Institute for Water Quality and
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Deliverable D 6.1

Outlines of a Handbook for International Nutrient Balances on Catchment Scales

including

Deliverable D 6.2

Minimum-Requirement-Data-List

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Minimum-Requirement-Data-List

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1. INTRODUCTION

This handbook is designed for nutrient management on larger catchment scales. As a consequence the methods introduced and suggestions given do not comprise the full content of the requirements of the European Water Framework Directive. Furthermore the handbook is restricted to N and P only. The focus is on the management of nutrient emissions into the hydrosphere in order to abate eutrophication. Acidification, climate change, healthy diet, etc. are out of scope of this handbook.

The Water Framework Directive (WFD 2000/60/EC) rules a cross border surface water related cultivation of river catchments. The achievement of a “good ecological status” of surface water is aimed at until 2015. Therefore implementation of the WFD strongly needs reliable and transferable methods for classifying water quality on the one hand but also methods which make detailed inventory of nutrient sources and emissions to surface waters in specific catchments possible for a further performance of international water management strategies to reduce or control nutrients. Emission modelling can be a proper method to come aware of the demands of the WFD. However, there are many unsolved problems yet, concerning process understanding, quantification of emission pathways, model choice, model configuration and also calibration of these models by well adapted load measurements in general but also data availability and consistence. These problems are heightened by international and in some cases even national heterogeneity concerning the methodological approach, data and stand of the art.

In this handbook developed as a part of the EU funded daNUbs Project (EVK 1-CT-2000-00051) results from nutrient balancing in the Danube basin (size: app. 800000 km^2) are presented. Results concerning nutrient balances for the time period 1998-2000 stem from the emission model MONERIS carried out for 388 catchments (< 100 km^2 to >16000 km^2) covering the whole Danube river basin. A detailed documentation of the outcomes of MONERIS emission modelling is presented in Deliverable 5.5. Furthermore results from 5 case study areas from Austria, Hungary and Romania, subdivided into 26 subcatchments with catchment areas from 40 km^2 to 2330 km^2 are presented. On the one hand this is necessary due to the structure of MONERIS where nutrient balances are calculated on subcatchment scale and so even large scale investigations include some meso scale sized catchments (see areas of the 388 sub catchments) on the other hand more precise outcomes from case study areas can light up further needs for large scale investigations. A detailed documentation of the investigations in the case study areas is presented in Deliverable 1.1, 1.3 + 1.4.

In general nutrient balances on catchment scale can be prepared in a wide range concerning spatial, temporal and process resolution. The most simples approach is input output measurement, where the catchment is dealt with as a black box. From such an approach no predicates about nutrient sources, processes, pathways or prognoses are possible. Nevertheless input output measurements are the only possibility to prove the reliability of models more applicable for nutrient management, such as emission models. In these models dependent on model philosophy a more or less detailed parameterization concerning nutrient sources and emission pathways to the river under specific natural and anthropogenic conditions must be provided. Concerning this, the reliability of nutrient balances on catchment scale strongly depends on process knowledge and especially on data availability, precision and the adoption of a monitoring programme to the specific conditions of the catchment (for example detection of high flow events). The use of proper emission models can guarantee an adequate cause analyse identifying necessary management strategies. Nevertheless a proper model calibration should be guaranteed by adequate load calculations (temporal resolution, long term data set). Furthermore special events like high flow events but even spatial heterogeneities (waterside dependent emissions from tributaries or point sources) should be considered in the monitoring strategy (see chapter 4).

- On the basis of project results, especially output from work package 1 (“nutrient balances for case study regions”), 2 (“erosion”), 4 (“monitoring improvement”) and 5 (“model application”), this report will focus on the following main subjects:

- a short grading of MONERIS concerning other emission models
- evaluation of main sources and pathways of nutrients
- possibilities for estimating nutrient emissions (main pathways)
- data availability on catchment scale
- data sets of major importance for nutrient balancing on catchment scale
- “Minimum requirement data list”

These main topics will be discussed for nitrogen and phosphorus separately as the basic sources, pathways and mobilisation and retention processes differ considerably. Analogical the weighting of input data does (Deliverable 5.5, Deliverable 1.3, Scheer et al., 2004).

2. NUTRIENT BALANCES – RESULTS FROM DANUBE EMISSION MODELLING

In the first part of this chapter a short summary of results concerning a model comparison is given (ATV-DVWK, 2004). Secondly an overview concerning the MONERIS model structure and results of the Danube project is given to communicate insight into model result accuracy.

A detailed overview and test of emission models and nutrient quantification tools is provided in the EC Project EUROHARP (Towards European Harmonised Procedures for Quantification of Nutrient Losses from Diffuse Sources; http://euroharp.org). – started in 2001.

2.1. EMISSION MODEL COMPARISON

In the model comparison three different emission models for mesoscale catchments (200 – 2000 km^2) were tested in three German river catchments:

- MOBINEG 2001 (F&N Umweltconsult, Version 0104)
- STOFFBILANZ (TU-Dresden, Institut für Geographie)
- MODIFFUS (FAL, Zürich-Reckenholz)

but also one emission model which was primarily conceived for larger catchments:

- MONERIS (Umweltbundesamt, Berlin/IGB- Berlin).

Scope of the ATV-DVWK project was to test these emission models in reference to quantify diffuse emissions to the surface water with respect to the WFD. So the main criteria were:

- Plausibility
- Accuracy
- Cause Analyses
- Data availability
- Applicability

Summarized for all emission models deficits were found concerning the quantification of diffuse phosphorus emissions. For MOBINEG 2001 and MODIFFUS the quantification of diffuse nitrogen emissions was insufficient either.

The insufficient results of phosphorus emission modelling can predominately be traced back to the fact that erosion modelling was unreliable in all cases.

Although the MONERIS model approach and results concerning the pathway erosion were most convincing, it was annotated that erosion is only calculated for arable land and erosion stem from woodland or “natural erosion” defining the natural background of erosion processes is neglected by the MONERIS erosion approach.

Another unsolved problem in the MONERIS erosion approach is the calculation of the Sediment Delivery Ratio (= SDR, discussed below) with the help of an empiric formula using mean slope data and a percentage of farmland. This simplification makes concerted measures impossible.

Comparing model results with load calculations for nitrogen and phosphorus MONERIS achieves convincing results for nitrogen (TN: deviation 15 % and NO_3-N: 20-30 %). For phosphorus MONERIS results show a much higher deviation (SRP: ~30% and TP: ~45 %).

Nevertheless in total the MONERIS approach produces the best agreement with load calculations in comparison to the other models. This is particularly remarkable due to the fact that the other emission models were especially designed for nutrient balancing on meso scale.

The assessment for the availability of data being used for MONERIS is also convincing. In comparison to the other model input data (higher parameterization and spatial resolution) most of the MONERIS data are easily available. They are listed up below and discussed in detail later:

- Soil map
- Slope
- Precipitation
- Discharge on subcatchment level
- Atmospheric N deposition
- Landuse
- Drainage Area
- Soil degradation
- N-surplus/ P-enrichment

The implementation of assumptions, estimates, expert knowledge etc. of numerous parameters increases uncertainty of model results. Therefore data availability is essential for emission modelling. In general the smaller the catchment the more data is available. The use of mean data on subcatchment base as performed in the MONERIS approach leads to a loss of spatial information. According to this, local measures for nutrient reduction can not be derived from MONERIS results. This fact points up the limited practicability of MONERIS concerning meso and small scale nutrient balancing; for large scale investigations this deficit is less important. In the contrary the lower resolution of data needed makes nutrient modelling on large scale practicable. Furthermore the model comparison highlights the problem of the tested meso scale models which are able to produce spatial high resolute information (based on a detailed data base) on the one hand but with a lower model result accuracy than MONERIS on the other hand, which makes the chance for accurate measures questionable.

As results from daNUbs project document the MONERIS approach is also suitable for meso scale catchments (for example Wulka catchment 384 km^2). Even in subcatchments down to 40 km^2 convincing model accuracy was achieved for nitrogen (chapter 2.2.2).

2.1.1. Swat application

The application of the SWAT 2000 model for P and N flows has a high demand on the input parameters. Besides this high demand and the problem of data availability the results obtained within the 5 catchment areas were not satisfying. A detailed documentation concerning the SWAT application in the case study areas is given in Deliverable 1.1, 1.3 + 1.4 and 2.1.

2.2. EMISSION MODELLING WITH MONERIS

The GIS oriented Model MONERIS (**MO**delling **N**utrient **E**missions in **RI**ver **S**ystems) was developed for the estimation of nutrient inputs by various point and diffuse sources into German river basins larger than 1000 km² for the periods 1983 to 1987, 1993 to 1997 and 1998-2000 (BEHRENDT et al., 2000; BEHRENDT et al., 2002). Within this project the model was applied to 388 sub-basins of the Danube . The estimations were made for the period 1998 to 2000.

The basic input into the model are data on discharges, data on water quality of the investigated river basins and a Geographical Information System integrating digital maps as well as statistical information for different administrative levels (chapter 6).

Whereas the inputs of municipal waste water treatment plants and of direct industrial discharges enter the river system directly,

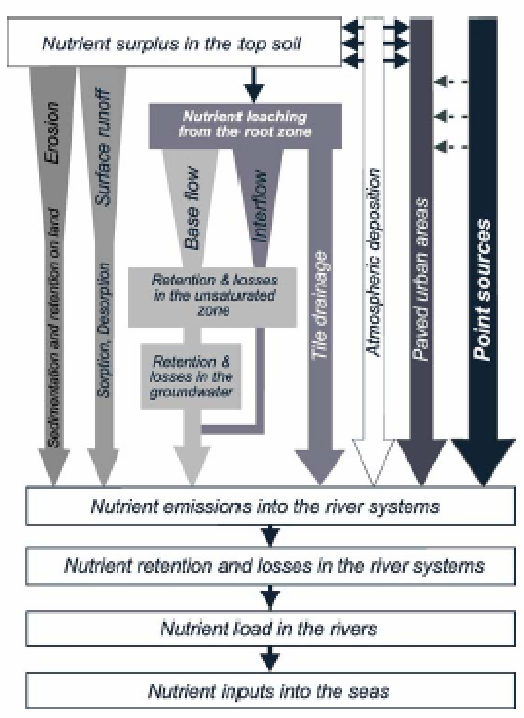


Figure 2.1: Pathways and Processes in MONERIS.

the sum of the diffuse nutrient inputs into the surface waters is the result of different pathways realized by several runoff components (see Figure 2.1).

The distinction between the inputs from the different runoff components is necessary, because the concentrations of substances within the runoff components and the processes within these runoff components are very different. Therefore MONERIS takes seven pathways into account:

- discharges from point sources
- inputs into surface waters via atmospheric deposition
- inputs into surface waters via groundwater
- inputs into surface waters via tile drainage
- inputs into surface waters via paved urban areas
- inputs into surface waters by erosion
- inputs into surface waters via surface runoff (only dissolved nutrients)

Within the diffuse pathways, various transformation, loss and retention processes are identified. To quantify and forecast the nutrient inputs in relation to their cause knowledge of these transformation and retention processes is required. This is not yet possible through detailed dynamic process models because the existing database is limited for medium and large river basins. Therefore, existing approaches of macro-scale modelling will be complemented and modified and, if necessary, attempts will be made to derive new applicable conceptual models for the estimation of nutrient inputs via the individual diffuse pathways.

An important step in the development of the individual sub-models 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.

In chapter 4 sub-models dealing with the main pathways are presented more in detail.

2.2.1. MONERIS model accuracy for the Danube catchment (large scale)

Comparison of observed and calculated nutrient loads

As shown by Billen & Garnier (1999) and Behrendt & Opitz (1999) the nutrient emissions into a river system can not be directly compared with the observed load because retention processes within the system of surface waters have to be taken into account. The MONERIS model includes the possibility to calculate this retention for phosphorus and nitrogen based on river parameters as specific runoff and hydraulic load (see chapter 4.6).

If these retention formulas are applied to the emissions into the Danube resp. its subcatchments the phosphorus and nitrogen load can be estimated and compared with the observed loads given above. The result for the investigated time period 1998-2000 of this comparison is presented in Figure 2.2 for dissolved inorganic nitrogen (DIN), total nitrogen (TN) and total phosphorus (TP), respectively.

Especially for the both nitrogen components the calculated loads agree well with the results of the measurements. The deviation between the measured and calculated loads exceeds a deviation of 50 % for only 9 (DIN) and 2 (TN) subcatchments (“only”).

The mean deviation is below 21 or 22 % for DIN and TN load, respectively. If the possible error of the observed load is taken into account (Del. 5.5), the real deviation can be assumed to be less than 20 %.

The highest deviation in observed nitrogen loads were found for the subcatchments within the Hungarian part of the Danube and some Romanian catchments. Because the sampling frequency for the Romanian catchments is only monthly, it can be assumed that the reason for the deviation is also an insufficient estimation of the mean annual DIN-load.

For phosphorus, Figure 2.2 shows that the deviation between calculated and observed loads is higher than for nitrogen. The mean deviation between calculated and observed loads was estimated to be 30 % which is about 10% higher than for nitrogen. In contrast to nitrogen the deviation is larger than 50% for 12 catchments and for about one third of the catchments larger than 30%. Additionally a clear tendency exists that the calculated P loads are below the observed loads.

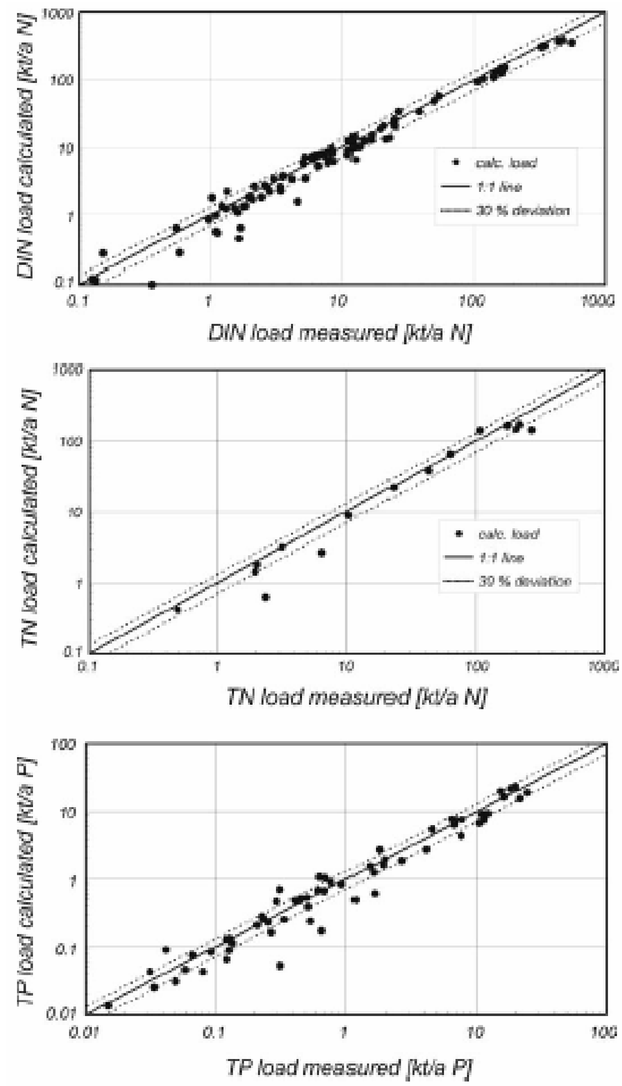


Figure 2.2: Comparison of observed and calculated Loads (DIN, TN and TP).

A more detailed analysis of the catchments with the high underestimation of the phosphorus load shows that most of these catchments are also located in the Hungarian part of the Danube. In general it can be assumed that the deviation results due to an overestimation of the retention by overestimation of the surface water area and underestimation of the total P-emissions especially for Kapos, Sio and Lonyai. Because the measurements for these catchments are showing mostly strong dilution functions for the concentration due to high point source discharges, it can be assumed that especially the point source discharges are underestimated for these catchments within the model due to the data base (see chapter 4.5.1).

The overestimation of the surface water area is a result of the application of the used equation in MONERIS, 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 this formula. In this case the retention will also be overestimated.

As figure 2.3 shows, the calculated nitrogen load increases continuously with increasing river length. For the stations of the Danube downstream of the Iron Gate an overestimation of the calculated TP-loads occurs in comparison to the observed TP-loads (see Figure 2.3). This indicates that the Iron Gate reservoir is an additional sink for phosphorus. If such an additional P retention in the catchment including the Iron Gate reservoir is taken into account and the amount of this retention is estimated by minimizing the deviation of the observed and calculated TP-loads for the Danube stations downstream of the Iron Gate, the P-retention of this reservoir is found to be about 8.5 kt/a P or 36 % of the TP-inputs into the reservoir. If the P retention in the Iron Gate reservoir is taken into account the calculated TP-load of the Danube to the Delta was 22 kt/a P for the period 1998-2000, which is 21% above the TP-load observed for the station Reni the last Danube station upstream of the Delta. The main processes of the phosphorus retention are sedimentation and sorption. From the model results it can be concluded that about 46 kt/a P or 68 % of the P-emissions into the surface waters will be retained. About 20% of this total retention would be realized within the reservoir at the Iron Gate.

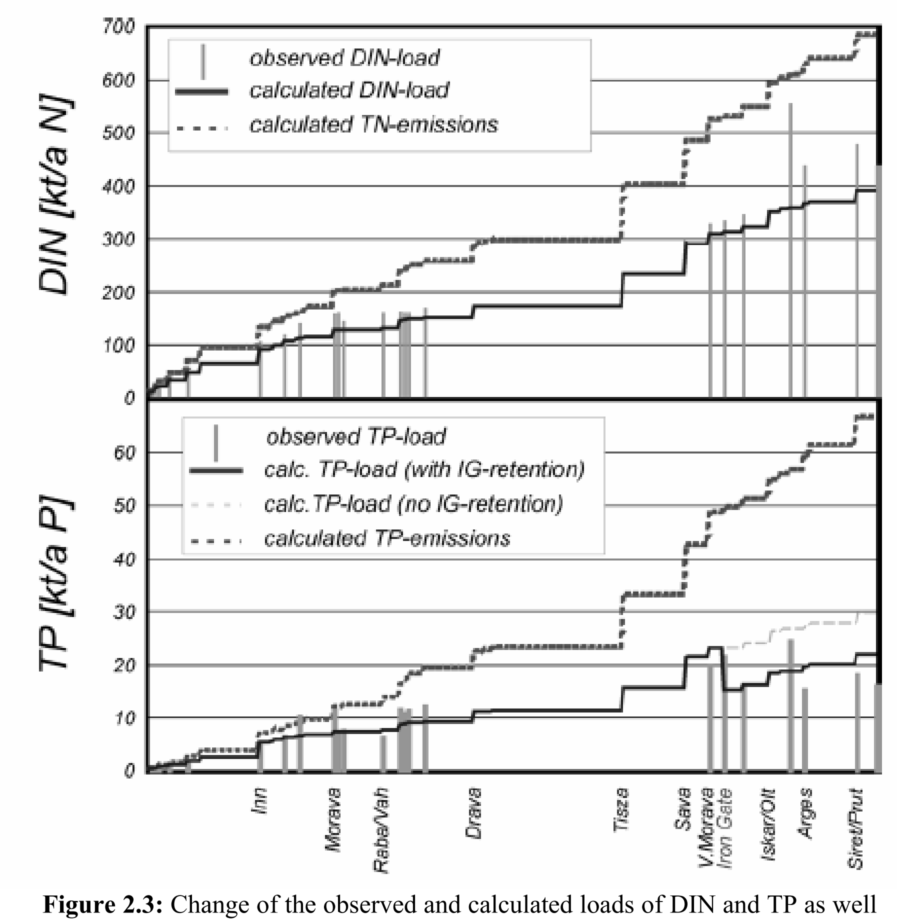


Figure 2.3: Change of the observed and calculated loads of DIN and TP as well as N- and P emissions along the Danube for period 1998-2000.

In general, the comparison of the calculated and observed nutrient loads shows that for nitrogen a sufficient level of agreement was already reached. Before the approaches used by the model for the different pathways are specified for specific conditions in the Danube river basin, it is necessary to improve the spatial resolution of the model so that nitrogen surpluses can also be calculated on a regional basis rather than simply on a national basis (see chapter

4.2.1). This requires the collection of more agricultural data at a regional level. For phosphorus the deviations between calculated and observed loads are about 30 %, which is quite high. Therefore the database (especially for point source discharges) as well as for some model approaches especially erosion and surface runoff have to be improved.

2.2.2. MONERIS model accuracy for the case study areas (meso scale)

The following chapter presents the results from the MONERIS calculations, which were obtained in the 5 CSA in the Danube catchment in respect to the retention approaches which are included in MONERIS. A detailed discussion is included in Deliverable 1.3. In MONERIS two approaches are included to estimate the retention of nitrogen and phosphorus in the river (see chapter 4.6.1). In both the ratio between the measured loads and the calculated emissions are used, on one approach this ratio is contrasted with the specific runoff (runoff subdivided by the area), on the other approach with the hydraulic load (runoff subdivided by the water surface area of surface waters). With the increase in specific runoff /hydraulic load the retention in the river should decrease in a certain range (see Figures: Graph / Range ± 30%).

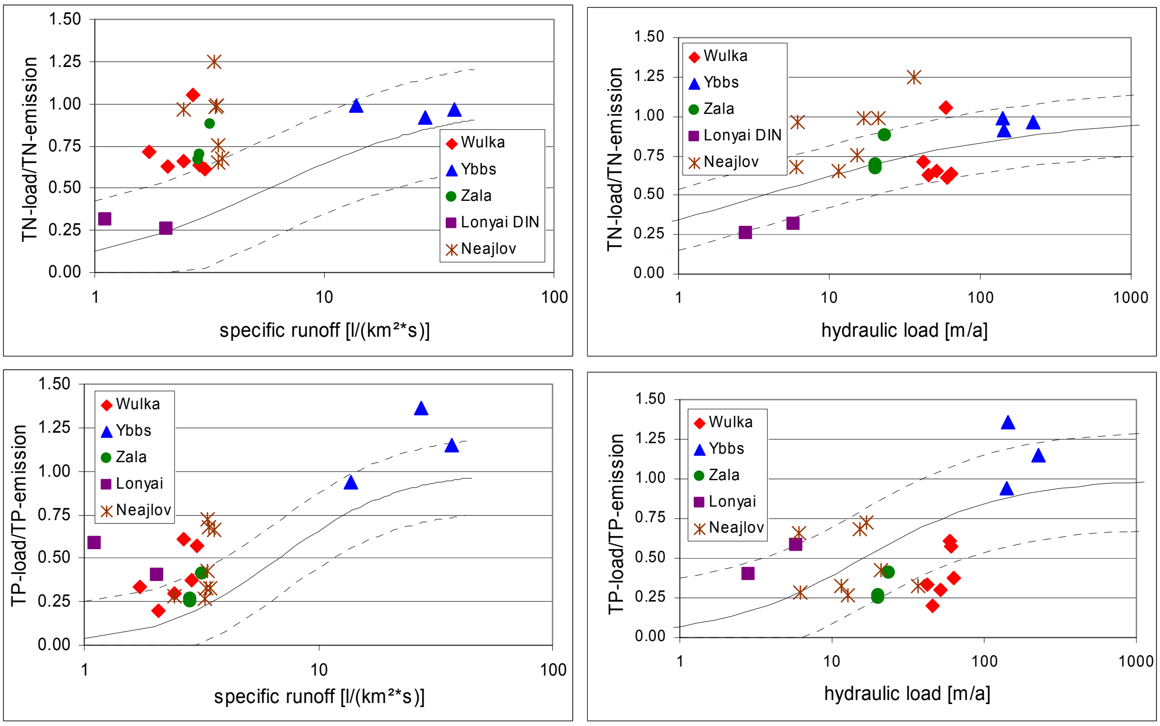


Figure 2.4: Retention of N and P in the river versus the specific runoff and the hydraulic load of the CSA.

Figure 2.4 shows the relation between the retention (relation between river load and total emissions) and the specific runoff as well as the hydraulic load for N and P. Expected retention and its ranges according to the MONERIS retention approaches are compared to the retention calculated for the different subcatchments of the case study investigations. The estimated P retention in the river for the different subcatchments does not show a good agreement with the expectations based on the retention approach with the specific runoff. Only for most of the subcatchment of the Ybbs and the Zala the retention is in a good line with the expectations. The main watershed outlet of the Ybbs catchment is out of the expected range. Emission estimates are below the measured instream loads. The reason for this is, that during the investigation period 2 major high flow events appeared, which can not be covered by the emission. Also the retention in the Lonyai catchment and in most of the Wulka and Neajlov subcatchments are not in line with the expected retention.

The N retention estimated with the approach based on the hydraulic load fits quite well with the expectations. For three subcatchments of the Neajlov point source emissions probably have been underestimated and therefore the load to emission coefficient is unrealistic high. At the Wulka in one subcatchment one point source emission only few km upstream the measuring point is influencing the load significantly. Retention figures therefore are not representative for the whole catchment. These four subcatchments have not been considered for further evaluation of data.
For the Lonyai catchment only DIN but no TN measurements were available. Including only DIN leads apparently to a higher retention.

The retention approach based on the hydraulic load shows better accordance of the P retention with the expectations compared to the approach based on the specific runoff. Most of the subcatchments of the CSA are within the range of ± 30% of the expected retention.

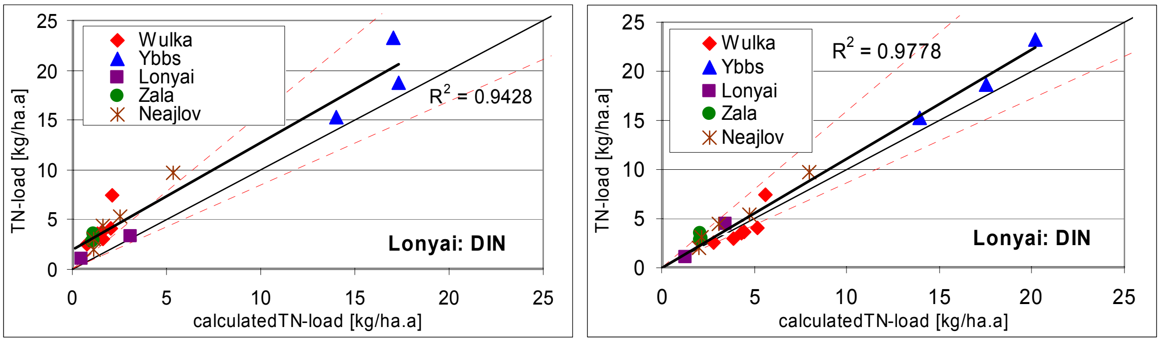


Figure 2.5: Comparison of calculated and measured nitrogen loads of different catchments with retention approach based on the specific runoff (left) and retention approach based on the hydraulic load (right)

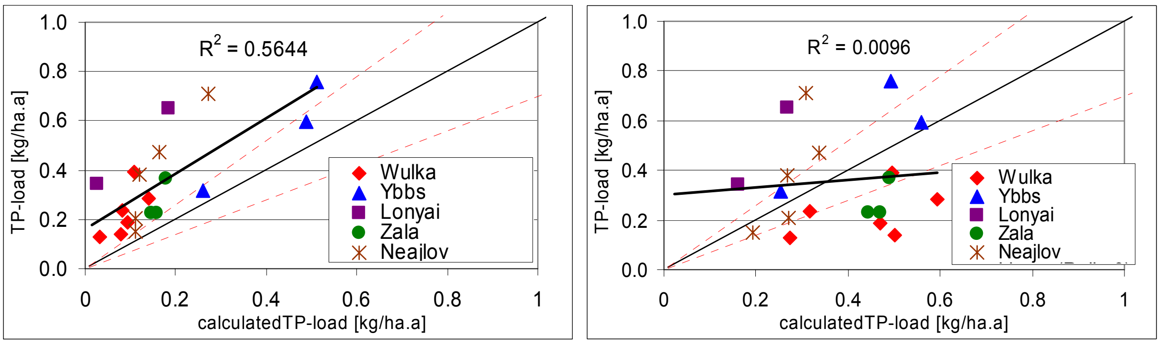


Figure 2.6: Comparison of calculated and measured phosphorus loads of different catchments with retention approach based on the specific runoff (left) and retention approach based on the hydraulic load (right).

In Figure 2.5 the calculated river loads using the two different retention approaches are compared to the river loads calculated from the measurements. The retention approach based on the hydraulic load fits well with the measurements and for most of the CSA the deviation is within the range of ±30%. The retention approach based on the specific runoff fits well for the Ybbs and the Lonyai catchment only. For the Wulka, the Zala and the Neajlov catchment the calculated river loads are smaller than the measured ones. Generally, the stability index is higher for the retention approach based on the hydraulic load. The trend line indicates a good estimation of the calculated loads using the retention approach based on the hydraulic load. Using the retention approach based on the specific runoff the calculated loads tend to be underestimated.

In regard to the phosphorus loads the retention approach based on specific runoff tends to underestimate the calculated loads (see Figure 2.6). Again, except the Ybbs catchment most of the catchments have a higher measured load than the calculated one. The retention approach based on the hydraulic load show a weak correlation to the phosphorus loads calculated from the measurements. For Ybbs and Lonyai the calculated river loads are below the calculated ones. For the Ybbs subcatchments this can be explained by an overrepresentation of high flow events during the investigations phase, for the Lonyai river by the high share of point source emissions, where the retention of phosphorus can be expected to be lower as from diffuse emissions. The calculated loads at the Wulka and the Zala are higher as the measured ones if the retention approach based on the hydraulic load is applied. There seems to be a tendency that the retention is overestimated. The calculated loads for the Neajlov catchment do not show a good correlation resulting in both over- and underestimations of the calculated P loads. The stability index is better for the retention approach based on the specific runoff. Generally, the retention approach based on the specific runoff tends to underestimate the calculated P load, whereas the retention approach based on the hydraulic load results a more scattered picture, where underestimation as well as overestimations appears.

From the results the mean deviation of the calculated loads in relation to the measured loads was calculated. All calculated mean deviations less than 30% are marked red (see Table 2.1).

Table 2.1: Mean deviation of the calculated N and P loads in relation to measured loads.

	Ybbs [%]	**Wulka** [%]	**Zala** [%]	**Lonyai** [%]	**Neajlov** [%]	**all data** [%]
N-load (specific runoff)	**14**	**60**	**65**	**31**	**54**	**47**
N-load (hydraulic load)	**9**	**22**	**31**	**13**	**18**	**16**
P-load (specific runoff)	**23**	**59**	**39**	**82**	**54**	**51**
P-load (hydraulic load)	**20**	**116**	**78**	**56**	**35**	**69**

It can be seen that for all the CSA the mean deviation was calculated for the N loads using the retention approach based on the hydraulic load are within the range of 30%. That means that using this retention approach the calculated loads are well in line with the measurements. For the calculated P loads using the retention approach based on the hydraulic load for one CSA (Ybbs) the mean deviation was less than 30%.

In general, for all the data a mean deviation within the range of 30% could be obtained for the N loads using the retention approach based on the hydraulic load. For the estimation of the P loads the mean deviation for all the data was higher than 50% using both retention approaches. This result is significantly worse than the one obtained for the different subcatchments of the Basin wide MONERIS application. A reason might be that as average the subcatchments of the case study investigation are much smaller as for the Basin wide application. Due to the more dynamic behaviour of P-loads in smaller catchments the deviation between calculations and measured loads tend to increase for smaller catchments.

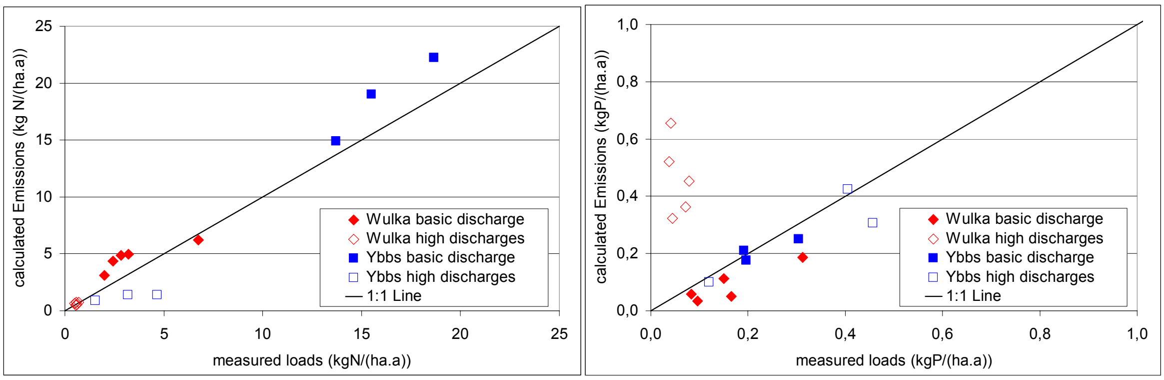


Figure 2.7: Comparison of calculated nitrogen and phosphorus emissions and measured loads for basic discharge and high discharges in the Austrian case study regions.

Above, the measured nitrogen loads of basic discharges and high discharges are plotted against calculated emissions related to basic or high discharge for different total (gross) subcatchments at the Wulka and the Ybbs. Points above the 1:1 line indicate retention during the considered conditions (emissions are higher than loads). Points below the 1:1 line indicate release of nutrients during the considered period if emission calculations and river loads are correct. For nitrogen at basic discharges retention is indicated for all subcatchments. Nitrogen emissions and river loads related to high discharges are much lower as emissions and loads related to basic discharge. For the subcatchments at the Wulka the emissions related to high discharges are almost equal to river loads related to high discharges. There is no indication for retention or release. For the subcatchments in the Ybbs loads related to high discharges are much higher than emission estimates. This would indicate release of nitrogen from the river sediments under high flow conditions. Nevertheless accuracy of data is not high enough to strongly support this indication.

For phosphorus the results are quite different. Loads related to high discharges are similar to the ones related to basic discharge. For the Ybbs emission estimates for basic and high discharges are similar to the loads under the same conditions in most of the cases. This is an indication that retention is insignificant under both conditions. Only in one case there is a week indication for release of phosphorus under high flow conditions. Totally different are the results for the Wulka. The overall retention in the river system is much more important in the Wulka catchment as in the Ybbs catchment. It is evident that at high discharges emissions in the Wulka catchments are much higher than river loads. Even if a certain uncertainty for the emission estimates as well as for subdivision of river loads must be considered the indication is strong that at the Wulka emissions at high discharge conditions (mainly erosion) is retained in the river system. At basic discharges river loads are significantly higher than emissions. Emissions at basic discharges are mainly from point source. Uncertainties are comparable low for this emission pathway. Thus this indicates strongly that part of the (particulate) phosphorus emitted and retained at high discharge is released at basic discharges and increases the river loads under this conditions.

A comparison of MONERIS model results from large scale and meso scale point up that nitrogen balancing using the model approach basing on hydraulic load is sufficient on both scales with a deviation less than 20 % in general. For phosphorus the high deviations found on large scale even increase on meso scale. Whereas on large scale a mean deviation between calculated and observed loads was estimated to be 29.7 % on meso scale the mean deviation was found to be 51 % using the approach based on specific runoff calculations. These results agree very well with the results from the model comparison (see chapter 2.1). They point out that model accuracy especially for phosphorus increases with increasing catchment size.

3. NUTRIENT SOURCES AND PATHWAYS

For evaluating the key nutrient sources and pathways on catchment scale a broad availability of data but also of process knowledge is necessary. Therefore results presented in this chapter belong to investigations made on different scales, both large scale handling the Danube catchment main sources (Del. 5.5) and more detailed data from the case study areas where a higher resolution of data and analysis give a more specific insight into the processes responsible for nutrient release or retention. Results of the case study investigations are presented in detail in Del. 1.3.

3.1. NUTRIENT SOURCES AND PATHWAYS FOR THE DANUBE CATCHMENT (LARGE SCALE)

For the present state (period 1998-2000) the model result was that a total of 756 kt/y nitrogen (N) and 68 kt/y phosphorus (P) are emitted by the different pathways into the surface waters of the Danube Basin. 82 % of N-emissions and 64 % of P-emissions come from diffuse sources. The dominant pathway are groundwater (44 %) and emissions from WWTP`s (23 %) for N- and erosion (35 %) as well as point source discharges (47 %) for P-emissions (Figure 3.1).

