Chapter 2

Managing River Sediments

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2.1 Hydrodynamics and Sustainable Sediment Management

2.1.1 Introduction

Sediments play an important role in river engineering and water resources management. In the past, many rivers in developed countries have been engineered by training and regulation works for navigation, hydropower generation and flood protection. In the past decades, municipal and industrial wastewater discharge and various diffusive sources from agriculture have caused a widespread contamination of river sediments by heavy metals, organic toxicants and agrochemicals. Meanwhile, many historically contaminated sites in rivers are localized and identified as a severe latent hazard for the river ecosystem (see Sect. 1.1.3). Most of the contaminated sites have been detected in low flowing water bodies which are either permanently or temporarily connected to the main river channel such as near bank groyne fields in waterways or harbors, river dead arms, flood plains and last not least flood retention reservoirs (Fig. 2.1). Many deposits are most likely to be resuspended and transported over a long distance by extreme discharges causing contamination of not yet polluted surface water bodies and unpolluted soils subject to flooding.

High discharges in rivers may cause the mobilization of contaminants deposited in such low flowing zones of river channels. The recent flood events in the river Odra in 1997, river Rhine in 1999 and river Elbe in 2002 have illustrated not only the devastating power of floods by damaging hydraulic structures and breaching dams but also the enormous erosion capacity of flowing water associated with the mobilization, transport and partial deposition of contaminated sediments in tidal harbors, estuaries and coastal areas. The precautionary as well as the nondeterioration principle calls for the development and implementation of an integrated sediment management aiming to reduce the risk of contaminated sediment mobilization and their impact on the environment according to the EU water framework directive.

Integrated Risk Assessment

Sustainable sediment management aims to reduce the risk of adverse impact and ecological damage by sediment associated toxicants and to improve the ecological status of surface water bodies. A comprehensive risk assessment, which is an essential contribution to the challenging task, requires an interdisciplinary approach to cope with the interacting physical, chemical and biological processes occurring on extremely differ-

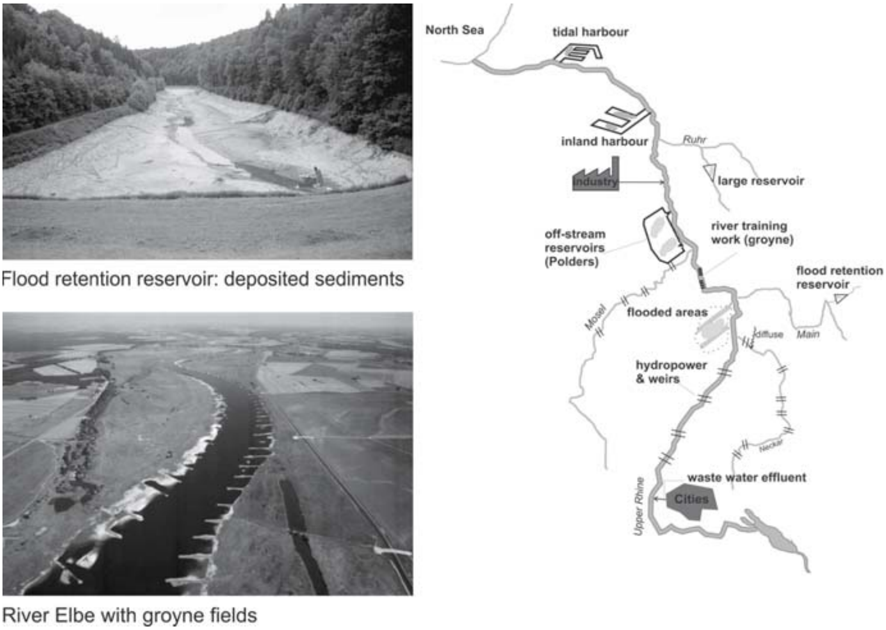

Fig. 2.1. Sources, sinks and pathways of contaminants in a large river basin

ent space and time scale (Kern 1997). Management strategies must include river engineering issues, and environmental problems and economic aspects on the local, regional and river basin scale (Carlon et al. 2000). Contaminated sediments deposited in groyne fields, harbors and water reservoirs can be mobilized in many ways, for instance by floods, maintenance dredging, partial or total emptying of reservoirs, revision or technical inspection of structures. After resuspension, fine sediments are mostly transported over long distances through the whole river system while simultaneously various other processes occur, such as mixing, dilution and loading by tributaries, fractional sedimentation, pollutants repartitioning as well as chemical and biological transformation and degradation.

Beside the quantitative aspects of sediment transport, such as river bed stabilization, habitat improvement, flood protection and navigation, the mobility and transport behavior of sediment bound contaminants and nutrients are emerging key issues of vital importance to future sediment management and surface water quality improvement.

Contaminant immission at a downstream site in terms of concentration and load depends on the catchment characteristics and the hydrological situation such as

- location and connectivity of contaminated sites in the catchment
- actual hydraulic conditions in the river channel network
- in-situ toxicity and total amount of contaminants mobilized upstream.

Floods play a dominant role in sediment erosion risk assessment because of their extreme erosion and transport capacity. Hence, there is a high probability that histori-

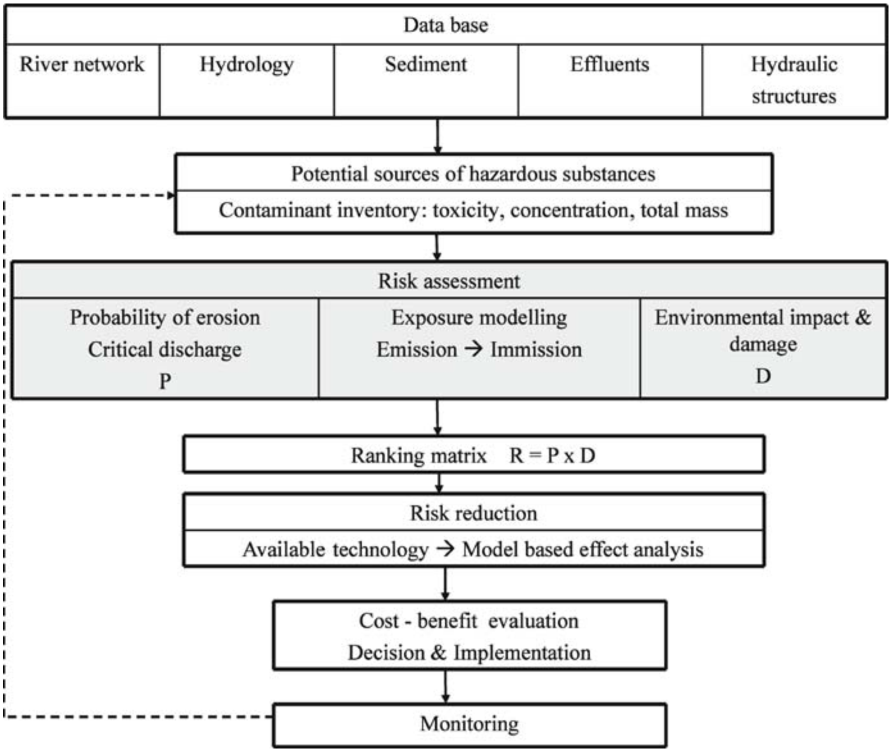

Fig. 2.2. Catchment related integrated sediment management concept

cally contaminated sediments in deeper layers can be resuspended and transported through the river basin to the estuarine and coastal waters. Another key factor is the sediment erosion stability because it controls the mobility and contaminant mass flux and hence the initial conditions for the subsequent transboundary transport process (Fig. 2.2). The site specific relationship between discharge, bed shear stress and sediment mass flux can be completely described by hydraulic modeling. Discharge statistics are directly transformed by hydrodynamics into bed shear stress statistics. Finally, erosion probability results from the convolution of the probability density function of the hydrodynamic bed shear stress and the sediment specific erosion resistance.

After the exploration of polluted sites and their contaminant inventory, a risk analysis must be performed to quantify the risk index *R* as the product of erosion probability and environmental damage or impact (Carlon et al. 2000).

Numerical exposure models are useful tools to describe the pathway and fate of mobilized contaminants and aim to quantify the spatial and temporal distribution of dissolved and particulate substances in the water column and the river bed as well, and to identify sedimentation zones in the river system (Baart et al. 2001).

Numerical modeling allows us to integrate different results and experience from engineering and natural science, and to simulate processes differing by some orders of magnitude both in space and time (Kern 1997). Individual scenario modeling provides data on the intensity and duration of exposure and, with statistical input data, they deliver information on exposure duration and frequency causing accumulation of deposited pollutants which can be used for a statistically based effect model (de Zwart 2005; Öberg and Bergbäck 2005).

The application of a contaminated sediment transport model requires a comprehensive data base including hydrological, morphological and sedimentological data as well as chemical and biological data to cover sorption, transformation and degradation processes. However, the complex effects of biofilms on sorption and biodegradation processes in a riverine environment cannot yet be modeled satisfactory (Flemming, this vol.).

In addition to the hydrological probability aspect, uncertainties in the model concept, model parameters and accuracy of the input data must be considered to account for the uncertainty of the exposure model results. Hence, the uncertainty of calculated immission data and the risk index *R* (Fig. 2.3) of course is significantly affected by the

- model type, deterministic/stochastic approach, dimensionality
- spatial and temporal resolution required, processes implemented
- number of hazardous sites involved
- quality of data about in-situ contaminants
- pathway and tributaries between emission and immission site
- physical and in particular, biochemical processes involved
- data base for model calibration and validation.

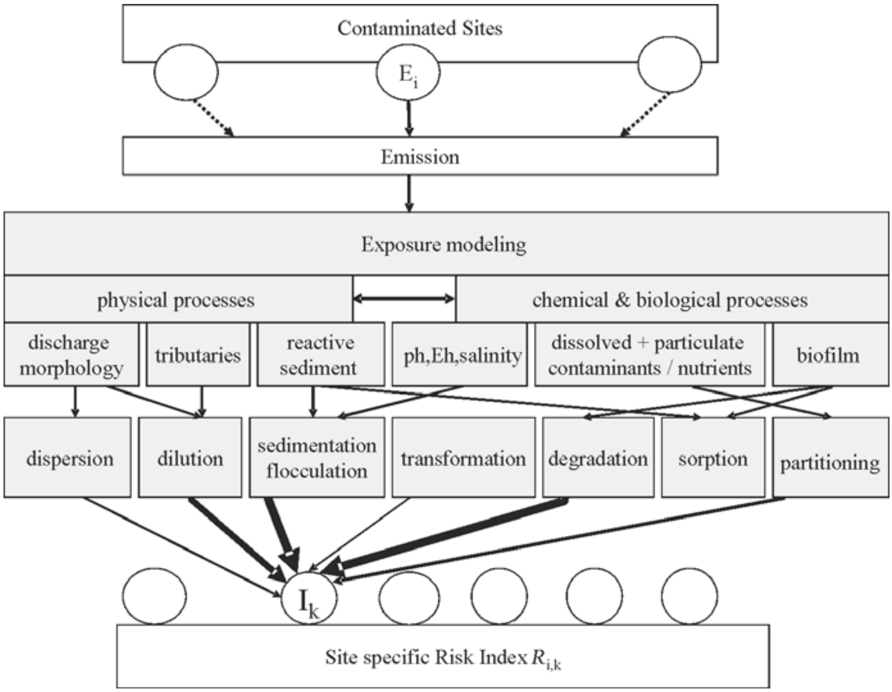

Fig. 2.3. Exposure modeling for site specific risk index evaluation

Based on the numerical results, an emission or immission related site specific risk index $R_{i,k}$ can be obtained by evaluating the impact of each emission site, marked by the index i, to a defined immission site or vise versa, by ranking the immission load of an individually considered site at risk, expressed by the index k. The matrix enables an optional ranking of the damage potential of contaminated sites as well as for receiving water bodies or flooding areas at risk. Both ranking figures provide useful information for remediation planning.

