Peter Heininger · Johannes Cullmann
Editors

Sediment Matters

Springer

Sediment Matters

Peter Heininger · Johannes Cullmann
Editors

Sediment Matters

Springer

Editors
Peter Heininger
Federal Institute of Hydrology
Koblenz
Germany

Johannes Cullmann
IHP/HWRP
Federal Institute of Hydrology
Koblenz
Germany

ISBN 978-3-319-14695-9 ISBN 978-3-319-14696-6 (eBook)
DOI 10.1007/978-3-319-14696-6

Library of Congress Control Number: 2014959852

Springer Cham Heidelberg New York Dordrecht London

Printed on acid-free paper

Springer International Publishing AG Switzerland is part of Springer Science+Business Media (www.springer.com)

Preface

Erosion, sedimentation processes and management in catchments, river systems and reservoirs have reached global importance. Sediment transport is a vital component of natural hydromorphological regimes. Contaminated sediments can have adverse effects on people, environment and economy. Sediment is a fundamentally important component of aquatic ecosystems. Where human activities interfere with sediment quantity or quality, sediment management becomes necessary. Sediment processes and their socio-economic and environmental impacts are many and varied, making it almost impossible to treat them all in the framework of a single book. Rather, the purpose of this book is to provide exemplary insights into the relevant aspects related to sediment and sediment management as they were presented and discussed during the 6th International Conference on Water Resources and Environment Research in Koblenz, Germany in June 2013. The research findings included in the individual chapters of this publication will allow readers to gain an overview of the relevant boundary conditions, drivers, processes and consequences of erosion, sediment transport and sedimentation at different scales. The inter-linkages of sediment dynamics and sediment quality with bio-geochemistry, ecology and human activities and their consequences for an effective sediment management are shown exemplarily in the various chapters of this book and allow to put individual questions and issues into a broader sediment perspective.

Our main acknowledgement goes to all authors of individual chapters. We also acknowledge the help of the reviewers, the lector and all who have provided the necessary support for this publication. We thank very much Mrs. B. Noll and Mrs. A.M. Conde Corral for their tireless patience and technical support during the writing and publishing of this book.

Koblenz, 2015

Peter Heininger
Johannes Cullmann

Contents

Introduction

Johannes Cullmann and Peter Heininger

1 Sediment as Part of the River Basin

Natural river basins are continuously evolving and adapting. Erosion, sediment transport and sedimentation have been key factors for landscape development, the genesis and degradation of soils, water quality, the evolution of aquatic habitats and the formation of river deltas for geological eras. Both small and substantial changes in sediment distribution, erosion, deposition, and transport are natural and necessary processes in aquatic ecosystems. The magnitudes of the sediment loads transported by rivers have important implications for the functioning of the system; for example through their influence on material fluxes, geochemical cycling, water quality, channel morphology, delta development, and the aquatic ecosystems and habitats supported by the river.

Erosion and sedimentation processes interact with human usages of river system services. Often, as a consequence of river training, inputs of energy can act only vertically in the direction of the river bed thus encouraging the depth erosion of the bed. Scouring increases where flow velocities are increased and is a frequent phenomenon downstream of sediment sinks. Local scour and sedimentation effects may dramatically impact on dams and bridges, and balanced sediment conditions are of paramount importance for the stability, reliability and functioning of hydraulic infrastructure. This becomes evident when looking at reservoir sedimentation, the silting of irrigation infrastructure or riverbed erosion of engineered streams. Walling (2006) estimates the total loss of worldwide reservoir volume due to sedimentation at a rate of 0.5–1 % per annum. Sediment trapping in reservoirs

J. Cullmann (✉) · P. Heininger
Federal Institute of Hydrology, Am Mainzer Tor 1, 56068 Koblenz, Germany
e-mail: cullmann@bafg.de

P. Heininger
e-mail: heininger@bafg.de

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_1

and check dams can lead to extremely dangerous situations like the failure of the 10.5 Mm^3 Balin Dam in Taiwan in 2007 (Kondolf et al. 2014).

Engineered structures strongly affect the hydraulic conditions and the morphology of rivers and estuaries. The sediment budget of a river is closely connected with its hydromorphology. Weakly developed hydromorphological features are indicators of a disturbed sediment budget. Vice versa, the hydromorphological characteristics of the river have influence on the sediment budget. The prevailing hydromorphological conditions, in turn, are crucial for the diversity of habitats and biota (Bábek et al. 2008; Collins et al. 2011; Langhammer 2010). Floodplains and marshes have been dramatically reduced worldwide, mainly due to dyke construction. One of their widely acknowledged functions is the sequestration of sediment and associated substances. This ecosystem function is severely reduced by the loss of floodplains (Ciszewski 2001; Walling et al. 1998, 2000).

Sediment is closely linked to water quality issues. High sediment concentration in water may call for a cost intensive purification process in order to guarantee the desired human use. Historical contamination from industrial and mining activities as well as present-day point- and non-point emissions may become sources of sediment contamination what inevitably leads to conflicts with human activities such as agriculture or fishing and can pose a general threat to aquatic communities. As a legacy of the past, sediment contamination in many aquatic systems—lakes, estuaries, and coastal oceans—represents a world-wide problem (Burton and Johnston 2010; Chapman and Wang 2001; Dagnino et al. 2013; Förstner and Salomons 2010; Heise 2009; Lair et al. 2009; Liu et al. 2000). Contaminated sediments from still-water zones can be mobilized during flood events and contaminant reaches far downstream from the actual source, ultimately impacting the marine environment (Bopp et al. 1998; Grousset et al. 1999; Heininger et al. 1998; Schwientek et al. 2013). Generally, there is an increasing recognition that fine sediment represents an important diffuse pollutant source in surface waters, due to its role in governing the transport of contaminants through fluvial systems and because of its impacts on aquatic ecology (Owens 2005). An understanding of the sources, behaviour, and storage of sediment-associated contaminants in rivers is therefore needed, so that appropriate strategies may be implemented to reduce and control both contaminant inputs into rivers and the detrimental effects associated with such contaminants within rivers and receiving systems. Suspended sediment should be considered in this context as well as floodplain deposits and channel bed sediment, when studying the temporal and spatial patterns of contaminant behaviour in river systems (Grabowski et al. 2012; Hu et al. 2014; Owens 2005; Salomons 2008).

2 Some Facts About Sediment Dynamics

Sediment influx into the oceans and related nutrient and pollutant fluxes are key parameters for global bio-geochemical processes. Generally, human activity can either enhance or decrease sediment dynamics. In natural systems, sediment balance is oscillating around a stable optimum. Virtually all suspended sediment is supplied to river systems either by terrestrial erosion or through the production of organic matter. It may be released, for example, as a consequence of heavy rainstorms, debris flow, collapse of local river banks and the continuous reshuffling of sediments in river channels. Sediments are accumulated wherever shear stress is below critical values and suspended sediments can be deposed with terminal velocity when conditions allow.

In river systems with anthropogenic impact sediment dynamics are often altered compared to the natural status. Syvitsky et al. (2005) showed that about 26 % of the global sediment transit is trapped in reservoirs. The Yellow River in China is a typical example for decreasing sediment dynamics. Suspended sediment load delivered to the China Sea was recorded to be about 1.1 Gt per year in the 1950s. This amount has decreased to about 0.4 Gt per year in 1990 (Walling 2009). More recent data indicate that the load may even be down to 0.15 Gt per year (Wang et al. 2007). This load reduction is accompanied by a proportional decrease in river flow mainly due to the abstraction of water for economic activities. The main reason for the decrease in sediment load is the trapping of sediment in reservoirs. An example for increasing sediment dynamics can be found in Walling (2005). The Rio Magdalena drains a 260,000 km^2 river basin in the Andes. The sediment load to the Carribean has increased by 40 % from 1975 to 1995. This is attributed to the fact that forest clearance and intensification of agriculture enhance the degradation of soils. In addition, mining activities contribute here to an increased sediment load of the river. A further relevant impact on sediment balance may result where sand extraction is a major source of income. Wang et al. (2007) estimate that as much as 110 Mt sand is extracted annually in the Yangtze catchment. The sediment load of the Yangtze River has decreased from about 500 Mt per year in the 1960s to about 200 Mt per year around the year 2000.

General conclusions can be drawn from these scenarios. The sediment transport into the oceans is decreasing on global scale. Intensification of silviculture, mining and agricultural activities without appropriate soil conservation management will inevitably lead to increased erosion and thus enhanced sediment supply to the rivers. The free sediment flow is increasingly disturbed in river basins, and the sediment storage in river system is increasing. Typically, hydraulic infrastructure like dams and weirs trap sediment and thus decrease the sediment load of rivers, even if erosion is accelerated at the same time. This phenomenon can be observed in many rivers throughout the world such as Danube, Mississippi, and Indus.

3 Sediment Management

The above mentioned general facts and relations clearly show that sediment management concepts are indispensable tools for provident and sustainable planning and operation of human activities in river basins today.

In order to provide operational services with adequate sediment management plans, several preconditions must be fulfilled. First of all sediment management must be based on accurate knowledge about erosion, the pathways of sediment transport into the river system and within the river system. Next, the quality of sediments is crucial for any kind of management. Polluted sediment often poses serious problems to water management authorities and its disposal can be costly. Thus the first step towards a sound management plan is a survey of pathways, a screening for pollutants and putting in place a monitoring system that captures sediment dynamics and quality with adequate spatial and temporal resolution. The sediment budget concept (Dietrich and Dunne 1978) provides a valuable framework for assisting the management and control of diffuse-source sediment pollution and associated problems, by identifying the key sources and demonstrating the importance of intermediate stores and the likely impact of upstream mitigation strategies on downstream sediment and contaminant fluxes (Walling and Collins 2008).

Hydrologists and geo-morphologists have recognized for over a century that the river basin is the fundamental unit of study in hydrology and fluvial geomorphology (Chorley 1969; Gregory and Walling 1973; Owens 2005; Walling and Collins 2008). Historically, sediment management was driven by quantity issues. Sediments were dredged to maintain waterways, or were extracted as a resource (sand, gravel, etc.). Currently, much of the thinking on sediment management and sediment risk assessment is concentrated on sediment quality and on the role of sediments in hydromorphology and ecology. It is the interdependence between the management of sediment quantity and quality that has to be effectively addressed in up to date sediment management concepts (Heise 2009; Heise and Förstner 2007; Owens 2005; SedNet 2003, 2007).

Erosion and sedimentation impact on different scales. The large scale sediment balance of a river system impacts on general ecological conditions like habitat, estuaries and near shore aquatic biota. Locally, erosion and sedimentation can be critical to pillars of bridges or culverts. Consequently, a sediment management plan should consider different scales and integrate the overall benefit of managing sediments. A further step in the design of a management plan is a thorough risk analysis for different single objectives in the overall objective function. This means that priority areas, critical infrastructure, threshold values for sediment quality and/or scouring/sedimentation and ecological indicators need to be agreed on and given a relative value in the overall objective function.

Different actors (nations, organizations, stakeholders) may have different objectives when they address sediments. A framework must be devised that allows goals and priorities to be balanced in a transparent way. Therefore, as a third step, the management plan needs to give concrete advice on how the different objectives can be reached, how they will impact each other, and a cost estimate. For example, in a management plan, objectives could be (a) to guarantee certain shipping channel geometry and (b) to enhance sediment transport through locks and weirs. These objectives might be contradictory and, if pursued alone, a single objective will likely prejudice the other objective. Therefore management plans need to encompass concrete measures that will be able to address a multi-objective target.

4 The Content of This Book

This book consists of 12 technical articles addressing key areas around the sediment issue. In five sections the following topics are addressed: sediment transport processes, modelling sediment transfer in rivers, sediment quality, sediment monitoring and sediment management in river basins.

4.1 Sediment Transport Processes

In the articles "*Sediment Transport in Headwater Streams of the Carpathian Flysch Belt: Its Nature and Recent Effects of Human Interventions*" and "*Aspects of Sediment Transport in Single-Thread and Anabranching River Channels in Flysch Carpathians* (*A Case Study from the Czech Republic*)" results of both empirical and modelling research of sediment transport in headwater streams are presented. Sediment transport and morphological processes are linked to watershed management challenges. In "*Sediment Transport Processes Related to the Operation of a Rapid Hydraulic Structure* (*Boulder Ramp*) *in a Mountain Stream Channel: A Polish Carpathian Example*" effects of infrastructure on sediment transport processes on a small scale are dealt with. As one aspect, the functionality of a hydraulic structure enabling the migration of fish and benthic invertebrates is discussed in terms of sediment continuity.

4.2 Modelling Sediment Transfer in Rivers

Modelling is specifically addressed in three chapters. "*Challenges in Modelling Sediment Matters*" provides an overview of the possibilities we have at hand when modelling sediment transport processes. In "*Suspended Sediment Estimation Using an Artificial Intelligence Approach*" different modelling techniques to the prediction of suspended sediment concentrations in rivers are discussed. Finally, in "*Projected Climate Change Impact on Soil Erosion and Sediment Yield in the River Elbe Catchment*" ensemble modelling is used to draw conclusions on the impact of climate change on soil erosion compared to that of potential land cover change.

4.3 Sediment Quality

The study of "*Water Quality and Sediment Management in Brahmaputra Basin of India: Impact of Agricultural Land Use*" is building a bridge between the impacts of land use and demographic pressure on sediment quality and, subsequently, on the surface and groundwater quality. Based on long-term monitoring data and a robust mixing model, "*Contamination of Sediments in the German North Sea Estuaries Elbe, Weser and Ems and Its Sensitivity to Climate Change*" provides a detailed study on suspended sediment as a source of pollution in tidal rivers. Different scenarios for extreme weather conditions in the future and implications for the water quality conditions of the estuaries are described.

4.4 Sediment Monitoring

In this section important aspects of sediment quantity and quality monitoring are addressed with one article each. The Chapter entitled "*Application of a New Monitoring Strategy and Analysis Concept of Suspended Sediments in Austrian Rivers*" tellingly depicts the components that are needed for effective monitoring of suspended sediments. It describes various measuring techniques as well as the validation of measurement in the Austrian case. "*Stream Sediment Geochemistry of the Areas Impacted by Mines, B1 Secondary Catchment of the Olifants Primary Catchment Area, South Africa*" deals with the relationship of the water quality and metal loadings on sediments. A comprehensive monitoring system was established to investigate the severity of mining activities in the catchment on the water resources and the ecosystem.

4.5 Sediment Management in River Basins

The chapter "*An Approach to Simulating Sediment Management in the Mekong River Basin*" provides a large scale example of possibilities and implications for sediment management. Alternative sediment management practices are discussed in view of the different interests pursued in the catchment. Finally, "*Sediment Management on River Basin Scale: the River Elbe*" describes a scientifically based integrated sediment management plan in support of a comprehensive management planning in a large international river basin.

5 Future Developments

Physical and chemical aspects of sediment processes are taught in universities around the globe. At the same time, sediment issues have dramatically changed their focus during the past decades. The view that sediment management is a purely quantitative operation that involves dredging and relocation of a certain amount of sediment per time is history. Sediment quality, the role of sediments in hydromorphology and ecology and the implications of the sediment conditions for ecosystem services have gained much awareness over the past years. Yet there are only few management plans that really involve sediments from source to final deposition in terms of quantity, quality and dynamics. Linking soil conversation practices and wetland management/restoration to sediment quality and thus the health of aquatic ecosystems is by far not the standard in recent sediment management approaches.

The great challenge for the international community lies in linking the solutions that have been developed in specific sectors such as soil conservation in the agricultural context or dredged material management for securing waterways and their navigability. This can only be achieved if several conditions apply:

1. Motivation
   Real examples that show convincing benefit of integrated management plans are the only way to engage authorities that tend to minimize their costs rather than to look at the overall benefit of integrated management.
2. Linking scientists and practitioners
   Real examples of added values arise when scientific findings can be combined with day to day practical procedures of executing agencies. A fine example for such a possible short circuit of science with management is the sediment management concept described in the chapter "*Sediment Management on River Basin Scale: the River Elbe*". To bring such concepts to bear, it is crucial that the international community improves networks for mutual capacity development. This can be fostered through initiatives like SedNet. The mission of SedNet is "to be a European network for environmentally, socially and economically viable practices of sediment management at river basin scale". A further example is the International Sediment Initiative (ISI)—a UNESCO initiative that aims at fostering sustainable water and sediment management on the global scale. These initiatives provide nuclei for outreach to river basin organizations and national services in order to provide opportunities to combine science and practice.
3. Investing in university education
   The vast majority of university based education in the field of sediments is historically linked to geology (isotope geology and morphology), Chemistry (inorganic as well as hydro- and environmental chemistry) and civil engineering (sediment control and sediment transport). The new paradigm of *integrated* sediment management is generally not yet well reflected in modern university education. Whilst writing this introduction, the authors could not find a course offer for integrated optimization of sediment management options during a half hour web search. This leads to the impression that there are either not enough possibilities for education, or these possibilities are not well communicated. Here, national science boards, education ministries and international organizations that are engaged in curricula development (i.e. UNESCO) are expected to act as catalysts for improving the situation.

Future developments should be addressed against a background of general facts and requirements like:

- Sediment transport into the oceans is decreasing on global scale
- Natural sediment flow is increasingly disturbed in river basins
- Sediment storage in river system is increasing
- Erosion from land surface is generally intensifying
- Scouring is a frequent phenomenon downstream of sediment sinks
- Sediment quality and water quality must be jointly considered
- In many rivers, sediment pollution is still on the rise and contaminant transfer may threaten environment and management goals far from the pollution source
- The sediment budget of a river and its hydromorphological conditions are two sides of the same coin
- Deficits in the sediment status both in terms of budget and quality threaten ecosystems and their services
- Sediment management needs to address the whole river basin and should integrate quantitative, qualitative and ecological aspects in a unified conception
- Sound, system-related sediment monitoring is a prerequisite for sustainable management.

These general facts and requirements simplify a wide variability of specific dynamics as well as cause and effect chains. They do not necessarily apply as a whole to all river basins. But they are suitable for setting the general scene and summarizing salient issues for designing appropriate sediment management plans. More details and discussion are to be found in the chapters of this book.

References

Bábek O, Hilschorvá K, Nehyba S, Zeman J, Famera M, Francu J, Holoubek I, Machá J, Klánová J (2008) Contamination history of suspended river sediments accumulated in oxbow lakes over the last 25 years in the Morava river (danube catchment area), Czech Republic. J Soils Sediments 8(3):165–176

Bopp RF, Chillrud SN, Shuster EL, Simpson HJ, Estabrooks FD (1998) Trends in chlorinated hydrocarbon levels in Hudson river basin sediments. Environ Health Perspect 106(4):1075–1081

Burton GA, Johnston EL (2010) Assessing contaminated sediments in the context of multiple stressors. Environ Toxicol Chem 29(12):2625–2643

Chapman PM, Wang F (2001) Assessing sediment contamination in estuaries. Environ Toxicol Chem 20(1):3–22

Chorley RJ (1969) The drainage basin as the fundamental geomorphic unit. In: Chorley RJ (ed) Water, earth and man. Methuen, London, pp 77–100

Ciszewski D (2001) Flood-related changes in heavy metal concentrations within sediments of the Biała Przemsza river. Geomorphology 40:205–218

Collins AL, Naden PS, Sear DA, Jones JI, Foster IDL, Morrow K (2011) Sediment targets for informing river catchment management: international experience and prospects. Hydrol Process 25(13):2112–2129

Dagnino A, Bo T, Copetta A, Fenoglio S, Oliveri C, Bencivenga M, Felli A, Viarengo A (2013) Development and application of an innovative expert decision support system to manage sediments and to assess environmental risk in freshwater ecosystems. Environ Int 60:171–182

Dietrich WE, Dunne T (1978) Sediment budget for a small catchment in mountainous terrain. Z Geomorphol 29:191–206

Förstner U, Salomons W (2010) Sediment research, management and policy. J Soils Sediments 10 (8):1440–1452

Grabowski RC, Wharton G, Davies GR, Droppo IG (2012) Spatial and temporal variations in the erosion threshold of fine riverbed sediments. J Soils Sediments 12:1174–1188

Gregory KJ, Walling DE (1973) Drainage basin form and process. Arnold, London

Grousset FE, Jouanneau JM, Castaing P, Lavaux G, Latouche C (1999) A 70 year record of contamination from industrial activity along the Garonne river and its tributaries (SW France). Estuar Coast Shelf Sci 48:401–414

Heininger P, Pelzer J, Claus E, Tippmann P (1998) Contamination and toxicity trends for sediments—case of the Elbe river. Water Sci Tech 37(6–7):95–102

Heise S (2009) Sediments in river basins from the perspective of SedNet, the European sediment network. J Soils Sediments 9:393–399

Heise S, Förstner U (2007) Risk assessment of contaminated sediments in river basins—theoretical considerations and pragmatic approach. J Environ Monit 9:943–952

Hu W, Huang B, Zhao Y, Sun W, Gu Z (2014) Distribution, sources and potential risk of HCH and DDT in soils from a typical alluvial plain of the Yangtze river delta region. China Environ Geochem Health 36(3):345–358

Kondolf GM, Gao J, Annadale GW, Morris GL, Jiang E, Zhang J, Cao Y, Carling P, Fu K, Guo Q, Hotchkiss R, Peteuil C, Sumi T, Wang HW, Wei Z, Wang Z, Wu B, Wu C, Yang CT (2014) Sustainable sediment management in reservoirs and regulated rivers: experiences from five continents. Earth's Future 2:256–280. doi:10.1002/2013EF000184

Lair GJ, Zehetner F, Fiebig M, Gerzabek MH, Van Gestel CAM, Hein T, Hohensinner S, Hsu P, Jones KC, Jordan G, Koelmans AA, Poot A, Slijkerman DME, Totsche KU, Bondar-Kunze E, Barth JAC (2009) How do long-term development and periodical changes of river-floodplain systems affect the fate of contaminants? Results from European rivers. Environ Pollut 157:3336–3346

Langhammer J (2010) Analysis of the relationship between the stream regulations and the geomorphologic effects of floods. Nat Hazards 54(1):121–139

Liu M, Baugh PJ, Hutchinson SM, Yu L, Xu S (2000) Historical record and sources of polycyclic aromatic hydrocarbons in core sediments from the Yangtze estuary. China Environ Pollut 110:357–365

Owens PN (2005) Conceptual models and budgets for sediment management at the river basin scale. J Soils Sediments 5(4):201–212

Salomons W (2008) Linking soils, sediments and catchment ecosystems. J Soils Sediments 8 (2):67–68

Schwientek M, Rügner H, Beckingham B, Kuch B, Grathwohl P (2013) Integrated monitoring of particle associated transport of PAHs in contrasting catchments. Environ Pollut 172:155–162

SedNet (2003) The SedNet strategy paper. In: Brils J (ed) TNO-MEP. The Netherlands, 22 p

SedNet (2007) Sediment management—an essential element of river basin management plans. Report on the SedNet round table discussion, Venice, 22–23 Nov (www.sednet.org)

Syvitski JPM, Vörösmart CJ, Kettner AJ, Green P (2005) Impact of humans on the flux of terrestrial sediment to the global coastal ocean. Science 308:376–380

Walling DE, He Q, Blake WH (2000) River flood plains as phosphorus sinks. The Role of erosion and Sediment Transport in Nutrient and Contaminant Transfer. Waterloo, Canada, IAHS

Walling DE, Owens PN, Leeks GJL (1998) The role of channel and floodplain storage in the suspended sediment budget of the river Ouse, Yorkshire, UK. Geomorphology 22:225–242

Walling DE (2005) Human impact on land-ocean sediment transfer by the worlds rivers. Geomorphology 79:192–216

Walling DE (2006) Human impact on land-ocean sediment transfer by the world's rivers. Geomorphology 79:192–216

Walling DE, Collins AL (2008) The catchment sediment budget as a management tool. Environ Sci Policy 2:136–143

Walling DE (2009) The impact of global change on erosion and sediment transport by rivers: current progress and future challenges. UN-Water World Water Development Report III

Wang Z, Li Y, He Y (2007) Sediment budget of the Yangtze River. Water Resour Res 43, W04401, doi:10.1029/2006WR005012

Part I
Sediment Transport Processes

Sediment Transport in Headwater Streams of the Carpathian Flysch Belt: Its Nature and Recent Effects of Human Interventions

Tomáš Galia, Jan Hradecký and Václav Škarpich

Abstract The paper summarizes results of both empirical and modelling research of bedload transport in headwater streams of the Czech part of the Western Carpathians. Flysch lithology (i.e. alternation of less resistant claystones and sandstones) affects bedload transport parameters in view of relatively fine-sized sediment supply resulting in low flow resistance of channels. Flood competence method (Q20 flood) and marked particle displacement method (up to Q1–2 flow) was applied to determine critical conditions for the incipient motion of grains in channel bed. The beginning of bedload transport in flysch headwaters under lower values of critical conditions when compared to other regions was confirmed by application of the criteria of unit stream power and unit discharge. The simulated values of bedload transport intensity (1D transport model TOMSED) during a high-magnitude flood in both supply-limited and transport-limited headwaters are significantly lower than it was observed in torrents of the Alpine environment. In relation to unsuitable contemporary watershed management affecting the sediment transport (large check-dams, removing of large wood from local channels), trends of accelerated incision are observed in most headwater streams as well as in lower-gradient piedmountain gravel-bed rivers of the Flysch Belt of the Western Carpathians. Approaches of contemporary local watershed management are presented and some recommendations for the maintenance of channel stability predisposed by soft lithology (e.g. application of artificial step-pool sequences, management of woody debris in channels) are proposed.

