Chapter 5

Catchment Modeling

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This chapter addresses different monitoring and modeling approaches to investigate and quantify sources and pathways of sediments, nutrients and pollutants within river catchments.

Within four sections, the authors apply a variety of methods focusing on issues of different temporal and spatial scales ranging from event-driven sediment transport in a small mountainous catchment to annual load and mass balance in large river basins. The goal of this chapter is to improve the knowledge in assessing source-related load balances in river catchments.

Starting point of the first chapter is the European Water Framework Directive (WFD) which takes a combined approach of emission limit values and environmental quality standards to pollution control. WFD requirements are identified and present concepts to calculate substance specific load and mass balance in river catchments are discussed with respect to their capacity and limitation in assessing point source and non-point-source pollution.

Section 5.2 presents a field study investigating the effects of land cover changes on sediment transport in an alpine watershed by means of a coupled, spatially distributed event-based rainfall-runoff-erosion model. The choice of the sediment transport formula may significantly affect the calculated magnitude of erosion and deposition. Hence, a better understanding of the driving mechanism of erosion in relation to overland flow is required.

Section 5.3 evaluates the flood dependent transport processes of suspended sediment and pollutants in the Middle Elbe River. Event-driven matter transport is due to the complex interaction of: inflow from tributaries with different pollutant load; the role of groyne fields acting either as a source or as a sink for particulates; and the deposition on floodplain areas. Transport of the trace metals depends on their sediment-water partitioning and the grain-size of the suspended sediment.

Section 5.4 aims at mean annual loads of phosphorus from diffuse and point sources in large river basins. The empirical emission model presented is based upon so-called phosphotopes which are regarded as homogeneous types of sub-areas representing discontinuous source areas for non-point phosphate inputs. As shown for the Ems and Rhine catchments this phosphotope modeling concept may identify hot spot areas of pollution and hence, may support options and plans for a river basin management and remediation measures according to WFD. Further experience in monitoring and modeling of pollutant emission and transport in river catchments over a large range of spatial and temporal scales is still necessary for the implementation. The following four papers show the broad spectrum of work and give inspiration for future research in the field of catchment-related pollution control and management.

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5.1 Catchment Modeling of Emissions from the Perspective of WFD Implementation

5.1.1 Introduction

The Water Framework Directive (WFD) sets ambitious objectives for the protection of European water resources (EC 2000). For priority substances and other pollutants environmental quality standards have been defined for surface water. An understanding of the sources and pathways of these substances within river catchments is crucial to establish effective monitoring programs and to develop emission control strategies as a part of cost-effective programs of measures. In this context, source-related emission modeling offers the potential to support WFD implementation, since these catchment models provide annual load balances and highlight the relevance of various pollutant sources and pathways (Fig. 5.1).

The objective of this chapter is to discuss the value of catchment-related emission modeling from the viewpoint of WFD implementation. Since this contribution is related to the SEDYMO research program (Westrich and Förstner 2005), emphasis is given to fluvial systems. Trace metals are chosen as a substance group of environmental concern which is particle-associated to a large extent. Instream processes such as erosion, sedimentation or biogeochemical transformation are in the focus of other con-

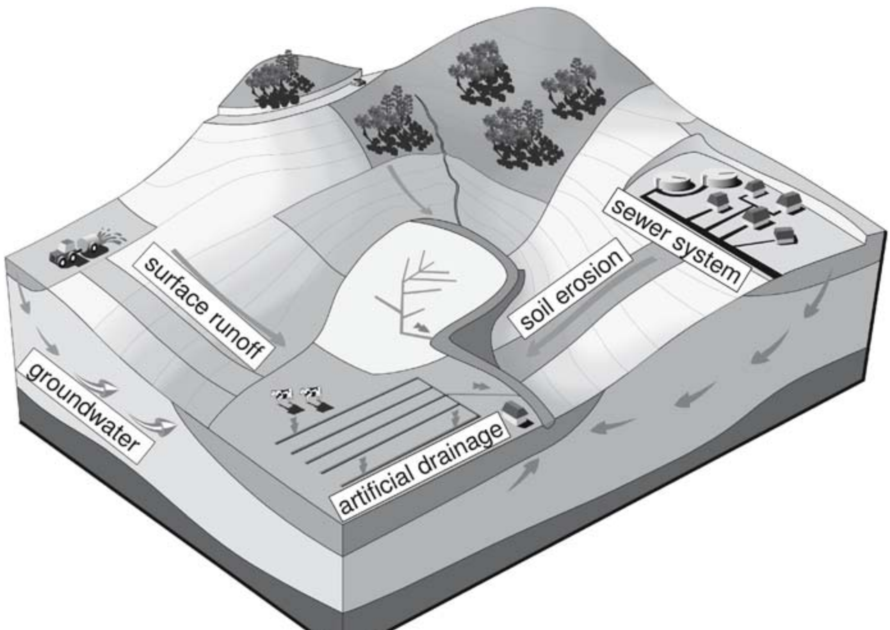


Fig. 5.1. Sources and pathways of pollutants in river basins

tributors and hence, are not considered here. The results presented are from the research project SAFE (Becker et al. 2005) and are related to the 1 800 km^2 catchment of the Erft River, a left tributary of the lower Rhine located in the western part of Germany.

In a first step, the WFD is briefly presented and its requirements for catchment modeling are described. Next, pathways of pollutants are analyzed with respect to data availability and their temporal process behavior. The capabilities of present modeling concepts which calculate substance load budgets in river catchments are investigated. The deficits, obstacles and perspectives of catchment models are considered.

5.1.2 The WFD and Its Requirements for Catchment Modeling

On 23 October 2000, the "Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy" (EC 2000), commonly known as the European Water Framework Directive (WFD), was adopted. The key elements and objectives of the Directive are as follows (Blöch 2004):

- protection of all waters aiming at good status, as a rule, at the latest by 2015, linked to a non-deterioration principle
- comprehensive monitoring systems for all waters
- coherent water management based on river basins
- combined approach of emission limit values and quality standards
- economic instruments to support environmental objectives
- mandatory public participation
- streamlining legislation, and ensuring a single, coherent managerial frame.

The Directive foresees a timetable for implementation using a stepwise approach consisting of: initial river basin analysis (finalized 2005), the implementation of monitoring programs by 2007, the establishment of programs of measures and river basin district management programs until 2009, the implementation of measures until 2012 and the achievement of good status of water resources until 2015.

Within the first river basin analysis, in Germany published in spring 2005, 60% of all surface water bodies were deemed to be at risk, 26% possibly at risk and 16% not at risk of failing the WFD objectives (BMU 2005). One major reason for waters being at risk was found to be deficits in hydromorphology, including river continuity. Another is related to discharges from point and diffuse sources affecting the water quality of surface waters, which is the focal point of this study.

The Combined Approach

The WFD takes a combined approach to pollution control first by setting emission controls to limit pollution at the source (e.g., waste water, agricultural fertilizers) and secondly, by establishing water quality objectives for bodies of water. In every case, the more stringent of the two will apply (De Toffol et al. 2005). Thus Member States will have to set down in their programs of measures both the limit values to control emissions from individual point sources and environmental quality standards (EQS) to limit the cumulative impact of such emissions as well as those from diffuse sources of pollution. For surface waters, EQS have been defined for priority substances and certain other pollutants considering the annual average (AA-EQS) and the maximum allowable concentration (MAC-EQS) (EC 2006).

A meaningful combined approach is strongly connected to a sound understanding of the origin, pathways and depots of pollutants within river catchments. Surface waters may receive loadings of pollutants from various point and diffuse sources. Knowledge about the pollutant origin and especially about the dominant input pathways is prerequisite to develop successful strategies for emission avoidance and emission control.

The relevance of external sources in comparison to the background values for the apparent instream pollution has to be investigated. In the past, especially particle-bound pollutants such as most of the heavy metals and hydrophobic organic micropollutants may have accumulated in aquatic sediments due to historical pollution. In this case, pollution of river bed has to be considered and may even far exceed the impact of external sources. These 'areas of concern' have to be identified within a river basin and assessed with respect to their ecological damage potential using a 'weight of evidence' approach (Heise et al. 2004; Heise et al. 2005).

The WFD Scales

It is noteworthy to recognize that the WFD's water quality targets are connected to different spatial scales. Here, three levels may be distinguished, which have to be embedded within a management framework (Fig. 5.2).

The protection of the marine environment is a major issue in Europe's water policy. This is manifested within international agreements such as HELCOM for the Baltic Sea (EC 1994a,b) and OSPAR for the North-East Altantic (EC 1997), which aim at a preservation of biodiversity of the marine ecosystem, protection from eutrophication, hazardous and radioactive substances and the effects of offshore exploration of oil and gas. From the marine viewpoint only the entire load of nutrients and pollutants from a river basin is important without considering its precise origin.

However, knowledge about the 'areas of concern' as well as the 'input pathways of concern' is required to set efficient management priorities on the catchment scale. Additionally, legislative and financial instruments rather than technical measures will be used at this scale to support the achievement of the environmental goals.

The other end of the WFD scale is defined by the single water bodies for which, as a rule, the good status has to be achieved by 2015. For these river sections, operational monitoring and cost-effective measures have to be developed. Hence, when considering water quality targets, a detailed understanding of the pollutant emission pathways is needed on the local scale.

On the regional scale, management units may be defined by groups of water bodies with similar conditions or impacts.

With respect to temporal discretization, substance load balances are needed at least on an annual basis. Firstly, this is to identify trends within the WFD reporting cycle, as the Directive demands river basin management plans to be established and updated every six years. Secondly, the annual basis allows to compare modeling results with the AA-EQS for priority substances and pollutants in surface waters.

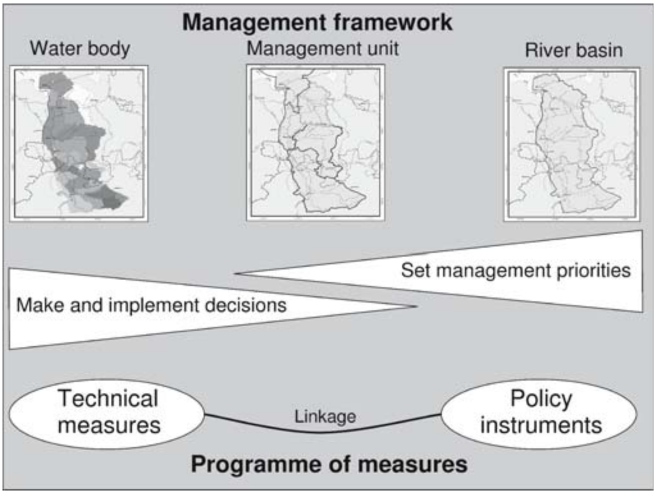


Fig. 5.2.
WFD management framework

The Contribution of Emission-Related Catchment Models

Modeling load balances of substance emissions within river basins supports for WDF implementation by:

1. Contributing to development of monitoring strategies
2. Supplementing available monitoring data
3. Facilitating data analysis of emission and instream pollution
4. Helping to verify complience with environmental quality standards
5. Identifying and assessing the relevance of pollutant pathways
6. Enabling development of instruments and measures for pollution control (e.g., by scenario analyses)
7. Providing a basis for implementation of the 'polluter pays principle'

5.1.3 Calculating Emission Balances in River Systems

Emission Pathways

To calculate the substance loadings which a water body receives is a challenging task since a network of numerous environmental pathways has to be considered (Fig. 5.3). Each pathway may be defined by three parts: mobilization from a specific source, a transport route, and the input to the water body. For example, the roofs of buildings may consist of zinc which is mobilized by rainfall and transported via roof runoff into a combined sewer system. Zinc may be discharged into a river in this example either via the outflow of the waste water treatment plant or, in case of overloading, via combined stormwater overflow. At this point, the question arises if it is advantageous or even necessary to describe the entire pathway (including mobilization, transport route and input) or if it would be sufficient or favorable to consider only the input. Generally,

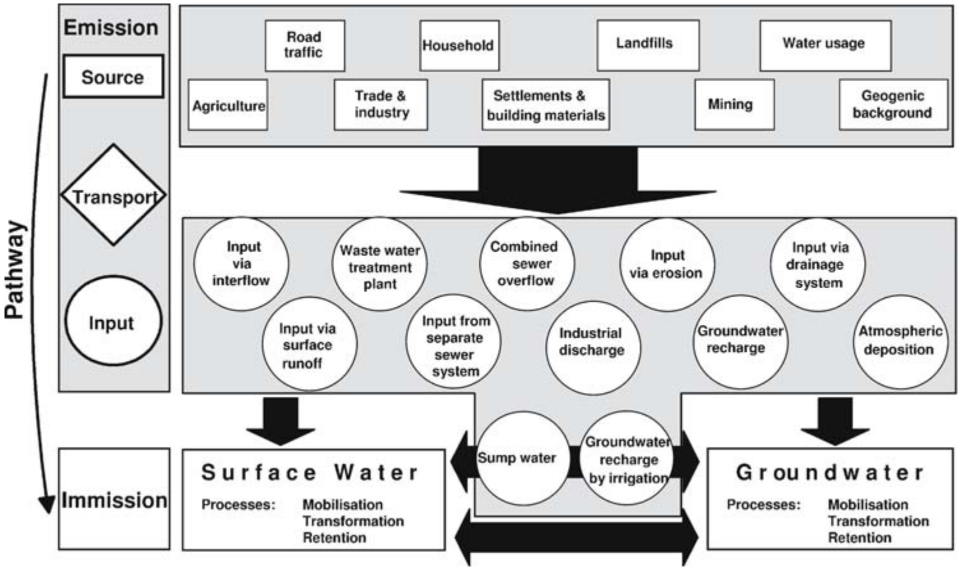


Fig. 5.3. Environmental pathways of pollutants in river basins

a clear answer cannot be given to this question since the choice of the model concept depends on both the objectives of the investigation and the availability of data. However, the recommendation can be given that a modeling approach which is limited to the input is sufficient if either the input is well known or if it is fed from heterogeneous sources which are not easily identifiable. A highly sophisticated process – based model describing the whole pathway seems to be less appropriate to be applied in large river basins than a conceptual model which is developed to set management priorities. On the other hand the same conceptual model developed for the large scale may be too coarse to support decisions on a water body scale.