The ecotoxicological aspects of a comprehensive risk analysis can be supported and substantially improved by using effect models which allow us to describe the dose/effect relationship, e.g., by application of Artificial Neural Network (Lek and Guegan 2000) or Fuzzy Logic approach (Ahlf, this vol.). The exposure model can, of course, be used for a risk reduction analysis investigating the effect of alternative remediation measures. A risk based sustainable sediment management strategy must, of course, try to find a source oriented solution instead of an end of pipe solution. Hence, the source related risk index is of first priority. After a cost-benefit evaluation a prioritization of remediation action can be performed as a rational basis for the decision on a cost effective solution for sediment improvement.

Experimental Methods

Because of the great variety of river characteristics, water chemistry, quality and biology it is evident that the sediment stability is very site specific and subject to seasonal variation (Paterson 1997). Therefore, experimental results cannot be transferred directly from one site to another. Because of a lack of a conclusive generic description of cohesive sediment erosion processes, experimental investigations on undisturbed sediments either in the laboratory or the field seem to be indispensable to gain site and river specific data (Lick et al. 1994; Zreik et al. 1998).

Sediment erosion stability controls not only the onset and source strength of particle mass flux but also the flux of the dissolved and colloidal components associated with the pore water. Immediately after erosion larger aggregates are exposed to strong turbulent shear forces disrupting the eroded lumps and generating new reactive surface for the exchange and transfer of adsorbed pollutants (Worch, this vol.). During the hydraulic transport, concentrations, grain size spectrum and most probably the chemical and biological milieu condition will also change and hence reactions and interactions between the particulate and dissolved phase accordingly.

To investigate some key processes with regard to fine sediment mobility and particulate contaminant behavior specific experimental equipment has been developed and applied as follows (Fig. 2.4):

- *SETEG system:* Depth profile of erosion threshold and erosion rate in a pressurized channel, sediment testing area 150 cm^2, bed shear stress up to 25 N m^{-2}, sediment core length up to 150 cm (Witt and Westrich 2003)
- *Differential Turbulence Column:* Concentration profile of different particle fractions, flocculation, desorption and remobilization of sediment bound contaminants, pollutants partitioning (Kühn, this vol.)
- *Mesocosm:* Erosion, sedimentation cycles under tidal like conditions, sorption process under controlled chemical conditions in the water column (Gust, this vol.)

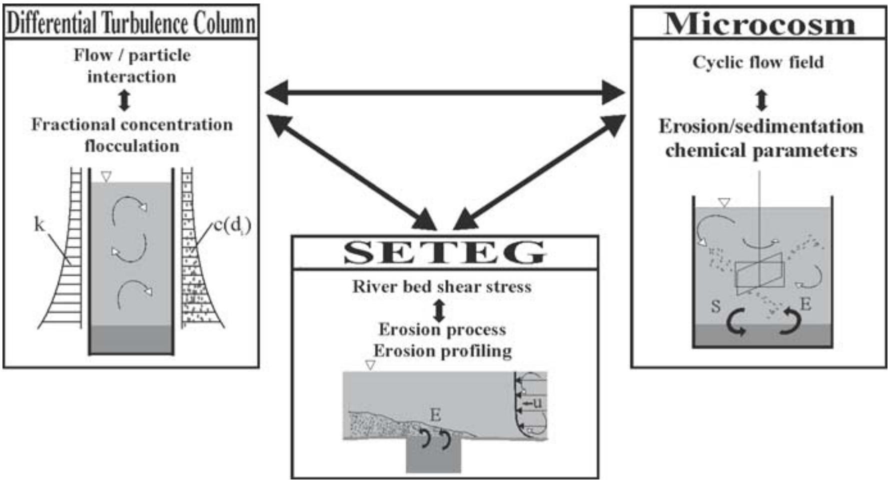

Fig. 2.4. Experimental techniques for investigating contaminated sediment stability and suspended particle/turbulent flow interaction

Complementary sediment stability tests can be performed to quantify scale effects of erosion and hence to facilitate the up-scaling of experimental laboratory data to the field and to compare laboratory data with in-situ measurements (Westrich and Förstner 2005).

- *Box Sampler:* Erosion tests in a flume, sediment testing area 30×70 cm^2, sediment depth 28 cm, maximum bed shear stress 20 N m^{-2}.
- *EROMOB:* Mobile in situ erosion testing equipment; sediment testing area 30×70 cm^2, maximum bed shear stress 10 N m^{-2} (Westrich and Schmid 2004)

Both instruments have a testing area ten times larger than the abovementioned SETEG equipment. The intercomparison of the abovementioned erosion testing methods with the inclusion of other methods like the CSM method (de Deckere et al. 2002) or the in-situ flume (Debnath et al. 2006) has not yet been concluded.

Experimental investigations have been performed using two parallel undisturbed sediment samples from the same spot with the aim of providing sediment depth profiles with a resolution of about 2 cm as follows:

- one sediment sample is used for erosion profiling after physical properties profiling such as grain size, bulk density, water and gas content,
- the other sample is used for chemical and biological parameter profiling.

For practical application it is advisable to restrict the analytical effort on sediment exploration and to model the erosion process by using a limited number of variables representing a significant percentage of the total variance of the sediment parameters.

A comprehensive indepth investigation was conducted to identify and quantify the relevant parameters by multivariate statistical analysis in order to find a relationship between sediment erosion behavior and measurable sediment properties (Gerbersdorf et al. 2005).

Spatial Variability of Sediments

Beside the physical properties, contaminated sediments exhibit a great spatial variability not only in the horizontal but also in the vertical direction which mainly indicates the history of the pollution. The deposition pattern reflects the spatial and temporal variation of the flow field. Fine reactive sediments with a fall velocity of the order of magnitude of some 10^{-4} m s^{-1} can only be found in zones where, for some time, the local bed shear stress was below the critical value of sedimentation to build up a certain sediment layer thickness which was protected by overlaying sediments or could withstand due to its erosion resistance. Because of the variation in discharge, suspended sediment inflow and river water pollution, large gradients of sediment properties can be detected only by a respectively high vertical resolution of a few centimeters. The critical erosion shear stress may change by a factor of 3 within sediment layers of 5 to 10 cm as shown in Fig. 2.5a, 5c and 5d for different sites.

Particulate contaminant profiles are not simply correlated to a single sediment parameter profile. Therefore, mobilization modeling must refer to depth profile data of both the sediment erodibility parameter and the contamination to ensure that the depth dependent contaminant source strength is captured by the model.

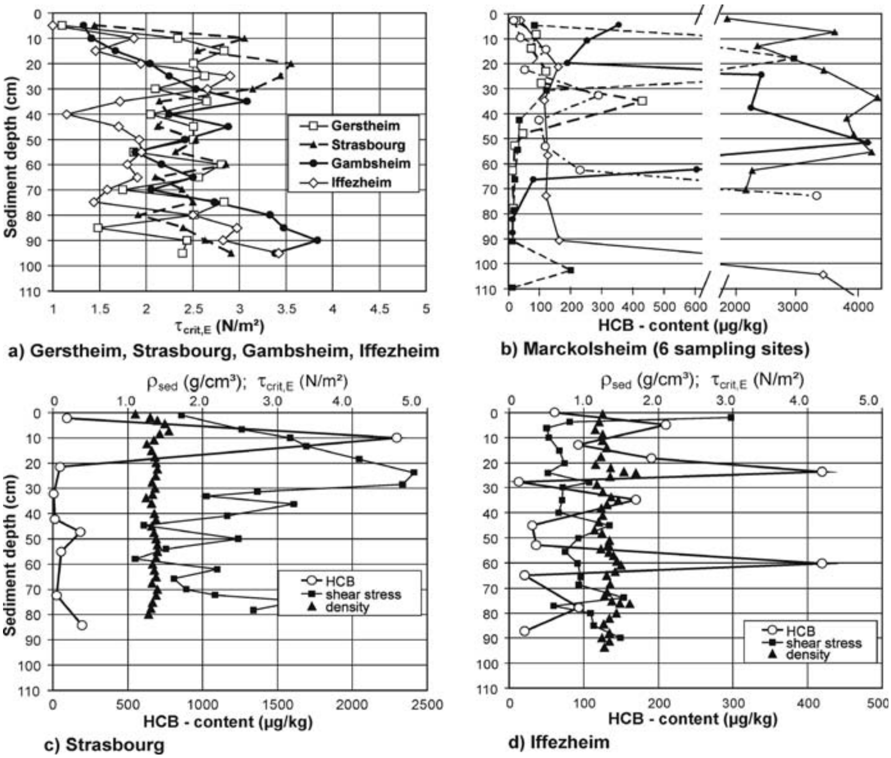

Fig. 2.5. Depth profile of sediment erosion parameters and particulate HCB content of Upper Rhine reservoirs (Westrich and Witt 2004)

With a sufficiently high number and spatial density of sediment samples a geostatistical analysis is advisable in order to improve the reliability of the model input data, to enhance the efficiency of sediment monitoring and to reduce the costs of maintenance dredging (Winkler and Stein 1997). In the case of a poor data base of sediment properties and contaminant inventory, simple interpolation and extrapolation can be applied which, of course, increases the uncertainty of the model output.

Apart from the variability of the critical shear stress the sediment samples from Marckolsheim, Strasbourg an Iffezheim (Fig. 2.5b, 5c and 5d) are typical for contaminated sites in river reservoirs as they illustrate the high spatial variability of the contamination of organic toxicants, e.g., *Hexachlorobenzene* (HCB). Similar large gradients of grain size, bulk density and gas content were also detected. Depth profiles of neighboring samples always exhibit some small difference, known as the nugget effect, in the semi-variogram (Asselmann 1997), which must be considered when combining erosion and biochemical measurements of two sediment cores. The large micro- and mesoscale heterogeneity of sediment parameters underlines the necessity of a high spatial resolution of samples for reducing the uncertainty of model input data.

Freshly deposited fluffy sediments show a small timeindependent erosion rate of some 10^{-6} kg m^{-2} s^{-1}, whereas consolidated sediments exhibit a linear progression of initial erosion rate caused by enhancing the erosive potential of the local disturbance of turbulent boundary layer (Witt and Westrich 2004). This phenomenon illustrates the difficulty of defining the erosion rate from smallscale laboratory experiments and for transferring the data to nature or to numerical models.

2.1.2 Contaminant Transport Modeling

Physically based numerical models have proven to be powerful tools for describing the pathway and fate of contaminants in surface water and hence the relationship between emission and immission (Onishi 1981). Moreover, predictive numerical transport models are used to anticipate the environmental impact of hydrological mobilization scenarios and to analyze the effect of optional remediation measures as well. They provide necessary information for assessing alternative riskreducing measures and estimating their efficiency. The model choice depends on the objective and the requirements in terms of spatial and temporal resolution and accuracy. Calibration and validation of contaminant transport models is a crucial task because of lack of appropriate data, especially on sorption processes and biochemical transformation.

