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## Research Paper

# Modelling of Nutrient Emissions in River Systems – MONERIS – Methods and Background

*key words:* nitrogen, phosphorus, in-stream retention, river basin management

### Abstract

MONERIS is a semi-empirical, conceptual model, which has gained international acceptance as a robust meso- to macro scale model for nutrient emissions. MONERIS is used to calculate nitrogen (N) and phosphorus (P) emissions into surface waters, in-stream retention, and resulting loads, on a river catchment scale. This paper provides the first (i) comprehensive overview of the model structure (both the original elements and the new additions), (ii) depiction of the algorithms used for all pathways, and for retention in surface waters, and (iii) illustration of the monthly disaggregation of emissions and the implementation of measures. The model can be used for different climatic conditions, long term historical studies, and for future development scenarios. The minimum validated spatial resolution is 50 km<sup>2</sup>, with a temporal resolution of yearly or monthly time steps. The model considers seven emission pathways (atmospheric deposition on surface waters, overland flow, erosion, tile drainage, groundwater, emissions from sealed urban areas, and point sources), and six emission sources (natural background, fertilizer application, nitrogen atmospheric deposition on arable land and other areas, urban sources, and point sources); and these are calculated separately for different land-uses. The pathway and source-related approach is a prerequisite for the implementation of measures to reduce non-point and point-source emissions. Therefore, we have modified MONERIS by the addition of a “management alternative” tool which can identify the potential effectiveness of nutrient reduction measures. MONERIS is an appropriate tool for addressing the scientific and political aspects of river basin management in support of a good surface water quality.

### 1. Introduction

The international North Sea Conference (NSC) and the Helsinki-Commission (HELCOM) enacted a 50% reduction of nitrogen (N) and phosphorus (P) loads from 1985 and 1987 to 1995 in the North Sea and the Baltic Sea. Despite this, water quality remains insufficient in European rivers, and consequently legislative initiatives have been put in place. Most important among them is the European Water Framework Directive (WFD), which came into force in 2000, to bring about good ecological and chemical conditions of water quality in groundwater and surface water bodies until 2015 (EUROPEAN PARLIAMENT AND COUNCIL OF THE EUROPEAN UNION, 2000; REKOLAINEN *et al.*, 2003).

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To determine effective measures for reducing emissions to, and loads in, surface waters, we can use tools that model nutrient sources and emission pathways, and scenarios for land use and management options (CHAPLOT *et al.*, 2004; KERSEBAUM *et al.*, 2003; KRAUSE *et al.*, 2008; VOLK *et al.*, 2008, 2009). Examples of such models include HSPF (BICKNELL *et al.*, 2001), AGNPS (YOUNG *et al.*, 1987), MIKE-SHE (REFSGARD, 1997), Soil and Water Assessment Tool (SWAT) (ARNOLD *et al.*, 1998; GASSMAN *et al.*, 2007), SWIM (KRYSAKOVA *et al.*, 1998), GROWA/WEKU (WENDLAND and KUNKEL, 1999) and STOFFBILANZ (GEBEL *et al.*, 2010a, b). Overviews of different eco-hydrological models are given in ARNOLD and FOHRER (2005); HORN *et al.* (2004); KRYSAKOVA and HABERLANDT (2002) and VOLK and STEINHARDT (2001).

Within the EU-project EUROHARP, eight nitrogen models and five phosphorus models were applied to 17 European catchments, to compare the models from the viewpoint of the calculated net nutrient loads and the partitioning of nutrient emissions at the catchment scale (KRONVANG *et al.*, 2009). This comparison was conducted because nutrient emissions on a catchment scale cannot be measured. In general, in the tested models although there was good agreement between observed and modelled nutrient loads, there were very large differences among the models in the modelled values for nutrient emissions and retention. Neither the simple, nor the complex, models were more consistent, nor in other terms delivered better results than the others; the limitations were those posed by the simplicity, or the data demand of the models.

The MONERIS model is described in detail in BEHRENDT (1988, 1996), BEHRENDT *et al.* (2000, 2002a); BEHRENDT and DANNOWSKI (2005); FUCHS *et al.* (2010) and VENOHR *et al.* (2009, 2010a).

Compared to other models like SWAT (ARNOLD *et al.*, 1998), SWIM (KRYSAKOVA *et al.*, 1998), GROWA/WEKU (KUNKEL and WENDLAND, 2002, 1997) and STOFFBILANZ (GEBEL *et al.*, 2010a, b), our MONERIS model (MODelling Nutrient Emissions in River Systems; BEHRENDT *et al.*, 2000) works with a moderate demand of input data, requires only a short computing time and is applicable to large river basins. Approaches used in recent models, range from export coefficients to detailed physically based equations. Consequently, model results vary from lumped total figures, to detailed information of a high temporal and spatial resolution. In these terms, MONERIS represents a good compromise between detailed, process-based models, and the application of simple approaches.

MONERIS calculates nutrient emissions for seven different emission pathways and six different emission sources (VENOHR *et al.*, 2010b). With this model, it is possible to identify long-term pathway- or source-related changes, as well as management options including cost-effectiveness-analysis. However, depending on the scientific task, other models might be chosen, such as SWAT or SWIM (higher temporal resolution) or GROWA/WEKU (focus on groundwater).

For the first version of MONERIS, the temporal resolution was 5-year periods, and the spatial resolution was 500 km<sup>2</sup>. The main driver for the first development of MONERIS was to identify sources and pathways of nutrient emissions for hydrological sub-catchments, later called analytical units (AU). The approaches for the considered pathways have been developed and calibrated separately from each other, and no further inter-calibration was made during later model runs. The approaches for calculation of in-stream retention have been calibrated with observed values or with data from models other than MONERIS. Thus we ensured that the calibrated retention approach did not include a systematic error of MONERIS, by filling the gap between the modelled emissions and the observed loads.

Since its inception in 1999, MONERIS has been applied to the whole of Germany (BEHRENDT *et al.*, 2000; VENOHR *et al.*, 2008a, b), numerous European river systems (*e.g.*, Axios, Danube, Daugava, Elbe, Odra, Po, Rhine, Vistula, see BEHRENDT *et al.*, 2000, 2003a; 2003b, BEHRENDT and DANNOWSKI, 2005; SCHREIBER *et al.*, 2005), and to river catchments in Canada (VENOHR *et al.*, 2010b), Brazil (VON SPERLING and BEHRENDT, 2007), Mongolia (HOFMANN *et al.*, 2010; HOFMANN *et al.*, 2011; MENZEL *et al.*, 2011) and China (XU, 2004).

The international application of MONERIS facilitates its continuous testing and development. The recent version of MONERIS works with a minimum spatial resolution of 50 km<sup>2</sup>, and calculates in yearly or monthly time steps. Emissions can be calculated on a land-use basis, rather than as a sum for the respective analytical unit. MONERIS has been extended by a “management alternative” tool, so that it can also be used to identify the potential of measures to reduce emissions and loads. The model includes a module, developed under the lead and in cooperation with the Technical University of Berlin, to calculate the costs originating from the implementation of measures, thus allowing cost-effectiveness-analysis (CEA). The description of the CEA module is in preparation, and will be published subsequently to the present model description. Completed by the “management alternative” tool and the CEA module, the model provides a comprehensive framework for river basin management.

The three objectives of our present paper are to: (i) provide a comprehensive overview of the MONERIS model and its new developments, including the methods, formulas and model parameters for all pathways, and for the in-stream retention, (ii) illustrate the monthly disaggregation of emissions, and (iii) to describe the “management alternatives” tool for the simulation of the impact of management changes on the nutrient emissions into surface waters.

The paper begins with a general overview of the model, and then proceeds to detailed description of the methods, including the algorithms for calculation of each pathway and those for retention in surface water; the paper is completed by presentation of two new features: the newly developed method to disaggregate the yearly results into monthly results, and the “management alternatives” tool.

## 2. General Model Descriptions

MONERIS has been developed at the Leibniz-Institute of Freshwater Ecology and Inland Fisheries (IGB; BEHRENDT *et al.*, 2000) as a semi-empirical, conceptual model for the quantification of nutrient emissions from point and diffuse sources into surface waters of river

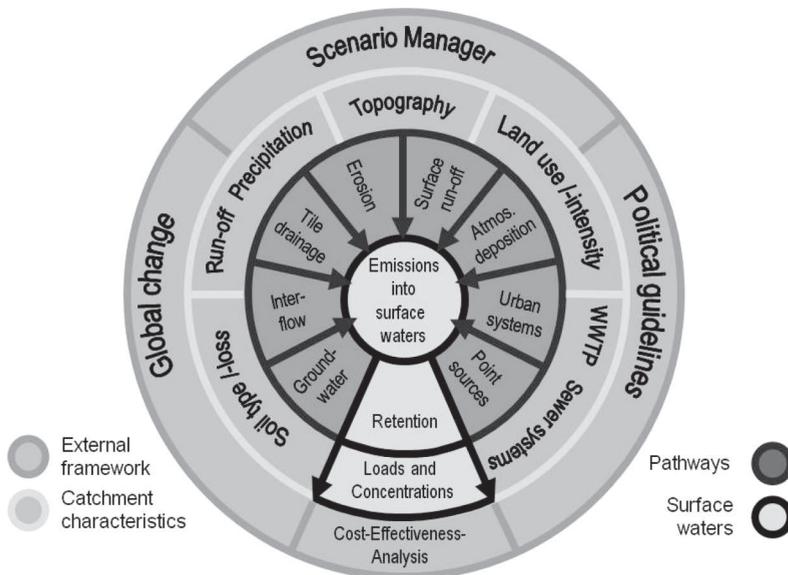


Figure 1. Structure of the MONERIS model showing the external framework, catchment characteristics, pathways, and surface waters.

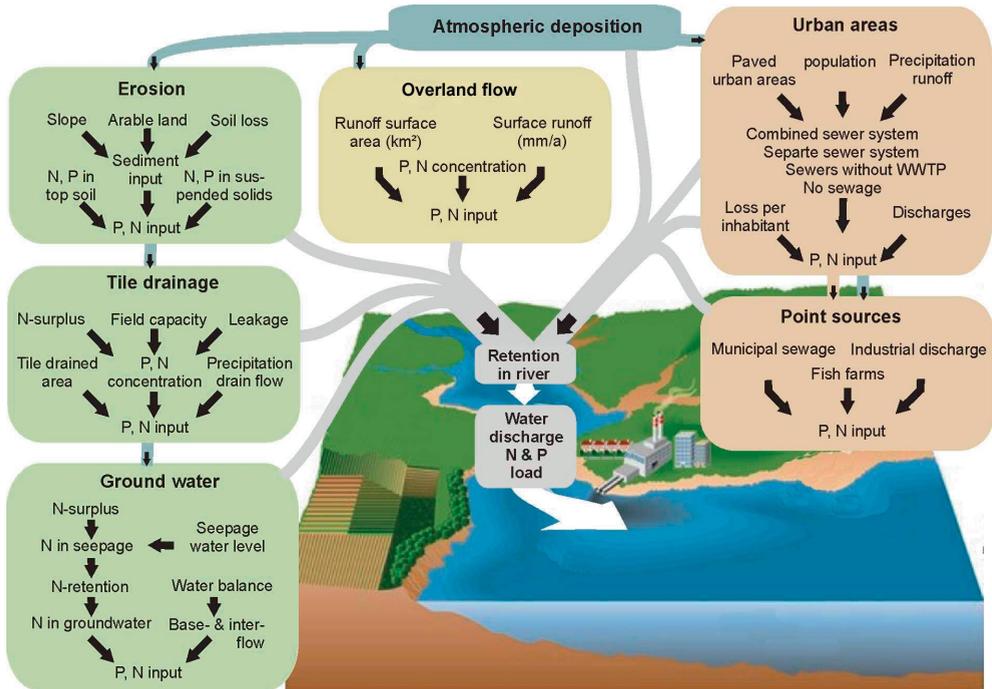


Figure 2. Pathways and processes for nutrient emission into surface waters (background picture modified after LOICZ, TURNER *et al.*, 1998).

systems (BEHRENDT *et al.*, 2000, Fig. 1). The model is based on catchment characteristics (Fig. 2) derived from digital maps and statistical reports. The application of MONERIS allows the regionally differentiated quantification of nutrient emissions into and the resulting loads in surface waters.

MONERIS considers six diffuse nutrient emissions pathways: direct emissions via atmospheric deposition on surface waters, overland flow, erosion, emissions via tile drainages, emissions via groundwater and interflow from unsealed areas and finally emissions from sealed urban areas and households (Fig. 2). Emissions from point sources are considered for municipal waste water treatment plants (WWTP) and industrial discharges.

For each pathway, the water component (for the erosion pathway the sediment yield) and the nutrient concentration for the different pathways are calculated separately. MONERIS calculates the total resulting net emissions entering surface waters, including retention and transformation processes in soils and groundwater. Subsequently, in-stream retention in surface waters is calculated separately for tributaries, main rivers and selected lakes to finally describe resulting loads in the surface waters of a river system. The basis for a model evaluation is the comparison of observed and calculated nutrient loads in the surface waters.

### 3. Methodology to Calculate Nutrient Emissions

Analytical units (AU) are the smallest modelling unit and represent the basis for the calculation of nutrient emissions and in-stream retention. Analytical units describe hydrological sub-catchments (based on the topography). For each analytical unit the hydrological con-

nection to a downstream analytical unit has to be defined and is considered in MONERIS by the ‘flow-net-equation’ (FNE). Artificial divides, e.g. by channels, of the river system (splittings) can be implemented in the FNE and are considered for the calculation of the in-stream retention and load. Due to the resolution of the input data analytical units must not be smaller than 1 km<sup>2</sup>. The validated minimum size for analytical units is 50 km<sup>2</sup>.

MONERIS requires the total runoff as input data. The runoff for every diffuse pathway except groundwater recharge is determined separately as a function of precipitation, while the groundwater recharge is calculated as the difference of the total runoff and the sum of the runoff from the other diffuse pathways.

The following land use categories are considered in MONERIS: arable land, grassland, natural covered areas (incl. forests), urban areas, wetlands, open land areas, open pit mine, water surface areas and other areas.

### 3.1. Calculation of the Water Surface Area

The area of surface water is needed to model the atmospheric deposition on surface waters as well as the later in-stream nutrient retention. Information on the lake area is in general available from topographical maps, whereas the surface area of river is, if at all, only given for large streams. An approach to calculate the surface area of (smaller) rivers was implemented in MONERIS.

The water surface area of rivers is calculated as a product of the mean width and the flow length for individual analytical units. For a subsequent calculation of in-stream retention, water surface areas of main rivers, tributaries, and lakes, have to be determined separately (Fig. 3). The calculation of river width is based on VENOHR *et al.* (2005), but was re-calibrated with an extended and modified data set. VENOHR *et al.* (2005) calculated the flow-

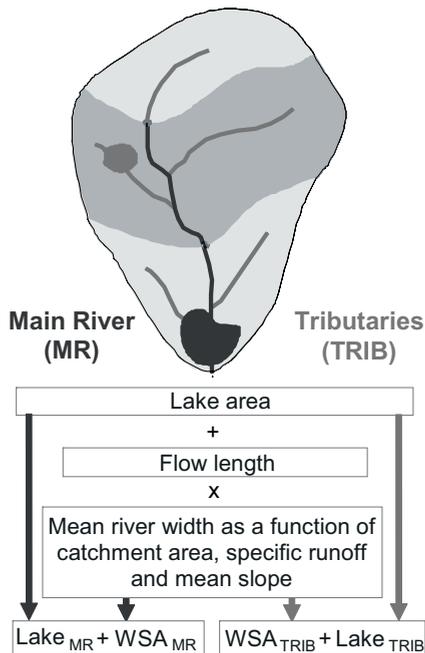


Figure 3. Calculation of the water surface area (WSA) of main rivers (MR) and tributaries (TRIB) in river systems.

length-weighted mean width of all tributaries within one analytical unit. In contrast to the approach by VENOHR *et al.* (2005), for the present calibration, the width has been classified in steps of 0.1 m, as the widths in maps were only given as classified data.

VENOHR *et al.* (2005) derived river widths from topographical maps and statistical reports, but the width of main rivers wider than 42 m were missing. In the recent, extended data set, 184 additional river stretches of large German rivers with widths up to 380 m have been added from Google-Maps. By combining both data sets, for main rivers, the widths of 462 river stretches have been now considered, and for tributaries the mean widths of rivers and ditches in 259 analytical units have been now considered. After classification of the width of tributaries into 0.1 m steps, there were 20 groups available for calibration of tributary widths.