Contaminant sources are usually classified into two general forms, point sources and non-point sources, each of which poses specific problems regarding monitoring, quantification and management. Point sources of pollution are those originating from a single, well defined location. As such, they are often readily identified and generally easily controlled and monitored. Examples of point sources of pollution relevant to water and sediment management in river basins include industrial discharge, effluent from waste water treatment plants (WWTPs) and combined sewer overflow (CSO). Non-point (diffuse) sources of contaminants are those originating from a wide area within the catchment. As their location of discharge into a water body is not well defined the identification, and in particular the control, of these sources presents a challenge to water pollution management. Examples of diffuse sources of pollution include atmospheric deposition and the input of contaminants via surface runoff, interflow, groundwater runoff and erosion.

It is important to realize that some of the input pathways are continuous processes whereas others are event-driven. In view of the fact that the WFD requires emission load balances on an annual basis, on the one hand, those pollutant inputs which are defined by continuous, almost uniform graphs of substance loadings are the easiest to quantify and to control. Runoff from WWTP, groundwater recharge or pumping of sump water in mining areas are examples of this kind of processes. On the other hand, the description and simulation of pollutant inputs defined by discontinuous pathways which are steered by low frequency, high impact (LFHI) events is most difficult. Input via soil erosion, surface runoff and CSO belong to this group. As their fate is strongly affected by erosion, particulate nutrients and pollutants are difficult to handle.

Model Validation

Validation of emission load balances is an issue of outstanding relevance. Since only few input pathways are monitored by measuring the discharged loadings, the overall load balances can only be checked by means of environmental monitoring data from the receiving water bodies. Here, two problems have to be envisaged: First, the water quality of a water body may be influenced or even controlled by historical sediment pollution rather than by external sources (see Sect. 5.1.2). Second, instream processes such as erosion, sedimentation and biogeochemical transformation processes may be important for particle-bound and/or reactive compounds. Today, most of the catchment models do not account for these internal riverine processes in detail but use constant or discharge-dependent retention factors instead. Therefore, presented comparisons assuming that the sum of emissions minus retention is equal to monitored loading may be inadequate due to the deficiencies in process-description for the water bodies themselves.

5.1.4 Obstacles and Strategies in Catchment Modeling

In this section different approaches to quantify emissions are discussed for selected pathways. As an example of a continuous point source, effluent of waste water treatment plants (WWTP) is considered. Soil erosion is highlighted as discontinuous pathway from non-point source which is controlled by LFHI events.

Emissions from Continuous Point Source: Input Via WWTP Effluent

In the following presentation, different possibilities of estimating annual effluent loads from WWTP are investigated and discussed for the substance group of trace metals.

Figure 5.4 provides a schematic view of a conventional WWTP receiving its inflow from the sewage system. Purification is accomplished through fixation of a portion of the inflowing trace metals in the sewage sludge. The fluvial system is affected by the WWTP effluent. Hence, the effluent seems to be the ideal location to monitor and calculate the pollutant loading.

Calculation, or better estimation of the mean annual pollutant load is given by the product of the annual WWTP runoff and the mean annual concentration of the regarded pollutant in the WWTP effluent. In practice, a surveillance program is either continuous or discontinuous at certain monitoring intervals providing e.g., hourly or daily values. In case of lacking monitoring data annual WWTP discharge can be esti-

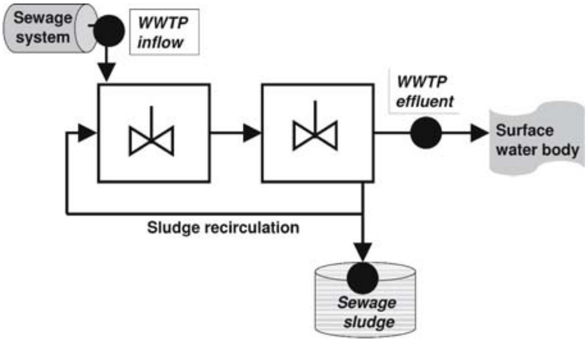
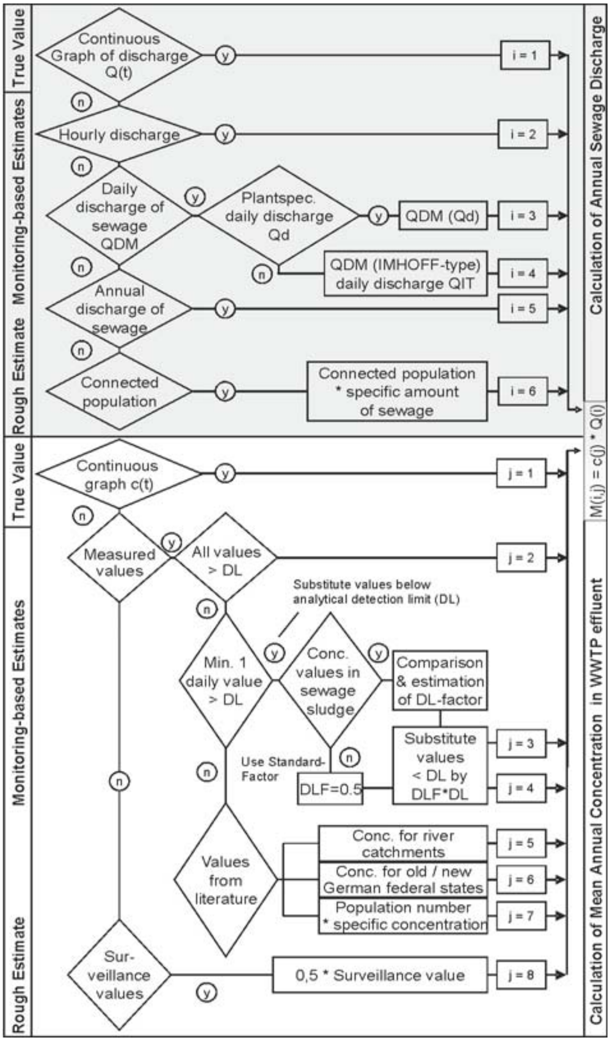


Fig. 5.4.
Points to quantify emissions from waste water treatment plants



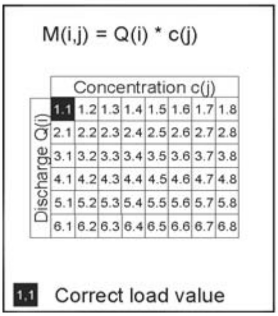


Fig. 5.5.
Stepwise approach to calculate annual WWTP effluent loads

mated by the design capacity of the plant or connected inhabitants (Zessner and Lindtner 2005), and annual concentration is usually derived either from literature values or from the surveillance value of the plant's operating permit, which can only provide a rough estimate of the true concentration. For calculation of both discharge and concentration, a decision tree can be set up which goes from greatest to poorest data availability (Fig. 5.5).

When combining the different estimates for the annual means of discharge and concentration, an $(n \times m)$-matrix can be set up where every matrix element refers to a specific load estimate. The upper left matrix element (1,1) refers to the correct load value, as it is determined by continuous monitoring, whereas the lower right one, e.g., matrix element (n, m), reflects the load estimate which is expected to be the least accurate, since it is not based on any monitoring data. Straightforward calculation of monitoring data generates a certain matrix element e.g., a specific annual load estimate. Depending on the given monitoring frequency this load estimate will be more or less accurate. From the matrix approach it becomes apparent that monitoring strategy has to be taken into account to ensure reliable emission data.

Another problem in practice is caused by concentration values below the analytical detection limit (DL). For trace metals, conventional analytical methods such as optic emission spectrometry (OES) might not be sensitive enough to detect trace metals in WWTP effluents. In this case a common procedure is to substitute values below DL to avoid data gaps in order to perform load calculations. The half detection limit value is most widely used to substitute analytical values below DL. For log-normally distributed analytical values, Fig. 5.6 depicts that this approach is only suitable if the minority of monitoring data is lower than DL. Otherwise, especially if only 10% or less of the

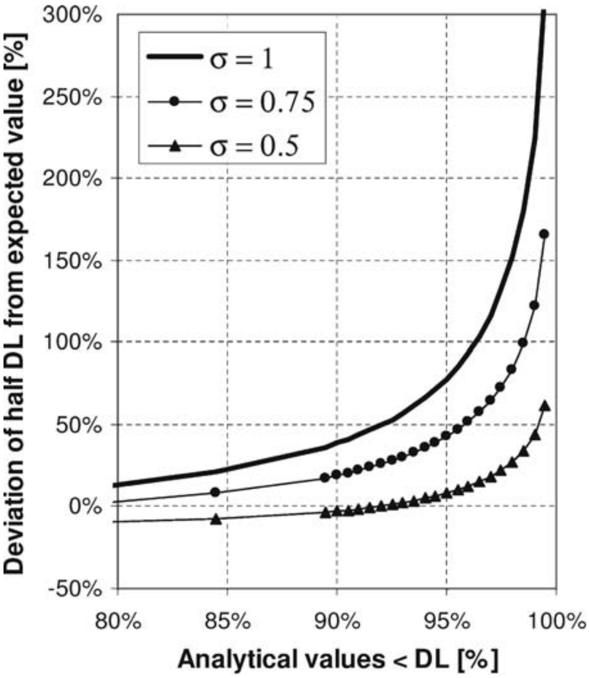


Fig. 5.6. The detection limit problem: deviation of the half detection limit value from the expected true value as a function of the percentage of values below detection limit. Results are given for different standard deviations σ assuming log-normally distributed analytical values

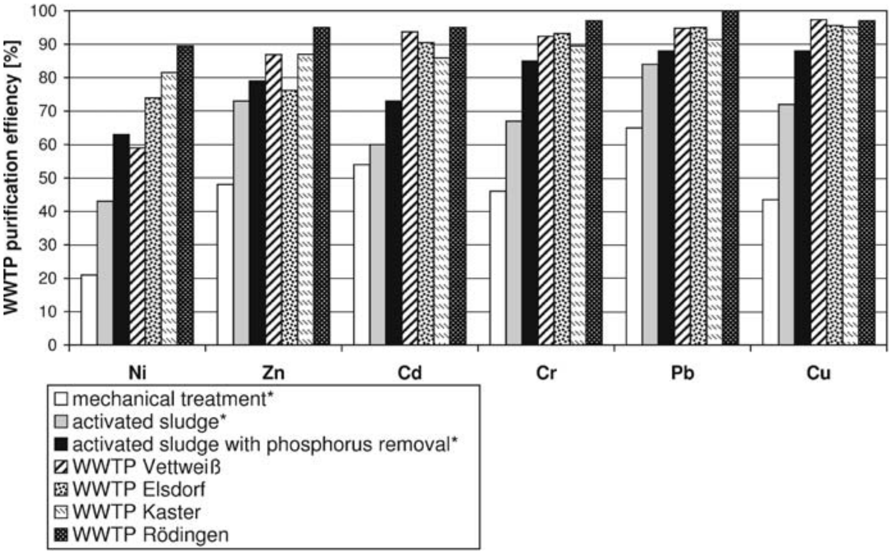


Fig. 5.7. The purification efficiency problem: trace metal retention efficiencies of WWTPs with different design characteristics. Mean literature values according to Fuchs et al. (2002) are indicated with * and compared with median values of three WWTPs (Vettweiss, Elsdorf and Kaster) using activated sludge with phosphorus removal and the membrane activated sludge WWTP at Rödingen

monitored data are detectable, the true concentration might be tremendously overestimated. In this case monitoring data are insufficient and do not form a basis for annual load estimates. For example, the WWTP surveillance program of the state of North Rhine-Westphalia, Germany, using conventional OES in combination with inductive coupled plasma (ICP-OES), revealed that 99% of concentration values for lead and cadmium were below DL (MUNLV 2006).

The solution to this difficulty is either to use more sensitive analytical methods such as mass spectrometry (ICP-MS) or to employ enrichment techniques prior to the trace metal detection. Since increasing analytical effort raises the costs of any surveillance program, it should be determined if there are further alternatives in calculating the pollutant loading of WWTP effluent.

DL problem may be avoided by considering the WWTP inflow or, even more promisingly, by using the sewage sludge where pollutants become enriched throughout the treatment process (Fig. 5.4). With either alternative, WWTP emissions can only be derived if the purification efficiency of the WWTP is well known. Thus, plant-specific data on pollutant retention efficiency have to be gathered. As a fall-back, literature values may be used instead if these are based upon the same design characteristic of the treatment plant. However, for WWTPs within the Erft River catchment, retention efficiencies for trace metals measured by Becker et al. (2005) appear to be considerably higher than reported by Fuchs et al. (2002) for WWTP in the Rhine catchment (Fig. 5.7).

Figure 5.8 depicts a comparison of mean annual loads of trace metals for 46 municipal WWTP plants in the Erft River catchment. The overall load of the examined trace metals derived from sewage sludge is at an average of approximately 5.0 t yr^{-1},

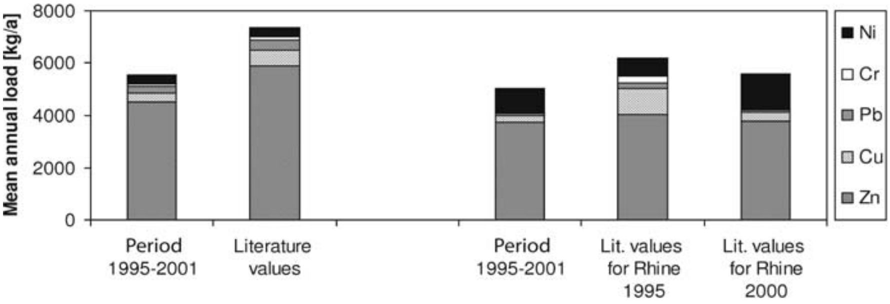


Fig. 5.8. Emissions from 46 WWTPs in the Erft River catchment. Comparison between the WWTP effluent method (*left*) and the sewage sludge approach (*right*) for trace metals

which is only slightly smaller than the load of 5.5 t yr^{-1} in the effluent. Based on this result, the sewage sludge method can be recommended as an alternative to the effluent method, despite significant deviations in value for individual trace metals such as nickel or lead. For both methods, literature values overestimate the values based on monitoring.

Emissions from Discontinuous Non-Point Source: Soil Erosion

As an example of a discontinuous non-point source, soil erosion is highlighted in this section. This pathway is significant for nutrients, especially phosphorus, and particle-bound pollutants and may form a large portion of the entire emissions as known for large river basins (Böhm et al. 2000).

Soil erosion depends on the characteristics of the catchment (hill slope, soil properties, vegetation) and those of the precipitation events (intensity, amount and course of rainfall). Erosion monitoring data is sparse since the events are unpredictable.