T. Galia (✉) · J. Hradecký · V. Škarpich
Faculty of Science, Department of Physical Geography and Geoecology,
University of Ostrava, Chittussiho 10, 710 00 Ostrava, Czech Republic
e-mail: tomas.galia@osu.cz

J. Hradecký
e-mail: jan.hradecky@osu.cz

V. Škarpich
e-mail: vaclav.skarpich@osu.cz

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_2

1 Flysch Structures as a Predisposing Factor for Channel-Forming Processes

From a lithological point of view, Flysch nappe structures of the Czech part of the Outer Western Carpathian (Silesian Nappe and Magura Nappe) are generally built by two types of sedimentary rocks: sandstones and much less-resistant claystones although some other rocks such as conglomerates and limestones also occur. Such structures strictly predispose sediment delivery into local channel segments with respect to specific sediment inputs and grain-size characteristics. Heterogenous mixture of bedrock layers and tectonically weakened zones affects chronic hillslope instability, while both shallow and deep landsliding is typical of the study area which represents the most landslide-affected region within the territory of the Czech Republic (Hradecký and Pánek 2008; Pánek et al. 2011). Valleys of high-gradient streams draining out the highest mountainous areas; the steepest channels are affected by small 'fire hose' effect-related debris flows, which are usually connected with high-magnitude flood events.

Headwater channels based in less resistant claystones are prone to accelerated incision and large bank failures, which are activated in case of limited sediment-supply conditions. Grain-size character of sediment supply in the flysch Western Carpathians is very rarely represented by sandstone boulders of diameters >0.5 m; cobbles and smaller grain fractions prevail. Median particle-sizes are around the value of 50 mm, while d_{90} usually varies between 120 and 300 mm, depending on the lithology of sediment supply (ratio of claystones in bed sediments) and channel gradient. This implies an occurrence of channel-reach morphologies related to grain-size characteristics of stream bed; the absence of larger boulders usually prevents the formation of regular step-pool morphology sensu Montgomery and Buffington (1997). Cascades and rapid channels with absent high water scours from steps to pools implying lower form resistance are quite typical of local flysch conditions. Absence of boulder fraction directly affects total flow resistance, which leads to higher potential transport capacity of local channels. This lithology-conditioned predisposition has also strongly contributed to recent trends of accelerated incision.

2 Bedload Transport in Small Flysch Mountain Catchments

2.1 Critical Conditions for the Commencement of Bedload Transport

Better understanding of the trigger of bedload transport at steep stream gradients is necessary for the improvement of watershed management connected with the protection of the property and human lives. As a result of low flow resistance in flysch-based channels, critical conditions (i.e. critical unit stream power, critical unit discharge, and critical shear stress) for the commencement of bedload transport of certain grain-size fractions have generally lower values than those obtained in other environments. Two methods were successfully used in two local steep headwater streams, the Malá Ráztoka Brook and the uppermost part of the Lubina River ($A < 2$ km^2), to determine critical conditions for the incipient motion of grains in channel bed. Flood competence method was based both on the measurement of sizes of cobbles and boulders transported during Q_{20} discharge and the reconstruction of geometrical parameters of channel during this event. The latter approach consisted in marking individual grains and subsequent observation of their movement following lower flow events that varied between annual discharge and bankfull discharge ($Q_{1–2}$). A flood with Q_{20} discharge (specific discharge ca 2 m^3 s^{-1} km^{-1}) set in motion almost all bed material of a diameter up to 0.3–0.4 m, which corresponds to d_{95}–d_{99} size fraction of bed surface (Galia and Hradecký 2011). Transport of marked particles with maximal diameters up to 0.1 m was observed during lower flows not exceeding Q_2 (Galia and Hradecký 2012).

Using a combination of data from flood competence and marked particle displacement methods back calculations of basic bed shear stress formulas were applied: $\tau_b = \rho \cdot g \cdot R \cdot S$ and $\tau_{ci} = \tau^*_{ci} \cdot (\rho_s - \rho) \cdot g \cdot d_i$, where τ_b means shear stress acting on the channel bed; τ_{ci} means critical shear stress for the movement of particle of diameter d_i; τ^*_{ci} means dimensionless critical shear stress or Shields parameter for d_i grain size; ρ means density of water; ρ_s means density of a solid particle; g is gravitational acceleration; R means hydraulic radius; and S means channel gradient. The application of dimensionless critical shear stress as a function of ratio d_{90}/d_i (Lenzi et al. 2006) led to the relationship $\tau^*_{ci} = 0.1(d_{90}/d_i)^{-0.52}$. The original relationship derived from the Rio Cordon torrent in Italian Dolomites ($d_{50} = 119$ mm and $d_{90} = 451$ mm) by Lenzi et al. (2006) indicated a significantly lower exponent (−0.737); the influence of general lower d_{90} percentiles in flysch torrents is reflected due to relatively fine character of local sediment supply.

The beginning of bedload transport in flysch headwaters under relatively low values of critical conditions was confirmed applying the criteria of unit stream power ω_{ci} and unit discharge q_{ci} for maximal transported particle diameter d_i in forms $\omega_{ci} = a \cdot d_i^b$ (d_i in mm) and $q_{ci} = a \cdot d_i^b$ (in m). Unit stream power ω is usually defined as $\omega = (Q \cdot \rho \cdot S \cdot g)/w$, where w means channel width. Figure 1 shows a comparison between the relationship observed in Czech Carpathian headwater channels, $\omega_{ci} = 0.57 d_i^{1.02}$ with respect to the boulders transported during Q_{20} flood and marked particle displacement during lower discharges, and the power trends obtained in other environments. It documents well that bedload transport of certain grain-size fraction in Alpine and Andine high-gradient channels with $a = 31.5$ and $b = 0.488$ (Mao et al. 2008) and Belgian gravel-bed streams with $a = 1.13$ and $b = 1.438$ (Petit et al. 2005) begins under much higher critical stream powers than that obtained in Carpathian flysch headwaters. Moreover, our trend is very close to the lower limit in $\omega_{ci} = a \cdot d_i^b$ relationship derived by Williams (1983) for a large worldwide set of gravel-bed streams ($a = 0.079$, $b = 1.3$).

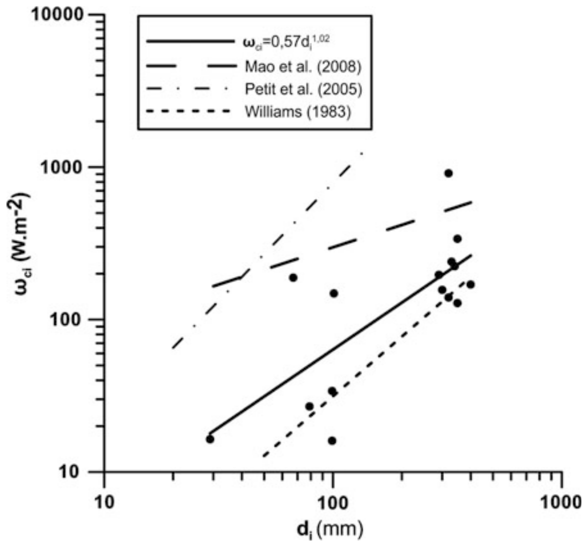
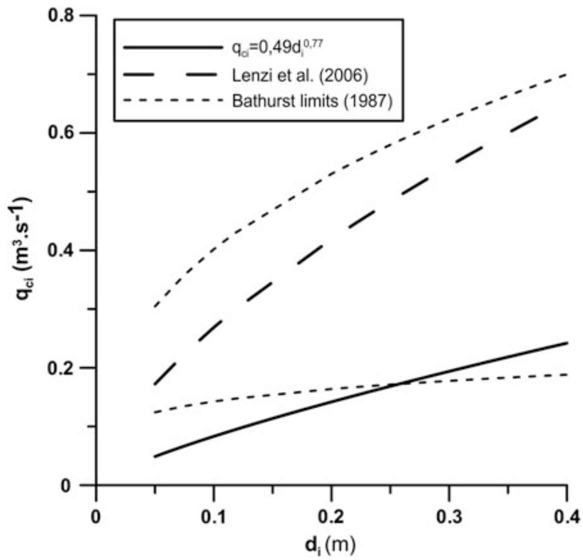


Fig. 1 Comparison of trends in critical unit stream power between flysch headwaters in the Carpathians and other environments

A similar situation arose when comparing the relationship for unit discharge $q_{ci} = Q/w$ for grain diameter d_i. Bathurst (1987) introduced coefficient a between 0.09 and 0.16 and exponent b in a range of 0.2–0.4 for the Rocky Mountains (USA) streams in $q_{ci} = a \cdot d_i^b$ relationship (d_i in mm). Lenzi et al. (2006) later substituted 1.176 for a and 0.641 for b after analysing grain motion in the Dolomitan Rio Cordon torrent (d_i in m). As for Carpathian flysch headwater streams, coefficient a was equal to 0.49 and b to 0.77 (d_i in m) when considering transport of large boulders during Q_{20} event and marked particles movement during lower discharges, again implying incipient motion of individual grain diameters under relatively low critical discharges (see Fig. 2).

Presented results document that in any case, bedload transport in flysch mid-mountains begins under relatively low discharges when compared with other small mountain streams with different predispositions. Nevertheless, a role of sediment supply in view of limited sediment supply and, on the contrary, limited transport capacity was not evaluated and our data are a combination of headwaters covering both conditions. Numerous papers documented a relationship between supply-limited conditions and the armouring of bed material resulting in a higher stability of bed surface (e.g. Whiting and King 2003; Hassan and Zimmermann 2012). In addition, higher stability of step-pool channels was observed under low sediment supply conditions (Recking et al. 2012) implying high critical conditions for the destruction of steps and incipient motion of the coarsest fraction.

Fig. 2 Comparison of trends in critical unit discharge between flysch headwaters in the Carpathians and other environments



2.2 Simulations of Bedload Transport

Direct bedload measurements in steep mountain channels are still rare. Limited sediment supply character together with important partitioning of form resistance in stepped-bed morphologies and influence of woody debris and bedrock outcrops lead to the overestimation of bedload transport volumes by conventional equations originally developed for gravel-bed rivers (e.g., Yager et al. 2007; Chiari and Rickenmann 2011). Modelling of bedload transport was conducted using the one-dimensional TOMSED model (Friedl and Chiari 2011) in two flysch headwater channels with a different regime of sediment supply. The Malá Ráztoka Brook (2.2 km^2) is characterised by limited sediment supply character with a significant occurrence of resistant sandstone outcrops (up to ½ of the total length of the longitudinal stream profile). On the contrary, Velký Škaredý Brook (1.06 km^2) can be described as transport limited due to a large number of sediment sources; bank failures occur in non-resistant claystone members in the downstream part of the stream profile, while debris flow accumulations are typical of the uppermost part. The modelled event was the 5/2010 flood (Q_{20}) peaking up to 4 m^3 s^{-1} in the Malá Ráztoka gauging station (2.02 km^2). The sediment erosion and deposition along the studied channels were mapped in the field shortly after the flood event and compared to the situation before the event. Simulated volumes of bedload transported material and bedload discharge were verified in accordance with these erosion or deposition in channels before and after the flood event due to missing bedload transport data from direct measurements.

The Manning equation with separated grain and form resistance to lower energy gradient (Rickenmann 2005) was applied to obtain mean velocities in channel

Table 1 Simulated bedload transport in Malá Ráztoka Brook and Velký Škaredý Brook during 5/2010 flood event

	Malá Ráztoka	Velký Škaredý
Basin area (km^2)	2.2	1.06
Peak discharge (m^3 s^{-1})	4	2
Peak bedload transport intensity (m^3 h^{-1})	10–40	25–50
Total bedload transport volume (m^3)	370–860	500–1,240

cross-sections. Limited erosional depth and estimation of direct sediment inputs from sediment sources (i.e. bank failures and hollows) during the flood event were also included in the simulations. Bedload transport equations of Rickenmann (2001) and Bathurst et al. (1987) were used to calculate bedload discharge. Table 1 shows resulting ranges of values of simulated bedload transport discharge and total bedload transport volumes during 5/2010 flood for selected channel cross-sections of Malá Ráztoka Brook (sediment supply limited) and Velký Škaredý Brook (transport limited). It implies that bedload transport intensity in small watersheds is dependent rather on direct sediment supply than on absolute value of peak discharges. The TOMSED model calculated lower values of maximal bedload transport intensity and total volumes of transported material for supply limited stream despite the occurrence of twofold peak discharge in the stream outlet.

The reconstruction of bedload transport in Austrian Alpine streams with slightly larger basin areas (6–10 km^2) and much higher peak discharges (up to 25 m^3 s^{-1}) has been done by Chiari and Rickenmann (2011) by means of the SETRAC numerical model (Rickenmann et al. 2006), a predecessor of the TOMSED model. They indicated much higher intensity of bedload transport with total volumes of transported sediments 16,000 and 33,000 m^3 during flood events than those simulated in our case (maximum 1,240 m^3 at some cross-sections of transport-limited Velký Škaredý Brook). Such a comparison is necessary for the understanding of contemporary management of local torrents, because the approach of torrential check dams traditionally used in local streams comes from the Austrian Alps. Relatively small volumes of potentially transported material together with low critical conditions for the beginning of bedload transport make channels predisposed by flysch lithology very prone to accelerated incision. Also, oversized check dams and bank stabilisation strongly contribute to such trends in many of local channels.

3 Contemporarily Watershed Management and Its Effect on Stream Hydromorphology

Since permafrost degradation in the early Holocene, forest cover has gradually developed in the whole area of Czech Carpathians. At the beginning of 16th century, the Wallachian colonisation brought the deforestation even of the ridge parts of the mountains and the establishment of the large grasslands for pastoral farming. Together with higher precipitation during Little Ice Age, this led to an increase in sediment delivery into stream segments, higher activity of debris flows in the steepest channel gradients, accumulative tendencies on alluvial cones and some of lower gradient gravel-bed streams were transformed from single-thread pattern into anabranching pattern (Šilhán and Pánek 2007; Wistuba and Sady 2011; Škarpich et al. 2013). Experience with such tendencies which were naturally accompanied by higher intensity of bedload transport at that time, probably led to fast adoption of Alpine torrential check dams since the beginning of 20th century. Nevertheless, approximately at the same time pastoral farming gradually declined and major parts of the Czech Carpathians were covered by spruce and beech agricultural forests again. Decrease in sediment supply caused by afforestation and, on the other hand, construction of channel stabilisation works resulted in accelerated incision of most local channel reaches during 20th century. Similar tendencies were also observed in lower gradient gravel-bed rivers in the forefields of the Western Carpathians (e.g. Lach and Wyżga 2002; Wyżga 2008; Škarpich et al. 2013). Mentioned changes in sediment supply and watershed management of headwaters is one of the important factors that trigger accelerated incision of these rivers.

Check dams and their maintenance together with bank stabilisation works still prevail in the contemporary management of local high-gradient channels, although some modern hydraulic rapid structures begin to appear. Check dams and road sluices act as barriers in sediment transport causing the discontinuity in sediment flux in the local longitudinal stream profiles. The accelerated incision and coarsening of bed material are usually observed downstream of these objects (Škarpich et al. 2010). In relation to contemporary watershed management, one should note again proneness of local channels to erosional processes due to low flow resistance connected with low critical conditions for the commencement of bedload transport and relatively small amounts of potentially transported material, as we documented in the previous chapter. Figures 3 and 4 show the situation during 5/2010 flood event (ca Q_{20}) when an undersized sluice was jammed by coarse material as it was unable to transport coarse fraction of sediments during flood culmination (Fig. 3). At the same time, the channel incised significantly downstream the sluice partially destroying the sluice due to backward erosion (Fig. 4). Gravel extraction from the upstream channel-reaches and bank stabilisation works downstream the sluice started immediately after the flood event.

A similar role is played by oversized concrete check dams, especially in headwater streams with recent limited sediment-supply conditions and bedrock outcrops occurrence in channel bed. Due to this deficiency in potential sediment sources, accelerated incision trends are usually observed downstream the constructed dams. In case of the presence of soft claystone bedrock lithology in downstream channel-reaches, incision continues into this bedrock and many difficulties arise concerning the stabilisation of the channel-reaches (Fig. 5).

Large woody debris has for a long time been recognised as an additional bed roughness element affecting sediment transport dynamics in small steep streams when individual logs act as steps in the stream longitudinal profile. These steps are

Fig. 3 A road sluice acting as a barrier to bedload transport during 5/2010 flood event due to its jamming (Lubina watershed). The *arrow* shows flow direction

Fig. 4 Accelerated incision downstream the sluice during 5/2010 flood event (Lubina watershed)

Fig. 5 A channel-reach incised 6 m into claystone bedrock downstream check dams in the Malá Ráztoka watershed (the upstream view)

important energy dissipators in nature streams, when they significantly reduce bedload transport intensity during high flow events. Presence of woody debris also improves stabilisation of sediment accumulations in channels (e.g. Gomi et al. 2003; Faustini and Jones 2003). Nevertheless, local channels are systematically cleaned from woody debris by forest management, especially at well-accessible sites related to lower channel gradients ≤0.1 m/m. The removal of large woody debris by local people resulted in a significant decrease in individual logs (minimal size 0.5 × 0.1 m) in active channels with the increase in watershed area when we tested 102 channel-reaches in the Czech part of the Western Carpathians (Fig. 6). A lack of woody debris in channels causes further decrease in total flow resistance, which probably also contributes to the acceleration of bed degradation. Moreover, ecological potential of woody debris in channels should not be neglected.

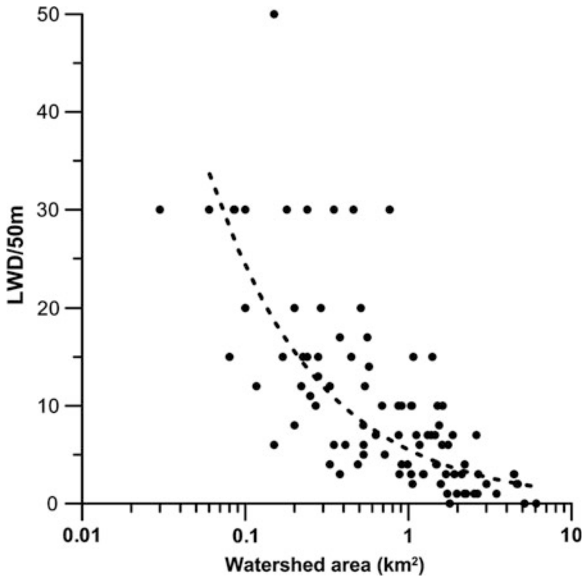


Fig. 6 Relationship between the amount of large woody debris on 50 m distance of the channel and watershed area

4 Recommendation for the Management of Flysch Headwaters

The following recommendations are based on our research and observations in flysch-predisposed watersheds; however, they can be applied in any of small channels predisposed by relatively soft lithology prone to accelerated erosional processes. The previous chapter demonstrated some unsuitable examples of watershed management in local headwaters. Classic torrent control works are probably still a functioning solution for channel-reaches in densely built-up areas to protect the property and lives during high-magnitude flood events. Nevertheless, modern rapid hydraulic structures should be adopted widely due to their better effect for aquatic habitats. Bank stabilisation works should only be implemented in sites of necessary protection (e.g. bridge constructions, road and railway communications). Such stabilisation works are contemporarily widely used in stream longitudinal profiles without reasonable arguments. Moreover, decreased connectivity in sediment fluxes between channels and adjacent hillslopes naturally contributes to the acceleration of erosional processes in local streams.

Experiments with stream bed stabilisation using artificial step-pool sequences have been performed since 1990s in the torrents of the Italian Alps. Such structures imitate natural step-pool morphology sensu Montgomery and Buffington (1997) and there is an effort to replace traditional concrete check dams with these low boulder check dams. Artificial step-pool structures are built for the safety degree for $Q_{20–30}$ and provide sufficient connectivity for aquatic organisms and sediment transport (Lenzi 2002). Restored step-pool channels are increasingly common also

Fig. 7 Artificial step-pool sequence in the Malá Ráztoka Brook

in the USA (Chin and Phillips 2007; Chin et al. 2009). Since May 2013 a similar experiment has been taken place in flysch Carpathians (the Malá Ráztoka watershed) through a construction of 13 step-pools from local sandstone boulders (Fig. 7). Key boulders in each step exceed a diameter of 0.4 m; this corresponds to the largest grain-size fraction transported during flood 5/2010 (Q_{20}) in the examined channel (Galia and Hradecký 2012). The geometry of artificial step-pool sequence is related to the relationship obtained for natural step-pool sequences $1 < (H/L/S) < 2$, where H is step height, L means distance between crests of steps and S means channel gradient (e.g. Lenzi 2001; Wohl and Wilcox 2005). The channel gradient of experimental channel-reach varies between 0.08 and 0.12 m/m; bankfull width is about 4 m and the height of artificial steps varies between 0.4 and 0.6 m. One hundred limestone grains of diameters $20 < d_i < 40$ mm were placed in the most upstream pool. They represented fine grain-size fraction and enabled us to observe the dynamics of this fine fraction through an artificial step-pool channel-reach. During two months, until June 2013, ca 2/3 of limestone grains were transported downstream from the uppermost pool. The longest travel distance was three step-pool sequences during ordinary Qa–Q_1 flows. That implies that these structures provide efficient connection of fine grain-size fractions during lower flows. It is planned to continue the experiment in order to monitor the stability of steps during higher flow events. Erosional and depositional trends in constructed step-pool sequences will be evaluated within repeated geodetic measurement.