The river width is calculated based on catchment area, specific runoff, and mean slope of the catchment (Eq. 1) and (Eq. 2). As the width of tributaries is only influenced by the local conditions, the calculation is based on catchment area, specific runoff and slope of the respective analytical unit. For main rivers, the entire hydrological catchment and the according mean specific runoff have to be considered, whereas the mean slope of the respective analytical unit is used.

$$W_{\text{TRIB}} = 0.082 \cdot A_{\text{AU}}^{0.0395} \cdot q_{\text{AU}}^{1.545} \cdot sl^{-0.025} \quad (\text{Eq. 1})$$

$W_{\text{TRIB}}$  = calculated mean river width of the tributaries, in m  
 $A_{\text{AU}}$  = area of the analytical unit, in km<sup>2</sup>  
 $q_{\text{AU}}$  = specific runoff of the analytical unit, in l/(s · km<sup>2</sup>)  
 $sl$  = mean slope (1000 m GRID) in analytical unit, in %

$$W_{\text{MR}} = 0.26 \cdot A_{\text{CAT}}^{0.49} \cdot q_{\text{tot}}^{0.45} \cdot sl^{-0.025} \quad (\text{Eq. 2})$$

$W_{\text{MR}}$  = calculated mean river width of the main river, in m  
 $A_{\text{CAT}}$  = area of the total catchment, in km<sup>2</sup>  
 $q_{\text{tot}}$  = mean specific runoff of the total catchment, in l/(s · km<sup>2</sup>)

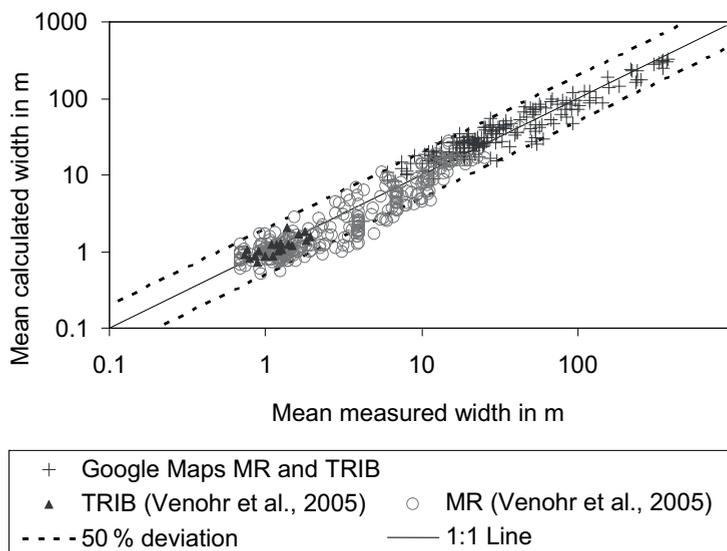


Figure 4. Comparison of measured and calculated widths for main rivers (MR) and tributaries (TRIB).

Table 1. Mean absolute deviation,  $r^2$ , and modelling efficiency (EF) of the calibration for main 462 rivers and 259 tributaries grouped in 20 classes.

	Main rivers $N = 462$	Tributaries $N = 20$
Mean absolute deviation in %	30.3	15.3
$r^2$	0.96	0.58
EF	0.94	0.50

Measured and calculated river widths are compared in Figure 4. The measured mean width varied between 0.7 m and 380 m for main rivers, and between 0.7 m and 1.9 m for tributaries. A mean deviation of 30.3% for main rivers, and 15.3% for tributaries, and as well as high regression coefficients (Table 1), document a good statistical agreement between measured and calculated widths.

We calculated the individual water surface area of main rivers and tributaries for a specific analytical unit, by multiplying the river width with the flow length. The flow length was derived from topographical maps, and was subsequently corrected according to the scale-dependent generalization of the map. With increasing scale, generalisation increases, such that smaller rivers and meanders are missing; thus the real length of the river, and consequently the water surface area, will be underestimated. Therefore, for 87 German catchments, we compared the lengths of rivers derived from maps with four different scales (1:25,000; 1:100,000; 1:250,000; 1:1,000,000), and derived scale factors for the main rivers ( $S_{MR}$ ) and tributaries ( $S_{TRIB}$ ) (Table 2) (VENOHR, 2006). For large scale maps the scaling factor can vary considerably, e.g., between DLM1000 and Bartholomew (BART1000). As the DLM1000 has been corrected with information from the DLM250 map, its scale factor is much smaller than that of the BART1000 map. Consequently, the detailedness of maps is not only dependent on the scale but also on the respective methods and data, which were used to generate the map. Scale factors thus cannot be transferred to other maps of the same scale.

Table 2. Scale factors for tributaries ( $S_{TRIB}$ ) and main rivers ( $S_{MR}$ ) for maps of different scales, based on 87 German catchments.

Maps	Scale	Scale factor	
		$S_{MR}$	$S_{TRIB}$
DTK25	1:25,000	1.00	1.00
UBA1000	1:100,000	1.11	1.83
UBA-OSU1000	1:100,000	1.11	2.10
DLM250	1:250,000	1.11	3.23
DLM1000	1:1,000,000	1.13	2.99
BART1000	1:1,000,000	1.18	8.40
DCW1000	1:1,000,000	1.17	6.28

### 3.2. Nutrient Emissions via Atmospheric Deposition on Water Surface Areas

In MONERIS, atmospheric deposition on water surface areas is calculated as a product of the water surface area and the area-specific atmospheric deposition rate of N and P. Atmospheric N deposition is considered separately for different land uses or as a mean value per

analytical unit, whereas for P deposition, only a mean deposition rate per analytical unit is considered. For a subsequent calculation of the total water balance in an analytical unit, the precipitation on, and the potential evaporation from, water surface areas are balanced.

### 3.3. Nutrient Emissions via Overland Flow

Emissions of dissolved fractions of nutrients via overland flow from unsealed areas is calculated separately for different land uses (arable land, grassland, naturally covered areas, open land areas, wetlands, open pit mine areas, and snow and ice covered areas) (Fig. 5).

Basis for the overland flow calculation is the specific runoff from these areas (Eq. 3).

$$q_{\text{spec}} = \frac{Q_{\text{AU}} - Q_{\text{U}} - Q_{\text{WSA}}}{A_{\text{GL}} + A_{\text{AL}} + A_{\text{NC}} + A_{\text{OA}} + A_{\text{S}} + A_{\text{WL}} + A_{\text{OPM}}} \cdot 1000 \quad (\text{Eq. 3})$$

$q_{\text{spec}}$  = specific runoff, in  $\text{l}/(\text{s} \cdot \text{km}^2)$

$Q_{\text{AU}}$  = mean annual runoff from analytical unit, in  $\text{m}^3/\text{s}$

$Q_{\text{U}}$  = mean annual runoff from sealed urban areas, in  $\text{m}^3/\text{s}$

$Q_{\text{WSA}}$  = annual precipitation on water surface areas, in  $\text{m}^3/\text{s}$

$A_{\text{GL}}$  = grassland, in  $\text{km}^2$

$A_{\text{AL}}$  = arable land, in  $\text{km}^2$

$A_{\text{NC}}$  = naturally covered areas, in  $\text{km}^2$

$A_{\text{OA}}$  = open land areas, in  $\text{km}^2$

$A_{\text{S}}$  = snow and ice covered areas, in  $\text{km}^2$

$A_{\text{WL}}$  = wetlands, in  $\text{km}^2$

$A_{\text{OPM}}$  = open pit mine areas, in  $\text{km}^2$

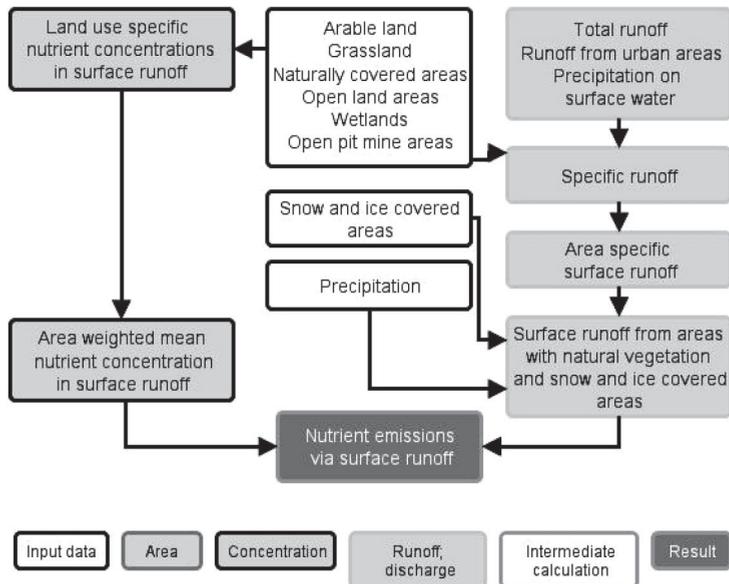


Figure 5. Basic scheme for calculation of dissolved nutrient emissions via overland flow.

The surface runoff is calculated as a function of the specific runoff of an analytical unit, following the approach by CARL and BEHRENDT (2006, 2008) and CARL *et al.* (2008) (Eq. 4). The use of this approach for test regions in the Danube catchment proved its applicability, and delivered results similar to those of hydrological models such as SWAT and DIFGA (CARL *et al.*, 2008).

$$q_{SR} = 0.0426 \cdot (q_{spec} \cdot 86.4 \cdot 0.365)^{1.2461} \quad (\text{Eq. 4})$$

$q_{SR}$  = specific surface runoff, in mm/yr

Equation 4 has not been validated for calculation of runoff from glaciers and snow-covered areas, and has to be considered by a separate approach. The US SOIL CONSERVATION SERVICE (1972) developed an approach to describe runoff from snow and ice covered areas (Eq. 5). Using data from Austrian alpine regions, ZESSNER *et al.* (2010) re-calibrated this approach and derived an exponent of 0.6, instead of the original value of 0.45.

$$Q_S = 4 \cdot A_S \cdot (PR_{yr} - 850)^{0.6} \frac{1000}{86\,400 \cdot 365} \quad (\text{Eq. 5})$$

$Q_S$  = surface runoff from snow and ice covered areas, in m<sup>3</sup>/s

$PR_{yr}$  = annual precipitation, in mm/yr

Equation 5 is only valid for annual precipitation exceeding 850 mm; in cases of lower precipitation, no runoff from snow and ice covered areas is assumed. Based on the specific surface runoff (Eq. 4), as well as runoff from snow and ice covered areas (Eq. 5) the total surface runoff from unsealed areas is calculated using equation 6.

$$Q_{SR} = q_{SR} \cdot (A_{GL} + A_{AL} + A_{NC} + A_{WL} + A_{OPM}) \frac{1000}{86\,400 \cdot 365} + Q_S \quad (\text{Eq. 6})$$

$Q_{SR}$  = total surface runoff from unsealed areas, in m<sup>3</sup>/s

### 3.3.1. Phosphorus

An extensive study on the P content and P absorption capacity of soils in the north-east German flatlands was conducted by PÖTHIG and BEHRENDT (1999), based on BRAUN *et al.* (1991) and WERNER *et al.* (1991). This study showed that the water soluble P concentration depends very strongly on the P saturation of the soil. PÖTHIG *et al.* (2010) made measurements at 429 sites in Germany and Switzerland, and derived an equation to calculate the dissolved P concentrations dependent upon the P saturation in soils (Fig. 6).

Since the P saturation in soils varies spatially and temporally, we modified the approach by PÖTHIG *et al.* (2010) for use in MONERIS. To consider changing P saturation in soils and the situation in a respective analytical unit, the ratio between the mean P accumulation and the maximum P accumulation during the entire calculation period is considered on an administrative unit level (Eq. 7).

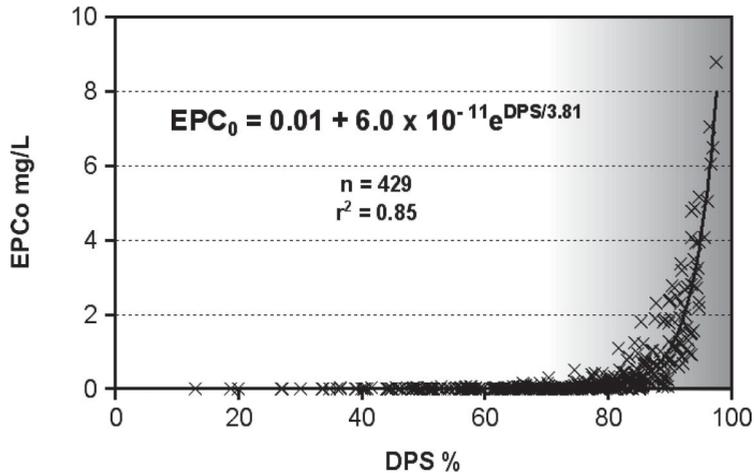


Figure 6. Dependence of the equilibrium P concentration (EPCo) on the degree of P saturation in the soil (DPS) (PÖTHIG *et al.*, 2010).

$$PA_{AU-COR} = PA_{AU-RY} \frac{PA_{AM-SY}}{PA_{AM-MAX}} \quad (\text{Eq. 7})$$

$PA_{AU-COR}$  = corrected P accumulation of a specific calculation year for each analytical unit, in kg/(ha · yr)

$PA_{AU-RY}$  = individual P accumulation for each analytical unit, considering the spatial distribution of P accumulation, in kg/(ha · yr)

$PA_{AM-SY}$  = mean P accumulation, on the level of an administrative unit, for the calculation year, in kg/(ha · yr)

$PA_{AM-MAX}$  = maximum P accumulation on the level of an administrative unit during the entire modelling period, in kg/(ha · yr)

Finally, a P change correction factor is calculated by equation 8.

$$PA_{AU-CF} = \frac{PA_{AU-COR}}{PA_{AM-MAX}} \quad (\text{Eq. 8})$$

$PA_{AU-CF}$  = correction factor to consider the temporal and spatial variability of P saturation in soils, in kg/(ha · yr)

In order to avoid unrealistically high P concentrations in surface runoff, a maximum P saturation of 97% is assumed, and the ratio of  $PA_{AU-CF}$  will therefore be limited by the model. Equation 9 calculates the P concentration in surface runoff, according to the approach by PÖTHIG *et al.* (2010) on the basis of temporarily and spatially corrected P accumulation.

$$P_{SR-AL} = 0.01 + 6 \cdot 10^{-11} \cdot \text{Exp}^{\left[ \frac{PA_{AU-CF} \cdot \text{DPS}}{3.81} \right]} \quad (\text{Eq. 9})$$

$P_{SR-AL}$  = P concentration in surface runoff from arable land, in mg/l

DPS = degree of P saturation in soils with arable land, in %

Using DPS values of 90% for arable land and 80% for grassland, at a correction factor  $PA_{AU-CF} = 1$ , the corresponding P concentrations of 1.1 mg/l and 0.1 mg/l in surface runoff are calculated by equation 9. These calculated concentrations correspond well to concentrations observed during storm water events (GELBRECHT *et al.*, 1996). For open land areas, naturally covered areas, open pit mine areas, wetlands, and for natural background conditions, a P saturation of less than 50% is assumed. This corresponds to a P concentration in surface runoff of 0.01 mg/l, following the approach by PÖTHIG *et al.* (2010). For snow and ice covered areas, a P concentration of 0.005 mg/l is assumed.

### 3.3.2. Nitrogen

In MONERIS, the N concentration in surface runoff from arable land is calculated on basis of the approach by WERNER *et al.* (1991). WERNER *et al.* (1991) did not take into account the N by atmospheric deposition (in form of  $NO_x$  and  $NH_y$ ), thus we make an addition to compensate for this (Eq. 10). For snow and ice covered areas, an N concentration of 0.1 mg/l is assumed. For land use other than arable land or snow and ice covered areas, the N concentrations in surface runoff are derived only from the atmospheric deposition.

$$C_{SR-N-AL} = 0.3 + \frac{AD_{NH_y} + AD_{NO_x}}{PR_{yr}} \quad (\text{Eq. 10})$$

$C_{SR-N-AL}$  = N concentration in surface runoff from arable land, in mg/l

$AD_{NH_y}$  = atmospheric deposition of  $NH_y$  fractions on arable land in  $kg/(km^2 \cdot yr)$

$AD_{NO_x}$  = atmospheric deposition of  $NO_x$  fractions on arable land in  $kg/(km^2 \cdot yr)$

The emissions via surface runoff are calculated by multiplying the respective P and N concentrations with the surface runoff for each land use type. The sum of these land use specific emissions is the total emission via surface runoff.

### 3.4. Nutrient Emissions via Erosion

The quantification of N and P emissions, via erosion, into surface waters is based on four parameters: (i) soil loss, (ii) slope, (iii) nutrient content in top soils, and (iv) land use. The erosion pathway of MONERIS follows two empirical approaches. First, MONERIS calculates sediment input as the product of the soil loss, and an empirical sediment delivery ratio (SDR, according to WALLING (1983, 1996)). Second, the modelled sediment input is multiplied with the nutrient content in the topsoil and an empirical enrichment ratio (ENR) (Fig. 7).