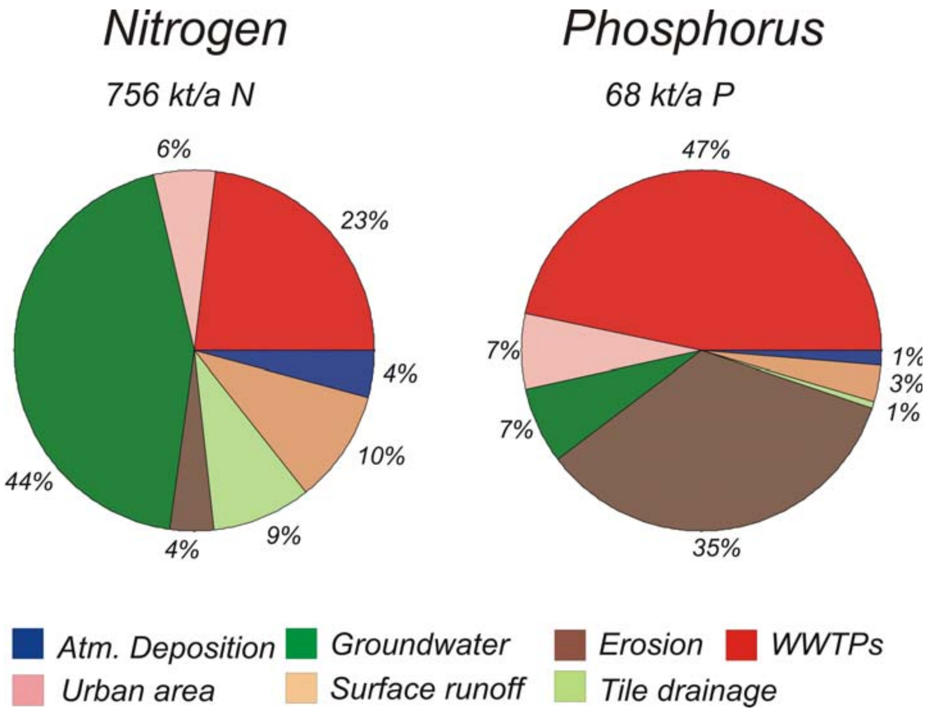
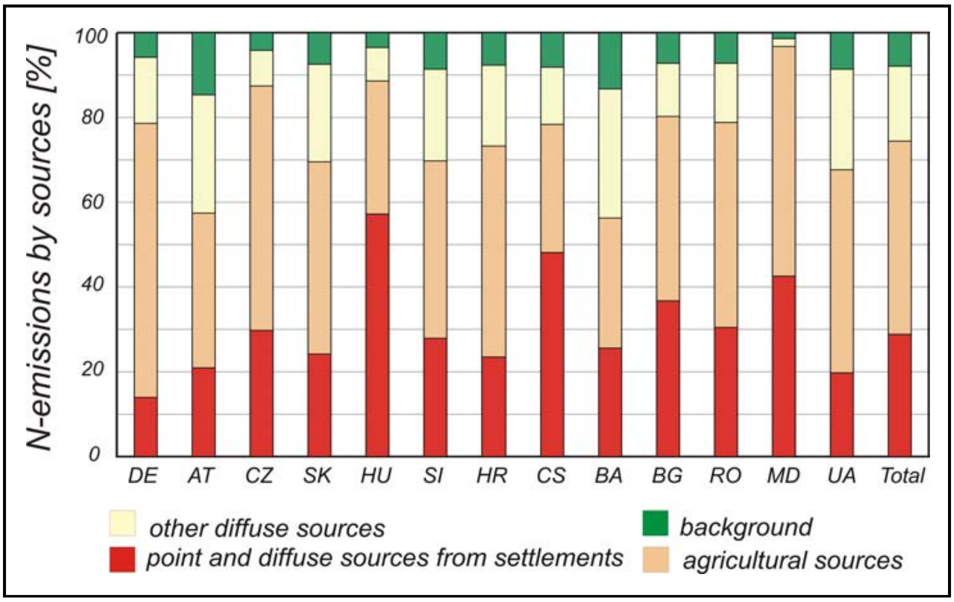


Figure 3.1: Nutrient emissions pathways for the total Danube in the time period 1998-2000.

Figure 3.2 demonstrates that the share of the different sources within the country parts of the Danube differ in a wide range. It is obviously that this differences results in different major task for possible reductions of the nutrient emissions in the countries. The high portion of the

P-emissions from urban settlements for all countries with exception of Germany and Austria indicates that a further improvement of waste water treatment would due to further significant reductions of the total P-emissions in the Danube. For nitrogen it was found out (see Figure 3.2) that agricultural emissions are for all countries with exception of Hungary, Serbia and Montenegro and Bosnia and Herzegovina the main source.



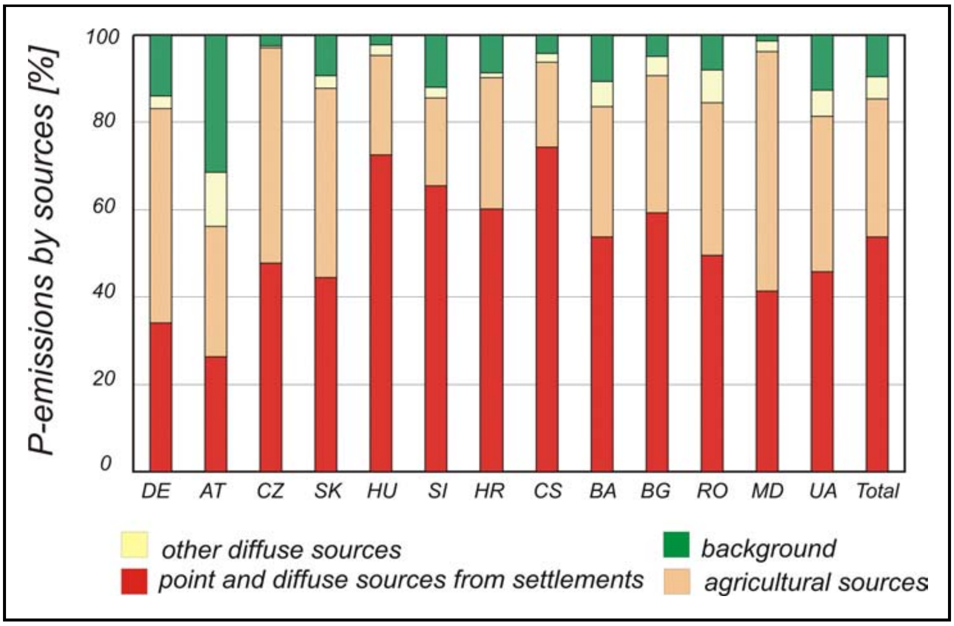


Figure 3.2: Nitrogen and Phosphorus from human sources and natural background into the Danube area of the different countries in the time period 1998-2000.

As can be concluded from a great variety of main pathways (not presented here) and the size of their shares in the Danube catchment either, nutrient balancing on catchment scale always will depend on "the case arising". This includes physical conditions in the catchment as well as anthropogenic influences. It is obvious that the variety will even increase with decreasing scale.

3.2. NUTRIENT SOURCES AND PATHWAYS FOR THE CASE STUDY AREAS (MESO SCALE)

In Table 3.1 the emissions from the case study areas are associated with the activities in the catchments and grouped to background emissions (non-influenceable emissions), diffuse emissions and emissions from point sources.

Table 3.1: N and P emissions by activities in the 5 CSA.

Nitrogen	background	agriculture	point sources + urban areas	other diffuse sources	TN
	[kg/ha*a]	[kg/ha*a]	[kg/ha*a]	[kg/ha*a]	[kg/ha*a]
Ybbs	3.30	13.30	1.00	2.80	20.4
Wulka	0.40	3.70	1.30	0.30	5.7
Zala	0.81	1.08	0.89	0.59	3.4
Lónyai	0.15	0.22	2.90	0.09	3.4
Neajlov	0.19	1.43	1.05	0.05	2.7
Phosphorus	**background**	**agriculture**	**point sources + urban areas**	**other diffuse sources**	**TP**
Ybbs	0.180	0.220	0.150		0.55
Wulka	0.030	0.480	0.120		0.63
Zala	0.030	0.690	0.080		0.80
Lónyai	0.030	0.220	0.350		0.60
Neajlov	0.006	0.364	0.105		0.48

Figure 3.3 shows the relative contribution of the different activities to the N and P emissions. In most of the case study regions agriculture is the main N emitter. For N especially in the Ybbs and the Zala catchment a high background emission was estimated which can't be influenced by human activities. Additionally to the major contributor agriculture also other diffuse sources have to be considered. These emissions are mainly due to deposition of NO_x from traffic, industry and room heating on non agricultural areas. In the Wulka catchment the background emissions are lower, but beside agriculture as main emitter also the point sources and urban areas contribute a considerable part of the total N emissions. In the Lonyai catchment due to the insufficient waste water treatment most of the N emissions come from point sources. In the Neajlov catchment most of the N emissions come from agriculture as well as from point sources and urban areas.

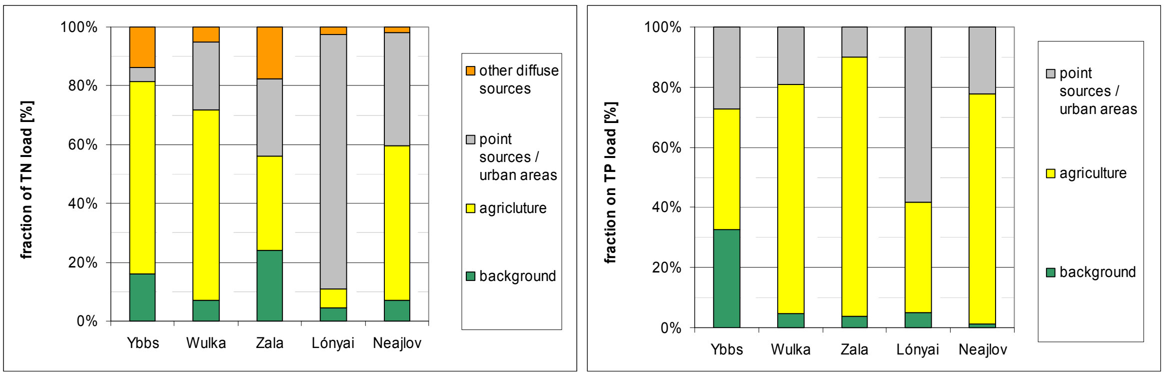


Figure 3.3: Fractions of N and P emissions by activities in the 5 case study regions.

In regard to the P emissions also the agriculture is the major source for most of the case study areas. In the Ybbs catchment, there are background emissions contributing $^1/_3$ to the total emissions. This is more than the contribution of point sources and urban areas. In the Lonyai catchment the contribution of point sources has the highest fraction on the total P emissions.

The main pathways for phosphorus emissions (see figure 3.4) in the different case study areas is erosion. Nitrogen is mainly transported by groundwater. Furthermore figure 3.4 illustrates the broad variety of share of main emission pathways in the different case study areas.

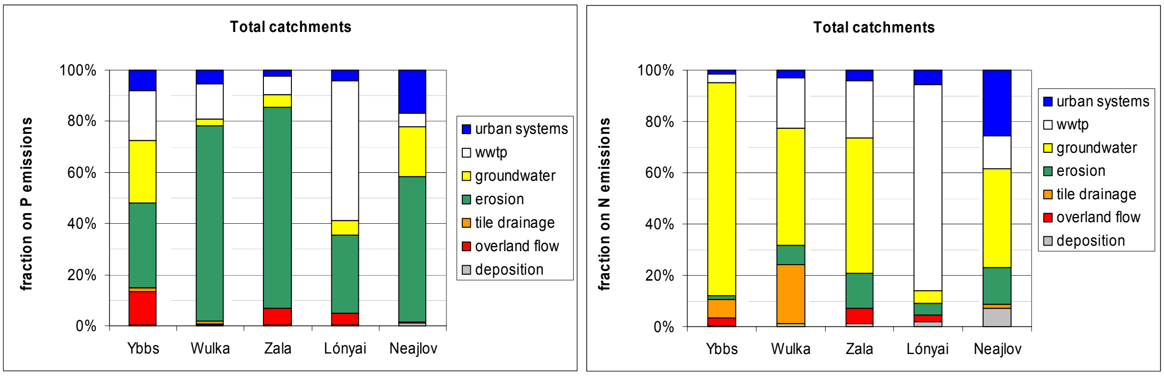


Figure 3.4: Emission pathways for nitrogen and phosphorus for all case study areas.

In the following the most important sources and pathways for nitrogen and phosphorus are discussed.

3.3. EVALUATION OF KEY PARAMETERS

The following statements derive from investigations in the case study areas.

3.3.1. Nitrogen pathways

Groundwater

In the Danube catchment groundwater is the major pathway for nitrogen emissions to the surface water. The quantity of emissions via groundwater depends (i) on the nitrogen surplus on agricultural soils, (ii) the surplus on non agricultural soils and (iii) the extent of retention/denitrification in soil and groundwater.

Ad (i) the key factors influencing the nitrogen surplus in agricultural areas are mainly of anthropogenic nature: intensity of agricultural production (mainly animal density and fertilizer application) and the optimisation (time and amount) of fertiliser application and agricultural practice.

Ad (ii) Key factors influencing surplus in areas not under agricultural use are anthropogenic as well: combustion processes mainly from traffic and industry as well as NH_3 emissions to the air from animal husbandry which lead to nitrogen inputs by deposition.

Ad (iii) denitrification in groundwater highly influences nitrogen emissions to surface waters. With nitrogen surpluses on soils between 15 and 75 kgN/(ha.a) as average values for catchments, the nitrogen emissions to surface waters vary between 1 and 25 kgN/(ha.a). Denitrification in soils and groundwater of a region may amount from 5 up to more than 50 kgN/(ha.a). That means that in some regions the ratio of the surplus reaching the surface water is fewer than 5 %. The influence of denitrification in soil and groundwater on nitrogen emissions to surface waters may be higher than the anthropogenic factors leading to nitrogen surpluses in agriculture. Quantification of denitrification in groundwater therefore is the most important value in the frame of a regional nitrogen balance for surface water emissions. The natural factors influencing denitrification in soils and groundwater are hydro-geological factors as groundwater recharge rates, flow velocities and residence time in the underground, but also carbon or pyrite availability and oxygen depletion in groundwater. Denitrification by heterotrophic and autotrophic bacteria, respectively, takes place in case of degradation of organic carbon or oxidation of pyrite (FeS) under absence of dissolved oxygen. The denitrification can either be limited by the availability of organic carbon (absence or very slow hydrolyses of particulate organic matter) or pyrite or by the competition between nitrate and dissolved oxygen as electron donor for carbon degradation (diffusion limitation). Thus, it can be expected, that denitrification rates will tend to be higher if (i) the organic carbon (or pyrite) availability is high as compared to oxygen supply (oxygen and nitrate consumption), (ii) the nitrogen surplus in soils is high (high ratio nitrate to oxygen in leakage water), (iii) amount of leakage water is low (again high ratio of organic carbon or pyrite and nitrate to oxygen in leakage water) and (iv) the residence time in groundwater is high (increasing reaction time). Factor (i) depends mainly on the geological unit and the groundwater recharge rates (amount of leakage water). In factor (ii) the nitrogen surplus and in factor (iii) the groundwater recharge rates are decisive, while factor (iv) is influenced by the hydrogeological situation as well as the amount of leakage water.

In chapter 4.1 possibilities to estimate nitrogen emissions from groundwater are presented.

Point sources

The nitrogen discharges to surface waters from point sources are highly influenced by the population density and to some extent by the industrial activity in a region. These factors determine the amount of nutrients in the waste water of a region. The head specific nitrogen emission per inhabitant is relatively constant, but may be influenced by industrial activity. The share of nitrogen that is discharged to surface water via point sources depends on the waste water infrastructure of a region. Connections to sewer systems increase the share of discharges into surface waters, while improvement of the treatment process from no treatment to mechanical treatment to biological treatment with carbon removal to biological treatment with nitrification and to biological treatment with nitrification/denitrification reduces the emissions. Possibilities to estimate emissions from point sources are documented in chapter 4.3.

Urban areas

Urban areas, as it is used as pathway in the MONERIS approach, includes three different pathways: emissions of nutrients from non sewered areas, emissions via combined sewer overflows and emissions via rain water sewers. While the emissions via rain water sewers and combined sewer overflows are usually below the calculation accuracy of total nitrogen emissions, discharges from non sewered areas may be relevant in cases, where a high portion of the population is not connected to sewer systems.

Tile drainage

As for emissions via groundwater, the nitrogen surplus on soils is an important factor influencing nitrogen emissions via tile drainage. Nevertheless, the main anthropogenic factor is the amount of tile drained areas, which significantly reduce the residence time in the underground and therefore the retention/denitrification of nitrogen in the underground. A big problem of quantification is that detailed data on the amount of drained areas are not available in most of the regions.

Overland flow

Overland flow (nutrient emissions via the surface runoff from non paved areas in dissolved form) is not in the focus of nitrogen balances as well. Specific emissions ranges between 0 and 0.6 kg N/(ha.a). The upper limit is found in regions with high deposition rates and a high share of surface runoff in the water balance as it can be found in the mountainous region of the Ybbs catchment.

Erosion

Nitrogen is transported in water mainly in soluble forms. As compared to the total emissions the content of nitrogen in relation to phosphorus is much lower. Area specific emissions are in the range of 0.1 to 0.6 kg N/(ha.a). Thus, erosion has not a significant effect in nitrogen balances and will be discussed in connection with phosphorus.

Deposition

The pathway deposition according to the MONERIS definition is restricted to the direct deposition of nutrients on the surface waters. With emissions of 0.04 – 0.20 kg N/(ha a) related to the total catchment area this pathway is irrelevant for nitrogen balances of land based catchments. It will become relevant only in regions with a high share of surface water area.

In contradiction to the direct deposition on the surface water area, deposition in general on agricultural and especially non agricultural areas is of significant importance as it contributes to the surplus on soils, which is basis for emissions via groundwater as it was already discussed at the emission pathway “groundwater”.

3.3.2. Phosphorus pathways

Erosion

The accuracy of phosphor balances of river catchments depends on the accuracy of the calculation of phosphorus inputs via soil particles by erosion. Therefore, in the daNUbs project an own workpackage (wp2) is specifically dedicated to erosion estimates. For details of key factors influencing erosion see chapter 4.2 or deliverable 2.1 and 2.2 of work package 2.

For regional nutrient balances concerning erosion the surplus on agricultural soils is an important factor in respect to phosphorus emissions via erosion. This surplus is mainly determined by the amount of fertiliser applied. In contradiction to nitrogen, for phosphorus not the surplus of a year determines the potential for the emissions but the accumulation of the surplus over the years leads to increasing phosphorus content in soils. Together with the enrichment ratio of phosphorus in eroded particles (enrichment of phosphorus in small, easier erodable particles leads to higher concentrations in eroded particles than in the soil) and the amount of soil particles discharged to the river system, finally the phosphorus content in soils influences the phosphorus emissions via erosion. As it was shown for the Austrian case studies, estimations of phosphorus content in soils based on the long term phosphorus surplus in agriculture and geogenic baseline concentrations leads to quite good average soil concentrations (Deliverable D1.3). The enrichment ratio for phosphorus can be determined by the relation between phosphorus content in soils and in suspended solids transported in the river system, if these data are available.

What should be mentioned is, that in addition to the accurate estimation of soil erosion from soils, for determination of the input into the surface water the determination of the sediment delivery ratio (ratio between eroded soil and soil particles transported into the river) is of decisive importance. Depending on catchment characteristics, the sediment delivery ratio may vary between 1 and 0.05 or even less, meaning that 100% of the eroded material reaches a specified outlet or only 5% may be detected. As the SDR is a ratio, it depends also on the amount of calculated gross erosion (which in turn depends on the soil erosion model employed). Soil erosion rates may vary in two orders of magnitude and similarly do SDR's. This means that both parameters are important drivers for the calculation of sediment loads. A calibration of erosion estimates is only possible by a comparison of the results of erosion calculation with the transported suspended solids in the river. A problem in this respect which is not solved yet is, how to distinguish between P-retention in the catchment and retention in the river system, or expressed in a other way: which part of the sediment delivery ratio describes sedimentation before sediments reach the river system and which part describes retention after sediments have reached the river system (e.g. sedimentation in the river bed or in flooded areas).

Stated before the MONERIS approach offers a practicable tool to determine the sediment delivery ratio till discharge to the river on the one hand and to estimate river retention based on an additional approach. Outcome of this approach will be discussed later on (chapter 4.4). Anyway, there is no way yet to check the plausibility of the differentiation between retention in the catchment and retention in the river.

Possibilities to estimate nutrient emission from erosion are presented in chapter 4.2.

Point sources and urban areas

For these two emission pathways the statements made for nitrogen are valid for phosphorus as well. Values for emissions from urban areas obtained in the case study regions vary between 0,02 – 0,09 kgP/(ha a) and contribute to the total emissions with less than 20 %. Even in regions with a high percentage of the population not connected to sewer systems the values are relatively low because of the high retention capacity of underground for phosphorus. P-emissions are high for points sources in those areas were a high population density coincidences with a high degree of connections to sewer systems and a low level P-removal at treatment plants.
Even if point sources only contribute to a small share to total average emissions at low flow conditions point sources can highly influence river P concentrations as has been shown in D1.3.

Groundwater

Phosphorus emissions to surface waters via groundwater may not be neglected in all cases. In cases with a high groundwater discharge or high phosphorus concentrations in groundwater the values rise up to about 0.16 kgP/(ha.a) which may be a significant contribution to the total emissions. The MONERIS model estimates phosphorus concentrations in the groundwater based on soil types. In the Austrian case study areas in three of four cases, where the data base was appropriate for comparison, calculated phosphorus concentrations in groundwater were in line with measured groundwater concentrations (see Deliverable D1.3).

Overland flow

As for nitrogen, the contribution of overland flow (nutrient emissions via the surface runoff from non paved areas in dissolved form) to the total emissions for surface waters will only be of relevance (>10 %), if the water discharge via the surface is high. Area specific emission values vary between 0.003 and 0.07 kgP/(ha.a). The MONERIS model estimates the concentrations of the surface runoff based on P-saturation in soils. It was not possible in the frame of case study investigations to check an applicability of this assumption for the case study regions.

Tile drainage and deposition

Emissions via these pathways are below 0.01 kgP/(ha.a) related to the total area and are of no importance as compared to other emissions via other pathways for the subcatchments of the case study regions. Tile drainage might become important under certain conditions (high portions of tile drained areas, high P-losses from soils).

4. APPROACHES FOR ESTIMATING NUTRIENT EMISSIONS

This chapter deals with hot spots of nutrient balancing where model accuracy is crucial for the overall result. Methodological problems are discussed and possibilities for estimating nutrient emissions are performed. Listed up this chapter will deal with

- nutrient surplus in topsoils (phosphorus and nitrogen)
- emissions via groundwater (nitrogen)
- emissions via erosion (phosphorus)
- municipal point sources (phosphorus and nitrogen)
- retention and transport in surface water (phosphorus and nitrogen)

Most of the presented approaches stem from the emission Model MONERIS. Following the Material accounting (MA) approach is presented to point up that it would be helpful for management measures if emission Model input data - often taken from statistics- would be more clearly attributed to its source. For example, when the atmospheric deposition would cause high emissions to surface waters in a further step the polluter has to be known to address possible measures.

4.1. REQUESTED ENLARGEMENT OF THE NUTRIENT MANAGEMENT DATA BASE BY THE "MATERIAL ACCOUNTING (MA)" APPROACH

Introduction

In order to derive efficient measures detailed information on the sources of emissions is required.

In general there are 2 possible strategies of reducing emissions:

- end-of-pipe technologies: e.g. sewage treatment plant
- front-end-measures: e.g. ban of P-containing washing powders, change of dietary habits, optimized agricultural practice

Usually a bunch of measures consisting of both strategies lead to the most efficient reduction programme.

The information required by the MONERIS approach in order to calculate the nutrient emissions into the groundwater and surface waters is not always sufficient to derive front-end-measures or certain information is only given implicitly. Especially information on emissions from private households, industry and agriculture are not included in detail or should be refined. This means that not the whole range of possible measures can be considered and therefore their impact not be calculated.

In the following section the processes and nutrient flows are depicted which have to be estimated in addition to the requirements of the MONERIS application in order to derive front end measures. These further investigations complete the data base obtained for the MONERIS application and increase the number of possible measures to reduce nutrient emissions into the Danube (a detailed description is annexed). The following pages give an overview on the flows of nutrients that shall be estimated in addition comprising mainly the topics "agriculture", "forestry", "waste water" and "N-emissions /deposition".

Agriculture

Agriculture is the central process for the regional nutrient management. A detailed scheme including also nutrient flows of minor importance is given in Figure 4.1 (detailed description of flows in the Annex).

It is useful to separate the process Agriculture into "farm" and "agricultural soil" to be able to clearly distinct between gaseous emissions from the farm or from the soil.

"Starting point" of the MONERIS approach to calculate emissions from diffuse sources is the *nutrient surplus* in agricultural soils - the MONERIS approach only comprises the "field balance" but not a "farm gate balance". It neglects the "output-goods" that leave the farm like meat or milk production, crops that are used for human or animal nutrition etc. as well as the "import" of "feed" into the farm. Furthermore gaseous N-emissions from the stables are not included (losses during storage lie between 15 to 20 % of the total amount of N (depending on the stable system (manure, dung, etc. and the kind of animal).

A farm gate balance offers more options for improving the regional nutrient management.

Direct discharge of manure can contribute a considerable nutrient load to surface waters but are not included in the MONERIS model. In addition manure treated in special waste water treatment plants is not included explicitly in the MONERIS model. However it can be easily implemented as a point source (this is somehow true also for direct discharges). However a clear source apportionment will not be possible.

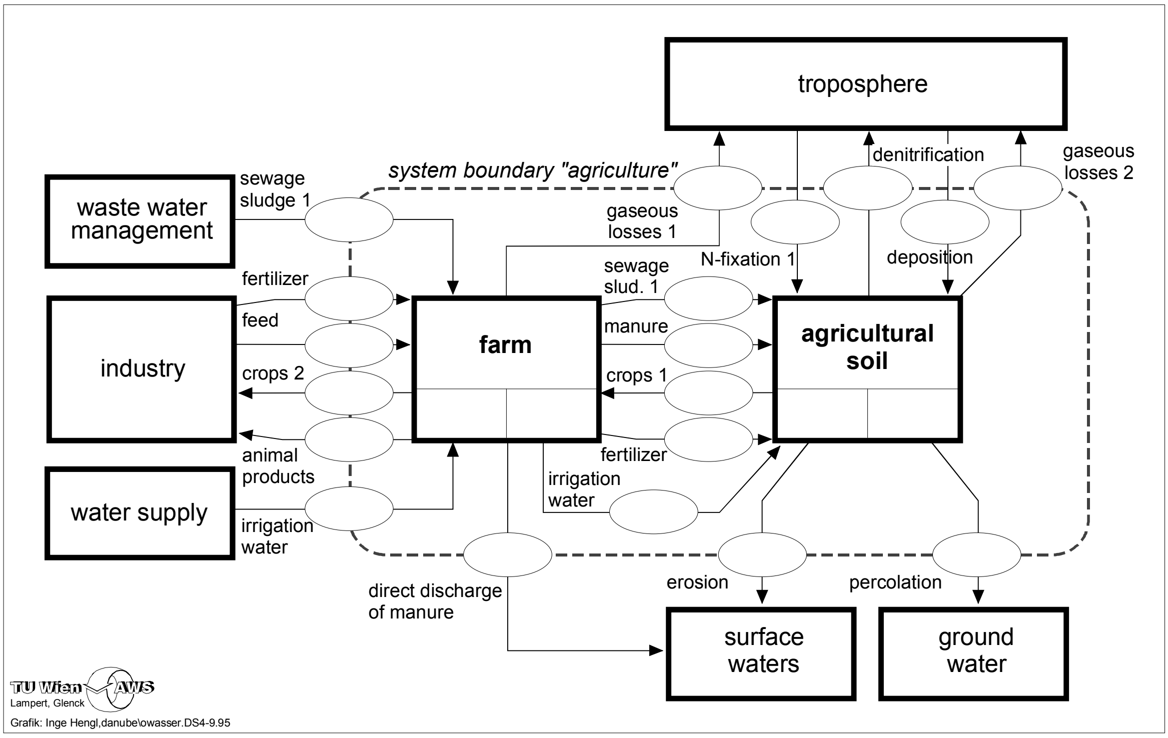


Figure 4.1: system „agriculture“.

On a regional level the (potential) nutrient flows “sewage sludge” and “irrigation water” are negligible. Sewage sludge could be important especially for P if the nutrient surpluses would be lower (or even zero in P-rich soils) and most of the waste water would be treated in plants with P-removal.

To calculate the potential impact of changes in the dietary habits it has to be estimated, how much area is used to produce one unit of animal protein and one unit of vegetarian protein.

Forestry

In the MONERIS model, it is assumed that the total amount of Nitrogen deposited on the forestry area percolates into groundwater. MONERIS does not consider the nutrient removal by the wood harvested or by denitrification nor the N-fixation by micro-organisms. As a consequence MONERIS tends to overestimate the nutrient surplus in forestry and as a consequence the nutrient emissions to groundwater.
As for “agriculture”, input and output flows as well as the change of the stock have to be calculated in order to calculate the impact of different measures.
Information about the wooded area and the exploitation of forest products should be available from regional wood harvesting statistics.

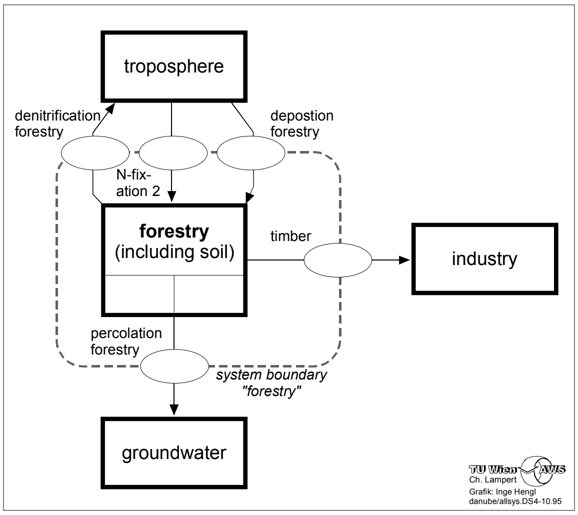


Figure 4.2: Process „Forestry".

In order to balance forestry the input flows *deposition* and *N-fixation* as well as the output flows *timber* and *denitrification* have to be calculated. The resulting N-surplus is partly a change of stock and partly is percolated to the groundwater – the part percolating to groundwater should be the "effective" input for the MONERIS calculation.

Table 4.1: Flows to be calculated in addition.

	Moneris
N-fixation	Missing
Deposition	implicitly
Timber	implicitly
Percolation	implicitly
Denitrification forestry	implicitly.

Waste water

Waste water from private households

A main source of N- and P-emissions from private households is due to the consumption of food. Dietary habits (i.e. the consumption of animal and vegetarian protein) strongly influence the need of agricultural area. To produce one unit of animal protein about 5 times more area is needed than to produce one unit of non-animal protein.

For P in addition the use of P-containing washing powder and dish-washer products can be of importance. These figures may vary in the individual countries depending on their nutrition customs and the use of phosphates in washing powder. MONERIS itself provides 3 possibilities how to calculate the P emissions. However the N-emission calculations are quite rough assuming an average N-flux of 11 gN/cap.d. This results in an underestimation of the N-emissions from inhabitants (at least for a western Europe diet).

In the MONERIS approach nutrient emissions from industrial waste water seems to be underestimated at least for P which is assumed to be 0 g P/PE.d. For Nitrogen a range between 0 - 7 g N/PE.d depending on the size of the treatment plant is given in MONERIS.

These emission estimates should be refined. The branches of industry where high turnovers of nutrients take place are: chemical-, fertiliser-, food-, pulp- and paper-industry, diaries, coking plants.

Detailed approaches to evaluate N-loads in waste water are presented in chapter 4.3

Gaseous emissions – Deposition

Deposition highly influences N-emissions from non-agricultural areas (Phare project: Nutrient balances for Danube countries: >10% of the total emissions into the Danube). MONERIS does not differentiate on the source of the deposition, therefore no related measures can be derived. Reactive N-compounds are emitted into the troposphere by agricultural activities (NH_3) as well as by combustion processes (NO_x) (traffic, energy conversion (heating, electricity, etc.)).

4.2. POSSIBILITIES FOR ESTIMATING NUTRIENT SURPLUSES IN TOPSOILS

4.2.1. Large scale approach

For calculating emissions from diffuse sources one outstanding input data is the nutrient surplus in the top soils (see figure 2.1). For all countries in the Danube basin the nutrient surplus of agricultural areas was estimated using the OECD method (OECD, 1997). The ***soil surface balance*** calculates the difference between the total quantity of nutrient inputs entering the soil and the quantity of nutrient outputs leaving the soil annually. The calculation of the soil surface balance, as defined here, is a modified version of the so-called *"gross balance"* which provides information about the complete surplus (deficit) of nutrients into the soil, water and air from an agricultural system.

Based upon the parameters (see below) and the coefficients given in the Appendix the nutrient surplus in the agricultural area was estimated by the following equations.

Nutrient Input = Fertilisers + Net Input of Manure + Other Nutrient Inputs

Nutrient Output = Total Harvested Crops + Total Forage

Nutrient Surplus = Nutrient Outputs - Nutrient Inputs

Nutrient Surplus per Hectare Agricultural Land = Nutrient Balance (tonnes of nutrient) divided by the Total Area of Agricultural Land (hectares)

The calculations for the different countries are based on the agricultural statistics and nutrient equivalents for livestock and crops. Because the result of the balance depends on the selected numbers of the nutrient equivalents, the same equivalents were used for the calculations in all countries. These harmonized nutrient equivalents are mainly based on those used in the Czech Republic and published in the database of the OECD (1999) as well as by BEHRENDT et al. (2002).

4.2.1.1 Input data processing

The estimate of the annual total quantity of ***nutrients inputs*** for the soil surface nitrogen and phosphorus balance includes the addition of:

- *inorganic or chemical nitrogen and phosphorus fertiliser*: quantity consumed by agriculture;

- *livestock manure nutrient production*: total numbers of live animals (cattle, pigs, sheep, goats, poultry, horses, and other livestock) in terms of different categories according to species (e. g. chickens, turkeys), sex, age and purpose (e. g. milk cow, beef cattle), multiplied by respective coefficients of the quantity of nitrogen and phosphorus contained in manure per animal and year (see Table 1, Appendix). The NH_3-volatilisation is considered in the calculation by a reduction of the gross livestock manure nitrogen production of 30%.
- *atmospheric deposition of nutrients:* total agricultural land area multiplied by a single coefficient of nutrient deposited per hectare. For the period 1985 to 1999 the results of the EMEP calculations for the individual countries were taken into account for the nitrogen deposition rates. For years before 1985, the same value as occurred in 1985 was used. For the phosphorus deposition a value of 0.5 kg/(ha·a) P was assumed;
- *biological nitrogen fixation:* area of harvested legume crops (e. g. field beans, soybeans, clover, alfalfa) multiplied by respective coefficients of nitrogen fixation/ha, plus t nitrogen fixation by free living soil organisms computed from the total agricultural land area multiplied by a single coefficient of nitrogen fixation/ha (see Table 2, Appendix);
- *nutrients from recycled organic matter:* quantity of sewage sludge applied to agricultural land multiplied by a single coefficient of nutrient content of sewage sludge. For the sludge a nutrient content of 1.5 kg/t N and 0.5 kg/t P was assumed;
- *nutrients contained in seeds and planting materials:* quantity of seeds and planting materials (e. g. cereals, potato tubers) multiplied by respective coefficients of nutrient content of seeds/planting materials.

The estimate of the annual total quantity of ***nutrient outputs,*** or nutrient uptake, for the soil surface nutrient balance additionally includes the following:

- *harvested crops:* quantity of harvested crop production (e. g. cereals, root crops, pulses, fruit, vegetables and industrial crops) multiplied by the respective coefficients of nutrient uptake to produce a tonne of harvested crop (see Table 3, Appendix);
- *forage crops:* quantity of forage crop production (e. g. fodder beets, hay, silage, and grass from temporary and permanent pasture) multiplied by the respective coefficients of nutrient uptake to produce a tonne of forage.

The nutrient balances were calculated for the long-term period 1950 to 1999 for Germany and the Czech Republic. Based on the FAO dataset a period from 1961 to 2000 could be considered for Austria, Hungary, Romania and Bulgaria. For the other countries the nutrient balance could be calculated only for the period 1992 to 2000, because FAO and national data were only available since the year of independence.

The full data set for the calculation of the nutrient balance was only available for 6 countries, at least for the years 1998, 1999 and 2000. Especially the production of fodder crops and/or permanent grassland is not taken into account in the published agricultural statistics of most countries. For these countries a factor between the total N production by forage and the gross nitrogen inputs by livestock manure was derived for the 6 countries where all data were available. The factor was estimated as 0.557 ($n=6$; $r^2=0.915$).

For Germany, Austria, the Czech Republic, Slovakia, Hungary and Romania a calculation of the nutrient surplus was possible also for smaller administrative units, based on the data crop and livestock statistics for regions or districts collected by the consultants. The nutrient surplus for the districts was estimated with the same method than for the countries. Because consumption of mineral fertilizer was often missing for the district level, the consumption data for districts was calculated by a distribution of the mineral fertilizer according to the procedure applied for German districts (BEHRENDT et al., 2002).