Due to the lack of appropriate measured data, either conceptual models or physically based models are employed to quantify soil erosion. The conceptual approach is generally based on the Universal Soil Loss Equation (USLE, Auerswald 1998) and provides mean annual values. Process oriented erosion models such as AGNPS (USDA 2006), EROSION 3D (Schmidt 1996; von Werner 2002), EUROSEM (Morgan et al. 2006) and WEPP (USDA 2006) simulate the physical processes of soil mobilization and soil transfer for discrete precipitation events. In the following both approaches are described and results are compared for the example of the upper Rotbach, a sub-basin of the Erft River, Germany.

Conceptual approach of erosion modeling. The USLE adapted to German conditions by Schwertmann et al. (1990), is employed to calculate the mean annual loss of soil (Eq. 5.1):

$$A = R \cdot K \cdot L \cdot S \cdot C \cdot P \tag{5.1}$$

A = Mean annual soil loss (t (ha a)$^{-1}$), R = rain erosivity factor (N h^{-1}), K = soil erodibility factor (t h (ha N)$^{-1}$), L = slope length factor (–), S = slope steepness factor (–), C = crop and management factor (–), P = protection measures factor (–).

Using Eq. 5.1 to estimate the potential erosion, the part of the mobile soil which is transported into the surface waters can be calculated if delivery and transfer of soil under the influence of heavy precipitation are taken into account. Various methods are available to determine the sediment delivery ratio, *(a)* accounting only for steep areas which are located close to rivers (Behrendt et al. 1999); *(b)* considering hill slope and the portion of farm land (see model MONERIS, Behrendt et al. 1999); or *(c)* classifying the hill slope by the probability with which the abutting, slopy area is connected to a rivulet.

When calculating the soil transfer by erosion it is always to be noticed that the portion of small grain-size fractions increases during the transport by surface discharge. This sorting is caused by preferential settling of larger soil particles and is considered by the enrichment ratio (ER, see Sect. 5.4.3) employing one of the following procedures: *(a)* constant substance-specific ER (Fuchs et al. 2002); *(b)* ER depending on the long term mean of the annual sediment input (Auerswald 1998); *(c)* ER accounting for the correlation between sorption surface and grain size of soil particles when simulating the graded transport of several grain size fractions; *(d)* ER given by the ratio of measured substance concentrations in river bottom sediment to the mean monitored soil contamination (Böhm et al. 2001).

The substance load by soil erosion, which is transported into the river during heavy precipitation, is calculated by Eq. 5.2 from the mean annual soil loss (Eq. 5.1), the sediment delivery ratio, the enrichment ratio and the substance concentration in the soil.

$$\mathrm{SE} = E\frac{\mathrm{SDR}}{100}\frac{\mathrm{ER}}{100}c_{\mathrm{soil}}\frac{A}{1\,000}R \tag{5.2}$$

SE = substance input via erosion (kg), E = potential erosion (t ha^{-1}), R = rainfall erosivity factor (%), SDR = sediment delivery ratio (%), ER = enrichment ratio (%), c_{soil} = substance concentration in the soil (mg kg^{-1}), A = area of farm land (ha).

Process oriented approach of erosion modeling. These models simulate the soil erosion in small catchments for single precipitation events. The model EROSION 3D (Schmidt 1996; von Werner 2002) which is employed here, is applicable in catchments of up to approximately 400 km^2. Spatial resolution of grid cells is between 5 and 20 m, temporal discretization is by time steps of 10 min. The model EROSION 3D consists of two main components, a GIS-module and the program core. The GIS-module is used for the digital relief analysis. The program core of the model takes into account, among others, the following processes of soil erosion: infiltration of precipitation according to Green and Ampt (Schmidt 1996); generation of runoff (excess of infiltration and retention in troughs); unsoldering of soil particles from the soil surface; transport of particles and deposition depending on the transport capacity of the surface runoff; enrichment of fine particles along the transport path.

Input parameters of deterministic erosion models are numerous and can be subdivided into three groups. Digital terrain model; soil parameters (distribution of grain sizes, storage density, content of organic carbon, initial water content, resistance against erosion, coefficient of roughness, degree of coverage, correction factor, associated precipitation situation); precipitation parameters (intensity, interval of discretization, duration of precipitation).

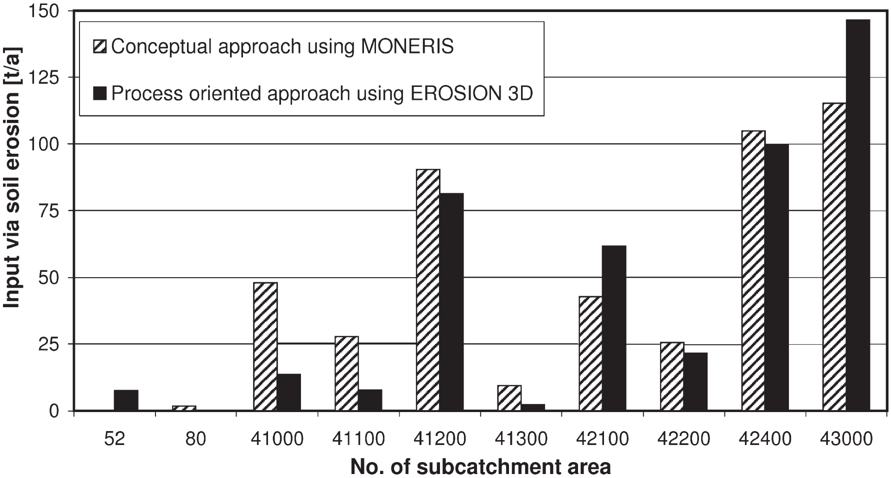


Fig. 5.9. Sediment input via soil erosion in different areas of the upper Rotbach catchment

Model application. For the year 2000, the annual sum of soil erosion and deposition in the upper Rotbach catchment, a river system in the slopy setting of the northern Eifel mountains. Figure 5.9 shows the results of the conceptual method following the approach of MONERIS (Behrendt et al. 1999) and those of the process oriented EROSION 3D model assuming that all the farmland is covered with winter wheat.

The comparison of the model results for the sub-catchments of the upper Rotbach shows remarkable resemblances. This applies to the spatial allocation of the erosion as well as for the absolute quantities.

Despite of these promising results, it must be stressed that matter input via soil erosion can only be assessed with large uncertainties. Most seriously, model validation by means of instream data is affected by sparse data and riverine sediment transport which may blot out the effect of soil erosion (see Sect. 5.1.3).

From the viewpoint of WFD both approaches are supplementary and thus, their combination is most promising. The conceptual approach may serve to assess the significance of the soil erosion pathway in comparison to other human pressures and impacts on the river basin scale. The process oriented approach is to investigate the event-driven effects of soil erosion and to evaluate the significance of measures, e.g., for soil conservation, on the local and regional scale.

5.1.5 Conclusions

From the perspective of WFD implementation, there is a substantial need for catchment modeling and in particular for substance load balances. These models contribute to the combined approach. They may support decisions to set up monitoring programs, to develop cost-effective instruments and measures for pollution control and to implement the 'polluter pays principle'. Application and further development of catchment models should address the following considerations:

1. Emission-related catchment models are designed to identify 'input pathways of concern' e.g., they aim at identifying the relevant external sources and impacts within a river catchment.
2. This type of models disregards instream contamination due to historical sediment pollution. Hot spots of sediment contamination which are referred to as 'areas of concern' have to be identified separately and require a risk assessment with respect to their environmental relevance (Heise et al. 2004; Heise et al. 2005).
3. Integrated water quality management should account for both, 'input pathways of concern' as well as 'areas of concern'. Hence, emission-related models cannot be used in isolation to arrive at management recommendations, especially when looking at particulate pollutants. These models should rather be part of the entire managerial framework.
4. WFD requires annual load balances on different spatial scales (river basin, management unit, water body). Although the existing models do not meet all the WFD requirements – especially on the regional scale – there is no need to recommend the development of a single, generally applicable catchment model. For one, the level of knowledge about transfers and processes in river basins varies significantly from one water body or management unit to another. For another, the availability of model input parameters is very different. Therefore, applying catchment models should not be reduced to the automatic use of a specific, straightforward algorithm. Instead, the regional applicability of a certain model should first be checked against the objective and the scale of application, the regional data availability and the level of knowledge about pollutant sources and pathways. A variety of emission-related models which offer data-dependent ways of calculating pollutant discharges are thus needed. Robustness checks and sensitivity analysis are strongly recommended to increase reliability of the calculated load balances.
5. At present, catchment models do not consider the fate of pollutants within rivers in an appropriate way. Especially for particle-bound and reactive compounds, either these models have to be further improved, or a coupling of emission-related catchment models with surface water quality models has to be envisaged to overcome the existing deficits. Hence, better collaboration between "catchment modelers" and "instream modelers" is recommended.

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5.2 Modeling the Effects of Land Cover Changes on Sediment Transport in the Vogelbach Basin, Switzerland

5.2.1 Introduction

Land use changes are often considered to be the reason for increased flood frequency and magnitude as well as enhanced erosion. In small mountain basins, rapid changes in vegetation land cover may occur due to anthropogenic or natural causes, with pronounced effects on hydrological processes and the local ecosystem. With deforestation or windstorm damage, an increased proportion of the land surface may be exposed to more intensive overland flow processes, providing an enhanced sediment supply to the fluvial system and increasing the risk for downstream areas during severe flood events. It is generally accepted that the adaptation of appropriate land use strategies will lead to a more effective prevention of runoff and sediment production. In addition, careful management of the vegetation cover may lead to the ecosystem being less exposed to windstorm damages and more effective in regenerating after disturbances. A sustainable forest management which minimizes the risks of floods and erosion and strengthens the ecosystem has to be based on a proper understanding and evaluation of basin response to changes in vegetation cover.

Unfortunately, direct observations of land use change effects are very limited, in particular in combination with adequate hydrological, sediment transport and erosion measurements. Therefore, the quantification of sustainable forest management practices often relies on mathematical modeling and scenario analysis. Scenarios play an important role because they use expert knowledge to identify realistic catchment changes, which are subsequently used to assess the relevant hydrologic response by means of model simulations. Thus, distributed rainfall-runoff models, coupled with sediment transport and erosion/deposition models and used together with land cover change scenarios, can help in a quantitative assessment and evaluation of erosion patterns resulting from catchment response to scenarios of different land uses.

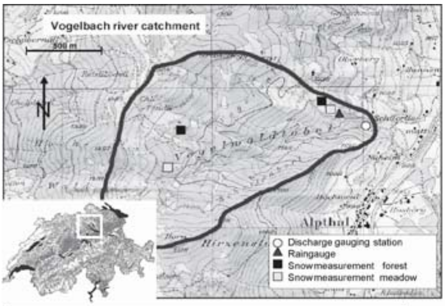
As part of a larger study (Kirsch and Burlando 2005) the present investigation focuses on the development and application of a coupled spatially distributed and GIS-integrated event-based rainfall-runoff-erosion model, and on its subsequent application to the analysis of catchment response to land cover change scenarios. Specifically this paper presents the investigation of the effects on overland and channel potential erosion and transport rates in a small mountain basin in Switzerland. The paper reports the results of a preliminary study which aims at showing the potential of modeling as a replacement of extensive, demanding and often unfeasible 1:1 investigations, whereas the study is designed as an investigation framework for ungauged basins, especially targeting the study of basin changes. It does not address and resolve open scientific issues concerning erosion and sediment transport issues which are still debated in the literature. Therefore, assumptions which have been proposed and extensively used in the literature are neither refuted nor proved, but studied from the prospective of modeling relevant changes within the basin over time. Conversely, while showing the power of mathematical modeling, the study expresses a word of caution on the (direct application) of equations available from the literature without reflecting, even in a qualitative fashion, on the physical processes.

The analyzed land cover change scenarios consist of hypothetical forest damages caused by windstorms similar to historically observed storms in Switzerland (BUWAL 2002). The results focus on the changes in hillslope and channel erosion rates, on the modification of the duration of the erosion event and of the total sediment load. The analyses were carried out by means of two different hillslope sediment transport approaches, to assess their sensitivity and their ability to predict erosion in space and time as influenced by land cover change.

5.2.2 The Case Study

The study catchment Vogelbach is located in the Alptal, central Switzerland (Fig. 5.10). Covering an area of 1.55 km^2, its elevation ranges between 1 060 m at the outlet and 1 545 m at the highest location. Average annual precipitation measured at the catchment weather station is around 2 100 mm yr^{-1} with highest precipitation in the summer. The average annual runoff from the catchment area is about 1 590 mm, resulting in an annual runoff coefficient of about 0.74. Based on station measurements, the rainfall DDF curves give for the return period 2.33 and 100 years the rainfall amount of 28 mm and 69 mm (1 hour duration), and 82 mm and 156 mm (24 hour duration). The mean runoff is about 0.078 $m^3\,s^{-1}$, while the highest recorded peak was about 6 $m^3\,s^{-1}$. The estimate of the centennial flood is approximately 11.5 $m^3\,s^{-1}$ (BWG, 2003).

Land cover consists mainly of forest (63%), pasture and meadow (31%). Hillslope steepness ranges between 20 and 40% with an average gradient of 37%. Most common soils in the basin are Gleysol and Regosol. The Swiss digital soil map (Geostat 1997) and specific literature (Burch 1994) indicate shallow soils (30–40 cm) with a large clay content causing a generally low infiltration capacity. Forested areas have a slightly higher infiltration capacitiy (compared to unforested areas) which leads to a lower degree of water logging. In general, geomorphologic properties and vegetation/land cover distribution within the Vogelbach catchment represent well the boundaries of hydrologically similar reacting areas. The erosion patterns and erosion intensities are also strongly related to hillslope morphology and vegetation cover; in par-



Runoff measurements	since 1967
Time resolution	10 min

Drainage Area	155 ha
Exposition	ESE
Altitude Range	1060 m – 1545 m
av. hillslope gradient	37 %
av. main channel gradient	18 %
Drainage density	ca. 8 km/km²
Forested area	63 %
Wetted area	25 %
Pasture area	12 %
Geology	Flysch

Fig. 5.10. Location, characteristics and instrumentation of the Vogelbach Basin

ticular steep hillslopes connected to the channels are susceptible to erosion, soil creep and shallow landslides. The main channel is incised into the hillslopes and has an average steepness of 18% and an average width of 5.5 m (Milzow et al. 2006). The longitudinal profile surveyed by Milzow et al. (2006) shows a step-pool morphology with occasional cascading sections and exposed bedrock. The median bed sediment particle size is around $d_{50} = 120$ mm, most bed material ranges between 10 and 1 000 mm in size. The catchment is instrumented with an automatic gauging station measuring discharge with a time resolution of 10 minutes. Precipitation is recorded with the same time resolution at a weather station located within the basin.