The aim is to quantify the concentration field of dissolved and particulate pollutants in the water body and to describe the areas subject to temporary or permanent contaminant deposition. Focusing on transport in river channels, the hydrodynamic model must be supplied directly by discharge data from gauging stations or by a hydrological catchment model. Deep water bodies like estuaries and tidal harbors very often require 3-D flow and transport modeling (Ditschke, this vol.). In contrast, many transport processes in lowland rivers can be described by depth averaged flow velocities and suspended sediment concentrations with 2-D models. Assuming fully mixed conditions, 1-dimensional advection/dispersion models can be applied to investigate large scale far field transport and long term processes.

Input Data Base

Beside the hydrological, hydraulic and sediment data for the hydrodynamic model part, the sorption parameters of the reactive particle fraction must be available in a data base for the description of the interaction between the aqueous and solid phase. A comprehensive data base must include information on different subjects, such as the following:

- discharge statistics at gauging stations
- river network, channel bathymetry
- digital terrain model of the flood plains
- hydraulic structures, location and operation
- suspended sediment concentration
- grain size/fall velocity spectrum of suspended matter
- fall velocity and sorption parameter of contaminated fraction
- site specific erosion threshold and erosion rate parameters
- sediment and pollutant specific sorption parameter
- biochemical degradation parameter

Model input data are of varying quality with respect to accuracy and density in space and time. Statistical information should be available, such as expected value and statistical variance, especially for chemical and biological parameters, to perform a sensitivity analysis and to facilitate an uncertainty assessment. Consistent field data of extreme events are very poor and, as a result, hard data for contaminant transport model calibration and validation and in particular model prediction are uncertain (Karnahl, this vol.).

Coping with Uncertainties

There are various sources of uncertainties: uncertainty in the model concept, the model parameters and the data itself. The impact of model parameter uncertainties on the prediction of reservoir sediment erosion by floods has been investigated by Li (2004). The boot strap method was applied to gain the mean value and statistical variance of the critical shear stress and the erosion rate from the experimental data. The numerical analysis was performed with a 1-D model (Kern 1997) using the Monte Carlo method. It reveals the predominant influence of the peak discharge and the flood duration. Referring to historical floods it has been shown that with a critical bed shear stress ranging from 3.5 N m^{-2} ±0.5 N m^{-2} with a variance of 0.12 the eroded sediment volume was higher when assuming spatially uncorrelated erosion data.

Sedimentation of fine suspended material is primarily dominated by the mean bed shear stress, fall velocity and concentration of the contaminated fraction. Modeling fractional sedimentation under natural conditions in flood plains is difficult and uncertain because of the influence of roughness elements, like vegetation, on the near bottom turbulence. However, in most cases, large flood plains show low flow velocities and hence are a significant sink of contaminants as experienced by the Elbe flood in August 2002.

2.1.3 Case Study: Upper Rhine

Site Description

The lower six hydropower stations on the Upper Rhine, built in the years 1961 to 1977, show a characteristic sedimentation pattern related to the individual layout of the hydropower channel, the weir channel and the ship lock (Fig. 2.6). A sustainable sediment management must be established to keep the required freeboard of the embankment for safety reasons and, in particular, to reduce the risk of erosion of highly contaminated sediments in deeper layers. The dynamic behavior of fine suspended sediments is very much controlled by the fact that the hydropower capacity is limited and, in the case of a flood, the river discharge is split and the surplus is thereby directed to the weirs which serve as a spillway. The operation rules are as follows:

- The discharge capacity of the hydropower station is $Q_{Turbine} = 1\,400$ m^3 s^{-1} and 1 100 m^3 s^{-1}. Most of time there is no discharge through the weir section except about 15 m^3 s^{-1} for ecological purpose.
- If the river discharge exceeds 1 400 m^3 s^{-1} and 1 100 m^3 s^{-1} the surplus discharge, i.e. the difference between Q_{River} and $Q_{Turbine}$, is directed to the weir channel.
- The headwater at the dam is kept at a constant level.

Study Objective

On the one hand, the main objective was to estimate the future risk of resuspension of historically deposited contaminants, mainly *Hexachlorobenzene* (HCB) and, on the other hand, to perform a retrospective analysis of the erosion and sedimentation of HCB during the last flood in May 1999 during which a substantial amount of particulate HCB was mobilized. The study was conducted using a 2-D numerical flow and transport model (Jacoub, this vol.). Each of the six reservoirs was investigated to estimate the HCB mass eroded and to quantify the cumulative contribution of the respective reservoirs to the total particulate HCB load released to the Lower Rhine and monitored at the German/Dutch border. Unfortunately, the pre-flood data of contaminated sediment zones were scarce. Water samples were taken only in front of the turbine section at the lowest hydropower station Iffezheim (Fig. 2.6) during the flood event. The inflow discharge hydrograph, the suspended sediment outflow concentration and the associated daily HCB load are given and used as boundary conditions (Fig. 2.7).

Since the flow velocities approaching the hydropower section are most of the time high enough, no deposition of fine contaminated sediment fraction takes place whereas, in the headwater of the weir section, the flow velocities are small to allow fractional sedimentation except the short period of erosive flood event. At the beginning of the rising hydrograph, fresh sediments can be deposited in the weir branch but shortly afterwards they are resuspended together with older sediments and flushed through the weirs. After the peak flow, when the flow velocities in the weir branch again become small enough, the inflowing suspended sediments and particulate contaminants start settling and remain deposited until the next flood event. Hence, the flood removes the previously deposited sediment top layer of some 10 cm and causes an in-

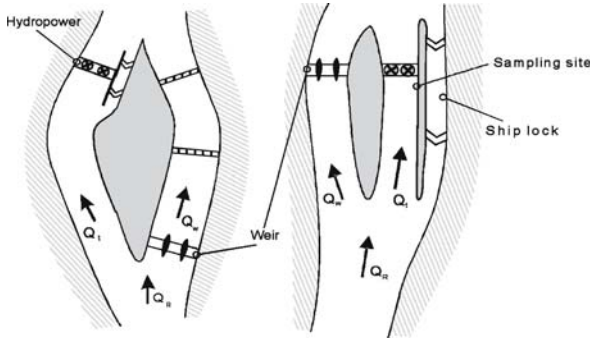

Fig. 2.6.
Layout scheme of two representative Upper Rhine reservoirs: Marckolsheim (*left*) and Iffezheim (*right*)

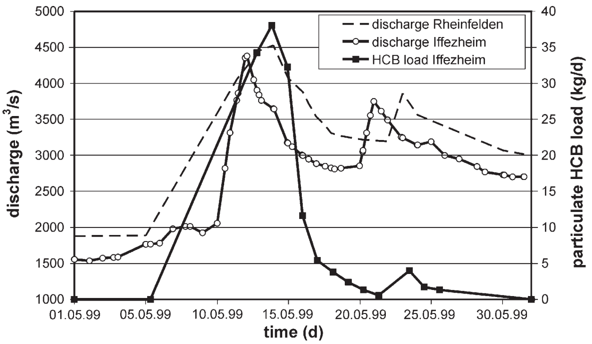

Fig. 2.7.
Discharge hydrograph and daily particulate HCB load during the flood in May 1999 (BfG in Heise et al. 2004)

put of currently mobilized contaminants which are almost completely deposited in the weir branch.

The reservoir Marckolsheim completed in 1961 was selected for an intensive field investigation based on 25 sediment sampling spots to cover an estimated total contaminated area of about 54×10^3 m^2. The spatial sampling density at Iffezheim was 12 samples representing a total area of 35×10^3 m^2 whereas in the Straßburg reservoir only 7 samples were taken of a total contaminated area of an estimated size of 23×10^3 m^2. Simple techniques were applied such as interpolation, total averaging under exclusion of extreme values etc. to assign to each node of the computational mesh the required sediment parameters for critical erosion shear, erosion rate and particulate contaminant concentration. The latter value was averaged over the erosion depth of some 12 cm resulting from the calibrated 2-D flow model. In the Upper Rhine reservoirs, the sediment bulk density varied from 1.1 to 1.7 g cm^{-3}, the critical erosion shear stress between 0.5 to 10 N m^{-2} and erosion rates between 10^{-3} and 10^{-5} kg m^{-2} s^{-1} were measured and used for numerical modeling accordingly (Witt 2004; Jacoub and Westrich 2006).

Despite the extensive field investigation at Marckolsheim, the uncertainty of the calculated eroded particulate HCB was unsurprisingly high. The main reason is the lack of pre-flood data on sediments and the high spatial variability of sediments and contaminants of the post-flood samples. The estimated HCB mass eroded during the flood ranges from 2.4 to 17 kg (Table 2.1). The latter value is far beyond the ICPR (International Commission for the Protection of the river Rhine) target value of 1.3 kg referring to a maximum permissible sediment contamination of 40 μg kg^{-1} for HCB.

Table 2.1. One-year sediment and HCB mass balance of the reservoir Iffezheim from 1 May 1999 to 30 April 2000 (Jacoub 2004)

Particle size (µm)	Input (10^3 t)	Output (10^3 t)	Suspension (10^3 t)	Deposited (10^3 t)	Eroded (10^3 t)	Volume (10^3 m^3)
20	3 674	3 670	1.9	29.5	27.2	1.95
60	1 775	1 689	2.5	103	18.9	69.78
100	593	508.7	2.0	101	18.5	68.83
Total deposited sediment volume:				140 560 m^3		
Averaged annual deposited sediment to be dredged:				120 000–140 000 m^3		
Particulate HCB concentration of 60 µm fraction in the river bed: Total averaged value:				18 µg kg^{-1}		
Averaged value of the weir section only:				290 µm kg^{-1} (220 µg kg^{-1} measured)		

Additional effort was spent on Iffezheim to present a detailed diagnosis of the transport dynamics of different suspended sediment fraction during low flow periods and in particular, during the flood from 5 to 30 May 1999 (Fig. 2.7). During low flow period ($Q_{Rhine} < 1\,500$ m^3 s^{-1}), no inflow of particulate HCB was measured; however, fine suspended particles can be transported by lateral dispersion into the left weir branch and deposited at different rates according to the fractional fall velocity as shown in Fig. 2.8 for the grain size: 20 µm, 60 µm and 100 µm (SS20, SS60 and SS100). If the partial discharge through the left branch exceeds about 1 500 m^3 s^{-1}, which corresponds to a total river discharge of about 2 900 m^3 s^{-1}, erosion starts reaching a maximum at the flood peak as depicted in Fig. 2.8 (lower right) and decreasing with falling hydrograph. The numerical results reveal that the inflowing particulate HCB which was assumed to be associated with the 60 µm sediment fraction starts settling after the beginning of the closure of the weir.

Even though the HCB-concentration of the riverbed before the flood was unknown and assumed to be zero, the numerical model results show good agreement when comparing the calculated value of 220 µg kg^{-1} with the HCB contamination of 290 µg kg^{-1} measured in the years 2001 to 2003 and averaged over the erodible top layer of about 10 cm.

According to the numerical results of the individual reservoirs investigated the total mass of HCB mobilized during the flood in all six reservoirs amounts to some 61 kg, which must be considered an underestimation because the computation was performed with the erosion depth averaged contamination measured after the flood as mentioned above (Table 2.2). It is also evident that the measured value of 145 kg HCB must be considered too low because the sampling site in front of the turbine section on the right hand side (Fig. 2.6) could not capture the sediments eroded in the weir channel. It rather represents the fractional load of HCB through the hydropower branch at Iffezheim.

The results of the post-flood retrospective study on the transport dynamics of particulate HCB, in conjunction with the volume of sediments deposited in the subsequent 12 months, provide useful information for future sediment management and contaminant mobilization risk assessment.