We expect that artificial step-pools can play an important role in mountain stream restoration. They can be an alternation to larger concrete dams in areas with a lower degree of flood protection for their low cost demands and functional connectivity for aquatic organism migrations and sediment transport. Moreover, accelerated incision as it is observed downstream of larger check dams is not expected if these much lower boulder dams are used.

There is also great potential for the application of individual logs in order to enhance aquatic environment. The topic of clearing the channels of woody debris by local people has been discussed in the previous chapter. We suppose that the presence of logs in headwater channels with relatively fine bed sediments may lead in view of sediment dynamics to (i) the deceleration of erosional processes in headwater streams by an increase in total bed roughness, although steep mountain channels are undoubtedly understood as erosion and transport segments of the fluvial network, and to (ii) a significant decrease in bedload transport intensity in specific channel-reaches with relatively unlimited sediment sources due to stabilisation of channel accumulations by woody debris.

A detailed evaluation of active sediment sources is the main goal for the determination of quality watershed management not only in view of heterogeneous geologic predispositions. Even neighbouring small headwater basins can significantly differ in the criteria of potential sediment supply during high flow events (i.e. their magnitude and grain-size parameters) consequently related to the intensity and total volumes of bedload transport. Streams with a wide occurrence of relatively resistant sandstone bedrock outcrops in beds have much smaller potential for the transport of larger volumes of sediments than streams based in soft claystone formations usually accompanied by shallow landslides and bank failures. In addition, the lithology of bed sediments strongly affects the development of bed resistance formations (i.e. steps or large boulders). Predisposition to debris-flow scours should carefully be assessed at channel gradients exceeding 0.20 m/m since this channel gradient is widely understood as a boundary between prevailing fluvial and colluvial processes (e.g. Gomi et al. 2003; Šilhán and Pánek 2010).

Acknowledgments This research was supported by the project of the University of Ostrava Foundation SGS04/PřF/2012 and SGS15/PřF/2013. Thanks are also extended to Monika Hradecká for the English language revision.

References

Bathurst JC (1987) Critical conditions for bed material movement in steep, boulder-bed streams. In: Erosion and sedimentation in the Pacific rim. Proceedings of the corvallis symposium, August 1987, IAHS Publ. no. 165, pp 309–318

Bathurst JC, Graf W, Cao H (1987) Bed load discharge equations for steep mountain rivers. In: Thorne CR, Bathurst JC, Hey RD (eds) Gravel-bed rivers. Wiley, New York, pp 453–477

Chiari M, Rickenmann D (2011) Back-calculation of bedload transport in steep channels with a numerical model. Earth Surf Proc Land 36:805–815

Chin A, Phillips JD (2007) The self-organization of step-pools in mountain streams. Geomorphology 83:346–358

Chin A, Anderson S, Collison A, Ellis-Sugai BJ, Haltiner JP, Hogervorst JB, Kondolf GM, O'Hirok LS, Purcell AH, Riley AL, Wohl E (2009) Linking theory and practice for restoration of step-pool streams. Environ Manage 43(4):645–661. doi:10.1007/s00267-008-9171-x

Faustini JM, Jones JA (2003) Influence of large woody debris on channel morphology and dynamics in steep, boulder-rich mountain streams, western Cascades, Oregon. Geomorphology 51:187–205

Friedl K, Chiari M (2011) A one-dimensional transport model for steep slopes. Manual tomsed version 0.1 beta. http://www.bedload.at. Accessed 11 Sep 2012

Galia T, Hradecký J (2011) Bedload transport and morphological effects of high-magnitude floods in small headwater streams—Moravskoslezské Beskydy Mts. (Czech Republic). J Hydrol Hydromechanics 59(4):238–250

Galia T, Hradecký J (2012) Critical conditions for beginning of coarse sediment transport in torrents of Moravskoslezské Beskydy Mts. (Western Carpathians). Carpathian J Earth Environ Sci 7(4):5–14

Gomi T, Sidle RC, Woodsmith RD, Bryant MD (2003) Characteristics of channel steps and reach morphology in headwater streams, southeast Alaska. Geomorphology 51:225–242

Hassan MA, Zimmermann AE (2012) Channels response and recovery to changes in sediment supply. In: Church M, Pascale MB, Roy AG (eds) Gravel-bed rivers: processes, tools, environments. Wiley, New York, pp 464–473

Hradecký J, Pánek T (2008) Deep-seated gravitational slope deformations and their influence on consequent mass movements (case studies from the highest part of the Czech Carpathians). Nat Hazards 45:235–253

Lach J, Wyżga B (2002) Channel incision and flow increase of the upper Wisłoka River, southern Poland, subsequent to the reafforestation of its catchment. Earth Surf Proc Land 27:445–462

Lenzi MA (2001) Step-pool evolution in the Rio Cordon, northeastern Italy. Earth Surf Proc Land 26:991–1008

Lenzi MA (2002) Stream bed stabilization using boulder check dams that mimic step-pool morphology features in Northern Italy. Geomorphology 45:243–260

Lenzi MA, Mao L, Comiti F (2006) When does bedload transport begin in steep boulder-bed stream? Hydrol Process 20:3517–3533

Mao L, Uyttendaele GP, Iroumé A, Lenzi MA (2008) Field based analysis of sediment entrainment in two high gradient streams located in Alpine and Andine environments. Geomorphology 93:368–383

Montgomery DR, Buffington JM (1997) Channel reach morphology in mountain drainage basins. Geol Soc Am Bull 109:596–611

Pánek T, Tábořík P, Klimeš J, Komárková V, Hradecký J, Šťastný M (2011) Deep-seated gravitational slope deformations in the highest parts of the Czech Flysch Carpathians: evolutionary model based on kinematic analysis, electrical imaging and trenching. Geomorphology 129:92–112

Petit F, Gob F, Houbrechts G, Assani AA (2005) Critical stream power in gravel-bed rivers. Geomorphology 69:92–101

Recking A, Leduc P, Liébault F, Church M (2012) A field investigation of the influence of sediment supply on step-pool morphology and stability. Geomorphology 139–140:53–66

Rickenmann D (2001) Comparison of bed load transport in torrents and gravel bed streams. Water Resour Res 37(12):3295–3305

Rickenmann D (2005) Geschiebetransport bei steilen Gefällen. Mitteilung 190 der Versuchsanstalt für Wasserbau. Hydrologie und Glaziologie, ETH Zurich, pp 107–119

Rickenmann D, Chiari M, Friedl K (2006) SETRAC—a sediment routing model for steep torrent channels. In: Ferreira R, Leal EAJ, Cardoso A (eds) River Flow 2006, vol 1. Taylor & Francis, London, pp 843–852

Šilhán K, Pánek T (2007) Blokovobahenní proudy v masivu Smrku (Moravskoslezské Beskydy; Česká republika). Geomorphologia Slovaca et Bohemica 1:56–64 (in Czech)

Šilhán K, Pánek T (2010) Fossil and recent debris flows in medium-high mountains (Moravskoslezské Beskydy Mts, Czech Republic). Geomorphology 124:238–249

Škarpich V, Galia T, Hradecký J, Peč J (2010) Identifikace (dis)konektivit vodních toků za využití makrogranulometrické analýzy korytových sedimentů (Moravskoslezské Beskydy). Geologické výzkumy na Moravě a ve Slezsku v r. 2010:199–204 (in Czech)

Škarpich V, Hradecký J, Dušek R (2013) Complex transformation of the geomorphic regime of channels in the forefield of the Moravskoslezské Beskydy Mts.: Case study of the Morávka River (Czech Republic). Catena 111:25–40, http://dx.doi.org/10.1016/j.catena.2013.06.028

Whiting PJ, King JG (2003) Surface particle sizes of armoured gravel bed streambeds: effects of supply and hydraulics. Earth Surf Proc Land 28:1459–1471

Williams GP (1983) Paleohydrological methods and some examples from Swedish fluvial environments. Geogr Ann 65A:227–243

Wistuba M, Sady A (2011) Walachian colonisation on mountain ridge recorded in alluvia of the Škorňanský stream (Moravskoslezské Beskydy, Western Carpathians, Czech Republic). Geomorphologia Slovaca et Bohemica 11(1):18–27

Wohl EE, Wilcox A (2005) Channel geometry of mountain streams in New Zealand. J Hydrol 300:252–266

Wyżga B (ed) (2008) Stan środowiska rzek poludniowej Polski i możliwości jego poprawywybrane aspekty. Polska akademia nauk, Instytut ochrony przirody, Krakow. ISBN 978-83-61191-12-4 (in Polish)

Yager EM, Kirchner JW, Dietrich WE (2007) Calculating bed load transport in steep boulder bed channels. Water Resour Res 43:W07418

Aspects of Sediment Transport in Single-Thread and Anabranching River Channels in Flysch Carpathians (A Case Study from the Czech Republic)

Václav Škarpich, Tomáš Galia and Jan Hradecký

Abstract Present-day state of channels shows a tendency towards the acceleration of processes linked with river bed lowering. Focusing on the Ostravice River and Bečva River basins in the Czech part of flysch Carpathians, the paper summarizes the results of energy potential aspects of contemporary Carpathian river channels. The study has been conducted with the use of the Bedload Assessment for Gravel-bed Streams (BAGS) spreadsheet-based program and unit stream power formula. Presented results show potential values of sediment transport with the identification of erosion and accumulation processes in channels. Selected channel cross-profiles include preserved gravel-bed reaches with an anabranching pattern as well as transformed reaches with accelerated deep erosion and occurrence of a single bedrock channel. The modelling shows potential transport trends in relation with the morphology of the channel. The results can be used to distinguish the reaches with erosion or accumulation trends. The modelling on cross-profiles with a low rate of fluvial erosion (anabranching channel pattern) shows a decrease in potential sediment transport. It is caused by increased or decreased (dis)connectivity in the longitudinal profile of the fluvial (dis)continuum system. It is influenced both by flow diversion through sub-channels and a decrease in sediment transport capacity. We define this area of active channel as a zone of reduced growth in transport capacity. In contrast, reaches with a single channel pattern show higher values of potential sediment transport caused by the absence of this zone of reduced growth in transport capacity. The results may be applied in form of a conceptual scheme to improve the management of local watersheds.

V. Škarpich (✉) · T. Galia · J. Hradecký
Department of Physical Geography and Geoecology, Faculty of Science,
University of Ostrava, Chittussiho 10, 710 00 Ostrava, Czech Republic
e-mail: vaclav.skarpich@osu.cz

T. Galia
e-mail: tomas.galia@osu.cz

J. Hradecký
e-mail: jan.hradecky@osu.cz

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_3

1 Introduction

Over the last century many fluvial systems have significantly been affected by changes (Kondolf 1997; Rinaldi and Simon 1998; Zawiejska and Wyżga 2010; Fryirs et al. 2007) both of the character of human interventions and the effect of climate fluctuation (Liébault and Piégay 2001). Several studies showing varied channel adjustment have analysed the response of rivers to human impact (Kondolf 1997; Rinaldi and Simon 1998; Liébault and Piégay 2001; Zawiejska and Wyżga 2010; Fryirs et al. 2007). River changes cause direct modification of fluvial processes within channels which induces adjustment in sediment transport characteristics depending on channel flow character (Wyżga 1993). In wider channels, the flowing water splits into in-bank and overbank flow, whereas in narrow and incised channels, the flowing water is transferred entirely within the channels. This incision along with narrowing is caused by flow concentration and increase in the energy potential of water flowing in channels. Moreover, it is intensified following accelerated river bed degradation (Wyżga 1993; Zawiejska and Wyżga 2010; Galia et al. 2012).

2 Background of Applied Equations

The research has been conducted using the Bedload Assessment for Gravel-bed Streams (BAGS) spreadsheet-based program. The extension calculates transport stage and potential of bed load transport with the use of six equations developed specially for gravel-bed rivers (Pitlick et al. 2009). In this case study the Parker (1990) surface-based equation was used because of lack of subsurface grain-size data. Several channel cross-profiles in the Ostravice River and the Bečva River basins were selected in order to create transport stage analysis. Transport stage is simply a ratio of bed shear stress to critical shear stress:

$$\varphi = \tau/\tau_r, \tag{1}$$

where τ is defined as bed shear stress and τ_r as critical shear stress. In fact, transport stage with higher values shows characteristics of channels with higher energy potential.

Similarly, available power supply or time rate of energy supply in channel can be evaluated by means of the analysis of unit stream power (USP) (according to Bagnold 1966; Wyżga 2001; Zawiejska and Wyżga 2010). USP (W m^{-2}) is characterized as the rate of energy dissipation against the bed and banks of a stream per unit downstream length and per unit channel width. It is given by the equation:

$$\omega = (\rho * g * Q * S)/b, \tag{2}$$

where ρ is the density of water (1,000 kg m^{-3}), g is acceleration due to gravity (9.8 m s^{-2}), Q is discharge (m^{3} s^{-1}), S is the channel slope and b is the channel width (Bagnold 1966). Channel gradients were defined for all cross-profiles of the studied stream longitudinal profile when we considered altitudes 50 m upstream and downstream from selected cross-profiles. Channel widths were defined for all cross-profiles from BAGS simulation of water level of the N_{yo} (N year occurrence) discharge. Selected study area includes preserved gravel-bed reaches with an anabranching pattern, transformed reaches with accumulation processes of sediments or accelerated deep erosion and incised single bedrock channels.

3 Regional Setting of the Rivers of Czech Flysch Carpathians

At the end of the 19th century, the rivers draining the Czech part of flysch Carpathians were characterised by an anabranching channel pattern in the piedmont zone. Regional geological settings predispose large sediment supply into the river systems (Menčík et al. 1983). In recent historical times, intensive diverse human interventions in basins and river channels have caused river system changes. Main changes observed in the rivers of Czech flysch Carpathians comprise channel narrowing (see Fig. 1) and incision (see Fig. 2). Such tendencies became highly intensive especially in the 20th century (Hradecký and Škarpich 2009; Škarpich et al. 2013). For example, the highest rate of incision observed in the channels of

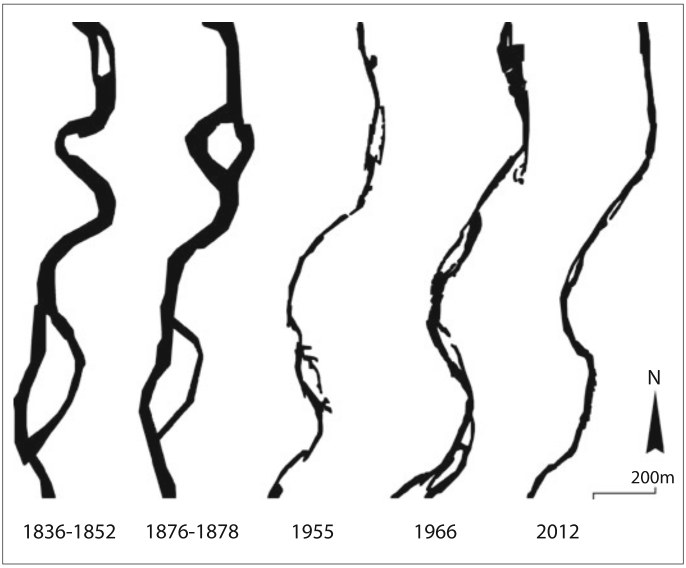


Fig. 1 The development of the Ostravice river channel pattern between 27.7 and 28.6 r km in the period of 1836–2012

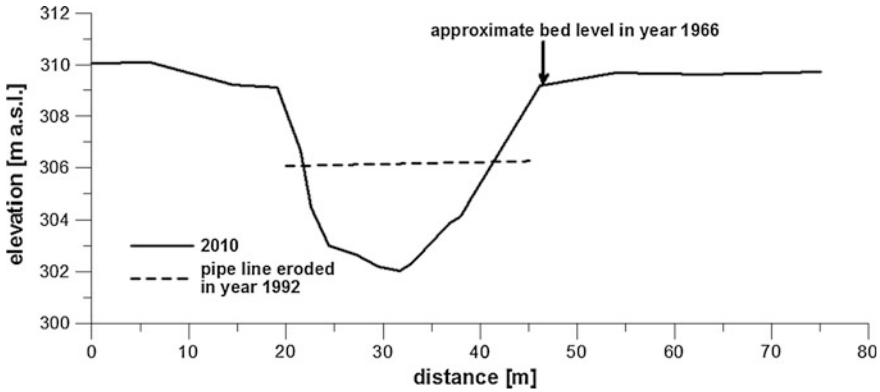


Fig. 2 Cross-profile at 2.31 r km in the area of massive incision of the Morávka R. (*left* tributary of the Ostravice R.) with the identification of incision starting in the year 1992

Czech flysch Carpathians was recorded in the Morávka River basin (the Morávka River is the main tributary of the Ostravice River)—ca 16 cm per year in the last 40–50 years (Škarpich et al. 2013). Mean value of incision of the channels in the piedmont zone in the whole Ostravice River basin varies from 3 to 6 cm per year.

Main causes of the incision and narrowing in the rivers of Czech flysch Carpathians were identified as (i) a decrease in sediment supply to the channels (related to land use and land cover changes in the study area as well as man-made channel bank stabilisation works affecting lateral connectivity within the river system) (Škarpich et al. 2011) and (ii) a high number of barriers (dams or weirs) influencing sediment transport through the river system in the longitudinal profile (Hradecký and Škarpich 2009; Škarpich et al. 2013). One of the main factors is also connected with the geological predisposition of flysch lithology to erosion in the bedrock of channels (Hradecký 2002; Škarpich et al. 2013).

4 Transport Characteristics of Anabranching and Single-Thread Channels

From the point of view of the energy potential of channels, we identify a big difference between anabranching and single-thread channels. Trends in sediment transport represent a good instrument for the identification of (dis)connected sediment transport reaches along the longitudinal profile. The analysis (see Figs. 3 and 4) shows transport trends in relation to the water depth. The modelling at cross-profiles with a high rate of fluvial erosion shows an increase in sediment transport. Figure 3 indicates energy potential (from the point of view of transport stage analysis) of a

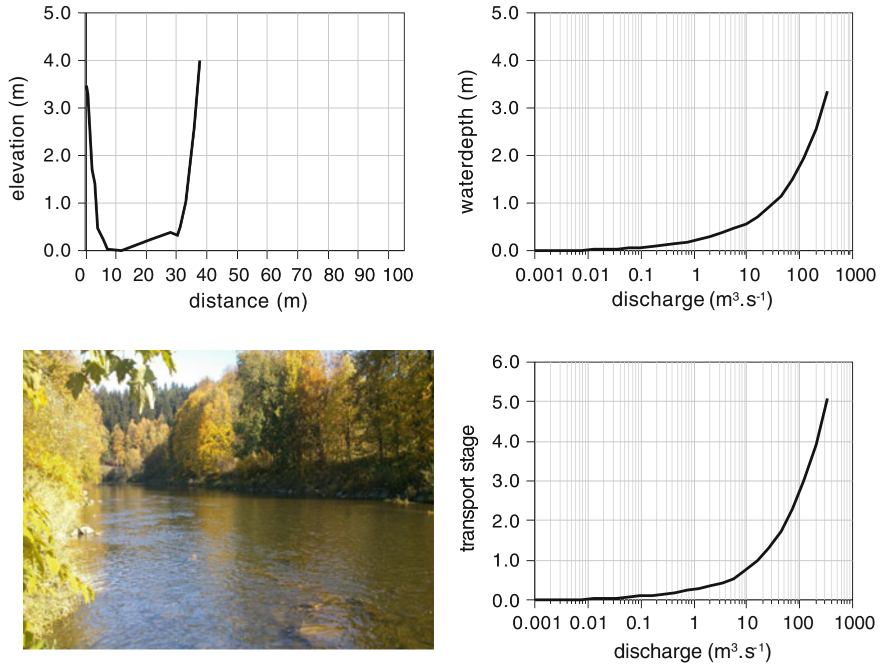


Fig. 3 Cross-profile, water depth during discharge and transport stage of the single-thread channel of the Bečva R. in the locality VB2 (for location see Fig. 6)

single-thread channel (the D_{50} of the surface bed layer particles in the studied profile was 24 mm). An increase in the growth of transport stage represents an increase in connectivity in the sense of higher energy potential for the transport of sediments. In case of sediment deficit, this energy is dissipated in erosion processes affecting the channel bed.

By contrast, the analysis of sediment transport in an anabranching channel (Fig. 4) revealed lower transport values (the D_{50} of the surface bed layer particles in this studied profile was 55 mm). Consequently, these values determine that energy dissipated on channel bed is lower and erosion processes play an unimportant role. This state is caused by flow diversion into sub-channels of the anabranching channel pattern. Generally, a decrease in the growth of transport stage represents a decrease in connectivity in the sense of the lower energy potential for the transport of sediments. We defined this area of flow diversion as a zone of reduced growth in transport capacity.

In conclusion, the analysis of an increase/decrease in the curve of transport stage indicates sediment transport potential and predisposition to erosion processes affecting the channels (especially channel incision).

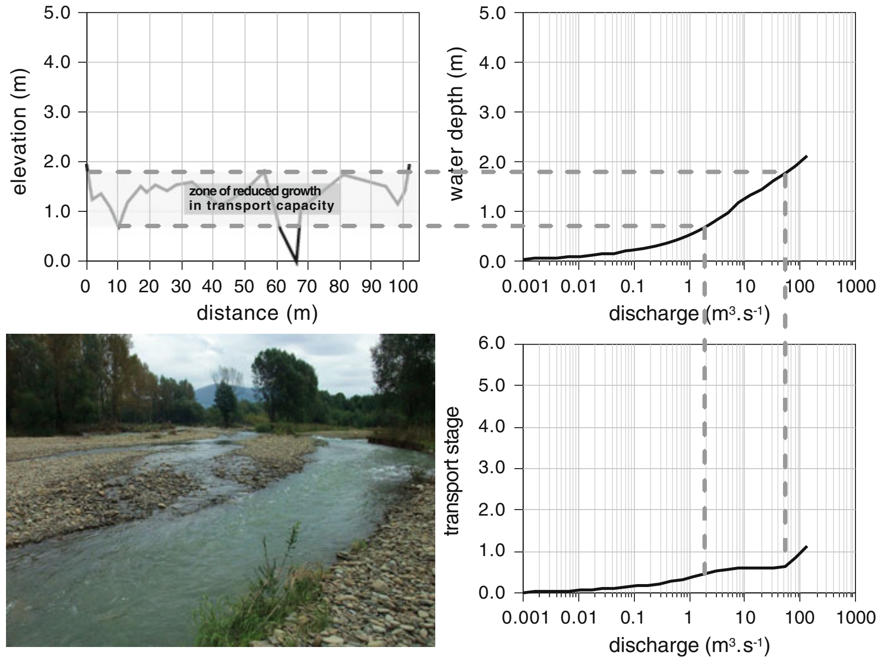


Fig. 4 Cross-profile, water depth during discharge and transport stage of the anabranching channel of the Morávka R. in the locality Mor3 (for location see Fig. 5)

5 Sediment Transport Aspects of the Channels in Czech Flysch Carpathians

5.1 A Case Study from the Ostravice River Basin

The Ostravice River is a gravel-bed stream of a length of 65.1 km that flows from steeper Moravskoslezské Beskydy Mts to a relatively flat piedmont. The Ostravice River is a right tributary of the Morava River. The drainage area is 827 km^2. The mean annual discharge of the river amounts to 15.5 m^3 s^{-1} at the Ostrava gauging station (for location see Fig. 5) where the basin area is 821 km^2. The main tributaries are the Čeladenka River, the Morávka River and the Lučina River.