Sediment transport measurements are carried out by hydrophones, automatically recording the impulses of bed load particles larger than 10 mm at the basin outlet. Suspended load monitoring is not provided within the current equipment of the catchment. To estimate the total sediment load, measurements and relations of the neighboring catchment (Erlenbach) are used. There, calibrated hydrophones and a sediment retention basin allow the monitoring of total sediment and bed load sediment transport (Rickenmann 1997).

Spatial data used within this study are commonly available data within Switzerland. The spatially distributed land cover data for the base scenario ("present state") are derived from the raster map "Arealstatistik72" (100×100 m) of the Geostat Database. This map is the basis for developing hypothetical land use change scenarios for forest damage and management. In addition, these data are expanded by information and data of Burch (1994) and Walthert et al. (2003). Several floods between 1986 and 1990 were used to calibrate and validate the hydrological model (see the following section), and simulate the impact on erosion induced by land cover changes. The analysis focuses here particularly on a flood event of 1986 with a peak of 4.7 $m^3 s^{-1}$, corresponding approximately to a 8 year return period flood. The total event precipitation was around 46 mm with a duration of 3 hours and a maximum intensity of 7.5 mm in 10 min. The runoff coefficient for this event was around 0.45. Sediment yield was derived using observation data for the same event in the Erlenbach catchment and is estimated to be within the range 1 100–1 900 m^3 of sediment. This range was estimated using two different approaches, both based on the relation between erosion-effective runoff volume and sediment volume. Rickenmann (1997) defined a linear relationship between these two parameters. The similarity of the two catchments is used for a simple transfer of the sediment volume by the erosion-effective runoff volume. The second approach assumes a different regression between erosion-effective runoff volume and sediment for the two catchments and uses an area-related transfer by the area-normalized erosion rate. The above mentioned range is defined by the different sediment volumes resulting from the two approaches.

5.2.3 The Modeling Framework

The setup of the modeling framework aims at providing a spatially explicit suite of models on the basis of which impacts of land cover changes on the spatial and temporal distribution of runoff and erosion can be analyzed. An event-based, spatially distributed hydrological model, coupled to an event-based, spatially distributed erosion model is developed for this purpose.

The Hydrological Model (Fest98mod)

The hydrological model used to simulate the catchment response is based on an early version of the FEST98RS model extensively presented in Montaldo et al. (2004; the reader is referred to this work for further details about the hydrological model). The model provides a convenient framework because of its event-based approach and its spatially explicit distributed nature and land cover parameterization. The spatially distributed nature of the model requires the availability of a digital elevation model (DEM), and of raster based thematic maps of land cover and soil characteristics, as well as spatially distributed precipitation input. The fundamental model components are the surface runoff production and concentration modules. Subsurface flow processes are modeled by a single or multiple linear reservoirs arranged in sequence.

Surface runoff production is determined by a modified SCS-CN method (SCS 1972) and routed through hillslope and channel cells by the variable parameter Muskingum-Cunge method (vMCM, Cunge 1969; Ponce and Yevjevich 1978). Both methods are implemented on a raster basis.

The CN method uses soil and land cover information to the derive soil storage capacity. Kuntner (2002) modified the method to allow accounting for intermittent precipitation and optimized the CN values for Swiss catchments by extensive calibration and validation. Excess rainfall is routed by the vMCM, differentiating between overland and channel flow by the area-threshold method. A fraction-factor approach separates the infiltrated rainfall into deeper percolating water and subsurface flow, which is routed to the outlet by a linear reservoir scheme. This connects the storage of the subsurface horizons linearly to the output Q ($m^3 s^{-1}$) at the outlet. The linearity is described by the storage coefficient k (s), which defines the decay of the storage S (m^3) over time $Q = (1/k)S$.

The vMCM is a nonlinear coefficient routing method representing an approximation of the diffusive wave. Its parameterization (essentially the wave celerity and the hydraulic diffusivity) is variable in space and time thus accounting for effective physical hillslope and channel properties (slope, esp. slope/channel width) and for the space-time variability of the inflow rate. Ponce (1983) and Brunner (1989) compared it against fully unsteady flow equations, and stated a good comparison over a wide range of conditions. To calculate overland flow, hillslope width is considered here equal to the DEM cell size (25 m in this case). Channels are conversely assigned a geometry in the form of a relation between depth and width, which is derived by direct survey or by scaling relationships dependent on the drainage area. Details about the equations can be found in Ponce and Yevjevich (1978), Ponce (1989), Ponce and Chaganti (1994), whereas a description of their implementation in rainfall-runoff modeling is found in Montaldo et al. (2004).

The parameters that could not be derived by direct knowledge of the basin characteristics were derived from literature and adapted to the Vogelbach by calibration and subsequent validation. The calibration of the model is thus based on Kuntner (2002) who carried out extensive calibrations over different scales, including the catchment considered in this study. Main task of Kuntner (2002) was the calibration of the CN-Numbers for Switzerland and the preparation of a Swiss-wide SCS-soil type map.

Relevant hydrological parameters for the rainfall excess were the CN-value, the Soil-type (derived) and the initial loss. All parameters of the flow routing module were in a first step derived from literature and maps or from previous field studies as in the case of the depth-width relation of flow area and the roughness for channel and overland flow, which were estimated on the basis of Chow (1973), Chaudhry (1993) and Engmann (1986). The storage coefficient for the subsurface routing was initially derived from the recession limb of the measured hydrographs. All parameters were altered in their physically meaningful range to obtain the best fit for the calibration events of Kuntner (2002). In this paper, one event is considered for calibration of the hydrological model, three further events for validation. All of the considered events were tested against several criteria, among them the Nash-Sutcliff efficiency (with values between 0.95 and 0.98).

The Sediment Transport and Erosion Model

The coupled erosion model determines event-based sediment transport and erosion/deposition rates both on hillslopes and in channels assuming transport limited conditions. Because of the likely very limited contribution of the suspended sediment fraction due to the coarse nature of the large part of the sediments, the study focuses only on bedload transport in the channel. In general, erosion/deposition is computed from the sediment mass balance for each cell: $\mathrm{d}q_S / \mathrm{d}x = D$, where q_S represents the sediment mass flow (kg $\mathrm{m}^{-1}\,\mathrm{s}^{-1}$), x the flow distance (m) and D the net erosion/deposition rate (kg $\mathrm{m}^{-2}\,\mathrm{s}^{-1}$). Following Molnar et al. (2006) and assuming transport limited conditions, sediment fluxes can be set equal to the sediment transport capacity T_c, that is $q_S \approx T_c$. Transport limited conditions give a maximum potential for sediment transport and erosion/deposition and allow us to identify the most sensitive areas and the overall erosion potential of the basin.

Channel (bedload) sediment transport is implemented by the Schoklitsch equation for bedload in steep mountain streams, in which the critical flow q_c ($\mathrm{m}^2\,\mathrm{s}^{-1}$) is estimated by an empirical relation based on channel gradient (S) and particle diameter d_{16} developed by Bathurst et al. (1987):

$$q_S = \frac{2.5}{\rho_S / \rho} S^{2/3}(q - q_c) \quad \text{with} \quad q_c = 0.21\sqrt{(gS^{-1.12})}\,d_{16}^{1.5} \tag{5.3}$$

Here, q ($\mathrm{m}^2\,\mathrm{s}^{-1}$) is the specific water discharge rate, S (–) the slope, ρ (kg m^{-3}) the density of water and ρ_S (kg m^{-3}) the density of bed material.

Many sediment transport equations exist for hillslope overland flow, depending on the prevalent flow hydraulics and sediment characteristics (e.g., Julien and Simons 1985; Prosser and Rustomji 2000). In this paper, two commonly applied formulae were selected to investigate the effects of land use changes over time.

The first relation studied is based on *shear stress* (e.g., Foster and Meyer 1972; Mitas and Mitasova 1998) and gives the specific volumetric discharge as:

$$q_S = \frac{K}{\rho_S} \cdot (\tau_0 - \tau_c)^{\mu} \quad \text{with} \quad \tau_0 = g\rho r_{hy} s \tag{5.4}$$

where K is the coefficient of effective transport capacity (s), τ_0 the bed shear stress (N m^{-2}), τ_c the critical shear stress (N m^{-2}), μ an exponent (–) and r_{hy} the hydraulic radius (m). The second relation is based on *runoff*, assuming laminar flow over hillslope surfaces. Referring to Julien and Simons (1985), Prosser and Rustomji (2000) and Kilinc (1972) the general formula is given by

$$q_S = \frac{\alpha}{\rho_S} S^{\beta} q^{\gamma} i^{\delta} (1 - \tau_c / \tau_0)^{\varepsilon} \tag{5.5}$$

with q the specific water discharge (m^2 s^{-1}), i the rainfall intensity (m), ρ the density of water (kg m^{-3}), β, γ, δ, ε exponents (–) and α a transport capacity factor (–). Transport limited condition implies $\tau_0 \gg \tau_c$. Julien and Simons (1985) discussed the characteristics of overland flow and noted that overland flow can occur under turbulent and laminar conditions. They stated that the relation of viscous to inertia forces in combination with small flow depth forces the flow to be laminar up to a certain Reynolds number Re, which is indicated by Shen and Li (1973) to be approximately $Re = 2\,000$, after which smooth turbulent flow starts. The assumption of laminar flow is supported by Hessel (2002), where in field measurements on plot sizes with similar (or even steeper) slopes, land cover and velocities, the Reynolds number seldom exceeded the critical value.

The assumption of transport limited (conditions) implies that as much material is supplied as can be transported by the runoff. Raindrop impact – one of the most important factors for sediment detachment – plays a minor role for the transport limited case. Raindrop impact may influence under certain circumstances the flow conditions and therefore the sediment transport rate indirectly. However, in this study we assume the raindrop impact on the soil surface to be negligible due to the predominantly large sediment size and because large part of the catchment area is forested. For this latter reason there is no evidence of mud- or debris-flow development on the steep slopes. In addition, the interaction of erosion and sediment transport with the vegetation coverage and the overland flow has been approached conceptually by defining a characteristic roughness for each cover, rather than modeling with more sophisticated techniques such as those discussed by Nepf (1999). This seems to be indeed reasonable because of the consistency with the modeling scale which is based on a 25×25 m raster, which is rather coarse for the representation of small scales erosion features.

Land Cover Scenarios

Several hypothetical scenarios of land cover changes were developed on the basis of an extensive literature review. They all originated from the actual land cover derived from the raster maps of "Arealstatistik72". In a comprehensive study (Kirsch and Burlando 2005) two main driving scenarios (windstorm, climate change) were developed to study their possible effects on land cover and in turn on runoff, sediment yield, and erosion/deposition patterns. Here, only the windstorm scenario is illustrated.

After the Vivian (1990) and Lothar (1999) wind storms, the Swiss Federal Office for the Environment carried out several studies which analyzed the behavior and coping strategies of ecosystems (natural vs. man influenced) after heavy windstorms, with a focus on the re-establishment and growth of vegetation (forest in particular) after

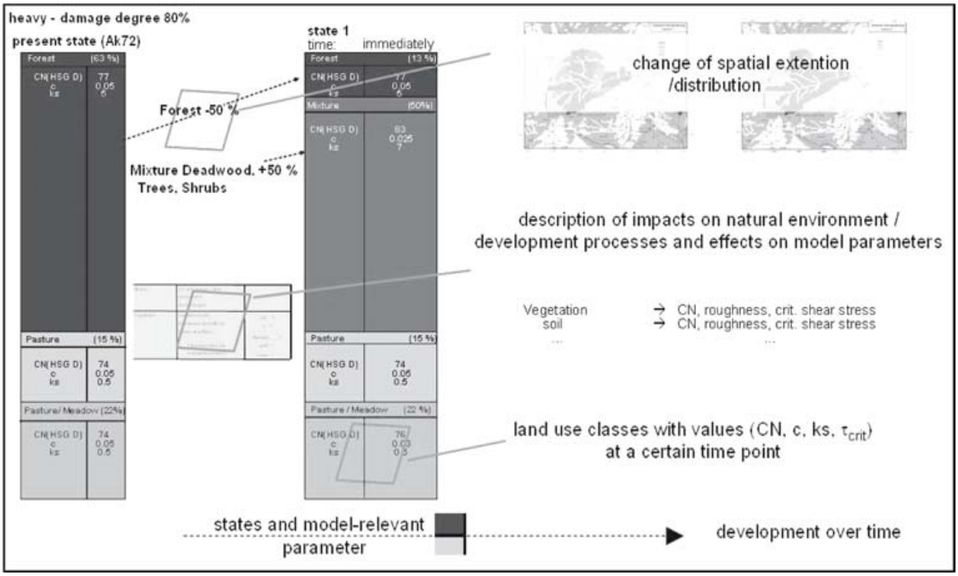


Fig. 5.11. Conceptualization of the time-space scenario evolution and the relevant model parameters and example of transition between phases (**a**) and (**b**) (adapted from Kirsch and Burlando 2005)

damaging events (e.g., Lässig and Schönberger 1997; Lässig and Motschalow 2000). On this basis, the developed windstorm scenario was divided into two subscenarios (removal vs. no removal of woody debris), of which only the latter is reported here. We developed a hypothetical, but qualitatively plausible evolution of vegetation cover in time and space after a damaging windstorm event, to describe the natural recovery of the forest. The land use change scenarios were translated into spatial and temporal changes of the model-relevant parameters, thus resulting in appropriate parameter sets for each condition within the scenario-development states. The scenarios were not only defined in terms of spatial and temporal changes but also in terms of different magnitudes of impact. Scenarios with a damage degree of 80% and 40% of the forest cover were considered at three different states in time, i.e. *(a)* immediately after the event and *(b)* 2 and *(c)* 7 years later. Figure 5.11 summarizes the construction and description of the parameter sets and their development over time – i.e. phases *(a)*, *(b)*, *(c)* – and space.

5.2.4 Results

A sensitivity analysis was carried out to illustrate the importance of input parameters and a baseline simulation was carried out to compare forest damage scenarios (for more details see Molnar et al. 2006; Kirsch and Burlando 2005; Hinz 2004). Table 5.1 lists the parameter values for the different scenarios and approaches considered in this paper. The investigated scenarios overall show similar impacts on runoff production, overland and channel flow velocity and depth (see Fig. 5.12). In the damaged areas, stronger runoff generation can be observed, associated with higher velocities due to smaller roughness.