Soil loss data for arable land, grassland, naturally covered areas, and snow and ice covered areas, are considered separately and are usually generated based on the Universal Soil Loss Equation (USLE; DIN 19708, 2005, WISHMEIER and SMITH, 1978). To calculate the long term annual mean soil loss, this equation requires additional geo-referenced input data, such as that on precipitation, soil, and land management practices. Although the underlying concept of the USLE was criticized (*e.g.*, NOVOTNY and CHESTERS, 1989), we use this approach in MONERIS because alternatives are data-demanding and often limited to small areas (DE VENTE *et al.*, 2007; VOLK *et al.*, 2010). The long term annual mean soil loss is used as input data in MONERIS and has to be derived in advance of a model setup.

Soil loss and sediment input are highly variable in space and in time. In MONERIS, the long term mean soil losses from arable land, grassland, and naturally covered areas are corrected by the precipitation ratio  $PR_{CF}$ , between the actual and the long term mean USLE R factor (Eq. 11). The R factor is calculated according to DEUMLICH (1993).

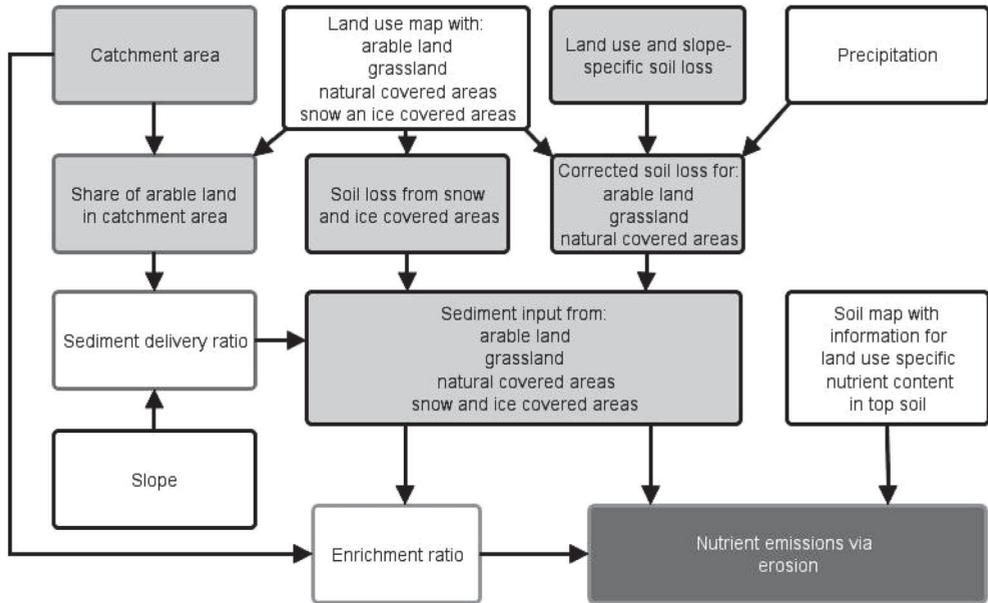


Figure 7. Main steps for calculating nutrient emissions via erosion.

$$PR_{CF} = \frac{0.152 \cdot PR_{S-SY} - 6.88}{0.152 \cdot PR_{S-LT} - 6.88} \quad (\text{Eq. 11})$$

$PR_{CF}$  = precipitation ratio as correction factor, dimensionless  
 $PR_{S-SY}$  = summer precipitation in specific study year, in mm/yr  
 $PR_{S-LT}$  = long term mean summer precipitation, in mm/yr

Not all areas are directly connected to surface waters and only a small fraction of eroded particles actually reaches the surface waters. Therefore, the share of areas that contribute to sediment input has to be defined, and is described by the Sediment Delivery Ratio (SDR). The SDR equation was derived from a detailed mapping in 29 catchments in eastern Germany. In a first step areas directly located at a surface water and with a slope orientated to it were derived as critical source areas. In empirical models, the spatial variability of SDR is usually explained by simple parameters such as average slope or area (DE VENTE *et al.*, 2007; WALLING, 1983, 1996). Accordingly, MONERIS uses the average slope and the proportion of arable land in each analytical unit to estimate the long term mean SDR (Eq. 12). Like this the share of areas contributing to emissions via soil erosion could be determined.

$$SDR_{AG} = 0.006684 \cdot (sl - 0.25)^{0.3} \cdot (20 + A_{AL\%})^{1.5} \quad (\text{Eq. 12})$$

$SDR_{AG}$  = sediment delivery ratio, in %  
 $sl$  = mean slope from 1000 m-DEM, in %  
 $A_{AL\%}$  = proportion of arable land, in %

Equation 12 is only suitable for agricultural areas with sheet erosion as the dominant erosion process (*i.e.*, the scope of the USLE). In case of no arable area (scenario on background conditions) it is assumed that on 20% of the catchment area erosion still occurs. Erosion may occur on arable land and on grassland but the spatial origin can often not clearly be distinguished. Equation 12 therefore has been re-calibrated under consideration of this share of 20% and to give the total share of arable land and grassland contributing to erosion. Equation 12 is applied when average slope angles are above 0.25%. For areas with a slope of < 0.25%, no erosion is considered in MONERIS and the SDR is set to 0. The approaches in MONERIS were calibrated with land use data derived from the Corine Land Cover Map (DEUTSCHES ZENTRUM FÜR LUFT- UND RAUMFAHRT, 2000).

In a second step the actual amount of eroded matter reaching surface waters had to be quantified. Therefore annual sediment loads (yields) at 23 gauges in southern Germany were used. At low flow conditions sediment loads to a considerable share consist of autochthonous material. USLE-based soil erosion data therefore were compared to long term mean annual critical sediment yields, following the method by BEHRENDT *et al.* (2000). Critical suspended sediment yields were calculated as the discharge-dependent fraction of total suspended solid yields (SSY) (Fig. 8).

From multi-year time-series of daily data on water runoff (Q) and suspended-solid concentrations (SSC), the mean values of SSY and SSC were obtained for runoff classes. Generally, these mean values of SSY and SSC increased above gauge-specific critical runoff (Q<sub>crit</sub>), and the critical yields were defined as the difference between regression models describing the SSY-Q and SSC-Q relationships for high and low discharge classes (Fig. 8; Eq. 13). Alpine catchments are very effective sediment suppliers, and the USLE underestimates their

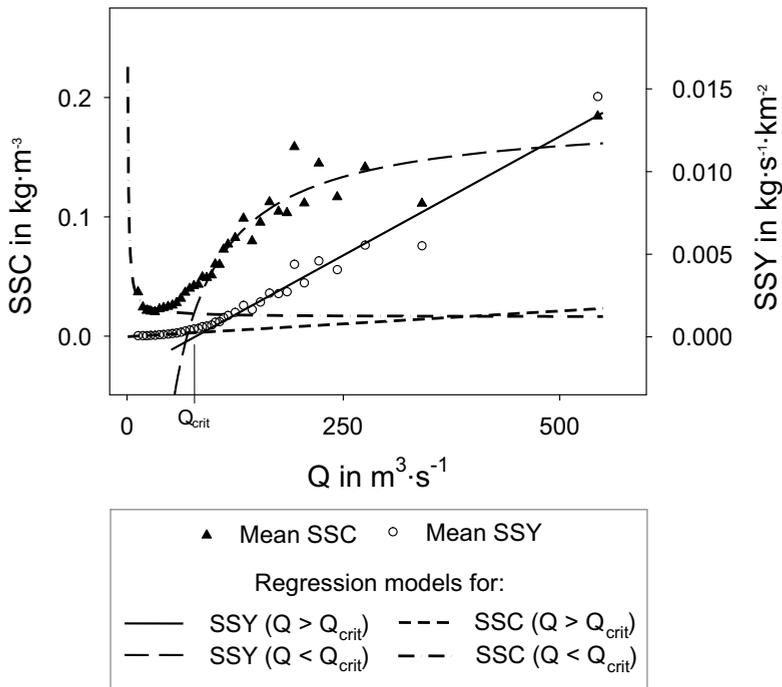


Figure 8. Calculation of critical suspended solids yields (Q = water discharge, SSC = suspended solids concentration, SSY = suspended solids yield, Q<sub>crit</sub> = gauge-specific critical runoff) (adapted from BEHRENDT *et al.*, 2000).

soil loss. On basis of monitored suspended solid loads in alpine regions BEHRENDT *et al.* (2000) derived a soil loss from snow and ice covered areas of  $400 \text{ t}/(\text{km}^2 \cdot \text{yr})$  (Eq. 13). The suspended solid yield by erosion from arable land, grassland, naturally covered areas, and snow and ice covered areas, is finally calculated by equation 13.

$$\text{SSY} = (\text{SL}_{\text{AL}} \cdot \text{A}_{\text{AL}} + \text{SL}_{\text{GL}} \cdot \text{A}_{\text{GL}}) \cdot \text{PR}_{\text{CF}} \cdot \text{SDR}_{\text{AG}} + \text{SL}_{\text{NC}} \cdot \text{A}_{\text{NC}} \cdot \text{PR}_{\text{CF}} + \text{SL}_{\text{S}} \cdot \text{A}_{\text{S}} \quad (\text{Eq. 13})$$

SSY = suspended-solids yield, in t/yr

$\text{A}_{\text{AL}}$  = arable land, in  $\text{km}^2$

$\text{A}_{\text{GL}}$  = grassland, in  $\text{km}^2$

$\text{A}_{\text{NC}}$  = naturally covered areas, in  $\text{km}^2$

$\text{A}_{\text{S}}$  = snow and ice covered areas, in  $\text{km}^2$

$\text{SL}_{\text{AL}}$  = soil loss from agricultural land, in  $\text{t}/(\text{km}^2 \cdot \text{yr})$

$\text{SL}_{\text{GL}}$  = soil loss from grassland, in  $\text{t}/(\text{km}^2 \cdot \text{yr})$

$\text{SL}_{\text{NC}}$  = soil loss from naturally covered areas, in  $\text{t}/(\text{km}^2 \cdot \text{yr})$

$\text{SL}_{\text{S}}$  = soil loss from snow and ice covered areas, in  $\text{t}/(\text{km}^2 \cdot \text{yr})$

$\text{PR}_{\text{CF}}$  = see equation 11

Soil erosion and sediment transport are selective processes in respect to particle size (SHARPLEY, 1980). Fine soil particles are preferentially transported to, and within surface waters, and are usually richer in adsorbed nutrients (nutrient content  $\text{NC}_{\text{TS}}$ ), than is the case with coarser soil particles (SCHEFFER and SCHACHTSCHABEL, 1989). As P accumulates in soils, BEHRENDT *et al.* (2000) developed an approach to estimate the P content in top soils, assuming an initial value of 150 mg/kg (soil mass) and a mean clay content of 21%. Top soil contents for open land, and snow and ice covered areas are assumed to be 150 mg/kg for P and 250 mg/kg for N.

The modelled sediment input is multiplied with the nutrient content of the topsoil and with an empirical enrichment ratio (ENR), to obtain the nutrient emission via soil erosion. The enrichment ratio is defined as the ratio of the nutrient content in suspended sediments to that in the bulk soil. For P, the variation of the calculated ENR in 27 catchments in the Danube basin is well explained by modelled sediment input (BEHRENDT *et al.*, 2000) (Eq. 14); this was in agreement with AUERSWALD (1989) and SHARPLEY (1980). Equation 14 is valid where  $\text{SSY}/\text{A}_{\text{AU}} \leq 1.0$ , otherwise  $\text{ENR}_{\text{P}}$  is set to 18.

$$\text{ENR}_{\text{P}} = 18 \cdot \left( \frac{\text{SSY}}{\text{A}_{\text{AU}}} \right)^{-0.47} \quad (\text{Eq. 14})$$

$\text{ENR}_{\text{P}}$  = enrichment ratio for P, dimensionless

$\text{A}_{\text{AU}}$  = analytical unit area, in  $\text{km}^2$

For N, an indirect approach had to be chosen because there were no data available on N content in suspended solids. From a study of 17 catchments WERNER and WODSAK (1994) determined that the mean N-P ratio of eroded soil and bulk soil was 2.35 and (Eq. 15) was adapted accordingly to estimate  $\text{ENR}_{\text{N}}$ , the enrichment ratio for N.

$$\text{ENR}_{\text{N}} = \frac{18}{2.35} \cdot \left( \frac{\text{SSY}}{\text{A}_{\text{AU}}} \right)^{-0.47} = 7.7 \cdot \left( \frac{\text{SSY}}{\text{A}_{\text{AU}}} \right)^{-0.47} \quad (\text{Eq. 15})$$

$\text{ENR}_{\text{N}}$  = enrichment ratio for N, dimensionless

The total nutrient emissions via erosion are calculated using the respective values for N and P, and for different land uses, by equation 16.

$$ER = \frac{NC_{TS}}{1000000} \cdot SL \cdot ENR \tag{Eq. 16}$$

- ER = nutrient emissions via erosion, in t/yr
- NC<sub>TS</sub> = nutrient content (N or P) in top soils, in mg/kg
- SL = soil loss from different land uses, in t/(km<sup>2</sup> · yr)
- ENR = N or P enrichment ratio, dimensionless

### 3.5. Nutrient Emissions via Tile Drainage Flow

N and P emissions, via tile drainages into surface waters, are calculated for tile-drained arable land and grassland, respectively. Three parameters are considered in MONERIS: (i) tile drain flow rate, (ii) size of tile drained areas, and (iii) mean nutrient concentration of the tile drain flow; the main calculation steps for nutrient emissions via tile drainages are shown in Figure 9.

#### 3.5.1. Tile Drainage Flow Rate

The tile drainage discharge rate is calculated as 50% of the winter precipitation and 10% of the summer precipitation following KRETSCHMAR (1977, Eq. 17). These values were supported by analysis of data for tile drainage from monitoring stations (HIRT *et al.*, 2011).

$$TD_{q-spec} = 0.5 \cdot PR_{WI} + 0.1 \cdot PR_{SU} \tag{Eq. 17}$$

- TD<sub>q-spec</sub> = area specific drain flow, in mm/yr
- PR<sub>WI</sub> = precipitation in winter, in mm/yr
- PR<sub>SU</sub> = precipitation in summer, in mm/yr

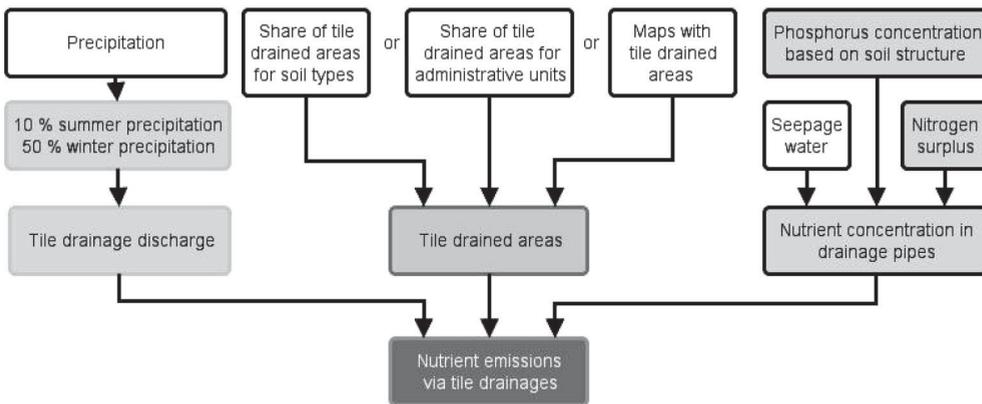


Figure 9. Main steps for calculation of N emissions via tile drainages.

### 3.5.2. Size of Tile Drained Areas

To determine the size of tile drained areas in an analytical unit, three different kinds of input data are used: (a) maps of tile drained areas (b) statistics on tile drained areas at an administrative level, and (c) calculated percentage of tile drained areas for different soil and site conditions.

In many areas, especially in Western Europe, there are little data available on location of tile drained areas (maps or statistics), and assessment of the percentage of tiled drained areas for different soil and site properties is the only way to derive the size of tile drained areas in an analytical unit. The percentage of tiled drained areas can be derived from digitalised tile drained data for representative areas. Based on spatial analysis of the soil and site properties on these representative tile drained areas, the percentage of tile drained areas could be transferred to a whole catchment area (BALZER, 2010; BEHRENDT *et al.*, 2000; HIRT *et al.*, 2005a, b).