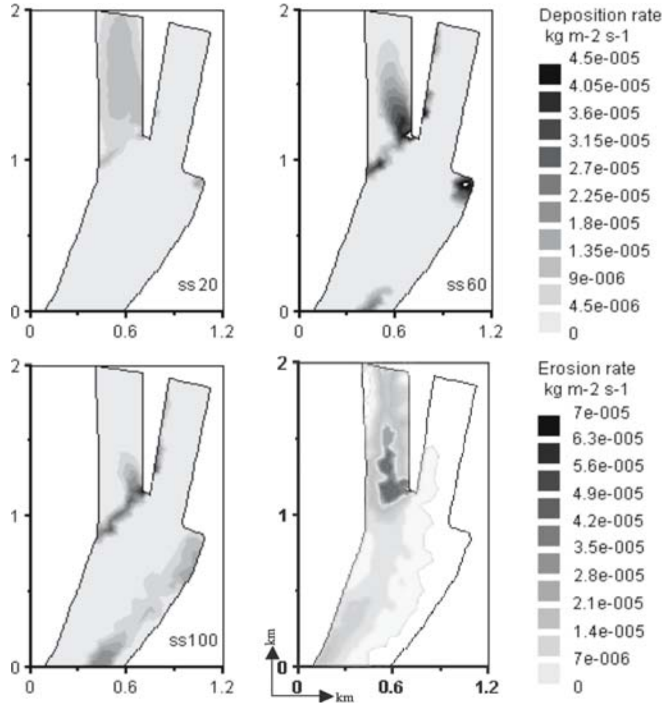

Fig. 2.8. Numerical results of the spatial distribution of the deposition rates of three grain size fractions at low river discharge ($Q = 1\,500\ m^3\,s^{-1}$) and erosion rates (*lower right*) at flood peak ($Q = 4\,250\ m^3\,s^{-1}$) for the Iffezheim reservoir (Jacoub 2004)

Table 2.2. Calculated total HCB mass eroded in the Upper Rhine reservoirs during the flood in May 1999 (Westrich and Witt 2004)

Hydropower station/reservoir	Contaminated area ($10^3\ m^2$)	Particulate HCB (kg)	Specific HCB mass content ($g\ m^{-2}$)	Uncertainty assessment
Marckolsheim	54	5	0.11	Underestimated
Rheinau	13	10	0.08	Estimated*
Gerstheim	178	14	0.08	Plausible
Strasbourg	230	23	0.10	Plausible
Gambsheim	68	6	0.09	Highly underestimated
Iffezheim	36	3	0.08	Highly underestimated
Total:		61 (145 measured)		Highly underestimated

The total amount of deposited sediments from 5 to 30 May 1999 was calculated to be 140 000 m^3 which is close to the averaged annual volume necessary for re-establishing the original channel geometry by maintenance dredging. The good agreement of the deposited sediment volume also confirms the computational results of the contaminated mass budget and hence the applicability of the contaminant transport model.

2.1.4 Conclusions and Outlook

Experimental investigations of physical, chemical and biological parameters must be performed to define erosion process, suspension and sedimentation of fine cohesive sediments and their role in the transfer of dissolved and particulate contaminants. However, the results need to be verified by field measurements.

Numerical models can considerably contribute to risk assessment by describing the pathway and fate of contaminants from the emission to the immission site. Predictive models provide results to be used for the design and analysis of alternative remediation measures and as basic information for further cost effective sediment quality improvement.

References

Asselmann NEM (1997) Suspended sediment in the river Rhine, the impact of climate change on erosion, transport and deposition ISBN 90-6809-254-5 (NGS)

Baart AC, Boon JG, Villars MT (2001) Generic model for contaminants (GEMCO), Z2725, Delft Hydraulics, Delft

Boudreau BP (1997) Diagenetic models and their implementation. Modeling transport and reactions in aquatic sediments, Springer-Verlag, Berlin

Carlon C, Norbiato C, Critto A, Marcomini A, (2000) Risk analysis applied to a contaminated site. Determination of risk based remedial targets. Ann Chim 90(5-6):349–358

Debnath K, Nokora V, Aberle J, Westrich B, Muste M (2006) Erosion of cohesive sediments: resuspension, bed load and erosion patterns from field experiments, to be published by Journal Hydraulic Research, ASCE

De Deckere, EM, van der Waal P, Andree S (2002) A flume study on the effect of Corophium volutator on the sediment stability. J Experimental Marine Biology and Ecology

De Zwart D (2005) Impact of toxicity on species, composition of aquatic communities: Concordance of predictions and field observations, Netherlands National Institue for public helth and the Environment (RIVM)

Formica SJ, Baron JA, Thibodeaux LJ, Valsaraj KT (1988) PCB transport into lake sediments: conceptual model and laboratory simulation. Environmental Sc. Technology, 22:1435–1440

Gerbersdorf S, Jancke T, Westrich B (2004) Physico-chemical and biological sediment properties determining erosion resistance of contaminated riverine sediments – Temporal and vertical pattern at the Lauffen reservoir river Neckar, Germany. Limnologia 5:132–144

Haag I, Westrich B (2001) Erosion investigation and sediment quality measurements for a comprehensive risk assessment of contaminated aquatic sediments. The Science of The Total Environment 266:249–257

Jacoub G (2004) Development of a 2-d numerical code of particulate contaminant transport in impounded rivers and flood retention reservoirs. Institut für Wasserbau, Universität Stuttgart, Mitteilungen Heft 133

Jacoub G, Westrich B (2006) Modelling transport dynamics of contaminated sediments in the headwater of a hydropower plant at the Upper River Rhine. Acta Hydrochimica and Hydrobiologica 33(3)

Kern U (1997) Transport von Schweb – und Schadstoffen in staugeregelten Fließgewässern am Beispiel des Neckars. Institut für Wasserbau, Universität Stuttgart, Mitteilungen, Heft 93, ISBN –3-921694-94-9 (in German)

Lek S, Guegan JF (2000) Artifical Neuronal Networks: application to ecology and evolution. Springer-Verlag

Li, Chien Chen (2004) Deterministisch-stochastisches Berechnungskonzept zur Beurteilung der Auswirkungen erosiver Hochwasserereignisse in Flussstauhaltungen. Universität Stuttgart, Institut für Wasserbau, Mitteilungen Heft 129

Lick W, Lick J, Ziegler CK (1994) The resuspension and transport of fine-grained sediments in Lake Erie. J Great Lakes, vol. 20, pp 599–612
Öberg T, Bergbäck B (2005) A review of probabilistic risk assessment of contaminated land. J Soils and Sediments 5(4):213–224
Onishi Y (1981) Sediment contaminant transport model. American Society of Civil Engineering. Journal of Hydraulic Division 107:1089–1107
Paterson DM (1997) Biological mediation of sediment erodibility: ecology and physical dynamics. In: Burt N, Parker R, Watts J (eds) Cohesive Sediments. Wiley and Sons, pp 215–229
Westrich B, Förstner U (2005) Sediment dynamics and pollutant mobility in rivers (SEDYMO), assessing catchment-wide emission–immission relationship from sediment studies. J Soil and Sediments 5(4):194–200
Westrich B, Schmid G, Entwicklung und Einsatz eines mobilen Gerätes zur in-situ Bestimmung der Erosionsstabilität kontaminierter Feinsedimente. Institut für Wasserbau, Universität Stuttgart, Report TB 2004/05-VA 49
Westrich B, Witt O (2004) Untersuchungen zum Resuspensionsrisiko von Sedimentablagerungen in ausgewählten Staustufen des Rheingebiets. IKSR Report, Institut für Wasserbau, Universität Stuttgart (in German, not published)
Winkler H, Stein A (1997) Optimal sampling for monitoring and dredging contaminated sediments. International Conference on Contaminated Sediments, vol. II, pp 1019–1028
Witt O (2004) Erosionstabilität von Gewässersedimenten mit Auswirkungen auf den Stofftransport bei Hochwasser am Beispiel ausgewählter Staustufen am Oberrhein. Institut für Wasserbau, Universität Stuttgart, Mitteilungen Heft 127, ISBN 3-933761-30-1
Witt O, Westrich B (2003) Quantification of erosion rates for undisturbed cohesive sediment cores by image analysis. Hydrobiologia 494 (1–3):271–276
Zreik DA, Krishnappan G, Germaine JT, Madson OS, Ladd C (1998) Erosional and mechanical Strength of deposited cohesive sediments. Hydraulic Engineering 124:1076–1085

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2.2 Requirement on Sediment Data Quality – Hydrodynamics and Pollutant Mobility in Rivers

2.2.1 Introduction

Three principal media can be used for aquatic monitoring: water, particulate matter and living organisms. With respect to particulate matter, characteristics have been noted such as: *(1)* good specificity to a given pollutant, *(2)* high sensitivity to low levels of pollution, *(3)* medium to low sample contamination risk, *(4)* short (suspended matter) or long to very long (deposited sediment) time span, respectively of information obtained.

The objectives of an assessment program for particulate matter quality can be (Thomas and Meybeck 1992):

- to assess the *present concentrations* of substances including pollutants found in the particulate matter and their variations in time and in space (*basic surveys*), particularly when pollution cannot be accurately and definitely shown from water analysis;
- to estimate *past pollution levels* and events (e.g., for the last 100 years) from the analysis of deposited sediments (*environmental archive*);
- to determine the direct or potential *bioavailability* of substances or pollutants during the transport of particulate matter through rivers and reservoirs (*bioavailability assessment*);

- to determine the *fluxes* of substances and pollutants to major water bodies (i.e., regional seas, oceans) (*flux monitoring*); and
- to establish the *trends* in concentrations and fluxes of substances and pollutants (*trend monitoring*).

The use of particulate matter as an assessment medium has several advantages, at least compared to the water phase, mainly due to the high sensitivity to low levels of pollution and the medium to low sample contamination risk. However, considering the complex system of a large river basin, a closer look is necessary both with respect to state-of-the-art of quality control and quality assurance in these water quality assessment procedures (Sect. 2.2.3) and specifically to quality requirements in relation to hydraulic sediment data (Sect. 2.2.4).

The European Water Framework Directive (WFD) monitoring objectives require compliance checking with Environmental Quality Standards (EQS) but also the progressive reduction of pollution (Sect. 1.1.3). The no-deterioration clause implies that trend studies should be foreseen for sediment and biota. However, *compliance monitoring for sediment* is not yet appropriate because of lack of the definition of valid Environmental Quality Standards (EQSediment) in a European context, analytical limitations and anticipated costs involved to obtain full spatial coverage (Anon 2004a, 2006).

Sediment *trend monitoring* may be both spatial and temporal, and may be related to the chemical and ecological status of a water body. Sediment monitoring may also play a part in *risk-assessment* (see Sect. 10.1).

In principle, it has been recognized that harmonization of sediment monitoring is particularly relevant at a river basin level. In particular, technical issues such as sediment collection, sample treatment, sediment analysis and reporting results will have to follow a common level of quality requirements. Major problem areas have been identified and discussed by the European thematic framework "Metropolis" (Metrology in Support of Precautionary Sciences and Sustainable Development Policies; Anon. 2004b), and comprise lack of representativeness, a high level of uncertainty, lack of metadata, and lack of traceability.