Several channel cross-profiles in the Ostravice River (OST1, OST2), the Morávka River (MOR1, MOR2, MOR3, and MOR4) and the Mohelnice River (MOH1) were used in order to compute transport stage (see Table 1). The Mohelnice River is a left tributary of the Morávka River. Selected channel cross-profiles include preserved gravel-bed reaches with an anabranching pattern (MOR2, MOR3) as well as transformed reaches with accelerated deep erosion (MOR1) and the occurrence of a single bedrock channel (MOR4). The OST2 cross-profile shows a slightly accumulative trend with well-developed lateral bars in the river channel, whereas the

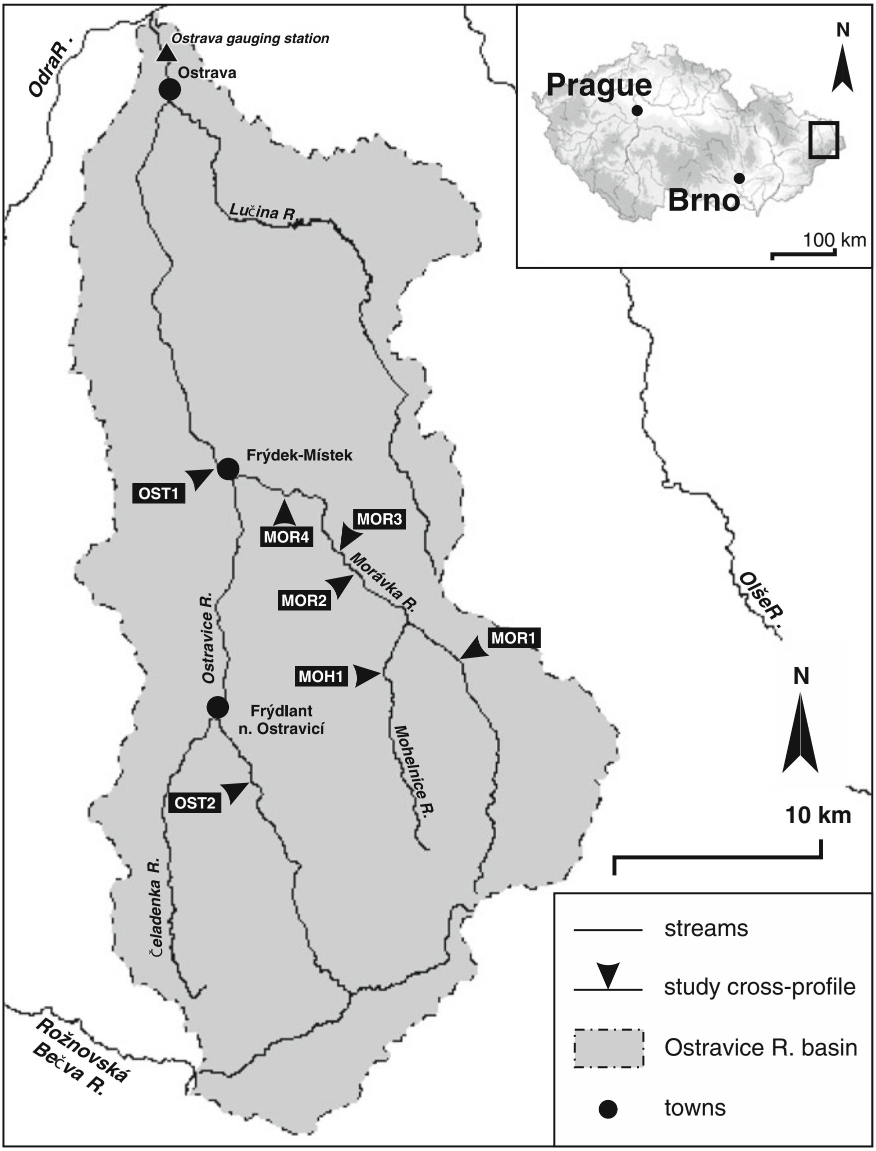


Fig. 5 Localisation of studied cross-profiles in the Ostravice R. basin

MOH1 and OST1 reaches have no significant trend in erosion or deposition. Lateral gravel bars can be found in the river channel (see Table 1, trend of channel development—erosion/accumulation).

The modelling at cross-profiles with a high rate of fluvial erosion (MOR1, MOR4) shows an increase in the values of sediment transport and unit stream

Table 1 Computed transport stage (φ) and unit stream power (ω) for the given discharge N-year occurrence and trend of channel development (prevailing channel processes: *E* erosion or incision, *A* accumulation, *E/A* erosion/accumulation) in the studied cross-profiles in the Ostravice R. basin (see Fig. 5)

Locality	Q_{1yo}		Q_{2yo}		Q_{5yo}		Q_{10yo}		Trend of channel development
	φ	ω (W m^{-2})	φ	ω (W m^{-2})	φ	ω (W m^{-2})	φ	ω (W m^{-2})	
OST1	2.47	185.97	3.19	299.03	3.77	464.93	4.30	598.77	E/A
OST2	1.61	74.54	2.05	114.93	2.62	180.37	3.05	239.54	E/A
MOR1	3.18	281.59	3.95	508.93	4.88	865.45	5.59	1171.58	E
MOR2	1.52	112.17	1.91	182.83	2.40	305.77	2.69	421.43	A
MOR3	0.61	35.91	0.70	58.54	0.94	97.90	1.14	134.93	A
MOR4	3.44	498.86	4.28	813.11	5.37	1359.88	6.28	1874.24	E
MOH1	0.98	106.35	1.28	185.01	1.65	331.24	1.95	478.58	A

power (see Table 1). Relatively high values of these parameters computed for the OST1 reach may indicate high intensity of erosion, however, no intense erosion processes can really be observed there (see Table 1). It can be caused by high sediment delivery from the adjacent slopes of the Moravskoslezské Beskydy Mts. By contrast, reaches with preserved anabranching development (MOR2, MOR3) show significantly lower values of these parameters (see Table 1). Finally, MOH1 and OST2 reaches show similar values of transport stage and unit stream power revealing a balanced trend in erosion and deposition (see Table 1).

5.2 A Case Study from the Bečva River Basin

The Bečva River is a 61.5 km-long stream that flows from the Moravskoslezské Beskydy Mts and through the Javorníky Mts and the Hostýnsko-vsetínská hornatina Mts. It is a left tributary of the Morava River and originally an anabranching gravel-bed stream. The Bečva River is created by two source streams, the northern Rožnovská Bečva River and the southern Vsetínská Bečva River. The drainage area is 1,620 km^2. The mean annual discharge of the river amounts to 17.3 m^3 s^{-1} at the Dluhonice gauging station (for location, see Fig. 6), where the basin area is 1,592 km^2.

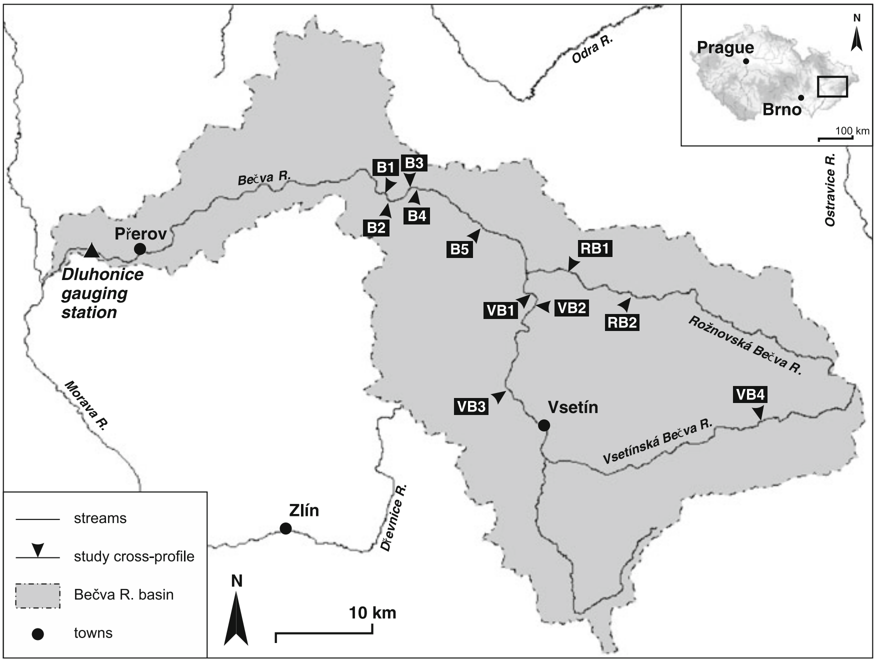


Fig. 6 Localisation of studied cross-profiles in the Bečva R. basin

The modelling at cross-profiles shows a similar condition of incised regulated or anabranching and regulated wide channels to that of the channels in the Ostravice River basin. Cross-profiles with a high rate of fluvial erosion (VB2, VB3, VB4 and RB2) show an increase in the values of transport stage (see Table 2). Relatively high values of the parameters of the unit stream power computed for the VB2, VB3, VB4 and RB2 reaches indicate high energy potential. These channels are incised to the extent of bedrock exposure.

Relatively higher values of the parameters computed for the VB1 and RB1 reaches may indicate higher erosion intensity, however, no intense erosion processes can really be observed there (see Table 2). Main reason is dissipated energy of flowing water in a wider regulated channel which decreases the energy for erosional processes. However, no exposed bedrock is visible in these channels.

On the contrary, reaches with an anabranching channel pattern (B3, B4 and B5) show significantly lower values of these parameters (see Table 2). The anabranching channel pattern was restored by floods in the year 1997. Bank stabilisation and channelisation works were destroyed by this flood and the channel was left in a state of spontaneous development. A contemporary decrease in transport stage is caused by flow diversion into the sub-channels of the contemporary anabranching channel pattern. B1 and B2 reaches (with a regulated wide channel) show similar values of the transport stage revealing a balanced trend in erosion and deposition (see Table 2). No intensive erosion processes can really be observed, yet accumulated lateral gravel bars can be found in the river channel (see Table 2, trend of channel development—erosion/accumulation).

Table 2 Computed transport stage for the given discharge N-year occurrence and trend of channel development (prevailing channel processes: *E* erosion or incision, *A* accumulation, *E*/*A* erosion/accumulation) of selected cross-profiles in the Bečva R. basin (see Fig. 6)

Locality	Transport stage			Trend of channel development
	Q_{1yo}	Q_{5yo}	Q_{10yo}	
B1	0.71	1.09	1.23	E/A
B2	1.08	1.67	1.89	E/A
B3—Černotín	0.82	1.26	1.43	A
B4—Černotín	0.68	1.04	1.18	A
B5—Choryně	0.36	0.56	0.63	A
VB1—Jarcová	1.34	1.82	2.01	E/A
VB2—Jarcová	3.37	4.61	5.08	E
VB3—Bobrky	2.61	3.67	4.03	E
VB4—Velké Karlovice	1.73	2.35	2.68	E
RB1—Hrachovec	1.33	2.02	2.30	E/A
RB2—Rožnov	1.86	2.71	3.12	E

6 Conclusion

Beskydian channels have recently been experiencing geomorphic transformation as a result of human intervention. Consequently, we observe prevailing channel reaches with erosion processes (especially incision of channels), whereas preserved anabranching river systems are rare. One of the main side-effects is prevailing hungry water effect (sensu Kondolf 1997). This state is caused by the absence of sediment load and the consequences of land use changes (in the sense of sediment stabilisation through the increase in forest cover), channelisation of river reaches and the impact of dams.

Another reason is an increase in the energy potential of flowing water which makes erosion processes intensive. A similar theoretical approach is described by Fryirs et al. (2007) who defined some reaches as boosters. These affect the acceleration of sediment transport and transport capacity of channels. In river reaches where sediments are missing the energy of flowing water is expended on accelerated erosion processes.

We have attempted to predict trends in erosion and deposition in local river channels on the basis of energy potential prediction. Modelling at cross-profiles with a high rate of fluvial erosion shows an increase in the values of transport stage and unit stream power. Trends in these parameters represent a good instrument for the identification of (dis)connected sediment transport reaches along the longitudinal profile and the detection of accelerated erosion or accumulation.

Present incision processes of the channels are accelerated by the synergy of local geological conditions (according to Škarpich et al. 2013) and increased transport capacity of rivers. Similar conditions were observed by Wyżga (1993) or (2001) and Galia et al. (2012).

Acknowledgments This research was supported by the project of the University of Ostrava Foundation SGS15/PřF/2013. Thanks are extended to Monika Hradecká for the English style corrections.

References

Bagnold RA (1966) An approach to the sediment transport problem from general physics. Geol Surv Prof Pap 422-I 1:231–291, Washington

Fryirs K, Brierley GJ, Preston NJ, Spencer J (2007) Catchment-scale (dis)connectivity in sediment flux in the upper hunter catchment, New South Wales, Australia. Geomorphology 84:297–316. doi:10.1016/j.geomorph.2006.01.044

Galia T, Škarpich V, Hradecký J (2012) Bedload sediment transport in connection with the geomorphological transition of gravel-bed streams in the Moravskoslezské Beskydy Mountains. Geografie 117:95–109 (in Czech with English summary)

Hradecký J (2002) Contribution to the morphodynamic chronology of Beskydian rivers (Morávka River 1780–1997). In: Kirchner K, Roštínský P (eds) Book of Abstracts 1. Faculty of Science Masaryk University of Brno, Brno

Hradecký J, Škarpich V (2009) General Scheme of the Western Carpathian Stream Channel Behaviour. In: Hartvich F (ed) Book of Abstracts 8. Czech Association of Geomorphologists, Kašperské Hory

Kondolf GM (1997) Hungry water: effects of dams and gravel mining on river channels. Environ Manag 21:533–551

Liébault F, Piégay H (2001) Assessment of channel changes due to long-term bedload supply decrease, Roubion river, France. Geomorphology 36:167–186. doi:10.1016/S0169-555X(00)00044-1

Menčík E, Adamová M, Dvořák J, Dudek A, Jetel J, Jurková A, Hanzlíková E, Houša V, Peslová H, Rybářová L, Šmíd B, Šebesta J, Tyřáček J, Vašíček Z (1983) Geologie Moravskoslezských Beskyd a Podbeskydské pahorkatiny (Geology of the Moravskoslezské Beskydy Mts and Podbeskydská pahorkatina Hilly land). ÚÚV v nakladatelství ČSAV, Praha. (in Czech, with English summary)

Parker G (1990) Surface-based bedload transport relation for gravel rivers. J Hydraul Res 28:417–436

Pitlick J, Cui Y, Wilcock P (2009) Manual for computing bed load transport using BAGS (Bedload Assessment for Gravel-bed Streams) software. General Technical Report RMRS-GTR-223, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO

Rinaldi M, Simon A (1998) Bed-level adjustment in the Arno river, central Italy. Geomorphology 22:57–71. doi:10.1016/S0169-555X(97)00054-8

Škarpich V, Hradecký J, Tábořík P (2011) Structure and genesis of the Quaternary filling of the Slavíč River valley (Moravskoslezské Beskydy Mts., Czech Republic). Moravian Geogr Rep 19:30–38

Škarpich V, Hradecký J, Dušek R (2013) Complex transformation of the geomorphic regime of channels in the forefield of the Moravskoslezské Beskydy Mts.: case study of the Morávka river (Czech Republic). Catena 111:25–40. doi:10.1016/j.catena.2013.06.028

Wyżga B (1993) River response to channel regulation: case study of the Raba river, Carpathians, Poland. Earth Surf Proc Land 18:541–556. doi:10.1002/esp.3290180607

Wyżga B (2001) Impact of the channelization-induced incision of the Skawa and Wisłoka rivers, Southern Poland, on the conditions of overbank deposition. Regulated Rivers Res Manag 17:85–100. doi:10.1002/1099-1646(200101/02)17:1<85:AID-RRR605>3.0.CO;2-U

Zawiejska J, Wyżga B (2010) Twentieth-century channel change on the Dunajec river, southern Poland: patterns, causes and controls. Geomorphology 117:234–246. doi:10.1016/j.geomorph.2009.01.014

Sediment Transport Processes Related to the Operation of a Rapid Hydraulic Structure (Boulder Ramp) in a Mountain Stream Channel: A Polish Carpathian Example

Karol Plesiński, Artur Radecki-Pawlik and Bartłomiej Wyżga

Abstract Rapid hydraulic structures—RHS—(called also boulder ramps) are modern, environment-friendly grade-control structures which mimic natural riffles and do not disturb longitudinal continuity of the stream for fish and benthic invertebrates. Due to the reduction of hydraulic gradient and backwater effect, such hydraulic structures change the pattern of sediment transport and deposition in the channel, facilitating persistence of alluvial streambed and the formation of gravel bars upstream and downstream of the structures. This is of key importance for preserving habitats for benthic invertebrates and the spawning ground of lithophilic fish if a stream has to be channelized. At the same time, properly designed rapid hydraulic structures must allow efficient transfer of sediment flux through their apron, helping to clean the structures of gravel and preventing their clogging. This study deals with observations and modeling of sediment transport in the vicinity of a rapid hydraulic structure in a mountainous gravel-bed channel. The study aims to: (i) show the effects of RHS on sediment transported along a stream channel, and (ii) to evaluate the performance of CCHE2D model in predicting sediment phenomena along the stream with rapid hydraulic structures. The studied structure is located in Porębianka Stream draining a flysch catchment in the Polish Carpathians. We measured and calculated hydraulic parameters characterizing the flow on and in the vicinity of the structure, such as velocity, dynamic velocity, shear stress, Froude number, Reynolds number and friction coefficient. The knowledge of those parameters allowed us, at the same time, to calculate sediment transport in the region of the structure using BAGS model for the Parker transport formula and parallel modeled the sediment transport with the CCHE2D model. The results show

K. Plesiński (✉) · A. Radecki-Pawlik
Department of Hydraulic Engineering and Geotechnics, University of Agriculture in Kraków, Kraków, Poland
e-mail: k.plesinski@ur.krakow.pl

B. Wyżga
Institute of Nature Conservation, Polish Academy of Sciences, Kraków, Poland

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_4

how the hydraulic structure (enabling the migration of fish and benthic invertebrates), operates in terms of sediment transport processes (basically, giving the answer to the question: what is the influence of RHS on sediment transport) which form the channel morphology in its vicinity. In that context the CCHE2D model is discussed with its advantages and impediments.

Keywords Rapid hydraulic structure · Boulder ramp · Sediment transport · Mountain stream · Polish carpathians

Notation

x,y	Horizontal Cartesian coordinates (m),
U, V	Depth-averaged flow velocities in x: and y-directions (m s^{-1}),
z_s	Water surface elevation (m a.s.l.),
g	Gravitational acceleration (m s^{-2}),
Txx, Txy, Tyx, Tyy	Depth-averaged turbulent stresses (N m^{-2}),
Dxx, Dxy, Dyx, Dyy	Dispersion terms due to the non-uniformity of flow velocity and the effect of secondary flow, which are important in curved channels,
τ_{bx}, τ_{by}	Bed shear stresses determined by the following equations $\tau_{bx} = \frac{1}{h^{1/3}} \rho g n^2 u \sqrt{u^2 + v^2}$, $\tau_{by} = \frac{1}{h^{1/3}} \rho g n^2 v \sqrt{u^2 + v^2}$ (N m^{-2}),
τ_{sx}, τ_{sy}	Represent the shear forces acting on water surface, usually caused by wind driving (N m^{-2}),
f_c	Coriolis coefficient (–),
C_k	Depth-averaged suspended-sediment load (kg s^{-1}),
$C_{*\mathrm{k}}$	The suspended-sediment transport capacity or the depth-averaged suspended-sediment at the equilibrium state (kg s^{-1}),
ε_s	Turbulence diffusivity coefficient of sediment, determined with $\varepsilon_s = \frac{v_t}{\sigma_c}$ (–),
σ_c	Turbulent Schmidt number, usually having a value between 0.5 and 1.0 or determined by van Rijn's method (–),
ω_{sk}	Settling velocity of sediment (m s^{-1}),
α	Non-equilibrium adaptation coefficient (–),
c_{bk}	The average concentration of bed load at the bed-load zone (–),
q_{bk}	The bed-load transport rate of size class k (kg s^{-1}),
q_{b*k}	The corresponding bed-load transport capacity or bed-load transport rate at the equilibrium state (–),
α_{bx}, α_{by}	Direction cosine components of the bed-load movement, which is assumed to be along the direction of bed shear (0). α_{bx} and α_{by} are corrected to consider the effects of helical motion and channel slope (Wu and Wang 2005),
R	Relative density of bed material in the water (kg m^{-3}),
ρ	Density of water (kg m^{-3}),

ρ_s	Density of sediment (kg m^{-3}),
h	Water depth (m),
I	Drop of water surface (–),
F_i	The percentage of the bed material fraction (%),
d_i	Grain diameter (m).

1 Introduction

Bed load transport in river channels is a fundamental process with applications to a wide variety of research problems, such as canal design (Lane 1955), palaeohydrological reconstructions (Church 1978), placer formation (Komar and Wang 1984; Li and Komar 1992), flushing flows (Milhous 1990; Kondolf and Wilcock 1992), and assessment of aquatic habitat (Buffington 1995; Montgomery et al. 1996; Buffington and Montgomery 1997). For hydraulic engineers, especially those responsible for hydraulic structures on mountainous streams, where the type of hydraulic structures applied has an influence on the resultant channel morphology, sediment transport problems are also a key issue (Yang 1996). Riffle and pool sequences are frequent features of natural gravel-bed streams. Mimicking this morphology, engineers developed rapid hydraulic structures, RHS (or boulder ramp hydraulic structures), which still operate as hydraulic structures, stabilizing channel planform, reducing channel slope and preventing bed erosion, but are similar to riffles, harmonizing with natural river landscape and allowing fish and invertebrates to migrate down and upstream (Fig. 1).

So far, only a few papers have described designing of such structures and hydraulic parameters related to them (Pagliara and Bung 2013; Radecki-Pawlik et al. 2013; Sattar et al. 2013; Radecki-Pawlik 2013). At the same time, even less attention has been given to the sediment transport within the region of RHS. This paper is an attempt to describe some sediment transport problems connected with an occurrence of rapid hydraulic structures in a gravel-bed stream. Our study tries to show the influence of such a structure on sediment transport processes. In that context, advantages and disadvantages of the CCHE2D model are discussed and compared with BAGS model as well as the classic Hjulström's diagram. The study aims: (i) to show the effects of RHS on sediment transported along the stream channel, and (ii) to evaluate the performance of CCHE2D model in predicting sediment phenomena occurring along the stream with rapid hydraulic structures. All measurements necessary to run the models and calculate other hydraulic parameters were performed in Porębianka Stream in the Gorce Mountains, Polish Carpathians, where several boulder ramps had been constructed.