Because the database is only available for administrative units (countries, districts or municipalities) these estimations were done at first for the administrative units. Secondly the total estimated nutrient surplus for the administrative units is calculated as a specific surplus for the agricultural area of these administrative units according to the CORINE landcover or the corrected USGS landcover map. By means of these maps the specific nutrient surplus per agricultural area of CORINE is used to estimate the nutrient surplus for the different sub-catchments of the Danube.

4.2.2. Meso scale approach

Nutrient surpluses in the topsoil were calculated for the case study areas based on the MONERIS approach (chapter 4.1.1). The most important differences were the higher spatial resolution of input data (municipality level) and the application of a netto mineralization tool from the emission model "Stoffbilanz" (Gebel, 2003).

In this model it is assumed that 50 % of the applied manure is going to the stable nitrogen pool and a mineralization from the stable nitrogen pool is calculated based on the soil type and the planted crops.

4.2.2.1 Input data processing

As stated above calculations of the surplus in agricultural soils were done based on statistical and literature data on the level of municipalities for a time span of almost 40 years (1964 to 1999). In between balances were calculated in time steps of 5 to 10 years dependent on the availability of the statistical data. As input fluxes into the soils application of mineral fertiliser, manure and sewage sludge as well as biological nitrogen fixation and deposition were considered. Outputs from the agricultural soils are nutrients in harvested products and the ammonia losses from manure after application. In addition potential influence of the stable nitrogen pool on the yearly balance was estimated with the calculation of the nitrogen net mineralization. Calculated surpluses are either denitrified or leached to the groundwater.

Problematic are the data for mineral fertiliser application. In Austria till 1995 information on the selling of mineral fertiliser on the district level exist. The problem with these data is, that especially in districts with larger towns the sales statistic may not coincident with the application. Since 1995 only data for fertiliser application on the level of federal states are published. Therefore following procedure has been applied. On the one hand the trend of the development of fertiliser use on federal state level since 1995 was assumed for the district level. Average sales statistics were used for application values on regional basis. On the other hand based on crop statistics the needed amount of fertiliser was calculated on municipal level assuming an application based on the guidelines for fertiliser applications. It was assumed that the fertiliser application lies between this target value and up to 20 % more. Finally the fertiliser application based on sales statistics together with availability of manure in a municipality was compared with the amount of fertiliser needed based on guidelines and the higher value was assumed to be the total fertiliser application in the municipality.

The manure application was calculated based on a transformation of the number of animals to animal units and specific nutrient contents in manure for the animal units. The amount of sewage sludge application was calculated in a way that for those treatment plants where sewage sludge is used in agriculture it was assumed that the sludge was equally distributed on the agricultural areas of the municipalities discharging their waste water to the treatment plant. For the biological nitrogen fixation it was assumed that the nitrogen in the harvested products from legumes stems from fixation. In respect to deposition data from measurements of wet deposition of stations in the region or close to the region were taken. For the dry deposition estimations from literature were used.

The nutrient content in harvested products was calculated based on literature data of nutrients in harvested products, the land use statistics (area used for production of certain crops) on municipality level and the statistics on crop yields which is available on district level. Ammonia losses from applied manure again were estimated based on specific values for animal units.

4.3. POSSIBILITIES FOR ESTIMATING NUTRIENT EMISSIONS FROM GROUNDWATER

4.3.1. Large scale approach

As stated in chapter 3 groundwater is the main pathway for nitrogen emissions to the surface water in the Danube catchment. For this reason the methodology used in the MONERIS approach to calculate nutrient emissions via groundwater is presented below in detail.

The nutrient inputs by groundwater are calculated from the product of the groundwater outflow and the groundwater nutrient concentration and include the natural interflow and the base flow. This is caused by the absence of methods to calculate the natural interflow separately. Figure 4.1 shows a scheme for the calculation of nitrogen emissions via groundwater.

The groundwater flow was calculated for each basin from the difference of the observed runoff at a monitoring station and the estimated sum of the other discharge components (drain flow, surface runoff, storm water runoff from paved urban areas and atmospheric input flow):

$$Q_{GW} = Q - Q_{DR} - Q_{RO} - Q_{URB} - Q_{AD} \qquad (1)$$

with:

Q_{GW} = base flow and natural interflow [m³/s]
Q = average runoff [m³/s]
Q_{DR} = tile drainage flow [m³/s]
Q_{RO} = surface runoff from non-paved areas [m³/s]
Q_{AD} = atmospheric input flow [m³/s].

Figure 4.3 illustrates the MONERIS approach for calculating nitrogen emissions by groundwater.

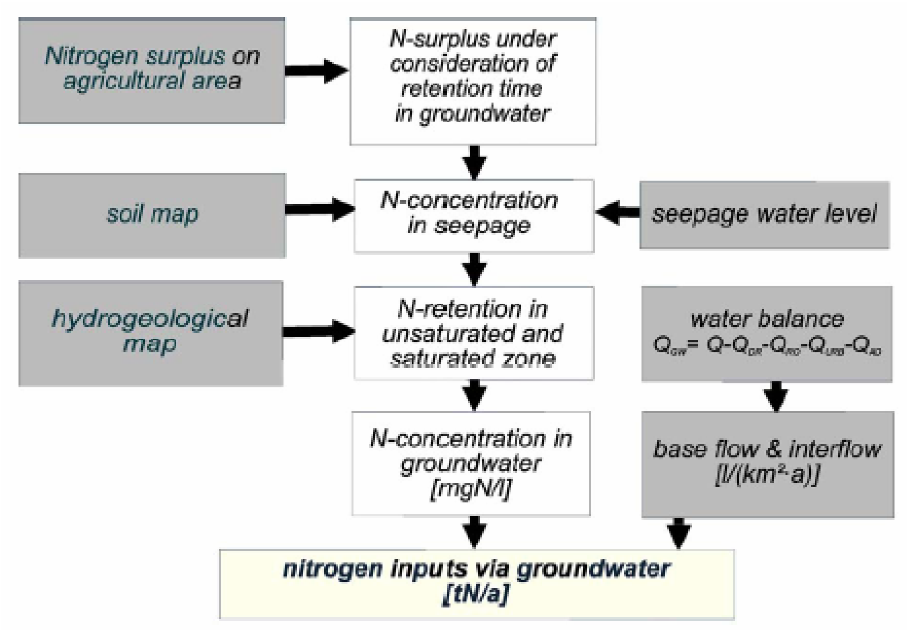


Figure 4.3: Nitrogen emissions via groundwater.

For the calculation of total phosphorus concentrations in groundwater it was determined that in accordance with Equations 2 and 3, nitrogen concentrations in groundwater are less than 5% of those in seeping water and the TP-concentrations in groundwater are 2.5 times greater than the SRP-concentrations:

$$C_{GW_{TP}} = 2.5 \cdot C_{GW_{SRP}} \quad \text{if } C_{GW_N} \leq 0.15 \cdot C_{LW_N}$$

$$C_{GW_{TP}} = C_{GW_{SRP}} \quad \text{if } C_{GW_N} > 0.15 \cdot C_{LW_N} \qquad (2/3)$$

with:

C_{GW_N} = nitrogen concentration in groundwater [g/m³],
C_{SW_N} = nitrogen concentration in seeping water [g/m³],
$C_{GW_{TP}}$ = TP-concentration in groundwater [g/m³] and
$C_{GW_{SRP}}$ = SRP-concentration in groundwater [g/m³].

The N-concentrations in the groundwater were also calculated from the potential nitrate concentration in the soil. The residence time of water and substances on their way from the root zone to the groundwater has to be taken into account for the groundwater pathway. The reasons are firstly that the level of losses (denitrification) can be time dependent, and secondly that the nitrogen surplus of agricultural land is also changing over time such that the nitrogen in groundwater flowing into surface waters is related to the N-surpluses in the past rather than the present.

A comparison between the regionalized residence times estimated for the Elbe catchment and its tributaries with the WEKU model KUNKEL & WENDLAND (1999) and the long term level of precipitation in these regions indicates that the residence time in the groundwater is dependent on the level of seeping water. Therefore it was assumed that the residence time of groundwater varies in a range between 5 and 50 years and the mean residence time of each sub catchment was estimated from a relation shown in the following equation (4):

$$\tau_{RES} = \frac{3000}{LW} \tag{4}$$

with:

τ_{RES} = mean residence time for the natural subsurface flow [a].
LW = seeping water quantity [l/(m²·a)].

This residence time was used to calculate the mean nitrogen surplus on agricultural area of each sub catchment as an average of the previous years between the investigated period (t_0:1998-2000) and the period t_0-τ_{RES}. These calculations were made for five year periods only. It was assumed that the residence time is not longer than 50 years also for such sub catchments where the equation 4 gives higher values.

Based on these results the nitrogen surpluses for the different basins were corrected according to the following formula (5):

$$N_{TSUR} = \frac{N_{SUR} \cdot A_{AG} \cdot CLS + N_{DEP} \cdot (A_{EZG} - A_{LN} - A_W - A_{IMP} - A_M)}{A_{CA} - A_W - A_{IMP} - A_M} \tag{5}$$

with

N_{TSUR} = total nitrogen surplus [kg/ha],
N_{SUR} = nitrogen surplus of agricultural areas [kg/ha],
CLS = correction factor for the long-term changes in surpluses,
N_{DEP} = atmospheric nitrogen deposition [kg/ha],
A_{CA} = catchment area [ha],
A_{AG} = agricultural area [ha],
A_W = total water surface area [ha],
A_{IMP} = impervious urban area [ha] and
A_M = mountain area [ha].

The N-surpluses thus estimated are used for the calculation of the overall potential nitrate concentrations in seeping waters for the areas contributing to base flow. For this, the first steps of the approach of FREDE & DABBERT (1998) are used. A condition for this is that the net-mineralization and immobilisation are negligible. Furthermore, it is assumed that there is no denitrification in the root-zone. Then, the following applies (6):

$$C_{LWPOT_{NO3-N}} = \frac{N_{TSUR} \cdot 100}{LW} \tag{6}$$

with

$C_{LWPOT_{NO3-N}}$ = potential nitrate concentration in seeping water for the total area with base flow [g/m³ N],
LW = seeping water quantity [l/(m²·a)].

The seeping water quantity (LW) is calculated from the water balance (see Equation 1) for each sub catchment. The nitrogen retention (mainly denitrification) in the soil, unsaturated zone and in the groundwater is calculated from the comparison of the regionalized groundwater concentrations of nitrate and the potential nitrate concentration in seeping water. This comparison was carried out for the whole area of Germany and it was found that the nitrogen retention is dependent on the level of infiltration water and the hydrogeological conditions.

The nitrate concentrations in groundwater can than be calculated from the nitrate concentrations in seeping water whilst taking account of the retention within the soil which depends on the hydrogeological rock types according to Equation 7 from BEHRENDT et al. (2000).

$$C_{GW_{NO3-N}} = \left(\sum_{i=1}^{4} \frac{1}{1 + k_{1i} \cdot LW^{k2i}} \cdot \frac{A_{HRT\,i}}{A_{CA}} \right) \cdot C^{b}_{LWPOT_{NO3-N}} \tag{7}$$

with:

$C_{GWNO3-N}$	= nitrate concentration in groundwater [g/m³ N],
b	= model coefficient for denitrification (0.627),
k_1 and k_2	= model coefficients
A_{HRT}	= area of different hydrogeologically rock types [km²].

At the end the nutrient emissions via groundwater are estimated from the product of the regionalized nutrient concentrations and the groundwater flow of the basins:

$$EGW_{N,P} = a \cdot Q_{GW} \cdot C_{GW_{N,P}} \tag{8}$$

with

$EGW_{N.P}$	= nutrient emissions via groundwater [t/a] and
a	= unit conversion factor.

The necessary data for nutrient emission calculations via groundwater are presented in chapter 5. Following a short presentation of results and a simple approach to check calculated nitrogen emissions via groundwater is given.

4.3.1.1 Approach to verify N concentrations in groundwater calculated by MONERIS

Nitrogen emissions in the Danube catchment refer to emissions from groundwater to a large extent and spatial distribution is rather heterogeneous (retention of more than 95 % of the total N-pool in areas with low rates of percolating water contrasted by nitrogen retention < 50 % in the upper Danube, as well as in the upper Drava and Sava). These results are especially important for the implementation of further measures to reduce the nitrogen concentrations in the Danube River and the load of Danube into the Black Sea. Therefore an evaluation is necessary using independent data that reflect the situation for groundwater N-emissions into the Danube. This can be done as proposed by Behrendt et al. (2000) who suggested that if a large number of observed concentrations of nitrate in groundwater wells was available this data could then be regionalised.

Because such data could not be collected within this study a comparison between the measured and the calculated nitrogen concentrations was carried out based on an indicator derived directly from the data of water quality monitoring of the different rivers in the Danube. As shown by Behrendt et al. (2003) such an indicator can be the mean concentration of nitrate in the rivers at low flow conditions and at low temperatures.

A further precondition is that the proportion of point sources to the total N-emissions in the river should be low. During theses conditions the flow in the river is dominated by the groundwater inputs and the denitrification within the water is assumed to be low because of low temperatures. Because the nitrogen input by groundwater is mostly nitrate this mean concentration levels can be used as a comparison for the calculated N-concentration of the groundwater emissions into surface waters. For this procedure monitoring data over a longer period (3 to 5 years) have to be used to establish a mean concentration, as the analysis of only a small amount of data per country can be distorted by the large number of countries involved.

Figure 4.4 shows the comparison between the observed nitrate concentration at low flow conditions and low temperature for the different water quality monitoring stations in the smaller rivers of the Danube, and the nitrogen concentration of the groundwater emissions calculated with MONERIS for the river catchments upstream of these monitoring stations. The comparison was carried out for all catchments with a proportion of point discharges to the total nitrogen emissions lower than 50 % and 20 %. Additionally to the procedure derived for the Odra, all rivers strongly influenced by snow melt in the high altitudes of the Alps were removed from the analysis.

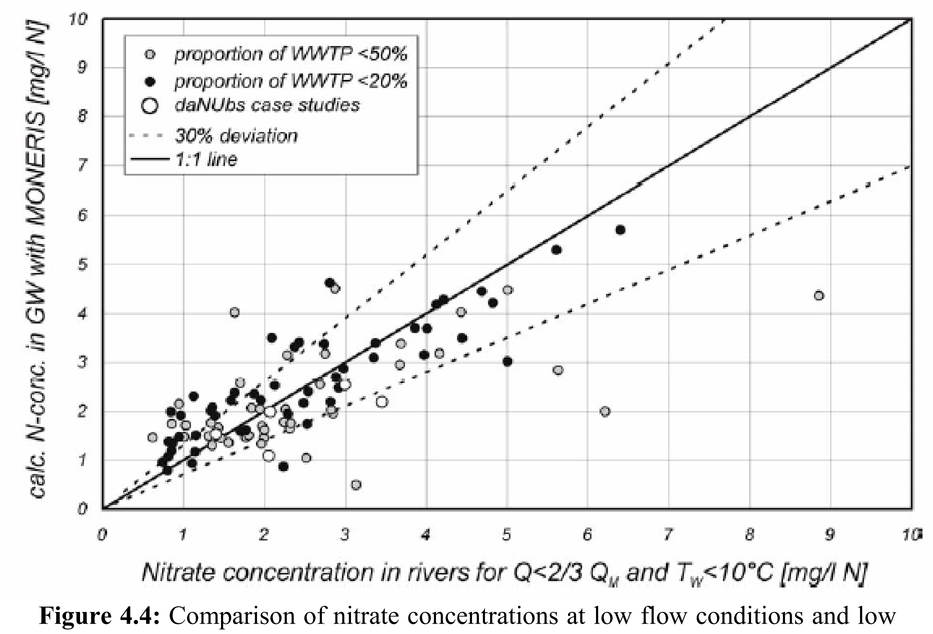


Figure 4.4: Comparison of nitrate concentrations at low flow conditions and low temperature for different monitoring stations and the calculated nitrogen concentrations in groundwater with the model MONERIS.

The similarity between the observed and calculated concentration is increasing for catchments with a share point source discharge lower than 20%. The mean deviation is reduced to 24 %, and only for 4 of the remaining 51 catchments was the deviation larger than 50%.

The calculated N-concentrations in the groundwater is strongly dependent on the accuracy of the input data, especially nitrogen surplus, within the catchment. Consequently the mean deviation between the observed and the calculated N-concentrations of groundwater is smaller for the parts of the Danube for which regionalised data of N surplus were available (23 % for Danube upstream of Hungarian-Yugoslavian border). This illustrates the need for a better resolution of the statistical data, especially for agriculture. Data for the national level are not sufficient to explain the differences of the nitrogen concentrations in groundwater, because this data does not reflect the differences of the intensity of agriculture within the country.

If these data are available for the whole Danube basin the model can be changed or calibrated to reduce the deviation between observed and calculated concentrations. Further, it would be very useful if more of the results of measurements in the smaller rivers of the Danube basin were available. This is also important because the nitrate concentrations in rivers at low flow and in winter time can be helpful in indicating possible long term changes of the N-inputs via groundwater.

4.3.2. Meso scale approach

The most important pathway for nitrogen emissions to surface waters is the groundwater. Quantitative assessment is based on the surplus on soils and on the denitrification in the soil and subsurface passage. Surplus on soils is derived based on an agricultural balance (field balance) and on deposition on non agricultural areas. The consideration of net-mineralization in agricultural soils (release of nitrogen form the organic nitrogen stock in soils) turned to be important in the cases of the Hungarian and Romanian case study areas, where the surplus suddenly dropped to close to zero at the beginning of the 90ies.

As shown before the MONERIS approach does not explicitly consider net-mineralization but covers it to some extent by using long term averages for nitrogen surpluses in soils as basis for the calculation of nitrogen emissions via groundwater.

Even more important for assessment of nitrogen emissions via groundwater as the surplus in soils is the quantification of the retention (denitrification) of nitrogen in soils and groundwater. In respect to emissions to surface water in comparison between regions the denitrification in groundwater may overrule the influence of the surplus in soils. The MONERIS retention approach works well on subcatchment level, but it shows a high sensitivity on the categorisation of geological units (consolidated-impermeable to unconsolidated-deep groundwater). Appropriate information on geology and categorisation according the MONERIS definitions therefore is prerequisite for a successful application.

For more detailed assessment of areas mainly contributing to nitrogen emissions within a subcatchment the MONERIS model is not applicable. More detailed approaches based on flowtime of groundwater and time dependency of denitrification have to be used. Based on a first application of an approach for two regions in the Austrian case study areas, a first differentiation of areas which contribute to the nitrogen emissions to surface waters and such which do not was possible (Figure 4.5).

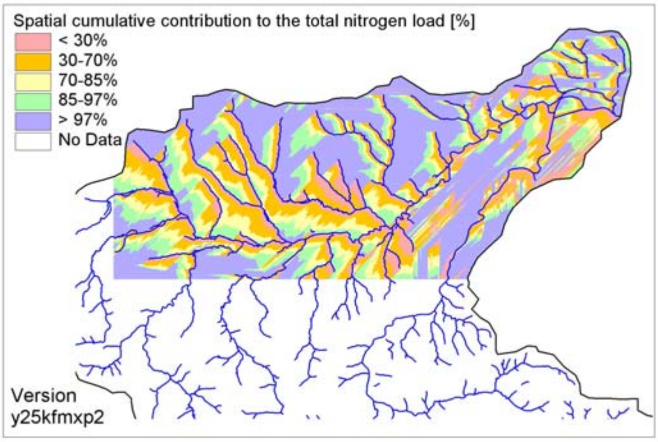


Figure 4.5: Spatial distribution of the areas with their cumulative contribution to the total nitrogen emissions via groundwater.

Even if the quantitative assessment has high uncertainties yet, the qualitative statement is possible, that in respect to discharges to surface waters in a catchment there are restricted areas with low flow time of groundwater to surface waters (low residence times) and which significantly contribute to nitrogen discharges to the surface waters. Other areas with long flow times of groundwater have negligible influence on river discharges of nitrogen.

4.4. POSSIBILITIES FOR ESTIMATING NUTRIENT EMISSIONS FROM EROSION

On catchment scale erosion often is the most important pathway for phosphorus emissions to the river. Therefore following an overview is provided concerning the needs and problems arising in regard to soil erosion modelling. However, until now there seems to be no proper way to estimate soil erosion on large scales and even on meso scale erosion calculation seems to be an unsolved problem yet.

As soil erosion is highly variable in the spatial as well as in the temporal domain, processes governing soil loss are different, too. On large watershed scale no direct measurements of the spatial distribution of soil erosion can be obtained (available only on plot and on small watershed scale). Furthermore the calculation of erosion on catchment scale by erosion models depends on a multitude of parameters (e.g. soil texture, total P content of the top soil, precipitation, landuse, surface runoff and slope) and accuracy of this input data decrease rapidly at larger scales. Due to these problems absolute numbers of soil loss at large and even on meso scales cannot be obtained with a sufficient degree of confidence. Following a short overview is given about the scale problem of erosion calculation (Strauss & Klaghofer, 2003):

Plot scale

At the spatial extension of a research plot detailed knowledge about factors which control soil loss is commonly available. Therefore work at this scale can be used to study interactions between soil loss and process factors, and to test basic assumptions used by different soil erosion models. Results available at this scale include those of research plots and rainfall simulation experiments.

Small watershed scale

At the small watershed scale, availability of data decreases rapidly. In most cases it is neither possible to measure the spatial pattern for each input parameter needed by particular erosion models, nor carrying out exact measurements of the spatial extension of soil loss within a small watershed. Instead, the necessary data input information is often obtained from maps. One example is the Austrian soil map at the scale of 1.25.000. Soils are classified according to their genetic status and genesis as usual in many other soil maps. Quantitative information on soil properties for one soil type is given only for one "representative" spot. As shown in Murer & Strauss (2003) even the same soil type can show significant heterogeneities concerning grain size classes. However, information about the extent of variance for particular soil units is not available at presence.

In addition to less detailed input data, there is a change of processes which control runoff and erosion at the small watershed scale (Lane et al., 1997). Due to the increasing amount of available runoff water the importance of linear erosion features increases. Poesen et al., (1996) and Auzet et al., (1995) evaluate their importance. Takken et al., (2001) describe the influence of tillage direction on soil erosion. Consequently, the connection between single fields along a hydrological flow path and the way field management is practised may completely override erosion processes as conceptualised in models such as the original USLE. A convenient way of obtaining information about spatial distribution and extent of soil loss within small watersheds is by mapping visible linear erosion features after rainfall events.

Large watershed scale

At the large watershed scale no direct measurements of the spatial distribution of soil erosion can be obtained but sediment production of such areas is usually measured at defined outlets. The quality of available data decreases again.

Erosion modelling approaches

Nevertheless, on the basis of a common data set, empirical models such as USLE (= Universal Soil Loss Equation), MUSLE (= Modified Universal Soil Loss Equation) or **MMF** (Morgan-Morgan-Finney model) may be used conceptually to obtain a relative ranking of soil loss risk (Strauss et al., 2003).

USLE

The USLE can be used to estimate soil loss with emphasis on sheet and rill erosion. It does not take sediment deposition into account. The equation can be expressed as follows:

$$A = R*K*L*S*C*P \qquad (9)$$

Where:

A = computed soil loss per unit area
R = the rainfall and runoff factor
K = the soil erodibility factor
L = the slope - length factor
S = the slope steepness factor
C = the cover and management factor
P = the erosion-control practice factor

Table 4.2: Input parameters for application of the USLE.

Factor	Input Parameter	Unit
K_{USLE}	Organic carbon content of upper soil layer	%
	Soil permeability	-
	Percentage of soil particle size < 50 µm	%
	Percentage of soil particle size < 125 µm	%
	Percentage of soil particle size < 2 µm	%
	Soil structure code	-
C_{USLE}	Soil cover for different periods throughout the year	-
	Relative contribution of rainfall energy for different periods of the year	-
P_{USLE}	Table values	-
LS_{USLE}	Slope	(m/m)
	Length of erosive hill slope	m

MUSLE

The MUSLE is a modified version of the well known USLE (Wischmeier and Smith, 1978). The main difference compared to the USLE is the replacement of the so called rainfall factor with a direct estimate of surface runoff and peak runoff rate. In addition the temporal resolution of the calculations is daily, whereas the USLE gives long term annual means. The authors claim that the introduction of direct estimates for surface runoff allows the extension of the original USLE to calculate sediment deposition.

The equation of the MUSLE is expressed as:

$$SED = 11.8 * (Q_{surf} * q_{peak} * Area_{hru})^{0.56} * K_{USLE} * C_{USLE} * P_{USLE} * LS_{USLE} * CFRG \qquad (10)$$

with:

- SED = sediment loss (metric t)
- Q_{surf} = surface runoff (mm/ha)
- q_{peak} = maximum surface runoff ($m^3 s^{-1}$)
- $Area_{hru}$ = Area of a hydrological response unit (km^2)
- K_{USLE} = K-Factor (soil erodibility factor) of the USLE ($t\ h\ ha^{-1}\ N^{-1}$)
- C_{USLE} = C-Factor (crop management factor) of the USLE (dimensionless)
- P_{USLE} = P-Factor (erosion control-practice factor) of the USLE (dimensionless)
- LS_{USLE} = LS-Factor (slope length and gradient factor) of the USLE (dimensionless)
- CFRG = coarse fragment factor of the USLE (%)

A set of additional variables is necessary to calculate these factors.

Table 4.2 gives an overview about these additional input parameters and their units for application of the MUSLE.

Table 4.3: Additional input parameters for application of the MUSLE.

Factor	Additional Input Parameter	Unit
Q_{surf}	Daily rainfall	mm
	Hydrologic soil group	-
	Soil water content at saturation	mm
	Soil water content at field capacity	mm
	Usable soil water content of soil profile	mm
	Land use	-
q_{peak}	Time of concentration	h
	Extreme amount of 0.5 h rainfall for a given month	mm
	Average amount of 0.5 h rainfall for a given month	mm
CFRG	Rock content of upper soil layer	%

Morgan-Morgan-Finney model (MMF Model)

The model separates the erosion process into two processes, the water phase and the sediment phase. The eroded sediment is compared to the transport capacity of the runoff and the smaller value is taken as mean soil loss/year. A detailed characterization is given in Morgan et al., 1984 and Morgan (2001).

Table 4.4: Input parameters for application of the MMF- Model.

Factor	Parameter	Unit
Rainfall	Annual or mean rainfall	mm
	Number of rain days/year	-
Soil	Soil moisture content at field capacity or 1/3 bar tension	%w/%w
	Bulk density of the top soil layer	Mg/m3
	Effective hydrological depth of soil	m
	Soil detachability index	g/J
	Cohesion of the surface soil	kPa
Landform	Slope steepness	degree
Land cover	Proportion of rainfall intercepted by vegetation	%/100
	Ratio of actual to potential evapotranspiration	%
	Ground cover	%
C_{USLE}	Soil cover for different periods throughout the year	-
	Relative contribution of rainfall energy for different periods of the year	-
	Plant height	m

Enrichment ratio (ER)

The enrichment ratio for phosphorus can be determined as the relation between phosphorus content in soils and in suspended solids transported in the river system, if this data are available. More information about the Enrichment ratio is given in chapters 4.4.1 and 4.4.2.

Soil Delivery ratio (SDR)

An unknown fraction of eroded soil passes through from the spot where erosion occurs until it reaches a channel system where it also may get deposited. As the transportation and deposition process from the erosion spot until the point of interest (usually a predefined watershed outlet) cannot be simulated any more, sediment yields are quantified using an empirical function, the sediment delivery ratio (SDR), expressed as the percentage of the gross soil erosion by water that is delivered to a particular point in the drainage system. SDR is sometimes referred as a transmission coefficient. It is computed as the ratio of sediment yield at the watershed outlet (point of interest) to gross erosion in the entire watershed (D 2.2). Depending on catchment characteristics, the sediment delivery ratio may vary between 1 and 0.05 or even less, meaning that 100% of the eroded material reaches a specified outlet or only 5% may be detected. As the SDR is a ratio, it depends also on the amount of calculated gross erosion (which in turn depends on the soil erosion model employed). Soil erosion rates may vary in two orders of magnitude and similarly do SDR's. This means that both parameters are important drivers for the calculation of sediment loads.

The SDR can be affected by a number of factors including sediment source, texture, nearness to the main stream, channel density, basin area, slope, length, landuse/land cover and rainfall-runoff factors. The relationship established for sediment delivery ratio and drainage area is known as the SDR curve. Coarser texture sediment and sediment from sheet and rill erosion have more chances to be deposited or to be trapped, compared to fine sediment and sediment from channel erosion. A small watershed with a higher channel density has a higher SDR compared to a large watershed with a low channel density. A watershed with steep slopes has a higher SDR than a watershed with flat and wide valleys. In order to estimate SDRs, the size of the area of interest should also be defined. In general, the larger the area size, the lower the SDR. However, there is no precise procedure to estimate SDR. In terms of the definition of sediment delivery ratio, the expression for computing sediment delivery ratio can be written as follows (Ouyang et al., 1997):

$$SDR = SY/E \tag{11}$$

with:

SDR = the sediment delivery ratio
SY = the sediment yield
E = the gross erosion per unit area above a measuring point

The SDR varies with the drainage area, slope, relief-length ratio, runoff-rainfall factors, land use, land cover and sediment particle size, etc. Empirical equations relating SDR with one or more factors reviewed in Ouyang et al., (1997) are tools to estimate SDR.

4.4.1. Large scale approach

Figure 4.6 shows the procedure for estimating nutrient inputs by erosion based on the soil loss rate, the sediment delivery ratio and the enrichment ratio of nutrients. The mean soil loss in each subcatchment is calculated with the help of the GIS. The sediment delivery ratios for the sub catchments are determined according to Equation 12 (BEHRENDT et al. 2000):

$$SDR = 0.012 * (SL_{CA} - 0.25)^{0.3} * A_{AR}^{\ 1.5} \tag{12}$$

with SDR = sediment delivery ratio [%],
SL_{CA} = mean slope from USGS-DEM [%] and
A_{AR} = area of arable land from CLC [%].

The sediment input due to erosion for the river basins is then calculated according to Eq. 13:

$$SDR = SOL * SED \quad (13)$$

with SED = sediment input [t/a] and
SOL = soil loss [t/a].

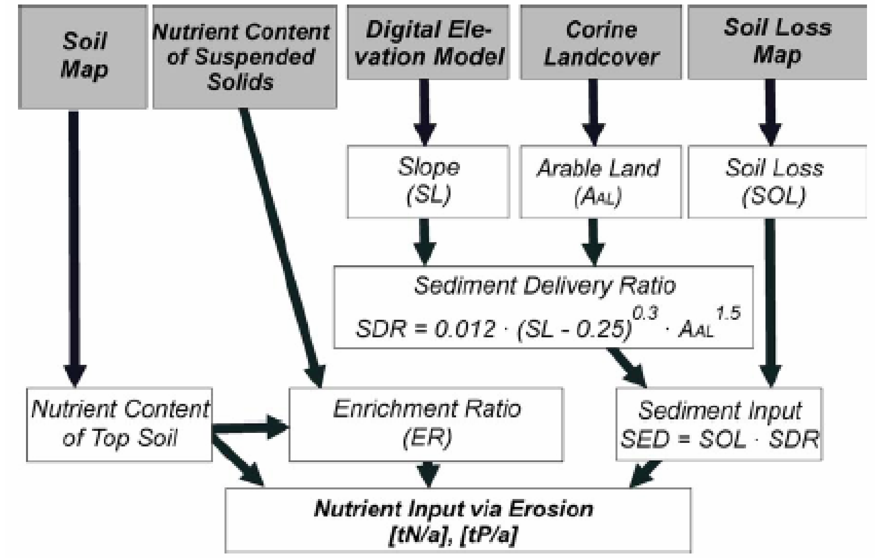


Figure 4.6: Nutrient emissions via erosion.

The enrichment ratio is calculated according to the equations from BEHRENDT et al. (2000):

$$ER_P = 18 \cdot \left(\frac{SOL}{A}\right)^{-0.47} \quad (14)$$

$$ER_N = 7.7 \cdot \left(\frac{SOL}{A}\right)^{-0.47} \quad (15)$$

with $ER_{N,P}$ = enrichment ratio for nitrogen and phosphorus.

The nutrient inputs by erosion were finally calculated as the product of the nutrient content of soil, the enrichment ratio and the sediment input:

$$EER_P = a * P_{SOIL} * ER_P * SED \quad (16)$$

$$EER_N = a * N_{SOIL} * ER_N * SED \quad (17)$$

with $EER_{N,P}$ = nutrient input via erosion [t/a] and
a = unit conversion factor.

4.4.2. Meso scale approach

In work package 2 different erosion models were tested for meso scale erosion and phosphorus losses estimates. In a second step methodological approaches to improve basic input data availability for erosion estimates on meso scale are presented. Chapter 6.3 provides an overview of available data and demanded data for adequate erosion estimates.

Model validation

To prove the validity of the assumptions within MUSLE and MMF measured data on rainfall simulation plots were compared to results obtained applying both models.

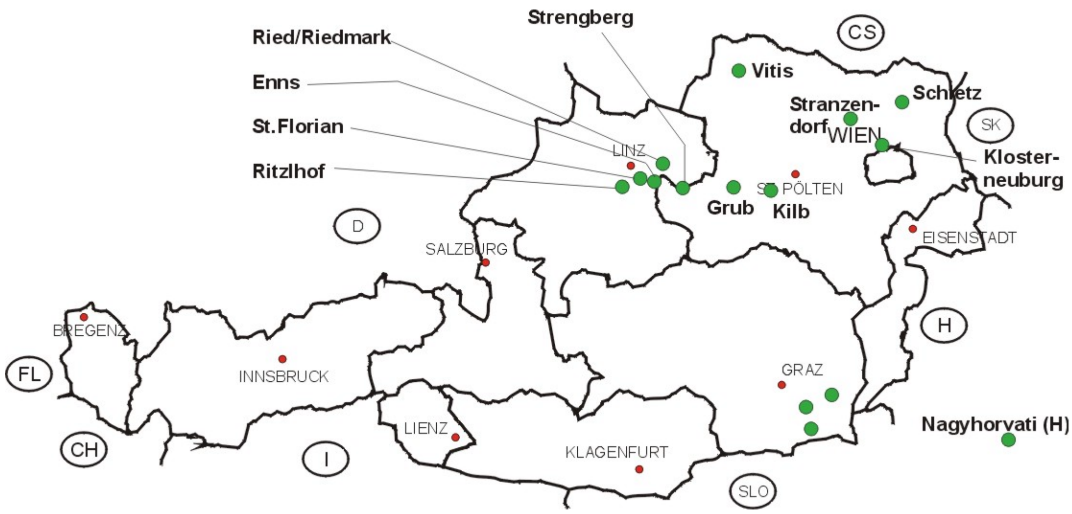


Figure 4.7: Sites at which rainfall simulation experiments have been conducted.

Plot sizes for the experiments varied between 10 m^2 and 12 m^2. A range of different soils occurring in those Austrian regions with the highest erosion risk was used.

Evaluation was only performed on the soil erosion component of both models, which means that values for runoff were not estimated but supplied from rainfall simulation results. In a first attempt we focused on the K-factor which describes the influence of soil properties on soil loss.

A comparison between measured K-factor values and USLE calculated K-factor values gives a correlation coefficient of 0.71 (n = 26) and a mean standard error for the prediction of K-factor values of 0.13. The implication of this result is that for any model application the K-factor cannot be estimated with a higher precision than K-factor ± 0.063 (68 % probability), and any grouping of these data needs to take that into account. This applies only for the case where exact measurements of the necessary input data for K-Factor calculation are available. For any application beyond this, this precision has to be seen as a "best case" assumption. The relationship between measured soil loss and soil loss resulted in the linear regressions (no intercept included) of *(18)* and *(19)*.