The concept of traceability (Quevauviller 2002) implies that measurement data are *(1)* linked to stated references *(2)* through an unbroken chain of comparison, *(3)* all with stated uncertainties. In the following the implications of the traceability concept for the quality control of sediment analysis will be demonstrated with special reference to the sampling of sediments (Sect. 2.2.2) and the combination of pollutant mobility data (Sect. 2.2.3) and hydrodynamic information (Sect. 2.2.4) at the river basin scale. In Sect. 10.1 "Quality assurance of ecotoxicological sediment analysis" these findings were extended on the influences arising from biological factors.

2.2.2 Sediment Sampling

Natural sediment formed during weathering processes may be modified quite markedly during transportation and deposition by chemicals of anthropogenic origin. Firstly, it must be noted that anthropogenic chemicals may be scavenged by fine sediment particles at any point from their origin to the final sink or their deposition. Secondly, to

Table 2.3. Development of particulate matter quality assessment in rivers in relation to increasing levels of monitoring sophistication (after Thomas and Meybeck 1992)

	Monitoring level		
	A	B	C
Suspended matter (SPM)	Survey of SPM quantity throughout flood stage (mostly when rising)	Survey of SPM quality at high flow (filtration or concentration)	Full cover of SPM quality throughout flood stage
Deposited sediment	Grab sample at station (end of low flow period)	Longitudinal profiles of grab samples (end of low flow period)	Cores at selected sites where continuous sedimentation may have occurred

Level A: Simple monitoring, no requirement for special field and laboratory equipment.
Level B: More advanced monitoring requiring special equipment and more manpower.
Level C: Specialised monitoring which can only be undertaken by fully trained and equipped teams of personal.

compute a geochemical mass balance for sediment-associated elements, it is imperative to derive, by measurement, a mass balance for the sediment in the system under evaluation.

To establish background levels of particulate matter composition, samples of bottom sediment should be taken in the upper reaches of the river basin. The effects of tributaries on the main river should be covered by sampling tributaries close to their junction with the main river. In practice different levels of monitoring sophistication can be distinguished (Table 2.3).

Study of Dated Sediment Cores

The study of dated sediment cores has proven particularly useful as it provides a historical record of the various influences on the aquatic system by indicating both the natural background levels and the man-induced accumulation of elements over an extended period of time. Marine and, in particular, lacustrine environments have the ideal conditions necessary for the incorporation and permanent fixing of metals and organic pollutants in sediments: reducing (anoxic) and non-turbulent environments, steady deposition, and the presence of suitable, fine-grained mineral particles for pollutant fixation. Various approaches to the dating of sedimentary profiles have been used but the isotopic techniques, using ^{210}Pb, ^{137}Cs and $^{239+240}Pu$, have produced the more unambiguous results and therefore have been the most successful (see review on "Historical Monitoring" by Alderton 1985).

Sampling and Filtration of Suspended Matter

Suspended-sediment sampler fall into three general categories (Anonymous 1982; Ongley and Blachford 1982; Horowitz 1991): *(i)* integrating samplers that accumulate a water-sediment mixture over time, *(ii)* instantaneous samplers that trap a volume of whole water by sealing the ends of a flow-through chamber, and *(iii)* pumping samplers that collect a whole-water sample by pump action. Integrating samplers usually are preferred because they appear to obtain the most representative fluvial cross-sectional samples. Cross-sectional spatial and temporal variations in suspended sediment and associated trace elements and their causes are discussed by Horowitz (1991).

Filtration may be carried out under positive pressure or vacuum; excessive pressure or vacuum should be avoided because this may cause rupture of algal cells and release of their intracellular contents into the filtered sample (Hunt and Wilson 1986). Filters having different structures, pore sizes, and composition are available (Brock 1983); the effective pore size of depth filters – having a complex system of channels – changes as the filter becomes more loaded with particles, whereas the effective pore size of screen filters is not affected by filter loading (Apte et al. 2002). Filtration and ultrafiltration can be used for size fractionation of aquatic particles, colloids, and macromolecules (Buffle et al. 1992).

Uncertainties. Handling of suspended sediments includes medium to high contamination risk, similar to the sampling and processing of water samples. Beside problems with filtration techniques (see above), it is important to minimize the time between sample collection and filtration because adsorption/desorption reactions involving particulates and bacterial activity can lead to changes in sample composition.

Handling, Preparation and Storage of Sediment Samples

A review of Mudroch and Azcue (1995) covers the major operations such as *(i)* measurement of pH and Eh (including a detailed description of equipment and solutions used in the measurements), *(ii)* subsampling for determination of cation exchange capacity, *(iii)* subsampling under oxygen-free atmosphere, *(iv)* sample mixing and subsampling into prepared containers and *(v)* sampling hazardous sediments and safety requirements.

Wet Samples

A general scheme for handling samples for tests and analyses on wet sediments is presented in Fig. 2.9 (Mudroch and Bourbonniere 1994). Samples for determination of *particle size-distribution* should not be frozen but stored at 4 °C. Tightly sealed plastic bags, glass jars, or other containers can be used to store samples prior to particle size analyses. Sediments with a high iron content should be stored in air-tight containers to avoid precipitation of iron oxides on particle surfaces and should be analyzed as soon as possible after collection.

Sediment samples for *geotechnical studies* can be stored at 4 °C in a humidity-controlled room, without any large changes in sediment properties for several months. Long cores, such as those collected by piston coring, can be cut into lengths suitable for storage, wrapped to preserve their original consistency, and stored in a refrigerated room.

Freezing has long been an acceptable preservation method for sediments collected for the determination of organic and inorganic constituents. It has been shown that rapid and deep-freezing can best maintain sample integrity and thus enable investigation for concentrations of contaminants. The lower the temperature of deep-freezing the better: a temperature of −80 °C is the suggested maximum.

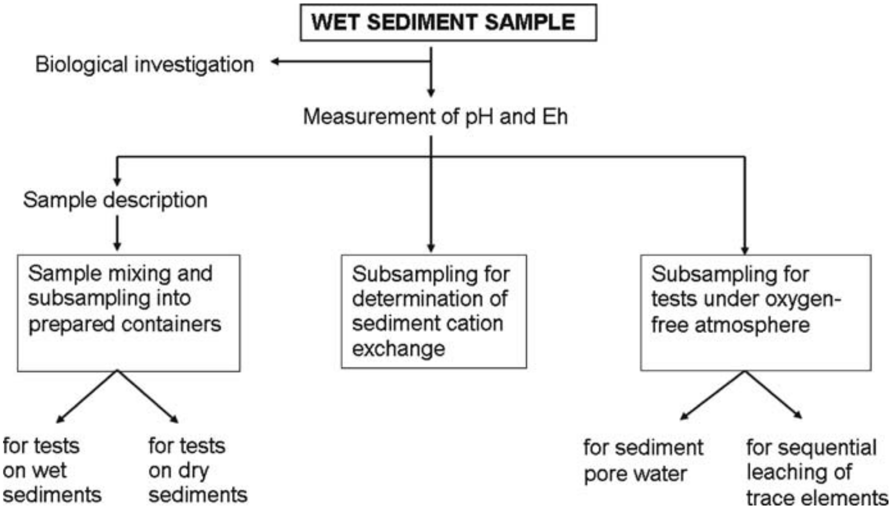


Fig. 2.9. Handling samples for tests and analyses on wet sediments (after Mudroch and Bourbonniere 1994)

Samples collected for investigations of *benthic organisms* are usually processed in the field by wet sieving through different size sieves. If, for any reason, the samples cannot be processed in the field, they should be stored at 4 °C in the dark and processed in the laboratory as soon as possible.

Dry Sediment Sample Preparation

Handling operations of dry sediments include drying, sieving, grinding, mixing, and homogenization. Three types of drying are commonly used to prepare solid samples prior to analysis (Mudroch and Bourbonniere 1994):

- *Air-drying* is rarely used for the preparation of sediments for pollution studies, since it may generate undesirable changes in sediment properties. For example, changes in metal availability and complexation were shown for samples that were air-dried.
- *Oven-drying* of sediments is usually carried out on samples collected for the determination of inorganic components, such as major and trace elements. Oven-drying is not acceptable for sediments which contain any volatile or oxidizable components, whether they be organic or inorganic, and may contribute to the alteration of even non-volatile organics.
- *Freeze-drying* can be used for drying sediments collected for the determination of most organic pollutants as well as for analyses of inorganic components, such as the major and trace elements. The principal advantages of freeze-drying for sediments are *(i)* low temperatures avoid chemical changes in labile components, *(ii)* loss of volatile constituents, including certain organic compounds, is minimized, *(iii)* most particles of dried sediments remain dispersed, *(iv)* aggregation of the particles is minimized, *(v)* sterility is maintained, and *(vi)* oxidation of various minerals or organic compounds is minimized or eliminated.

Anoxic Sediment Treatment

Anoxic sediment samples require different sampling preservation techniques such as oxygen exclusion. Drying and freezing (also freeze-drying) of the samples should be avoided for material designated for extraction procedures. If total analyses or strong acid digestion is planned, the sediment is dried at 60 °C, crushed and stored; for mass calculations, reweighing after drying at 105 °C may become necessary. For a more differentiated approach, in particular for solid speciation studies on anaerobic samples, the following pretreatment scheme was developed (Kersten and Förstner 1987):

- Samples were taken immediately from the center of the material (collected with a grab or corer) with a polyethylene spoon, filled into a polyethylene bottle up to the surface.
- Immediately after arriving at the laboratory, sediments were inserted into a glove box prepared with an inert argon atmosphere. Oxygen-free conditions in the glove box were maintained by purging continuously with argon under slight positive pressure.
- Extractants were deaerated prior to the treatment procedure.

Quality Control

Containers and other equipment used in handling sediment samples after retrieval can be a significant source of contamination (Mudroch and Azcue 1995). For example, plastics contain plasticizers that can be potential contaminants in the determination of organic compounds. Glass, porcelain, stainless steel, Teflon, or Teflon-coated instruments should be used in handling sediment samples to be analyzed for organic components. Wide-mouth amber or clear glass jars and bottles with aluminum foil or Teflon-lined caps are the best containers, but certain compounds (e.g., phenols) can adsorb to these surfaces. Metal containers, spoons, or other equipment may contaminate samples that will be analyzed for metals and trace elements. If both organic and metal analysis are required for a given sediment sample, a Teflon container is recommended.

Since standard sampling and preparation techniques are not available for sediments, results from sediment analyses and in particular their application for sediment quality criteria (SQC), depend in a special way from a high level of quality control (QC) and quality assurance (QA) both in field and laboratory (Keith 1991). QC in *planning* includes choice of *(i)* sampling locations, *(ii)* sampling procedures, and *(iii)* material; quality control in *field sampling* covers *(i)* sample collection, *(ii)* sample handling, *(iii)* cleaning procedures, *(iv)* transport, *(v)* preservation, and *(vi)* storage.

Two techniques can be used for QC in sediment sampling (Mudroch and Azcue 1995):

1. Collection of more than one sediment sample at selected sampling sites using identical sampling equipment, such as multicorers, as well as using identical field subsampling procedures, handling and storage of the samples, and methods for sediment analyses.
2. Subdivision of the collected sample into a few subsamples and treatment of each subsample as an individual sample. The results of chemical analyses of all subsamples indicate the variability due to the sampling and analytical techniques and sediment heterogeneity within a single collected sample.

The control samples used in sediment studies include sampling, transport, sampling equipment, etc., and control samples for laboratory procedures. Contrary to water sampling, sediment sampling generally does not require the use of blanks.