### 3.5.3. Mean Nutrient Concentration in Tile Drainage Runoff

The N concentration in tile drain outlets (Eq. 18), and the potential nitrate concentration in the seepage water, is calculated based on the approach of FREDE and DABBERT (1998) using the regionally differentiated N surplus. The nutrient concentration of seepage water is expected to correlate with those of the tile drainage flow. The soil boundary condition is that net mineralization and net immobilisation are both negligible. To consider denitrification in soils the nitrate concentration in the tile drainage flow is reduced by an exponent of 0.85 for arable land, and 0.7 for grassland (BEHRENDT *et al.*, 2000). The calculated N concentrations in the tile drainage runoff, according to (Eq. 18), correspond to BEHRENDT *et al.* (2000).

$$TD_{TNC} = \frac{(IM_{Nsurp})^{0.85}}{TD_{q-spec}} \cdot 100 \quad (\text{Eq. 18})$$

$TD_{TNC}$  = N concentration in tile drainage flow under arable land (for grassland the denitrification exponent of 0.85 has to be substituted by 0.7), in mg/l

$IM_{Nsurp}$  = N surplus, in kg/(ha · yr)

The P concentration in tile drainage discharge for different soil structures was determined based on a literature study by BEHRENDT *et al.* (2000), and is summarized here in Table 3. Tile drained fens and bogs are expected to be degraded ecosystems where mineralisation of organic matter occurs, and thus they are treated as nutrient sources rather than as nutrient sinks. The phosphorous concentration in tile drainage runoff from bogs varies significantly according to regional conditions, and has been calibrated for different model applications.

Table 3. P concentrations in tile drainage runoff in four different soil structures.

Soil structure	P-concentration in mg/l
Sandy	0.20
Loamy	0.06
Fen	0.30
Bog	2.00

3.6. Nutrient Emissions via Groundwater

Nutrient emissions via groundwater are calculated as the product of area weighted land use and soil structure specific groundwater concentrations, and the groundwater discharge rate, defined as the sum of natural interflow and base flow.

3.6.1. Nitrogen

The N surplus on agricultural land and its change during groundwater residence time is the dominating parameter for calculating the N concentration in groundwater. In MONERIS, the mean N concentrations in the groundwater are calculated based on mean long term conditions. Here, the groundwater residence time can either be estimated based on mean long term conditions (Eq. 21) or is considered as external derived input data. The modelled N concentration in groundwater also consider N uptake in the root-zone, later retention in groundwater, as well as the seepage water quantity in the individual analytical units (Fig. 10). The N emissions via groundwater to the surface waters are calculated as the product of the groundwater N concentrations, and the groundwater flow, calculated for the respective year.

Groundwater recharge is calculated for all unsealed areas, except water surface areas, open pit mine areas and tile drained areas (Eq. 19). The groundwater recharge is calculated for each analytical unit, as the difference between the mean long term total runoff (input data of MONERIS) and the calculated mean long term runoff from diffuse pathways (Eq. 19).

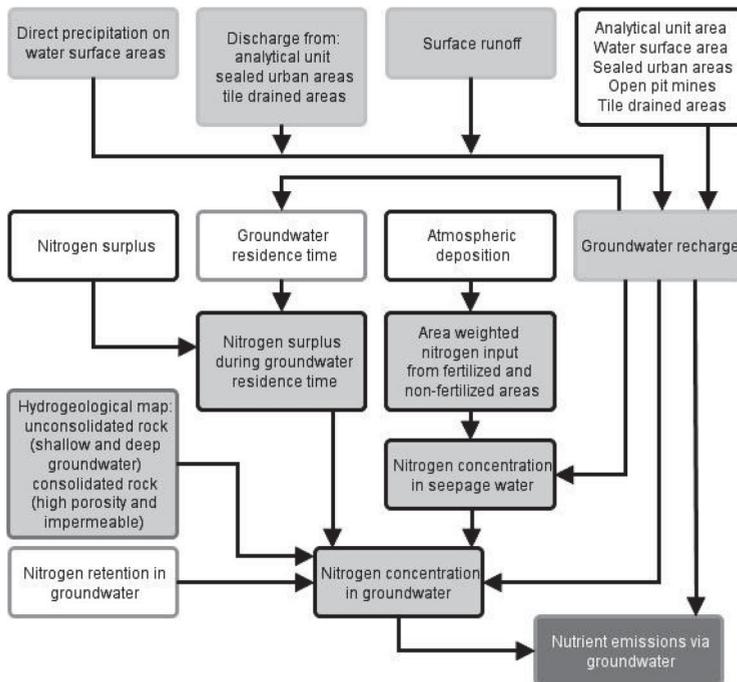


Figure 10. Main steps for calculation of N emission via groundwater.

$$q_{\text{GW}} = \frac{Q_{\text{AU}} - Q_{\text{WSA}} - Q_{\text{SR}} - Q_{\text{TD}} - Q_{\text{urban}}}{A_{\text{AU}} - A_{\text{WSA}} - A_{\text{urban}} - A_{\text{OPM}} - A_{\text{TD}}} \cdot \frac{86400 \cdot 365}{1000} \quad (\text{Eq. 19})$$

- $q_{\text{GW}}$  = mean long term groundwater recharge, in mm/yr  
 $Q_{\text{AU}}$  = mean long term runoff from an analytical unit, in m<sup>3</sup>/s  
 $Q_{\text{WSA}}$  = mean long term direct precipitation on surface waters, in m<sup>3</sup>/s  
 $Q_{\text{SR}}$  = mean long term surface runoff from unsealed areas, in m<sup>3</sup>/s  
 $Q_{\text{TD}}$  = mean long term discharge from tile drainages, in m<sup>3</sup>/s  
 $Q_{\text{urban}}$  = mean long term discharge from sealed urban areas, in m<sup>3</sup>/s  
 $A_{\text{AU}}$  = analytical unit area, in km<sup>2</sup>  
 $A_{\text{WSA}}$  = water surface areas, in km<sup>2</sup>  
 $A_{\text{urban}}$  = sealed urban areas, in km<sup>2</sup>  
 $A_{\text{OPM}}$  = open pit mine areas, in km<sup>2</sup>  
 $A_{\text{TD}}$  = tile drained areas, in km<sup>2</sup>

By calculating groundwater recharge as a residual of the remaining flow components, the water balance in an analytical unit should not be altered. However, in some cases, precipitation and runoff show inconsistencies, and the runoff from the diffuse pathways is higher or much lower than the total runoff in an analytical unit, resulting in negative or very high groundwater recharges. In order to avoid large deviations from the runoff (input data of MONERIS), a corrected groundwater recharge ( $q_{\text{GW-corr}}$ ) is defined by a maximum value of 1.5 times the sum of interflow and base flow calculated by CARL *et al.* (2008) (Eq. 20), and a minimum value of 25 mm/yr.

However, the total runoff per analytical unit is considered as input data in MONERIS. The simple approach to calculate the groundwater recharge avoids, in most cases, a modification of the total runoff by the model, that can be applied to any river system and supports the link to hydrological models.

$$q_{\text{GWmax}} = 0.146 \left[ \frac{Q_{\text{AU}} - Q_{\text{SR}}}{A_{\text{AU}}} \cdot 86.4 \cdot 365 \right]^{1.1247} + 1.176 \left[ \frac{Q_{\text{AU}} - Q_{\text{SR}}}{A_{\text{AU}}} \cdot 86.4 \cdot 365 \right]^{0.8535} \quad (\text{Eq. 20})$$

$q_{\text{GWmax}}$  = groundwater recharge, in mm/yr

The mean groundwater residence time for each analytical unit is considered to reflect the change of land use intensities, in other words, the N surplus on agricultural areas. If groundwater residence time data are not available, they will be estimated by MONERIS using equation 21 (SCHREIBER *et al.*, 2003). Equation 21 has been derived from the long term comparison of N surpluses and N concentrations for groundwater monitoring stations in the Rhine, Elbe and Odra basins (BEHRENDT *et al.*, 2000). Depending on the period for which data were available, this correlation is only valid for a maximum of 50 years (SCHREIBER *et al.*, 2003), which is supposed to be long enough to reflect recent land use changes. Calculated groundwater residence times are aggregated into five-year classes, due to the simple approach (Eq. 21) and the limited availability of spatially distributed data for N surpluses.

$$\text{GW}_{\text{RT}} = \frac{3000}{q_{\text{GW-corr}}} \quad (\text{Eq. 21})$$

$\text{GW}_{\text{RT}}$  = groundwater residence time, in yr  
 $q_{\text{GW-corr}}$  = corrected mean long term groundwater recharge, in mm/yr

For the groundwater residence time, a mean N surplus is calculated. Because spatially distributed data on annual N surplus is often not available, MONERIS considers two different sources for N surplus data: (i) N surplus for each analytical unit for a specific year ( $N_{SY}$ ), and (ii) annual time series of N surplus on a sub-national or national administrative level ( $N_{AM}$ ).

For modelling the mean N surplus in an analytical unit during the groundwater residence time, the mean  $N_{AM}$  during the groundwater residence time is compared to  $N_{AM}$  of the specific calculation year. The change of N surplus on an administrative level ( $N_{AM-GR}/N_{AM-SY}$  ratio) is also for the respective analytical unit (Eq. 22).

$$N_{AU-GR} = N_{AU-SY} \cdot \frac{N_{AM-GR}}{N_{AM-SY}} \quad (\text{Eq. 22})$$

$N_{AU-GR}$  = mean N surplus in an analytical unit during groundwater residence time, in kg/(ha · yr)

$N_{AU-SY}$  = N surplus in an analytical unit for a specific calculation year, in kg/(ha·yr)

$N_{AM-GR}$  = mean N surplus in the respective administrative level during groundwater residence time, in kg/(ha · yr)

$N_{AM-SY}$  = N surplus in the respective administrative level for a specific calculation year, in kg/(ha · yr)

For non-fertilized areas (naturally covered areas, wetlands, open land areas, open pit mine areas, snow and ice covered areas and unsealed urban areas), the mean N surplus is not used for the calculations; instead the mean long term atmospheric deposition is used. From this, the area-weighted mean N input of fertilized and non-fertilized agricultural areas is calculated. The mean N concentration in the seepage water is calculated by equation 23.

$$C_{N-LW} = \frac{N_{in}}{q_{GR-corr}} \cdot 100 \quad (\text{Eq. 23})$$

$C_{N-LW}$  = mean N concentration in seepage water, in mg/l

$N_{in}$  = mean N input on soils, as the area-weighted mean of fertilized and non-fertilized agricultural areas, in kg/(ha · yr)

Estimation of the nitrate concentration in groundwater requires a catchment-specific model for N retention in the unsaturated and saturated vadose zone. To determine this, BEHRENDT *et al.* (2000) compared the nitrate concentration in seepage water to the groundwater nitrate concentrations of 217 monitoring stations in Germany. The N retention rates were shown to depend on the seepage water rate and on hydro-geological conditions, and led to the development of equation 24. Significant differences in the groundwater concentrations in different hydro-geological rock types could be shown and have been considered in the constants  $k_1$  and  $k_2$  of equation 24 and Table 4.

$$C_{GW-NO3} = \left[ \sum_{i=1}^4 \frac{1}{1 + k_1 \cdot q_{GR-corr}^{k_2}} \cdot \frac{A_{HG_i}}{A_{AU}} \right] \cdot C_{N-LW}^{0.637} \quad (\text{Eq. 24})$$

$C_{GW-NO3}$  = nitrate concentration in groundwater, in mg/l

$A_{HG_i}$  = areas of hydro-geological rock types, in km<sup>2</sup>

$k_1, k_2$  = model constants for different hydro-geological conditions, dimensionless

Table 4. Two model constants (Eq. 24) used to estimate N retention in four hydro-geological rock types according to BEHRENDT *et al.* (2000).

Hydro-geological rock types	$k_1$	$k_2$
Unconsolidated rock, shallow groundwater	2752.221	-1.54004
Unconsolidated rock, deep groundwater	68 561.63	-1.95861
Consolidated rock, high porosity	60.22649	-0.90311
Consolidated rock, impermeable	0.012733	0.661513

The seepage water concentrations for different land use categories are modelled following equation 23. The resulting groundwater concentrations for different land uses are calculated by applying the mean groundwater N retention (Eq. 24).

Additionally to the N emissions described above, dissolved organic N (DON) emissions via groundwater from wetlands and forests are calculated. DON is calculated using the corrected values for groundwater recharge underneath naturally covered areas and wetlands. VENOHR (2006) and VENOHR *et al.* (2010a) derived DON concentrations of 2–6 mg/l for wetlands, and of 0.5–2.5 mg/l for forests, based on studies in Irish, German and Canadian river systems. In spite of large differences in DON concentrations between different catchments, VENOHR (2006) and VENOHR *et al.* (2010a) observed a strong increase of DON concentrations with decreasing temperatures.

### 3.6.2. Phosphorus

The mean dissolved reactive phosphorus (SRP) concentrations in groundwater are calculated as the area-weighted mean of sandy, silty and loamy soils, and fens and bogs, in the respective analytical unit. The share of degraded and natural fens and bogs is considered separately for each analytical unit. Model parameters for mean SRP concentrations in water from different soil structures and from fens and bogs have been derived by BEHRENDT *et al.* (2000) from data for 217 groundwater monitoring stations in Germany, published data by BRAUN *et al.* (1991), DRIESCHER and GELBRECHT (1993) and WERNER *et al.* (1991) (Eq. 25).

For arable land and grassland (without tile-drained areas) the mean P groundwater concentration is calculated by equation 25. For naturally covered areas, wetland, open land areas and unsealed urban areas, the mean concentration of SRP in the groundwater is set to 0.02 mg/l also derived on basis of data published by BRAUN *et al.* (1991), DRIESCHER and GELBRECHT (1993) and WERNER *et al.* (1991).

$$C_{\text{GWAGRI-P}} = \frac{0.1 \cdot A_S + 0.03 \cdot (A_C + A_L + A_{SI}) + 0.1 \cdot A_{FD} + 0.02 \cdot A_{FN} + 0.5 \cdot A_{BD} + 0.035 \cdot A_{BN}}{A_S + A_L + A_{FD} + A_{FN} + A_{BD} + A_{BN}} \quad (\text{Eq. 25})$$

$C_{\text{GWAGRI-P}}$  = P concentration in groundwater under agricultural land use, in mg/l

$A_S$  = areas of sandy soil, in km<sup>2</sup>

$A_C$  = areas of clay soil, in km<sup>2</sup>

$A_L$  = areas of loamy soil, in km<sup>2</sup>

$A_{SI}$  = areas of silty soil, in km<sup>2</sup>

$A_{FD}$  = areas of degraded fens, in km<sup>2</sup>

$A_{FN}$  = areas of natural fens, in km<sup>2</sup>

$A_{BD}$  = areas of degraded bogs, in km<sup>2</sup>

$A_{BN}$  = areas of natural bogs, in km<sup>2</sup>

In aerated groundwater, there are usually only small differences between the total P concentration and the SRP concentration. In contrast, in anaerobic groundwater, the total P concentrations can be up to five times higher than the SRP concentration (BEHRENDT, 1996; DRIESCHER and GELBRECHT, 1993). To distinguish between aerobic and anaerobic conditions, the ratio between nitrate concentrations in groundwater and in seepage water is used. If this ratio is less than 0.1, indicating anaerobic conditions, the total P concentration in the groundwater is assumed to be 2.5 times higher than the SRP concentration (BEHRENDT *et al.*, 2000). For aerobic conditions the total P concentration is assumed to equal the SRP concentration.

### 3.7. Nutrient Emissions via Urban Systems

Nutrient emissions from urban systems are calculated for sealed urban areas connected or not connected to sewer systems, as well as for households not being connected to sewer systems or waste water treatment plants (WWTP). Households connected to sewer systems and to WWTP are not accounted for in the calculated emissions from urban systems as they should be considered in the WWTP inventory. Storm water events generating high runoff from sealed areas are crucial for emissions from urban systems. Here, the increased discharges from combined sewers can often not be handled by WWTP and have to be stored in the sewer systems. If the storage capacity in combined sewer systems is exceeded, the excessive water amount is bypassed directly, allowing raw sewage from households, commercial use, and streets to reach surface waters during overflow events. Beyond combined sewer systems, MONERIS considers four more pathways for nutrient emissions from urban areas (Fig. 11): rainwater collected via separate sewer systems, decentralized treatment plants, sewer systems without WWTP, and not connected inhabitants and areas.

For calculation of the N and P concentrations in sewages from households, MONERIS uses inhabitant-specific emissions. For N, inhabitant-specific emissions ( $\text{Inh}_N$ ) vary between 9 and 12 g/(inhabitant · d) (ATV, 1997, LINDTNER and ZESSNER, 2003; WERNER *et al.*, 1991), and can be adapted to the situation in the particular river system. For P, inhabitant-specific P emissions are set to 1.65 g/(inhabitant · d), excluding phosphates from laundry and dishwashers. Since the usage of phosphates in dishwasher and laundry detergents varies from country to country, the total specific values for dissolved P emissions from humans and detergents are considered on a country level.