*Soil loss (USLE) = 0.73 (± 0.07) * measured soil loss (n = 26, r = 0.61)* *(18)*
*Soil loss (MUSLE) = 0.60 (± 0.05) * measured soil loss (n = 26, r = 0.75)* *(19)*

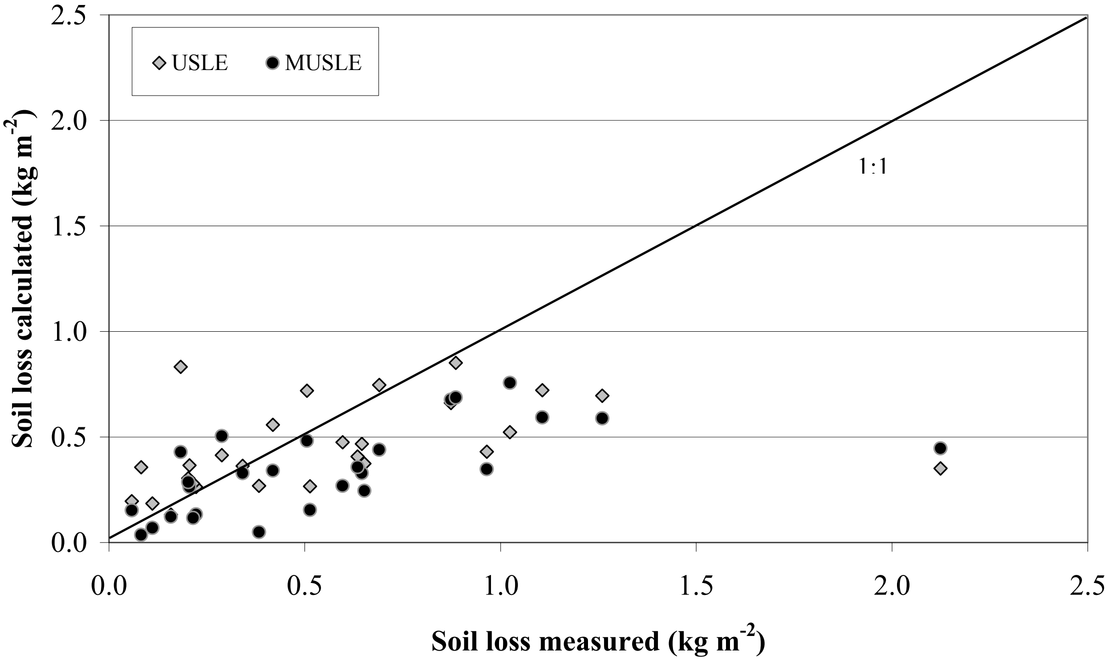


Figure 4.8: Comparison of measured erosion data obtained via rainfall simulation experiments with calculated erosion data using the MUSLE and the USLE equations.

A similar evaluation was done for the MMF model. This resulted in the equation

*Soil loss (MMF) = 1.098 (± 0.28) * measured soil loss (n = 26, r = 0.85)* (20)

This result implicates that for the MMF model application the erosion value cannot be estimated with a higher precision then ±0.52 (95% probability).

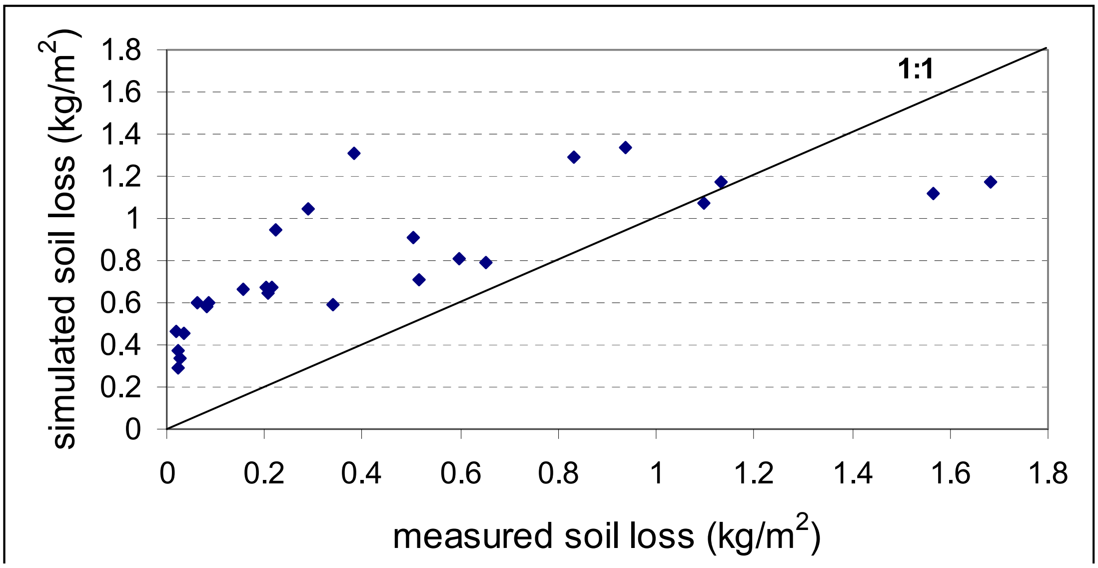


Figure 4.9: Comparison of measured erosion data obtained via rainfall simulation experiments with calculated erosion data using the MMF equation.

Calibration

As a basis for calculation and calibration of surface runoff the knowledge of the regional water balance is essential. Water balance calculations for the Ybbs and the Wulka river basin have been carried out by IHGW (2003) using the water balance model Difga2000 (Schwarze, 2001). This enabled to separate total flow into slow groundwater flow, fast groundwater flow and direct flow. Direct flow rates of Difga2000 were then used for calibration of runoff for both soil erosion models. The drainage basins of the river Ybbs and Wulka were divided into several subbasins. These were used for calibration. In addition river basin outlet data were used to validate calibrated results. Hydrologic calibration for the SWAT model was performed by IHGW (Deliverable D1.1.).

Table 3.5: Runoff calibration results for the different subbasins (north to south) and the main outlet of the river Ybbs, average percentage of total river discharge; simulation period 1991-1997.

Surface Runoff (%)	**Main outlet**	Subbasins **Krenstetten** arable land	 **Ybbsitz** grassland	 **Opponitz** forested	 **Lunz/Ois** forested
Baseflow separation Difga	28.6	32.5	22.9	29.4	31.2
Soil erosion model MMF	32.0	32.5	22.4	29.4	31.5
Soil erosion model SWAT	37.0	56.0	39.0	37.0	23.0

After calibration MMF again is able to describe the regional water balance of the Wulka catchment appropriately. This is also confirmed by a correlation coefficient of 0.96 between measured and calculated runoff values

Simulated sediment loads

MUSLE

Simulation of sediment loads using the MUSLE approach was completely done within SWAT. Basis for the simulation were the calibrated water flow components (for a discussion of problems see Deliverable 1.1). Consequently we used identical evaluation periods for our model application

According to the SWAT handbook (Neitsch et al., 2001) simulated sediment yield should be comparable to measured values once the water balance calibration was done. We did, therefore, not include any instream calculations to handle sediment routing or deposition in the stream but compared only the sediment load which was transported into the stream. This is equal to the assumption, that all sediment which reaches the stream also leaves the watershed. We also did not account for stream bank erosion, which, under specific circumstances may contribute considerably to the total sediment production of a river (DeRose et al., 2002). However, riparian cover, the most important control factor for riverbank erosion, is quite dense for the Ybbs as well as the Wulka river catchments. Therefore we deem riverbank erosion for our environments to be low compared to soil erosion.

MMF

Differing from SWAT, runoff and soil erosion was calculated for individual grids (see model description). For ease of comparison, results of MMF were averaged on the same subbasin level (73 subbasins) as used by SWAT. Similarly to our SWAT calculations we assumed no changes of sediment yield as soon as soil erosion has delivered into the stream. This assumption is supported by the fact, that field inspection in the Ybbs river basin did not confirm big amounts of retention, main channel processes were not calculated. To compare results, we used the same simulation periods as for the modelling in SWAT (1991-97).

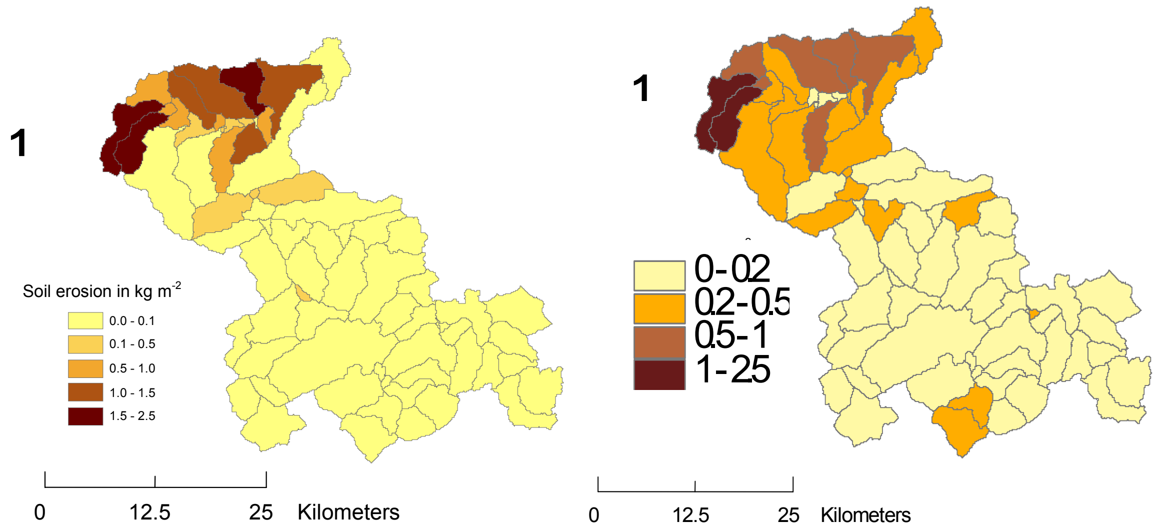


Figure 4.10: Simulated sediment yield (t/ha/a) for various subcatchments of the Ybbs river catchment using the MUSLE approach of SWAT and the MMF model.

Model Comparison and comparison of simulated and measured sediment loads

A comparison of areas affected by erosion gives a relatively good agreement between results calculated with MUSLE and those obtained with MMF. The results of this comparison of MMF and MUSLE for the period 1991-1997 indicate a general agreement on soil loss risk estimation. Results of both models reflect the landuse pattern with low erosion rates in the alpine areas and higher erosion rates in areas with more intense agricultural landuse.

However absolute values of soil loss differ considerably. One reason for this are the different structures of MMF and MUSLE models. While the MUSLE concept in the SWAT system uses only one input value for each parameter to calculate erosion for a hydrological response unit, the MMF model calculates erosion on a grid basis (25 m²). Within one subbasin a lot of grid cells exist, which do not have any erosion (such as pasture, forests but also agricultural areas with low erosion rates such as footslopes or basin areas).

Figure 4.11 demonstrates the general agreement in calculated soil losses for both models for higher calculated soil losses. However, SWAT exhibits a tendency to estimate higher soil losses compared to MMF for those subbasins with a higher soil loss risk. This can be confirmed by the slope value of the linear regression between results of both models which is 0.6 (1 indicates perfect agreement). This compares well with results for the USLE that demonstrate a general overestimation of model predictions at higher soil loss risks (Risse et al., 1993; Strauss and Klaghofer, in press). For particular subbasins, considerable variation in results between the models occurs. For areas with low erosion rates calculated by SWAT and higher erosion rates calculated by MMF, this may be explained by the fact, that SWAT calculates single input values for each HRU or subbasin. In heterogeneous areas with very different landuse intensities SWAT uses those landuse parameters with the greatest aerial extension. Therefore, small areas with a high erosion risk may be neglected, whereas MMF uses all grid values of a subbasin for calculation of average soil loss.

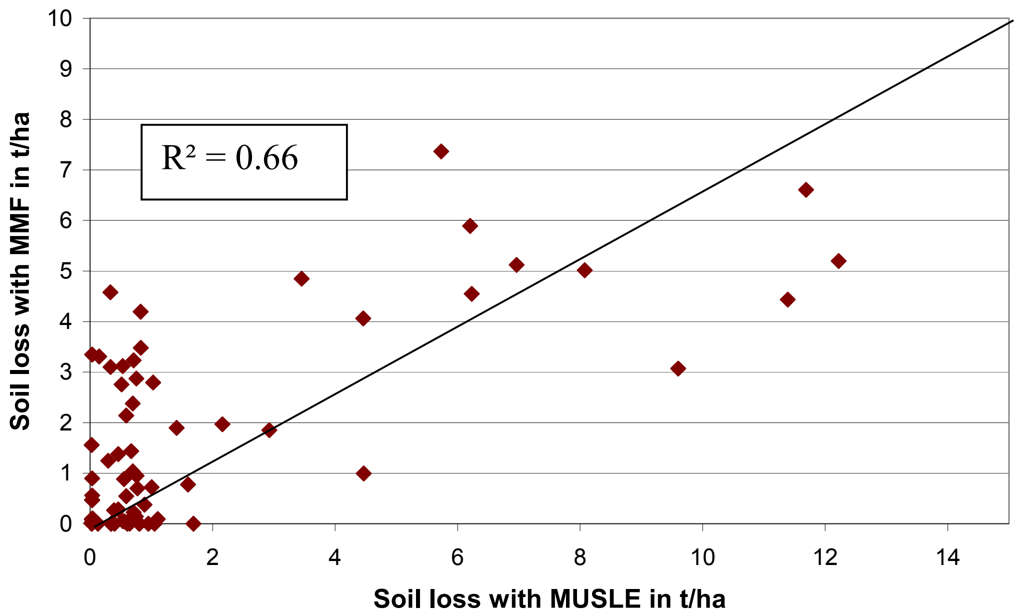


Figure 4.11: Relationship of soil loss calculated with MUSLE as compared to soil loss calculated with MMF (both in kg m^{-2}) at subbasin level.

Model results may also be compared to measured data at those locations, where monitoring had taken place. Procedures to calculate sediment loads are presented in Deliverable D 1.3. A comparison of calculated versus measured loads at the three stations where a detailed monitoring programme had taken place (Table 4.6) reveals that calculated sediment loads for station Opponitz have the best agreement for both models. The relatively high yield in this forested mountainous area is caused by very steep slopes (average more than 30°). MUSLE overpredicts sediment loads for the other stations hugely, while MMF overpredicts only for station Krenstetten. The better fit of MMF to measured data is not surprising as for the MMF also sediment load had been calibrated (at stations Opponitz and Krenstetten). The fact that even with a calibration of MMF at Krenstetten no better fit could be done indicates that with a further calibration unrealistic input data would have been necessary. The overpredictions of sediment load with MUSLE indicate that the assumptions of the model that no further inclusion of a sediment delivery ratio is necessary due to inclusion of an explicit runoff term, do not hold. According to this assumption, the huge differences between measured and calculated loads could only be explained by retention in the river itself. However, field inspection in the Ybbs river basin did not confirm such big amounts of retention. We therefore conclude, that redistribution of soil inside the subbasins constitutes the majority amount of soil erosion.

Table 4.6: Comparison of calculated MUSLE and MMF (already calibrated) values and measured sediment load at the stations Opponitz, Krenstetten and Greimpersdorf (mean values for the years 91-97).

Station	**Sediment load (t/ha/a)**		
	Calculated - MMF	**Calculated - MUSLE**	**Measured**
Opponitz	0.4	0.5	0.4
Krenstetten	1.0	6.2	0.4
Greimpersdorf	0.7	2.7	0.7

Wulka

MUSLE

The approach used in MUSLE uses the spatial distribution of calculated sediment yield for the Wulka river catchment on the basis of 45 subwatersheds. Compared to calculated sediment loads for the Ybbs river, these values are in general lower. Also, no big spatial differences are to be observed. Although landuse is mostly dominated by arable land owning to flat slopes and low precipitation soil erosion is not high. Mean precipitation is almost half of Ybbs catchment. Whenever slopes increase over 10 (in the south and in the north) landuse changes into forest.

MMF

A simulation for the Wulka catchment was also done using the MMF model. As already discussed above, to make the models comparable, results of MMF were averaged on the same subbasin level (45 subbasins) as used by SWAT (see figure 4.12).

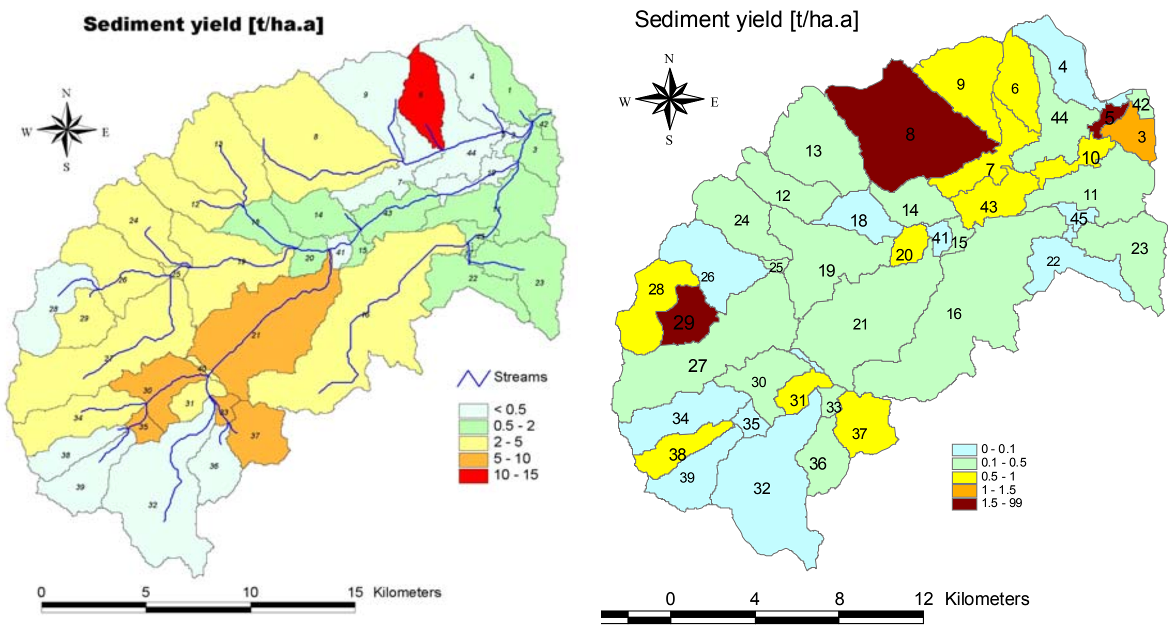


Figure 4.12: Simulated sediment yield (t/ha/a) for various subcatchments of the Wulka river catchment using the MUSLE approach of SWAT and the MMF model.

Comparison simulated – measured sediment loads

Comparing simulated with observed sediment loads a similar tendency as compared to the Ybbs river catchment can be observed. Again, simulated sediment load of SWAT is about an order of magnitude higher compared to measured values (Table 4.7). Compared to the Ybbs river, the different subwatersheds do not differ very much in terms of land use and general environmental conditions. This may be seen very well at the similar sediment loads of the different measurement points in the watershed. This result is also reproduced by the MUSLE. It can be concluded, that a subdivision of this kind of watershed into smaller units is not necessary.

The spatial pattern of soil erosion in the two models is similar. Higher soil erosion occurs in subbasins dominated by vineyards and low erosion in forested areas in the south.

Table 4.7: Comparison of calculated MUSLE and MMF (already calibrated) values and simulated sediment loads for different subwatersheds of the Wulka river catchment (mean values of the years 92-97).

Station	Sediment load (t/ha/a)		
	Calculated - MMF	**Calculated - MUSLE**	**Measured**
Walbersdorf	0.2	1.9	0.2
Wulkaprodersdorf	0.2	2.9	0.2
Schuetzen	0.4	2.4	0.2
Trausdorf	0.3	2.8	0.2
Oslip-Eisbach	1.0	2.1	0.1
Nodbach	0.2	2.0	0.1

As the water flows down the different subcatchments, water flow and sediment load are supposed to increase. These results are calculated sediment loads by MMF. However, measured loads at the different subbasins do not follow this pattern, which is surprising. The results of the two erosion models can not be compared directly due to the fact that in contrast to MUSLE, results of sediments in MMF model have been calibrated and therefore are quite similar to measured data with the exception of subwatershed Oslip. We tried to find reasons for this behaviour. However, runoff rates fit very well and landuse is not really different to other subbasins. In addition, drainage density which could strongly influence sediment yield in the MMF model is similar to other subbasins. So at the moment we are not able to explain this behaviour. In addition it appears that the measured loads at Oslip are quite low.

Simulated phosphorus load

With increasing contribution of soil erosion out of agricultural land, the importance of this pathway to phosphorus inputs into aquatic ecosystems increases. Therefore accurate prediction of sediment delivery is an important and effective approach to predict phosphor discharge into river systems. We calculated phosphorus loads for the various monitoring points of the river Ybbs and Wulka and compared these values to phosphorus loads calculated with two different algorithms connected to the different erosion models. Phosphorus concentrations to be used for the load estimation were calculated as total phosphorus (mg/l) minus total phosphorus of the filtered sample (mg/l). The subsequent calculations therefore correspond to the phosphorus load which is attached to sediment > 0.45 µm.

MUSLE

To simulate total phosphorus loads for the Ybbs and the Wulka river catchments using the MUSLE results as a basis, as a first approach the algorithms implemented in the SWAT interface were used. Main task was to simulate particulate phosphorus as this is attached to erosion. Nevertheless future efforts should also consider dissolved P for two reasons: a) at least in agriculturally used areas dissolved P loads makes up to 20% of the total load and b) whereas particulate P loads may or may not contribute to eutrophication processes immediately, dissolved P is very likely to do so.

Loading of total P is calculated with basically three equations (Neitsch et al., 2001):

$$SedP = 0.001 * conc_{sedP} * \frac{\text{sed}}{\text{area}} * e \qquad (18)$$

$$e = 0.78 * (conc_{sed})^{-0.2468} \qquad (19)$$

$$conc_{sed} = \frac{\text{sed}}{10 * \text{area} * \text{Q}} \qquad (20)$$

with: SedP = amount of P transported with sediment (kg/ha)
$conc_{sedP}$= concentration of P in the top 10 mm of soil
sed = sediment yield on a given day (t)
area = area of hydrological response unit (ha)
e = P enrichment ratio
$conc_{sed}$ = concentration of sediment in surface runoff (t sediment/m^3 water)
Q = surface runoff (mm)

To calculate P content of the topsoil, the default settings proposed by SWAT were used in a first attempt

Ybbs

Table 4.8: Calculated MUSLE (SWAT) and measured total P loads for the Ybbs river watershed (period 91-97).

4.4.2.1. Station	P load (kg/ha/a)	
	Simulated	**Measured**
Opponitz	0.3	0.3
Krenstetten	2.0	0.5
Greimpersdorf	0.9	0.7

A comparison between measured and simulated P loads reveals a good agreement for stations Greimpersdorf and Opponitz and an overprediction of simulated P loads for station Krenstetten. This reflects very well the situation of sediment prediction. There, the deviations between measured and simulated P loads are smaller due to the underprediction of P concentration in the sediment. In fact, this leads to the result that for station Greimpersdorf, measured and predicted P loads are quite close (correct results for the wrong reason).

Wulka

Similarly to the methodology applied for the Ybbs river catchment, P loads for the Wulka river catchment have been calculated.

Table 4.9: Calculated MUSLE and measured total P loads for the Wulka river watershed (period 1992-97).

Station	**P load (kg/ha/a)**	
	Calculated	**Measured**
Walbersdorf	0.7	0.4
Wulkaprodersdorf	1.2	0.3
Schuetzen	1.0	0.5
Trausdorf	1.2	0.3
Oslip-Eisbach	0.8	0.3
Nodbach	0.9	0.2

MMF

Phosphorous loads were also calculated within the MMF Model approach. To do so, an additional algorithm governing movement of phosphorus attached to soil particles from land areas to the stream network was implemented in the model structure.

Eroded sediments usually contain much higher sediment-bound P compared to the topsoil in the upland source area (Sharpley, 1984). The higher phosphorous content in the sediments is due to selective processes of erosion and transportation of fine sized particles. Sediment discharge is enriched by clay particles. Furthermore phosphor in the soil is preferentially attached to colloidal particles. Phosphorous enrichment ratios are used to take this into account, to quantify the increase in particulate P in the eroded sediment compared to that in the upland area topsoil.

To take the differences between total P content in the topsoil and the P content in the river sediment into account enrichment ratios were calculated as the concentration of total P in the simulated sediment divided by that in the soil for each subbasin. Obtained enrichment ratios were then averaged for each case study area.

$$E = \frac{SEDP}{P_{soil} \times SED} \times 10^6 \qquad (21)$$

Where E = P enrichment ratio
SEDP = measured amount of P transported with sediment (kg/a)
P_{soil} = amount of total P in the topsoil (mg P/kg)
SED = simulated sediment load (kg)

Obtained mean enrichment ratios for the two case study areas were 1.2 (s_x = 0.97) for the Wulka river and 1.0 (s_x = 0.40) for the Ybbs river. The standard deviations of these values are given in parentheses. They indicate considerable variation for the different subbasins.

Results of Deliverable D2.1 have employed best available data to simulate soil erosion and nutrient loading in the case study areas using different modelling approaches. Although, the approaches were quite different, all of them used similar input parameters which in general can be identified as being key factors for the description of erosion and nutrient loading. Besides the well known key factors for onsite soil erosion (slope, landuse, soil properties), water flow has been identified to be crucial for the calculation of total loads and soil nutrient contents are a further data source which is needed.

Table 4.10: Calculated MMF and measured total P loads for the Ybbs and the Wulka river watershed.

Station	P load (kg/ha.a)	
	Calculated MMF	**Measured**
Ybbs:		
Greimpersdorf	0.5	0.7
Krenstetten	0.6	0.5
Opponitz	0.4	0.3
Wulka:		
Schuetzen	0.5	0.5
Walbersdorf	0.2	0.4
Wulkaprodersdorf	0.3	0.3
Trausdorf	0.3	0.3
Oslip-Eisbach	1.5	0.3
Nodbach	0.4	0.2

Possibilities to estimate large scale key factors from meso scale investigations

For estimation of sediment and nutrient loads at large scales (national, international) however, the data on key factors which were used for the case study areas are usually not available with the same accuracy and detail. This leads to the question of possibilities to estimate these key factors - which in turn are used as input for the different models – with data of less accuracy. In the following, some of these approaches are highlighted (D2.2).

Top soil P

This parameter mostly is not accessible in national soil databases. One possibility is to calculate total P out of CAL P (D2.2), which is analysed in the national soil monitoring programme. Other approaches for the different countries might be necessary to calculate TP, because different data basis with different parameters for phosphorus can be available there.

Nevertheless for the northern part of the Ybbs catchment (Amstetten) some values for plant available phosphorus (measured as CAL- P in Austria and some other European countries, see ÖNORM L 1087, 1993) and total P have been measured (KS Bodenuntersuchung Amstetten, data submitted by IWA of TU Vienna). These data were assumed to be representative for the region. For the areas not covered by the data some measurements on plant available P exist (given by AGES). It was therefore necessary to calculate a relationship between plant available and total P amounts. Two additional data sets were used for this. Values for plant available P (measured in CAL) and total P had been measured within another European project (DESPRAL) and also within a project carried out by BOKU (data given by Institute for Soil Science). All these data have been pooled and used to develop the relationship given in Formula 22. As a variety of different European soils are covered in this data set, it could be used for other regions as well.

$$TP = 2.0338 * lP + 61.07 \qquad r^2 = 0.5, n = 232 \qquad (22)$$

where: TP = total P content in the topsoil in mg/100g
lP = plant available P content measured as CAL-P in mg/100g

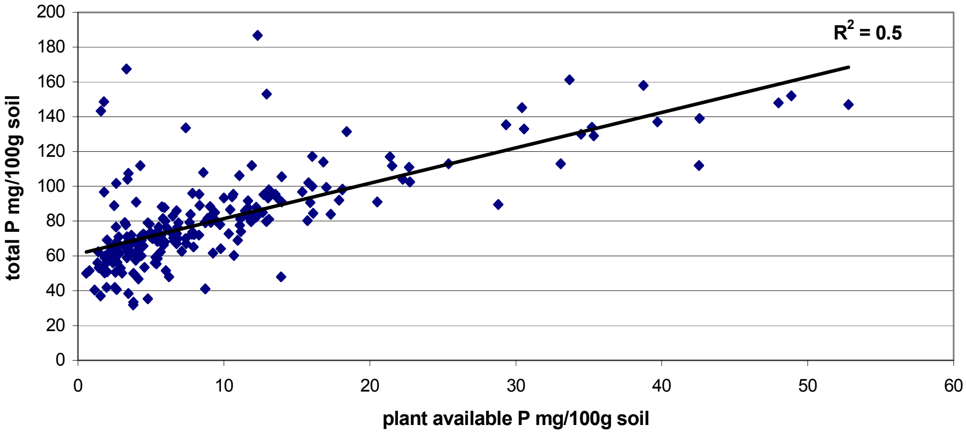


Figure 4.13: Relationship between measured total P and labile P (measured in CAL) loads for different soil types (arable land).

Slope

Figure 4.14 reveals the relationship between mean slope of the subbasins within the Ybbs river catchment and the grid resolutions of 25m, 50m, 100m, 200m, and 400m. As expected, the slope tends to smooth out at coarser grid sizes. The extent of this effect is dependant on the degree of steepness. Based on the findings on decrease of slope with increasing grid size it is possible to develop equations which may be used to correct different grid sizes. However, as the difference between a 25m grid and a 50 m grid was only about 5-10% for the range of slopes in the Wulka river catchment, and the effects on calculated soil loss was low, we did not set up such kind of equations to correct these data. But this fact should be taken into account by all modelling procedures: the coarser the spatial resolution is the higher is the error underlying a logarithmic relation. A set of equations to correct mean slope of catchments when only coarse grid data are available is proposed in Deliverable D 2.2.

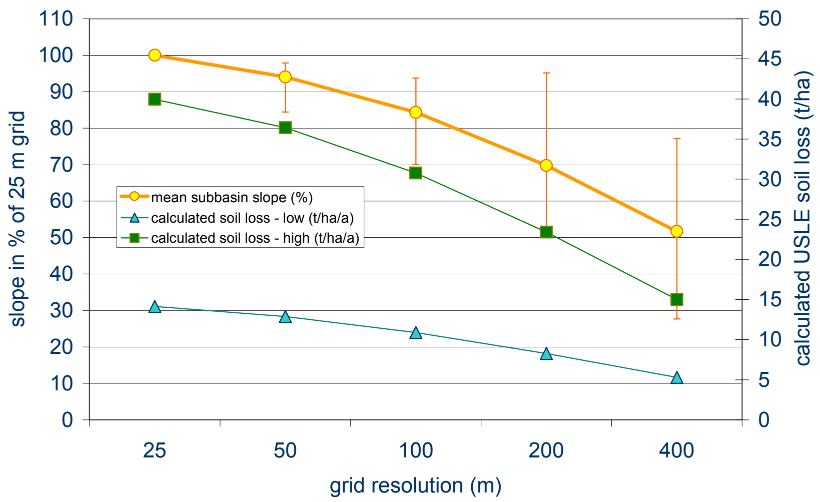


Figure 4.14: Effect of different grid sizes on mean slopes at the subbasin level of the Ybbs river watershed, bars indicate standard deviation. Effect of slope changes on soil loss calculated with the USLE, using a high erosion risk assumption (high) and a low erosion risk assumption (low).

4.5. POSSIBILITIES FOR ESTIMATING POINT SOURCE EMISSIONS

4.5.1. MONERIS approach

In MONERIS the emissions from point source emissions are calculated using statistical data of WWTP`s on national level.

For the estimation of the nutrient inputs by municipal waste water treatment plants and by industrial discharges in the Danube catchment the ICPDR inventory for the year 2000 was used. This inventory includes only the largest point sources and represents for each country about 75% of the total point source emissions into the river systems of Danube.

To estimate the total nutrient discharges by WWTP and direct industrial inputs, a correction based on the available database for each country was made.

For Germany the national WWTP inventory of Bavaria and Baden-Württemberg and for Austira the national WWTP inventory was used to calculate the total nutrient discharges by point sources within the German and Austrian part of the Danube.

For Slovakia and Hungary the WWTP database collected by the consultants was also used. Since this was larger than the ICPDR inventory, it was assumed that the inventory for these both countries was nearly complete.

For the other countries the missing 25% of point source discharges were calculated from the inventory (includes 75% of discharges). This amount was divided by the population of the catchments within the countries investigated where information on WWTP`s was not available.

This specific point source emissions per inhabitant and country is overlayed by the remaining population within the sub-basins to calculate the total nutrient emissions by point sources.

One problem encountered was that the assumption that the ICPDR inventory includes about 75% of the total point source emissions of each country is not actually well defined in relation to specific nutrients and could differ for nitrogen and phosphorus.

4.5.2. Approach basing on standard values

Usually point source emissions are calculated based on the data from the specific enterprises. In many cases those data will not be available, can not be collected with an acceptable effort for all treatment plants or settlements in a region or data are not of acceptable quality. Thus it will be necessary to make assumptions and estimations based on the information available in order to derive realistic emission loads for whole regions. In such case it can not be the goal to calculate accurate values for every single treatment plant or settlement with a high temporal resolution, but to derive appropriate estimates to describe the average situation as realistic as possible. Based on the calculation of nitrogen and phosphorus discharges to waste water produced per inhabitant and the evaluation of a detailed data set of 76 municipal treatment plants in Austria this chapter develops a method for the estimation of nitrogen and phosphorus loads in the influent and the effluent of municipal waste water treatment plants. Further on the accuracy of the method will be evaluated on a set of data from additional 29 treatment plants.

Material and Methods

For calculation of the specific nitrogen and phosphorus discharges from inhabitants to the waste water system two approaches have been used. On the one hand the input of nitrogen and phosphorus into households have been calculated based on the consumption of food, detergents and water in Austria (statistical data) and the related average concentrations of the nutrients in these goods. On the other hand these inputs into households have been balances with data from literature on the production of faeces, dish wash residuals, solid waste discharges and emissions to the air. Further on data from 76 waste water treatment plants with average loadings between 5,000 and 350,000 pe (pe…population equivalent = 60 g BOD_5/d) have been used to derive the variation of population equivalent specific influent loads of nutrients as well as of nutrient elimination rates in the treatment plants. The data sets were collected by Lindtner *et al.*, 2002 and consist of information on design values of the treatment plants (e.g. design capacity in pe, volumes of aeration tanks, of primary and secondary clarifiers and sludge treatment devices), connected inhabitants, sludge production and composition, discharge as well as BOD, COD, N, P influent and effluent concentrations and loads (monthly averages from two to seven measurements every week over one year). In a first step the consistency of the data was checked using detailed P, COD and N balance for every treatment plant. Only treatment plants with complete and consistent data have been used for further evaluation. The drop out rate was somewhere around 50 % with a dependency on the parameter considered. For the remaining treatment plants the distribution and relation of different parameters where investigated in order to derive standard values, that can be used for estimation of yearly averages of effluent and influent loads for nitrogen and phosphorus of other treatment plants in the case, that these data are missing. For calculation of population equivalent specific nitrogen and phosphorus influent loads a method developed by Andreottola *et al.* (1994) with presentation of these specific values in dependency of the relation between connected inhabitants and actual BOD influent loads (expressed as pe) was used. As reference a population equivalent (pe) of 60 g BOD_5/d was assumed. In addition, another set of 41 treatment plant data has been used for the validation of the method for load estimations that was developed based on the standard values derived before. Again, the first step was a plausibility check of the data set. Data from 29 treatment plants remained for further consideration. For these treatment plants estimations for yearly effluent and influent loads of nitrogen and phosphorus have been done based on basic information of different detail and on standard values for e.g. specific nutrient loads in the influent and removal rates. Results of the estimation have been compared to the measured values and the average deviation for all treatment plants and the standard deviation of the deviations for single treatment plants have been calculated. Further on clusters of different numbers of treatment plants have been formed and the average deviation of the measured values and the estimated once for these clusters have been calculated in order to derive information on how many treatment plants have to be considered together in order to get realistic results of estimations based on standard values.

Development of standard values

Influent loads

13 to 15 g N/(inh.d) and 1.8 to 2.4 g P/(inh.d) are consumed by households in Austria, mainly by food but to some extend by detergents and water (nitrogen only) (details see Lindtner, Zessner, 2003). Most of this amount is discharged to the waste water mainly via urine, but via

faeces and dish wash residuals as well. Altogether the discharge to the waste water can be assumed with 11 – 13 g N/(inh.d) and 1.6 -2.0 g P/(inh.d). The rest is discharged to solid waste or released to the air (nitrogen only).

Figure 4.15 and Figure 4.16 show the nitrogen and phosphorus loads per pe of different treatment plants in dependency of the relation between connected inhabitants (inh) and average loading of the treatment plant expressed as pe (based on Andreottola *et al.*, 1994). A relation inh/pe of 1 would indicate discharges from inhabitants (households) only. A relation of inh/pe of 0 indicates only industrial discharges. Values in between show a mixture of household and industrial discharges as usually found in municipal waste water.