Fig. 1 Rapid hydraulic structure (RHS) in Porębianka stream under low water level conditions in summer 2011 (*photo* by K. Plesiński)

2 Study Area

The catchment of Porębianka Stream is underlain by flysch rocks. It is situated in the Polish Carpathians about 60 km south of Kraków. Porębianka is a 4th-order, 15.4 km long, flashy stream characterized by frequent flood events that mostly occur during summer. It is a right-bank tributary to the Mszanka River (Fig. 2). The bed material of the stream consists of sandstone and mudstone pebbles and cobbles with a subordinate proportion of sandy to silty particles. Hydrological characteristics of Porębianka were determined on the basis of records at a gauging station located in the middle course of the stream. The highest mean monthly stage occurs in April due to snow melt. The lowest flows occur in February and October. The amplitude of water stage in Porębianka is 151 cm (Korpak 2008). This high variability of water stage reflects low retention potential of the flysch bedrock and partial deforestation of the catchment.

Along the stream, deep channel incision has occurred over the few last decades, mostly caused by illegal in-channel mining of gravel. The morphology of Porębianka channel is also influenced by check-dams which divide the stream into different channel stretches, with incision occurring downstream of the check-dams and bed aggradation upstream of them (Kościelniak 2004; Korpak 2007). Some proportion of flow runs through gravels underlying the channel bed (cf. Carling et al.

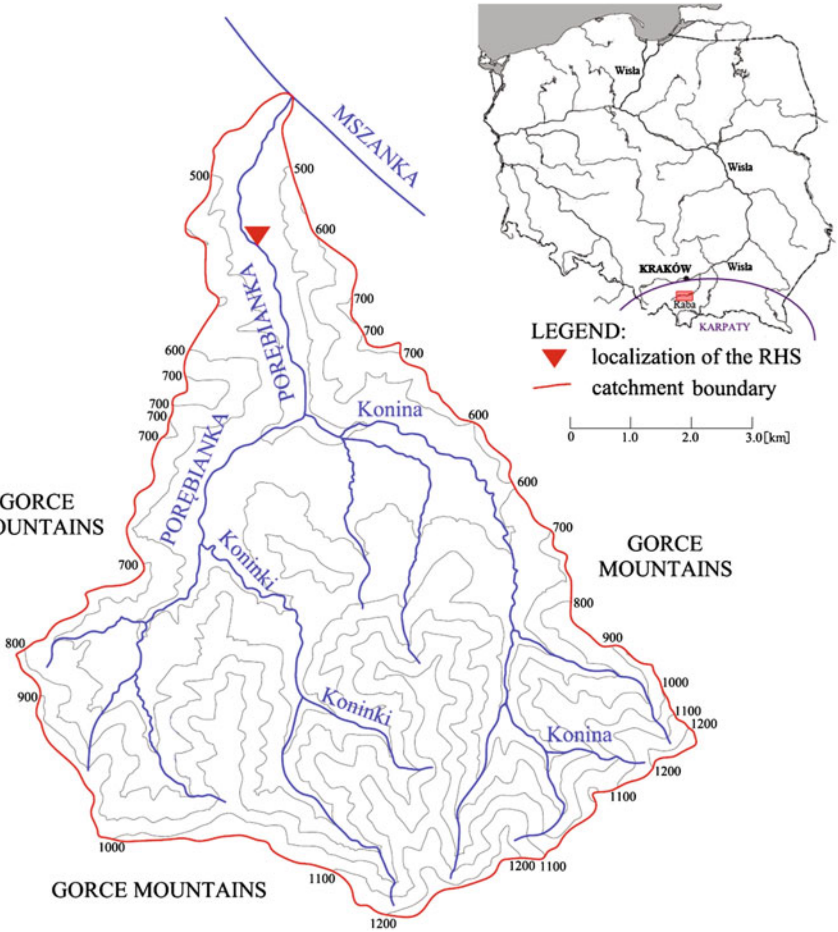


Fig. 2 Location of the study area

2006; Lach and Wyżga 2002), but this was not analyzed in this paper. In the lower course of the stream, its morphology is influenced by a number of rapid hydraulic structures (Fig. 1) that were built to stabilize the stream bed and change the sediment transport conditions. Thus, we decided to investigate this problem and look closely into the bed-load transport occurring along the reach of Porębianka, where the rapid hydraulic structures had been constructed. Measurements of hydraulic and sedimentary characteristics of Porębianka were performed in the period comprising elevated flows higher than the mean as well as flood flows. They made it possible to check, through the analysis of bed-load transport, sedimentation and erosion processes, how the examined boulder ramp (RHS) affected the hydrodynamics and morphology of the channel upstream and downstream of the structure.

3 Methodology

In this study we applied two methods of modeling and calculating sediment transport. These were: the CCHE2D model (Jia and Wang 2009; Wu 2001) and the BAGS model (Wilcock et al. 2009) with the Parker equation (Parker 1990). Below we present a short description of them.

3.1 CCHE2D Model

CCHE2D has two versions based on the Efficient Element Method (Jia and Wang 2009; Wu 2001) and Finite Volume Method (Wu 2004). Both adopt the following 2-D shallow water equations:

The continuity equation:

$$\frac{\partial Z}{\partial t} + \frac{\partial (hu)}{\partial x} + \frac{\partial (hv)}{\partial y} = 0 \tag{1}$$

and the momentum equations:

$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + v\frac{\partial u}{\partial y} = -g\frac{\partial Z}{\partial x} + \frac{1}{h}\left[\frac{\partial (h\tau_{xx})}{\partial x} + \frac{\partial (h\tau_{xy})}{\partial y}\right] - \frac{\tau_{bx}}{\rho h} + f_{Cor}v \tag{2}$$

$$\frac{\partial v}{\partial t} + u\frac{\partial v}{\partial x} + v\frac{\partial v}{\partial y} = -g\frac{\partial Z}{\partial y} + \frac{1}{h}\left[\frac{\partial (h\tau_{yx})}{\partial x} + \frac{\partial (h\tau_{yy})}{\partial y}\right] - \frac{\tau_{by}}{\rho h} + f_{Cor}u \tag{3}$$

CCHE2D models compute the total sediment transport using a single governing equation or separately calculate the bed-load and suspended-sediment transport using two equations. The latter approach is introduced here. The depth-averaged transport equation of suspended sediment is

$$\frac{\partial (hC_k)}{\partial t} + \frac{\partial (uhC_k)}{\partial x} + \frac{\partial (vhC_k)}{\partial y} = \frac{\partial}{\partial x}\left(\varepsilon_s h\frac{\partial C_k}{\partial x}\right) + \frac{\partial}{\partial y}\left(\varepsilon_s h\frac{\partial C_k}{\partial y}\right) + \alpha\omega_{sk}(C_{*k} - c) \tag{4}$$

The bed-load transport is determined by the following equation:

$$\frac{\partial\left(\delta_b \underset{bk}{\bar{c}}\right)}{\partial t} + \frac{\partial (\alpha_{bx}q_{bkx})}{\partial x} + \frac{\partial (\alpha_{by}q_{bky})}{\partial y} + \frac{1}{L_t}(q_{bk} - q_{b*k}) = 0 \tag{5}$$

The bed deformation is calculated by the equation:

$$(1 - p')\frac{\partial z_{bk}}{\partial t} = \frac{\alpha\omega_{sk}(C_k - C_{*k}) + (q_{bk} - q_{b*k})}{L_t} \tag{6}$$

3.2 BAGS Model

A primer accompanying BAGS (Bedload Assessment in Gravel-bedded Streams) model (basically the software) was written to facilitate computation of sediment transport rates in gravel-bed rivers. BAGS provides a choice of different formulas and supports a range of different input information. It offers the option of using measured transport rates to calibrate a transport estimate. BAGS can calculate a transport rate for a single discharge or for a range of discharges (Pitlick et al. 2009). With the software, the following methods can be used:

1. The substrate-based equation of Parker-Klingeman-McLean
2. The substrate-based equation of Parker-Klingeman
3. The surface-based equation of Parker
4. The procedure of Bakke
5. The two-fraction equation of Wilcock
6. The surface-based equation of Wilcock and Crowe

In this paper we used the Parker's method described in detail below. In this method (Parker 1990) the following parameters are the most important:

Equivalent diameter:

$$d_{sg} = \exp\left(\sum_{i=1}^{n} F_i \ln(d_i)\right) \tag{7}$$

Value of shear stress:

$$\tau_{sg}^{*} = \frac{h \cdot I}{R \cdot d_{sg}} \tag{8}$$

Value of effective motion function:

$$\varphi_{sg0} = \frac{\left(\tau_{sg}^{*}\right)}{0.0386} \tag{9}$$

$$\delta_{\varphi} = \sqrt{\sum_{i=1}^{n}\left(\left(\frac{\ln\left(\frac{d_i}{d_{sg}}\right)}{\ln(2)}\right)^2\right)F_i} \tag{10}$$

$$\omega = 1 + \frac{\delta_{\varphi}}{\delta_{\varphi 0}\left(\delta_{sg0}\right)}\left(\omega_0\left(\varphi_{sg0}\right) - 1\right) \tag{11}$$

when some parameters are taken from the Bags-manual.

For each grain diameter d_i from the sieve curve, we calculate parameters:

$$\delta_i = \frac{d_i}{d_{sg}} \tag{12}$$

$$g_{0i} = (\delta_i)^{-0.0951} \tag{13}$$

$$\varphi_i = \omega \cdot \varphi_{sg0} \cdot g_{0i} \tag{14}$$

Dimensionless transport of bed-material load is calculated with the equation:

$$W^* = 0.0218\sum_{i=1}^{n} G(\varphi_i)F_i \tag{15}$$

where for $\varphi > 1.59$:

$$G(\varphi_i) = 5{,}474\left(1 - \frac{0.853}{\varphi}\right)^{4.5} \tag{16}$$

for $1 \leq \varphi \leq 1.59$:

$$G(\varphi_i) = \exp\left\{14.2(\varphi - 1) - 9.28(\varphi - 1)^2\right\} \tag{17}$$

and for $\varphi < 1$:

$$G(\varphi_i) = \varphi^{14.2} \tag{18}$$

The total value of the transport of bed-material load for all fractions is next calculated from:

$$W_c(h) = \frac{\sqrt{g}(h \cdot I)^{1.5}}{R}\sum_{i=1}^{n} F_i W^* \tag{19}$$

A number of empirical formulas were proposed to calculate the rate of bed-load transport in mountain watercourses. Vanoni (1975) mentioned that most of available methods of predicting and calculating sediment flux are far from satisfactory. However, in this paper we applied the above-mentioned models to recognize at least a range of influence of boulder ramps (RHS) on sediment transport phenomena. Parallel to the numerical simulation with use of CCHE2D and BAGS models, we performed field measurements of the changes to channel morphology and selected hydrodynamic and granulometric parameters of the study stream to provide extensive plausibility check to the modeling results and determine boundary conditions of the models. Thus, the field measurements were concentrated on measuring bed-material size (sieving analysis) to obtain grain-size curves used in the model validation procedures. Also a geodetic survey and linear measurements were needed to know a detailed shape of RHS and the channel morphology, so such measurements were conducted in the field as well.

Numerical modeling of hydrodynamic and morphological conditions was carried out for the flood flow $Q = 55\ m^3\ s^{-1}$ which was the largest flood event observed during the measurement period. That flood occurred in May and June 2010. Field observations indicated that the flood flow was high enough to initiate bed-load transport and to modify channel morphology upstream and downstream of the boulder ramp selected for the numerical modeling. The modeling focused on a 200 m-long channel section comprising the boulder ramp and short parts of the stream upstream and downstream from the hydraulic structure. The values obtained from the modeling were compared with the classical Hjulström's diagram showing correlation between hydrodynamic parameters and the size of grains that either remain stationary or are transported on the channel bed.

4 Results and Discussion

Below, for clarity of understanding, the results are shown on graphs and in tables and are described along the chapter to explain the results from the carried out analysis. Figure 3 shows flow velocities for the measured discharge of $55\ m^3\ s^{-1}$. They have been obtained on the basis of numerical modeling employing CCHE2D model. In the channel above the boulder ramp, flow velocity in the lateral and central parts of the channel was similar, ranging from 1.90 to 2.20 $m\ s^{-1}$. Two gravel bars occurred upstream of the ramp: one in the central part of the channel and another close to the left bank. Velocity of the flow over the centrally situated bar ranged from $V \approx 2.30\ m\ s^{-1}$ in its lower parts to $V \approx 1.90\ m\ s^{-1}$ over the highest point of the bar. The maximum velocity over the lateral bar was much lower, amounting to $V \approx 1.50\ m\ s^{-1}$. The presence of both bars apparently influences the bed roughness.

The highest velocity $V = 4.25\ m\ s^{-1}$ indicated by the modeling with CCHE2D model occurred at the end sill of the rapid hydraulic structure, in the downstream part of the triangular trough constructed across the middle part of the structure along

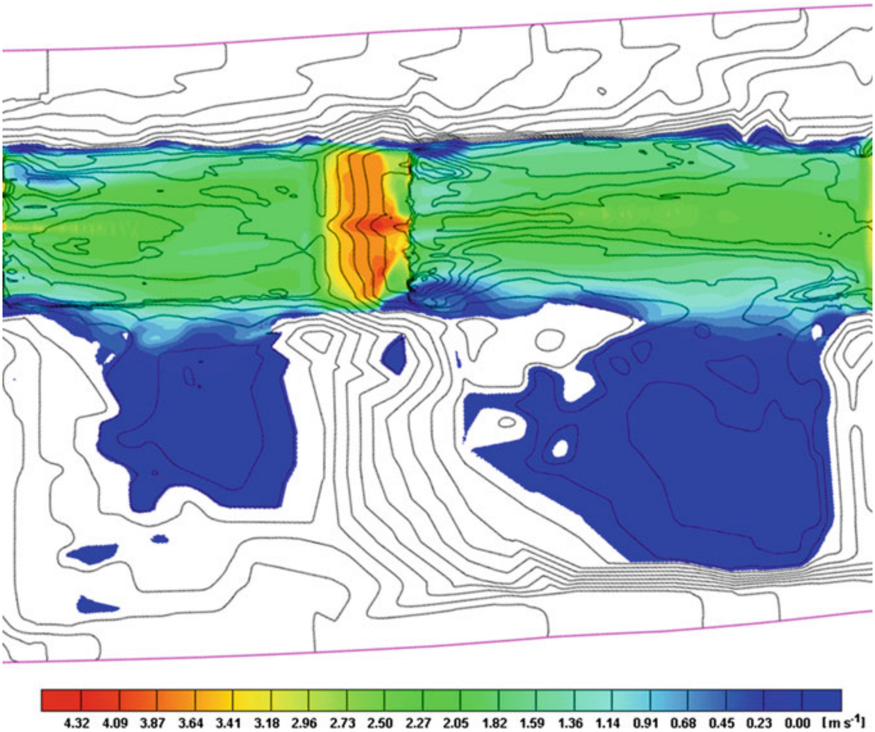


Fig. 3 Flow velocity in a 170 m stream stretch obtained by the numerical modeling with CCHE2D

its steep gradient. At the same time, a lower value of V = 2.00 m s^{-1} was observed at the crest sill of the structure. In the middle part of the energy dissipating pool, velocity amounted to V ≈ 3.60 m s^{-1}, whereas in its left and right bank it was lower, equaling V = 2.74 m s^{-1} and V = 2.63 m s^{-1}, respectively. Average velocity indicated by the modeling to occur along sides of the steep, boulder-paved apron of the structure was similar irrespective of the detailed location and amounted to V ≈ 3.70 m s^{-1}.

Next, part of the channel located downstream of the rapid hydraulic structure was analyzed. Here, the highest velocity, ranging from 2.40 to 2.60 m s^{-1}, was obtained in the middle of the channel, just below the energy dissipating pool. Downstream from the boulder ramp, the formation of two lateral bars was noticed. The velocity of flow over the bars ranged from 1.50 m s^{-1} in their distal parts to 1.90 m s^{-1} in the proximal parts. Downstream of the bars, the velocity V ≈ 2.00 m s^{-1} was observed. Immediately downstream of the energy dissipating pool, channel forms resembling plunge pools were noticed. Velocity in these locations amounted to V ≈ 1.15 m s^{-1}.

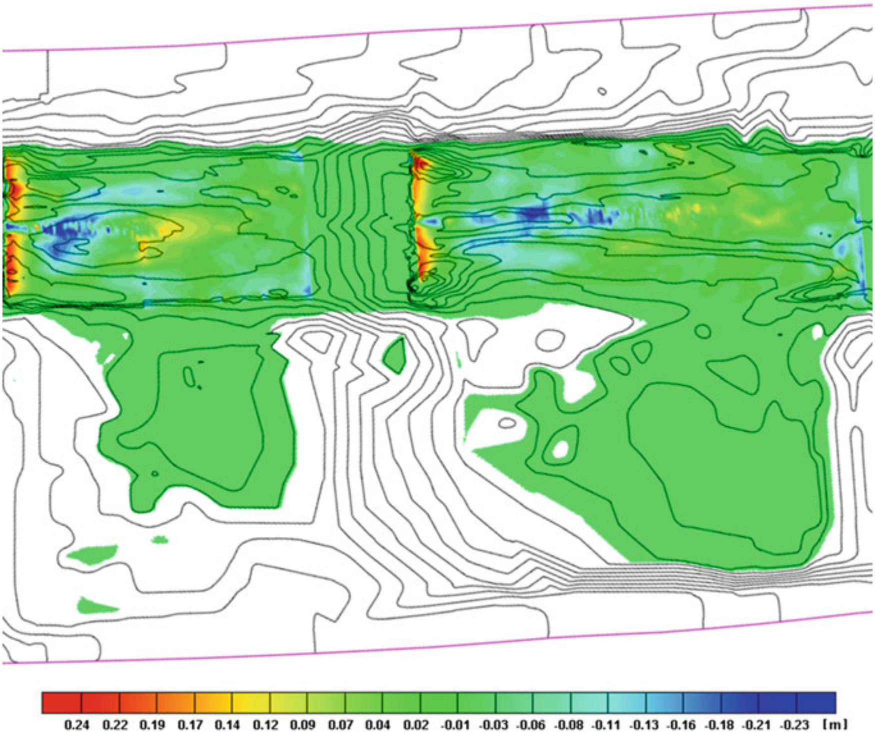


Fig. 4 Bed changes in a 170 m stream stretch obtained by the numerical modeling with CCHE2D

Subsequently, we analyzed changes in morphology of the bed upstream and downstream of the rapid hydraulic structure, that were indicated by the CCHE2D model. Upstream of the structure, a proximal part of the central bar was eroded ($\Delta H \leq -0.13$ m), and the eroded material was deposited in the distal part of the bar ($\Delta H \leq 0.10$ m). Bed material was also eroded along both lateral parts of the central bar: $\Delta H \approx -0.09$ m along the left side of the bar and $\Delta H \approx -0.06$ m along its right side. Bed scouring was also recorded immediately upstream of the boulder ramp and the eroded material was deposited downstream of the structure.

In the main channel downstream of the boulder ramp (Fig. 4), erosion of the stream bed is observed: $\Delta H \approx -0.12$ m. It may be associated with narrowing of the channel here. At the same time, along both sides of the channel, scouring of two lateral bars is observed (ΔH ranging from -0.02 to -0.06 m). In the distal parts of the bars, accumulation of sediment is observed, resulting in the aggradation of the bed by $\Delta H \approx 0.06$ m. Sediment deposition was also observed in the plunge pools formed immediately below the energy dissipating pool. This deposition caused partial infilling ($\Delta H \leq 0.20$ m) of the plunge pools.

Model CCHE2D enables a detailed analysis of changes to bed morphology on the background of changes in flow velocity and bed shear stress, at the same time

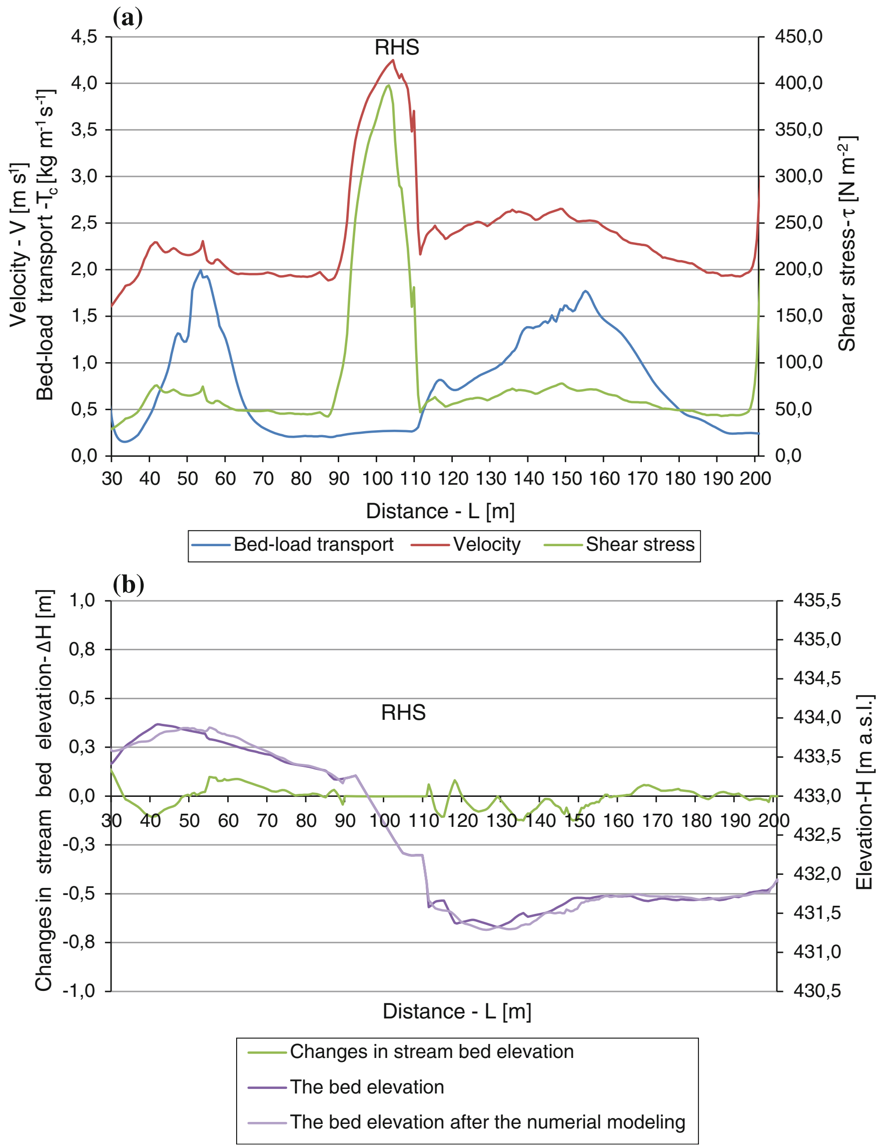


Fig. 5 **a**, **b** The hydrodynamic parameters and bed-load transport obtained with numerical modeling with CCHE2D (*above*) and the observed changes in the elevation of the stream bed (*below*)

calculating the rates of bed-load transport. The analysis of the transport rate obtained from the modeling was done in the following way. First, a general analysis comprised the channel section L = 30–202 m (Fig. 5a, b), indicating relationships between morphological changes and the rates of bed-load transport. Second, parameters of the bed-load transport, hydrodynamic parameters (velocity, bed shear stress), erosion and deposition were analyzed in different parts of the investigated stream section (channel upstream of the rapid hydraulic structure, the structure itself, and the channel downstream of the structure). Results of the detailed analysis, performed with use of CCHE2D model, were subsequently compared with the results obtained by means of the classical Hjuström's diagram.