2.2.3 Traceability in Chemical Sediment Analysis[1]

Chemical analyses on sediments, including suspended particulate matter and porewater, are efficient tools in water-quality management (surveillance, survey, monitoring); in this context they refer – in order of increasing complexity – to different objectives, such as preliminary site characterization, identification of chemical anomalies, establishment of references and identification of time changes (chemical, biological), calculation of mass balances, and process studies (Golterman et al. 1983). Chemical analyses are also used to directly characterize contaminated in situ sediments and dredged materials in relation to various treatment techniques.

In the view of the traceability concept, a 'basic sequence' of measurements consists of three steps, which can be considered as an unbroken chain of comparisons (Sect. 2.2.1):

1. *Sampling and sample preparation.* Project planning, sampling stations, sampling devices, handling and storage, and quality control are not standardized, but well-documented in all aspects (Mudroch and Azcue 1995).
2. *Grain size as a characteristic sediment feature.* Sampling on fine-grained sediment (Horowitz 1991) and grain size normalization with 'conservative elements' such as Cs, Sc, Li and Al (reflecting clayey material content) is recommended as standard approach (Förstner 1989).
3. *Analytical procedures.* Reference sediment materials are commercially available. While direct species analysis is still limited, standardized extraction schemes for metals and phosphorus in sediments as well as certified reference materials for comparisons were developed under the auspices of BCR/IRMM.

Further steps in chemical sediment analysis are split up with regard to specific purposes – sediment quality assessment including biological effects (see Sect. 10.1), coupling of sediment quality data with erosion risk evaluation (*(4)* and *(5)* below); chemical changes following resuspension of anoxic sediments (*(6)* and *(7)* below); and modeling of chemical sediment data (*(8)* below).

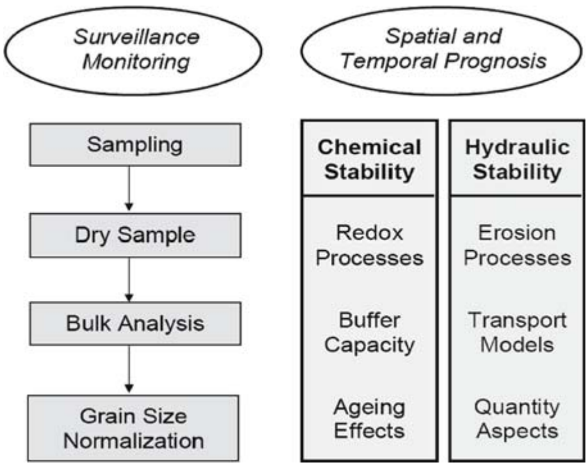
Due to the particular dynamics of fluvial processes, hydraulic parameters such as the critical shear stress of erosion processes form the primary input factors for investigating and predicting large-scale dispersion of contaminants in flood-plains, dike foreshores and polder areas. Unlike problems related to conventional polluted sites, the risks here are primarily connected with the depositing of contaminated solids on soils in downstream regions. Short-term issues include the fate of sediment associated contaminants when sediment is deposited upland and a better understand-

[1] Based on Förstner U (2004) Trends in Analytical Chemistry 23:217–236.

ing of the impact on ground water, water and soil ecosystems. Medium/long-term issues will focus on integration of quality and quantity aspects, and to determine the sediment transport processes at the river basin scale as a function of land and water use and hydrological (climate) change (see Sect. 1.1). A schematic view of the combined assessment of chemical and hydrodynamic effects in river sediments is presented in Fig. 2.10.

4. *Erosion effects.* Sediment physical parameters and techniques form the basis of any risk assessment in this field. Sampling of flood-plain soils and sediments is affected by strong granulometric and compositional heterogeneities arising from the wide spectrum of flow velocities at which the sediments were eroded, transported and deposited. Standardized fractionation schemes and respective reference materials can be useful for studying ecotoxicological aspects of resuspended sediments. Sediment quality issues should include experimental designs for the study of chemical and biological effects during erosion and deposition (Kosian et al. 1999). Coupling of erosion experiments with investigations on aging effects as well as on the mobilization of chemicals from porewater and labile sediment phases (below) could provide a valuable tool in the decision-making process for remediation techniques.
5. *Aging effects.* 'Diagenetic' effects, which apart from chemical processes (sorption, precipitation, occlusion, incorporation in reservoir minerals and other geosorbents such as char, soot and ashes) involve an enhanced mechanical consolidation of soil and sediment components by compaction, loss of water and mineral precipitations in the pore space, may induce a quite essential reduction of the reactivity of solid matrices. The methodologies developed so far (Corbisier et al. 1999; Thompson et al. 1999; Verbruggen et al. 2000; Zhang et al. 2001) could influence the traceability aspects in the field of ecological/chemical risk assessment (Sect. 10.1) and in relation to erosion stability/pollutant mobility both in situ and river basin wide, and these informations will also affect the decision-making process for remediation techniques (Sect. 1.1.3).

Fig. 2.10.
Assessment of combined chemical and hydrodynamic effects



The dramatic effects of stormwater events on particle transport can coincide with rapid and far-reaching chemical changes, in particular, by the effects of sulfide oxidation on the mobilization of toxic metals. The objectives of this research fall under the category 'process studies', usually involving a relative high degree of complexity (Sect. 1.2). Such field and laboratory studies as well as the models using these data are indispensable for long-term prognosis of erosion and chemical mobilization risks arising from subaqueous deposition and capping, both favorable technologies for dredged material and in situ sediments (Förstner 2003).

6. *Anoxic/oxidized samples.* Changes of the forms of major, minor and trace constituents cannot be excluded, when the sediment is transferred from its typical anoxic environment to chemical analysis via normal sample preparation. On the other hand a comparison of extraction data from the original and oxidized samples could be used for worst-case considerations in respect to potential metal release during sediment resuspension or subsequent to up-land deposition of dredged material.
7. *Capacity controlling properties.* Both pH and redox potential in sediment/water systems are significant parameters for mobilization and transformation of metals or phosphorus. Criteria for prognosis of the middle- and long-term behavior of these and other substances should, therefore, include the abilities of sediment matrices for producing acidity and for neutralizing such acid constituents (Sect. 6.5).
8. *Modeling.* The data of critical trace metals and matrix components, as determined from original samples, can also be used in models and in this way, sequential extractions can serve as effective conformational tools to reduce the complexity of the natural system (Wallmann et al. 1993). Pore water analytical data can be applied in geochemical models for short-, medium- and long-term predictions (Parkhurst and Postma 1999). Transport and reaction models consider advective, dispersive and diffusive transport mechanisms as well as ad- and desorption processes (Landenberger 1998).

In total, the traceability of 'further steps' (4–8 above) is less pronounced than that of the three steps (1–3 above) of the 'basic sequence'. However, in the light of the economic value of these analyses for developing and executing far-reaching management plan, coordinated efforts should be undertaken to improve this situation. Short-term measures should range from organized propagation of results from on-going research ('aging effects'), official documentation of techniques and instruments in a relative new field ('erosion effects') and state-of-the-art procedures ('modeling', e.g., analytical data from pore water), via extension of standardized extraction schemes and reference materials (prescription for handling 'anoxic sediments' for fractionation, certification of specific constituents like Ca, S and Fe(II) for the study of 'capacity controlling properties'), up to the development of new reference materials ('pore water'). With regard to the quantity aspect of contaminated sediments in river basin scale, chemical inventories of interim deposits like mining residues, river bank, polder and flood plain deposits, fillings of river-dams and lock-reservoirs, should be given high priority.

2.2.4 Hydraulic Data Quality

Hydrological, hydraulic and sediment data are usually collected from different sources, such as water authorities, institutes and agencies. In many cases, the original measuring data are not accessible; they have already been processed and aggregated so to present them as daily, monthly or annually averaged values in terms of discharge, concentration, load etc. Very often, they are communicated without any technical specification of the sampling site, sampling technique, data-processing method and uncertainty assessment. Little is known about the representativity of the measuring points in flow cross-sections and only sparse information is given on the sampling frequency which is especially necessary for flood events. Gauging stations must provide calibrated discharge rating curves, which normally do not cover the upper range of extreme discharges with overbank flow.

With the aim of quantifying transport rates, flow velocities in the cross-section and the respective fractional concentrations of contaminated sediment must be measured along vertical transects to capture the entire water depth and to allow the calculation of the total transport rate in a given cross-section as shown by the following relationship

$$Q_s = \int_A \int u \cdot c \cdot \mathrm{d}A \tag{2.1}$$

$$Q_s = Q \cdot C^A + \int_A \int u'' \cdot c'' \cdot \mathrm{d}A \tag{2.2}$$

$$u'' = U^A - u \tag{2.2i}$$

$$c'' = C^A - c \tag{2.2ii}$$

$$Q_s = \alpha \cdot Q \cdot C_1 \tag{2.3}$$

with the discharge Q in $m^3\,s^{-1}$, the total suspended sediment transport rate Q_s in $kg\,s^{-1}$, the total flow cross-section A, the cross-section averaged suspended sediment concentration C^A, the cross-section averaged flow velocity U^A which equals Q/A, the local flow velocity u in $m\,s^{-1}$ and the respective suspended sediment concentration c in $kg\,m^{-3}$. When only the cross-section averaged data Q and C^A is referred to, a systematic error is obtained where the second term in Eq. 2.2 which represents the differential advection is neglected. This term can be determined only by extensive measurements over the whole cross-section of the flow. The differential advection term vanishes only if there is a uniform flow velocity or a homogeneous concentration of suspended matter, i.e. if u'' or c'' equals zero. In the case of a fixed monitoring station the measured concentration C_1 must be weighted by a site-specific and time-dependent factor α to account for the non-homogeneous distribution of the suspended sediments.

The sampling frequency must be adjusted to the discharge in order to provide an appropriate temporal resolution of the process and to account properly for the discharge as a key parameter for the evaluation of the transport rate and mass balance. The evaluation of single flood events often requires a sampling frequency of the order of magnitude of hours. The spatial density of sampling points depends on the local gradients of both the flow field and the contaminated suspended sediment concentration.

The weakest link in the information chain seems to be the sediment. There are only a few data available on the grain size of suspended sediment, on fall-velocity distribution and sediment erosion stability. Various instruments have been developed and applied for in-situ or on-site particle fall-velocity measurements. Advanced experimental techniques including optical methods with digital image processing are presented and discussed by Eisma et al. (1997). A variety of experimental methods has been developed for cohesive sediment erosion tests. A comprehensive overview of in-situ erosion measurement devices is reported by Cornelisse et al. (1997), the performance of selected erosion devices is described by Gust and Müller (1997). However, a detailed comparison of the various methods is not yet available (see Sect. 2.1).

Vertical profiles of sediment properties and contaminant concentration, including dissolved and colloidal substances in the pore water, must be available to quantify the total mass flux of deep erosion processes. Therefore, the variability in the vertical sediment parameters has to be taken into account, which increases the variance of the computational output. The more processes involved along the river pathway, the higher the variance of the transport quantities involved and the larger the gap between the best- and worst-case assumption for sediment management.

Model Parameter Uncertainty Assessment

Numerous physically based models have been developed to describe the effect of flood events on river morphology and sediment transport, but most of them are deterministic and do not account for the uncertainties involved in the input variables and model parameters. Therefore, it is necessary to apply statistical methods to assess and improve the reliability of model results. In most stochastic approaches, probabilistic distributions of the input variables and model parameters are used for an uncertainty assessment. However, in most cases, the data set is not sufficient to determine the statistics in a conventional way. The integration of the stochastic concept into a deterministic model provides a useful alternative to cope with such uncertainties.