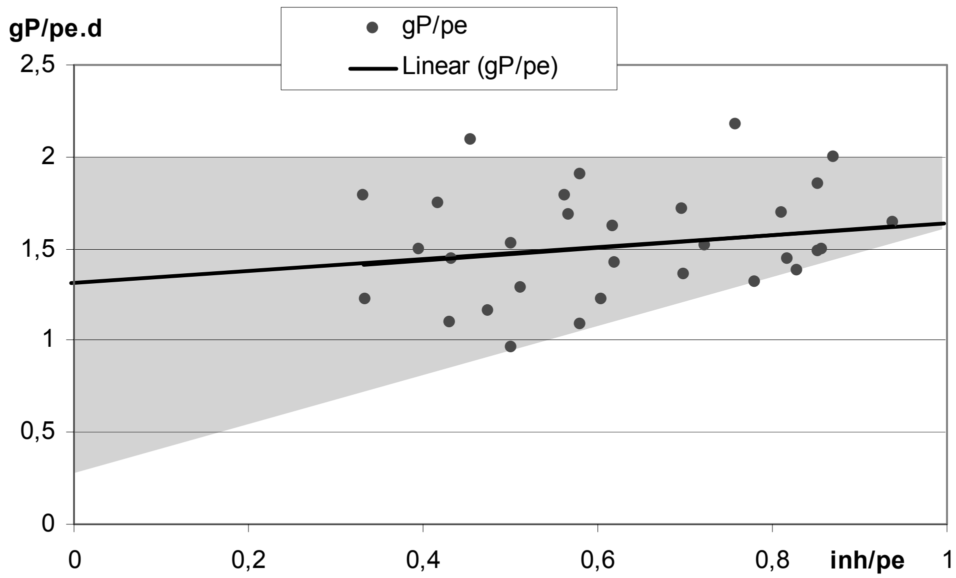


Figure 4.15: Specific P-loads dependent on the inhabitant to population equivalent relation (1 pe = 60 g BOD5/d).

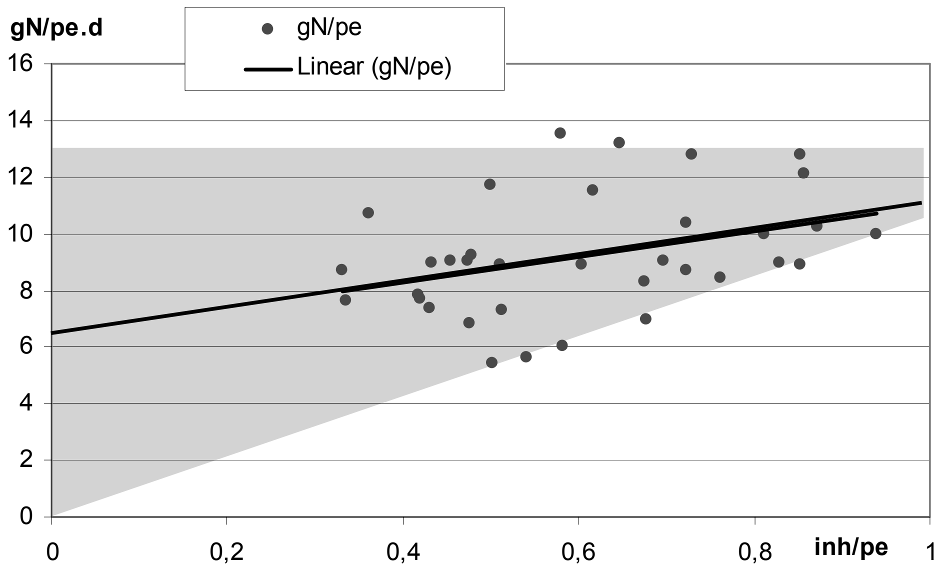


Figure 4.16: Specific N-loads dependent on the inhabitant to population equivalent relation (1 pe = 60 g BOD_5/d).

Following conclusions can be drawn from these figures in order to derive standard values for load estimations:

- In waste waters stemming mainly from households average values of 11 g N/(pe.d) and 1.6 g P/(pe.d) are found. This is well in line with the calculations of inputs form households to waste water of 11 – 13 g N/(inh.d) and 1.6 – 2.0 g P/(inh.d).
- The specific contribution of industries to municipal waste water varies between 0.3 to 2.0 gP/(pe.d) and 0 to 13 g N/(pe.d) with average values of 1.3 g P/(pe.d) and 6.5 g N/(pe.d).
- As average values for municipal waste water (contributions from household and industry) this leads to specific influent loads of 1.5 g P/(pe.d) and 8.8 g N/(pe.d). These values are close to values that Nowak (2000) derived from data of 73 treatment plants (1.53 g P/(pe.d) and 9.3 g N/(pe.d)).

These standard values can be used for estimation of N- and P-influent loads if the actual pe (BOD, COD)-loading (and the connected inhabitants) of treatment plants are known. In case this information is not available, additional assumptions have to be made. The design loads of a treatment plant will be known in most of the cases, the number of connected inhabitants can be derived form statistical data. Figure 4.17 shows which relation between the design capacity or the number of connected inhabitants with the actual loading (pe) of a treatment plant can be expected in the Austrian situation.

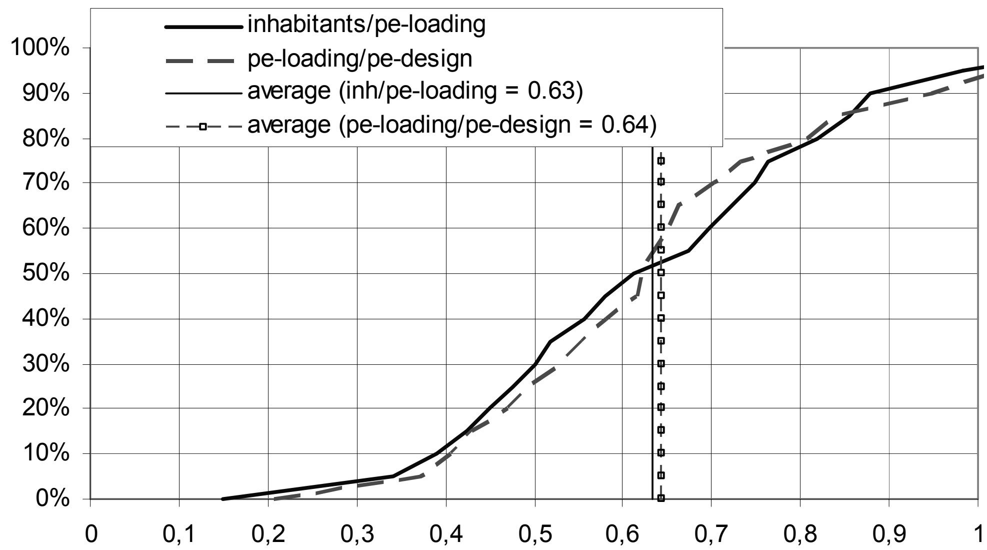


Figure 4.17: Probability curves of the relation between connected inhabitants and actual pe-loading and between actual pe-loading and the pe-design capacity.

The relation between inhabitants and actual pe-loading as well as relation between the actual pe-loading the design capacity expressed as pe varies in a way that 80 % of the treatment plants lie in the range of 0.4 to 0.9. The average values that can be assumed as standard values for estimations of the pe-loading are 0.63 and 0.64 respectively.

Effluent loads

If no data on effluent loads exist they can be estimated based on influent loads (measured or estimated) and typical removal rates. In the following the data set of treatment plants was used to derive those typical removal rates for nitrogen and phosphorus based on information on specific volume of aeration tank and treatment target (carbon removal only = C-wwtp, nitrification = N-wwtp, nitrification/denitrification = ND-wwtp, phosphorus removal = P-wwtp).

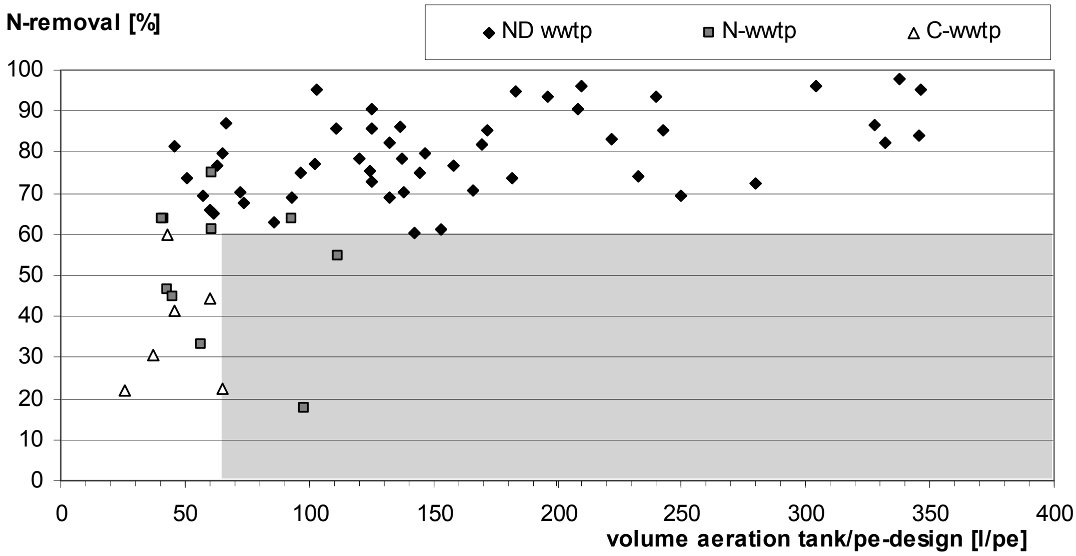


Figure 4.18: N-removal rates in dependency of the specific volume of the aeration tank.

Figure 4.18 shows the relation between nitrogen removal rates and the available volume of the aeration tank per design capacity expressed as pe. Following conclusions can be drawn:

- A nitrogen removal of more than 60 % can be expected if the specific volume of the aeration tanks is more than 65 l/pe. The removal rate for nitrogen shows an increasing tendency with increasing specific aeration tank volume. For treatment plants designed for nitrification/denitrification an average removal rate of 80 % N as yearly average can be expected.
- The nitrogen removal rates for treatment plants designed for carbon removal or nitrification only vary in a wide range. No relation to specific aeration tank volume can be seen. For treatment plants designed for carbon removal only, an average removal of nitrogen of 35 % has been observed. This is significantly more than it is removed by excess sludge. Partly nitrification and denitrification can be expected.
- For treatment plants designed for nitrification an average removal rate of 50 % was observed. Again, partly denitrification happens in most of these cases as well.

For phosphorus evaluation of treatment plant data in respect to removal rates only can be done for treatment plants with additional P-removal (P-precipitation and/or bio-P). Treatment plants without P-removal hardly exist in Austria anymore, thus no representative data set was obtained. On the basis of information from literature a removal of P of 0.6 g/(pe.d) can be assumed. For treatment plants with P-removal two approaches can be used. Either removal rates as percentage of the influent load can be used or the effluent load is calculated based on estimated effluent concentrations and the discharge volume. The evaluation of the data set from treatment plants showed that both approached lead to the same variation of results. In respect to removal rates an average value of 85 % P-removal was obtained. In respect to typical effluent concentrations an average value of 0.75 mgP/l can be assumed. In both cases the variance of results is relatively small. 90% of the treatment plants lie within a range of ± 10 % of these values.

Validation of estimations based on standard values

Influent loads

For validation of the approach for load estimation based on standard values a data set of 29 treatment plants has been used. Following assumption have been made in order to calculated influent loads based on different basic information and standard values and to compare the results of this estimation with the measured loads:

1. Only the design capacity (pe) of a treatment plant is known: The influent loads to the treatment plants are calculated based on an average loading of 63 % of the design capacity and specific loads of 8.8 g N/(pe.d) and 1.5 g P/(pe.d).
2. Design capacity and connected inhabitants are known: An average pe-loading of the treatment plant is calculated from the average of 0.63 of the design capacity and 1/0.64 of the connected inhabitants. The N and P influent loads are calculated based on the formulas derived from figures 4.15 and 4.1.6:

   g N/(pe.d) = 4.5 (inh/pe) + 6.5 and g P/(pe.d) = 0.3 (inh/pe) + 1.3.
3. The BOD-influent load (actual pe-load) is known from measurements: N and P loads can be estimated based on specific values of 8.8 g N/(pe.d) and 1.5 g P/(pe.d).
4. The nitrogen or the phosphorus influent loads are known from measurements: Phosphorus or nitrogen loads can be estimated based on the relation of N:P = 6:1 (= 8.8 gN/d / 1.5 gP/d).
5. The BOD-influent load (actual pe-load) and the connected inhabitants are known: N and P influent loads can be estimated based on the formulas derived form figures 4.15 and 4.1.6:

   g N/(pe.d) = 4.5 (inh/pe) + 6.5 and g P/(pe.d) = 0.3 (inh/pe) + 1.3.

Table 4.11 shows the deviations between measured influent loads and the loads that were estimated based on the assumptions specified above. The table shows average deviation of the whole set of treatment plants as well as the standard deviation of the deviations for single treatment plants. For comparison the absolute values of the measured loads are shown as well.

Table 4.11: Deviation between measured influent loads and the loads that were estimated based on the different assumptions specified above. Deviations are expressed as average deviation of all treatment plants and as standard deviation of the deviation for single treatment plants.

Assumed basic information:	Average ± standard deviation	Average ± standard deviation
1. design load	0.2 ± 3.1 g N/(pe.d)	0.0 ± 0.6 g P/(pe.d)
2. design load and inhabitants	-0.2 ± 2.5 g N/(pe.d)	0.0 ± 0.5 g P/(pe.d)
3. BOD load	-0.7 ± 1.7 g N/(pe.d)	-0.1 ± 0.2 g P/(pe.d)
4. N or P load	0.1 ± 1.7 g N/(pe.d)	0.0 ± 0.3 g P/(pe.d)
5. BOD load and inhabitants	-0.8 ± 1.9 g N/(pe.d)	-0.1 ± 0.2 g P/(pe.d)
Average of measured loads	9.5 g N/(pe.d)	1.6 g P/(pe.d)

It can be seen that the average deviation between estimated and measured values for the whole set of treatment plants is relatively small (< 10 % of total measured load) for all different basic assumptions. For single treatment plants the deviation may be much higher (20 – 40 %). Estimations are significantly better if at least one from BOD-load, P-load or the N-load in the influent is known.

Effluent loads

For estimation of effluent loads the same assumptions for the estimation of influent loads have been used as described before. Based on this estimated influent loads effluent loads have been estimated and compared to the measured effluent loads. For estimation of the effluent loads elimination rates for nitrogen and phosphors have been assumed dependent on the treatment target of the plant. For treatment plants designed for carbon removal only, elimination rates for nitrogen of 35 % have been assumed, for treatment plants with nitrification 50 % and for plants with nitrification/denitrification 80 %. For plants operated with additional P-removal removal rates of 85 % have been assumed. For plants without additional P-removal the removal was estimated with 0.6 g P/(pe.d). Table 4.12 shows the average deviations between estimated and measured values as well as the standard deviation of the deviation of single treatment plants and for comparison the absolute value of the average of the measured effluent loads. Again the estimations based on different basic assumptions specified above are compared.

Table 4.12: Deviation between measured effluent loads and the loads that were estimated based on the different assumptions as specified above. Deviations are expressed as average deviation of all treatment plants as well as standard deviation of the deviation for single treatment plants.

Assumed basic information:	Average ± standard deviation	Average ± standard deviation
1. design load	0.1 ± 1.3 g N/(pe.d)	-0.01 ± 0.17 g P/(pe.d)
2. design load and inhabitants	0.0 ± 1.2 g N/(pe.d)	0.00 ± 0.16 g P/(pe.d)
3. BOD load	0.1 ± 1.3 g N/(pe.d)	-0.03 ± 0.15 g P/(pe.d)
4. Nor P load	0.3 ± 1.3 g N/(pe.d)	-0.01 ± 0.16 g P/(pe.d)
5. BOD load and inhabitants	0.0 ± 1.3 g N/(pe.d)	-0.03 ± 0.15 g P/(pe.d)
Average of measured loads	2.4 g N/(pe.d)	0.25 g P/(pe.d)

Again the estimation for all treatment plants fits well to the measured value. The deviations are less than 13 % of the measured values. For single treatment plants the deviations might be very high (up to more than 70 % to the measured value). This result is independent from the basic information that has been used for estimation of influent loads. An estimate based on the design capacity and treatment target only has the same accuracy as an estimate based on measured BOD-loads in the influent and the treatment target of the plant.

If the average estimation for 29 treatment plants fits well to the measured values and the estimations for single plants do not fit, the question rises, how many plants have to be considered together in order to achieve reliable results. Therefore, clusters of different numbers of treatment plants have been formed and the deviation between estimated and measured values for these different clusters has been calculated. Figure shows these calculations for example of BOD-load and connected inhabitants as basic data for the calculation of the effluent loads. It clearly can be demonstrated that the deviation decreases with the increasing number of plants considered together in a cluster. If we take more than 10 plants together (e.g. of a river basin or a region), the deviation of this calculations as compared to measurements can be expected to stay below ± 20 % in 90 % of the cases.

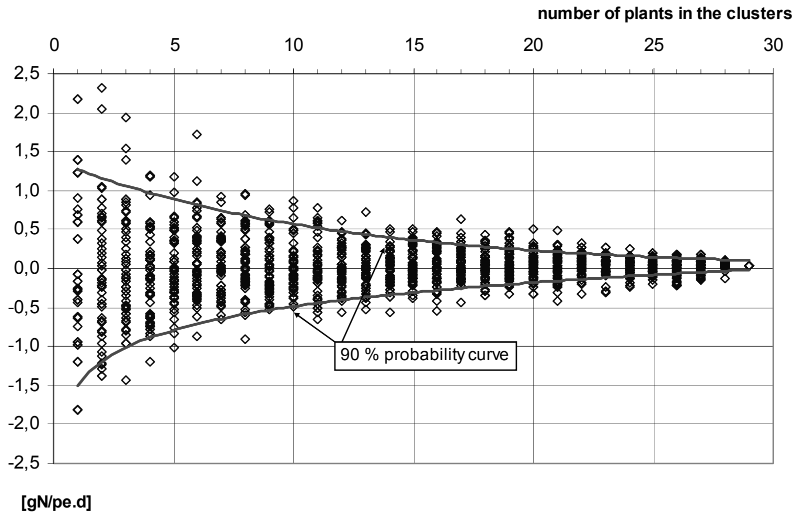


Figure 4.19: Deviation of estimated from measured specific nitrogen loads dependent of the number of treatment plants considered together in a cluster (estimations of influent loads were based on BOD-load in the influent and connected inhabitants).

Finally for whole Austria estimation only based on the design capacity and design targets (with or without nitrification/denitrification or P-removal) as well as standard values derived above was done. The calculations in most of the cases fitted very well with the reported data from the different federal states. For whole Austria (without Vienna) own estimations matched with official data from authorities with a deviation of less than 10 % for N and P in influent and effluent. Details are published in Lindtner, Zessner (2003).

Conclusions

In order to obtain completed data sets on yearly N and P influent and effluent loads of waste water treatment plants in catchments, regions or countries it will be necessary in many cases to make estimations for treatment plants, where no measured data exist.

In this paper standard values for N and P influent loads have been derived. As average values for municipal waste water (contributions from household and industry) this leads to specific influent loads of 1.5 g P/(pe.d) and 8.8 g N/(pe.d). Further relations between actual loadings and design capacity and between connected inhabitants and actual loadings as well as nutrient elimination rates were presented. All these relation may vary in a wide range.

If averages of relations of different parameters are used as standard values for estimation of influent or effluent loads deviations, between estimates and measured values may be significant for single treatment plants. Nevertheless, if estimates are done for a higher number of plants together the reliability of estimates increases significantly. If more than 10 pants are considered together, the expected deviation between estimations and measurements stays beyond ± 20 % for the estimation of yearly effluent loads.

Estimates of influent or effluent loads never will be able to replace measurements at the different treatment plants. But for regional studies estimates based on the method presented are an efficient tool to complete data sets and to check the plausibility of existing data.

Nevertheless the assignability of the standard values obtained with the described method is very limited. Results do only correspond to Austrian circumstances. In other regions or countries with specific premises (e.g. new standard values have to be deduced. with other standard

4.6. POSSIBILITIES FOR ESTIMATING RETENTION AND TRANSPORT IN SURFACEWATER

4.6.1. Estimates of retention and transport in the Danube catchment (large scale)

River loads

In the emission model MONERIS for each of the investigated sub basins where data on concentrations of nutrients and discharges were available, annual nutrient load was calculated according to the Equation 24. This method for the calculation of load is also favoured by OSPAR (1996) for the calculation of loads into the North Sea. In a comparison of five various methods to estimate annual nutrient load for English rivers, Littlewood (1995) showed that only this method gave reliable load estimates.

$$L_y = a \cdot \frac{Q_y}{\sum_{i=1}^{n} q_i} \sum_{i=1}^{n} q_i c_i \qquad (24)$$

with

L_y	= annual load [t/a].
a	= unit conversion factor,
n	= number of data,
Q_y	= mean annual flow [m^3/s],
q_i	= measured flow [m^3/s] and
c_i	= measured concentration [mg/l].

From the annual values, the mean load for the studied time period 1998-200 was estimated according to Equation 25:

$$L_P = \frac{1}{p} \cdot \sum_{i=1}^{p} L_T \qquad (25)$$

with

L_P	= average annual nutrient load in the studied period [g/s],
P	= number of years with measuring data in the study period.

Retention in Rivers

When comparing the estimated nutrient emissions and the load in the catchment areas, considerable variation was found (Behrendt, 1996b; Behrendt & Opitz, 1999) which could not be explained by an underestimate of the load or an overestimate of the inputs (Behrendt & Bachor, 1998). These differences were instead due to retention and loss processes within the river systems e.g. sedimentation, denitrification and plant uptake.

On the basis of data for nutrient emissions and loads in 100 catchment areas with a size of 100 to 200.000 km², an empirical model was therefore derived (Behrendt & Opitz, 1999) for the retention of nitrogen and phosphorus in relation to the specific runoff or the hydraulic load in the catchment area. The basis for the model is the mass balance of a catchment area whereby the observed nutrient load for a time period of one or more years is the result of the balance of the sum of all inputs from point and diffuse sources and the sum of all retention and loss processes:

$$L_{N,P} = ET_{N,P} - R_{N,P} = \sum EP_{N,P} + \sum ED_{N,P} - \sum R_{N,P} \qquad (26)$$

with

$L_{N,P}$ = nutrient load [t/a],
$ET_{N,P}$ = total nutrient input [t/a],
$R_{N,P}$ = loss or retention of nutrients [t/a],
$EP_{N,P}$ = nutrient input via point sources [t/a] and
$ED_{N,P}$ = nutrient input via diffuse sources [t/a].

After adjustments of Equation 26 we get the following:

$$\frac{L_{N,P}}{ET_{N,P}} = \frac{1}{1 + R_{L_{N,P}}} \qquad (27)$$

with $R_{L_{N,P}}$ = load weighted nutrient retention. For the description of possible relationships between retention (R_L) and possible driving forces a power function is selected.

$$R_{LN,P} = a * X^b \qquad (28)$$

with

a, b = model coefficients.

On the basis of the available data, there are relationships between retention and specific runoff and also the hydraulic load in the catchment areas. In addition to the retention derived only for the load of inorganic dissolved nitrogen (DIN) a corresponding relationship was found for total nitrogen (TN).

The following models are used for the calculation of retention of TN, DIN and TP:

TN: $R_{LN} = 1.9 * HL^{-0.49}$ $n = 56, r^2 = 0.52$

with HL = hydraulic load [m/a].

DIN: $R_{LN} = 5.9 * HL^{-0.75}$ $n = 100, r^2 = 0.65$

TP: $R_{LP} = 26.6 * q^{-1.71}$ $n = 89, r^2 = 0.81$

with q = specific runoff [l/(s*km^2)].

If these approaches are applied, the nutrient load can be calculated from the nutrient inputs for all studied catchment areas (Equation 29) and the results can be compared with measured loads.

$$L_{N,P} = \frac{1}{1+R_{L_{N,P}}} \cdot ET_{N,P} \tag{29}$$

4.6.2. Estimates of retention and transport in the case study areas (meso scale)

River loads

Load calculations are of decisive importance in the frame of regional nutrient balances. As shown before (e.g. Zessner and Kroiss, 1999) for phosphorus the concentrations in the rivers increase with increasing river discharges. Therefore, results of load calculations highly depend on the inclusion of monitoring at high flow events as well as on the method used for load calculation. Accordingly at Wulka and Ybbs high flow events had been monitored. In Figure 4.20 for sampling points at the Wulka the relation between the discharge and the TP-load is plotted. The load increases with increasing discharge over proportional. Similar are the results for the other measuring points.

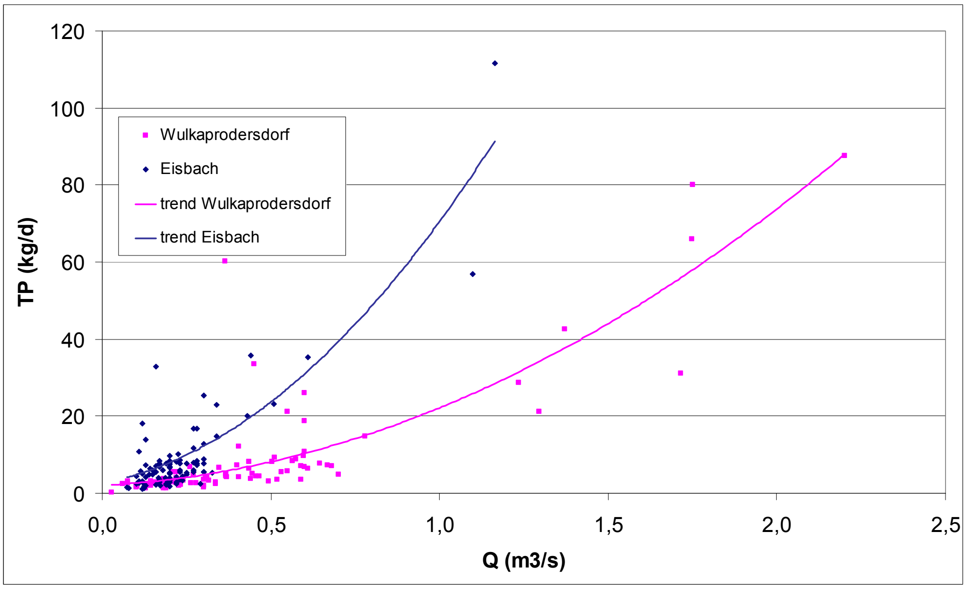


Figure 4.20: Discharge to TP-load relation for the sampling points Wulkaprodersdorf and Eisbach in the Wulka catchment.

In order to consider this relation and to evaluate possible impact on the calculation method on results of load estimates, loads have been calculated with two approaches:

1.: Based on averages on days with measurements and a correction in the ratio between the mean flow of the period for which loads are calculated and the average of the flow of the days where measurements exist. The used formula is:

$$L_j = \frac{\sum_{i=1}^{n} Q_i . c_i}{n} \times \frac{MQ}{Q_m} \times 31{,}5$$

(30)

L_j	average yearly load	[t/a]
Q_i	discharge on sampling day	[m³/s]
c_i	measured concentration	[g/m³]
n	number of measurements	
MQ	mean discharge of the considered period	[m³/s]
Q_m	mean discharge of sampling days	[m³/s]

2.: Based on a function that describes the relation between discharge and daily load (see Figure 4.20). This function is used to create daily loads based on daily discharges. Yearly loads can be calculated based on these “syntactic” loads. In some cases it might be necessary to create two functions: one for lower discharges and one of higher discharges. This method guarantees that weighted averages of loads according to the frequency of discharges are calculated.

Results of load calculations are shown in Table 4.1.3. Especially for SS and TP loads calculated with method 1 are significantly higher than those calculated with method two. The reason is that due to special sampling at high flow situations the number of samples at high flow is over represented. This leads to higher results for load calculations in the case that loads increase over proportional with discharge. In case that sampling at high flows is underrepresented the results of method 1 would be lower than those with method 2. For dissolved N-, P-parameters and SiO_2 the differences are much less pronounced, because the relation between loads and discharges is more or less proportional (linear regression).

Table 4.13: Results form load calculations (1998-2002) at different sampling points of the Wulka catchment.

	Method	Q	SS	TP	TPfilt	TN	anorgN	SiO_2
		mm	kg/ha.a	kg/ha.a	Kg/ha.a	kg/ha.a	kg/ha.a	kg/ha.a
Walbersdorf	1	95	119	0,40	0,12	3,8	2,9	9,2
Walbersdorf	2	95	59	0,23	0,1	3,0	2,4	7,2
Wulkaprodersdorf	1	66	49	0,15	0,08	3,3	3,3	6,7
Wulkaprodersdorf	2	66	38	0,14	0,07	3,5	2,7	6,3
Trausdorf	1	93	172	0,37	0,19	4,5	3,4	10,0
Trausdorf	2	93	63	0,29	0,18	4,2	3,2	9,1
Nodbach	1	54	65	0,19	0,07	3,1	2,2	5,0
Nodbach	2	54	27	0,12	0,07	2,6	1,8	4,1
Eisbach	1	86	44	0,44	0,27	7,5	6,1	11,2
Eisbach	2	86	31	0,39	0,26	7,4	6,4	9,9
Schützen	1	78	207	0,31	0,13	4,4	4,1	4,8
Schützen	2	78	55	0,19	0,11	3,7	3,3	6,9

In the following considerations results obtained with method 2 were used.

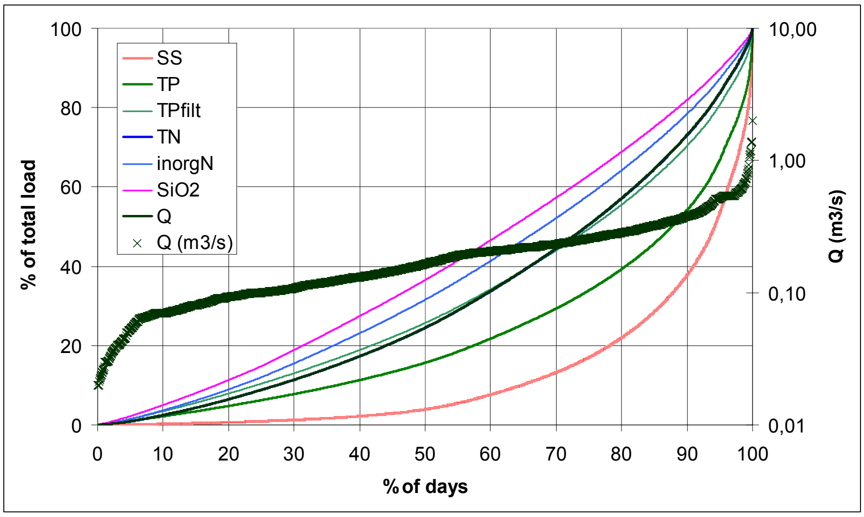


Figure 4.21: Load duration curve and contribution to the total transported loads at shares of days at the station Walbersdorf at the Wulka.

Figure 4.21 to Figure 4.24 show probability curve for discharges at the measuring stations in the Wulka catchment as well as the contribution to the total transported loads at shares of days. For instance in case of Walbersdorf at the Wulka it can be seen that in the period 1998 -2002 in 80 % of the days only 22 % of the SS-load was transported. The rest was transported in only 20 % of the days. For TP within 80 % of the days 40 % of the load was transported. The other parameters are more similar to the relation of the water discharge were within 80 % of the days about 60 % of the water volume is discharged. Other examples show the Nodbach, the Eisbach and Wulka at Schützen. While the Nodbach is similar to the Wulka at Walbersdorf, at Eisbach and Wulka at Schützen the loads are more equally distributed over the discharges because the influence of point sources is more pronounced.

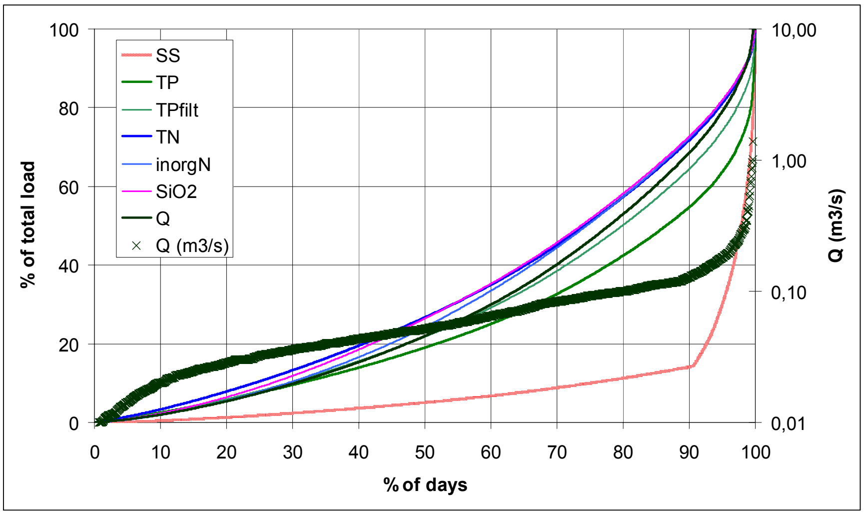


Figure 4.22: Load duration curve and contribution to the total transported loads at shares of days at the Nodbach in the Wulka catchment.

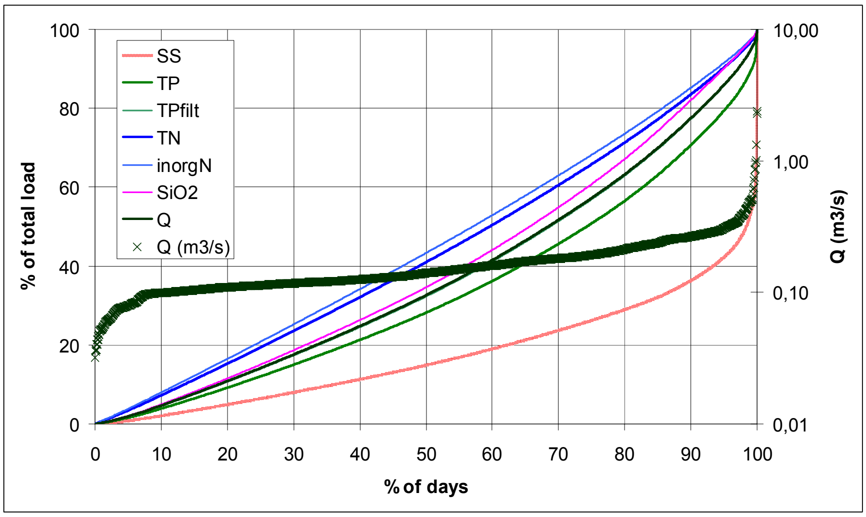


Figure 4.23: Load duration curve and contribution to the total transported loads at shares of days at the Eisbach in the Wulka catchment

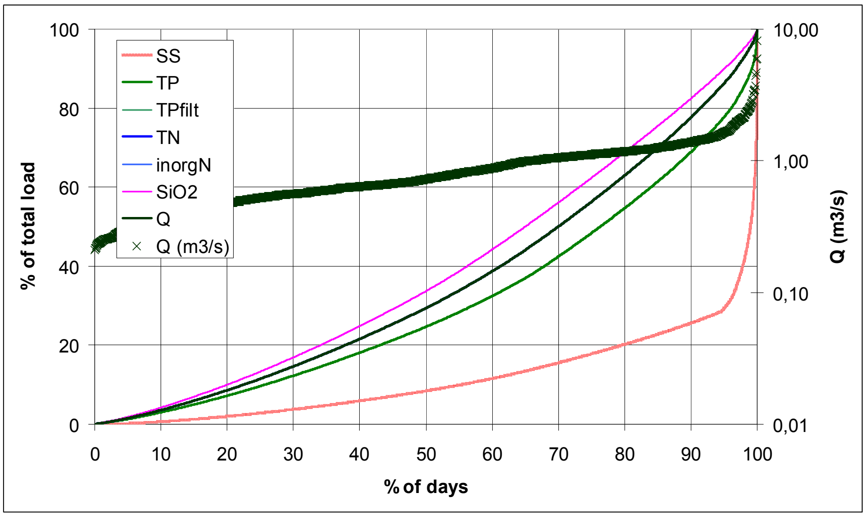


Figure 4.24: Load duration curve and contribution to the total transported loads at shares of days at the station Schützen at the Wulka

River retention

Nitrogen

Results indicate, that the retention (difference between emissions and transported loads) in the surface waters of the case study regions lies usually between 0 and 2 kgN/(ha.a) if it is related to the total catchment area. As compared to the retention/denitrification in soil and groundwater (5 – more than 50 kg N/(ha.a)) this is relatively small. Nevertheless, retention in surface waters may influence river loads (1 – 20 kg N/(ha.a)) very significantly. In the case study areas it reduces river loads up to 75 %. Retention may be storage of nitrogen in the river corridor (wetlands, dams, floodplains) or release from the water course by denitrification. It is not possible yet to distinguish between these two forms of retention quantitatively. There is evidence that denitrification plays the major role.