For the detailed analysis, the study section was divided into 9 segments in such a way that the values of average velocity and shear stress were similar in a given segment:

$$
\begin{aligned}
&\mathrm{L} = 30{-}35\ \mathrm{m}, \quad \mathrm{V} = 1.8\ \mathrm{m\,s^{-1}}, \quad \tau = 36\ \mathrm{N\,m^{-2}},\\
&\mathrm{L} = 35{-}55\ \mathrm{m}, \quad \mathrm{V} = 2.2\ \mathrm{m\,s^{-1}}, \quad \tau = 69\ \mathrm{N\,m^{-2}},\\
&\mathrm{L} = 55{-}65\ \mathrm{m}, \quad \mathrm{V} = 2.0\ \mathrm{m\,s^{-1}}, \quad \tau = 54\ \mathrm{N\,m^{-2}},\\
&\mathrm{L} = 65{-}90\ \mathrm{m}, \quad \mathrm{V} = 1.9\ \mathrm{m\,s^{-1}}, \quad \tau = 48\ \mathrm{N\,m^{-2}},\\
&\mathrm{L} = 90{-}110\ \mathrm{m}, \quad \mathrm{V} = 3.7\ \mathrm{m\,s^{-1}}, \quad \tau = 275\ \mathrm{N\,m^{-2}}\text{—segment with the RHS}\\
&\mathrm{L} = 110{-}135\ \mathrm{m}, \quad \mathrm{V} = 2.5\ \mathrm{m\,s^{-1}}, \quad \tau = 60\ \mathrm{N\,m^{-2}},\\
&\mathrm{L} = 135{-}155\ \mathrm{m}, \quad \mathrm{V} = 2.6\ \mathrm{m\,s^{-1}}, \quad \tau = 72\ \mathrm{N\,m^{-2}},\\
&\mathrm{L} = 155{-}180\ \mathrm{m}, \quad \mathrm{V} = 2.3\ \mathrm{m\,s^{-1}}, \quad \tau = 58\ \mathrm{N\,m^{-2}},\\
&\mathrm{L} = 180{-}202\ \mathrm{m}, \quad \mathrm{V} = 2.0\ \mathrm{m\,s^{-1}}, \quad \tau = 46\ \mathrm{N\,m^{-2}}.
\end{aligned}
$$

The general analysis of changes to the morphology of stream bed at the flow Q = 55 $m^3\ s^{-1}$ (Fig. 5a, b) indicated deposition of material in the channel segment L = 30–35 m. In the next segment, L = 35–50 m, scouring of the proximal part of a central bar, leading to bed erosion (ΔH = −0.10 m) was observed. At the same time, most of the entrained material was deposited in the distal part of the bar (L = 50–88 m). The amount of bed aggradation decreased downstream, amounting to ΔH = 0.10 m at the point L = 55 m and to ΔH = 0.01 m in the segment L = 80–88 m. The rate of bed-load transport amounted to Tc = 2.00 kg s^{-1} in the area of the gravel bar and to Tc = 0.21 kg s^{-1} in the vicinity of the boulder ramp.

Along the rapid hydraulic structure, where neither erosion nor sedimentation was noticed, the rate of sediment transport was Tc = 0.25 kg s^{-1}. This value was slightly higher than in the channel above the RHS (Tc = 0.21 kg s^{-1}). At the point L = 89–90 m, the model indicated erosion (ΔH = −0.04 m) that was not observed in the field. These points are special as they occur at the boundary of the channel with the rapid hydraulic structure. The results from the modeling, indicating erosion at these points, probably might be due to computational system error within the CCHE2D model, occurring on the boundary between the non-erodible surface of the RHS and the erodible part of the stream bed.

In a short section (L = 111–112 m) located immediately downstream of the RHS, the model indicated bed material deposition that, in fact, was not observed. In our opinion, this might also be associated with changes in bed erodibility where the model incorrectly identifies sediment transport conditions.

Along the section L = 113–160 m, bed erosion of varying size was recorded. There were sections, approximately 10 m long, where the scale of erosion was $\Delta H = -0.10$ to -0.12 m, but they alternated with shorter stretches (up to 1 m in length) with small or even negligible erosion $\Delta H \leq -0.01$ m. In the section L > 160 m, both deposition and accumulation of bed material was observed.

The detailed analysis of the data obtained from numerical simulation consisted in their presentation on the classical Hjulström's diagram which is a classic way of presentation of sedimentation results. This aimed at extensive plausibility check of the numerical model to calculate the transport of grains of particular size through comparison of the results with general fields of sediment accumulation, erosion and transport on the diagram.

Figure 6 presents the bed-load transport derived from the CCHE2D model. The input data are defined as follows: d < 0.025 m—15 %, d ≈ 0.035 m—35 %, d ≈ 0045 m—25 % d ≈ 0.055 m—55 %, d > 0.065 m—10 %. It can be observed that all the sediment fractions are subjected to transport regardless of their size. According to the Hjulström diagram, transport of sediment should first involve fine and subsequently coarse particles. In contrast, CCHE2D model predicts mass sediment transport and does not take into account fluctuations in the flow velocity.

The Hjulström diagram (Fig. 7) shows the dependence of the diameter of particles sitting on the stream bed to the mean flow velocity obtained from the modeling. It is done along individual study reaches. This made possible to identify the stream reaches with predomination of sedimentation, transport and erosion. Based on the data presented in the Hjulström diagram, we could compare the results of the numerical modeling performed with CCHE2D model with the classical approach of Hjulström.

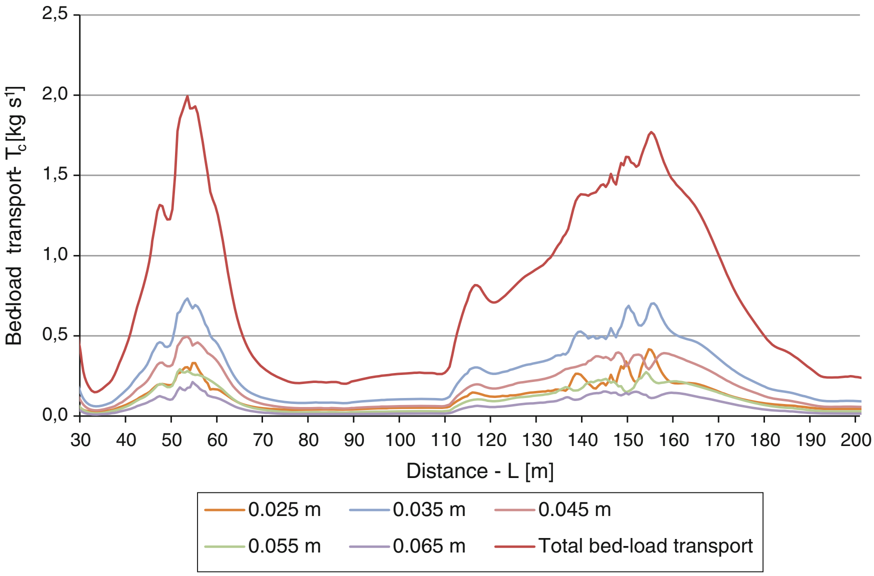


Fig. 6 Distribution of transported material in individual sections of the stream channel

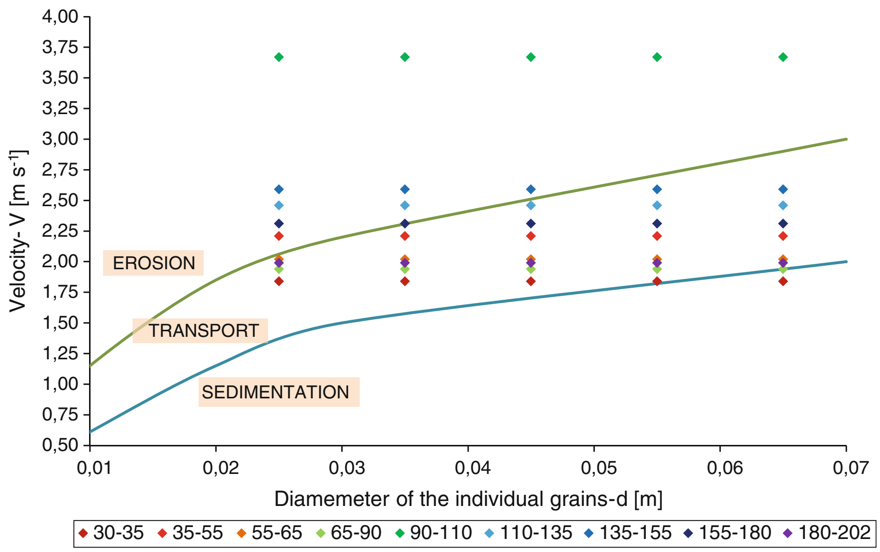


Fig. 7 Hjulström's diagram showing bed material conditions in each section of the stream channel

According to the Hjulström's diagram (Fig. 7), in the section L = 30–35 m grains with the diameter d ≤ 0.055 m should be in motion, while those larger than d ≥ 0.065 m might be deposited. According to CCHE2D model, the deposition of all fractions occurred along this section (Fig. 7). According to the Hjulström's diagram, in the next section (L = 35–55 m) fine sediment (d ≤ 0.025 m) should be eroded and grains with the diameter d ≥ 0.035 m should be in motion. According to CCHE2D model, erosion of the channel bed and movement of the grains of all sizes present in the bed material (d = 0.025–0.065 m) took place (Fig. 7; Table 1).

Along the section L = 55–65 m, the Hjulström's diagram indicates bed-load transport, whereas according to the numerical model CCHE2D, only sedimentation occurred. Flow velocity in that section reached up to V = 2.00 m s^{-1}, so it was slightly lower than in the previous section (L = 35–55 m, V = 2.20 m s^{-1}), where erosion occurred. It can be concluded that for that part of the channel, the CCHE2D model was sensitive to the increase in velocity but rather not to its absolute values.

Along the next section L = 65–90 m, sedimentation of all fractions was indicated by the CCHE2D model. However, according to the Hjulström's diagram, sedimentation of coarse grains d ≥ 0.065 m should occur, while grains of smaller diameter d ≤ 0.055 m should be transported.

In the section with the rapid hydraulic structure (L = 90–110 m), erosion should occur according to the Hjulström's diagram. However, the construction of the rapid hydraulic structures with use of large boulders does not allow to observe that phenomenon in CCHE2D model. While defining boundary conditions of the model, the sloping bed of the RHS was defined as "non-erodible", otherwise this structure would have been completely washed out.

Table 1 Summary of bed-material conditions obtained based on the Hjulström's diagram and data from the model that were observed during the simulation carried out for the flow Q = 55 $m^3\ s^{-1}$

Section L [m]	Average velocity[1] V [$m\ s^{-1}$]	Bed-material conditions for the individual grains according to Hjulström's diagram[2] d [mm]			Bed-material conditions according to the CCHE2D model[3]		Compatibility[4]
		Erosion E	Transport T	Sedimentation S	morphodynamic	V [$m\ s^{-1}$]	
30 – 35	1.8	-	25 – 55	≥ 65	S	↑ 1.60 – 1.85	partial
35 – 55	2.2	≤ 25	≥ 35	-	E	↑ 1.85 – 2.20	partial
55 – 65	2.0	-	25 – 65	-	S	↓ 2.20 – 1.95	-
65 – 90	1.9	-	25 – 55	≥ 65	S	↔ 1.95 – 1.95	partial
90 – 110	3.7	25 – 65	-	-	T	-	bd
110 – 135	2.5	≤ 35	≥ 45	-	E	↑ 2.15 – 2.55	partial
135 – 155	2.6	≤ 45	≥ 55	-	E	↔ 2.55 – 2.55	partial
155 – 180	2.3	≤ 25	≥ 35	-	S	↓ 2.55 – 2.10	-
180 – 202	2.0	-	25 – 65	-	S	↓ 2.10 – 2.00	-

[1] Average velocity for each section

[2] Sediment fractions subjected to erosion, transport and sedimentation in particular stream sections according to the Hjulström's diagram, d (mm)—individual grain diameter

[3] Summary of bed material conditions (erosion, transport and sedimentation) according to CCHE2D model and the values of velocity at the beginning and end of each section: *E* bed erosion; *T* bed-load transport; *S* sedimentation; *BR* the phenomena were not observed; ↑ increasing velocity along particular stream section; ↔ no change of velocity along particular section; ↓ decreasing velocity along particular section

[4] Compatibility of bed-material conditions indicated by CCHE2D model and the Hjulström's diagram: + Compatibility of models—green color; *Partial* compatibility of models—yellow color; – no compatibility of models—red color; *bd* no data

The Hjulström's diagram indicates that downstream of the rapid hydraulic structure (L = 110–180 m), bed erosion and entrainment of small sediment fractions should occur, coupled with the transport of coarse sediment fractions delivered from the channel upstream of the structure. This is inconsistent with the results of the numerical modeling, because according to CCHE2D model, bed erosion and setting in motion of small sediment fractions occurred along the section L = 110–155 m. These phenomena accompany the increase in velocity from V = 2.15 $m\ s^{-1}$ to V = 2.55 $m\ s^{-1}$. In the section L = 155–202 m, where the velocity suddenly decreases from V = 2.55 $m\ s^{-1}$ to V = 2.00 $m\ s^{-1}$, the model indicated sedimentation, which was also incompatible with the Hjulström's diagram.

Finally, we compared processes occurring in the sections L = 35–55 m and L = 155–180 m to demonstrate that the model CCHE2D is more sensitive to changes in velocity than to the absolute value of the parameter. Based on the data obtained from the model, along the first section bed erosion and sediment entrainment took place at increasing velocity from V = 1.85 m s^{-1} to V = 2.20 m s^{-1}. Along the second section L = 155–180 m, the model indicated sedimentation, despite the higher average velocity V = 2.30 m s^{-1} associated with the decreasing tendency of the velocity from V = 2.55 m s^{-1} to V = 2.10 m s^{-1}.

For an extensive plausibility check of the values of bed-load transport obtained with CCHE2D model, we used a simple BAGS model. It is used to calculate the rates of sediment transport according to different formulas rather than to simulate them with a numerical software. One has to bear in mind that in BAGS model all formulas for sediment transport calculations are giver directly in Excel files, so the user has no doubts which formulas and which parameters are used. It gives a confidence when using this software. Our calculations were based on the Parker's equation (Parker 1990) and the results from these calculations and those obtained with CCHE2D model are compared in Table 2. As the results obtained with use of the Parker's equation are most sensitive to channel slope, the calculations were performed for three different slope values. The first value, S = 0.0055, is the channel slope between two successive rapid hydraulic structures. The second value, S = 0.0139, is the average slope of the apron of the structure. The third value, S = 0.0073, is the hydraulic gradient calculated by CCHE2D model.

Figure 8 compares bed-load transport rates calculated by means of CCHE2D model with those obtained with use of BAGS software for the discharges Q = 25, 55, 90 and 190 m^3 $s^{-1.}$

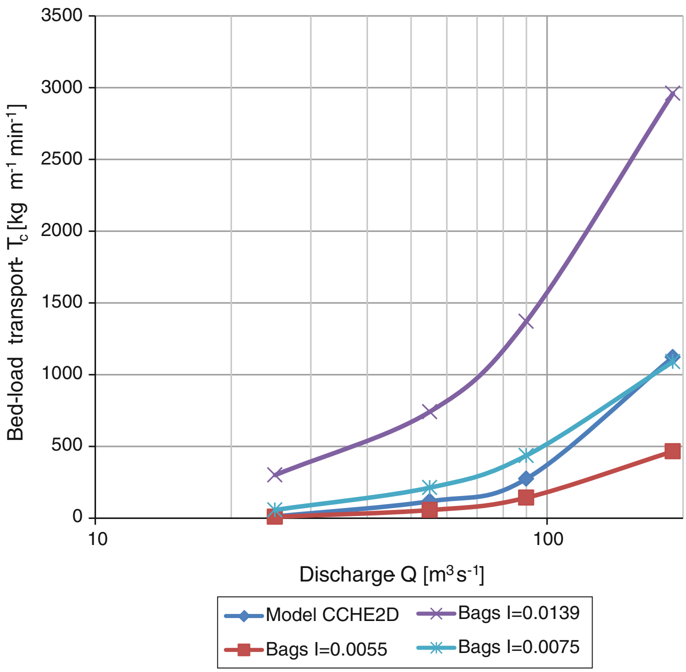


Fig. 8 Comparison of bed-load transport rates calculated by means of CCHE2D and BAGS models for changing flood discharge and different values of channel slope

Table 2 Bed-load transport rates (kg m^{-1} min^{-1}) calculated by means of CCHE2D and BAGS models for particular flood discharges and channel slope S = 0.0055 (channel slope between two successive RHS, measured in the field)

Q (m^3 s^{-1})	Model CCHE2D	Model BAGS
25	11.58	8.99
55	117.37	56.16
90	273.87	142.31
190	1120.97	465.22

A comparison of the results obtained by means of the two models indicates that bed-load transport rates calculated by the CCHE2D model were always higher, irrespective of the value of channel slope used in the BAGS model for the Parker's equation (Table 2). We analyzed bed-load transport values for 4 different discharge values. The first value of Q = 25 m^3 s^{-1} is the flood flow of 50 % probability. It is higher than the bankfull discharge and is the flow at which bed-load transport commences in the stream. The value of Q = 55 m^3 s^{-1} is the largest flood event observed during the measurement period. The third value of Q = 90 m^3 s^{-1} is the flood flow of 10 % probability, and the fourth value of Q = 190 m^3 s^{-1} is the flood flow of 1 % probability. The considerable discrepancies between the results obtained by means of the two models confirm the objection of Vanoni (1975) that available methods of predicting the rates of bed-load transport in rivers are far from satisfactory.

5 Conclusions

The main purpose of this paper was to show the influence of a rapid hydraulic structure (also called a boulder ramp) on sediment transport phenomena in a stream channel and to compare the results of two models predicting bed-load transport rates. Based on the performed analysis and calculations, the following conclusions can be formulated:

1. CCHE2D numerical model used to calculate bed-load transport rates and indicate erosion, transport and sedimentation phenomena in the stream section with the rapid hydraulic structure gives results which do not agree with bed-material conditions indicated on the Hjulström's diagram which is a classical tool for considerations of sedimentary phenomena in river channels.
2. The places with marked differences in the interpretation of erosion, transport and sedimentation of bed material between the classical Hjulström's approach and that of the CCHE2D model are located on the boundaries between stream bed and the hydraulic structure.
3. The rates of bed-load transport obtained with CCHE2D model are considerably overestimated in comparison with those calculated by means of the Parker's

equation in the BAGS model. In contrast to CCHE2D model, the formulas used to calculate sediment transport rate in the BAGS model are apparent to the users, which allows for greater confidence. The disparity of the obtained results most likely reflects a crucial importance of hydraulic gradient in the calculations of bed-load transport rates and some simplifications at defining boundary conditions in the CCHE2D model.

4. Predicting the phenomena of erosion, transport and sedimentation of bed material and calculating bed-load transport rates in a gravel-bed channel by means of a single numerical model, especially a 2-dimensional one such as CCHE2D, gives results that may be incompatible with field observations and with predictions obtained by means of the classical Hjulström's approach based on extensive empirical evidence. Thus, we suggest use of at least two models for these tasks and an extensive plausibility check of the simulation results through comparison with field observations of morphological changes in the stream.

References

Buffington JM (1995) Effects of hydraulic roughness and sediment supply on surface textures of gravel-bedded rivers. MS thesis, University of Washington, Seattle, 184 pp

Buffington J, Montgomery DR (1997) A systematic analysis of eight decades of incipient motion studies, with special reference to gravel-bedded rivers. Water Resour Res 33:1993–2029

Carling P, Whitcombe L, Benson I, Hankin B, Radecki-Pawlik A (2006) A new method to determine interstitial flow patterns in flume studies of sub-aqueous gravel bedforms such as fish nests. River Res Appl 22:691–701

Church M (1978) Palaeohydrological reconstructions from a Holocene valley fill. In: Miall AD (ed) Fluvial sedimentology, Can Soc Petrol Geol, vol 5, pp 743–772, Calgary, Alberta

Jia Y, Wang S, (2009) Development of a water infrastructural system chemical spill simulation model (WIS-CSSM). Technical report no. NCCHE-SERRI-TR-2009-01

Komar PD, Wang C (1984) Processes of selective grain transport and the formation of placers on beaches. J Geol 95:637–655

Kondolf GM, Wilcock P (1992) The flushing flow problem. EOS Trans AGU 73:239

Korpak J (2007) Morfologiczna rola budowli regulacyjnych w górskich systemach fluwialnych. Doctoral thesis. Kraków. Manuscript

Korpak J (2008) Rola maksymalnych wezbrań w funkcjonowaniu systemów uregulowanych koryt górskich. Land Anal 8:41–44

Kościelniak J (2004) Zmiany funkcjonowania górskich systemów korytowych w wyniku przeprowadzonych regulacji hydrotechnicznych. Prace Geograficzne IGiPZ PAN, vol 200, pp 59–62, Warszawa

Lane EW (1955) Design of stable channels. Trans Am Soc Civ Eng 120:1234–1279

Lach J, Wyżga B (2002) Channel incision and flow increase of the upper Wisłoka river, Southern Poland, subsequent to the reafforestation of its catchment. Earth Surf Proc Land 27:445–462

Li Z, Komar PD (1992) Selective entrainment and transport of mixed size and density sands: flume experiments simulating the formation of black-sand placers. J Sediment Petrol 62:584–590

Milhous RT (1990) The calculation of flushing flows for gravel and cobble bed rivers. In: Chang HH (ed) Hydraulic engineering, proceedings of the 1990 national conference, vol 1, pp 598–603, Am Soc Civ Eng, New York

Montgomery DR, Buffington JM, Peterson NP, Schuett-Hames D, Quinn TP (1996) Streambed scour, egg burial depths and the influence of salmonid spawning on bed surface mobility and embryo survival. Can J Fish Aquat Sci 53:1061–1070

Pagliara S, Bung D (2013) Design of low-head structures. International workshop on hydraulic design of low-head structures. FH Aachen University of Applied Sciences, p 230

Parker G (1990) Surface-based bedload transport relation for gravel rivers. J Hydraul Res 28 (4):417–434

Pitlick J, Cui Y, Wilcock P (2009) Manual for computing bed load transport in gravel-bed streams. Gen Tech Rep RMRS-GTR-223. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 40 p

Radecki-Pawlik A (2013) On using artificial rapid hydraulic structures (RHS) within mountain stream channels—some exploitation and hydraulic problems. In: Rowiński P (ed) Experimental and computational solutions of hydraulic problems, 32nd International school of hydraulics, Series: Geo Planet: Earth and planetary sciences, monograph, Springer, Netherland, pp 101–1155, 299

Radecki-Pawlik A, Plesiński K, Wyżga B (2013) Analysis of chosen hydraulic parameters of a rapid hydraulic structure (RHS) In: Porębianka stream, polish carpathians. In: Pagliara S, Bung D (ed) International workshop on hydraulic design of low-head structures, FH Aachen University of Applied Sciences

Sattar AM, Radecki-Pawlik A, Ślizowski R (2013) Using genetic programming to predict scour downstream rapid hydraulic structures from experimental results. In: Pagliara S, Bung D (ed) International workshop on hydraulic design of low-head structures 2013, FH Aachen University of Applied Sciences

Vanoni VA (1975) Sediment transport. New York, p 745

Wilcock P, Pitlick J, Cui Y (2009) Sediment transport primer estimating bed-material transport in gravel-bed rivers, United States Department of Agriculture Forest Service Rocky Mountain Research Station General Technical report RMRS-GTR-226

Wu W (2001) CCHE2D sediment transport model version 2.1. NCCHE Technical report. NCCHE-TR-2001-03

Wu W (2004) CCHE2D depth-averaged 2-d numerical modeling of unsteady flow and nonuniform sediment transport in open channels. J Hydraul Eng ASCE 130(10):1013–1024

Wu W, Wang S (2005) Development and application of NCCHE'S sediment transport models. US-China workshop on advanced computational modelling in hydroscience and engineering, Sept 2005, Oxford, Mississippi, pp 19–21

Yang C (1996) Sediment transport: theory and practice. McGraw Hill, USA, p 396

Part II
Modelling Sediment Transfer in Rivers

Challenges in Modelling Sediment Matters

Hafzullah Aksoy

Abstract Rainfall and runoff induced erosion and sediment transport in hydrological watersheds are complex processes. This process has great importance in scientific research studies and engineering practice. The amount of sediment transported within the watershed is needed for hydrological and environmental problems. Sediment transport over a watershed can be estimated by time series analysis, empirical or mechanistic equations, monitoring, sampling, surveying, remote sensing or geographical information systems. As monitoring and sampling sediment transport process are costly and not easy to implement yet, modelling has become an alternating tool used for estimating sediment transport. Data-based empirical models as well as process-based hydrological models are available for this purpose, yet modelling is difficult and challenging. Challenges encountered in the modelling are the variability in the estimate of sediment calculated by each model, data requirement for the calibration of model parameters, complexity in the calibration and validation stages of the process-based models, uncertainty in the transport capacity approach used in model construction, etc. In this chapter, these challenges related to the modelling sediment matters are discussed with an emphasis on the process-based sediment transport models. A case study on Buyukcekmece dam reservoir in the greater municipality region of Istanbul, Turkey shows that order of magnitude different outputs are obtained when data-based models are used for estimating sediment transport in hydrological watersheds. Process-based models were paid particular attention on their microtopographical structure, parameterization and data requirement.