Table 5.1. Parameter values for the base scenario simulation (see also Fig. 5.11)

Rainfall-runoff model			**Erosion model**		
$CN_{(forest/meadow/pasture)}$	77/74/76	(–)	$d_{16\ (channel)}$	22	mm
$k_{h\ (forest/meadow/pasture)}$	0.5/5/5	$m^{1/3}\ s^{-1}$	$\rho_{s\ (channel/hillslope)}$	2000/800	$kg\ m^{-3}$
k_{ch}	25	$m^{1/3}\ s^{-1}$	$\tau_{c\ (forest/meadow)}$	60–100/15	Pa
λ	0.05	(–)	α (Eq. 5.5)	6×10^4–2×10^5	
			β (Eq. 5.5)	1.66	
			χ (Eq. 5.5)	2.035	
			K (Eq. 5.4)	5×10^{-5}–1×10^{-4}	
			μ (Eq. 5.4)	1–1.5	

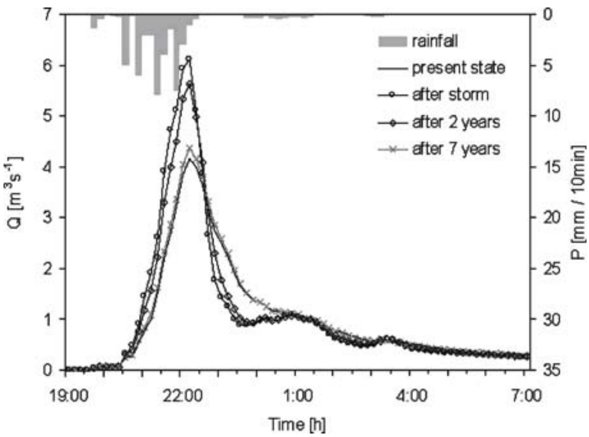


Fig. 5.12. Hydrograph comparison for the base and windstorm scenarios

This leads to faster flow concentration and higher peak flows as well as a faster flood wave rise, implying a potential increase of erosion in the channel. After 2 and 7 years, as the vegetation partially re-establishes, the basin response converges to that of the base scenario. This behavior reflects the changes of the model parameters corresponding to the land cover modifications.

Channel bedload sediment transport responds to the resulting flow scenarios accordingly. The 80% damage scenario (illustrated by Fig. 5.13a) produces stronger impacts on the hydrograph and sediment transport than the 20% one. Conversely, the simulation of overland flow erosion in response to scenarios shows a different behavior depending on the sediment transport formula used. Figure 5.13b shows that the shear stress and the runoff approach produce similar sedimentographs, but instantaneous values and volumes differ by a scale factor of about 0.5. The shear-stress approach was calibrated by Hinz (2004), and therefore assumed to be in the right order of magnitude. However, this suggests that a crucial point for any catchment scale quantitative modeling of overland sediment transport capacity is connected with the application and choice of sediment transport formulas. Their dependence

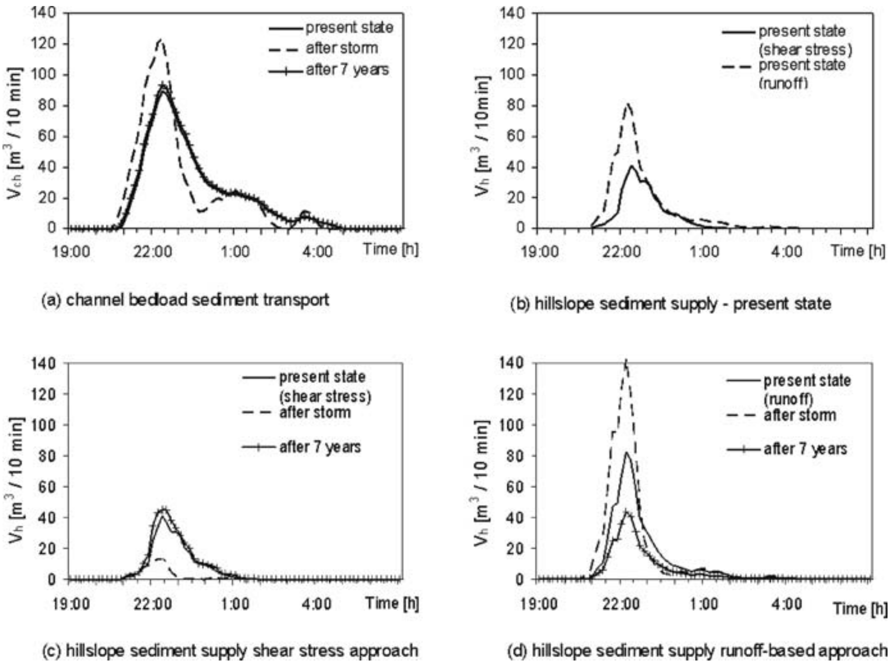


Fig. 5.13. Comparison of model results for different modeling approaches and different scenarios

on flow velocities, depths or flow rates reflects the changes of the surface hydraulics with a direct transfer to modeled erosion rates and transport capacities. This is well illustrated by Fig. 5.13c and d, showing, for a damage degree of 80%, a 28% decrease in hillslope sediment supply following the event when the shear stress based equation is used. Lower overland flow depths are the result of the balance between a drop in surface resistance and an increase in surface runoff following the event. On the other hand the runoff based equation predicts an increase in hillslope sediment supply due to the increase in surface runoff. As vegetation re-establishes in time, the volume of overland flow returns almost to the original conditions. The transport capacity volume still increases up to 118% because of the non-linear effect due to $\gamma \neq 1$ in Eq. 5.5.

5.2.5 Concluding Remarks

This study illustrates how scenarios and models may be developed and used to investigate in a spatially and temporally explicit fashion the impacts of land cover change on runoff, sediment yield and erosion/deposition patterns at the catchment scale. A distributed hydrological model was applied to several events with good results. The coupled hillslope erosion model that relates the transport rate to shear stress and flow depth, shows that especially hillslope sediment transport has to be tackled carefully.

Indeed, the choice of the sediment transport formula may significantly affect the magnitude of the erosion/deposition in relation to its functional dependence on the hydraulics of overland flow. In this sense, the modeler can influence the results by the way in which the equations are chosen and the model parameters are adjusted to reflect land cover change scenarios. Equations 5.4 and 5.5 use different hydraulic variables to determine the erosion rates. In the case of the shear stress approach, changes in surface roughness on hillslopes in the scenarios affect flow depth, which may decrease despite an increase in surface runoff following forest damage. The runoff approach directly reflects the increase in surface runoff due to the land use change. Both approaches can be used after calibration to match observed volumes in time-static modeling, meaning modeling based on the current state of the catchment. The differences shown in the resulting sedimentographs as well as in the predicted response to land cover changes (time-variable modeling) point out the importance of understanding the driving mechanism of erosion in relation to overland flow processes. The decision which of the two approaches is more appropriate depends on the laminar or turbulent flow characteristics and on soil erodibility. It is apparent from the literature that this question requires further experimental investigations across different scales to clarify open issues. To address these connections additional field studies have to be carried out in which shallow overland flow on different slopes and land cover is observed and analyzed together with sediment transport. Among these, erosion studies on natural hillslopes carried out by means of silt fences can be mentioned, as well as plot-size studies to evaluate man-made land cover changes, to study local effects for different slopes, soils and land cover combinations. This is necessary to overcome the limitations of insufficient or weak calibration and validation of existing models due to limited or lacking field data. In particular this targets conceptual models improving their predictive capabilities and acceptance.

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5.3 Transport and Fate of Dissolved and Suspended Particulate Matter in the Middle Elbe Region during Flood Events

5.3.1 Introduction

Although flood events are natural events, they may have an impact on the condition of a river system, i.e. when polluted deposits are involved in the process. The quality of the transported matter is mainly affected by the origin of the flooding water as well as the remobilization of deposits within the catchment area (e.g., groyne fields, lock-and-weir systems, mining and industrial areas, sewage plants). Transport and fate of a contaminant in the water body are significantly influenced by the ratio of the concentrations in the dissolved and the particulate phase. Depending on their morphology, floodplains act as sink for suspended matter (Engelhardt et al. 1999; Friese et al. 2000; Hanisch et al. 2005; Costa et al. 2006).

In the Middle Elbe groyne fields are the characteristic morphological feature. The resuspension of deposited, not yet consolidated groyne field sediments can lead to the first SPM (suspended particulate matter) peak in the course of a flood, a long time before the flood crest has arrived (Spott and Guhr 1996; Baborowski et al. 2004). All further peaks are the result of matter flow and inputs further upstream of the location, e.g., tributary inflows. Therefore the remobilization processes in groyne fields upstream of a sampling location play the most important role for the right starting time of a flood sampling campaign (Baborowski et al. 2005).

The assessment of sediment dynamic and pollutant mobility in rivers requires the integration of various experimental and modeling techniques (Förstner et al. 2000; Förstner 2004). They have to include aspects of erosion (Gust and Müller 1997; McNeal et al. 1996; Haag et al. 2001), transport (Brunk et al. 1997) and deposition (Asselmann and Middelkoop 1995; Kronvang et al. 2002; Krüger et al. 2006). In the paper results of investigations on transport and deposition in the middle part of the river Elbe are presented.

5.3.2 Aim

Former flood investigations along the Elbe River are not consistent with respect to sampling locations, begin of sampling, sampling times and methods. Furthermore they often do not consider the travel time of matter transport within the river (Baborowski et al. 2005). To overcome these deficits, investigations were performed considering both, local specific erosion values of the discharge as well as the travel time of the flood wave along a river stretch. The results provide a basis for a better understanding of the flood dependent transport processes in the Middle Elbe.

To assess the risk potential of the polluted river SPM for the floodplains, a well described floodplain (Büttner et al. 2006) was chosen to show exemplarily the potential sink function of floodplains for SPM transported with the flood wave.

5.3.3 Study Site

The investigations were carried out in the middle part of the river Elbe at river km 318, left bank near Magdeburg, and at river km 454, left bank in Wittenberge (German mileage). The considered stretch covers 136 km of the total length of the German part of the Elbe, which altogether amounts to 730 km.

There are approximately 1 950 typical groynes located between the two sampling sites. Along the whole course of the river Elbe in Germany there are approximately 6 900 groyne fields. The geographical position of the sampling sites is shown in Fig. 5.14.

The Sampling Site Magdeburg is part of the monitoring program of the International Commission for the Protection of the Elbe (IKSE/MKOL). The water quality at this site depends on inputs of stretches of the upper Elbe (Czech part, Dresden industrial region) as well as inputs from the polluted tributaries Mulde and Saale. Their confluences are 59 km and 27 km upstream of the location respectively. Therefore under normal discharge conditions the water quality at this point represents the pollution situation of the Middle Elbe.

Between sampling site Magdeburg and the Sampling Site Wittenberge no further significant pollution input occurs. The total retention area between both sites is around 19 700 ha (IKSE/MKOL 2001). The Floodplain Schönberg (upstream of Wittenberge) covers an area of ~200 ha.

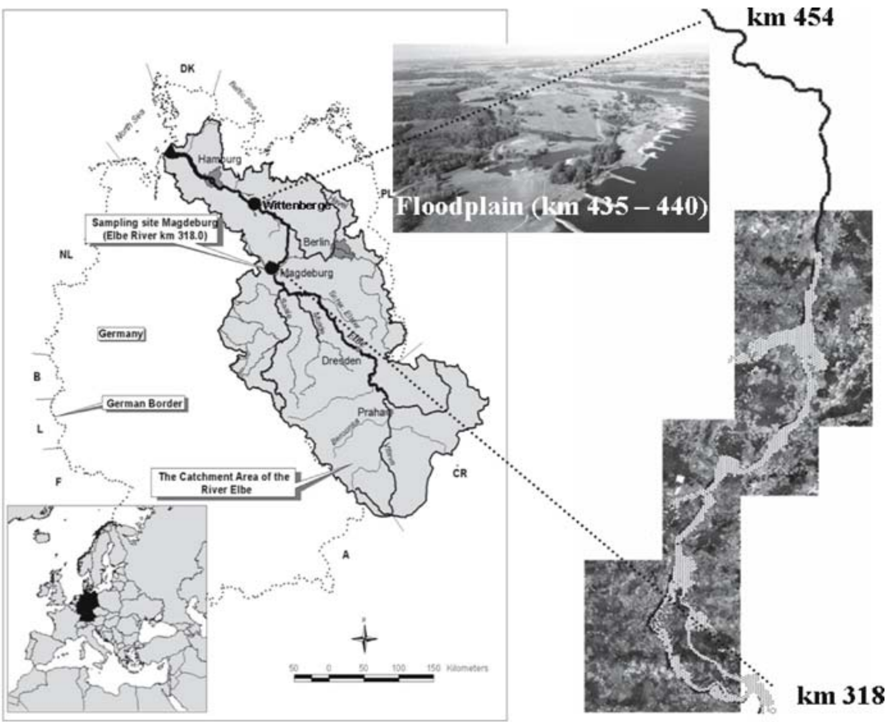


Fig. 5.14. Catchment area of the river Elbe with position of the sampling sites

5.3.4 Methodology

Investigations of two different flood waves were performed during a spring flood in February and March 2005, to get a better understanding of the main pollution potentials and pollution pathways in the Middle Elbe catchment area.

Sampling Strategy

Daily water samples were taken at the sampling sites in Magdeburg and Wittenberge considering local specific discharge threshold values as starting point for the measurement campaign.

For the sampling site Magdeburg at river km 318 the discharge threshold is about 800 $m^3 s^{-1}$ (Baborowski et al. 2004; Spott and Guhr 1996). For Wittenberge a threshold value of 1 080 $m^3 s^{-1}$ was chosen. At this critical value the Elbe water begins to flow into the floodplain.

The investigations were supplemented by monitoring deposited matter in the floodplain Schönberg using artificial lawns as sediment traps (Friese et al. 2000; Krüger et al. 2006). The artificial lawns aim at the simulation of the natural soil coverage (pasture or mown grassland). With respect to the natural topography, the traps were arranged at sampling points which cover a typical spectrum of morphological units in riparian areas like bayou of flood channel, not drained depression and plateau positions.

Main Parameter Water Samples

Dry weight of SPM was measured according to German Industrial Standards (DIN 38409 part H2, filtration onto Whatman GF/F glass fiber filter). The particle size distribution of SPM was counted in the range of 2 to 200 µm using an optical instrument, based on single particle evaluation (Baborowski 2002).