In respect to denitrification from surface waters in general the same factors are decisive as for groundwater. Denitrification by heterotrophic bacteria takes place in case of degradation of organic carbon under absence of dissolved oxygen. The denitrification can either be limited by the availability of organic carbon or by the competition between nitrate and dissolved oxygen as electron donor for carbon degradation. In contradiction to the groundwater, oxygen supply in river water is much better and the reaction time is much shorter. Thus, except in cases of heavy oxygen depletion by high inputs of organic matter, denitrification in the free flowing phase of a river will be of minor importance. Denitrification will take place in side arms of wetlands, where lower residence times coincidences with high availability of carbon (algae production) and reduced supply of oxygen, and in sediments of the riverbed, as it was demonstrated in the Austrian case study areas Ybbs and Wulka (see deliverable D1.3, Austrian part, chapter 5.3). In case of downwelling river water through the sediments, the factors availability of carbon (sedimentation of suspended solids, e.g. algae biomass), reduced oxygen supply and increased reaction time appear and lead to significant reduction of nitrogen concentrations. The relevance for the reduction of river loads will depend on the amount of water that is exchanged between surface and groundwater. The quantitative assessment of this amount was not possible within the investigations of the case studies. Qualitatively it can be stated, that in natural near river systems this exchange will be more relevant as in more canalised rivers and the relevance of this exchange will tend to increase with an increasing relation between the area surface area of the river bottom and the river discharge.

Consequently the MONERIS model uses the hydraulic load (surface water discharge of a subcatchment subdivided by surface water area) as parameter for the determination of the retention in the surface water. As it could be shown for the case study investigation on subcatchment level, based on this approach a good match between calculated river loads (emission estimations minus retention in the river) and measured river loads could be obtained. The average aberration between measured and calculated loads was 16 % for 23 subcatchments of the case study investigations. An important prerequisite for this calculation is a sound determination of the surface water area. The approach of MONERIS based on the determination of the river retention from a correlation to the areas specific runoff of a catchment needs no detailed determination surface water area, but this approach showed a much lower accordance between calculated and measured river loads. Results differ significantly from the one obtained with the “hydraulic load approach”. The average aberration between calculated and measured river loads of 23 case study subcatchments was 47 % with a clear tendency of a systematic error towards overestimation of retention and therefore lower calculated river loads as compared to measured ones.

Specific Nitrogen loads in the rivers of the case study areas vary between 1 and 20 kg N/(ha.a). Highest values are in those areas with high N-surpluses in soils and low denitrification in groundwater due to hydrogeological conditions. The measurements of nitrogen loads are not problematic, because the influence of special events on the yearly load is small. A sampling system of biweekly sampling should be appropriate for realistic results. Loads between neighbouring years can differ significantly in correlation with the different mean water discharge. In contradiction with former practice, dissolved inorganic nitrogen is not sufficient to determine total nitrogen. Organic nitrogen forms (dissolved and particulate) shall be considered. For the different case study regions organic forms of nitrogen contribute with 10 to 40 % to the total annual nitrogen loads in the rivers.

Phosphorus

Results indicate, that the retention (difference between emissions to the river and transported loads in the river) in the surface waters of the case study regions lies between 0 and 0,7 kgP/(ha.a), if loads are related to the total catchment area. As compared to the retention in the catchment, this is relatively small. Nevertheless, retention in surface waters may influence river loads (0,2 – 1 kg P/(ha.a)) significantly. In the case study areas it reduces river loads by up to 75 %. Retention is storage of phosphorus in the river and the river corridor (wetlands, dams, floodplains). Together with emission estimates via erosion, quantification of retention in the river system is the most decisive value of a phosphorus balance for a river system.

In respect to transport of phosphorus from the catchment to the receiving Sea there is no immediate influence between phosphorus emissions via erosion and the transported loads. Retention in the river system decisively influences this transport to the Sea. Erosion events are usually in coincidence with high flow or flood events in upstream parts of a river catchment. Particle-bound phosphorus is mobilised from the catchment (erosion) and the river bottom and transported at peak discharges to downstream sections, where retention by sedimentation of particles takes place. On the one hand this retention is a transport to flooded areas. In this case it can be considered as more or less long term retention. On the other hand sedimentation takes place in the riverbed, in case the tractive effort of the river is reduced. In this second case the P-pool in the sediments of the sedimentation area will be increased. If anaerobic conditions in the sediment appear, part of the phosphorus will be transformed to soluble ortho-phosphate and will continuously contribute to the phosphorus transport to the receiving Sea. Part of the P-retained in the river sediment will be mobilised by resuspension at the next bigger high flow event. All together, these alternating processes of suspension, transport, export to flooded areas or sedimentation in the river bed with partly solution and partly resuspension at the next event decreases the total phosphorus transported downstream and decreases the share of the phosphorus transport during high flow events on the total loads transported in the more downstream parts of a catchments as compared to the more upstream parts.

MONERIS model uses the hydraulic load (surface water discharge of a subcatchment subdivided by surface water area) and the areas specific runoff of a catchment as parameters for the determination of the retention in the surface water. The results of calculation of river loads differ significantly between the two approaches. For both approaches the average aberration between calculated river loads and measured ones is as average for 23 subcatchments considered in the case study investigations about 70 % (hydraulic load) and about 50 % (specific runoff) respectively. While using the retention approach with the area specific runoff this aberration clearly shows a tendency towards an underestimation of calculated loads as compared to the measured loads, the approach using the hydraulic load shows a week tendency of overestimating the calculated loads as compared to the measured loads. Definitely the relation between erosion estimates, retention in the catchment and in the river system is the main reason for the uncertainty of these calculations. In addition, the measurement of the river load is a factor of uncertainty for phosphorus. These uncertainties tend to increase with a decreasing catchment size, because the dynamic of transport processes in a river increases with decreasing catchment area. This might be an explanation why the fit between measured and calculated river loads is much better for the Basin wide application of MONERIS and the case study investigations.

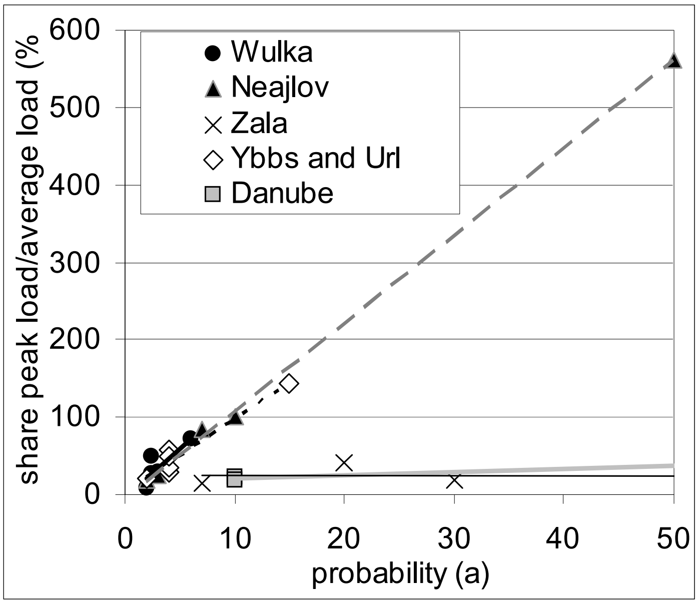


Figure 4.25: Relation between event probability and share of phosphorus loads transported during the event as compared to total average yearly load.

It was clearly demonstrated, that high flow events significantly influence the P-load in river systems. Most of the P-loads are transported within few days of a year. High flood events can transport P-loads in the order of magnitude of yearly average loads. Transported loads vary a lot between different years, depending on the discharge situation of the year. Using averages over some years (e.g. 5) reduces the influence of these fluctuations. Nevertheless, it is of high importance to include high flow events into load monitoring and calculation.

The influence of high flow events on the phosphorus transport decreases with increasing catchment area and is of much higher importance in tributaries than in the Danube River.

For monitoring of P-loads this means that flood events have to be specifically addressed in tributaries anyway. In a large river the importance of event oriented load monitoring depends on the time scale considered. For calculations of yearly loads monitoring at flood events is still decisive. If average loads over 5 years and more are taken into consideration, monitoring at flood events is less decisive, unless the probability of events increases significantly due to change of landuse practices in the catchment or climate change.

5. REQUIREMENTS FOR AN ADEQUATE MONITORING STRATEGY TO PROVIDE NUTRIENT LOAD CALCULATIONS ON LARGE SCALES

Data precision is essential to provide reliable nutrient balances on catchment scale. This includes data precision of available data sets concerning regional data but also of load measurements at specific strategic (e.g. borders) or geographical/hydrological (e.g. confluence of rivers, reservoirs, sub catchment boarder etc.) points.

As mentioned before comparing model results with load measurements is the only way to prove the reliability of model output.

Based on the results of the daNUbs project requirements for an adequate monitoring system will be formulated. For the Danube river on practice this means to upgrade the recent DWQ TNMN (= Danube Water Quality Transnational Monitoring Network) monitoring programme which started in 1996.

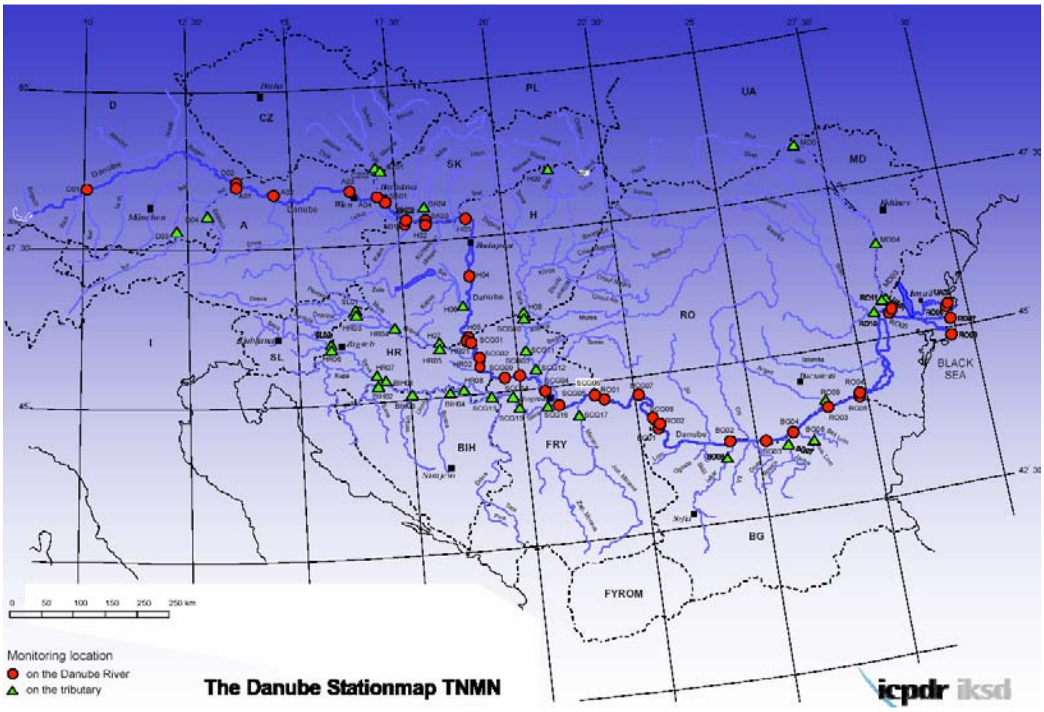


Figure 5.1: The Danube Stationmap of the TNMN; Source: icpdr.

In the TNMN 78 sampling stations are included and 52 parameters are monitored. The minimum sampling frequency is determined to be 12 times per year for chemical analyses and 2 times per year concerning biological parameters.

For N and P-balances the following determinands are currently analysed

- Ammonium-nitrogen
- Nitrite-nitrogen
- Nitrate-nitrogen
- Organic nitrogen
- Ortho-phosphate-phosphorus
- Total phosphorus

The results of the study on determination of the bioavailable phosphorus forms indicated that the particulate phosphorus is an important part of phosphorus loads, therefore, analysis of the sediment (suspended solids) bounded phosphorus by the analysis of the unfiltered water samples is necessary.

According to results of own measurements presented in Deliverable 4.4 and 4.5 the following recommendations are made:

- During stable flow conditions the difference between calculated weekly loads based on daily concentration measurements or based on one per week concentration measurement are low. The calculated weekly loads based on daily concentration measurements or based on one per week concentration measurement differ considerably during flood conditions.

- As the loads are significantly different during flood, therefore, additional sampling (as "Special Surveys") in the regular TNMN schedule improves the load calculation of nutrients (total-N and total-P) during the flood events. If the forecast has time advantage, the start of the frequent measurement is a day before the floodplain is inundated. In the case of forecast without enough time to start the frequent measurement before the floodplain is inundated, the sampling should be started immediately.

- Monitoring of flood events have to be specifically addressed in tributaries

- The concentrations of suspended solids and particulate nitrogen and particulate phosphorus species strongly increase during floods. This implies that high flow events contribute more than proportionally to the annual load of total nitrogen, total phosphorus. The relation between flow and total nitrogen is loose. Ammonium concentration increases while nitrate decreases with rise of flow.

- It is recommended to include additional sampling into the normal monitoring schedule during flood events to improve load estimations of nutrients total N and total P.

- It was concluded that the vertical differences in the concentrations in the surface and deep layer are generally smaller (mostly less than 10 %) than the differences along the cross section. The dissolved components did not show typical differences in the vertical distribution, but there was an exception for ammonium in a period in the Gabcikovo area.

- The load surveys shall enable to estimate the effect of retention of nutrients (total nitrogen and total phosphorus) within reservoirs. Nutrient retention (accumulation in the sediment and/or uptake by biota) could be studied in reservoirs in the frame of special studies to understand and describe accumulation, degradation and mobilisation processes, however, pollutant load calculations should be done in representative free-flow cross-sections along the river.

- When the cross-sectional heterogeneity is significant, and the load must be calculated, such as at border transect between two countries, than the water flow velocities should be measured, or calculated from available graphs, at the sampling locations for the pollutant measurement. The representative portion of the water discharge should be used for the load calculation by multiplying with the relevant concentration value.

6. DATA IN THE DANUBE CATCHMENT

6.1. AVAILABLE DATA PRECISION IN THE DANUBE CATCHMENT (LARGE SCALE)

The used and developed maps are presented in Deliverable 5.5.

6.1.1. Spatial input data

For the research project the following data were made available as geo-referenced datasets that could be integrated into the GIS. For GIS presentation of these data and the calculation results, the Lambert Equal Area Azimuthal projection was used with the central meridian 20° E and the latitude of reference 55° N.

The **river network** was taken from the *Environmental Research Systems Institute* (ESRI) "Digital Chart of the World" (1:1 Million, 1991/1992).

The **catchment boundaries** were constructed according to the position of the river monitoring stations from the Trans National Monitoring Network (TNMN, Water Quality in the Danube River Basin 1997, TNMN Yearbook) published by the *International Commission for the Protection of the Danube River* (*ICPDR*, 2000) as well as from the position of selected monitoring stations of the Danube countries. Additionally, the catchment area boundaries from digital databases from ***Hungary*** (*Institute of Water Pollution Control, VITUKI*), ***Romania*** *(Romanian Waters, National Administration*) and ***Slovenia*** (*Environmental Agency*) were used for delineation of the catchments.

The **geographical location of the monitoring stations** in the river network was derived from sources of very different content and quality. The primary information came either as stored coordinates in various geographic reference systems and/or as verbal descriptions of the location relative to water bodies and towns. In some individual cases, existing information about the location on the left or right river-bank was not considered because of the small scale of the river network map.

The **digital elevation model** (DEM) GTOPO30 from the *United States Geological Survey* (*USGS*) has a resolution of 30 arcsec (about 925 m x 570 m, resampled to 200 m x 200 m) and was used for the delineation of the catchment borders and to give an overview of the relief in the Danube river basin.

For **land use** classification, data from CORINE Land Cover (CLC) (*European Environment Agency* (EEA 1995) with a resolution of 100 m x 100 m were used, as well as data from CORINE Land Cover (CLC) (Land Cover, *European Commission*, 1996, CORINE Land Cover of Europe, *European Topic Centre on Land Cover*, Kiruna, Sweden, 1997) with a spatial resolution of 250 m x 250 m for the part of ***Switzerland***. This data includes land use data from satellite images for the years 1989-1992 for Switzerland from the *State Statistics Offices*, the PHARE-Program of the European Union and the *European Topic Centre on Land Cover*. As the information on land use is missing from CORINE for ***Croatia, FR Yugoslavia (Serbia and Montenegro), the Republic of Moldova and Ukraine***, additional information on land cover was taken from the USGS (*United States Geological Survey,* GLCC - Version 2, 1997) and used to identify land use classes in these countries according to CORINE land cover. The spatial resolution of the USGS land cover map is 1000 m x 1000 m.

The Digital **Soil Map** of the World (DSMW, FAO 1997) based on the FAO/UNESCO Soil Map of the World. The original scale of 1 : 5 000 000 was used in terms of physicochemical parameters such as soil texture, drainage class and nitrogen content in the upper soil layer.

A **hydrogeological map** of Europe from the *National Institute of Public Health and the Environment (RIVM)* was used for the differentiation of consolidated and unconsolidated rock regions within the Danube catchment area.

Several **hydrometeorological** input data digital maps were created. **Precipitation data** were obtained from interpolated distribution of (monthly values, 1998-2000, spatial resolution of one arc/degree) available from the *Global Precipitation Climatology Centre* (GPCC) of the *German Weather Service* (RUDOLF et al., 2003). **Mean annual precipitation** and **mean annual runoff** were made available by *Geodaten, Analyse & Integration* and are derived from maps at the scale 1: 200 000 published by the *Regional Co-operation of the Danube Countries (Regionale Zusammenarbeit der Donauländer)*, 1986, in "Die Donau und ihr Einzugsgebiet" part 3, Map III/3. These maps are based on meteorological and discharge data recorded mainly for the period 1931-1970. The Mean annual runoff map was used for the calculation of the specific runoff for those catchments without data on runoff.

Data on **atmospheric deposition** of nitrogen oxides and ammonium with a resolution of 50 km for the year 1999 were derived from the results of the *Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe (EMEP)* coordinated by the *Chemical Coordinating Centre (CCC) (Norwegian Institute for Air Research, NILU*) and were used for calculating the total nitrogen deposition in the investigated area.

The data input for **soil erosion** was provided by a digital map from *National Institute of Public Health and the Environment (RIVM)* (1995) based on the Universal Soil Loss Equation (USLE) with a resolution of 1 km.

The borders of the **administrative areas** (districts, regions, and countries) in the Danube Basin were available for the year 1999 from MACON, "Professional Maps and Data Sets", (1:1 Million, 1999).

For **population density** a digital map was created with the information on population figures (for the year 1999) available from the different national statistical offices (internet) and completed with the information from national statistical offices supplied by the national consultants (Austria). In addition a further map of population density was created from the LandScan 2000 Global Population Database developed by *Oak Ridge National Laboratory* (ORNL). The LandScan data set is a worldwide population database compiled on a 30" X 30" latitude/longitude grid. Census counts (at subnational level) were apportioned to each grid cell based on likelihood coefficients which are based on proximity to roads, slope, land cover, night time lights, and other data sets. The LandScan files are available via the internet in ESRI (*Environmental Research Systems Institute*) grid format.

A digital map with the location of waste water treatment plants (**WWTP)** from the WWTP Inventory 2000 of the *ICPDR* was created for the calculation of the input of point sources (municipal waste water treatment plants and industrial dischargers) in the river system of the investigated catchments. The ICPDR Inventory includes for each country only the largest point sources and about 75% of the total point source emissions into the river system of the Danube. This inventory was supplied by national inventories of ***Germany*** (*LfW Bavaria,* München *and LfU Baden-Württemberg,* Karlsruhe), ***Slovakia*** (by the *Water Research Institute,* Bratislava), **Austria**, (Federal Environmental Agency), Hungary (by the *Department of Sanitary and Environmental Engineering, Budapest University of Technology and Economics, BUTE*) and from the inventory (former study within the PHARE-Project EU/AR102A/91 (1997) "Nutrient Balances for Danube Countries") supplied by the *Institute for Water Quality and Waste Management, University of Technology Vienna*. The latter was used to supply information on WWTP ***the Czech Republic, Slovakia, Hungary, Slovenia, Bulgaria, Romania*** and the ***Ukraine*** which are not included in the inventories mentioned above.

6.1.2. Data for calculating point source emissions

For the emission from point sources data from the ICPDR Inventory together with data of the reference year 2000 were used. The inventory of municipal discharges includes emissions which total at least about 75 % of the national COD loads transported in sewers and discharged into the riverine environment, irrespective of the type of treatment. The type of treatment ranges from no treatment at all to mechanical treatment, the removal of organic carbon and up to the removal of phosphorus and nitrogen. For the Federal Republic of Yugoslavia ***(Serbia and Montenegro)*** the inventory only contains the data on municipal discharges for the year 1996. This is due to the fact that at the time of compilation of the ICPDR inventory the ***Federal Republic of Yugoslavia (Serbia and Montenegro)*** did not yet actively participate in, and contribute data to, the ICPDR. The inventory of industrial discharges includes the most relevant types of industry: food-, chemical-, pulp and paper-, fertilizer-, mining-, iron and steel-, metal surface treatment-, textile-, leather industry and large agricultural plants. In each case only the best data available were included, but inevitably the 'quality' of the data varies according to the methods used by national experts for identifying emissions from individual plants. Depending upon the country and source of emission, the values in the emission inventories of the respective countries may be based upon continuous or periodical measurements; permit values, or values estimated by some other means. Typical values included in the emission inventory of municipal discharges alongside the information regarding name of discharger, geographical location, river basin and main river are:

- raw water load (TPE)
- current treatment
- current capacity of WWTP (TPE)
- volume of wastewater discharge (Tm^3/a)
- total load discharged into receiving waters (BOD, COD, N, NH4-N, P) (t/a).

Typical values included in the emission inventory of industrial discharges alongside the information on name of the plant, geographical location, river basin, main river and sector are:

- raw water load (TPE)
- volume of wastewater discharge (Tm^3/a)

- discharged pollutants in t/a (e.g. COD, BOD5, NH4-N, NO2-N, NO3-N, TN, PO4-P, TP, SS) (t/a).

In addition to the ICPDR inventory database the national WWTP inventory of ***Bavaria and Baden-Württemberg*** and Austria was used to calculate the total nutrient discharges by point sources for the ***German and Austrian part of the Danube***. For the recorded WWTP similar information and typical values to those of the ICPDR are included in the database, such as:

- Plant capacity as inhabitant equivalents
- Treated wastewater volume per year
- Nitrogen parameters (concentration, yearly load)
- Phosphorus parameters (concentration, yearly load)

Additional information on waste water treatment plants (WWTP) (164 locations, values for the year 2000) was also supplied by national consultants for ***Slovakia*** (*Water Research Institute*, Bratislava), as well as information on major direct industrial discharger (1996 to 2000, summarized for major subbasins in Slovakia). For ***Hungary*** information on 495 WWTP was supplied by *Department of Sanitary and Environmental Engineering, Budapest University of Technology and Economic (BUTE*). For the other countries ***(Czech Republic, Slovenia, Bulgaria, Moldava, Ukraine)*** additional information was made available by the national consultant for Austria (*Institute for Water Quality and Waste Management, University of Technology Vienna)* from former studies within the framework of the PHARE Project EU/AR102A/91 (1997) “Nutrient Balances for Danube Countries”. Calculation of the missing point source discharges was carried out as described in chapter 3.1.1. The loads for nitrogen and phosphorus taken into account for the respective Danube country are totalled (see Deliverable 5.5, table 2.3).

6.1.3. Monitoring data for surface water

The **water quality** database from the *Trans National Monitoring Network* (*TNMN*) established by the *ICPDR* (*International Commission for the Protection of the Danube River*) comprises concentration and discharge values for the period 1996-2000. Under the Danube River Protection Convention (DRPC), the *Monitoring, Laboratory and Information Management Expert Group* (MLIM/EG) is “operating” the *Trans National Monitoring Network* for water quality in the Danube River Basin (see chapter 5).

When the project started within the structure of the TNMN 61 sampling stations in total were available selected from the national monitoring networks and based on criteria and objectives agreed between the countries. Of these, 31 were situated on the Danube river and 30 on the tributaries of the Danube. For the years 1996-1997 the data on discharge and river load varied between bimonthly and monthly values for the different stations, while for some stations data is even more irregular. The available values per year therefore vary from 3 to 12 for this time period. Starting from 1998-2000 daily discharge values were available from the TNMN database, and biweekly to monthly values for the river load (with exceptions of some stations, e.g. in Bosnia-Herzegovina). For the calculation of the river loads the period of 1998-2000 was considered. Investigated pollutants which were determined according to TNMN are Ammonia (NH_4), Nitrite as Nitrogen (NO_2), Nitrate as Nitrogen (NO_3), Phosphates (PO_4), Total

Phosphorus (TP) and other pollutants were not considered for the calculations. Data on suspended solids (SS), temperature, DOC, TOC were also available from this database.

For this study, a total of 35 monitoring locations in the ***German Danube catchment*** area were chosen including the 4 stations also monitored in the TNMN (see Deliverable 5.5, Table 2.4). If the discharge was measured at another station from the water quality, then some conversion is necessary. To calculate the discharge at the water quality station the flow at the discharge monitoring station is multiplied with a conversion factor (see Table 2.4, Deliverable 5.5). If the conversion factor is not known, it was determined from the relationship of the catchment areas of water quality monitoring and discharge monitoring stations. This conversion procedure was also applied for the flows at the water quality monitoring stations of the other Danube countries according to the catchment areas determined by GIS (Geographical Information System) and the data supplied by the national consultants. The calculations for this study are based on the data for the period 1998 – 2000.

For the ***Austrian part of the Danube catchment*** another 49 monitoring stations were chosen for this study in addition to the 4 stations in the TNMN. An overview of the stations and the available data of discharge and water quality is given in Deliverable 5.5, Table 2.5. Bimonthly measurements of nutrient concentration (Ammonia (NH4), Nitrite as Nitrogen (NO2), Nitrate as Nitrogen (NO3), Phosphates (PO4), Total Phosphorus (TP)), suspended solids (SS) and temperature were (basically) available from June 1993. Monthly values for the same parameters were available from 1996 until June/July 2001.

Daily discharge data were available for 21 stations for the time period 1971-1999. Of the 35 stations selected within this study for the ***Hungarian part of the Danube catchment*****,** data on water quality and discharge were provided for 30 monitoring stations by the *Institute of Water Pollution Control* at *VITUKI* in Budapest. The data were monitored weekly to biweekly (nutrients), runoff, temperature, suspended solids, chlorophyll (as well as other parameters) and are available for the period 1995-2000. An overview of the stations and the available data of discharge and quality data is shown in Deliverable 5.5, Table 2.6.

For 14 monitoring stations of the 16 stations selected for the ***Slovenian part of the Danube catchment***, data on water quality and discharge were provided by the national consultant**.** The data were monitored for some stations bimonthly in major parts irregularly (nutrients). Daily discharge values were available from runoff monitoring stations for the period 1994-1998 and for some of the stations also for the year 2000 and 2001. An overview of the stations and the available data of discharge and water quality measurements is shown in Deliverable 5.5, Table 2.7.

For this study a total of 46 monitoring stations were chosen for the ***Romanian part of the Danube catchment***. In the TNMN 11 monitoring stations are included. For the remaining 35 stations selected additionally from the national monitoring network, daily discharge values were available for the period 1994-1999 as well as monthly values on nutrient concentrations, temperature, and in some parts on suspended solids for the same period. Table 2.8 in Deliverable 5.5 gives an overview of water quality and discharge data in the Romanian part of the Danube river which could be used.

6.1.4. Administrative data

Administrative data were collected at the municipality or district level. With the help of GIS datasets of the administrative units, this information was used in the GIS on an area basis and could be aggregated for the various catchment areas. Data on **population**, **land use**, **cultivation**, and **livestock** numbers for municipalities or districts for the year 1999 were available in tabular form. Data were supplied by the national consultants as well as by the IGB from the information of different statistical offices via the internet.

6.1.5. Agricultural data

The top soil **nutrient surplus** at the agricultural area for the ***German part*** of the Danube Basin has been taken for the period 1950-1999 from BEHRENDT et al. (2002).

For the ***Czech part***, the nutrient surplus was supplied for the year 2000 on a district basis by the *Water Research Institute, branch BRNO.*

For **Austria*, Slovakia, Hungary and Romania*** the nutrient surplus was calculated on a district level for 1999 by IGB according to the OECD methodology (OECD, 1997) and based on the statistical data on district level provided by the national consultants of the respective countries.

For the other Danube countries ***(Slovakia, Slovenia, Croatia, Bosnia-Herzegovina, Yugoslavia, Bulgaria, Moldavia and the Ukraine)*** the nutrient surplus at the agricultural area was calculated on a country basis for the period 1961-2000 according to the OECD methodology (OECD, 1997) based on data from FAO statistics (*FAOStat 98 – 1961-1998* and for the period 1999-2000 based on data available from statistics of the FAO homepage via internet: http://apps.fao.org/cgi-bin/nphdb.pl?subset=agriculture).

Information on **tile drainage** was made available from different sources. Partly the national consultants provided the information on the share of tile drainage at total land in agricultural use by administrative units (different levels) or as major basin wide figures. Only for ***Slovenia*** the exact locations as well as the extent of tile drained areas were available. For those countries where such information was missing, the percentage of tile drained areas was estimated on the basis of the FAO soil map and figures given in this database on drainage capacity for the different soils. Based on the figures from different sources the percentage of tile drained areas on the total land in agricultural use was aggregated for each investigated catchment as an area weighted mean according to CLC and the catchment boundaries.

6.2. EVALUATION OF KEY DATA FOR NUTRIENT BALANCES

Nutrient surplus in Topsoils

The surplus on agricultural soils can be quantified based on a soil surface balance (see chapter 4.2 or 5.5. of deliverable D1.3; Austrian part). More advantageous for evaluation of scenarios with respect to changes of the agricultural productivity, is the “farm gate balance” (see chapter 4.1). In general, the accuracy of these calculations is appropriate. Nevertheless, it has to be stated, that the OECD method for the soil surface balance does not consider a net-mineralization. Especially in regions with a sudden drop of the production intensity net mineralization can significantly influence on the nitrogen available for leakage to groundwater on agricultural soils and can not be neglected anymore as it has been shown in the Hungarian case study investigations (see chapter 5.4. of deliverable D1.3; Hungarian part). The MONERIS approach works with long term averages of the nitrogen surplus and thus, covers the effect of time delay of effects of changes in management practice on nitrogen emissions to some extent, without taking net-mineralization into account explicitly (D1.4).

Groundwater

For further development of this submodel it seems to be necessary that statistical data for agriculture are available for, at least, the district level of the countries. If this data is available for the whole Danube basin the model can be changed or calibrated to reduce the deviation between observed and calculated concentrations. Further, it would be very useful if more of the results of measurements in the smaller rivers of the Danube basin were available. This is also important because the nitrate concentrations in rivers at low flow and in winter time can be helpful in indicating possible long term changes of the N-inputs via groundwater.

On a smaller scale even more important for assessment of nitrogen emissions via groundwater as the surplus in soils is the quantification of the retention (denitrification) of nitrogen in soils and groundwater. In respect to emissions to surface water in comparison between regions the denitrification in groundwater may overrule the influence of the surplus in soils. The MONERIS retention approach works well on subcatchment level, but it shows a high sensitivity on the categorisation of geological units (consolidated-impermeable to unconsolidated-deep groundwater). Appropriate information on geology and categorisation according the MONERIS definitions therefore is prerequisite for a successful application.

Erosion

The data input for soil erosion was provided by a digital map from *National Institute of PublicHealth and the Environment (RIVM)* (1995) with a resolution of 1 km. (Figure 6.1).

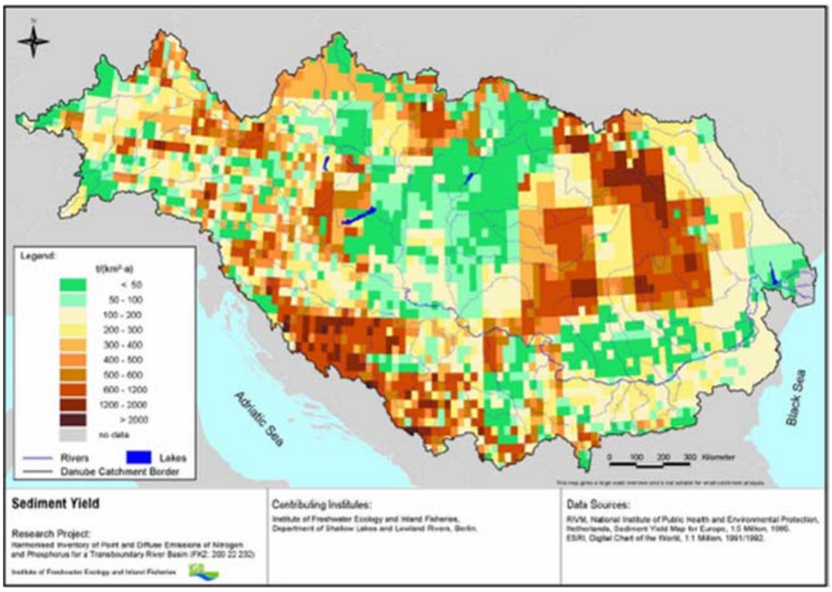


Figure 6.1: Sediment Yield in the Danube catchment.

The estimated emissions of nutrients into surface waters by erosion are strongly dependent on the quality of the soil loss map used as the starting point for the calculation. The used map (figure 6.1) is based on the application of the Universal Soil Loss Equation (USLE) for grid cells of 5 km. For the calculation of the nutrient inputs into the river system of the Danube the approach used in MONERIS (BEHRENDT et al., 2000) was applied in relation to the sediment delivery ratio (SDR) and the enrichment ratio (ER).

It should be possible to derive a better soil loss map, but for that the existing digital soil map of Europe in a scale of 1:1 Million prepared by the Soil Bureau of the JRC in Ispra and more detailed information on the grown crops (C-factor of the USLE) on the district or municipal level are necessary.

Soil data

The spatial resolution of the soil map determines the accuracy of the results, because soil erodibility effects detachment processes a lot. Therefore the scale of the European soil map (ESB, 1998) is too large and it is not suitable for an application and modelling of soil erosion within the case study watersheds. However for simulation of sediment delivery in the entire Danube River catchment the European soil map may be sufficient.

To calculate soil erosion especially soil texture is important, because texture is the main factor influencing erodibility (see also results on the calculation of K factors in Deliverable 2.1). All soil parameters in the MMF model (soil detachability, bulk density, field capacity, cohesion, effective hydraulic depth) connected to soil can also be generated in some way out of soil texture. For any model application therefore texture has to be identified as good as possible in spatial and quality resolution. For our work, the Austrian soil mapping system at a scale of 1:25.000 proved to be sufficient. Nationally available data in other countries may do as well (see the Hungarian case study in Deliverable D2.1). However, a unified approach to obtain the necessary input data would be highly desirable. For sediment and nutrient estimation in the Danubs River the data of the European Soil Thematic Strategy map is to coarse, more detailed information would be desirable.

Only available data on soil texture for entire Europe is the European Soil Map. This map proposes a unique methodology for all European States for collecting the same basic data on soil characteristics in the scale of 1:1.000000. However this map might be to coarse. Due to the fact the final competition on the Soil Geographical Database of Europe 1:250.000 (Dudal, R. et al., 1993) is still not foreseeable, more detailed information on texture is necessary on a national level. Examples of data available at national scale (1. 25000; Austria, Hungary) show, that type and amount of information is much better compared to the European soil map. It is however an enormous task to compile this information. In consequence chemical information of soil types are not available on a appropriate scale too. To give an example a lot of soil chemical data are available for a grid of 4km x 4km covering the whole Austrian territory with the exception of total P. Various algorithms has been developed to obtain the needed values from other sources of information, but this is not really satisfactory. In addition some information on the dissolved fraction of phosphorus in soil is necessary. Various monitoring points all over Europe exist, but the measuring procedures and consequently the measured phosphorus values differ. It would be desirable to bring these measurements to an unique level.

However, as the different procedures to obtain plant available (as a surrogate for dissolved) phosphorus is linked to national tests for optimisation of fertilisation, no such harmonisation is to be expected (see results of COST action 832). Therefore a methodology to link the different analytical methods to amounts of dissolved phosphorus in surface runoff has to be developed.

Chemical soil properties

To calculate phosphorus loads the total amount of total phosphor for the topsoil is needed. The application of the simulation models in Austria indicated the mayor problem in the availability of phosphor data. As the model uses the amount of total phosphor this information is mostly not accessible. In the national soil data basis contains different P forms except total P. That might well be so in all other European Counties. Therefore we developed a methodology (D2.1) to calculate total P out of CAL P.

Availability of data on total nitrogen contents in topsoil is usually better due to the high importance of nitrogen for plant growth. However, nitrogen loading due to soil erosion is relatively unimportant.

Landuse data

Due to the fact that landuse is one of the main factors which influence the distribution of elements in the landscape, the spatial resolution and the content of landuse information is highly important and in consequence specifies the spatial resolution of a model. The only information of landuse available on a European level is the CORINE dataset. In this project the methodology of collecting the landcover data was the same for all European countries. Therefore the maps can be used as basis for land cover. To improve the existing spatial and temporal resolution of landuse data for arable land, the CORINE data may be combined with agricultural surveys (see Figure 6.2).

