about 2.1 for nitrogen and 2.2 for phosphorus.

In contrast to the observed data and the calculated DIN load for observed flow in the 1990s (increase of DIN load), this load is decreasing (-25%) for long term flow conditions. This illustrates that for nitrogen small and moderate reductions of human influences were overridden by increased flow during the 1990s. The range of the possible variation of the nitrogen load of the Danube at Reni at the highest and lowest flow conditions is about 40% (see Figure IV.8). Therefore human influenced trends of nitrogen loads can be only identified for short time periods or individual years, if their changes are also in this order of magnitude. For smaller changes the identification of such trends needs longer time series or the support of models or other methods that take the variation of flows into account (see e.g. STALNACKE & GRIMVALL, 2001).

The TP load for long-term flow conditions from 1955 to the 1990 was by a factor of 1.7 higher than for observed flows. For long-term flow conditions the present TP load is only 10% higher than estimated for 1955. The influence of extreme flow conditions on the TP load at Reni is similar as for nitrogen. Because the reduction of human indicators was larger for phosphorus (38%) than for nitrogen (14%), the calculated TP loads decrease in the 1990s. Furthermore, it was found that the maximum of the TP load is changing from 1985 to 1990 (similar to the emissions).

According to the model calculations the establishment of the Iron Gate barrage is of minor importance to the nitrogen load in the Danube but important for the TP load (see

Figure IV.8). If the assumed TP retention of 32% within the Iron Gate barrage is realistic (VAN GILS et al., 2004), the influence of this measure is larger than the influence of different flow conditions.

- *Comparison with former studies*

A comparison of the results of this study with other analyses for the whole Danube and for the total time period of 50 years is not possible. All published studies cover only the time period from the end of 1980s to the mid of the 1990s. Estimations of the nutrient emissions into the Danube were published within the PHARE-study (EU/AR102A91, 1997; see also SOMLOYDY et al., 1999) and the HASKONING study (HASKONING, 1994). ZESSNER & VAN GILS (2002) revised the emission situation estimated by EU/AR102A91 (1997) for 1992 and enlarged the time period for the nutrient emissions in the Danube for the situation around 1988 and 1996/1997. GARNIER et al. (2002) applied the RIVERSTRAHLER model for the Danube for the period 1988 to 1991 and 1994 to 1996. The methods used for the estimation of the nutrient emissions are different. All earlier studies were done only on a country level or large subcatchments.

The results of all of these studies are shown in Table IV.8. Beside the results of the for-

Table IV.8: Comparison of the nutrient emissions and loads within the Danube according to the results of different studies for time periods at the begin and in the mid of the 1990s (Total – toal emissions; diffuse – emissions from diffuse sources; point – point source discharges; DIN – calculated load of dissolved inorganic nitrogen; TN load – calculated load of total nitrogen; TP load – calculated load of total phosphorus).

time period	Nitrogen					Phosphorus				Source
	total	diffus	point	DIN load	TN load	total	diffus	point	TP load	
	kt/y	kt/y	kt/y	kt/y	kt/y	kt/y	kt/y	kt/y	kt/y	
1992	677	455	222			93	53	40		HASKONING (1994)
1988-1992	1130	830	299		447	150	100	50	46	ZESSNER & VAN GILS (2002)
1988-1991	1257	591	666	858	936	178	29	149	51	GARNIER et al. (2002)
1988-1992	843	587	257	375	473	110	34	77	26	this study
1996/1997	898	584	314			108	46	62		ZESSNER & VAN GILS (2002)
1994-1996	972	456	516	644	710	116	29	87	35	GARNIER et al. (2002)
1993-1997	769	569	200	381	461	83	36	47	25	this study

mer studies the table includes also the findings of our analysis for the similar time periods.

The HASKONING study did not include about 20% of the Danube basin (Bosnia-Herzegovina and Serbia and Montenegro), therefore the estimated nutrient emissions are the lowest compared with the other results.

Compared with the results of ZESSNER & VAN GILS and GARNIER et al. (2002) the estimated total nutrient emissions of our study are the lowest for both time periods. Garnier et al. (2002) calculated the highest emissions. The difference between the model results is lower for the period of the mid of the 1990s.