Keywords Empirical models · Process-based models · Sediment transport · Calibration · Parameter · Transport capacity

H. Aksoy (✉)
Department of Civil Engineering, Istanbul Technical University, Maslak, Istanbul, Turkey
e-mail: haksoy@itu.edu.tr

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_5

1 Introduction

Sediment transport due to erosion is an important process ending with soil loss in hydrological watersheds. Besides its countless effects, from engineering point of view, it causes loss of storage volume in river reservoirs where eroded sediment deposits. Therefore it has always been an important scientific and engineering problem.

When sediment transport problem is dealt with for quantification purposes, numerous observation techniques and computational methods have been developed for theoretical and practical uses. The amount of sediment transported over a watershed can be calculated either by analyzing a time series (Phien and Arbhabhirama 1979; Phien 1981); correlating the collection of available data (Tingsanchali and Lal 1992); using data-based empirical equations or mechanistic equations such as Einstein, Laursen, Colby, Goncharov, Egiazaroff and Bagnold (Yalin 1972; Bogardi 1974; Garde and Ranga Raju 1977; Simons and Senturk 1992); monitoring, sampling and surveying (Araujo et al. 2006); or remote sensing and using geographical information systems (Baban and Yusof 2001). Modelling is another tool used for sediment transport. Widely used data-based empirical models such as USLE (Wischmeier and Smith 1978) as well as process-based hydrological watershed models such as KINEROS (Smith 1981), SHESED (Wicks 1988), and WEHY (Kavvas et al. 2004, 2006) are available to accommodate erosion and sediment transport modules by which sediment, eroded by rainfall or flow and transported over the hillslope and through the existing river channel to the reservoir, can be predicted. Also soft computational techniques such as artificial neural networks (Tayfur 2002) and wavelet functions (Aksoy et al. 2004) were found useful in establishing sediment transport models for forecasting or simulation purposes (Aksoy and Kavvas 2005).

In this chapter, computational methods are reviewed with an emphasis on empirical equations and process-based models to include a synthesis summary and to put forward challenges related to sediment transport processes. One challenge is that empirical equations can give order of magnitude different sediment amount. Another challenge can be related to the transport capacity of overland flow causing erosion and sediment transport. Data requirement for the process-based models and difficulties in their calibration and validation stages also are important issues to be discussed.

2 From Empirical Equations to Process-Based Models

From the review papers on the state-of-the art of sediment transport models (Zhang et al. 1996; Bryan 2000; Merritt et al. 2003; Aksoy and Kavvas 2005), it is understood that some models are similar as they are based on the same assumptions and some are distinctly different. Models so far developed can be categorized according to different criteria that may encompass process description, scale, and technique of solution (Singh 1995). A model may be based on an empirical framework to be called empirical model. If the model is constructed by using mass conservation equation of sediment, it is then called physically- or process-based erosion and sediment transport model. As a well known example for the empirical models, the USLE was designed as a tool to be used in the management practices of agricultural lands. Although its development is based on data from the United States, it has been used widely all over the world with modifications and revisions.

Limitations of the USLE due to its empirical nature forced modelers to use distinctly different alternatives based on the physical description of the erosion and sediment transport processes. KINEROS (Smith 1981), WESP (Lopes 1987), SEM (Storm et al. 1987), SHESED (Wicks 1988) and EUROSEM (Morgan et al. 1998) can be considered quick examples for the process-based sediment transport models. As summarized in Table 1, a process-based model may use lumped or distributed inputs to generate lumped or distributed outputs, respectively. A distributed model is constructed by using partial differential equations whereas a lumped model is expressed by ordinary differential equations (Singh 1995).

Models in Table 1 are all deterministic where the erosion and sediment transport processes are formulated by deterministic differential equations. None of these models can yet consider the stochasticity included in the erosion and sediment transport processes. One simplification used in the modelling is to reduce the number of dimensions of the governing equations. The model is called one- or two-dimensional depending on the number of dimensions of the mass conservation equation used in the model. Increase in the number of dimensions brings more intensive computations by the model. The partial differential equation governing the process may even be reduced to an ordinary partial differential equation (Aksoy and Kavvas 2001). The case for most of the existing process-based erosion and sediment transport models is the unsteady state where the time derivative of sediment concentration is considered.

A model is called an event-based model if it is used for the simulation of sediment produced by one single rainfall-runoff event. A continuous model is used for the simulation of sediment due to many consecutive rainfall-runoff events occurring during a season or longer time period. Initial and boundary conditions become very important in cases where the model simulates erosion and sediment transport continuously. Continuous simulation models generate large numbers of small events that may not cause significant runoff or soil loss. Models with rilled structure perform better in the simulation of the rainfall-runoff-sediment transport processes in the watershed due to the great effect of the micro-topography on overland flow and hence erosion and sediment transport. From experimental studies (Govindaraju et al. 1992) it is seen that flow and sediment discharges in rills are greater than those on interrill areas. However, smoothing irregularities (rills and interrill areas) over a hillslope is a common simplification in the modelling. Some models are capable of distinguishing among the sediment size. Single-size erosion

Table 1 Process-based erosion and sediment transport models

Model	Lumped	Distributed	Stochastic	Deterministic	1-D	2-D	Steady state	Unsteady state	Event based	Continuous	Rilled	No rilled	Single size	Multi size
ANSWERS		X		X	X		X		X			X		X
LISEM		X		X	X		X		X		X		X	
CREAMS		X		X	X		X		X		X		X	
WEPP		X		X	X		X			X	X		X	
EUROSEM		X		X	X			X	X		X		X	
KINEROS		X		X	X			X	X			X	X	
KINEROS2		X		X	X			X	X			X		X
RUNOFF		X		X	X			X	X			X		X
WESP		X		X	X			X	X			X	X	
CASC2D-SED		X		X		X		X	X			X		X
SEM		X		X		X		X		X		X	X	
SHESED		X		X		X		X		X		X	X	

and sediment transport models can only predict sediment transport for a representative grain size and can give the total sediment mass leaving the catchment. The sediment size distribution is very important in sediment quality since pollutants are usually sorbed to finest particles. This is achieved in multi-size models.

3 Challenges in Modelling

3.1 *Order of Magnitude Different Outputs with Empirical Equations*

For practical purposes, sediment yield of a watershed can be calculated by empirical equations using watershed characteristics. Following methods were taken from the literature and a case study was demonstrated to see how similar or differently these methods behave.

Method 1: Based on 80-year observation from USA and China,

$$G = 1,421\,A^{-0.229} \tag{1}$$

was derived for the estimation of sediment yield from a watershed (Erkek and Agiralioglu 1993). In Eq. (1), A is the watershed area (km^2) and G is the sediment yield (m^3/km^2-year).

Method 2: Hydrographic analysis from 16 dams operated by the State Hydraulics Works (DSI with its Turkish acronym) resulted in

$$Q_s = 1,906.26\,A^{0.9526} \tag{2}$$

where A and Q_s represents watershed area (km^2) and mean annual sediment amount (m^3/year), respectively (Gogus and Adiguzel 1991).

Method 3: A logarithmic regression line between mean annual sediment (Q_s) and the watershed area (A) of streamflows in Turkey was provided by Yurtsever et al. (1978) as

$$\log Q_s = -0.97688 + 1.103091 \log A \tag{3}$$

Equation (3) is based on streamflow and sediment discharge data from 52 gauging stations operated by the former Electrical Power Resources Survey and Development Administration (EIE with its Turkish acronym) and uses A, watershed area (km^2) as the independent variable to calculate Q_s, mean annual sediment amount (10^6 t/year).

Method 4: The ratio of reservoir storage volume to the watershed area is referred to as theoretical storage depth. Sediment deposited in reservoirs depends on hydrological, meteorological, topographical characteristics of the watershed as well as its land use and land cover. A family of curves was given depending on all these characteristics to calculate sediment volume accumulated (Bogardi 1974).

Method 5: Percentage of siltation per unit storage volume plotted against the reservoir volume per unit annual sediment load can be used for calculating the siltation in reservoirs. This is a curve starting with a maximum initial value of siltation ratio decreasing exponentially to become an asymptote to zero (Bogardi 1974).

Method 6: Investigations on the reservoir siltation have shown that the rate of siltation or storage volume loss depends on the flow recharging the reservoir. From results of observations made on a number of reservoirs, the relation of annual siltation loss per unit catchment area against the ratio of the original storage volume to the entire annual inflow has an increasing curve and a steady-state portion. These observations can be presented as alternatives to compute annual siltation per unit watershed area (Bogardi 1974).

3.2 Case Study

In order to compare the aforementioned empirical approaches, Buyukcekmece dam reservoir is selected as an application area (Fig. 1). Büyükcekmece dam reservoir is one of the most important domestic and sanitary water sources for the greater

Fig. 1 Buyukcekmece dam reservoir watershed

metropolitan area of Istanbul, Turkey. Its catchment area is 620 km^2 with an average elevation changing between 80 and 90 m. Plain, particularly agricultural areas are dense (80 %) whereas woodlands are poor (20 %) in the watershed. Ten-percent of plain areas have been urbanized.

Previous studies on Buyukcekmece dam reservoir showed that a 100-year dead volume was estimated as 24×10^6 m^3 by DAMOC (1971) and as 20×10^6 m^3 by DSI (1974). 134,000 and 13.4×10^6 m^3 dead volumes at annual and 100-year time scales, respectively, were calculated using bathymetry maps of years 1943 and 1966. The dead volume was calculated as 186,000 and 18.6×10^6 m^3 for annual and 100-year time period, respectively, when the sediment yield criterium (300 m^3 per km^2-year) used by DSI (1974) for catchments in Turkey is considered. Due to the poorly gauged data and in order to be on the safe side, DAMOC (1971) and DSI (1974) recommended 24×10^6 and 20×10^6 m^3, respectively, for the dead volume. Dead storage volumes calculated by the above mentioned methods and compiled from the related studies are presented in Table 2.

In the meantime, Kapdasli et al. (1996) made bathymetric measurements in the dam reservoir by using an echo-sounder device. A volume of 15.3×10^6 m^3 was calculated from this measurement and its comparison to the 1966 bathymetry map. Siltation corresponding to the 30-year period from 1966 to 1996 occurred mainly after 1985 when the river was separated from the Marmara Sea by the dam construction.

Results from the reports of DAMOC (1971) and DSI (1974) and empirical equations showed that the 100-year sediment volume varied from 5×10^6 to 87.5×10^6 m^3. The average value of sediment amount obtained from all these methods is 31×10^6 m^3. When the minimum and maximum values are considered outliers and not used, average sediment amount becomes 18.8×10^6 m^3 for the 100-year period. This value is clearly low comparing the value of 15.3×10^6 m^3 calculated for 30-year period from depth observation of Kapdasli et al. (1996).

Table 2 Dead volumes of Buyukcekmece dam reservoir for 100-year period

Method	Reference	Dead volume (10^6 m^3)
Lake bathymetry maps	(1)	13.4
Sediment yield criteria	(1)	18.6
DSI	(1)	20.0
DAMOC	(2)	24.0
Method 1	(3)	20.0
Method 2	(4)	87.0
Method 3	(5)	5.0
Method 4	(6)	12.4
Method 5	(6)	87.5
Method 6	(6)	23.0

(1) DSI (1974), (2) DAMOC (1971), (3) Erkek and Agiralioglu (1993), (4) Gogus and Adiguzel (1991), (5) Yurtsever et al. (1978), (6) Bogardi (1974)

3.3 Transport Capacity of Overland Flow

Sediment transport capacity of overland flow is the maximum flux of sediment that flow is capable to transport. All process-based soil erosion models contain a sediment transport equation. Many existing models use either a bed load or a total load formula originally developed for rivers while some use simple empirical formulas.

Early approaches to the sediment transport capacity have used the shear stress (Yalin 1963), stream power (Bagnold 1966), or unit stream power (Yang 1972). Alonso et al. (1981), after comparison of nine sediment transport formulas, suggested the use of Yalin (1963) equation in computing the sediment transport capacity for overland flow. Nearing et al. (1989) used a simplified function of the hydraulic shear stress acting on the soil for calculating the sediment transport capacity of flow. Tayfur (2002) analysed those approaches and concluded that the unit stream power could be selected for the simulation of unsteady state erosion and sediment transport from very mild bare slopes and, under low rainfall intensities, it could also be used to simulate loads from mild and steep slopes. For the very steep slopes, the shear stress and stream power models could be used. The stream power and the shear stress models could also be employed in order to simulate sediment load from mild and steep slopes under high rainfall intensities. In a similar study, Zhang et al. (2009) concluded that the stream power seems to be a preferred approach for estimating transport capacity for steep slopes.

A general relationship between variables that affect the sediment transport capacity was developed by Julien and Simons (1985) as

$$q_s = \alpha\, S^{\beta}\, q^{\gamma}\, r^{\delta} \left(1 - \frac{\tau_c}{\tau}\right)^{\varepsilon} \tag{4}$$

where q_s is sediment discharge, S slope, q discharge, r rainfall intensity, τ_c critical shear stress, τ actual shear stress, α a coefficient and β, γ, δ, ε exponents to be determined from laboratory or field experiments. When τ_c remains very small compared to τ and when it is considered that the sediment transport capacity of turbulent flow in deep channels is not a function of rainfall then Eq. (4) reduces to

$$q_s = \alpha\, S^{\beta}\, q^{\gamma} \tag{5}$$

Prosser and Rustomji (2000) addressed the same equation for the sediment transport capacity. As q is a function of the upslope contributing area, sediment discharge is evaluated completely by topographic factors. From examination of many studies based upon flumes, laboratory and field plots and rivers, β and γ, exponents of S and q in Eq. (5), were found to be bounded by 0.5 and 2.0, as lower and upper limits, respectively. When one single combination is desired, a median value of 1.4 can be used for both exponents. The sediment transport capacity (T_c) of overland flow was also found to be proportional to the overland flow discharge (q) only, as $T_c \sim q^{\gamma}$, where γ ranged between 1.2 and 1.5. Then the sediment concentration (C_s) in the runoff becomes $C_s \sim q^{\gamma-1}$ (Novotny and Chesters 1989).

Abrahams et al. (1998) obtained a regression equation for the transport capacity of overland flow by combining results of laboratory experiments.

This analysis shows that the transport capacity is a challenging issue in the sediment transport models. Models might behave differently for each transport capacity equation used. Therefore, selection of the transport capacity equation is a major issue in the development of the sediment transport models.

3.4 Data Requirement

Data needed for a model dramatically increase with how complex the model is. Distributed models, in particular, need more data than any other type of the models. Erosion and sediment transport models contain non-physical parameters. It is already a difficult task to collect erosion and sediment data from a watershed or from a specific hillslope in a watershed. Data collection becomes much harder for detailed models such as one considering the microtopographical details of rill-interrill area distinction.

Input data for erosion and sediment transport models include outputs from hydrological (rainfall-runoff) models. Therefore, in order to be able to run any erosion model it is first required to run a hydrological model so that the hydrological outputs can be supplied as input for the erosion model. Before an erosion model is run, either a hydrological model must be run or the outputs of the hydrological model must be supplied.

3.5 Calibration and Validation

A process-based rainfall-runoff-sediment transport mathematical model developed by Aksoy et al. (2011) is presented below to demonstrate challenges in calibration and validation stages of such models. The model is process-based and deterministic. It uses the two-dimensional mass conservation equation simplified with kinematic wave approach. The model considers the rill-interrill area interaction as in Fig. 2, therefore has two components; one for interrill area feeding the rill, and the other is for the rill itself. The model is solved numerically.

Rainfall over an interrill area becomes runoff first over the interrill area (1 in Fig. 2) and flows towards rills. When the runoff reached a rill (2 in Fig. 2) it joins the runoff in the rill to run further downstream to the channel (3 in Fig. 2). The flow joins, in the channel, to the channel flow to run further downstream to reach a lake or the ocean (4 in Fig. 2). Runoff over interrill area (1 in Fig. 2) might reach the channel (3 in Fig. 2) directly without getting the rill (2 in Fig. 2) depending on the lateral slope.

Model parameters come from hydrologic (rainfall-runoff) and sediment transport components of the model. They are calibrated through a least square method to

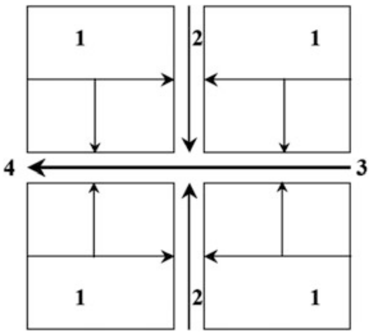


Fig. 2 Layout of interrill area, rill and channel; microtopographical scheme over the hillslope used for the 2-dimensional model of Aksoy et al. (2011)

minimize the difference between observed and calculated values of flow and sediment discharges by using data from a laboratory rainfall simulator and an erosion flume (Aksoy et al. 2012, 2013; Arguelles et al. 2013). Model parameters are listed in Table 3. Two (β and ε) out of 10 parameters were fixed as frequently made in literature (Singh and Regl 1983; Lopes and Lane 1988; Laguna and Giraldez 1993; Franchini 1994; Govindaraju 1995) for the sake of making calibration simple. Thus, calibration stage of the model has eight parameters; four for rainfall-runoff component, four for sediment transport component. The rainfall-runoff parameters are Chezy coefficients for interrill area and rill (C_Z and C_{ZR}), limit infiltration capacity and recession parameter of infiltration model (f_c and k_h). Parameters related to the sediment transport component are rainfall erosion parameter (α), runoff erosion parameters in the interrill area (σ and η), and runoff erosion parameter in the rill (σ_R).

Table 3 Model parameters

Parameter	Dimension ($M^xL^yT^z$)	Definition	Remarks
C_Z	$L^{1/2}T^{-1}$	Chezy coefficient in interrill area	
β	–	Exponent	Taken as $\beta = 1$
α	$ML^{-(\beta+2)}T^{\beta-1}$	Erodibility factor of soil under rainfall effect	Dimension depends on β (ML^{-3} for $\beta = 1$)
σ	L^{-1}	Erosion coefficient in interrill area	
ε	–	Exponent	Taken as $\varepsilon = 1$
η	$M^{1-\varepsilon}L^{\varepsilon-1}T^{2\varepsilon-1}$	Detachability factor of soil under flow effect	Dimension depends on ε (T for $\varepsilon = 1$)
C_{ZR}	$L^{1/2}T^{-1}$	Chezy coefficient in rill	
σ_R	L^{-1}	Erosion coefficient in rill	
f_c	LT^{-1}	Limit infiltration capacity	
k_h	T^{-1}	Infiltration model recession parameter	

In the calibration procedure, rainfall-runoff parameters were determined in such a way that the hydrograph is best approximated while the sediment transport parameters were fitted by simulating the sedigraph. Calibration is based on 32 out of 40 experimental data sets taken from a rainfall simulation study performed at laboratory scale (Aksoy et al. 2012). Eight experiments were used for validation of the model.

It is important to calibrate rainfall-runoff and sediment transport components separately as quite high number of combinations can be generated between so many high numbers of parameters. This might create a well done hydrograph while the sedigraph performs poorly. Oppositely, the sedigraph can be well preserved while the hydrograph showed a poor performance. Therefore, hydrograph and sedigraph are calibrated separately to achieve considerably well simulations.

While doing the calibration, it is first checked if the total flow is preserved. At the same time, time varying structure of the hydrograph was taken into consideration; i.e., ascension curve, steady-state period and recession curve of the hydrograph were paid particular attention. This has been achieved by calibration of rainfall-runoff parameters, Chezy coefficients in the interrill area and in the rill (C_Z and C_{ZR}), and parameters of the infiltration model (f_c and k_h). Similarly, sediment transported from the simulated hillslope was approximated by calibrating parameters in the sediment transport component of the model. These parameters are rainfall erosion parameter (α), runoff erosion parameters for the interrill area (σ and η), and runoff erosion parameter in the rill (σ_R). Similar to the rainfall-runoff component, parameters were calibrated in such a way that both the time varying structure of the sedigraph and the total amount of transported sediment are conserved. The calibration ended with a set of calibration parameters as given in Table 4.

An obvious challenge seen from Table 4 is that parameters change within a wide range of values. This has been the case in literature as summarized in Table 5. In one hand, among the calibration parameters of the hydrologic component of the model are soil characteristics and infiltration capacity, both physically meaningful. On the other hand, parameters related to the sediment transport do not have such clear physical definition. In Table 5, literature values compiled for sediment transport models are listed. As can be seen, these parameters have various values; even the same parameter might change within two orders of magnitude. For example; σ was taken 0.30 m^{-1} by Singh and Regl (1983) while it was given a value of 10 m^{-1} by Govindaraju (1995). Considering Foster (1982), Tayfur (2001) proposed an interval changing between 3 and 33 m^{-1}. However, it can be noted that parameters calibrated as in Table 4 are in accordance with the literature data in Table 5.