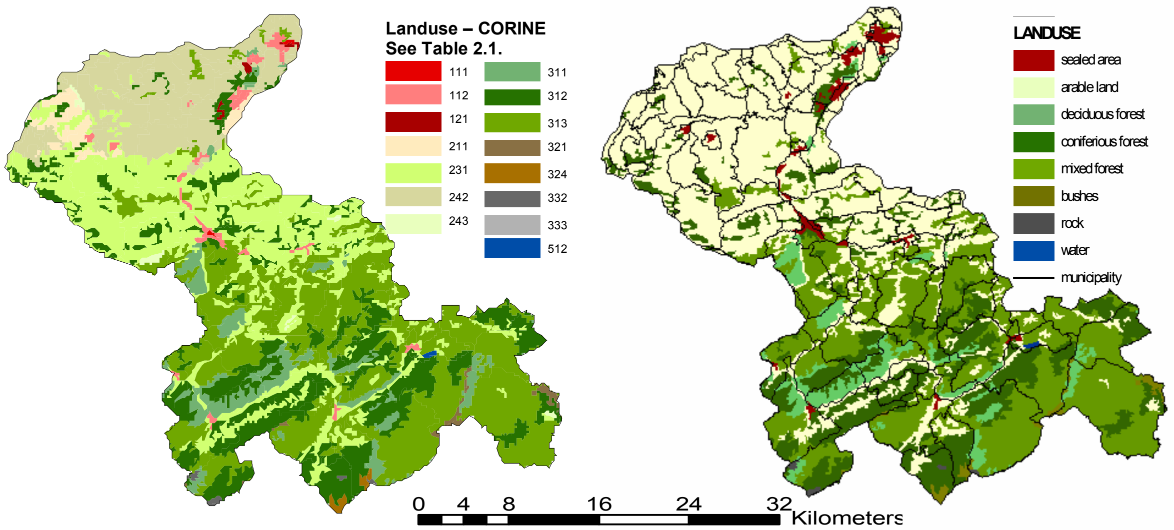


Figure 6.2: Comparison of landuse map CORINE and new map including statistical data for the Ybbs watershed.

Class two of the CORINE classification scheme has to be divided into crop species with different influence on soil erosion at least into 4 groups (erosive plants, non erosive plants, vine and orchard and grassland). Having obtained the information about specific landuse, the necessary parameter values for the input data of MMF (land cover, ground cover, plant height, ratio of actual to potential evapotranspiration, C-factor, rainfall interception) and USLE (C-factor) may be calculated using transfer functions.

As already mentioned in the previous chapter the only spatial information of landuse available on a European level is the CORINE dataset. But the resolution of this data set as well as the big uncertainty included in the maps because of map errors and a improper classification of landuse, CORINE data alone are not suitable for the estimation of soil erosion as proposed in our approach. Furthermore a serious drawback is that arable land is not portioned sufficiently in different landuse management groups, thus more information on landuse is necessary. To overcome some of the problems a combination of the CORINE dataset and national agricultural surveys is proposed. A similar approach has already been done for Germany (Erhard et al., 2004). Agricultural surveys or statistics are available in all Danube countries. However the problem in using them is that surveys in the different European countries are based on different principles, of spatial resolutions and accuracy. Table 6.1 tries to give an overview about the available Agricultural Surveys in the different Danube states, their spatial resolution and their applicability. In order to use them they have to be brought on a unique level.

Table 6.1: Availability of Agricultural Surveys in the Danube Countries.

Country	Agricultural Survey	Resolution	Applicability
Albania	FAO statistics	-	-
Austria	Austrian Agricultural-statistical Survey, 1999	municipality	+
Bosnia Herzegovina	Yugoslav Daily Survey 1998	~	-
Bulgaria	Agricultural Census 2003	~	~
Croatia	Agricultural Census 2003 Yugoslav Daily Survey 1998	municipality	-
Czech Republic	Agricultural Census, 2000	region	+
Germany	Land Survey, StBA 1997	municipality	+
Hungary	Agricultural Census, 2000; Fruit tree and vineyard basic survey, 2001	municipality	+
Italy			
Yugoslavia	Yugoslav Daily Survey 1998	~	-
Macedonia	FAO statistics	-	-
Moldavia	no	-	-
Poland	Agricultural Census 2002	+	-
Romania	General Agricultural Census, 2003	~	~
Serbia	Yugoslav Daily Survey 1998	~	-
Slovak Republic	Farm structure Census, 2002; Fruit and wine statistic, 2002	municipality	+
Slovenia	Agricultural Census, 2000	~	-
Switzerland	Agrarstatistik 2001		
Ukraine	Agricultural Survey, 2003	~	+

+ quality of information is very good, for either the classification of different crop groups and the spatial resolution at least at municipality level

- spatial resolution is too small, data availability only per region or classification of crop groups is too small

~ provided information on the survey is not clear

Nevertheless there is still an important gap of information due to grassland. In CORINE data the definition and the delimitation of grassland is rough. One possibility to obtain identical coverage information would be the usage of remote sensing technologies, probably combined with other ground based information. However, at present there is no such known activity.

Climate data

Precipitation represents the driving force of the water and energy circle and should be used with caution. For application of the MMF model monthly rainfall data are necessary. In our study, they have been taken from available gauging stations. For areas with rapid changes in climatic conditions (such as the Ybbs river catchment) the spatial resolution of climatic data certainly plays a much higher role in calculation of soil erosion but to give a recommendation on net density is difficult. A possibility to enter changes in precipitation due to changes in elevation (a precipitation-laps-factor) is given in Deliverable 1.1. In addition to monthly precipitation, mean rainfall amount per rainy day and typical rainfall intensities are needed for MMF. For application of the USLE functions are available to relate mean rainfall amounts to rainfall erosivity.

All over Europe there is quite a dense net of climatic stations, therefore the applicability of this data to sediment estimations of the Danube Basin in general is deemed good. However, additional measurements on rainfall intensity is more loose, but still in an accurate resolution for applicability to Danube Basin calculations for soil erosion. The main problem again is to unify the data obtained at national level for a European wide application

Surface runoff (mm)

For using the sediment delivery concept developed within the framework of this deliverable concentration of sediments in the surface runoff are used. Therefore the amount of surface runoff in a particular watershed is needed. Different techniques exist to separate the different flow components (Deliverable D1.1) which may be used to estimate the amount of surface runoff out of the total river discharge. With regard to a quite dense system of river discharge measuring points in the Danube River this should not be a problem. The accuracy of the separation techniques is to be discussed elsewhere (see deliverable 1.1).

Hydrological data are sufficiently available in the Danube basin. There is quite a dense net of gauging station measuring water discharge of the Danube River and its tributaries. At 94 stations alongside the Danube River water discharge is measured. However it would be desirable for calibration propose to have additional data on sediment yield, phosphorus and nitrate loads in the Danube River at the tributary outlets. At present flow proportional sediment and nutrient data, which are essential for estimation of total loads are rare. The Austrian water quality ordinance (WGEV, BGBl. Nr. 338/1991) for instance requires water quality sampling at two months intervals. This is not sufficient to catch amounts of sediment and phosphorus which mainly leave the watersheds during high flow events.

Slope

To simulate soil erosion in a GIS, spatial information of landform and relief energy is necessary. The proposed algorithm to overcome errors that occur due to spatial inadequate accuracy is trying to handle the problem that for all adjacent states of the Danube River only a digital elevation model with grid resolution of 1000 m is available.

At the time of writing Deliverable D2.2, the only digital elevation model available at European scale was with a spatial resolution of 1000m. Without correcting for slope, application of this grid leads to huge differences in the evaluation of actual soil erosion risk. Using the proposed methodology to correct for these differences is making the best of available data. However, still considerable errors apply. The more detailed a DEM the better the results of the models, therefore a better resolution on a European DEM would be highly desirable to reduce errors associated with slope estimation. For an additional including of routing processes for water or sediments, a spatial resolution of the digital elevation model of at least 50m is required. Results of model application have shown, however, that this is not a prerequisite to obtain good model results. Therefore, smaller resolutions of DEM's may be sufficient as well.

The potential slope length (Emmett, 1978) which is based on the drainage density of a watershed is used as a surrogate for the flow lengths of water within a landscape in the calculation of sediment delivery ratios. As potential slope length is the basis for sediment yield calculation the accuracy of river length and the watershed size has to be high.

For calculation of the potential slope length per subbasin a precise length of the river and the size of the watershed, wherein the sediment yield is calculated, is needed. To define the size of the watershed a digital elevation model is necessary. As already discussed above, this is only available at a coarse scale of 1:1.000000 and therefore not suitable for watershed calculations. National DEM's usually are available at finer scales but as river catchments do not match with country borders there is a big problem in connecting DEM's of different countries with different spatial resolution (unification of available data).

Point sources

For quantification in many cases specific data from the different points sources can be used. If this is not the case, emissions can be estimated based on information on inhabitants connected to sewer systems or population equivalents discharged to waste water, specific nitrogen emissions to waste water and typical removal rates of different treatment levels as it is presented in chapter 4.5. Accuracy of determination of point source emission in general is the highest as compared to other emission pathway. Estimations on specific values should be used to check existing data based on measurements. Based on this check it is possible to detect erroneous data.

Urban areas

Based on head specific nutrient emissions it is possible to estimate the amount of nutrients in the produced waste water relatively accurate. The problem is that usually it is not known where the waste waters from septic tanks and pits go to. Estimations based on expert knowledge have

to be made. Usually most of it is leaking to the underground and groundwater and only a small part is reused in agriculture or transported to treatment plants by lorry.
The main factor controlling the emissions to the surface waters is the retention of nutrients in the leakage water in the underground and groundwater. A systematic investigation on the relation between amount of retention and the hydro-geologic situation in a region does not exist yet. Usually it is assumed, that this retention as average value is higher than 50 %. As investigation within the Hungarian case study region show this retention might be even much higher (60 – 95 % for nitrogen and > 95 % for phosphorus) in specific cases (see Deliverable D1.3, Hungarian part, chapter 4.6). This explains why on site disposal of waste water with leakage to the underground is a relatively effective "measure" in respect to nutrient discharges to surface waters. But of course it is a relevant factor for groundwater pollution (e.g. hygienic aspects).

6.3. COMPARISON OF DATA USED ON LARGE AND MESO SCALE

Table 6.2 provides an overview of data used for MONERIS emission modelling on large and on meso scale, which can be interpreted as **minimum data requirement list**. In a further step a model result comparison concerning emission modelling for the case study areas (large scale parameterisation and meso scale parameterization) should outline the importance of input data using a concrete example.

The main aspects will be:
- Comparison of the results concerning average annual loads
- Comparison of the results concerning main pathways
- Comparison of the used data

Table 6.2: MONERIS input data; large scale and meso scale application.

Maps	Large scale				Meso scale			
Data	**Avail able**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**	**Avail lable**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**
River net	yes	ESRI digital	1991/ 1992	1:1 Mio	yes	digital maps		1: 50000
Catchment boundaries	yes	TNMN digital	1997		yes	digital maps		1: 50000
Administrative boundaries	yes	USGS digital			yes	digital maps		1: 50000
DEM	yes	USGS digital		200m x 200m	yes	digital maps		25m * 50m
Topographic	yes				yes	digital maps		1: 50000
Land use	yes	CORINE, USGS digital		100m x 100m/ 250m x 250m/ 1000 x 1000m	yes	digital maps	actual	At least 1: 100000
Hydrogeology, Geomorphology	yes	RIVM		Map of Europe	yes	digital maps		1: 50000
Soil types	yes	FAO, UNESCO	1997	1. 5 Mio	yes	digital maps		1:25000

Drained areas	partially		1997	1. 5 Mio	partially	digital maps		different
Eroded areas (soil loss map)	yes	RIVM	1995	1000x 1000 m	no			
Location of monitoring stations	yes			different maps	yes	digital maps	actual	
Locations of municipal and industrial discharges	yes		2000/ 1997		yes	digital maps	actual	
Statistical data	**Large scale**				**Meso scale**			
Data	**Avail able**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**	**Avail able**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**
Crop statistics (area, yield)	yes				yes	digital data	since 1960	municipality
Fodder production	no	factor			yes	digital data	since 1960	county
Fodder consumption					no	estimates		
Livestock no. or animal units	yes		1999	municipality /district	yes	digital data	since 1960	municipality
Manure production and use (application, direct discharges, treatment plant)					no	estimates		
Mineral fertilizer application					Part.	digital data	since 1945	county/federal state
Food production (meat, milk, eggs and non-animal food)					yes	digital data	since 1960	county
Population	yes		1999	country	yes		since 1960	municipality settlement
Food consumption	no				yes		since 1960	county
Use of detergents (washing powder, dish washing etc.)	yes			country	yes			county
Spec. P and N emissions to sewer systems (based on pop. Or pop. Equivalent)	yes			country	no			
Application of sewage sludge	no				yes		since 1990-	municipality
Waste water statistics	**Large scale**				**Meso scale**			

Data	Avail able	Source/ Format	Time span/ Actu-ality	resolution	Avail able	Source/ Format	Time span/ Actu-ality	resolution
Pop. Connected to sewer systems	part.		2000	country	yes	analog/ digital data	since 1971	municipality
Pop. Connected to WWTP`s	part.		2000	country	yes	analog data	Actual	municipality
Population connected to septic tanks and pits					yes	analog data	Since 1971	municipality
Inventory of point dis-charges	**Large scale**				**Meso scale**			
Data	**Avail able**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**	**Avail able**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**
Location	part.		2000	country	yes	analog data	actual	
Capacity of WWTP	part.		2000	country	yes	analog data	actual	
Actual loading	part.		2000	country	yes	digital data	actual	2 to 5 d /week
Population connected	part.		2000	country	yes	analog data	actual	
Treatment stages	part.		2000	country	yes	analog data	actual	
Inflow and effluent loads (discharge Q, N,P, org. carbon)	part.		2000	country	yes	digital data	actual	2 to 5 d /week
Monitoring data	**Large scale**				**Meso scale**			
Data	**Avail able**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**	**Avail able**	**Source/ Format**	**Time span/ Actu-ality**	**resolution**
River discharge	yes		1998-2000		yes	digital data		hourly
Groundwater level					yes	digital data		daily
Precipitation	yes		1998-2000	1. 200 000	yes	digital data		hourly
Air temperature	yes		1998-2000	1. 200 000	yes	digital data		
Conc. of substances in rivers	yes		1998-2000	biweek-ly/monthly	yes	digital data	since 1991	daily/ monthly
Conc. of N and P in drainage water	no				no			
Conc. of					yes	digital	since	three

substances in groundwater						data	1991	monthly
Conc. of N, P and silica in topsoil	yes		1997	1. 5 Mio	yes	analog data		4 km grid
N deposition	yes		1999	50000 x 50000m	yes	analog data	1986	unregular
P deposition	yes		1999	50000 x 50000m	no			
N+P+silica content in detergents: cleaning processes (washing powder, dish washing)					yes		actual	country
N emissions by traffic, energy supply, room heating etc.	no				yes		1986/ actual	country

Comparison of Model results on Large scale and Meso scale

Comparison of average annual emissions

Figure 6.3 expresses that the Wulka and Lonyai catchment can be compared on large scale and meso scale with respect to their catchment size (deviations < 6 %), while model results in the Zala catchment are not comparable, due to high differences in catchment sizes. The much larger catchment size used on large scale application (2690 km^2 compared to 1529 km^2 on meso scale application) is a consequence of Kiss Balaton Wetland region being excluded in the meso scale modelling approach. Because of the great physical and anthropogenic heterogeneities of the areas not included in the meso scale calculations even a comparison on ha weighted emissions is not suitable. Therefore, only results from the Wulka, Ybbs and Lonyai catchment will be compared. Nevertheless, even the Ybbs catchment area used for meso scale application is 17 % smaller than the area assumed for large scale applications caused by the use of a more upstream WGEV gauging station as catchment border in the large scale calculations. In the case of the Ybbs catchment a ha-weighted approach will prepare more accurate results.

Total annual phosphorus emissions in the Wulka catchment based on meso scale data pool result in lower emissions (57 %) compared to results using the large scale data pool, while it is the other way round in the Lonyai catchment with higher TP emissions calculated in the meso scale approach (146 % of large scale TP emissions). In the Ybbs catchment the calculated TP emissions are both 62 t/a by using the different data pools. Taking into account the area related differences the large scale application results in annual TP emissions of 0.46 kg/ha a and the meso scale application in TP emissions of 0.56 kg/ha a, a deviation of 17 %.

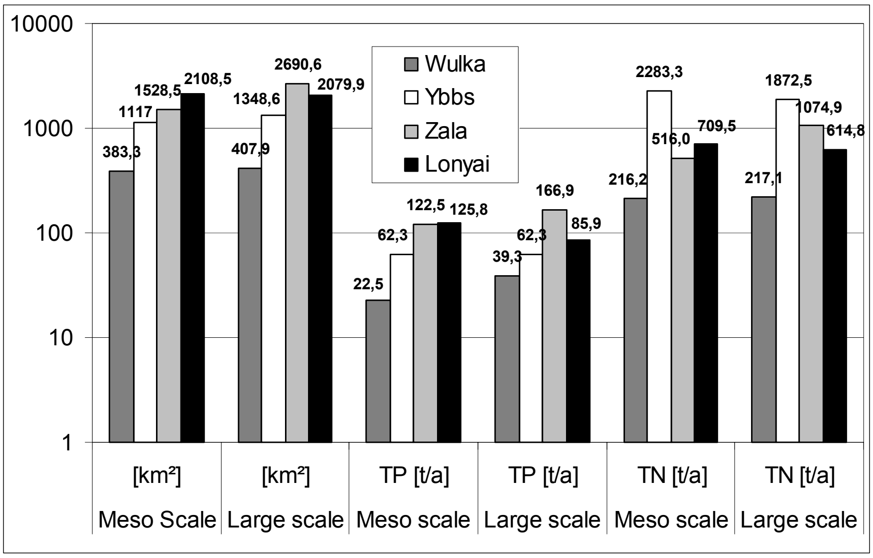


Figure 6.3: MONERIS annual emission results (TP and TN) of four different catchments using different data pools (Large scale and Meso scale), including catchment sizes; (logarithmic scale).

The total annual nitrogen emissions in the Wulka catchment calculated with the different data pools differ only in a negligible range (216 t/a meso scale, 217 t/a large scale). Even in the Lonyai catchment the calculated annual TN emissions vary only by 13 %, with higher emissions calculated using the meso scale data pool. Calculated annual TN loads in the Ybbs catchment are 18 % higher using the meso scale data pool. Taking into account the area related differences the large scale application results in annual TN emissions of 13.9 kg/ha a and the meso scale application in TN emissions of 20.3 kg/ha a, a deviation of 31 %.

Generally spoken, deviations in TP emissions based on calculations from meso scale data pool and large scale data pool are much higher (17 - 46 %) than the deviations found for TN emissions (< 31 %).

Comparison of main emission pathways

Figure 6.4 outlines the amounts of annual TP pathway related emissions calculated with different data pools (meso scale, large scale). From both calculations erosion and WWTP`s can be concluded to be the most important pathways for TP emissions.

In the Wulka and the Lonyai catchment differences in erosion emission calculations are most significant using meso scale and large scale data pools. Erosion in the Lonyai catchment in the large scale application is set to zero, while on basis of meso scale input data an annual emission of 38 t TP/a is calculated. Even TP emissions from WWTP`s vary significantly in the Lonyai catchment with higher emissions calculated in the meso scale application. In Wulka catchment the TP emissions calculated on basis of large scale data are higher than the emissions calculated with meso scale data in general. As stated above the deviations in erosion emission calculations are most significant, with large scale emission calculations extent meso scale emission calculations approximately by factor 2.

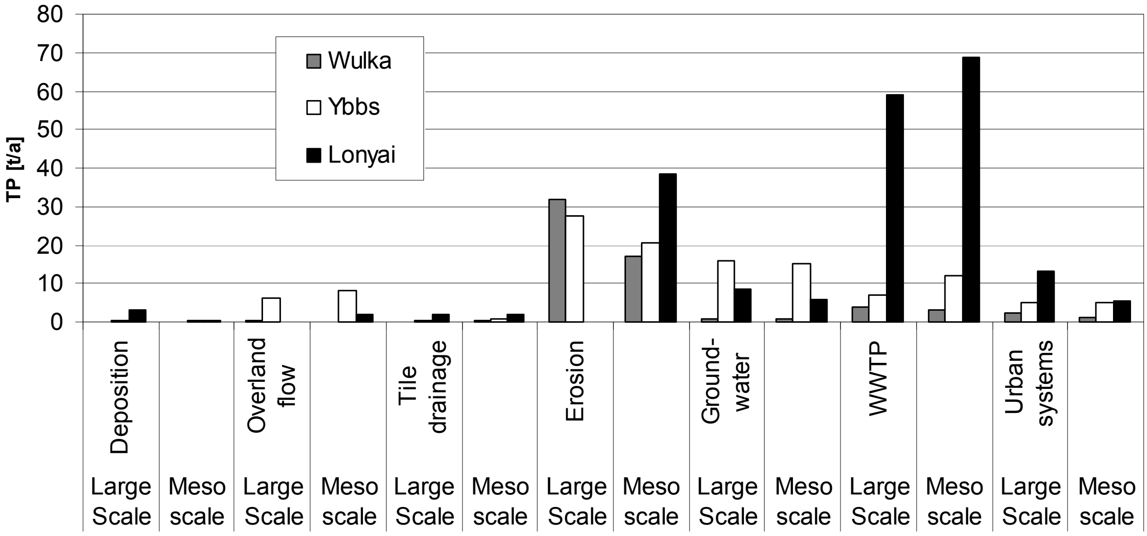


Figure 6.4: Comparison of TP emission pathways (t/a) in different catchments (Wulka, Ybbs and Lonyai) calculated with different data (large scale application, meso scale application).

In the Ybbs catchment beneath TP emissions from erosion and WWTP`s, groundwater TP emissions are important additionally (Figure 6.5). These results are reproduced in applications, large scale and meso scale, either. Nevertheless, TP emissions calculated on basis of meso scale data are higher with the exception of emissions calculated for erosion. The most significant deviation can be found at TP emission calculations from WWTP`s.

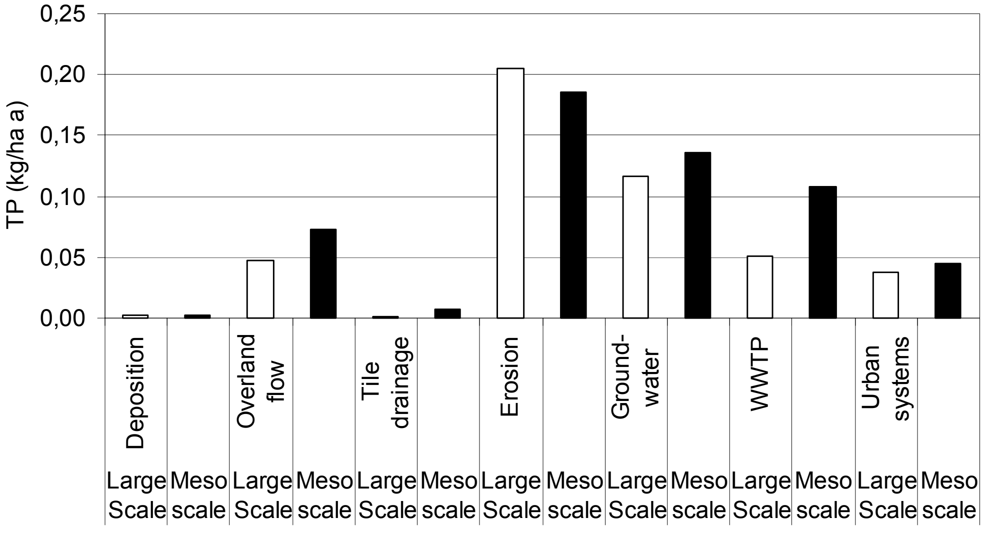


Figure 6.5: Comparison of TP emission pathways (kg/ha a) in Ybbs catchment calculated with different data pools (large scale application, meso scale application).

Summarized it can be concluded that results from TP emission calculations on both data pools (large scale, meso scale) tend to be comparable with respect to account for the main pathways in the different catchments. Only in the Lonyai catchment the lack of erosion emissions in large scale calculations differs from this trend. Nevertheless, total numbers of main emission pathway calculations differ significantly. It is obvious that the deviations in the main pathways (especially emissions from erosion but also from WWTP`s) using the different data pools lead to the significant deviations of annual TP emission calculations. Summarized the hot spots causing the most significant deviations in the catchments TP emission calculations using different data pools are:

erosion → Wulka and Lonyai (deviations in TP annual emissions ~ 40 %),

WWTP`s → Ybbs (deviation in TP annual emissions < 15 %).

TN emission calculations in the Wulka catchment show a distinct analogy with marginal higher loads calculated at large scale application in general (figure 6.6). Only TN emissions from groundwater show a contrary trend with significant higher TN emissions calculated using the meso scale application (factor 2). These parallel feed deviations lead to a good analogy of the annual TN emissions calculated with large scale and meso scale application in the Wulka catchment (figure 6.3). Furthermore, both applications reproduce groundwater, tile drainage and WWTP´s to be the major pathways for TN emissions in the Wulka catchment. In the Lonyai catchment major emissions stem from WWTP`s (large scale and meso scale application). However in the meso scale applications results exceed the large scale application results by factor 2. TN calculations from urban systems, tile drainage and deposition in the large scale application result in emissions of more or less 100 t/a each. Using the meso scale application these pathways are negligible. In the Ybbs catchment both applications account groundwater for being the most important pathway of TN emissions. In accordance to results from the Wulka catchment the meso scale application in the Ybbs catchment calculates significantly higher TN groundwater emission compared to the large scale application. While the large scale application points out overland flow to be of some importance for TN emissions in the Ybbs catchment the meso scale data result in erosion being also an important emission pathway, with factor 0.1 compared to groundwater emissions, however.

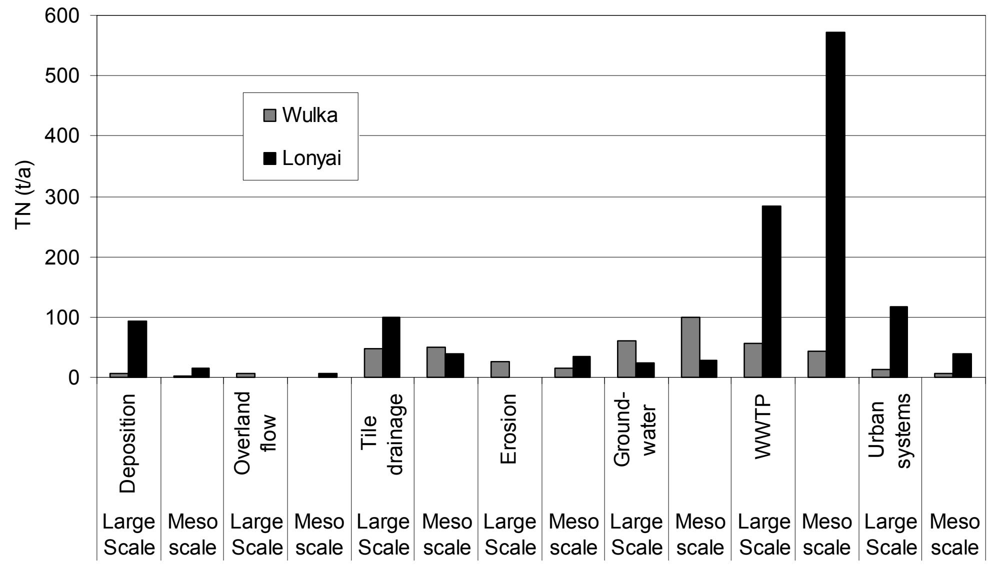


Figure 6.6: Comparison of TN emission pathways (t/a) in different catchments (Wulka and Lonyai) calculated with different data (large scale application, meso scale application).

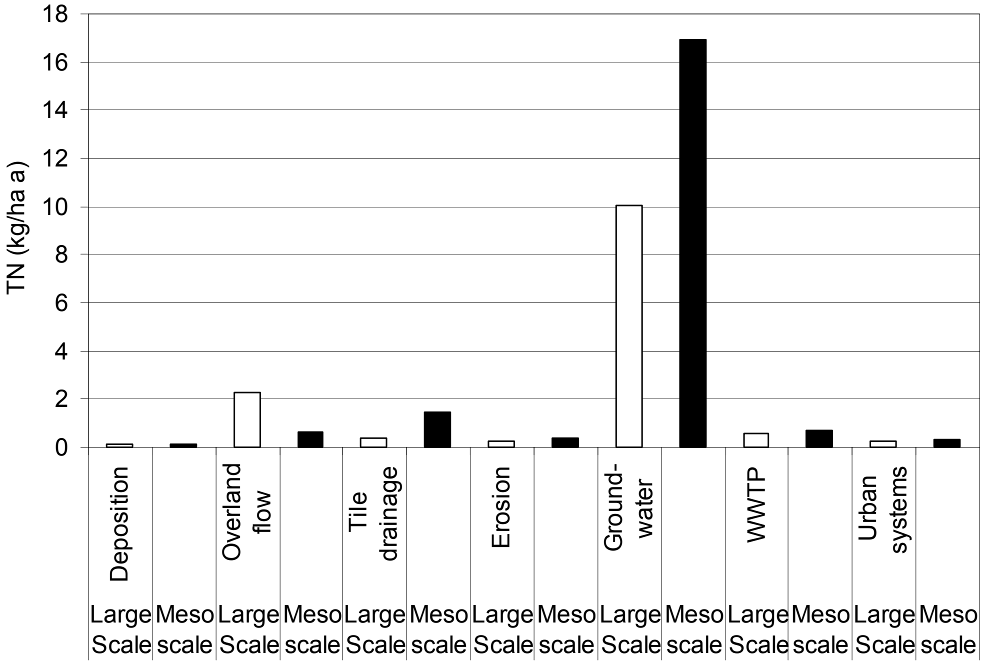


Figure 6.7: Comparison of TN emission pathways (kg/ha a) in Ybbs catchment calculated with different data pools (large scale application, meso scale application).

Results from the comparison of the major TN emission pathways calculated with different data pools (large scale application, meso scale application) express the main pathways to be reproduced by both data sets. Nevertheless, results from meso scale application are much higher concerning the dominant pathways with respect to emissions from WWTP`s (Lonyai) and emissions from groundwater (Wulka and Ybbs). Comparing the TN emission pathways of secondary importance a great variation must be stated with respect to meso scale data application and large scale data application. It is obvious that the deviations in the main pathways (especially emissions from groundwater but also from WWTP`s in Lonyai catchment) using the different data pools cause deviations of annual TN emission calculations. Summarized the hot spots causing the most significant deviations in the catchments TN emission calculations using different data pools are:

Groundwater → Ybbs, Wulka,

WWTP´s → Lonyai.

In a further step the relevant data will be compared to afford statements about the importance of special input data and to give some insight into the sensitivity of the used Model.

Comparison of the used main data

As can be concluded from the chapter above the comparison of data will concentrate on the most important input data for erosion and WWTP´s concerning TP emissions and groundwater and WWTP´s concerning TN emissions.

The calculation of TP and TN emissions from WWTP`s strongly depends on the accuracy of the used inventory. It is obvious that a more coarse inventory lacking for example small and new WWTP`s would result in calculation of lower nutrient emissions. Data base used for the large scale application is the ICPDR inventory of point sources prepared for almost the total Danube catchment. For meso scale application the national inventory was used preparing more detailed information. Although in the large scale application the attempt was made to account for the WWTP`s not being quoted by estimate additional emissions from population density, the higher emissions calculated from WWTP`s using the meso scale application in the Ybbs and the Lonyai catchment express the more detailed data base in the meso scale application and an underestimation in the accommodation approach in the large scale application.

The evalution of data used with respect to TP erosion emission calculation has to focus on the following parameters:

- Enrichment ratio calculated from the phosphorus content of the top soil and of suspended solids

- Sediment Delivery Ratio (SDR) calculated from the slope and the fraction of arable land

- Sediment input (SED) calculated from the SDR and the soil loss

Table 6.3 gives an overview of the main input data with respect to TP erosion emission calculations. The SDR in the Lonyai catchment (large scale application) is set to zero because of the very low average slope. From the formula of SDR (see figure 4.6) used in the MONERIS approach you can deduce that only slopes > 0.25 % will result in calculations of the SDR. That means that for erosion emission calculations a slope > 0.25 % is a model limiting condition. So especially in catchments with low average slopes an exact slope calculation is of great importance to calculate the SDR, a major erosion input parameter. In the Wulka catchment the SDR using the meso scale approach is higher than the SDR using the large scale application. One reason are the slope data, which are much higher in the meso scale application (9,1 % meso scale compared with 1,7 % large scale) due to the use of different grid sizes discussed in chapter 4.4.2. Nevertheless, the Sediment input (SED) calculated by SDR multiplied with soil loss is higher in the large scale application in the Wulka catchment due to higher soil losses. The soil loss data in the large scale application are taken from the soil erosion map (RIVM, 1995) while in the meso scale application data from own investigations were used (D2.1, D2.2). It is obvious that the precision of the soil erosion map for large scale erosion estimates is of extraordinary importance.

Table 6.3: Major input data for erosion calculations in Lonyai and Wulka catchment (large scale application = (ls) and meso scale application (=ms)).

	Top soil P [mg/kg]	**Mean specific soil loss [t/(ha*a)]**	**Slope [%]**	**Arable land [%]**	**SDR [%]**	**ER**	**SED [t/a]**
Lonyai (ls)	660	0,2	0,02	71,6	0	18,0	0
Lonyai (ms)	435	0,7	0,97	68,6	6,1	8,8	10359
Wulka (ls)	685	3,5	1,68	59,1	6,1	5,6	4949
Wulka (ms)	674	1,8	9,08	57,0	9,8	3,3	2003

The deviations in data used in meso scale and large scale applications like top soil P and the share of arable land are negligible.

The most important input data for TN emission calculations from groundwater are:

- The N surplus
- The water balance
- The residence time

Detailed information are provided in chapter 4.3.1. A comparison of data used on large scale and meso scale application points out that the higher N surplus calculated from field balances on meso scale in both the Wulka and the Ybbs catchment is essential for the higher emissions from groundwater compared to the large scale application using data from national level. Furthermore, as already stated in chapter 4.3.1., the geological data are of significant importance to calculate groundwater residence times and hence nitrogen retention.

Summarized, it can be concluded that the main emission pathways (erosion and WWTP`s for TP and groundwater for TN) lead to the main deviations in the meso scale and large scale application whereas the deviations for TP are much higher due to large uncertainties in erosion emission calculations. Responsible for these deviations is the reliability of the WWTP inventory in the case of WWTP`s; for erosion TP emission calculations the slope and the soil loss data exactness are of significant importance, while for groundwater TN emission calculation the N surplus (field balances versus national data) and the resolution of the used geological maps to calculate groundwater N retention seem to be the most influencing data sets in the case of this meso scale and large scale application comparison.

7. SUMMARY

This handbook is designed for nutrient management on larger catchment scales. As a consequence the methods introduced and suggestions given do not comprise the full content of the requirements of the European Water Framework Directive. Furthermore the handbook is restricted to N and P only. The focus is on the management of nutrient emissions into the hydrosphere in order to abate eutrophication. Acidification, climate change, healthy diet, etc. are out of scope of this handbook.

This handbook for nutrient management on catchment scale is mainly based on work carried out within EU funded DaNUbs Project (Nutrient Management in the Danube Basin and its impact on the Black Sea) (EVK 1-CT-2000-00051). For 388 subcatchments in the Danube basin (100 – 16000 km^2) the MONERIS nutrient emission model was applied and in 5 case study areas more detailed investigation on the nutrient cycle were carried out. Knowledge on nutrient sources, relevant processes in the environment and pathways are crucial for an efficient nutrient management.

Results from the daNUbs project express that the emission model MONERIS is a suitable tool to reproduce nutrient balances (nitrogen and phosphorus) on large scale. For an application on meso scale and even smaller scale MONERIS achieves convincing results concerning nitrogen load calculations. Nevetheless, phosphorus load calculations on smaller scales result in significant deviations. This means that MONERIS results can be used for a first overview accessing the main nutrient sources and pathways in fact- but if a more sophisticated question concerning the spatial and temporal resolution should be solved other models have to be used. However outcome from meso scale model comparison (chapter 2) point up that results from models attributed to this scale have to be challenged critically.

MONERIS model output accuracy for nitrogen loads show mean deviation of < 20 % on large and even on meso scale. The dominant pathway for nitrogen emission is groundwater.

Phosphorus loads calculated with MONERIS show mean deviations from load measurements of 30 % on large scale and meso scale. On smaller scale (sub catchments of case study areas), deviations can increase to 50 %. The dominant pathways for phosphorus emissions are erosion and WWTP`s. As was clearly expressed the higher deviations for phosphorus are on a large extent due to inaccuracies calculating emissions from erosion while emissions of nitrogen mainly caused by exfiltrating groundwater is well reproduced by most emission models. As shown in chapter 4.4 the insufficient calculation of phosphorus emissions by erosion is due to a number of uncertainties concerning main input data increasing with increasing scale.