For calculation of N emissions, MONERIS uses the area-specific value of 4 kg/(ha · yr) from litter and excrements, in addition to the values for atmospheric deposition. The area-specific P emission from sealed urban areas by atmospheric deposition, litter, and excrement, amounts to 2.5 kg/(ha · yr) (BROMBACH and MICHELBAACH, 1998).

#### 3.7.1. Calculation of Sealed Urban Areas

The total urban area can be deduced from a land use map. The percentage of this urban area that is sealed can then be calculated based on the population density, using the approach of HEANEY *et al.* (1976) (see Eq. 26). For equation, a maximum population density of < 150 inhabitants/ha is defined, which corresponds to a maximum of 92% of sealed urban areas.

$$A_{\text{SUA}} = 9.6 \cdot (0.4047 \cdot \text{POP}_{\text{DENS}})^{(0.573 - 0.0391 \cdot \log(0.4047 \cdot \text{POP}_{\text{DENS}}))} \quad (\text{Eq. 26})$$

$A_{\text{SUA}}$  = sealed urban areas, in %

$\text{POP}_{\text{DENS}}$  = population density, in inhabitants/ha for  $\text{POP}_{\text{DENS}} < 150/\text{ha}$

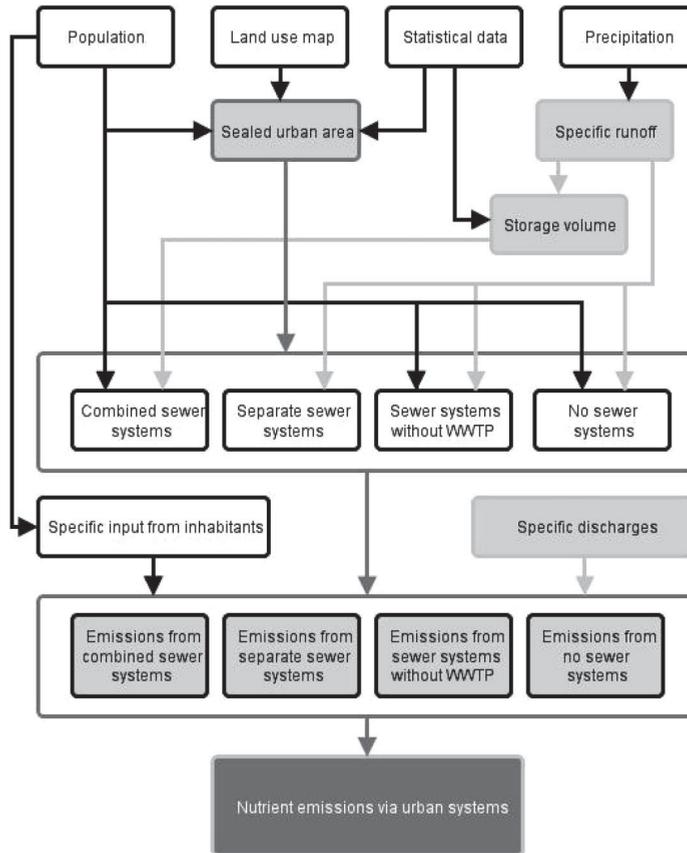


Figure 11. Main steps for calculation of nutrient emissions via urban systems (diagram excludes the pathway for decentralized treatment plants).

Percentages of the sealed urban areas are proportionally allocated to the corresponding sewer systems, according to the length of the combined and separate sewer systems, and the connection rates of inhabitants.

### 3.7.2. Runoff from Sealed Urban Areas

The runoff rate from sealed urban areas that is generated from precipitation is calculated according to HEANEY *et al.* (1976) (Eq. 27). The higher the percentage of the area that is sealed, the larger is the runoff rate, and thus the larger the part of precipitation reaching the sewer systems.

$$Q_{R-SUA} = 0.15 + 0.75 \cdot A_{SUA} \cdot 0.01 \quad (\text{Eq. 27})$$

$Q_{R-SUA}$  = runoff rate from sealed urban areas, dimensionless

Accordingly, the area-specific runoff from sealed urban areas can be calculated, depending on the size of the sealed urban area, the runoff rate, and the annual precipitation. The total runoff from sealed urban areas is calculated as the sum of runoff from all urban pathways.

### 3.7.3. Nutrient Emissions via Combined Sewer Systems

During heavy rainfall events, WWTP cannot handle the entire water coming from combined sewer systems and have to be stored in the sewer systems itself. Is the storage volume of the combined sewer system exceeded, excess water is bypassed directly, without treatment, into surface waters via the combined sewer overflow (CSO).

The discharge rate during CSO events depends on the storage volume in the combined sewer system. BROMBACH and MICHELBAACH (1998) and MEISSNER (1991) anticipated that a storage volume of 23.3 m<sup>3</sup>/ha corresponds to a storage capacity of 100%. The precipitation-runoff rate is assumed to be 1 l/(ha · s) (q<sub>R</sub>) for all analytical units. The discharge rate according to MEISSNER (1991) is described in equation 28:

$$Q_{R-CSO} = \left[ \left( \frac{\frac{4000 + 25 \cdot q_R}{0.551 + q_R}}{\frac{CSV}{100} \cdot 23.3 + \frac{36.8 + 13.5 \cdot q_R}{0.5 + q_R}} \right) - 6 + \frac{PR_{yr} - 800}{40} \right] \quad (\text{Eq. 28})$$

- Q<sub>R-CSO</sub> = discharge rate via CSO, in %
- PR<sub>yr</sub> = annual precipitation, in mm/yr
- q<sub>R</sub> = precipitation-runoff rate, in l/(ha · s)
- CSV = storage volume of combined sewer systems, in %

For calculating the discharge by CSO events, the number of days with heavy rainfall events is calculated according to MOHAUPT *et al.* (1998) on the basis of the annual precipitation (Eq. 29).

$$DHR = PR_{yr}^{2.5} \cdot 0.0000012 \quad (\text{Eq. 29})$$

DHR = number of heavy rainfall events, in d/yr

The effective number of heavy rainfall days (Eq. 30) describes the number of days, which actually cause a CSO event. Here, the discharge rate via CSO is calculated with and without (CSV = 0) considering the storage volume in the combined sewer system. Consequently, in combined sewer systems with a high storage volume, heavy rainfall events less often cause an overflow event.

$$DHR_{EFF} = DHR \cdot \frac{Q_{SP}}{Q_{SP-nostorage}} \quad (\text{Eq. 30})$$

- DHR<sub>EFF</sub> = effective number of days of heavy rainfall events, in d/yr
- Q<sub>SP</sub> = CSO discharge rate with consideration of storage volume in sewer systems, in %
- Q<sub>SP-nostorage</sub> = CSO discharge rate without consideration of storage volume in sewer system, in %

Calculation of the nutrient emission via CSO uses the nutrient loads in the combined sewer systems of an entire day. Although a CSO event may not be of an entire day's duration, sewage from households stays in the combined sewer systems for several hours or even days.

To calculate the total discharge during CSO events, three sources are taken into account: (i) water from households (inhabitants), (ii) water from commercial areas, and (iii) precipitation on areas connected to combined sewer systems. The total annual discharge in combined sewer systems generated by precipitation is calculated by equation 31.

$$Q_{\text{CSO-P}} = A_{\text{CS}} \cdot \text{PR}_{\text{yr}} \cdot Q_{\text{R-SUA}} \cdot 1000 \quad (\text{Eq. 31})$$

$Q_{\text{CSO-P}}$  = discharge in combined sewer systems generated by precipitation, in  $\text{m}^3/\text{yr}$

$A_{\text{CS}}$  = sealed urban areas connected to combined sewer systems, in  $\text{km}^2$

$Q_{\text{R-SUA}}$  = see (Eq. 27)

For calculating the amount of waste water from commercial areas, the approach of BROMBACH and MICHELBAACH (1998) and MOHAUPT *et al.* (1998) is applied. This approach assumes that 0.8% of the total urban area is commercial area, having a runoff rate of 1 l/(ha · s) for 10 hours after heavy rainfall events (Eq. 32).

$$Q_{\text{CSO-CA}} = \left( 1 \cdot 0.8 \cdot A_{\text{CS}} \cdot \frac{10}{24} \cdot \frac{86400}{1000} \right) \quad (\text{Eq. 32})$$

$Q_{\text{CSO-CA}}$  = discharge in combined sewer systems generated from precipitation on commercial areas, in  $\text{m}^3/\text{d}$

The discharges from inhabitants in households connected to combined sewers are calculated on basis of an inhabitant-specific daily waste water production, which is assumed to be 130 l/(inhabitants · d) (Eq. 33) (BROMBACH and MICHELBAACH, 1998; MOHAUPT *et al.*, 1998).

$$Q_{\text{CSO-Inh}} = \text{Inh}_{\text{CS}} \cdot \frac{130}{1000} \quad (\text{Eq. 33})$$

$Q_{\text{CSO-Inh}}$  = discharge in the combined sewer system generated from inhabitants, in  $\text{m}^3/\text{d}$

$\text{Inh}_{\text{CS}}$  = number of inhabitants in households connected to combined sewer systems

The total discharge from inhabitants and from precipitation from sealed urban areas emitted via CSO events is calculated using equation 34. The influence of external water intrusion into sewer systems is excluded, as it is assumed that the sewer systems act as donors of water during heavy rainfall events.

$$Q_{\text{CSO-total}} = ((Q_{\text{CSO-Inh}} + Q_{\text{CSO-CA}}) \cdot \text{DHR}_{\text{EFF}} + Q_{\text{CSO-P}}) \cdot \frac{Q_{\text{R-CSO}}}{100} \quad (\text{Eq. 34})$$

$Q_{\text{CSO-total}}$  = total discharge via spillways during CSO, in  $\text{m}^3/\text{yr}$

$\text{DHR}_{\text{EFF}}$  = see equation 30

For the calculation of the P concentration in combined sewer systems during overflow events, the P emissions from commercial areas are also considered. The N and P concentra-

tions in combined sewer systems at overflow events are calculated according to equations 35 and 36 respectively.

$$TNC_{CSO} = \frac{\left( Inh_{CS} \cdot Inh_N + (AD_N + SUA_N \cdot 100) \frac{Q_{R-SUA} \cdot A_{CS} \cdot 10}{365} \right) \cdot DHR_{EFF}}{Q_{CSO-total} \cdot \frac{100}{Q_{R-CSO}} + \frac{A_{CS} \cdot 23.3}{100000} \cdot DHR_{EFF} \cdot \frac{CSV}{100}} \quad (\text{Eq. 35})$$

$TNC_{CSO}$  = TN concentration during CSO, in mg/l

$Inh_N$  = inhabitant-specific N emissions, in g/(inhabitant · d)

$AD_N$  = atmospheric  $NO_x$  and  $NH_y$  deposition on urban areas, in kg/(km<sup>2</sup> · yr)

$SUA_N$  = specific TN emissions from sealed urban areas by litter, animal excrement, and traffic, in kg/(ha · yr) (currently set to 4 kg/(ha · yr))

$Q_{R-SUA}$  = see equation 27

$CSV$  = storage volume of combined sewer systems, in %

$DHR_{EFF}$  = see equation 30

$Q_{R-CSO}$  = see equation 28

$$TPC_{CSO} = \frac{\left( Inh_{CS} \cdot Inh_P + SUA_P \cdot 100 \frac{Q_{R-SUA} \cdot A_{CS} \cdot 10}{365} \right) \cdot DHR_{EFF}}{Q_{CSO-total} \cdot \frac{100}{Q_{R-CSO}} + \frac{A_{CS} \cdot 23.3}{100000} \cdot DHR_{EFF} \cdot \frac{CSV}{100}} \quad (\text{Eq. 36})$$

$TPC_{CSO}$  = TP concentration during CSO, in mg/l

$Inh_P$  = inhabitant-specific P emissions, in g/(inhabitant · d)

$AD_P$  = atmospheric P deposition on urban areas, in kg/(km<sup>2</sup> · yr)

$SUA_P$  = specific TP emissions from sealed urban areas by litter, animal excrement, and traffic, in kg/(ha · yr) (currently set to 2.5 kg/(ha · yr))

$DHR_{EFF}$  = see equation 30

$QR_{CSO}$  = see equation 28

Finally the nutrient emissions via combined sewer systems during overflow events is calculated as the product of the nutrient concentration in combined sewer systems at overflow events and the total discharge via CSO from sealed urban areas and households connected to combined sewer systems.

### 3.7.4. Nutrient Emissions via Rainwater-Collecting Separate Sewer Systems

Separate sewer systems collecting black water from households and industries are connected to WWTPs, and the waste water of such separate sewer systems is considered within the pathway point sources. Discharge from sealed urban areas including commercial areas connected to separate sewer systems collecting rain water is calculated in equation 37.

$$Q_{SS-total} = A_{SS} \cdot PR_{yr} \cdot Q_{R-SUA} \cdot 1000 \quad (\text{Eq. 37})$$

$Q_{SS-total}$  = total discharge from rainwater-collecting sewer systems, in  $m^3/yr$   
 $A_{SS}$  = sealed urban areas connected to separate sewer systems, in  $km^2$   
 $Q_{R-SUA}$  = see equation 27  
 $PR_{yr}$  = annual precipitation, in  $mm/yr$

Loads from rainwater-collecting separate sewer systems are partly treated in clarifier basins and in retention soil filters before being discharged to surface waters. Information about the share of the clarifier basins and retention soil filters has to be given as input data in MONERIS. The emissions finally depend on the subsequent retention in the clarifier basins and in retention soil filters. The retention in clarifier basins and in retention soil filters are set to 35% and 85% for N, and 35% and 45% for P, respectively.

Nutrient emissions via separate sewer systems are calculated based on area-specific emissions and atmospheric deposition by equations 38 and 39.

$$TNC_{SS} = \left( \frac{AD_N}{PR_{yr}} + \frac{SUA_N \cdot 10000}{PR_{yr} \cdot Q_{R-SUA}} \right) \cdot \left( 1 - \frac{0.35 \cdot RCB}{100} \right) \cdot \left( 1 - \frac{0.85 \cdot RSF}{100} \right) \quad (\text{Eq. 38})$$

$TNC_{SS}$  = N concentration from separate sewer systems after retention, in  $mg/l$   
 $RCB$  = share of retention clarifier basins in separate sewer system, in %  
 $RSF$  = share of retention soil filter in separate sewer system, in %  
 $Q_{R-SUA}$  = see equation 27

$$TPC_{SS} = \left( \frac{SUA_P \cdot 10000}{PR_{yr} \cdot Q_{R-SUA}} \right) \cdot \left( 1 - \frac{0.35 \cdot RCB}{100} \right) \cdot \left( 1 - \frac{0.45 \cdot RSF}{100} \right) \quad (\text{Eq. 39})$$

$TPC_{SS}$  = N concentration from separate sewer systems after retention, in  $mg/l$

The nutrient emissions via separate sewer systems are calculated as product of the nutrient concentration from separate sewer systems after treatment, and the discharge from rainwater-collecting separate sewer systems.