For the risk assessment of contaminated sediment resuspension, various sources of uncertainties must be considered. The most significant contribution to the uncertainty is due to discharge hydrology, which is known as the hydrological risk. Additional uncertainties originate from the imperfection of the model concept and, in particular, from the erosion-related sediment properties including the erosion threshold and erosion rate. Each of the quantities exhibits a specific measuring inaccuracy and shows a high spatial variability in nature. For an environmental impact assessment, the in-situ sediment contamination, sorption, transformation and degradation processes must be described based on chemical and biological parameters, which are subject to significantly higher uncertainties compared to physical parameter uncertainties. Therefore, any quantity calculated at the far downstream end of the contaminant pathway is subject to all the uncertainties included and hence exhibits the cumulative effect of the uncertainties involved (see Fig. 2.11). The impact of the uncertainties of physical, geochemical and biological parameters on the results is linked to and superimposed on the hydrological occurrence probability. Assuming statistical independence of the

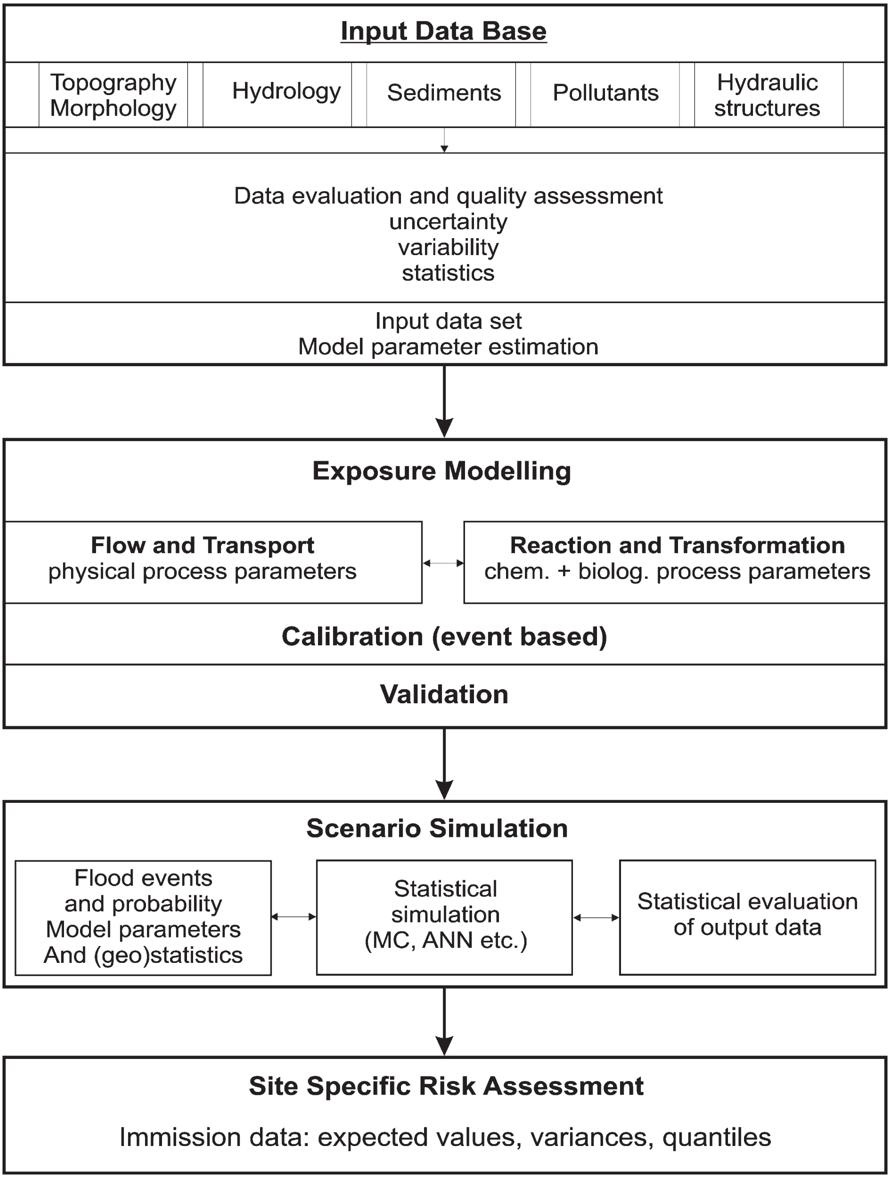


Fig. 2.11. Origin and transmission of uncertainties

parameters, the Gaussian law of error transmission can be applied for estimating the final uncertainty of the quantities. It clearly shows that the uncertainty of the target quantity can be much larger than any of the input parameters.

To cope with the uncertainty of measurement based model parameters and the effect of the variability of hydrological input variables, the use of a stochastic concept for assessing the uncertainty and improving the reliability of the model results is advisable. Stochastic concepts can be applied and integrated into a deterministic hydrodynamic transport model (Fig. 2.11).

The statistical components of the problem can be treated, for instance, by the Monte Carlo method which allows a statistical evaluation of the output. The bootstrap method (Efron and Tibshirani 1993) is an effective method for evaluating field data, especially when the amount of available data does not allow a conventional statistical analysis.

Since the hydrological component makes the dominant contribution to the sediment resuspension risk assessment, the probability, i.e. the uncertainty with respect to time, of any model based quantity increases substantially with the complexity and size of the catchment area. Large catchments consisting of different individual subcatchments with contaminated sites subject to potential resuspension are difficult to capture because of their weakly correlated regional or local flood events. Therefore, one can hardly predict the probability of an immission in terms of concentration of polluted sediments deposited at a site which is far down-stream from the emission site.

Case Study: Erosion Capacity of Floods

To demonstrate the influence of the sediment erosion parameters numerical computations with a 1-dimensional transport model based on the Monte Carlo simulation method were carried out for the 11-km stretch of the river Neckar barrage Lauffen, focusing on floods with various discharge hydrographs and a spatial variability of the sediment erosion properties (Li 2004). Historical flood events were selected from the data series covering the last 50 years to show the effect of the peak flood hydrograph characterized by the discharge and the flood volume on the erosion capacity of the flood. In addition, the uncertainty of the eroded sediment mass caused by the statistical variation of the sediment erosion parameters is quantified.

Two field studies on sediment erosion were made in 1997 and 1998. Altogether, 29 sediment cores were taken for experimental sediment erosion tests. In total, 460 data on the critical shear stress of sediment erosion are then available.

Each value from the collected data is regarded statistically independently. Using the non-parametric bootstrapping method (Efron and Tibshirani 1993) a certain number of bootstraps of the sample mean of the collected field data was produced. Each bootstrapped sample mean is assumed to be the mean critical erosion shear stress of the whole river reach, and was put into the function (Eq. 2.4) suggested by Kuijper et al. (1989) for calculating the erosion rate (E)

$$E = M\left(1 - \tau_0 / \tau_{\mathrm{crit.\,E}}\right)^n \tag{2.4}$$

where M is the erosion coefficient, n the erosion exponent, τ_0 the actual bed shear stress, and $\tau_{\mathrm{crit.\,E}}$ the critical erosion shear stress.

A data set of discharges covering almost 50 years from 1950 onwards was available for numerical simulations. The concentrations of inflowing suspended sediment as a function of the discharge were calculated using an experimentally determined power law function (Kern 1997). A field study of the flood event from 28 October 1998 to

4 November 1998 provided another data set of discharges and corresponding suspended sediment concentration (Haag et al. 2002). In the study of Li and Westrich (2004), the influence of all three sediment parameters in Eq. 2.4 was systematically investigated by a local sensitivity analysis. In the following, the discharge hydrograph and the critical erosion shear stress are considered stochastic. The influence of the two variables on the sediment erosion capacity of the flood is estimated by applying a 1-dimensional flow and sediment transport model (Kern and Westrich 1996) to the 11-km lock-regulated stretch of the river Neckar. The erosion parameters M and n were regarded as constant and set at 7.5×10^{-4} kg m^{-2} s^{-1} and 3.2 respectively.

Four historical flood events between 1950 and 1994 with different peak discharge and duration are chosen to demonstrate the effect of the shape of the hydrograph and the variability of the critical erosion shear stress on the sediment erosion potential of floods. The duration of each flood event is 10 days. 50 long term simulations were carried out for a period of 45 years using the measured discharges from 1950 to 1994. At the end of the 45-year simulation period, the impact of the respective flood event was investigated. For each simulation following the Monte Carlo method, the critical erosion shear stress was statistically determined and assumed to be constant in the entire river reach.

Figure 2.12 shows the calculated quantiles of eroded sediment for four historical flood events. Q_{max} stands for the flood peak discharge. Flood (a) differs from flood (b) in the larger flood volume and longer erosion duration. Both floods (a) and (b) show higher peak flow rate than floods (c) and (d) and hence exhibit much higher eroded sediment mass. The slope of the line indicates the effect of the variability of critical erosion shear stress. Among the four flood hydrographs in Fig. 2.12, flood (a) has the largest spreading of eroded sediment mass. A comparison between floods (a) and (b) shows that the larger flood volume of the hydrograph (a) seen in Fig. 2.13 results in a significant larger erosion capacity than flood hydrograph (b). The sediment mass eroded by flood (a) is about 53 000 tons larger than that of flood (b). The calculated results in case (a) spread over 2 000 tons, i.e. ±1 000 tons, whereas the spreading of

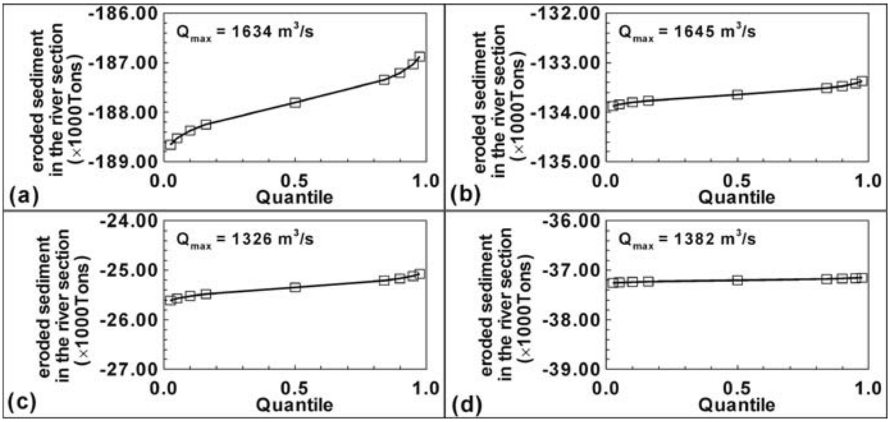


Fig. 2.12. Calculated sediment mass eroded by historical flood events with statistical variation of critical erosion shear stress from 2 to 10 N m^{-2}

flood (c) is, as expected, only about 1 000 tons, i.e. ±500 tons maximum. In comparison to both flood events (c) and (d) in Fig. 2.12 which have about the same peak discharge of 1 634 and 1 645 $m^3 s^{-1}$ respectively, more sediment is eroded by the flood in 1998 even though it had a lower peak discharge of 1 055 $m^3 s^{-1}$. The effect of the variability of the hydrograph, that is the peak discharge and duration of erosive discharge is evident.