It is problematic to judge the reasons for the deviation: on the one hand the calculation of the emissions and the transformation and retention are not “precise” on the other hand also the load calculations based on measured data are not exact (chapter 4.6 and chapter 5) . However, it can be said, that the larger the catchment, the less important is a high sampling frequency as extreme runoff events are much less important than in small catchments. For large catchments a biweekly sampling frequency is sufficient. Nevertheless special attention has to be paid to the monitoring of P-loads at high flow events in sampling and load assessment.

MONERIS applies two methods to calculate the retention for phosphorus and nitrogen in the river: one based on the specific runoff and the other one on the hydraulic load. Using the retention approach, based on the specific runoff the calculated N and P-loads tend to be underestimated.

The estimated TN and TIN loads deviate by about 20 %, the P-loads by about 30 % from the observed values on the large scale. In the meso scale for all the data a mean deviation within the range of 30 % could be obtained for the N loads using the retention approach based on the hydraulic load. For the estimation of the P loads the mean deviation for all the data was higher than 50 % using both retention approaches.

The reflection of “unusual” high point source contribution in the retention approaches shall be improved. Therefore if point sources contribute significantly to the river discharge the MONERIS retention approaches have to be modified.

In addition potential sinks (additional retention) like reservoirs are not included in the MONERIS retention and have to be taken into account.

The quantity of emissions via groundwater depends (i) on the nitrogen surplus on agricultural soils, (ii) the surplus on non agricultural soils and (iii) the extent of retention/denitrification in soil and groundwater.

The influence of denitrification in soil and groundwater on nitrogen emissions to surface waters may be higher than the anthropogenic factors leading to nitrogen surpluses in agriculture. Quantification of denitrification in groundwater therefore is the most important value in the frame of a regional nitrogen balance for surface water emissions. The natural factors influencing denitrification in soils and groundwater are hydro-geological factors as groundwater recharge rates, flow velocities and residence time in the underground, but also carbon or pyrite availability and oxygen depletion in groundwater.

The nutrient discharges to surface waters from point sources are highly influenced by the population density, the consumption of P-containing detergents and to some extent by the industrial activity in a region and the waste water infrastructure of a region (sewer system, level of treatment).

N-Emissions from urban areas via rain water sewers and combined sewer overflows are usually below the calculation accuracy of total nitrogen emissions, discharges from non sewered areas may be relevant in cases, where a high portion of the population is not connected to sewer systems.

Tile drained areas significantly reduce the residence time in the underground and therefore the retention/denitrification of nitrogen in the underground. Often the required data is not available.

Overland flow (nutrient emissions via the surface runoff from non paved areas in dissolved form) can be neglected for N. In addition erosion is not an important source for N-emissions into surface waters.

Direct deposition of nutrients on the surface waters is irrelevant for nitrogen balances of land based catchments. It will become relevant only in regions with a high share of surface water area. However, deposition on agricultural and especially non agricultural areas is of significant importance as it contributes to the surplus on soils, which is basis for emissions via groundwater.

Erosion processes heavily influence the P-balances. However, the estimation of P-emissions is related to a high degree of uncertainty. Starting point is the P-surplus resp. the resulting accumulation of the surplus over the years which lead to increasing phosphorus concentrations in soils. The enrichment ratio of phosphorus in eroded particles (enrichment of phosphorus in small, easier erodable particles) leads to higher concentrations in eroded particles as the average soil concentration. Only partly the soil eroded finally ends up in the river system. This "onland-retention" has to be considered by using a Soil Delivery Ratio. Depending on catchment characteristics, the sediment delivery ratio may vary between 1 and 0.05 or even less, meaning that 100% of the eroded material reaches a specified outlet or only 5% may be detected. As the SDR is a ratio, it depends also on the amount of calculated gross erosion (which in turn depends on the soil erosion model employed). Soil erosion rates may vary in two orders of magnitude and similarily do SDR's. This means that both parameters are important drivers for the calculation of sediment loads. Calculation of erosion at the mesoscale is possible if appropriate tools are employed. At the mesoscale, calculation of erosion primarily needs consideration of the available databases. If not extra derived for project purposes, the necessary input data usually are not available with an accuracy that is needed to estimate on site erosion. Therefore, assumptions to derive regional (for instance at community level) estimations of erosion have to be made, turning on site calculations (although these are already difficult) into risk estimations. Being a ratio between gross erosion and a measured sediment load at a specified point it is obvious, that the sediment delivery ratio is strongly dependant on the estimated soil loss rates, which in turn are influenced by the databases employed. This makes sediment delivery ratios empirical tools for a given data set of erosion and sediment load estimation. To derive transboundary erosion estimations it is therefore essential to either have a joint database for erosion calculation or to derive transfer functions for input parameters in order to harmonise the different data sets.

P-emissions are high for points sources in those areas were a high population density coincidences with a high degree of connections to sewer systems and a low level P-removal at treatment plants. Even if point sources only contribute to a small share to total average emissions at low flow conditions point sources can highly influence river P concentrations.

Values for emissions from urban areas obtained in the case study regions vary between 0,02 – 0,09 kgP/(ha.a.) and contribute to the total emissions with less than 20 %.

Phosphorus emissions to surface waters via groundwater have to be considered in areas with a high groundwater discharge or high phosphorus concentrations in groundwater (up to about 0.16 kgP/(ha.a) which may be a significant contribution to the total emissions).

The contribution of overland flow (nutrient emissions via the surface runoff from non paved areas in dissolved form) to the total emissions for surface waters is only relevant (>10 %), if the water discharge via the surface is high.

Emissions via tile drainage and deposition are below 0.01 kgP/(ha.a) related to the total area and are of no importance as compared to other emissions. Tile drainage might become important under certain conditions (high portions of tile drained areas, high P-losses from soils).

For increasing inaccuracies of phosphorus calculations on smaller scale one possible reason is the more dynamic behaviour of P-loads in smaller catchments leading to increased inaccuracies in load measurements. For monitoring of P-loads this means that flood events have to be specifically addressed in tributaries anyway. In a large river the importance of event oriented load monitoring depends on the time scale considered. For calculations of yearly loads monitoring at flood events is still decisive. If average loads over 5 years and more are taken into consideration, monitoring at flood events is less decisive, unless the probability of events increases significantly due to change of landuse practices in the catchment or climate change.

Data availability is essential for emission modelling. On large scales a lower resolution of the data is sufficient for nutrient modelling, however local measures for nutrient reduction can not be derived any more. While data precision for nitrogen load balances is sufficient in general, data for phosphorus load balances have to be improved. In general, especially improved input data of the main emission pathways (erosion and WWTP`s for phosphorus and groundwater and WWTP`s for nitrogen) can contribute to more precise nutrient balances. Nevertheless, these data are often not available or deviations of data sets available (possibly caused by different resolution or by different sources) can lead to significant deviations in nutrient balances. Therefore an international consistent data pool is strongly recommended to guarantee cross border nutrient balances in terms of WFD implementation.

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APPENDIX

METHODOLOGY FOR MATERIALS ACCOUNTING

In the early 1980s Baccini and Brunner developed a concept for assessing the anthropogenic metabolism of regions[1]. They combined existing scientific methods and new approaches to connect and interrelate soil, water and air with the anthroposphere in a holistic manner.

Current environmental protection requires adequate instruments for:

- the early recognition of the potential accumulation or depletion of materials in the environment,
- the identification of the most effective point of control of harmful concentrations and flows and
- integrated resources and environmental management

By modelling a system such as an urban region and its processes one can determine the total or partial amount of goods that enter, stay or leave the region relative to time. This is necessary to improve the management of given materials (sewage sludge, manure, etc.) and to determine concentrations of given elements (nutrients and/or chemicals) which may be crucial or toxic to the region, to its inhabitants and to the environment in this region. When it is possible to simulate such systems one can better understand and control the input and output flows of anthropogenic processes.
To fulfil these requirements, the main sources, flows, stores and emissions of anthropogenic and natural materials in regions must be determined. Material Accounting (MA) is one of the methods used to derive such requirements.

The metabolism of the anthroposphere is characterised by material[2] management systems[3], consisting of substances[4], goods[5] and processes[6].

Material flows into a given system are called *Imports,* material flows leaving a given system are called *Exports*. *Inputs* are material flows into a given process. Material flows from a given process are called *Outputs*. If the input-mass of goods into a process exceeds the output-mass of goods from a process *stocks* are formed. The stocks will increase until the fluxes are nearly equal (but not necessarily synchronous).

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[1] A *Region* is a more or less autonomous network of ecosystems and the anthroposphere. The *Anthroposphere* can be defined as an open system consisting of processes ("Industry & Trade", "Households", "Agriculture", "Forestry", etc.) connected with fluxes of goods. It is driven by man's biological and cultural needs. The anthroposphere is the part of the ecosystem where activities of humans take place. The subsystem *Environment* consists of the compartments " Hydrosphere ", "Lithosphere" and "Atmosphere". It runs by solar energy and the interactions of regional and global ecosystems.
[2] *Materials*: comprises substances and goods
[3] *System:* open assemblage consisting of materials, goods and processes.
[4] *Substance:* a chemical element or their compounds (characterised physically, chemically and economically within goods).
[5] *Goods:* consist of substances or mixtures of substances with functions valued by man (food, washing powder, sewage sludge, etc.).
[6] *Process:* denotes the transformation, transport or storage of a good.

Materials accounting is based on the law of mass conservation for chemical elements. This allows one to balance inputs, outputs and reservoirs within a certain period of time.

Sometimes it is impossible or inaccurate to analyse directly the input into a process. For instance the input into waste treatment is very heterogeneous and therefore sampling and analysis would be very costly. By analysing the easily accessible materials the missing products can be calculated by transfer coefficients. *Transfer coefficients* (dimensionless) express the partitioning of the total input between different outputs for a given process.

The flow of every material can be calculated by three types of information:
- the mass-fluxes of the input goods
- the material concentration of the input good
- the transfer function of this material in each process.

Measured concentrations in water, air and soil can be used to asses the present situation. This information, however do not depict changes, the speed at which possible changes occur or where the material flows come from. That means that with the traditional measuring methods it is possible only to determine environmental problems, which have been caused in the past or, sometimes, today. Early recognition of future loadings, as well as the identification of the effective means enables the decrease of loadings. Both are not possible with todays methods to monitor surface and groundwater quality.

Measures for environmental protection have to regard the whole system and have to be based on the principle of prevention. This approach aims at a long-term sustainability of (regional) material fluxes by optimising and controlling the input, the output, the stocks as well as the use of materials in the (regional) anthroposphere.

The prerequisite to develop a nutrient emission control policy for the groundwater and for rivers is to asses major inputs of nutrients into the water systems. The study of various activities that release material fluxes into the environment are an important base for a materials management program.
The methodology of materials accounting is a tool to reach this aim.

Steps of Materials Accounting

Usually materials accounting technique comprises five major steps:
1. **Definition of the objectives and questions**: Formulation of the problems, the goals and the questions to be answered by the study.
2. **System identification**: A highly complex system (region, plant, etc.) consisting of some hundred and up to 100,000 components has to be structured and condensed so that the amount of information is small enough to be collected and treated but still sufficiently large to allow that the goals of the project are fulfilled (*System identification*). Each system has its own typical characteristics (system boundary[7], processes, goods and their interrelationships) depending on the objectives of the project and the selected materials.

7 *System boundaries*: *Boundary in time*: the accounting period.. *Boundaries in space*: have to be defined as well horizontal as vertical. The horizontal boundary typically is a catchment area of a river, political boundaries (country, nation, ..), or a plant. The vertical boundary defines the height and depth of the considered system.

The first step consists in listing and defining goods and processes, the next step is to link the goods and processes.

3. **Data acquisition and calculation**: The provisional balance tries to estimate the material fluxes with the use of existing and/or easily available data that do not have to be specific for the region. By means of a provisional balance the processes and goods essential for the system are determined. The results of the provisional balance enable the selection of those processes and fluxes of materials and goods that have to be studied with first priority.
4. **Sensitivity analysis** of the data: The sensitivity analysis allows the identification of the key variables of the system and the estimation of effects due to variations of input variables.
5. **Presentation and documentation**: All results are combined and presented by means of simple graphs and tables. The primary data should remain secure and easily accessible. Modelling different scenarios would be useful to assess the impact of various measures on the regional stocks and flows of selected materials in view of environmental loads or of resource depletions.

Issues of scaling

Balancing material fluxes and estimating stocks in a region may rise some difficulties:

- dealing with a large set of data,
- using different boundaries (e.g. hydrological borders and political ones),
- facing different temporary stocks (e.g. retention) and
- non-availability of (reliable) data on a small scale but on a large scale or vice versa.

In practice the methodology of materials accounting operates on a number of different scales and both upscaling and downscaling has to be taken into account. Upscaling means the extension or extrapolation of estimates from a smaller region to a larger one. Down-scaling is the transfer of estimates from a larger region to a smaller one.

System Identification

As a first step the system boundaries have to be defined in space and time.
The geographical boundary has to be defined for each system.

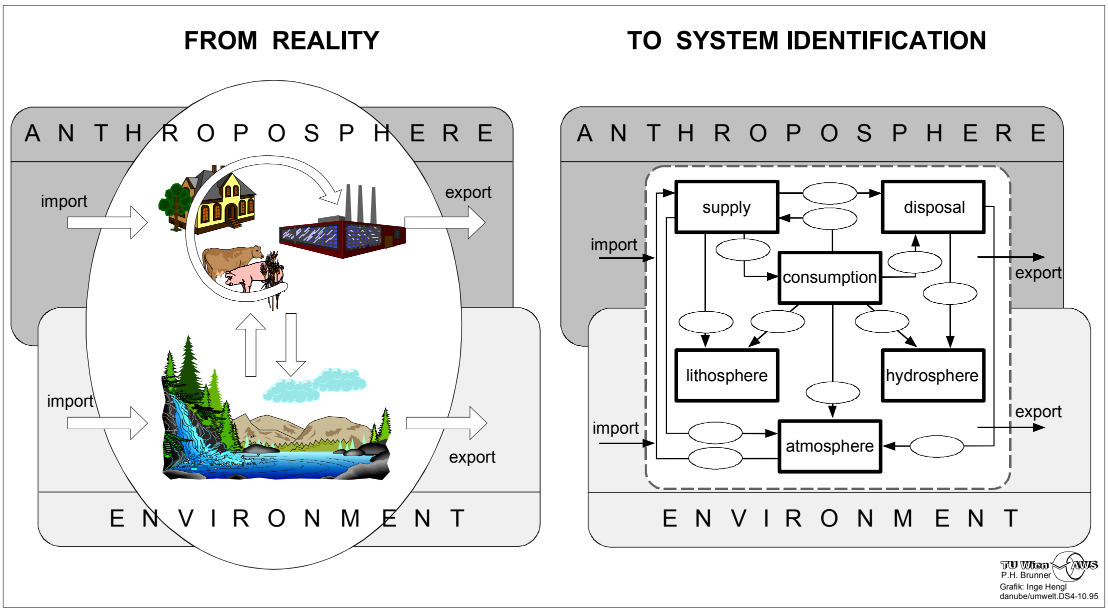

Figure 1: The different systems to describe the material flows.

The time boundary is the calendar-year (e.g January 1st - December 31st 2004), depending on the data availability. If there is no data available for the chosen year, data and information from previous years have to be employed.

The space boundary is the catchment area considered.

The next step is to select and connect processes and goods.

For some goods data on amounts and concentrations are readily available. Other goods and material fluxes can only be estimated. For these fluxes it is important to state the assumptions that have been made to quantify them.

The diagram below shows the system definition used in the (Phare project: Nutrient balances for Danube countries:

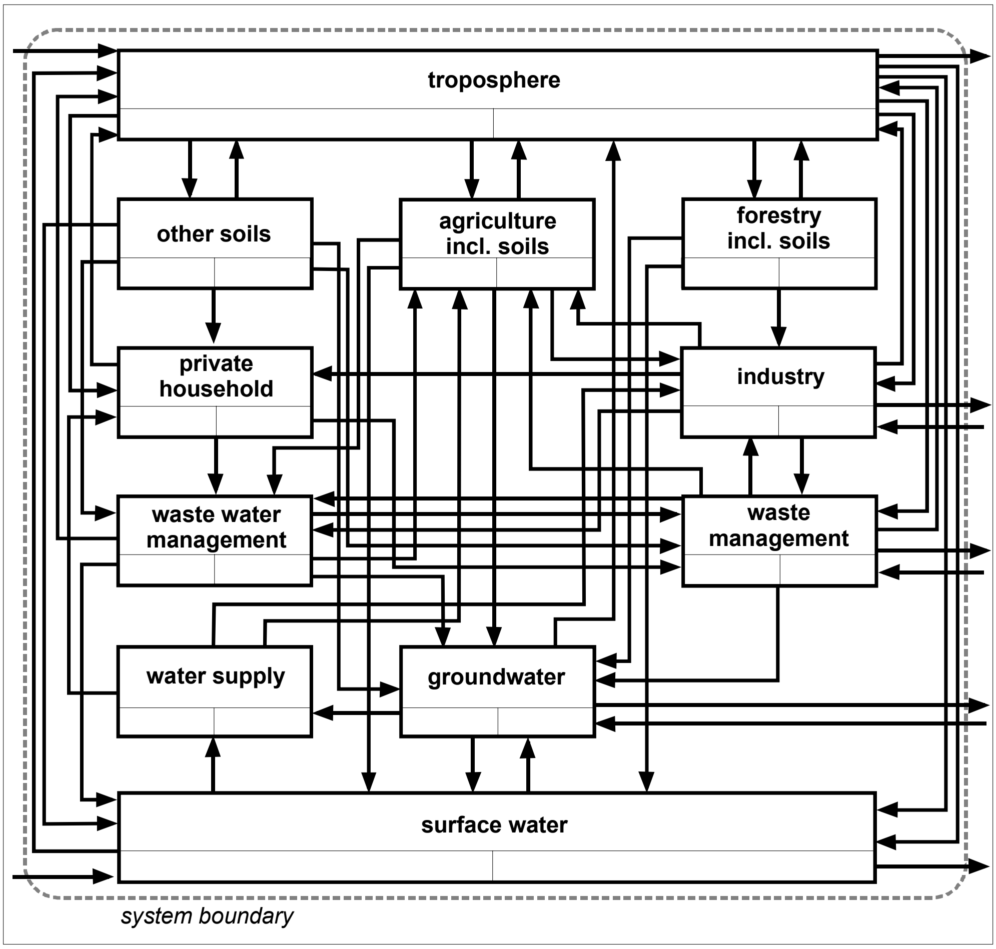

Figure 2: The processes and linkages of flows of the system „region" in the Phare project: Nutrient balances for Danube countries.

The approaches given should be considered as advices or examples but not as the only possible way to estimate the nutrient flows.

In tables presented an overview will be given, which flows are already included in the MONERIS calculations explicitly (exp.) or implicitly (imp.) and which ones are missing (mis.).
The following detailed process descriptions should serve as a guideline to obtain further information on the nutrient flows in the case study areas in view of future options for the efficient protection of surface and groundwater.

7.1.1. Agriculture

Agriculture is the central process for the regional nutrient management. A detailed scheme including also nutrient flows of minor importance is given in **Figure 3**.

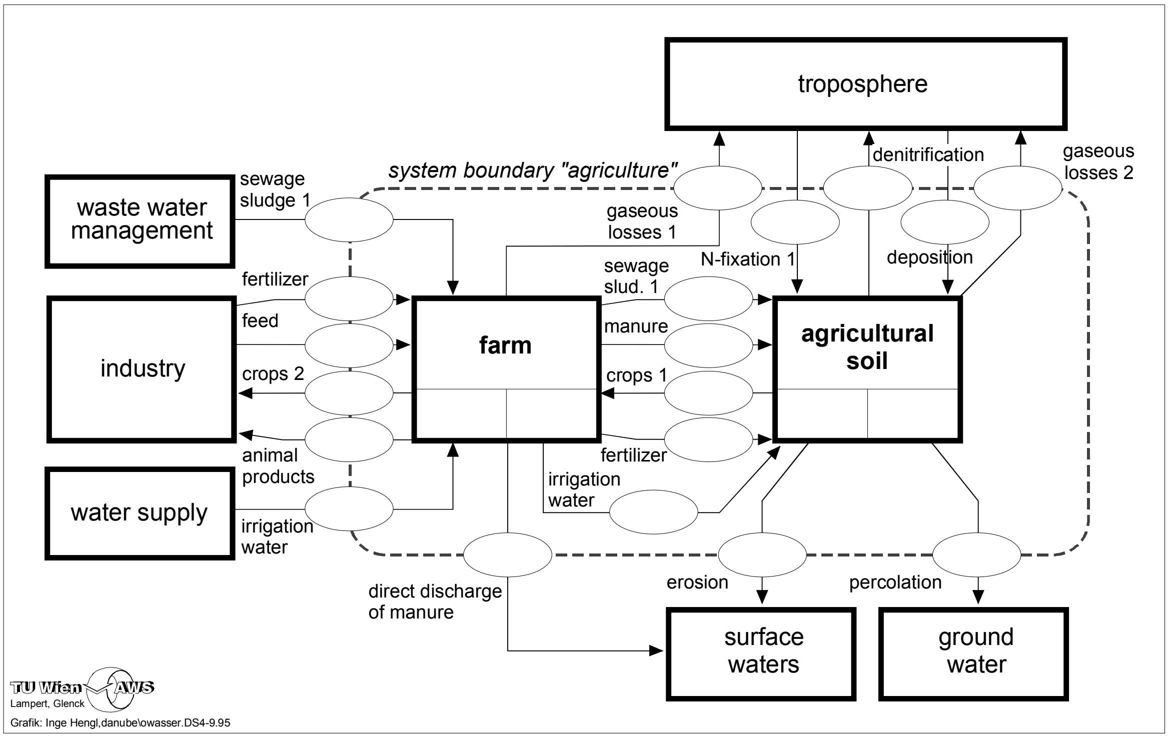

Figure 3: system „agriculture“

It is useful to separate the process Agriculture into “farm” and “agricultural soil” to be able to clearly distinct between gaseous emissions from the farm or from the soil.

“Starting point” of the MONERIS approach to calculate emissions from diffuse sources is the *nutrient surplus* in soils and is calculated as follows:

	N	Moneris	P	Moneris
Input	Organic fertilizer	exp.	Organic fertilizer	exp.
	Mineral fertilizer	exp.	Mineral fertilizer	exp.
	N-fixation by micro-organisms	exp.		
	Atmospheric deposition	exp.		
Output	Harvested products	exp.	Harvested products	exp.
Surplus	**N-surplus**		**P-surplus**	

Flows to be calculated in addition

Feed for livestock	mis.
Gaseous NH_3 losses farm	mis.
Gaseous NH_3 losses soil	imp.
Direct discharges of manure	mis.
Animal products	imp.
Crops for nutrition/ind. purpose	imp.
Feed (import/export)	mis.
Denitrification agric. soil	mis.
N, P-Stock in plough layer N, P change of stock	imp. calc. for P

On a regional level the (potential) nutrient flows "sewage sludge" and "irrigation water" are negligible. Sewage sludge could be important especially for P if the nutrient surpluses would be lower (or even zero in P-rich soils) and most of the waste water would be treated in plants with P-removal.

Organic fertilizers: In order to estimate the N- and P-content of manure the number of animals per species within the region has to be known. It is possible to claim a range for the gross-nutrient-content of the excreta for each species. To calculate the nutrient amount applied to soils we have to assume a ratio for losses of volatile NH_3 emissions during the storage and the application. Usually the losses during storage lie between 15 to 20 % of the total amount of N (depending on the stable system (manure, dung, etc. and the kind of animal) *(gaseous losses 1)*. The *gaseous losses of* NH_3 during the application again amounts to 15 to 20% of the total amount of N *(gaseous losses 2)*.

Calculating the amount of organic fertilisers applied on soils it has to be investigated if excreta are discharged directly into surface waters or treated in sewage treatment plants!

To conclude:
Organic fertilizer = Nutrient content of the excreta minus gaseous losses during storage minus gaseous losses during the application minus direct discharges minus treated manure.

For P no losses are assumed (except in the case of direct discharges or treatment in treatment plants).

(Mineral) fertilisers: Usually, information can be obtained only on a highly aggregated level and there is no distinction between the application on arable land or grassland. In general mineral fertilisers are added on arable land rather than on grass-land.
A transformation of the information available from a national or a county level into a regional scale can be very inaccurate. In order to obtain more realistic values the crops real needs have to be estimated. These needs mainly depend on the present nutrient pools in the soils (soluble P-fraction, organic matter content, etc.) and the expected yields of the various crops.

crops real needs - nutrient input by organic fertilizer = amount of mineral fertiliser

The method illustrated above neglects the economic situation in a region. Depending on the purchase power of the farmers the amounts of mineral fertilizers applied can vary on a broad scale.

One possibility to cross-check the results obtained is to contact regional suppliers of mineral fertilizers.

N-fixation: various crops are growing in symbiosis with particular micro-organisms, which are able to fix considerable amounts of atmospheric N. Additionally, certain micro-organisms, that live solitary in the soil, are able to fix molecular Nitrogen.

Atmospheric deposition: in some regions, especially those with a high density of industry, traffic, power plants or animal farms, the deposition of N-compounds (NO_x, NH_y) can amount to considerable N-flows (in certain cases more than 50 kg N/ha.a). The deposition of N-compounds can constitute an important nitrogen flux (problems with measurements of the total deposition have to be considered).

Sewage sludge:

Direct discharge of manure: Direct discharge of manure can contribute a considerable nutrient load to surface waters. These amounts should be calculated if manure discharge is a well known agricultural practice in the case study area.

If manure is treated in special waste water treatment plants the nutrient flow of the effluents into the river system have to be calculated.

Harvested products (crops 1): This good comprises the N, P-removal from all harvested crops for feed and for nutrition/industrial purposes, the latter being defined as *crops 2.* These two groups of harvested products shall be calculated separately.
If the harvest statistics of the nation or region under investigation are known the nutrient removal can be calculated easily by multiplying the amounts with the N, P-concentrations of the various crops. (This statistic can also be used to estimate the fixation of atmospheric nitrogen by symbiotic micro-organisms mentioned above.).

If feed is imported into the region (net-balances of feed!) these amounts have to be considered (we have to take into account the possibility that regions may produce either more or less feed than necessary for the regional livestock or the regional population. *Feed* in Figure comprises only the imported or exported feed. If the regional production of feed exceeds the regional demand this flow would be negative.

To calculate the potential impact of changes in the dietary habits it has to be estimated, how much area is used to produce one unit of animal protein and one unit of vegetarian protein.

Animal Products: Animal products comprise the production of meat, milk and eggs. N, P-removal by producing animal biomass can be estimated by means of slaughter statistics or derived from the numbers of fattening cattle, pigs and poultry and the average time of fattening. The production of other animal products such as milk and eggs can be learned from national statistics (if available) or estimated by multiplying the average production of one individual with the number of individuals in the case study area. (we have to take into account the possibility that regions may "export" a considerable amount of animals that are butchered outside the case study area – again net balances should be made).

Denitrification: In the Moneris calculation it is not differentiated where denitrification takes place. Denitrification takes place in (temporary) anoxic zones in the unsaturated zone or in the groundwater. As this process is influenced by the NO_3-concentration in the soil, a change of the agricultural practice could result in a decrease of the denitrification. In order to calculate the trend of the nutrient stock of the agricultural soils (in the plough layer), estimations on the denitrification in agricultural soils should be made.

Nutrient stock in soils: The N and P content of the top soil is crucial to calculate the nutrient flow by erosion. The nutrient stock of soils is built up by the amount of both organic and inorganic N and P. To estimate this stock recent measurements of nutrient-concentrations in soils (if available) should be used. If measurements are not available in the region the soil-concentrations of comparable regions (soil, hydrology, agricultural practice) should be used. This way of calculating the nutrient stock differs from the calculation of the nutrient stock by Moneris. Moneris defines a "starting point concentration in the soil" (i.e. for P in "western Germany": 565 mg P/kg in 1975). Based on this "starting point" the change in the soil concentration is calculated by integrating the annual P-surpluses (mineral fertilizer + organic fertilizer – harvest removal). Using both approaches the results can be compared.

Data of the process agriculture suggested to be investigated:

	unit
land use pattern in the region	%, ha
number of livestock for each species	number
production of animal products (meat, milk, eggs)	t/year, number/year
N-, P-concentration of animal products	mg N,P/kg
total manure/species	t/year
N-, P-concentration of manure	mg N,P/kg DM
NH_3-losses during storage	% of total N-content
NH_3-losses during application	% of total N-content
manure discharged to surface waters	% of total manure
manure discharged to sewer systems	% of total manure
harvest specified to different crops	t/year
cultivation area of the different plants	ha
imported feed (net-amounts)	t/year
N-, P-concentration of crops (grass)	mg N,P/ kg DM
mineral fertilisers applied to grassland	t N,P/year, kg N,P/ha
mineral fertilisers applied to arable land	t N,P/year, kg N,P/ha
deposition of N compounds on agricultural land	kg/ha
denitrification rate of agricultural soil	kg N/ha.a
N-, P-concentration of soils	mg N,P/ kg DM

7.1.2. Forestry

In the MONERIS model, it is assumed that the total amount of Nitrogen deposited on the forestry area percolates into groundwater. Moneris does not consider the nutrient removal by

the wood harvested or by denitrification nor the N-fixation by micro-organisms. As a consequence Moneris tends to overestimate the nutrient surplus in forestry and as a consequence the nutrient emissions to groundwater.
As for “agriculture”, input and output flows as well as the change of the stock have to be calculated in order to calculate the impact of different measures.
Information about the wooded area and the exploitation of forest products should be available from regional wood harvesting statistics.

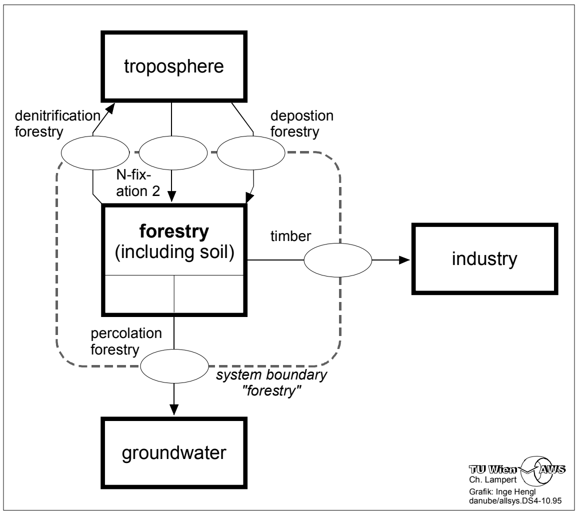

Figure 4: Process „Forestry“

Flows to be calculated in addition:

	Moneris
N-fixation	Missing
Deposition	implicitly
Timber	implicitly
Percolation	implicitly
Denitrification forestry	implicitly

In order to balance forestry the input flows *deposition* and *N-fixation* as well as the output flows *timber* and *denitrification* have to be calculated. The resulting N-surplus is partly a change of stock and partly is percolated to the groundwater – the part percolating to groundwater should be the “effective” input for the Moneris calculation.

Deposition forestry: in some regions, especially in those with a high density of industry, traffic or power plants, the deposition of N-compounds (NO_x, NH_y) can amount to more than 50 kg N/ha. The deposition of N-compounds can constitute an important nitrogen flux (problems with measurements of the total deposition have to be considered).

N-fixation: Some species of trees are growing in symbiosis with micro-organisms which are able to fix atmospheric nitrogen. Additionally, certain micro-organisms, that live solitary in the soil, are able to fix molecular Nitrogen. The fixed N-amounts in forest soils should be estimated.

Timber: this good describes the entire amount of wood that was cut down within the period of one year and removed from the forestry area. Regional or national statistics will provide information about the amount of deforested wooded area or amounts of timber harvested. Usually this flow amounts to 5 to 10 kg N/ha.a.

Denitrification: molecular nitrogen (N_2) and N_2O are released from forest soils. In order to calculate the change of stock, these amounts shall be estimated.

data needed for forestry calculations:

	unit
wooded area	km^2
structure of the wood (species)	%, km^2/species
harvested timber	t; t/species
estimated stock of wood	t/ha
estimated annual increase of wood	t/ha.a
N-, P-concentration of wood	mg N,P/kg DM
N-, P- concentration of soil	mg N,P/kg DM
N-fixation by micro-organism	kg N/ha.a
N-deposition	kg N/ha.a
Denitrification	kg N/ha.a

7.1.3. Waste water

Waste water from private households

A main source of N- and P-emissions from private households is due to the consumption of food. Dietary habits (i.e. the consumption of animal and vegetarian protein) strongly influence the need of agricultural area. To produce one unit of animal protein about 5 times more area is needed than to produce one unit of non-animal protein.
For P in addition the use of P-containing washing powder and dish-washer products can be of importance. These figures may vary in the individual countries depending on their nutrition customs and the use of phosphates in washing powder. MONERIS itself provides 3 possibilities how to calculate the P emissions. However the N-emission calculations are quite rough assuming an average N-flux of 11 gN/cap.d. This results in an underestimation of the N-emissions from inhabitants (at least for a western Europe diet).
In the MONERIS approach nutrient emissions from industrial waste water seems to be underestimated at least for P which is assumed to be 0 g P/PE.d. For Nitrogen a range between 0 - 7 g N/PE.d depending on the size of the treatment plant is given in Moneris.
These emission estimates should be refined. The branches of industry where high turnovers of nutrients take place are: chemical-, fertiliser-, food-, pulp- and paper-industry, diaries, coking plants.
Detailed approaches to evaluate N-loads in waste water are presented in chapter 4.3

7.1.4. Gaseous emissions – Deposition

Deposition highly influences N-emissions from non-agricultural areas (Phare project: Nutrient balances for Danube countries: >10% of the total emissions into the Danube). MONERIS does not differentiate on the source of the deposition, therefore no related measures can be derived. Reactive N-compounds are emitted into the troposphere by agricultural activities (NH_3) as well as by combustion processes (NO_x) (traffic, energy conversion (heating, electricity, etc.)).

Flows to be calculated in addition :

	Moneris
NH3 emissions agriculture	imp.
NOx-emissions combustion processes	mis.

For combustion processes a subdivision into "Traffic", "Energy conversion in power plants", "private households and public institutions", "Industry" is useful.

The NO_x-emissions of combustions can be calculated by using specific emission factors.

Despite that these estimations will be quite rough, they will support to cross-check the data concerning N-deposition.

required data:

source of energy used for industry, energy conversion in powerplants, traffic, private households and related amounts	kind of fuel, t, m^3
Combustion technology	
applied cleaning technology	efficiency

SWAT APPLICATION

For the calculation of detailed water balances on the regional scale in the daNUbs project the SWAT 2000 model was used, which is able to simulate the hydrologic cycle on the basin level. This distributed parameter, continuous time model is able to consider impacts from the present landuse, of the soil, of the shallow groundwater and of anthropogenic factors on the hydrological cycle of catchments.

The water balance calculations using the SWAT 2000 model were performed prior to distinguish the different runoff components are calculated by the model: the surface runoff, the lateral flow and the base flow. The model was calibrated and validated for the case study regions based on measured river discharge data. But due to the large number of input files (10 files for specification of basin, subbasin, HRU (Hydrologic response units) and river parameter) containing lots of diverse physically based parameter and the high demand on input data (daily, monthly or annual measurements) it was difficult to run the model and to achieve satisfactorily model performances. Due to the generation of HRU's out of soil and the landuse map (every combination of soil type / landuse type generates a HRU) the number of calibration sensitive parameter increased in that way, that a high effort was necessary to be put in the calibration and validation of the model (trial and error method + use of an automatic calibration routine, comparison with hydrograph separation techniques) to raise the model performance into an acceptable range. Once hydrological calibrated, the model was able to calculate the water balance and the runoff components distribution.

Additionally, the model simulates nutrient and pesticide routing and transformation into the river and in the river itself. Therefore in a second step it was tested if the SWAT nutrient tool is suitable for mesoscale nutrient balances (application for the study sites in Austria and Hungary).

It was found that the calculation of nutrient balances in the case study regions by SWAT was problematic with respect to process reproduction and calibration. Some major processes for nitrogen and phosphorus retention in groundwater and surface water are not included in the model making it difficult for an application in respect to quantification of nutrient fluxes on the case study level. E.g. for nitrogen, the retention (retardation) in the groundwater by denitrification is considered in the soil profile only and thus nitrate is tracked through the groundwater without any degradation. For phosphorus, calculated phosphorus river loads do not consider the fate of inorganic phosphorus interacting with sediment during channel transport (Kovacs, 2004). Additionally, a hydrological calibration of the model is not sufficient for an application for reproduction of nutrient balances, which means the model needs a calibration in respect to water quality too.

A detailed documentation concerning the SWAT application in the case study areas is given in Deliverable 1.1, 1.3 + 1.4.