Heavy metals and arsenic (As) were analyzed in filtered (<0.45 µm) and unfiltered samples. The filtered samples were acidified with HNO_3, the unfiltered digested with HNO_3/H_2O_2 in a microwave equipment.

Aluminum (Al), iron (Fe), manganese (Mn) and zinc (Zn) were determined by optical emission spectrometry with inductively coupled plasma (ICP-OES), while arsenic, cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), nickel (Ni), uranium (U) and zinc (Zn) were measured using mass spectrometry also with inductively coupled plasma (ICP-MS).

Floodplain Sediments

To estimate the sediment deposition during the flood artificial lawns which act as sediment traps were arranged at 7 positions (5 replicates) on the floodplain.

After drying, the collected sediments were analyzed by means of energy dispersive X-ray fluorescence (EDXRF) with regard to their contents of heavy metals and arsenic.

Total organic carbon of the sediments was analyzed in triplicate with a C(H)NS analyzer (Elementar vario EL) after drying at 105 °C and grinding of the samples.

Load Calculation

The deposition (D) between Magdeburg and Wittenberge (taking into account a constant travel time of 2 days) was calculated as:

$$D = \sum_i \left[F_M(t_i) - F_W(t_{i+2}) \right] \ , \quad i = 1,\ldots,n \tag{5.6}$$

with

- $F_M(t_i) = Q_M(t_i)C_M(t_i)$ and $F_W(t_i) = Q_W(t_i)C_W(t_i)$
- F_M, F_W: contaminant load at Magdeburg and Wittenberge, respectively
- Q_M, Q_W: discharge at Magdeburg and Wittenberge, respectively
- C_M, C_W: contaminant concentration at Magdeburg and Wittenberge, respectively
- t_i: i^{th} date

5.3.5 Results and Discussion

The particle size distribution of the transported SPM exemplarily is given in Fig. 5.15 for the Magdeburg sampling site. In agreement with the results of prior studies (Baborowski et al. 2004; Baborowski 2002) changes of the particle number concentrations in the range between 2 and 20 µm dominated during both flood waves. Particles < 20 µm constitute the largest portion of the total particle number (Fig. 5.16). This fraction is especially important for the transport of pollutants which tend to sorption. Their portion of mass of the SPM is small.

From Fig. 5.15 the remobilization of sediments at the early stage of the flood can be seen. Thereby the peaks of particle number concentrations of both grain size fractions as well as the concentration of SPM occurred at the same time (Fig. 5.16). While the concentrations of SPM and of the particle fraction > 20 µm decreased continuously after the maximum is reached, the particle concentrations < 20 µm remained elevated for longer, especially during the second wave. This indicates the supply of fine grained sediments from the tributaries Mulde and Saale. Owing to the storage capacities in its lower reach (Mulde reservoir, Saale lock-and-weir systems), a shift towards fines rather than coarse particles has been demonstrated in prior investigations (Baborowski et al. 2004). Hence, the flood dependent matter transport in the middle part of the river Elbe

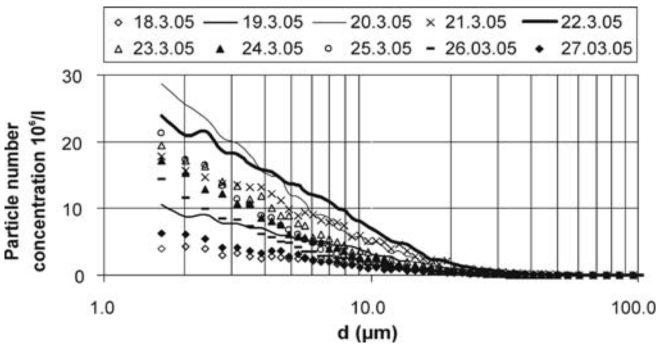


Fig. 5.15. Particle size distribution of the SPM at river km 318

Fig. 5.16.
Particle number concentration (*N*) and dry substance (*DS*) of the SPM

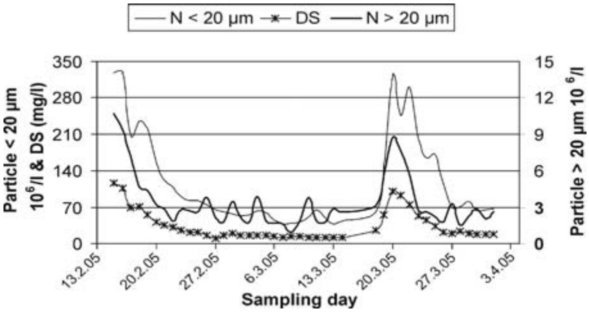
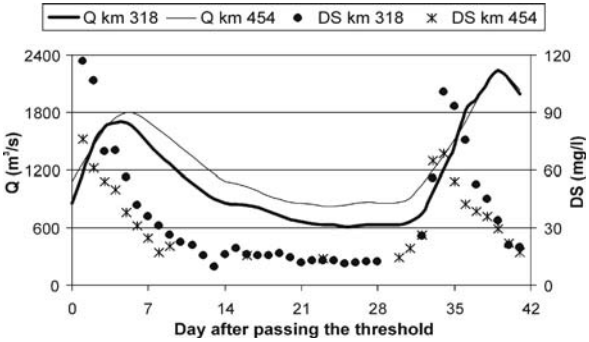


Fig. 5.17.
Discharge (*Q*) and dry substance (*DS*) of SPM at river km 318 and river km 454



seems to be rather limited by transport capacity of the groyne field sediments than by supply of sediments from the tributaries Mulde and Saale.

The travel time of the Elbe River between Magdeburg and Wittenberge was about 2 days. Therefore, if flow data of river km 318 are compared with flow data of river km 454, the corresponding time series are shifted by two days to account for the time the water takes to flow along this river stretch. Consequently the data on the x-axis are normalized by means of the time since the threshold has been exceeded for the first time.

As expected the highest SPM concentrations at both sampling sites were measured directly after passing the discharge threshold due to remobilization of groyne field sediments (Fig. 5.17).

This result is important for transport modeling. To determine initial conditions as well as boundary conditions in numerical pollution transport modeling, suspended sediment concentrations are required. However, if the number of available data is insufficient, it is necessary for reliable model calculations to provide the typical graph of SPM concentration based on the sparse data available. Therefore, local knowledge of the typical course of SPM during a flood event is essential. E.g. it has to be taken into account during all steps of model processing that the SPM maximum occurs prior to the maximum discharge (Lawler et al. 2006). Uncertainties in forecasting flood extend and depositions are often caused by uncertainties in boundary conditions rather than uncertainties in model parameters (Lane 1998).

Moreover the results, shown in Fig. 5.17, indicate decreasing SPM values along the river stretch between river km 318 and 454. Corresponding to the SPM, the concentrations of heavy metals and As decreased as well (Fig. 5.18).

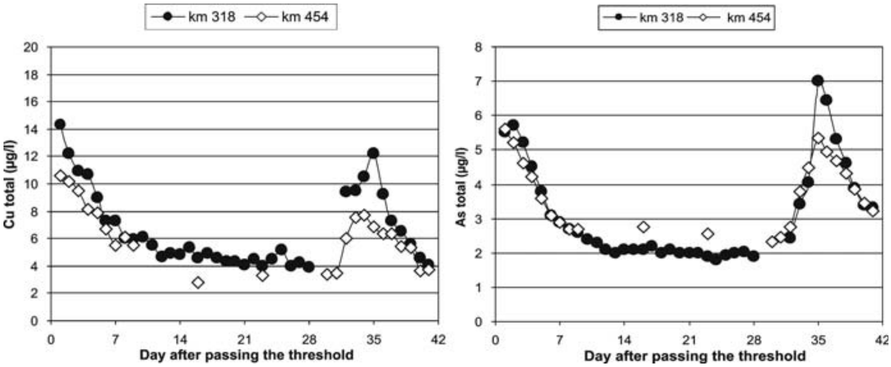


Fig. 5.18. Graph of Cu (*left*) and As (*right*) at river km 318 and river km 454

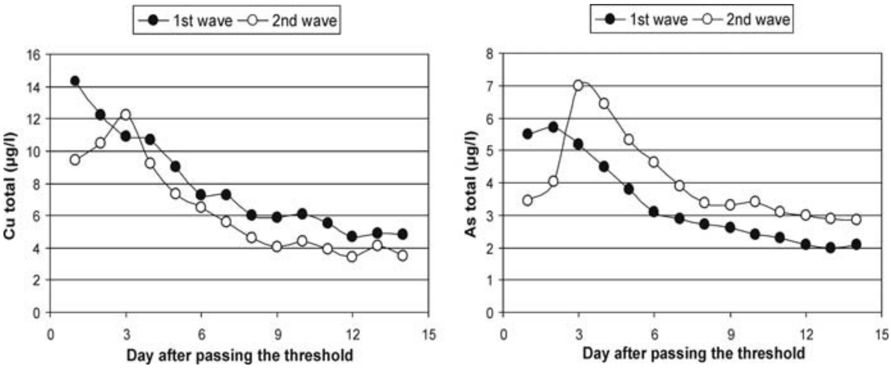


Fig. 5.19. Comparison of both waves for Cu (*left*) and As (*right*) at river km 318

Graphs of concentrations during the first and the second wave show a different course. To give an example, the comparison of both waves is presented for the sampling site Magdeburg in Fig. 5.19. The maximum concentrations of Cu and As do not show the same values in both waves and also occur at different times after passing the discharge threshold, indicating the influence of different sources of pollution.

The ore and companion elements Cu and As are typical pollutants originating from former mining activities in the catchment areas of the Elbe tributaries Mulde and Saale. Despite of a significant decrease of pollution in the river Elbe over the last fifteen years, contaminations with heavy metals, e.g., in sediments, are still present in the river basin (Heise et al. 2005). During flood the pollutants of these secondary deposits are remobilized by erosion and hence, appear in the water body. Due to regional geographic peculiarities in the Middle Elbe as well as results of prior studies, the element As can be considered as indicator for pollution inputs of the river Mulde, whereas Cu acts as indicator for inputs of the river Saale (former copper shale mining). Therefore, together with the evaluation of the discharge development in the Elbe and its tributaries (data not shown) during the flood, the dominating influence of the Saale on the first wave and of the Mulde on the second wave is reflected in Fig. 5.19.

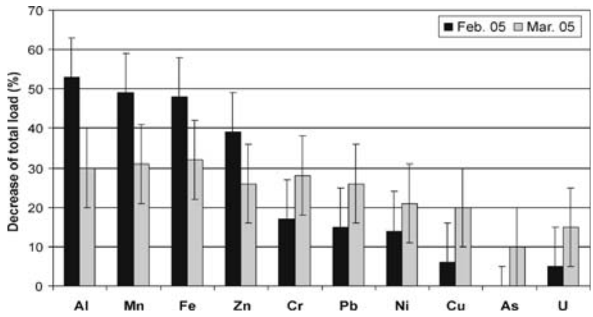


Fig. 5.20.
Decrease of total element load between river km 318 and river km 454

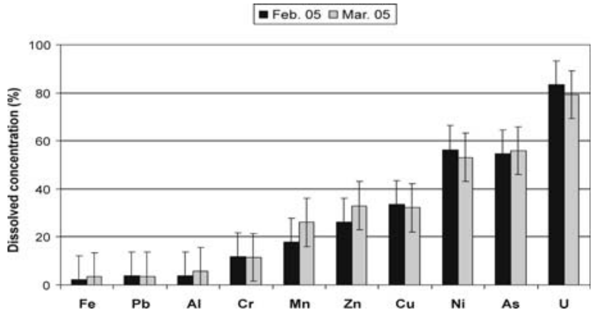


Fig. 5.21.
Percentage of transport in dissolved phase

An overview of the longitudinal matter transport is given in Fig. 5.20. It shows that over the entire flood event, pollutants are retained in the river reach under consideration.

The trace metal loads decreased by up to 50% between the two sampling sites. Thereby, the decrease of element loads varied significantly between the pollutants. It was highest for clay reference compounds (Al, Fe, Mn). The trapping of the clay fraction by sedimentation is evident and much larger than observed for the other, also mainly in particulate form transported elements Cr, Pb and Zn (Fig. 5.21). On this occasion a grain-size effect seems to be the reason for the different settling behavior of the pollutants. In contrast, the distribution between dissolved and particulate phase is more decisive for the observed different transport behavior of the elements Ni, Cu, As and U. Especially Ni, As and U were mainly transported in the dissolved fraction (55 to 80%, Fig. 5.21), responsible for the transport to the sea.

Moreover, the decrease of element loads for Al, Fe, Mn and Zn was higher during the first wave than observed for the second wave (Fig. 5.20). This can be interpreted as indication that these elements are originate to a larger extent from the groyne fields and exhibit, in contrast to the other elements, a hysteresis behavior. Cyclically decreasing SPM and element concentrations in the case of Elbe flood waves following each other in quick succession have been described in detail by Spott and Guhr (1996). However, this result together with the findings discussed for Fig. 5.16 underlines the hypothesis that the transport in the Middle Elbe is rather limited by transport capacity than by supply of sediments.

In summary, the reduction of loads can be explained by high retention due to sedimentation in floodplains, old arms and slack water zones. For example, the SPM retention between river km 318 and km 454 was calculated to be about 19 000 t for the first

Table 5.2. Composition of the sediments deposited on the floodplain Schönberg

Parameter	Unit	1st wave, $n = 7$ Minimum–maximum	2nd wave, $n = 7$ Minimum–maximum
Deposition	$t\ ha^{-1}$	0.09 – 1.8	1.0 – 4.0
Si/Al	–	3.1 – 4.0	3.6 – 4.6
Al_2O_3	%	7.8 – 13.8	10.4 – 11.8
SiO_2	%	39.4 – 54.9	45.8 – 53.9
Fe_2O_3	%	4.4 – 6.7	5.0 – 6.1
C	$g\ kg^{-1}$	71 – 134	80 – 115
N	$g\ kg^{-1}$	6.7 – 14.1	8.2 – 12
S	$g\ kg^{-1}$	2.2 – 4.7	2.5 – 3.3
Mn	$mg\ kg^{-1}$	492 – 2556	1775 – 4593
Zn	$mg\ kg^{-1}$	700 – 3350	943 – 1182
As	$mg\ kg^{-1}$	26 – 56	47 – 63
Cr	$mg\ kg^{-1}$	83 – 128	106 – 123
Cu	$mg\ kg^{-1}$	104 – 173	96 – 128
Ni	$mg\ kg^{-1}$	42 – 67	49 – 84
Pb	$mg\ kg^{-1}$	158 – 1138	141 – 163
U	$mg\ kg^{-1}$	0 – 4	2 – 4

flood wave. This is about 25% of the total SPM load passing km 318. Balancing these findings with the total floodplain area of 19700 ha (MKOL/IKSE 2001), a deposition of nearly 1 t ha^{-1} can be estimated for the first flood wave.