### 3.7.5. Nutrient Emission from Sealed Urban Areas and Inhabitants that Are Connected to Sewer Systems, but not to Municipal WWTP

The calculation of emissions from areas and households connected to sewer systems but not to WWTPs, (in the following called “connected to sewer systems only (OS)”), are conducted analogous to the approaches used for combined and separate sewer systems. In contrast to separate sewer systems, no treatment in clarifier basins and in retention soil filters is assumed (these treatments apply only in separate sewer systems). Also in contrast to combined sewer systems, in OS systems the total discharge is considered (not only the discharge at overflow events). Discharge from sealed urban areas connected to sewer systems only is calculated in equation 40.

$$Q_{OS-total} = A_{OS} \cdot PR_{yr} \cdot Q_{R-SUA} + \left( \text{Inh}_{OS} \cdot \frac{130}{1000} \right) \cdot 365 \quad (\text{Eq. 40})$$

$Q_{OS-total}$  = discharge from sealed urban areas and inhabitants that are connected to sewer systems only, in  $m^3/yr$   
 $A_{OS}$  = sealed urban areas connected to sewer system only, in  $km^2$

$Inh_{OS}$  = number of inhabitants connected to sewer systems only

$Q_{R-SUA}$  = see equation 27

Nutrient concentrations from sealed urban areas and from inhabitants connected to sewer systems only are calculated with equations 41 and 42.

$$TNC_{OS} = \frac{(Inh_{OS} \cdot Inh_N) \cdot 365 + (AD_N + SUA_N \cdot 100) \cdot Q_{R-SUA} \cdot A_{OS} \cdot 10}{Q_{OS-total} \cdot 86400 \cdot 365} \quad (\text{Eq. 41})$$

$TNC_{OS}$  = TN concentration in discharge from inhabitants and sealed urban areas connected to sewer systems only, in mg/l

$Q_{R-SUA}$  = see equation 27

$$TPC_{OS} = \frac{(Inh_{OS} \cdot Inh_N) \cdot 365 + SUA_N \cdot Q_{R-SUA} \cdot A_{OS} \cdot 1000}{Q_{OS-total} \cdot 365 \cdot 86400} \quad (\text{Eq. 42})$$

$TPC_{OS}$  = TP concentration in discharge from inhabitants and sealed urban areas connected to sewer systems only, in mg/l

$Q_{R-SUA}$  = see equation 27

### 3.7.6. Nutrient Emission from Sealed Urban Areas and Inhabitants that are Connected to DCTP with or without Sewer Systems

Decentralised wastewater treatment plants (DCTP) are onsite or cluster wastewater systems that are used to treat and dispose relatively small volumes of wastewater, generally originating from individual or groups of dwellings and businesses that are located relatively close together. Onsite and cluster systems are commonly used in combination. Different DCTP are distinguished in the model, according to their technical status and how loads are discharged to the surface waters. The DCTP can be constructed according to different legal regulations. In Germany under DIN 4261 01 (1991) (DIN = German industry norm), without wastewater aeration, a retention capacity of 10% for N and 7% for P can be achieved, and according to DIN 4261 02 (1984) with wastewater aeration, a retention capacity of 15% for N and 13% for P can be achieved. Emissions can be discharged directly via sewer systems or pipes and ditches, or indirectly via infiltration to soil and groundwater. In the first case, no further retention subsequent to the DCTP is assumed. For the second case, additionally to the retention in the DCTP, retention during the soil and groundwater passage is considered. The retention in soil and groundwater is calculated following the method described for the groundwater pathway. Analogous to the calculation for the other sewer systems, inhabitant-specific TN and TP are considered for DCTP (Eq. 43).

$$TN/TP_{DCTP} = \frac{RN/P_{SGW}}{100} \cdot Inh_{DCTP} \cdot Inh_N \cdot 0.365 \cdot \left(1 - \frac{R_{DCTP}}{100}\right) \quad (\text{Eq. 43})$$

$TN/TP_{DCTP}$  = nutrient emissions from DCTP, in t/yr

$RN/P_{SGW}$  = nutrient retention in soil and groundwater analogue to equation 24, in %

$Inh_{DCTP}$  = inhabitants connected to one of the different DCTP types (as described above)

$R_{DCTP}$  = nutrient retention capacity of DCTP, distinguished for DCTP with or without waste water aeration, in %

### 3.7.7. Nutrient Emissions from Sealed Areas and Households neither Connected to Sewer Systems nor to WWTP

In rural areas or in older settlements, households are sometimes neither connected to a sewer system nor to WWTP. Households, should, nevertheless, be connected to a septic tank. For this it is assumed that 90% of the dissolved inhabitant-specific emissions are transported to a WWTP (Eqs. 44 and 45). For the remaining 10% from septic tanks, and for emissions from not connected sealed urban areas, the retention in soil and groundwater is taken into account.

$$TN_{NC} = \frac{RN_{SGW}}{100} \cdot \left( Inh_{ST} \cdot Inh_N \cdot \frac{0.365}{1000} \cdot \left( 1 - \frac{FR_{WWTP}}{100} \right) + \left( \frac{AD_N + SUA_N \cdot 100}{1000} \right) \cdot A_{NC} \right) \quad (\text{Eq. 44})$$

$TN_{NC}$  = TN emissions from households and sealed urban areas neither connected to sewer systems or WWTP, in t/yr

$Inh_{ST}$  = number of inhabitants in households connected to septic tanks

$FR_{WWTP}$  = fraction of dissolved specific inhabitant emissions transported to WWTP, in %

$A_{NC}$  = sealed urban areas neither connected to sewer system nor to WWTP, in km<sup>2</sup>

$$TP_{NC} = \frac{RP_{SGW}}{100} \cdot \left( Inh_{ST} \cdot Inh_P \cdot \frac{0.365}{1000} \cdot \left( 1 - \frac{FR_{WWTP}}{100} \right) + \left( \frac{SUA_P \cdot 100}{1000} \right) \cdot A_{NC} \right) \quad (\text{Eq. 45})$$

$TP_{NC}$  = TP emissions from households and sealed urban areas neither connected to sewer systems or to WWTP, in t/yr

### 3.8. Nutrient Emissions via Point Sources

Nutrient emissions via point sources are taken from an inventory with information on individual waste water treatment plants (WWTP). Additionally, lumped discharges from industrial direct dischargers and remaining smaller point sources can be considered for each analytical unit. For the WWTP inventory data on the discharge, the TN and TP concentrations and the size as inhabitant equivalents are needed. Temporal changes in discharges from waste water treatment plants can be considered by factors for individual years in each analytical unit. For the later calculation of retention in surface waters it can also be considered whether point sources discharge into tributaries or directly into the main river of an analytical unit. In MONERIS usually all emissions are assumed to first reach the tributaries. In reality, WWTPs are normally located at large streams and therefore discharge into these main rivers. In case that it is known whether the WWTP discharge into the tributaries or the main river, it can be separately considered for the calculations.

### 3.9. Retention in Surface Waters

In surface waters, retention – which is the sum of all nutrient transformation and loss processes – is an important element of the nutrient cycle. In MONERIS we consider net nutrient retention rather than the contribution of individual chemical and biological transformation, or non-permanent retention processes.

Retention processes include nitrification-denitrification, plant uptake, sedimentation, and decomposition of dissolved organic matter. For N, the dominant retention process is denitrification, whereas for P it is sedimentation (MULHOLLAND *et al.*, 2000; SAUNDERS and KALFF,

2001b; SEITZINGER, 1988; SVENDSEN and KRONVANG, 1993; TRISKA *et al.*, 1994; VENOHR, 2006). For dissolved organic nitrogen (DON), retention in surface waters is assumed to be negligible.

For the retention calculation, it is assumed that: (i) nutrient emissions are evenly distributed throughout the catchment, (ii) all emissions reach the tributaries before the main rivers, and (iii) tributaries discharge into the main river at the outlet of the respective analytical unit. Thus, nutrient loads coming from upstream catchments are subject to the retention in the main river of the downstream analytical unit. In addition, for the situation where lakes are flushed by the main river at the outlet of an analytical unit, an additional retention can be considered for all incoming loads from tributaries and main rivers.

To model N retention, the THL-approach by VENOHR (2006) is used, which considers water temperature (T) and hydraulic load (HL) (Eq. 46). The P retention in tributaries is calculated using the approaches developed by BEHRENDT and OPITZ (2000) and BEHRENDT *et al.* (2000), which quantifies the retention as a function either of the hydraulic load or of the specific runoff, and calculates a mean retention based on both of these parameters. The P retention in main rivers is calculated according to the hydraulic load only (specific runoff is not suitable for the retention calculation in main rivers, as it does not change with the length of the respective river stretch (VENOHR, 2006)).

The retention processes in specific lakes or river stretches are closely related to the nutrient residence time, and thus also to the flow velocity (Eq. 46). As shown in equation 46, residence time and flow velocity can mathematically be transferred into the hydraulic load, which is used for the retention calculation with MONERIS. Water depth and flow velocity are both needed to calculate the residence time and are difficult to model, even on a large scale, in contrast to the water surface area, which can be easily calculated and is often also available from maps or statistical reports (see Chapter 3.1). Consequently, the hydraulic load has been chosen as modelling parameter for the retention calculation, to allow a wider applicability of the model.

$$\frac{z}{\tau} = \frac{z}{\frac{V}{Q_a}} = \frac{z}{\frac{WSA \cdot z}{Q_a}} = \frac{Q_a}{WSA} = HL \quad (\text{Eq. 46})$$

- $z$  = mean depth of a water body, in m
- $\tau$  = mean residence time in a water body, in yr
- $V$  = mean volume of a water body, in  $\text{m}^3$
- $Q_a$  = specific runoff, in  $\text{m}^3/\text{yr}$
- $WSA$  = water surface areas,  $\text{m}^2$
- $HL$  = hydraulic load, in  $\text{m}/\text{yr}$

### 3.9.1. Basic Approach to Model Retention in Surface Waters

The retention is based on a general mass balance equation for mixed reactors (Eq. 47).

$$\frac{dC(t)}{dt} = \frac{Q}{V} \cdot (C_{in}(t) - C(t)) - C(t) \cdot k \quad (\text{Eq. 47})$$

- $C(t)$  = calculated nutrient concentration as a function of time, in  $\text{mg}/\text{l}$
- $C_{in}(t)$  = mean nutrient concentration of emissions, in  $\text{mg}/\text{l}$
- $Q$  = runoff, in  $\text{m}^3/\text{s}$
- $k$  = retention rate, in  $\text{s}^{-1}$

If the speed of the contributing retention processes is much higher than the mean residence time in a surface water body, steady state conditions can be assumed for the retention calculation. Under these conditions, equation 47 can be simplified to:

$$\frac{Q}{V} \cdot (C_{in} - C) - C \cdot k = 0 \quad (\text{Eq. 48})$$

If we assume that  $Q \cdot C_{in}$  are the emissions,  $Q \cdot C$  is the load and  $C \cdot V \cdot k$  is the retention. Equation 48 can also be written as in equation 49, which is the basic structure for the retention approach in MONERIS.

$$L = \frac{1}{1 + R_L} E \quad (\text{Eq. 49})$$

$L$  = load in surface waters, in t/yr

$E$  = emissions to surface waters, in t/yr

$R_L$  = load-weighted retention coefficient ( $R/L$ )

If steady state conditions are given,  $N$  retention can be calculated with equation 50. On an annual basis this approach is used to calculate TN and DIN retention.

$$R_{L-TN/DIN} = 1 - \frac{1}{1 + a \cdot e^{bT} \cdot HL^{-1}} \quad (\text{Eq. 50})$$

$R_{L-TN/DIN}$  = load weighted retention for TN and DIN, in %

$T$  = water temperature, in °C

$a$  = constant, in yr/m

$b$  = constant, in  $K^{-1}$

For P only TP retention is calculated. As discussed above, BEHRENDT and OPITZ (2000) used hydraulic load and specific runoff for the P retention approach (Eqs. 51 and 52), and the mean of both is used in MONERIS.

$$R_{L-PHL} = 1 - \frac{1}{1 + a_1 \cdot HL^{-1}} \quad (\text{Eq. 51})$$

$$R_{L-Pq} = 1 - \frac{1}{1 + a_2 \cdot q^{-1}} \quad (\text{Eq. 52})$$

$R_{L-PHL}$  = load weighted P retention, using HL as driving parameter, in %

$R_{L-Pq}$  = load weighted P retention, using specific runoff as driving parameter, in %

$q$  = area specific runoff, in  $l/(s \cdot km^2)$

$a_1$  = constant, yr/m

$a_2$  = constant, in  $(s \cdot km^2)/l$

### 3.9.2. Calibration of the Retention Approaches

Parameter  $b$  (Eq. 50) describes the temperature dependence of N retention, reflecting the increase in biological activity with increasing temperature. Different methodological approaches have determined values for parameter  $b$  of between  $-0.04$  and  $-0.11$  (HILL, 1983; MOHAUPT, 1985; SAUNDERS and KALFF, 2001b; TOMS *et al.*, 1975; VAN LUIJN *et al.*, 1999), with a mean of  $-0.067$ .

To verify the values for  $b$  described in the literature, VENOHR (2006) statistically analysed the change of N concentrations with temperature at 49 monitoring sites in Germany from 1983 to 1997 (Fig. 12). Figure 12 shows the ratio between the N concentration at a given temperature and the concentration at the mean temperature, and shows how this ratio changes with changing water temperature. The average ratio of all available measurements for all monitoring stations was calculated subsequently for groups of  $0.5\text{ }^{\circ}\text{C}$  steps, resulting in 54 classes. The statistical analysis delivered a value of  $-0.056$  for  $b$  ( $r^2 = 0.86$ ;  $N = 54$ , not shown in Figure 12), which is in the lower range of published mean value of  $b$  (see above). However, VENOHR (2006) showed that a value for  $b$  of  $-0.067$  does not deliver results significantly different from those using  $-0.056$  for  $b$ . Finally,  $-0.067$  for  $b$  is used in MONERIS to represent the temperature dependence of N retention.

The N retention approach ( $R_{L-DIN/TN}$ ) was calibrated based on N concentration and temperature measurements from 59 catchments in Germany (39 (DIN) and 20 (TN)) for 1993–1997 (VENOHR, 2006). The 5-year mean nutrient emissions calculated with MONERIS compared with the 5-year mean of observed loads were used to calibrate the N retention approach (Table 5).

According to (VENOHR, 2006), the exponent for the hydraulic load or the specific runoff used to calculate  $R_{L-DIN}$ ,  $R_{L-TN}$ ,  $R_{L-Pq}$ ,  $R_{L-PHL}$ , has been set to a value of  $-1$  in order to avoid scaling effects, when applying the approaches to river catchments of different size. This was considered for the calibration of the N retention approaches. For P BEHRENDT and OPITZ (2000) derived exponents for  $q$  and  $HL$  different from 1. Therefore, for the application in MONERIS the approaches were re-calibrated with the dataset described by BEHRENDT and OPITZ (2000).

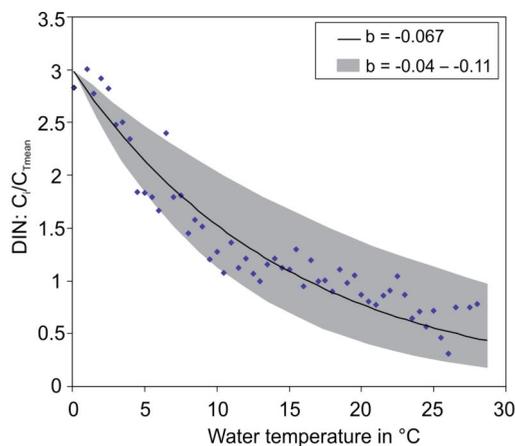


Figure 12. Dependence of the ratio  $DIN C_i : DIN C_{Tmean}$  (DIN concentrations of an individual measurement ( $C_i$ ), and the DIN concentration at the mean water temperature ( $C_{Tmean}$ ) of a monitoring station), on the water temperature. Temperatures grouped in  $0.5\text{ }^{\circ}\text{C}$  classes (VENOHR, 2006).

Table 5. Parameter for calculating DIN, TN and TP retention with Eqs. 51–53.

Retention approaches	a	Mean absolute Deviation in%	$r^2$	Modelling Efficiency	N	Equation
R <sub>L-DIN</sub>	8.58	22.3	0.71	0.57	38	51
R <sub>L-TN</sub>	4.74	26.9	0.76	0.67	20	51
R <sub>L-Pq</sub>	8.77	21.6	0.98	0.84	89	52
R <sub>L-PHL</sub>	15.91	31.0	0.97	0.73	89	53

### 3.10. Monthly Disaggregation of Emissions

MONERIS disaggregates annual emissions to monthly values, rather than calculating monthly emissions directly. This is appropriate because many area-covering input data (such as N surplus, inhabitants, and atmospheric deposition) are only available on an annual basis or only show a small inner-annual variation. The MONERIS approach to disaggregate the emissions describes the total monthly emissions and the resulting monthly loads, rather than modelling dynamics of the individual pathways.

For tile drainages, groundwater, and other diffuse pathways, the mean annual concentrations calculated by MONERIS are used. For the disaggregation of nutrient emissions from annual to monthly values, it is assumed that the nutrient concentrations do not change within years. Consequently, precipitation and runoff are the main driving factors for the seasonal variability of nutrient emissions. For runoff and precipitation, monthly input data is considered. Water temperature and incoming short wave radiation at ground level are used for the monthly retention calculation.