One can conclude from the numerical investigation that each flood event has its own individual erosion capacity. Therefore, in order to assess the impact of flood events with a given return period, the whole statistical ensemble must be simulated to provide the required statistical answer. For instance, when assessing the impact of a 100-year flood event on the resuspension of contaminated sediments in a confined river stretch, a series of synthetic hydrographs with varying peak discharge and duration must be investigated and analyzed to provide a probabilistic answer to how much sediment can be eroded by such an event. The joint effect of peak flow rate and the duration of the erosive flood discharge determines the concentration and load of resuspended sediments and, in conjunction with the contamination of the sediments, it also controls the event-related total contaminant load.

Fifty simulations were carried out for the flood event from 28 October to 4 November 1998 using the measured hydrograph with a peak discharge of 1 055 $m^3 s^{-1}$ and 50 samples of critical erosion shear generated using bootstrap sampling. The value of the critical erosion shear stress was given by a random process for each simulation. The critical erosion shear stress was assumed to be constant both in space and time.

The impact of the spatial variability of critical shear stress on the erosion process can be demonstrated by the flood in 1998 (Fig. 2.14). Assuming the critical erosion shear stress varies in space, the usual case in the field, the range of the statistical results is significantly increased and amounts to 8 000 tons of eroded sediments. The maxi-

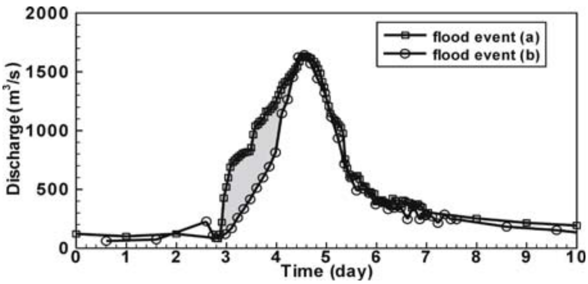


Fig. 2.13.
Hydrograph of two selected historical flood events with equal peak flow (Li 2004)

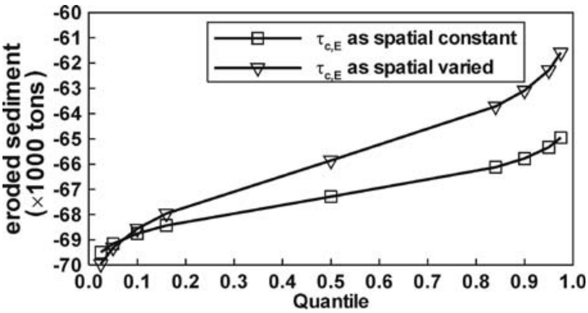


Fig. 2.14.
Quantile of calculated sediment mass eroded during flood event 1998 with a peak flow rate of 1 055 $m^3 s^{-1}$:
a spatially constant and
b spatially varied (Li 2004)

mum deviation from the mean value is about ±4 000 tons, which means a maximum uncertainty of about ±6% related to the expected quantity of 65 000 tons. The graph indicates that the statistical expected value of the eroded sediment mass is increased by about 1 500 tons due to only the spatial variability of the critical erosion shear stress. Moreover, the spatial variability of the sediment parameters does not only increase the variance but also shifts the expected mean value of the eroded sediment mass to a higher level. Only if a constant concentration of contaminants in $\mu g\ kg^{-1}$ across the sediment layers is assumed is the variance of the mass of resuspended contaminants the same as it is for the total eroded sediment mass.

References

Alderton DHM (1985) Sediments. In: Historical Monitoring, pp 1-95. MARC Technical Report 31. Monitoring and Assessment Research Centre, University of London, UK

Anonymous (1982) National Handbook of Recommended Methods for Water-Data Acquisition. Chapter 3 – Sediment pp 3-1 – 3-100. U.S. Geological Survey

Anonymous (2004a) Expert Group on Analysis and Monitoring of Priority Substances (AMPS). WFD AMPS Sediment Monitoring Guidance Discussion Document, AMPS and SedNet, Draft Version 1 from 16 April 2004

Anonymous (2004b) Evaluation of Current Gaps and Recommendations for further Actions in the Field of Environmental Analysis and Monitoring. METROPOLIS (Metrology in Support of EU Policies) Position Paper, March 2004, 8 p. Verneuil-en-Halatte/France

Anonymous (2006) Proposal for a Directive of the European Parliament and of the Council on Environmental Quality Standards in the Field of Water Policy and Amending Directive 2000/60/EC. Commission of the European Communities, Brussels, 17.7.2006. http://ec.europa.eu/environment/water/water-dangersub/pdf/com_2006_397_en.pdf

Apte SC, Batley G, Maher WA (2002) Monitoring of trace metals and metalloids in natural waters. In: Burden FR, McKelvie I, Förstner U, Guenther A (eds) Environmental Monitoring Handbook, Chapter 6. McGraw-Hill, New York

Brock TD (1983) Membrane Filtration: A User's Guide and Reference Manual. Springer-Verlag, Berlin

Buffle J, Perret J, Newman J (1992) The use of filtration and ultrafiltration for size fractionation of aquatic particles, colloids, and macromolecules. In: Buffle J, Van Leeuwen HP (eds) Environmental Particles, vol. I, Chap 5. Lewis Publ Chelsea MI

Corbisier P, van der Lelie D, Borremans B, Provoost A, de Lorenzo V, Brown N, Lloyd J, Hobman J, Csöregi E, Johannsson G, Mattiasson B (1999) Whole cell- and protein-based biosensors for the detection of bioavailable heavy metals in environmental samples. Anal Chim Acta 387:235–244

Cornelisse JM, Mulder HPJ, Houwing EJ, Williamson HJ, Witte G (1997) On the development of instruments for in-situ erosion measurements. In: Burt N, Parker R, Watts J (eds) Cohesive Sediments. John Wiley & Sons, pp 175–186

Efron B, Tibshirani R (1993) An Introduction to the Bootstrap. Chapman and Hall, New York

Eisma D, Dyer KR, van Leussen W (1997) The in-situ determination of the settling velocities of suspended fine-grined sediment – a review. In: Burt N, Parker R, Watts J (eds) Cohesive Sediments. John Wiley & Sons, pp 17–44

Förstner U (1989) Contaminated Sediments. Lecture Notes in Earth Sciences 21, Springer-Verlag, Berlin, 157 p

Förstner U (2003) Geochemical techniques on contaminated sediments – river basin view. Environ Sci Pollut Res 10:58–68

Förstner U (2004) Traceability of sediment analysis. Trends Anal Chem 23(3):217–236

Golterman HL, Sly PG, Thomas RL (1983) Study of the Relationship between Water Quality and Sediment Transport. Technical Papers in Hydrology no. 26, 231 p. UNESCO, Paris

Gust G, Müller V (1997) Interfacial hydrodynamics and entrainment functions of currently used erosion devices. In: Burt N, Parker R, Watts J (eds) Cohesive Sediments. John Wiley & Sons, pp 149–174

Haag I, Hollert I, Kern U, Braunbeck T, Westrich B (2002) Flood event sediment budget for a lock-regulated river reach and toxicity of suspended particulate matter. Proc 3rd Intern Conf on Water Resources and Environment Research (ICWRER), Dresden, Germany, July 2002

Horowitz A (1991) A Primer on Sediment-Trace Element Chemistry. 2nd ed. 136 p. Lewis Publ. Chelsea/Mich

Hunt DTE, Wilson AL (1986) The Chemical Analysis of Water: General Principles and Techniques. 2nd ed. Royal Society of Chemistry London

Keith L (1991) Environmental Samples and Anayses: A Practical Guide. 143 p. Lewis Publ. Chelsea/Mich

Kern U (1997) Transport von Schweb- und Schadstoffen in staugeregelten Fließgewässern am Beispiel des Neckars. Mitteilung des Intituts für Wasserbau, Heft 93, Universität Stuttgart

Kern U, Westrich B (1996) Mobilität von Schadstoffen in den Sediment staugeregelter Flüsse – Natur versuche in der Stauhaltung Lauffen, Modellierung und Abschätzung des Remobilisierungsrisikos kontaminierter Altsedimente. Wissenschaftlicher Bericht Nr. 96/23 (HG 237), Institut für Wasserbau, Universität Stuttgart

Kersten M, Förstner U (1987) Effect of sample pretreatment on the reliability of solid speciation data of heavy metals – implication for the study of diagenetic processes. Mar Chem 22:299–312

Kosian PA, West MS, Pasha CW, Cox JS, Mount DR, Huggett RJ, Ankley GT (1999) Use of nonpolar resin for reduction of fluoranthene bioavailability in sediment. Environ Toxicol Chem 18:201–206

Kuijper C, Cornelisse JM, Winterwerp JC(1989): Research on erosive properties of cohesive sediments. J Geophys Res 94(C10):341–350

Landenberger H (1998) CoTReM – A Multi-component Transport and Reaction Model (in German). Section of Geochemistry and Hydrogeology, University of Bremen, Germany

Li CC (2004) Deterministisch-stochastisches Bemessungskonzept zur Beurteilung der Auswirkungen erosiver Hochwasserereignisse in Flußstauhaltungen. Mitteilungen des Instituts für Wasserbau, Heft 129, Universität Stuttgart, ISBN 3-933761-32-8

Li CC, Westrich B (2004) Modeling reservoir sediment erosion by floods for assessing the effect of uncertainties and variability of input parameters. Presentation held at 6th Inter Conf on Hydroscience and Engineering (ICHE-2004), May 30–June 3, Brisbane, Australia

Mudroch A, Azcue JM (1995) Manual of Aquatic Sediment Sampling. 219 p. Lewis Publ, Boca Raton

Mudroch A, Bourbonniere RA (1994) Sediment preservation, processing, and storage. In: Mudroch A, MacKnight SD (eds) Techniques for Aquatic Sediments Sampling. 2nd ed., pp 131–169, Lewis Publ. Boca Raton

Ongley E, Blachford N (1982) Application of continuous-flow centrifugation to contaminant analysis of suspended sediment in fluvial systems. Environ Technol Letts 3:219–228

Parkhurst DL, Postma CAJ (1999) User's Guide to PHREEQC (Version 2) – A Computer Program for Speciation, Batch-Reaction, One-Dimensional Transport, and Inverse Geochemical Calculations. Water-Resources Investigations Report 99-4259. U.S. Geol. Survey, Denver, Colorado, USA

Quevauviller Ph (ed) (2002) Methodologies for Soil and Sediment Fractionation Studies. 180 p. The Royal Society of Chemistry Cambridge UK

Thomas R, Meybeck M (1992) The use of particulate material. In: Chapman D (ed) Water Quality Assessments. A Guide to the Use of Biota, Sediments and Water in Environmental Monitoring. Chapter 4, pp 121–170, Chapman & Hall, London

Thompson HA, Parks GA, Brown GE jr (1999) Dynamic interaction of dissolution, surface adsorption and precipitation in an aging cobalt(II)-clay-water-system. Geochim Cosmochim Acta 63: 1767–1779

Verbruggen EMJ, Vaes WHJ, Parkerton TF, Hermens JLM (2000) Polyacrylate-coated SPME fibers as a tool to simulate body residues and target concentrations of complex organic mixtures for estimation of baseline toxicity. Environ Sci Technol 34:324–331

Wallmann K Kersten M, Gruber J, U. Förstner U (1993) Artifacts in the determination of trace metal binding forms in anoxic sediments by sequential extraction. In: Rauret G, Quevauviller P (eds) Sequential Extraction of Trace Metals in Soils and Sediment. Int J Environ Anal Chem 51:187–200

Zhang H, Zhao F-J, Sun B, Davison W, McGrath SP (2001) A new method to measure effective soil solution concentration predicts copper availability to plants. Environ Sci Technol 35:2602–2607