The calculations of the deposition in the sediment traps of the floodplain Schönberg underline these findings (Table 5.2).

The average SPM deposition was 0.98 t ha^{-1} for the first wave and 2.02 t ha^{-1} for the second wave. The overall retention between km 318 and km 454, basing on the sediment trap investigations of both waves for the whole period of floodplain inundation can be estimated for Zn: 68 t, Pb: 13 t, Ni: 3 t, Cu: 7 t, As: 3.3 t and U: 0.15 t.

This demonstrates a potentially risk for the land use resulting from the function of floodplains as a sink for polluted SPM transported to the sea. Nevertheless, the determined deposition rates and contamination loads indicate that single floods in the area of the Lower Middle Elbe can hardly influence the quality of the soils in the central floodplain whose pollutant loads are the result of a pollutant input for decades and centuries. On the one hand, still today the contamination of the recent flood sediments corresponds to a large extent to the quality of the topsoils (Krüger et al. 2005). On the other hand, 0.09–4.0 t ha^{-1} sediment input (Table 5.2) with an assumed bulk density of 1 g cm^{-3} (after Krüger et al. 2005) solely amounts to a maximum of 0.6% of the 0–10 cm soil depth interval that is used as standardized soil sampling. This is not sufficient to alter the pollution state of this soil depth interval significantly.

5.3.6 Conclusions

Discharge related investigations allow a better understanding of the flood dependent matter transport. In view of both, local specific erosion thresholds as starting times for the measurement at a sampling site as well as the travel time along the regarded river stretch, different pollution pathways in the river system can be detected. Furthermore a more detailed description of the floodplains sink function for sediments and pollutants becomes possible, if instream and floodplain investigations are combined.

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5.4 Modeling P-Fluxes from Diffuse and Point Sources in Heterogeneous Macroscale River Basins Using MEPhos

5.4.1 Introduction

As the inventory of the EU-water framework directive (EU-WFD) revealed in 2004, high nutrient inputs are a major concern for most inland and coastal waters in Germany. Consequently the "good status" in water quality will not be achieved for a high percentage of water bodies. To improve water quality until 2015 the EU-WFD demands the drawing-up of detailed river basin management plans and programmes of measures until 2009. This provokes a demand for suitable instruments and models resp. to be applied for the area- and pathway-differentiated quantification of nutrient inputs. Model applications should aim at the localization of critical source areas within river basins as a basis for proposing measures of emission control which are adapted to site properties.

In the following the new empirical emission model MEPhos is presented, which enables a systemic quantification of P-loads in large heterogeneous river basins (Tetzlaff 2006). MEPhos differentiates between diffuse and point sources as well as between eight pathways.

The model is applied for the macroscale basins of the river Ems (12 900 km^2) and parts of the river Rhine (12 200 km^2). The river Ems basin is to be found in the north-western German lowlands and is dominated by sandy and boggy soils under intensive agricultural use. The investigation area Rhine is characterized by upland conditions with forest and grassland use. Furthermore the catchment comprises of sub-regions with high industrial and population densities (partly up to 2 900 citizens per km^2).

5.4.2 MEPhos Model Description

The new model MEPhos consists of following features (Tetzlaff 2006, Fig. 5.22):

- Area-differentiated modeling approach for the quantification of mean annual P-inputs from point and diffuse sources to surface waters
- Differentiation between the eight pathways artificial drainage, groundwater-borne runoff, erosion, wash-off, rainwater sewers, combined sewer overflows, municipal waste water treatment plants, industrial effluents
- Consideration of P-retention in running and standing waters

The area-differentiated modeling approach to quantify diffuse P-inputs via artificial drainage, groundwater-borne runoff, erosion and wash-off is based on phosphotopes (Fig. 5.22). Phosphotopes are regarded as homogeneous types of sub-areas representing discontinuous source areas for diffuse phosphate inputs. Analogous to hydrotopes,

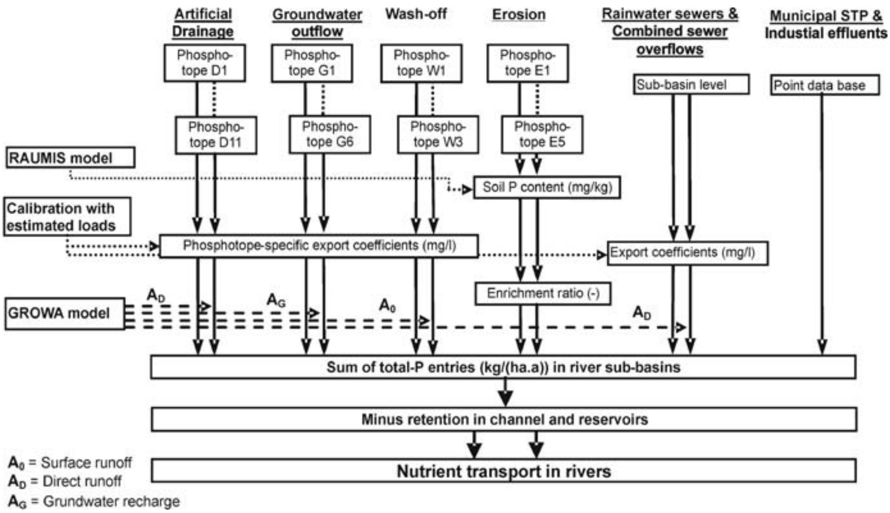


Fig. 5.22. Overview of the MEPhos model

phosphotopes include a set of different parameters, which control P-emissions. These parameters like soil types, land use and hydraulic connectivity to surface waters can be quantified for macroscale investigation areas by already available data sets from federal and state authorities as well as by results from hydrologic and agro-economic models. Phosphotopes are processed in GIS by classification and intersection of data sets. Details are given in Tetzlaff et al. (2007).

After modeling mean annual P-inputs via all eight pathways the emissions are summed up for river sub-basins related to water quality gauges (Fig. 5.22). Then the mean annual load of an upstream sub-basin is added and the P-retention subtracted (Eq. 5.7). This enables a validation of modeled P-loads by comparison with mean annual P-loads estimated from measured water quality and discharge data.

$$F_{EZG} = \left[\left(\sum F_{Dr} + \sum F_{Gw} + \sum F_{Abschw} + \sum F_{Eros} + \sum F_{Tk}\right) + \sum F_{KA} + \sum F_{ID} + \sum F_{Mw}\right] + F_{OL} - R_F - R_S \tag{5.7}$$

F_{EZG} = estimated load for river sub-basin, F_{Dr} = P-inputs via artificial drainage, F_{Gw} = P-inputs via groundwater-borne runoff, F_{Abschw} = P-inputs via wash-off, F_{Eros} = P-inputs via erosion, F_{Tk} = P-inputs via rainwater sewers, F_{KA} = P-inputs from municipal waste water treatment plants, F_{ID} = P-inputs from industrial effluents, F_{Mw} = P-inputs via combined sewer overflows, F_{OL} = P-load of an upstream river sub-basin, R_F = P-retention in running waters, R_S = P-retention in standing waters.

The retention of P due to settling and biogeochemical transformation within surface waters is modeled separately for running and standing waters. Behrendt and Opitz (2000) have developed an approach to describe the retention in rivers integratively, which is applied in the MEPhos model (Eq. 5.8). The coefficients a and b account for the influence of the catchment size and were quantified by Behrendt and Opitz (2000).

$$F = \frac{1}{1 + R_F} E \quad \text{and} \quad R_F = a q^b \tag{5.8}$$

F = modeled load for running waters (t a^{-1}), R_F = factor to describe P-retention in running waters (–), E = sum of modeled P-emissions (t a^{-1}), q = discharge per unit area (l s^{-1} km^{-2}), a, b = coefficients (–).

Equation 5.8 cannot record the increased P-retention occurring e.g. in reservoirs due to reduced flow velocity, increased travel time as well as higher sedimentation rate (Alexander et al. 2002; Kirchner and Dillon 1975). As a consequence, the particulate P-load transported as suspended matter decreases. According to Molot and Dillon (1993) P-retention in reservoirs and enbarraged lakes can be calculated as follows (Eq. 5.9):

$$R_S = \prod \exp(-k_r\, q^{-1}) \tag{5.9}$$

R_S = P-retention in reservoirs, k_r = retention coefficient ("settling velocity") (m a^{-1}), q^{-1} = reciprocal areal hydraulic load of reservoirs (a m^{-1}).

The following section focuses on modeling P-inputs to surface waters via erosion and presents an example of MEPhos results for erosion-related inputs.

5.4.3 Modeling Sediment and P-Inputs to Surface Waters Via Erosion

P-inputs to surface waters via erosion are controlled by soil loss from arable land within river basins, sediment delivery ratio, P-content in the top soil as well as by P-enrichment during the erosion process (Frede and Dabbert 1999; Auerswald 1989). Therefore diffuse P-inputs to surface waters via erosion are modeled after Eq. 5.10, based on the universal soil loss equation adapted to German conditions (Schwertmann et al. 1990):

$$E = (R \cdot K \cdot LS \cdot C) \cdot S \cdot PG \cdot ER \tag{5.10}$$

E = mean annual P-inputs via erosion (kg ha^{-1} a^{-1}); R = rain erosivity factor (N ha^{-1} a^{-1}); K = soil erodibility factor (t h ha^{-1} N); LS = combined slope length and steepness factor (–); C = crop and management factor (–); S = sediment delivery ratio (%/100); PG = content of total-P in the top soil (mg kg^{-1}); ER = enrichment ratio (–).

Because of the high sensitivity of the factors L and S (Auerswald 1989) they are not quantified by standard literature values, but calculated as LS-factor using algorithms of Moore and Wilson (1992). The slope shape is another important relief feature, because it controls the concentration of surface runoff (Auerswald et al. 1988). For modeling erosion with MEPhos the slope shape is taken into account by correction factors after Prasuhn and Grünig (2001) for the modification of the LS-factor.

With regard to proposals for efficient eutrophication reduction measures (Sect. 5.4.5), modeling of diffuse P-inputs by MEPhos has to be performed on the basis of phosphotopes, i.e. area-differentiated. Phosphotopes for modeling erosion inputs represent source areas for release of sediment and particulate-P and are made up by erodible arable land which is hydraulically connected to the network of rivers and flow paths. Morphological flow paths are modeled using a highly-resolved digital terrain model and the algorithm D Infinity (D∞) (Tarboton 1997). The parameters needed for the application of D∞ are calibrated with flow paths mapped in a test area in order to guarantee a representative and reliable simulation of flow paths. After selecting those flow paths, which are connected to the river network, the remaining paths are buffered with stripes of 2×30 m width according to findings of Sommer and Murschel (1999) and Fried et al. (2000) resp. Only those sub-areas contained in the 60 m strips are regarded as hydraulically connected. Then phosphotopes for modeling P-entries via erosion are derived by intersecting the data sets of erodible arable land with the buffered network of flow paths and rivers. Depending on their erosion potential five different phosphotopes are distinguished. By performing this process of disaggregating the eroding and hydraulically connected arable land, sediment deliveries to surface waters are also determined. MEPhos sediment deliveries vary between 3 and 29% with a mean of 11%. Werner et al. (1991) estimated the mean sediment delivery of former Western Germany at 8%.

According to Eq. 5.10 the level of particulate P-inputs results not only from the sediment delivered to surface waters, but also from the P-content of the top soil and the enrichment ratio. The calculation of the total-P content in the top soil is based on P-surpluses and clay content following Behrendt et al. (1999). P-surplus is modeled on a county level employing the agro-economic model RAUMIS developed and run by the German Federal Agricultural Research Centre (IAP and FAL 1996), the clay content is given by soil maps on a scale of 1:50 000 (Fig. 5.22).

For considering the selective effect of erosion induced by water an enrichment ratio has to be determined. For this a method is employed in the MEPhos model requiring measured water quality, above all about the P-content of suspended matter (Behrendt et al. 1999). According to Eq. 5.11 the enrichment ratio for river sub-basins is given by:

$$ER = \frac{P_S}{P_{OA}} \tag{5.11}$$

ER = enrichment ratio (–), P_S = P-content of suspended matter given at discharges above a "critical" level (mg kg^{-1}), P_{OA} = P-content in the top soil of sediment delivering arable land (mg kg^{-1}).

Then the mean annual P-entry (1995–1999) via erosion is modeled using Eq. 5.10. An example of the erosion modeling results is shown for eastern parts of the river Rhine basin in Fig. 5.23.

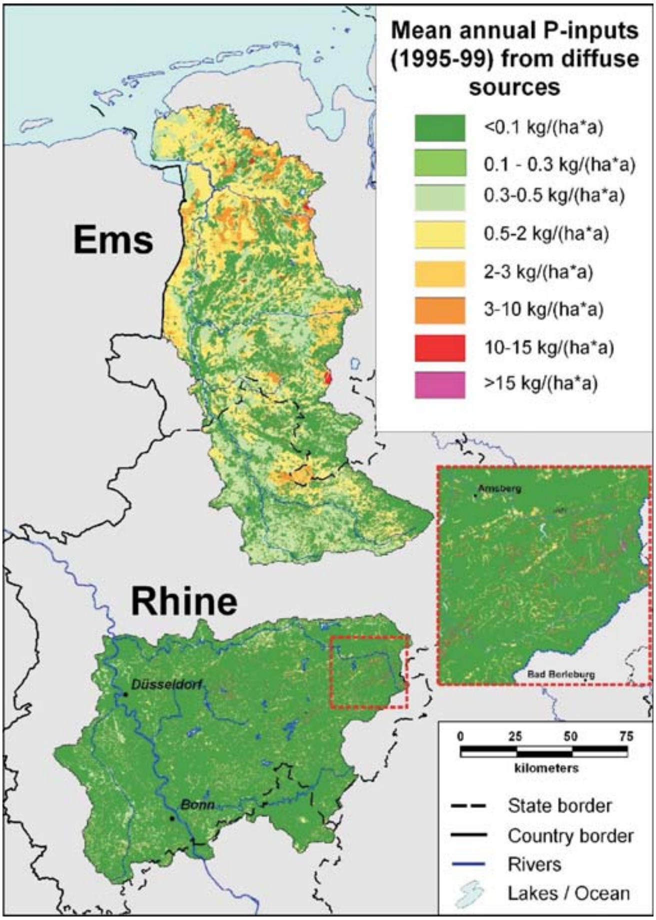
A check for validity of MEPhos modeling results can be performed by a comparison with mean annual P-loads estimated from measured data on both water quality and discharge. However, this requires quantification of the total sum of diffuse and point P-inputs via all eight pathways (see following chapter). A validation of modeling results for one single pathway is not feasible.