For diffuse pathways, the mean annual concentrations have been calculated based on annual diffuse emissions and mean annual runoff (Eq. 53).

$$C_a = \frac{E_a}{Q_a} \cdot \frac{1000000}{86400 \cdot 365} \quad (\text{Eq. 53})$$

$C_a$  = mean annual concentration from pathway(s), in mg/l

$E_a$  = annual emissions from pathway(s), in t/yr

$Q_a$  = mean runoff from pathway(s), in m<sup>3</sup>/s

To calculate monthly runoff for the pathways, a variety of approaches are used. Monthly surface runoff, runoff from urban systems, and precipitation on surface waters, is estimated according to the distribution of runoff during the year (Eq. 54). Although, these pathways are driven by precipitation, for the disaggregation to monthly values the generated runoff has to be taken into account.

$$Q_{\text{diff-M}} = Q_{\text{diff-Y}} \cdot \frac{Q_M}{Q_Y} \quad (\text{Eq. 54})$$

$Q_{\text{diff-M}}$  = monthly runoff as the sum of monthly runoffs from surface runoff, urban systems, and direct precipitation on surface waters, in m<sup>3</sup>/s

$Q_{\text{diff-Y}}$  = annual runoff as the sum of annual runoffs from surface runoff, urban systems, and direct precipitation on surface waters, in m<sup>3</sup>/s

$Q_M$  = monthly runoff, in m<sup>3</sup>/mo

$Q_Y$  = annual runoff, in m<sup>3</sup>/yr

Note that  $Q_{diff}$  is the part of total runoff without the runoff from tile drainages and groundwater.

Monthly flow rates from tile drainages are calculated as a percentage of the monthly precipitation (Eq. 55), based on HIRT *et al.* (2011) following KRETSCHMAR (1977). There is marked seasonal variation in the percentage of precipitation leading to tile drainage runoff, being much higher in winter than in summer; for sites in north-west Europe, this is shown in Figure 13. In case, the annual flow from tile drainages is calculated according to (Eq. 17), it is obligatory to use this equation also on a monthly basis to avoid differences between the annual sum and the sum of the monthly tile drainage flows.

$$Q_{TD-M} = RR_{TD} \cdot PR_M \cdot A_{TD} \cdot 1000 \tag{Eq. 55}$$

$Q_{TD-M}$  = monthly runoff from tile drainages, in m<sup>3</sup>/mo

$RR_{TD}$  = share of the monthly runoff rate on the monthly precipitation, in %

$A_{TD}$  = tile drained areas, in km<sup>2</sup>

$PR_M$  = monthly precipitation, in mm/mo

Runoff from groundwater is calculated as a residual of the total runoff from an analytical unit, minus the discharge originating from surface runoff, urban areas, direct precipitation on surface waters, and from tile drainages (Eq. 56). If the total monthly runoff is not available from other sources, *i.e.* models, it can be calculated by using the mean yearly runoff corrected by the ratio of mean monthly to mean yearly runoff for the next representative gauging station.

Equation 17 can be used for an annual model application. For the monthly disaggregation however, equation 55 should be applied. In this case the annual sum of the monthly tile drainage flow should be used instead.

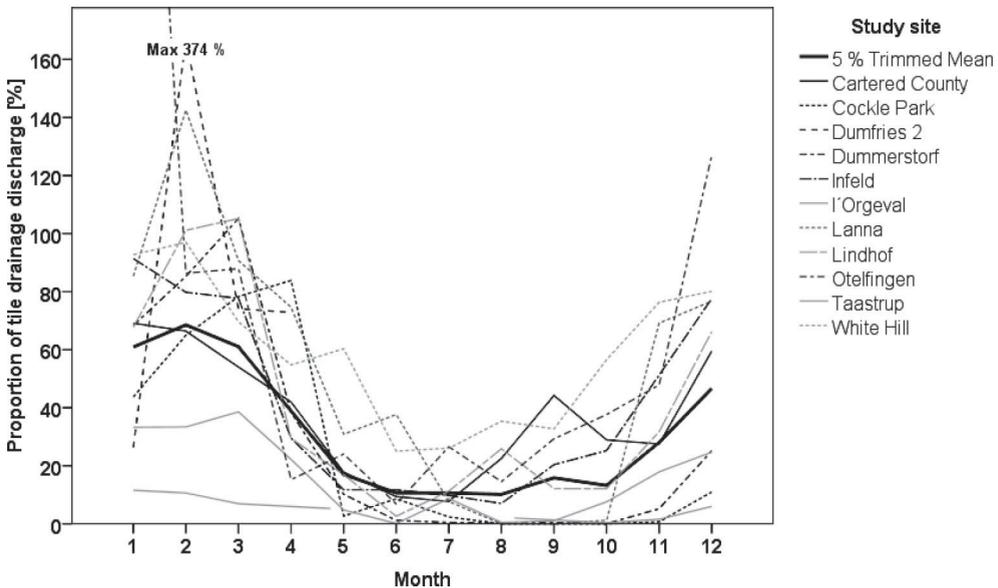


Figure 13. Monthly changes in percentage of precipitation leading to runoff from tile drainages; data is for 11 sites in north-west Europe, and their mean (solid black line) (HIRT *et al.*, 2011).

$$Q_{\text{GW-M}} = Q_{\text{tot-M}} - Q_{\text{diff-M}} - Q_{\text{TD-M}} \quad (\text{Eq. 56})$$

$Q_{\text{GW-M}}$  = mean monthly discharge from groundwater, in  $\text{m}^3/\text{s}$

$Q_{\text{tot-M}}$  = mean total runoff from sub-catchment, in  $\text{m}^3/\text{s}$

$Q_{\text{diff-M}}$  = mean monthly runoff as the sum of monthly runoffs from surface runoff, urban systems, and direct precipitation on surface waters, in  $\text{m}^3/\text{s}$

Emissions from point sources, (waste water treatment plants and industrial direct dischargers), were assumed to remain constant during the year and be equally distributed among the twelve months.

### 3.10.1. Monthly Retention Calculation

For TN, the major retention processes considered in the monthly calculations are the coupled nitrification-denitrification. For DIN other transformation processes like plant uptake and subsequent mineralisation are considered, too. The involved aquatic organisms can vary from phytobenthos, to macrophytes in smaller rivers and ditches, and phytoplankton in larger rivers and lakes. The transformation process between DIN (dissolved inorganic nitrogen) and PON (particulate organic nitrogen) are reversible, and can have a major influence on the fractioning of N in surface waters. On an annual basis, however, the resulting net effect is assumed to be negligible (SAUNDERS and KALFF, 2001b; SVENDSEN and KRONVANG, 1993; VENOHR, 2006).

The transformation of DIN to PON by uptake and the later mineralisation back to DIN does not effect the TN concentrations. For calculating monthly TN retention, the coupled nitrification-denitrification is considered to be the only relevant process. For calculating monthly DIN retention, uptake by aquatic organisms must also be considered. N uptake by aquatic organisms increases with increasing water temperature, decreasing flow velocity, and with available sunlight (LAMPERT and SOMMER, 1993; MULHOLLAND *et al.*, 2001). The available sunlight for a water body can vary widely, and is dependent on the local conditions, (for example shadows from clouds, trees, and houses, and self-shading by macrophytes within the water body, can retard much of the available sunlight). In general, the available sunlight is strongly correlated with the incoming short wave radiation at surface level. Hence, in MONERIS, calculation of the monthly DIN retention uses the incoming short wave radiation at surface level as the parameter representing monthly differences in available sunlight; differences in shading are neglected. Data on incoming short wave radiation for Europe, is available as a 15 km grid, with monthly values for May 2007 to April 2010, from the European-wide CM-SAF map provided by the German Weather Service (Deutscher Wetterdienst, DWD) ([www.cmsaf.eu](http://www.cmsaf.eu)).

For the calculation of monthly DIN-retention, we used the annual THL-approach for the calculation of TN-retention (Eq. 50 TN retention) to describe the dominant nitrification-denitrification, and the minor share of retention accounted for by sedimentation. In addition to the existing THL approach, a new term has been introduced to describe the N uptake by aquatic organisms. Water temperature, hydraulic load, and global radiation are used as parameters in the new Temperature-Hydraulic-Load-Radiation (THLR)-approach (Eq. 57).

$$\text{RM}_{\text{THLR-DIN}} = \frac{1}{1 + (5.7 + 0.025 \cdot R) \cdot e^{0.067 \cdot T} \cdot \text{HL}^{-1}} \cdot 100 \quad (\text{Eq. 57})$$

$\text{RM}_{\text{THLR-DIN}}$  = monthly DIN retention calculated with the THLR approach, in %

R = incoming short wave radiation at surface level, in  $\text{W}/\text{m}^2$

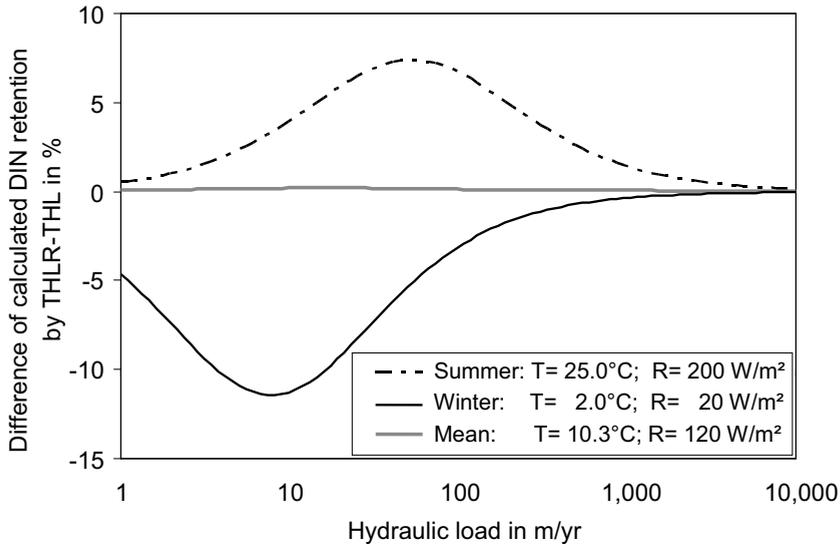


Figure 14. Comparison of DIN retention calculated with the THLR (Temperature Hydraulic Load Radiation) and THL (Temperature Hydraulic Load) at different hydraulic loads: data presented for summer (dashed line), winter (solid black line), and annual mean conditions.

From the CM-SAF map, the mean monthly radiation over the period from May 2007 to April 2010 was calculated for Germany. By calculating an area weighted mean, a mean monthly radiation of  $120 \text{ W/m}^2$  was derived.

Calibration of the THLR-approach assumed that the mean annual DIN retention of the THL-approach equals the monthly DIN retention of the THLR-approach at mean annual conditions. This is the case when using  $120 \text{ W/m}^2$  as the mean monthly Radiation,  $10.3 \text{ }^\circ\text{C}$  as the mean water temperature, and factors of 5.7 and 0.025 for the THLR-approach.

Compared to the THL-approach, the THLR-approach delivers a higher DIN retention in summer, especially for hydraulic loads between 10 and 200 m/yr. In summer, for low hydraulic loads ( $< 10 \text{ m/yr}$ , which are favourable conditions for N uptake by aquatic organisms); the two approaches deliver about the same retention, as radiation apparently does not limit the N uptake by aquatic organisms. At medium hydraulic loads (10 m/yr to 200 m/yr), the increased radiation of summer has the strongest effect on the modelled retention, and the THLR-approach delivers a 7.5% higher retention than the THL approach. At high hydraulic loads ( $> 200 \text{ m/yr}$ ), flow conditions are increasingly unfavourable for N uptake by aquatic organisms, and the differences between the retention calculated by the two approaches decrease. During winter, the situation is reversed, and the THLR-approach delivers a lower retention than does the THL approach, as low radiation limits N uptake by aquatic organisms at low hydraulic loads (Fig. 14).

For P, sedimentation is the dominant retention process. Only total P retention is calculated, and thus the changing shares of the P fractions that are accounted for by other processes, such as uptake by aquatic organisms, are omitted. The same equation (Eqs. 51 and 52) and parameters are used for calculation of both the monthly and the annual P retention.

## 4. Implementation of Measures in MONERIS

MONERIS uses integrated “management alternative” settings in order to calculate the potential of regionally differentiated measures to reduce nutrient emissions and resulting loads. Hereby it is possible to select or combine one or more measures, which can be implemented to individual or groups of analytical units. The measures implemented in MONERIS are considered by their net effect on the nutrient emissions, rather than the change in the respective transformation or transport processes.

In order to distinguish between the effect of the measure and the hydrological conditions of a specific year, the measures are only considered in the model for mean long-term dry and wet hydrological conditions. For the three hydrological conditions annual precipitation and runoff data are differentiated, all remaining input data, however, refer to the most recent available calculation year.

For the calculation of the N emissions, the development of the N surplus on agricultural land during the groundwater residence time has to be considered. Thus, the development of future N surpluses can be considered by the model.

The modelled potential of measures to reduce emissions can serve as basis for the development of management alternatives with a minimum resolution of an analytical unit. MONERIS, however, does not consider the technical, temporal or spatial feasibility of these measures, hence boundary conditions for the implementation of measures need to be reviewed with the appropriate national and state authorities or catchment area commissions and organisations.

The implemented measures are grouped into five categories: changes in land use (a), land use intensity (b), sewer systems (c), DCTP (d), and WWTP P or N (e). All given figures on the reduction potential or efficiency of certain measures can be changed by model users and can only be understood as a value to begin analysis for a certain study catchment.

### 4.1. Changes in Land Use

Measures from this category change the distribution or the management of a certain land use, such as tile drained areas or sloping areas prone to erosion.

#### 4.1.1. Conversion of Sealed to Unsealed Urban Areas (in %)

This measure simulates a reduction of the sealed urban areas. By this measure the discharge in sewer systems generated from precipitation is reduced and also the occurrence of combined sewer systems overflow events is reduced. Consequently, the groundwater recharge will increase by the same amount as discharge in sewer systems is decreased. The considered nutrient retention in the soil and groundwater leads to an additional reduction of the emissions. The discharge from WWTP is not affected by this measure.

#### 4.1.2. Conversion of Arable Land to Grassland

This measure assumes the conversion from arable land to grassland. A conversion of grassland to arable land (negative area portions) is not implemented in the model. The share of arable land to be converted to grassland can be defined separately for the slope classes <1%, 1–2%, 2–4%, 4–8%, >8%. By converting arable land to grassland nutrient emissions via erosion and overland flow will be reduced. For the converted arable land the stand-

ard approaches and model parameters used for grassland are applied. This measure should preferably be applied for areas with steep slopes.

If the share of tile drained areas is different for arable land and grassland this measure will also change the emissions via tile drainages. This does not affect the N surplus or P accumulation on agricultural land.

#### 4.1.3. Reduction of Tile Drained Areas on Agricultural Used Areas

The reduction of tile drained areas can be considered separately for arable land and grassland, but it cannot be defined for specific soils or slope classes. Tile drainages act like a short cut in the water circle and lead to an increased (artificial) interflow, delivering high nutrient loads. By this measure the leakage to the soils and groundwater increases, accompanied by an increased retention, compared to interflow.

The effects of rewetting of intensively used arable land, such as increased risk of P redissolution (GELBRECHT *et al.*, 1996), is not considered within the model.

#### 4.1.4. Soil Loss Reduction on Arable Land

By this measure the application of existing practises to reduce soil loss, such as conservation tillage and contour ploughing, can be simulated. Practices to reduce soil loss differ in their effectiveness, in dependence on the site characteristics like soil type, slope or precipitation (SCHMIDT *et al.*, 2002). Therefore, the effect of the soil loss reduction can be defined separately for the slope classes < 1%, 1–2%, 2–4%, 4–8%, > 8% and for each analytical unit. The reduction efficiency of such measures, directly reducing the calculated soil loss, can separately be defined by the model user for any slope class and analytical unit.

#### 4.1.5. Construction of Buffer Stripes

By applying this measure, the model simulates the construction of buffer stripes in order to reduce the share of eroded material entering surface waters. The user can define the share of agricultural land in a respective analytical unit, for which the emissions via erosion should be reduced. Consequently, the sediment delivery ratio (SDR, see Eq. 12) is reduced by this share. In dependence of the assumed width of the buffer stripe erosion will be reduced by 10 to 100% (RADERSCHALL *et al.*, 1996).

This measure does not consider any land use changes, for example to substitute the area need to construct buffer stripes. It is also not possible to select specific areas within a respective analytical unit, for which an increased soil loss was found. For this measure the percentage share of tributaries for which buffer stripes are assumed and the respective width of the buffer stripe can be defined.

When combining measures to reduce soil loss from arable land and the construction of buffer stripes, both measures are considered in addition to each other, and cannot be applied beside of each other for selected parts of arable land separately.

#### 4.1.6. Retention Ponds for Tile Drained Areas

This measure assumes that discharges from tile drainages enter a retention pond before being released to surface waters. N and P retention in retention ponds are calculated considering the tile drainage discharges, the retention pond size and the hydraulic load of retention

ponds. The sufficient N retention capacity can be attained by adjusting the area of retention ponds in relation to the tile drained areas. Research results suggest that a retention pond size of 150 m<sup>2</sup> per ha tile drained arable land provides a sufficient retention capacity.