In spite of these large differences the results of this study agree very well with the results of GARNIER et al.(2002) for the diffuse emissions of nitrogen and phosphorus. On the other hand MONERIS results fit better with the values given by ZESSNER & VAN GILS for the point source discharges. In general, the difference is larger for the point source discharges than for the diffuse nutrient emissions.

The reduction of the nitrogen load from the period 1990 to the period 1995 is the lowest for the MONERIS results (9%) followed by the results of ZESSNER & VAN GILS (2002) (20%) and GARNIER et al. (2002) (23%). Similar to our results, ZESSNER & VAN GILS (2002) found that the reduction of point nitrogen discharges was larger than for the diffuse sources. GARNIER et al.(2002) estimated the same reduction for both sources of nitrogen (23%). For phosphorus the largest reduction of the total emissions was estimated by GARNIER et al. (2002) (35%) followed by similar values given by ZESSNER & VAN GILS (2002) (28%) and this study (25%).

ZESSNER & VAN GILS (2002) estimated the nutrient loads from the emissions using the same equations as given by BEHRENDT & OPITZ (1999) and used in MONERIS, therefore the retention expressed as percentage of the total emissions is very similar. The nutrient retention within the surface waters of the Danube according to the RIVERSTRAHLER model varies between 32 and 34 % for DIN and 26 and 27% for TN but this retention takes into account only the retention in the main stream sections. The retention calculated in this study is about 20% (DIN) and 16% (TN) higher. In contrast to nitrogen the amount of the phosphorus retention is for both models very similar (Riverstrahler: 70-71%; MONERIS: 70-76%).

The deviation of observed and calculated loads is for the RIVERSTRAHLER model for both time periods between 16 and 18%. In both cases, the calculated loads are larger than the observed DIN loads. This can be caused by overestimation of the emissions or underestimation of the retention within the surface waters. For MONERIS the deviations are 2-6%.

The date source used for the observed DIN load differ, too. GARNIER et al. (2002) used the observed loads given by COCIASU et al., 1996 and pers. comm.) for the Station at Sulina (located in one of the main branches of Danube in the Delta). On the other hand, ZESSNER & VAN GILS (2002) and this study used the data for the station at Reni (about 100 km upstream) (see VAN GILS et al., 2004).

Both loads are very different, especially for the first time period (Sulina: 723 kt/y; Reni: 352 kt/y DIN). Because the RIVERSTRAHLER model was calibrated for this time period and for the higher load for Sulina, the calculated nitrogen emissions are much higher as for MONERIS.

IV.4 Conclusions

The model results of changing nutrient emissions of different point and diffuse sources as well as loads in the Danube in the last 50 years can explain the observed situation, which is characterised by observed loads of 8 different monitoring stations located along the Danube River. Because the deviation between observed and calculated nutrient loads is only 12% for DIN and 25% for TP for all time periods and stations, it can be assumed that the model results reflects the changes of the pollution situation within a range of uncertainty.

The identified changes for the nutrient loads of the Danube into the Black Sea are relatively low over time. But the changes of the ecological state of the Western Black Sea in the last 40 years are described as large (Mee, 2001; Lancelot et al., 2002; Horstmann et al., 2003). This indicates a high sensitivity of the ecological state of the Western Black Sea to changes of the nutrient loads of the Danube.

In general, it is assumed that the estimation of diffuse sources and retention in surface waters is much more complicated and uncertain than the calculation of point source discharges. But the comparison of results from different models shows that the variability of diffuse nutrient emissions and estimated P retentions is much lower than that of point source discharges. Therefore, the establishment of a better database for the point source discharges and their changes is a main task for the improvement of the models for such large river basin as the Danube.

The difference among the observed loads in the most downstream part of the Danube are very large in the 1990s; a harmonisation of the measurements to get a better and unified database for the calibration of the models is urgently needed.

IV.5 Acknowledgements

The results presented stem from the project "Nutrient management in the Danube Basin and its impact on the Black Sea" supported under contract ENVK1-CT-2000-00051 within the 5th EU Framework Programme. Details of this project can be found on the project homepage http://www.danubs.tuwien.ac.at, ELOISE No 455/38.

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