5.4.4 Total P-Inputs from Diffuse and Point Sources and Validation of Model Results

Diffuse P-inputs via the pathways artificial drainage, groundwater-borne runoff and wash-off are also modeled with MEPhos and are added to the erosion-related P-inputs. As shown in Fig. 5.23, mean annual P-inputs vary between <0.1 and >15 kg P ha^{-1} a^{-1} in both investigation areas. The river Ems basin is characterized by larger and coherent source areas. P-inputs between 0.5 and 3 kg P ha^{-1} a^{-1} are typical for the northern part and can be traced back to phosphotopes of the category "deep-ploughed cultivated raised bog soils under agricultural use" emitting P via artificial drainage. This pathway accounts also for the highest diffuse P-emissions of more than 10 kg P ha^{-1} a^{-1}, which result from raised bog soils under grassland use.

In the sub-basins of the river Rhine low P-inputs of <0.1 kg P ha^{-1} a^{-1} dominate, which result from geogenic background, represented by phosphotopes of the groundwater pathway. Medium and high P-inputs appear as small spots and are a consequence of erosion on arable land (Sect. 5.4.3, Fig. 5.23). P-inputs via erosion vary between less than 0.1 and more than 15 kg ha^{-1} a^{-1}. Erosion-related P-inputs are characterized by a small spatial extent of sediment delivery areas with highly differing input levels (Fig. 5.23, enlarged section). The small spatial extent of the phosphotopes results from the disaggregation of erodible arable land, its intersection with buffered flow paths and the application of highly-resolved data sets, above all the digital terrain model with a resolution of 10×10 m^2. High erosion-related P-inputs are to be found mainly in sub-regions with steep slopes and widespread arable land use (Fig. 5.23). For the Ems and Rhine investigation areas mean annual P-inputs of 2.94 kg km^{-2} a^{-1} and 14.8 kg km^{-2} a^{-1} resp. are modeled. The difference reflects the contrasting natural and land use conditions between the two areas (Sect. 5.4.1).

Fig. 5.23.
Mean annual P-inputs (1995–1999) from diffuse sources in the investigation areas Ems and Rhine (enlarged: spatial patterns of erosion induced P-inputs in eastern parts of the river Rhine sub-basin)



After modeling mean annual P-inputs via all other pathways diffuse and point source emissions can be summed up. This leads to mean annual total loads for the period 1995–1999, which amount to 1 666 t a^{-1} in the river Ems basin and to 1 574 t a^{-1} for the river Rhine sub-basin. Figures 5.24 and 5.25 show the relevance of all eight pathways for the mean P-input.

In the entire river Ems basin diffuse P-emissions account for 87% of all P-inputs to surface waters. It can be stated for nearly all of the 56 sub-basins that diffuse inputs dominate and that the pathway artificial drainage is of highest importance in this lowland river basin. The percentage of inputs via artificial drainage differs between 16 and 89% in total, it is above 50% in most cases (Fig. 5.24). The highest loads from non-point sources are modeled for the lower reaches of the river Ems and its tributaries, which corresponds with the decreasing population density and the increasing intensity of agricultural activities from south to north. Due to the origin of these high loads from artificially drained agricultural land, i.e. as mainly soluble reactive and therefore highly algae-available P, the agricultural activities on raised bog soils result in a tre-

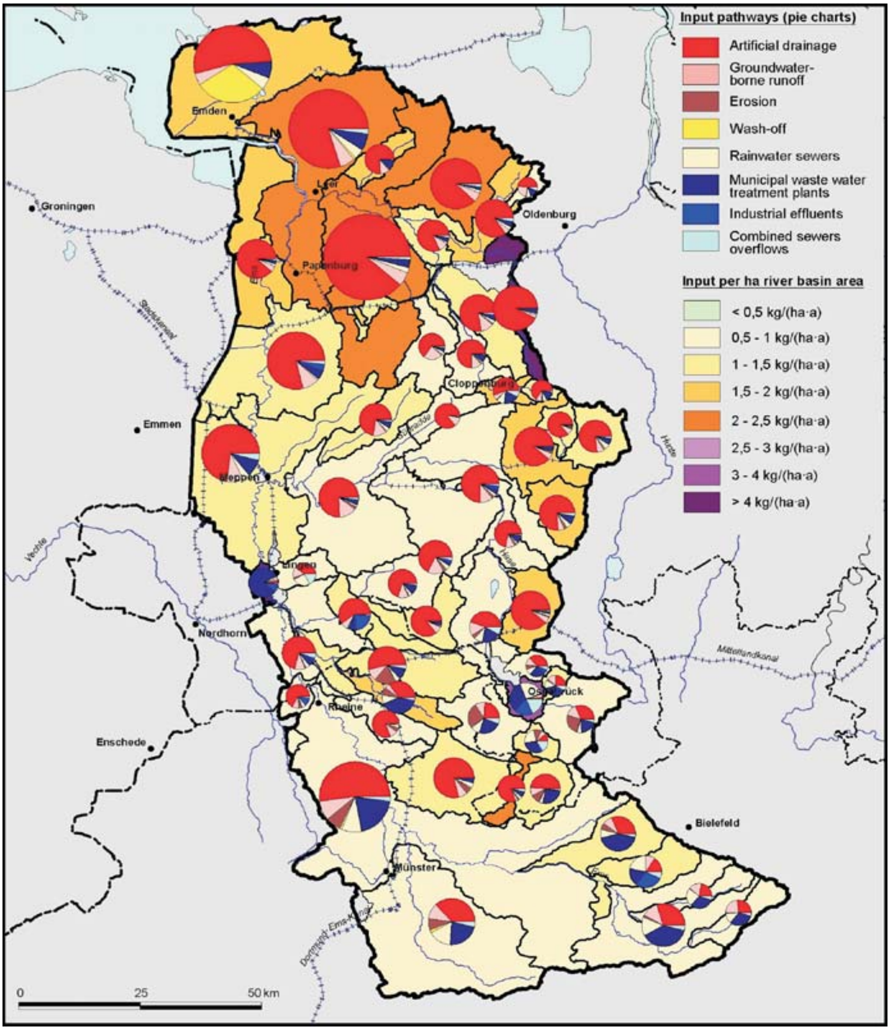


Fig. 5.24. Mean annual P-inputs (1995–1999) of sub-basins and relevance of pathways for the river Ems basin

mendous local and regional eutrophication potential within surface waters. Furthermore the P-retention during fluvial transport is relatively low because of the short distance between the source areas and the mouth of the river Ems. As a consequence, also the wadden sea receives a high phosphate input from diffuse sources.

While the mean total P-inputs (1995–1999) in the investigation area Rhine equal those in the Ems area, the mean relation between diffuse and point source inputs of 32:68 differs significantly. The main reasons are the high densities of population and industry, above all at the river Rhine and along the lower reaches of the river Ruhr.

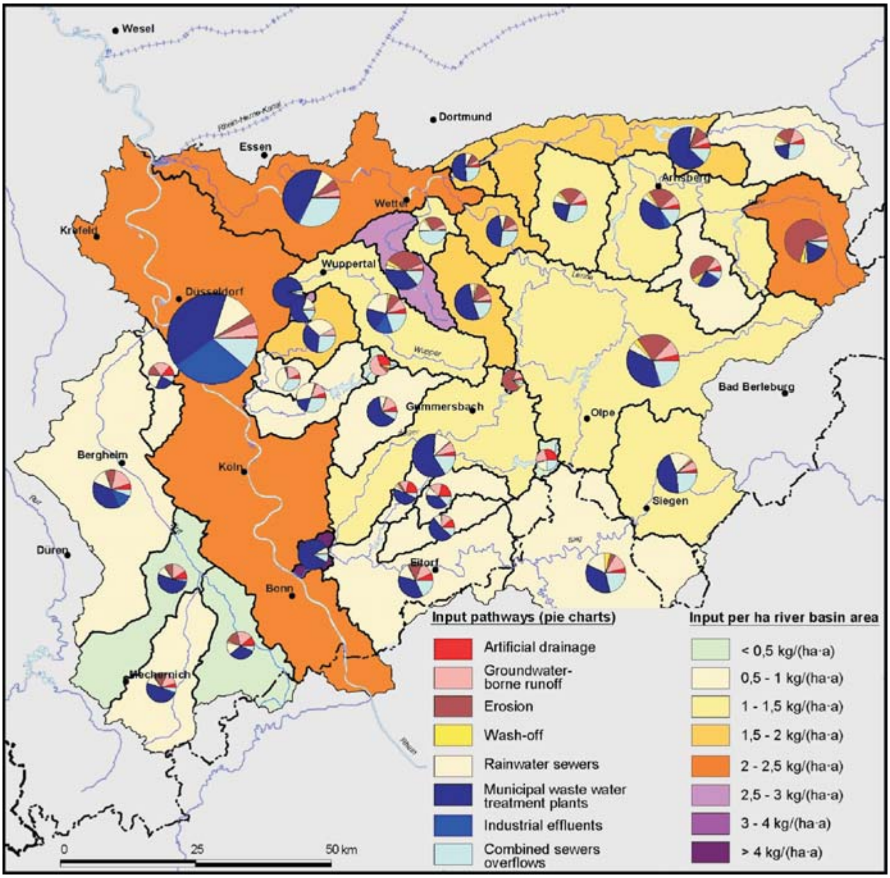


Fig. 5.25. Mean annual P-inputs (1995–1999) of sub-basins and relevance of pathways for the river Rhine sub-basin

High emissions via combined sewer overflows play an important role, too. When the MEPhos modeling results are examined for river sub-basins, areas where diffuse emissions are a major concern are revealed also for the investigation area Rhine (Fig. 5.25). They are located in the upper reaches of the rivers Erft and Ruhr, i.e. in the south-west and east of the investigation area. As a result of arable land use on steeper slopes, partly on loess soils, soil erosion is responsible for these increased diffuse P-loads.

The validity of the MEPhos modeling results is tested by a comparison with mean annual P-loads estimated from measurements of daily discharge and monthly total-P-concentration during the time period 1995–1999. When selecting the gauging stations attention was paid to achieve a great variability with respect to basin size, natural conditions and population density. Furthermore the extent of gaps in the time series of the measured data should be as small as possible. These requirements are fulfilled for 32 resp. 24 gauge-related sub-basins with sizes between approximately 50 and

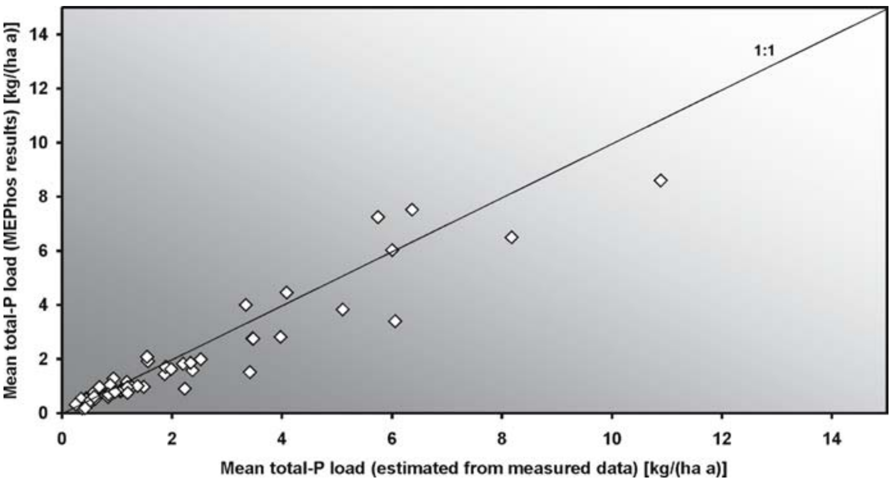


Fig. 5.26. Validation results of modeled mean annual P-inputs (1995–1999)

500 km^2 in the investigation areas Ems and Rhine. Mean annual P-loads for the period 1995–1999 are estimated following the OSPAR method (OSPAR Commission 1998). The general validation process is explained in Sect. 5.4.2 and Fig. 5.22. The validation results are displayed in Fig. 5.26.

The mean annual P-loads differ between ca. 0.1 kg ha^{-1} a^{-1} and ca. 11 kg ha^{-1} a^{-1} (Fig. 5.26). In its entirety the diagram shows a good correlation between gauging data and modeling results. The mean deviation amounts to 7.3%, the coefficient of determination reaches 89%. For 8 sub-basins the deviation is below 10%, for 20 of 56 sub-basins below 20%. Errors in this order of magnitude are within the usual variation range of empirical models. Although the most recent data sets with the highest spatial resolution and information content were used in this project, unavoidable measuring and interpolation errors are undoubtedly involved. Systematic errors causing the deviations could not be identified. Because of the restricted data availability in the field of urban drainage and sanitation, the model routines for the quantification of mean annual P-inputs via rainwater sewers and combined sewer overflows have to get by on a limited number of parameters. With respect to these restrictions the model results can be regarded as valid.

5.4.5 Conclusions and Management Options for Tackling Eutrophication

The MEPhos model in its existing form is suitable for the quantification of mean annual P-loads from diffuse and point sources in large river basins. Based on valid model results management options for the reduction of eutrophication can be proposed. When aiming at reasonable cost-effectiveness-relations the focus of reduction measures has to be on "hot spots", i.e. critical source areas with small spatial extent and high emission. In the river Ems basin the phosphotope "drained raised bog soils under grassland use" takes up less than 4% of the basin area and emits about 30% of all diffuse P-inputs.

Land use changes with respect to nature conservation could be one possibility to lower P-emissions. In the river Rhine sub-basin P-inputs via erosion make up ca. 35% of all diffuse inputs, originating from about 19% of the basin area. Erosion protection measures both on sediment delivering land and along water courses (on-site and off-site) are regarded as suitable management options. But due to the fact that in the river Ruhr basin a series of reservoirs act as sediment sink and due to the dominance of point sources, tackling diffuse sources in the river Rhine sub-basin has only little effect on the general water quality situation. A far larger improvement would be achieved by taking technical measures for small and medium sewage treatment plants, e.g., P-elimination techniques, and by extension of measures against combined sewer overflows.

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