Although this measure has been implemented in the model to calculate the effect of retention ponds, also other practices, such as in-field groundwater denitrification reactors, bioreactors or reactive swales and grassed waterways can be considered with this measure. This measure does not consider any land use changes, for example to substitute the area need to construct retention ponds.

#### 4.1.7. Reconstruction of Wetlands in Main Rivers

The reconstruction of wetlands, for example by dike back-shifting, is only possible for very few selected locations. The retention on wetlands, when being flooded, or in wetlands, when being flushed by groundwater, is very complex and difficult to model. In MONERIS only the first situation of flooding wetlands is implemented and considered by an increase of water surface areas. This leads to an increased retention due to reduced hydraulic load. The dyke back shift area as well as the gained water surface areas can be defined for each analytical unit separately. In the model the grassland of the respective analytical unit will be reduced by the amount of gained water surface areas. If the grassland area is exceeded, arable land will be reduced respectively.

#### 4.1.8. Stream Flow Restoration in Tributaries

By the stream flow restoration the flow length of tributaries will be increased, which in turn leads to an increased water surface area and an increased in-stream nutrient retention. The increase of the flow length can be defined for this measure.

### 4.2. *Changes in Land Use Intensities*

This category contains measures to reduce N surplus, N deposition and the ban of phosphate in detergents.

#### 4.2.1. Reduction of the Nitrogen Surplus and Maximum Use of Fertilizer and Manure

By the reduction of N surplus on agricultural land the N concentrations in tile drainages and in the groundwater will be reduced. While the N concentration in tile drainages could be reduced immediately, the groundwater concentration will be reduced delayed, in dependence of the groundwater residence times.

Three options can be chosen for this measure: 1. the reduction of N surplus in percent, 2. the reduction of N surplus in kg (ha · yr) and 3. defining a maximum N surplus allowed on the agricultural land. The first two options reflect the effect of good farming practices, which will reduce N surplus by a certain amount. The 3<sup>rd</sup> option can be selected to model the effect of the fulfilment of legal frameworks like the EU-Nitrate Directive.

The model is not able to consider negative N surplus. If the measure settings for option two would result in negative N surplus, the model replaces these N surpluses by a value of zero. If the N surplus in a specific analytical unit is lower than the maximum N surplus assumed for option three, the original value will be used by the model.

#### 4.2.2. Reduction of Atmospheric $\text{NH}_y$ and $\text{NO}_x$ Deposition

By this measure, the atmospheric deposition of  $\text{NH}_y$  and  $\text{NO}_x$  can be reduced separately, reflecting the different main sources for  $\text{NH}_y$  (agriculture) and  $\text{NO}_x$  (traffic, households, burning). The reduction of atmospheric deposition affects directly the N surplus on agricultural land, N concentrations in surface runoff, emissions from sealed urban areas and the deposition on water surface areas. Major restriction of this measure is that the causers of atmospheric emissions are often located outside of the studied river catchment.

#### 4.2.3. Use of Phosphate-Free Detergents

While phosphate in laundry detergents in most European countries is already banned, phosphates in automatic dishwasher detergents are increasingly used, with increasing numbers of sold dishwashers. The EC plans to prohibit all use of phosphates in detergents by 2013, European countries outside the EU at least plan the ban of phosphates in laundry detergents. By this measure a full ban of phosphates in the respective detergents is assumed and is reflected in the reduction of inhabitant specific P emissions (BEHRENDT, 1994). The amount of reduction depends on the level of usage, which is considered on countrywide basis in the model. Moreover P concentration from waste water treatment plants with less than 10,000 population equivalents is reduced by the share of phosphates from detergents.

### 4.3. Measures to Reduce Discharge Volumes from Sewer Systems

This category offers measures to reduce untreated discharge from urban systems via sewer systems by increasing the share of areas connected to sewer systems, the increase of storage volumes in combined sewer systems and additional constructions to treat loads from rainwater collecting separate sewer systems.

#### 4.3.1. Increased Storage of Combined Sewer Systems

This measure assumes the increase of storage volume in combined sewer systems to decrease the risk of overflow events resulting from storm water events. 100% of storage equals a water storage volume of 23.3 m<sup>3</sup>/ha (of sealed urban areas) in combined sewer systems (see Chapter 3.7.). For this measure the storage volume should be given in percent in relation to a storage volume of 23.3 m<sup>3</sup>/ha. If the original storage volume in a specific analytical unit is higher than the storage volume assumed for the measure, the original value will be used by the model.

#### 4.3.2. Filtration of Water from Separate Sewer Systems

This measure defines the share of loads in rainwater collecting separate sewer systems, which is treated in soil retention filters or clarifier basins before discharged to surface waters. For soil retention filters, a retention capacity of 80% for N and 45% for P is used as a standard value. For clarifier basins N and P retention is set to 35% of the incoming load. If the original share of treated loads in a specific analytical unit is higher than assumed for the measure, the original value will be used by the model.

### 4.3.3. People Connected to Sewer Systems are also Connected to WWTP

This measure expects that all households and areas (inhabitants) connected to sewer systems are also connected to a WWTP. This measure does not consider households and areas that are unconnected, connected to DCTP or septic tanks. The measure defines the number of inhabitants in households only connected to sewer systems as zero. By this measure households and areas only connected to sewer systems will be connected to a virtual WWTP (sewer category 4).

### 4.3.4. Portion of People Connected to Sewer Systems and WWTP

With this measure the minimum share of households and areas (inhabitants) connected to sewer systems and WWTP on the total inhabitants is defined. If the original share of connected households and areas (inhabitants) in a specific analytical unit is higher than assumed for the measure, the original value will be used by the model. If the share of connected households and areas (inhabitants) is increasing, the share of households and areas (inhabitants) connected to decentralized treatment plants DCTP or to septic tanks will be reduced respectively. Measure 4.3.3 is calculated first and will be considered in the calculated share of connected inhabitants.

## 4.4. Measures for Decentralized Treatment Plants (DCTP)

Measures from this category reduce the nutrient emissions via decentralized treatment plants. DCTP are small-scale WWTPs and differ according to the technical treatment status between older systems without waste water aeration (DIN 4261 Part 1) and the recent standards with waste water aeration (DIN 4261 Part 2) (see urban systems).

### 4.4.1. Technical Status of the DCTP

This measure simulates that all DCTPs are constructed according to the standards of DIN 4261 Part 2, for which retention rates of 15% for N and 13% for P are assumed. This measure does not change the type how loads are discharged to the surface water (sewer system, pipes/ditches, or soil/groundwater).

### 4.4.2. DIN2 with Additional Phosphorus-Removal for DCTP

With this measure it is simulated that for all DCTPs constructed according to DIN 4261 Part 2 an additional P-removal is implemented. Thus P will be further reduced by 80% in the biological and mechanical pre-treated waste water of the DCTP.

### 4.4.3. DCTP Transformed to Virtual WWTP

With this measure it is simulated that all DCTPs in a respective analytical unit will be converted into new, to be built WWTPs (virtual WWTP). In MONERIS the measure can be applied separately for DCTPs either discharging via sewer systems or via pipes/ditches or soil/groundwater. The strongest improvement will result for DCTP discharging via sewer systems and pipes/ditches, whereas for DCTP discharging via soil/groundwater the effect

Table 6. Maximum concentrations of P and N for 5 WWTP size classes according to the EC Urban Waste Water Treatment Directive (EUROPEAN PARLIAMENT AND COUNCIL OF THE EUROPEAN UNION (Ed), 1991).

WWTP size class	Inhabitant equivalents	Phosphorus mg/l	Nitrogen mg/l
2	2,000–5,000	≤ 6.0	≤ 60.0
3	5,000–10,000	≤ 6.0	≤ 60.0
4	10,000–50,000	≤ 2.0	≤ 15.0
5	50,000–00,000	≤ 2.0	≤ 15.0
6	> 100,000	≤ 1.0	≤ 10.0

might be negative, as retention in soils and groundwater can be higher than in the virtual WWTP.

#### 4.4.4. Virtual WWTP with Additional Phosphorus-Removal

Additionally to the previous measure for the virtual WWTP a P removal is assumed with this measure. Like this, the minimum nutrient reduction of the virtual WWTP is 80% for P and 40% for N.

#### 4.5. Measures for the Reduction of Phosphorus and Nitrogen Concentrations from WWTPs

Measures from this category define the N and P effluent concentrations for WWTPs larger than 2,000 population equivalents. Generally, in EU-countries N and P concentrations should correspond to the target values of the EC Urban Waste Water Treatment Directive (EUROPEAN PARLIAMENT AND COUNCIL OF THE EUROPEAN UNION, 1991, Table 6). Maximum outlet concentrations can be defined separately for different size classes. If the original outlet concentration of an individual WWTP is lower than the assumed maximum concentration, the original value will be used by the model.

If no individual WWTP inventory is available, a reduction of loads from non-classified WWTP in percent can be defined. The N and P loads from WWTPs are calculated as the product of discharge and concentration (see point sources). The discharge of a WWTP will not be changed by this measure.

## 5. Validation

The validation of models like MONERIS is difficult as areas covering information on emissions and retention (in the catchment and in surface waters) are usually not available. Monitoring data, (e.g., concentrations, runoff, matter fluxes) in surface waters or for a specific pathway always reflect the situation at a specific location and time. An up-scaling of these values to an analytical unit level and to a month or a year is difficult and is not always representative for the whole period and area.

For developing of MONERIS the approaches for the different pathways have been derived and calibrated separately, and if data were available, validated against these. A determination of an absolute error of emissions calculated for a specific analytical unit is therefore in most cases not possible.

A model comparison conducted in the EU-project EUROHARP (see introduction) showed that MONERIS compared to the other models delivered balanced results, often close to the mean values of all models (KRONVANG *et al.*, 2009). Nevertheless, none of the models delivered more consistent or in other terms better results than the other models.

Consequently, the validation of MONERIS results is often restricted to a comparison of modelled and observed loads. Observed loads are calculated on base of measured runoff and nutrient concentration at the gauging stations (BEHRENDT, 2000). In Figure 15 mean calculated loads of the single years 2003–2005 were compared to the observed loads for German river basins (FUCHS *et al.*, 2010). Due to a uniform deviation at the 1:1-line, no systematic error is detectable. A comparison of modelled and observed loads from MONERIS applications all over the world delivered mean deviations between 15% and 35%. This equals the error to be expected in the observed loads (ZWEYNERT, 2009).

The load comparison in general shows higher deviations for small (< 50 km<sup>2</sup>) analytical units. Two reasons can be found to explain this: (i) data or (ii) model insufficiencies. In smaller analytical units individual site conditions (*e.g.*, soil, slope, and location of tile drained areas) and events (*e.g.*, heavy rainfall) stronger influences the total emissions than in larger catchments. Area covering input data, *e.g.*, precipitation or atmospheric deposition, are often results from other models transferring point information to the area. In case data from administrative units (statistical reports) are used the spatial resolution of these units is coarser than for the analytical units. In this case the specific situation in different analytical units might not be explained by the input data. On the other hand, if detailed information is available, the approaches used in MONERIS, developed to describe mean situations in river systems, are not able to consider these data and to model the specific situation.

Spatial resolution of MONERIS is limited to ca. 10 km<sup>2</sup>, but the validated minimum size for analytical units is 50 km<sup>2</sup> (BEHRENDT *et al.* 2003b; ZWEYNERT, 2009). On a temporal scale a monthly resolution can be achieved. The quality of results is profoundly dependent on the resolution and quality of input data. Not for all countries the data quality is as high as in the EU countries. Thus, MONERIS can display only a part of its potential, if only a data set of reduced data quality is available. MONERIS was tested successfully in different climatic regions. Nevertheless, geographical limitations are given for areas under arid condition, since MONERIS is not able to calculate emissions with a negative groundwater discharge.

## 6. Conclusions

The motivation for this first complete description of the nutrient emission model MONERIS, and its methods and algorithms, was the need to provide an up-to-date comprehensive and scientific-based overview for modellers and model users. The model MONERIS is based on data for river flow and water quality, as well as a set of geo-referenced data including digital maps and extensive statistics. Whereas diffuse emissions into surface waters are calculated as the sum from different pathways and their individual flow components, the point emissions from wastewater treatment plants and industrial sources are taken from an inventory. The particular value of the MONERIS model concept is the strictly separated calibration and validation of the approaches for the different emissions pathways, which is achieved by using data sets independent from other results of MONERIS, thus avoiding the effect of inter-calibration.

Due to its flexibility, the MONERIS model was extended, in collaboration with the Karlsruhe Institute of Technology (FUCHS *et al.*, 2010), to allow calculation of heavy metals and polycyclic aromatic hydrocarbons. An additional improvement has been the enhancement of temporal and spatial resolution. On a temporal scale, a monthly resolution is now possible, and the spatial resolution has been enhanced to a validated minimum size of 50 km<sup>2</sup>

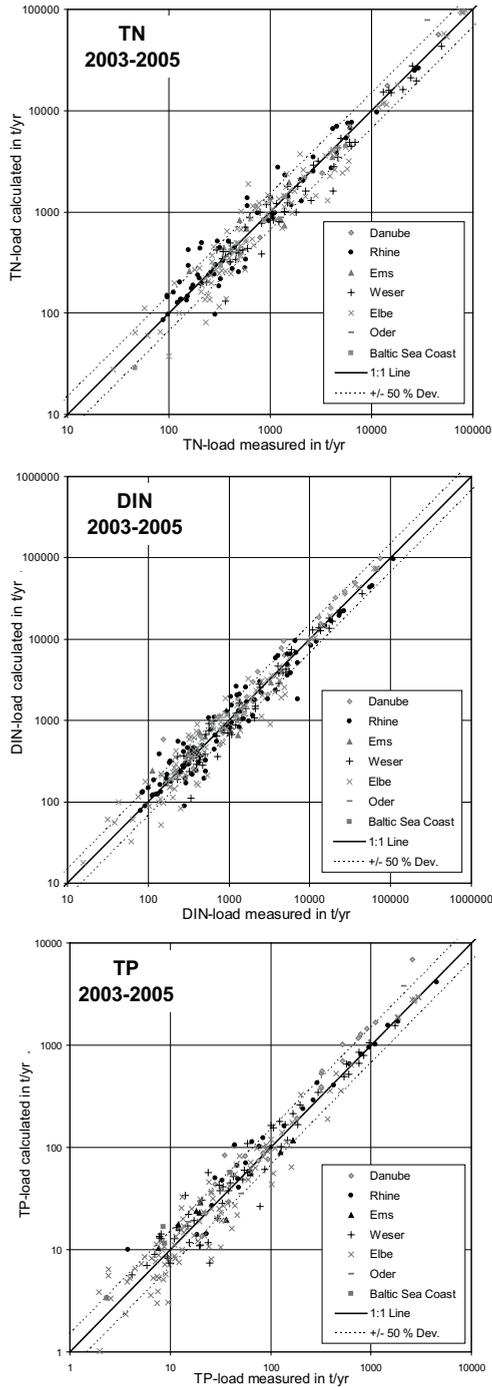


Figure 15. Comparison of calculated and observed annual loads for total nitrogen (TN), dissolved inorganic nitrogen (DIN), and total phosphorus (TP), for 6 rivers and two coastal areas for 2003–2005.

(BEHRENDT *et al.* 2003b; ZWEYNERT, 2009). Initially, MONERIS was created to calculate nutrient emissions in meso- to macroscale river basins, on a temporal resolution between 5-year mean values and yearly values.

The quality of model results is strongly dependent on the resolution and quality of the input data. In EU countries, the quality of the data input is high. However, this is not always the case for data from some non-EU countries. Even if there is limited data available for some catchments, MONERIS can still be applied, contrasting with the very detailed, process-based models which could not be used in such circumstances. The application of MONERIS was successful in a wide range of catchments, including those with cool-humid or hot-dry climates, intensively to extensively used landscapes, and highly populated to scarcely populated.

With the “management alternative” tool, MONERIS can calculate the effect of measures. However, there are restrictions for deriving detailed management plans, because MONERIS works on the level of an analytical unit, and does not consider whether site conditions are suitable for a particular management alternative. Nevertheless, information from management plans (such as those proposed by local studies or by Federal States) can be aggregated, and taken into account for scenario calculations.

Further developments for MONERIS may include calculating with negative water balances (such as occur under dry climate conditions), and with negative nitrogen surpluses (as could be used for historical modelling, or extensively used areas), and bridging the gap between abiotic drivers (flow velocity and light availability) and the response of aquatic biota. Beyond that, currently new approaches are under development, *e.g.*, to consider the effect of wetlands on the nutrient balance, to improve the SDR-approach for erosion, and to extend the catalogue of management options.

After 15 years of development, and numerous international applications, the MONERIS model is confirmed as an appropriate tool for nutrient emission modelling and river basin management on large scales, and under a wide range of different catchment characteristics.

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