Table 4 Calibration of model parameters

r (mm h^{-1})	S_y (%)	S_x (%)	S (%)	C_Z (m$^{1/2}$ s^{-1})	C_{ZR} (m$^{1/2}$ s^{-1})	f_c (mm h^{-1})	k_h (s^{-1})	α (kg m^{-3})	σ (m^{-1})	σ_R (m^{-1})	η (s)
45	5	5	7.07	5.81	8.53	0.81	1.39E-02	16.0	2.88	3.51	0.190
45	5	10	11.18	6.34	7.44	10.04	3.10E-03	16.0	3.50	3.72	0.200
45	5	20	20.62	0.88	5.20	0.50	5.74E-03	35.0	3.99	4.44	0.234
45	10	10	14.14	0.95	6.72	0.50	5.10E-03	17.0	3.31	3.62	0.210
45	10	15	18.03	2.40	5.80	0.50	2.49E-03	21.0	3.80	4.25	0.220
45	15	15	21.21	0.60	5.00	3.02	4.10E-03	28.0	4.04	4.20	0.220
45	15	20	25.00	0.44	4.10	0.50	5.55E-03	60.0	5.00	4.62	0.160
45	20	20	28.28	0.29	3.30	1.55	5.48E-03	53.0	8.92	9.52	0.261
65	5	5	7.07	5.59	6.09	4.43	9.10E-03	16.0	3.13	4.11	0.192
65	5	10	11.18	2.99	9.58	3.92	3.60E-03	17.0	4.29	4.25	0.200
65	5	20	20.62	0.82	3.59	0.50	1.60E-02	38.0	0.30	7.90	0.220
65	10	10	14.14	2.50	9.13	4.32	2.70E-03	9.0	8.59	4.29	0.207
65	10	15	18.03	3.14	4.89	2.79	2.91E-03	13.0	0.40	5.20	0.220
65	15	15	21.21	1.15	7.19	0.50	6.16E-03	72.0	0.07	0.28	0.222
65	15	20	25.00	0.65	3.29	0.50	3.88E-03	81.0	3.32	15.54	0.230
65	20	20	28.28	0.55	2.85	0.50	4.53E-03	84.0	3.47	13.60	0.240
85	5	5	7.07	5.14	8.28	7.16	4.10E-03	32.0	3.00	3.60	0.193
85	5	10	11.18	5.94	8.53	0.50	1.72E-03	42.0	4.55	3.17	0.197
85	5	20	20.62	1.60	5.09	0.50	7.53E-03	78.0	3.20	3.70	0.225
85	10	10	14.14	1.49	5.11	0.50	2.09E-03	37.0	0.15	3.46	0.210
85	10	15	18.03	1.41	5.04	0.50	2.20E-03	40.0	0.10	2.80	0.220
85	15	15	21.21	1.98	4.19	0.50	2.56E-03	47.0	0.01	9.00	0.230
85	15	20	25.00	1.31	3.09	4.76	4.82E-03	117.0	0.11	8.43	0.240

(continued)

Table 4 (continued)

r (mm h^{-1})	S_y (%)	S_x (%)	S (%)	C_Z (m$^{1/2}$ s^{-1})	C_{ZR} (m$^{1/2}$ s^{-1})	f_c (mm h^{-1})	k_h (s^{-1})	α (kg m^{-3})	σ (m^{-1})	σ_R (m^{-1})	η (s)
85	20	20	28.28	1.17	2.70	7.70	3.97E-03	114.0	0.07	9.87	0.250
105	5	5	7.07	3.92	8.28	0.50	1.69E-03	5.0	2.91	3.67	0.190
105	5	10	11.18	5.59	4.09	0.50	4.12E-03	45.0	0.04	0.17	0.210
105	5	20	20.62	3.59	3.09	0.50	2.81E-03	56.0	0.01	0.98	0.240
105	10	10	14.14	2.84	5.84	10.62	1.83E-02	48.0	0.05	0.22	0.220
105	10	15	18.03	2.60	5.09	0.50	2.29E-03	58.0	0.06	0.10	0.230
105	15	15	21.21	1.28	5.09	4.29	5.55E-03	73.0	2.54	14.70	0.250
105	15	20	25.00	2.28	3.18	0.50	5.56E-03	157.0	11.20	1.72	0.260
105	20	20	28.28	1.60	2.47	0.59	2.31E-03	172.0	10.00	1.30	0.270

Table 5 Sediment transport model parameters existing in the literature

Reference	α (dimension/unit changes with β)	β	σ (m^{-1})	σ_R (m^{-1})	η (dimension/unit changes with ε)	ε
(1)				3–33		
(2)		2	0.30	0.03		1.5
(3)		2				1.5
(4)					0.001–0.5 $kg^{-1/2}m^{1/2}s^2$	1.5
(5)		1				1.5
(6)		2	0.10	0.01–1		
(7)			10			1
(8)	104–106 kg^{-4} s^{-1} (for β = 2)	1–2	0.36			
(9)	0.0006–0.0086 kg^{-2} mm^{-1} (for β = 1)	1–2	3–33			
(10)	1 kg^{-4} s^{-1}	2	1.3		0.06 s	1

(1) Foster (1982), (2) Singh and Regl (1983), (3) Lopes and Lane (1988), (4) Kavvas and Govindaraju (1992), (5) Laguna and Giraldez (1993), (6) Franchini (1994), (7) Govindaraju (1995), (8) Sharma (1998), (9) Tayfur (2001), (10) Aksoy et al. (2003)

Hydrographs and sedigraphs are calculated by the model using calibrated parameters in Table 4. Figure 3 shows an example for calibration. It is seen that hydrograph and sedigraph are well preserved. Although higher differences between experimental data and fitted model are observed in the sedigraph, it is considered a well done calibration when calibrations reported in the literature are analyzed. See, for instance, SHESED model by Wicks (1988), Wicks et al. (1992), Bathurst et al. (1995); MULTSED model by Wicks et al. (1988); and WESP model by Lopes (1987) and Lopes and Lane (1988).

One challenge that Table 4 arises is the uncertainty in the parameters taking values in a wide range. Assigning a representative value for each individual parameter becomes a hard decision to make. For example, the average cannot be a

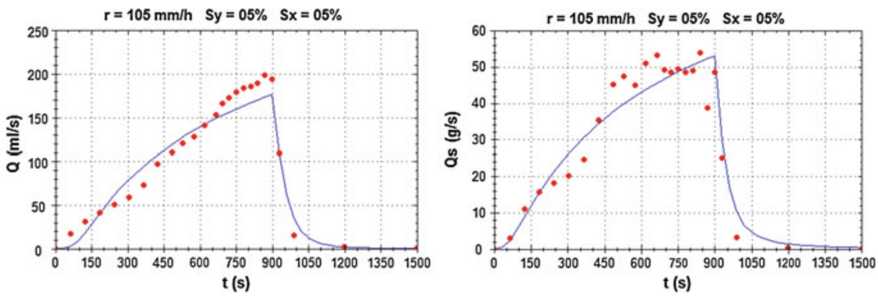


Fig. 3 Calibrated hydrograph and sedigraph for the experiment of r = 105 mm h^{-1}, S_y = 5 %, S_x = 5 % for calibration parameters as in Table 4

Table 6 Experiment characteristics and parameters used for the validation of the model

r(mm h^{-1})	S_y (%)	S_x (%)	S(%)	C_Z (m$^{1/2}$ s^{-1})	C_{ZR} (m$^{1/2}$ s^{-1})	f_c (mm h^{-1})	k_h (s^{-1})	α(kg m^{-3})	σ (m^{-1})	σ_R (m^{-1})	η (s)
85	10	20	22.36	2.43	4.58	2.34	5.19E-03	73.4	3.03	5.00	0.235

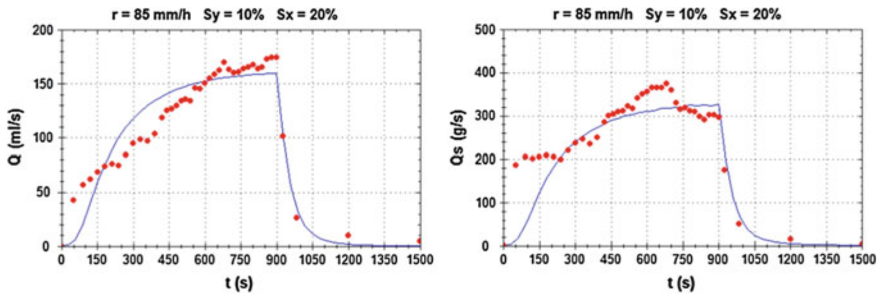


Fig. 4 Validated hydrograph and sedigraph for the experiment of r = 85 mm h^{-1}, S_y = 10 %, S_x = 20 % for validation parameters as in Table 6

good representative value to select. Therefore, it was investigated if the parameters can be linked to any such physical variables as rainfall intensity and topographical slope. This investigation showed that C_Z and C_{ZR} among the parameters from the rainfall-runoff component, and α and η from the sediment transport component were well regressed on rainfall intensity (r) in mm h^{-1}and slope (S) in percent as follows.

$$C_Z = 2.571 + 0.0522r - 0.2047S \tag{6}$$

$$C_{ZR} = 5.896 + 0.0257r - 0.1566S \tag{7}$$

$$\alpha = -65.92 + 0.6471r + 3.7692S \tag{8}$$

$$\eta = 0.16222 + 0.000098r + 0.002872S \tag{9}$$

The rainfall-runoff-sediment transport model was validated by using parameters as calibrated in Table 4. C_Z, C_{ZR}, α and η were regressed on rainfall intensity and slope as given in Eqs. (6–9) while average values were used for the remaining four parameters. As decided previously two parameters were taken constant (β = 1 and ε = 1). Parameters used are listed in Table 6, and validated hydrograph and sedigraph are given in Fig. 4 from which it can be said that the model satisfactorily performed.

4 Extension of Sediment Transport Models to a Sediment-Bound Pollutant Transport Model

The content of a pollutant in a soil is usually given in grams of the pollutant in grams of the soil. Non-point pollution is caused by mankind's activities on the land and differs from the natural erosion and sediment movement. For example; erosion and sediment transport caused by cutting a forest down is considered pollution, while a mudslide, caused by an earthquake, is not. Sediment concentrations two orders of magnitude lower than the natural erosion are not tolerable if they are caused by non-point pollution. Pollutant or nutrient yield can be calculated by multiplying the sediment yield with a potency factor, which is pollutant content of the sediment. However, for detailed studies, even use of lumped models is avoided in water quality studies as the delivery process is not such simple because of that related parameters represent a hydrologic stochastic process. It is therefore suggested to take the stochastic structure of the nonpoint pollution processes into account and also to establish the statistical characteristics of the processes (Novotny and Chesters 1989).

Sediment yield of a stream is strongly related to the flow. Flow is monitored in streams more frequently than the sediment concentration or phosphorus loads. Therefore, relations between flow and sediment or phosphorus are usually based on some regression equations. For example; sediment-turbidity relationship is used to convert a time-series of turbidity to suspended sediment concentration. If turbidity is missing for a period but flow has been measured at that period, the suspended sediment-flow relationship can be used to fill the gaps in the data (Green et al. 1999).

Phosphorus transported by the flow is much more than that associated with the soil since phosphorus is mainly associated with finer particles (Quinton 1999). It is known that phosphorus mainly moves with sediment by being attached to the surface of sediment particles. Therefore, it is reasonable to assume the sediment transport process as an indicator of phosphorus transport. Chemical properties are other factors that should be taken into account in the soil detachment processes, yet none of the existing models do so.

Akan (1987) studied pollutant washoff by overland flow on impervious surfaces. Ashraf and Borah (1992) worked on the modelling of pollutant transport in runoff and sediment. Yan and Kahawita (1997, 2000) and Wallach et al. (2001) studied modelling pollutants in the overland flow at the hillslope scale.

A model called Sediment-Phosphorus-Nitrogen-Model (SPNM) was developed by Williams (1980) for simulating contribution of agriculture to water pollution. SPNM was designed to predict sediment, P, and N yields for individual storms and to route these yields through streams. The model computes the total sediment yield predicted by the Modified Universal Soil Loss Equation (MUSLE). The P model predicts average annual P yields. The N model simulates both organic and inorganic N yields associated with the sediment and runoff. The organic N model has the same structure as the P model because both N and P are transported with sediment. The organic N tends to associate with fine clay whereas P tends to associate with coarse clay and silt as well as fine clay. The nitrate concentration in surface and subsurface flow are modeled separately. SPNM gave good results for sediment yield. Results for nutrients were found realistic.

AGNPS (Young et al. 1989) has a subcomponent for estimating P, N and chemical oxygen demand (COD). Chemical transport calculations are divided into soluble and sediment adsorbed phases. Nutrient yield in the sediment-adsorbed phase is obtained by multiplying the total sediment yield in a cell by the nutrient content in the field soil and the enrichment ratio, which is a function of sediment yield. Soluble nutrient yield is estimated by multiplying total runoff by the mean concentration of the nutrient at the soil surface during runoff and an extraction coefficient of nutrient for movement into runoff.

SHETRAN (Ewen et al. 2000) is a reactive solute transport model. Three main components in SHETRAN are water flow, sediment transport and solute transport. Flow is assumed not to be affected by sediment transport and sediment transport not to be affected by solute transport. Therefore, the three components are independent of each other. SHETRAN models a single complete river basin. It has a stream link and column structure. River network is modeled as stream links and the rest of the basin is modeled as a set of columns. Transport along the links and vertical transport in the columns are the two main movements. There is also lateral movement between cells in neighbouring columns. Later, Birkinshaw and Ewen (2000) developed a nitrogen transformation component and integrated it into the SHETRAN.

5 Conclusion

Sediment transport as a result of erosion in hydrological watersheds is an important challenging process against which people take measures. It has a particular practical importance as water resources structures are directly affected from the sediment transport by siltation of the reservoirs. Sediment transport has always been an interesting topic to study not only because of its importance in engineering practice but also due to its complex mechanism. One particular importance is determination of the amount of sediment eroded and transported within hydrological watersheds; over the hillslopes or within the concentrated flow courses in watersheds. For this aim, sediment transport is monitored and sampled from time to time. However, in order to arrive at conclusive deterministic relationships between flow and sediment transport, simultaneous records of runoff and sediment discharge are needed for long periods of time. In this case, monitoring and sampling become hard, expensive and time-consuming. Therefore, for quantification of sediment transport, computational methods have been developed for theoretical and practical purposes.

Sediment transport models are either data-based empirical models or they are based on simple concepts or complex processes. The former can make rough estimates order of magnitude different than each other while the latter is expected to approach the reality better with a higher cost for parameterization and data requirement. This becomes a challenge when the there is either not enough data or no data at all in case of ungauged basins. Another challenge can be related to the transport capacity of overland flow that initiates erosion and sediment transport within watershed and water courses. Difficulties in the calibration and validation stages of models are other issues to be considered.

Acknowledgments The German Alexander von Humboldt (AvH) Foundation and IHP/HWRP Secreteriat supported the author to attend the 6th International Conference on Water Resources and Environment Research organized by Federal Institute of Hydrology, Germany and held from 3 to 7 June 2013 in Koblenz, Germany. This chapter was derived from the author's presentation in this conference. However, material and method in this chapter are mostly based on the international project 'Development of a hillslope-scale sediment transport model' bilaterally supported by TUBITAK (The Scientific and Technological Research Council of Turkey, project no. 108Y250) and Korea Research Foundation (KRF). Assistance by Dr. Ebru Eris, Ege University, Izmir, Turkey, to the author for preparation of this chapter must also be acknowledged.

References

Abrahams AD, Li G, Krishnan C, Atkinson JF (1998) Predicting sediment transport by interrill overland flow on rough surfaces. Earth Surf Proc Land 23:1087–1099

Akan AO (1987) Pollutant washoff by overland flow. ASCE J Environ Eng 113(4):811–823

Aksoy H, Kavvas ML (2001) A physically based erosion and sediment transport component for watershed hydrologic models. Hydrologic Research Laboratory, Department of Civil and Environmental Engineering, University of California, Davis, California

Aksoy H, Kavvas ML (2005) A review of hillslope and watershed scale erosion and sediment transport models. Catena 65:247–271

Aksoy H, Kavvas ML, Yoon J (2003) Physically-based mathematical formulation for hillslope-scale prediction of erosion in ungauged basins. IAHS Publ 279:101–108

Aksoy H, Akar T, Unal NE (2004) Wavelet analysis for modelling suspended sediment discharge. Nord Hydrol 35(2):165–174

Aksoy H, Unal NE, Gedikli A, Cokgor S, Eris E, Yilmaz M (2011) Development of a hillslope-scale sediment transport model. TUBITAK project no. 108Y250, Final report, December 2011, p 173 (in Turkish)

Aksoy H, Unal NE, Cokgor S, Gedikli A, Yoon JY, Koca K, Inci SB, Eris E (2012) A rainfall simulator for laboratory-scale assessment of rainfall-runoff-sediment transport processes over a two-dimensional flume. Catena 98:63–72

Aksoy H, Unal NE, Cokgor S, Gedikli A, Yoon JY, Koca K, Inci SB, Eris E, Pak G (2013) Laboratory experiments of sediment transport from a bare soil with rill. Hydrol Sci J 58 (7):1505–1518

Alonso CV, Neibling WH, Foster GR (1981) Estimating sediment transport capacity in watershed modelling. Trans ASAE 1211–1220:1226

Araujo JC, Güntner A, Bronstert A (2006) Loss of reservoir volume by sediment deposition and its impact on water availability in semiarid Brazil. Hydrol Sci J 51(1):157–170

Arguelles ACC, Jung M, Pak G, Aksoy H, Kavvas ML, Yoon J (2013) Modelling of overland flow using areally-averaged local-scale interrill and rill flow equations. Water Sci Technol 68 (5):1188–1194

Ashraf MS, Borah DK (1992) Modelling pollutant transport in runoff and sediment. Trans ASAE 35(6):1789–1797

Baban SMJ, Yusof KW (2001) Modelling soil erosion in tropical environments using remote sensing and geographical information systems. Hydrol Sci J 46(2):191–198

Bagnold RA (1966) An approach to the sediment transport problem for general physics. U.S. Geol Surv Prof Pap 442 1:37

Bathurst JC, Wicks JM, O`Connell PE (1995) The SHE/SHESED basin scale water flow and sediment transport modelling system. In: Singh VP (ed) Computer models of watershed hydrology. Water Resources Publications, Littleton, pp 563–594

Birkinshaw SJ, Ewen J (2000) Nitrogen transformation component for SHETRAN catchment nitrate transport modelling. J Hydrol 230:1–17

Bogardi J (1974) Sediment transport in alluvial streams. Akademiai Kiado, Budapest

Bryan RB (2000) Soil erodibility and processes of water erosion on hillslope. Geomorphology 32:385–415

DAMOC (1971) Master plan and feasibility report for water supply and sewarage for the Istanbul region. Prepared for WHO, as executing agency for UN Development Program

DSI (1974) Development and planning of Buyukcekmece lake for water supply to Istanbul city. Report of DSI Istanbul XIVth Regional Directorate, Istanbul (in Turkish)

Erkek C, Agiralioglu N (1993) Water resources engineering. Beta, Istanbul (in Turkish)

Ewen J, Parkin G, O'Connell PE (2000) SHETRAN: distributed river basin flow and transport modelling system. ASCE J Hydrol Eng 5(3):250–258

Foster GR (1982) Modelling the erosion process. In: Haan CT, Johnson HP, Brakensiek DL (eds) Hydrologic modelling of small watersheds, ASAE monograph no. 5, St. Joseph, MI, pp 297–380

Franchini M (1994) Combined analytical solution of overland flow and sediment transport. Water Resour Manage 8:225–238

Garde RJ, Ranga Raju KG (1977) Mechanics of sediment transportation and alluvial stream problems. Wiley Eastern, New Delhi

Gogus M, Adiguzel F (1991) Sedimentation of reservoirs in Turkey. Proc XIth Civil Eng Tech Congr Turk Chamber Civil Eng Istanbul 1:369–383 (in Turkish)

Govindaraju RS (1995) Non-dimensional analysis of a physically based rainfall-runoff-erosion model over steep slopes. J Hydrol 173:327–341

Govindaraju RS, Kavvas ML, Tayfur G, Krone R (1992) Erosion control of decomposed granite at Buckhorn Summit. Final project report. University of California, Davis

Green TR, Beavis SG, Dietrich CR, Jakeman AJ (1999) Relating stream bank erosion to in-stream transport of suspended sediment. Hydrol Process 13:777–787

Julien PY, Simons DB (1985) Sediment transport capacity of overland flow. Trans ASAE 28 (3):755–762

Kapdasli S, Aksoy H, Fer I, Mutlu T, Unal NE, Aydin Gakko A, Alp M (1996) Hydrology and sediment transport problem of Buyukcekmece Lake. Watershed Symposium, ISKI, ITU, Water Foundation, Istanbul, 1–3 June 1997 (in Turkish)

Kavvas ML, Govindaraju RS (1992) Hydrodynamic averaging of overland flow and soil erosion over rilled hillslopes. IAHS Publ 209:101–111

Kavvas ML, Chen ZQ, Dogrul C, Yoon JY, Ohara N, Liang L, Aksoy H, Anderson ML, Yoshitani J, Fukami K, Matsuura T (2004) Watershed environmental hydrology (WEHY) model based on upscaled conservation equations: hydrologic module. ASCE J Hydrol Eng 9(6):450–464

Kavvas ML, Yoon JY, Chen ZQ, Liang L, Dogrul EC, Ohara N, Aksoy H, Anderson ML, Reuters J, Hackley S (2006) Watershed environmental hydrology model: environmental module and its application to a California watershed. ASCE J Hydrol Eng 11(3):261–272

Laguna A, Giraldez JV (1993) The description of soil erosion through kinematic wave model. J Hydrol 145:65–82

Lopes VL (1987) A numerical model of watershed erosion and sediment yield. PhD thesis, The University of Arizona

Lopes VL, Lane LJ (1988) Modelling sedimentation processes in small watersheds. IAHS Publ 174:497–508

Merritt WS, Latcher RA, Jakeman AJ (2003) A review of erosion and sediment transport models. Environ Model Softw 18:761–799

Morgan RPC, Quinton JN, Smith RE, Govers G, Poesen JWA, Auerswald K, Chisci G, Torri D, Styczen ME (1998) The European soil erosion model (EUROSEM): a dynamic approach for predicting sediment transport from fields and small catchments. Earth Surf Proc Land 23:527–544

Nearing MA, Foster GR, Lane LJ, Finkner SC (1989) A process-based soil erosion model for USDA-water erosion prediction project technology. Trans ASAE 32(5):1587–1593

Novotny V, Chesters G (1989) Delivery of sediment and pollutants from nonpoint sources: a water quality perspective. J Soil Water Conserv 44(6):568–576 (November–December)

Phien HN (1981) Reservoir sedimentation with correlated inflows. J Hydrol 53:327–341

Phien HN, Arbhabhirama A (1979) A statistical analysis of the sediment accumulation in reservoirs. J Hydrol 44:231–240

Prosser IP, Rustomji P (2000) Sediment transport capacity relations for overland flow. Prog Phys Geogr 24(2):179–193

Quinton JN (1999) Detachment and transport of particle—bound P: process and prospects for modelling. Paper presented at the COST action on Phosphorus meeting, Cordoba, 13–15 May 1999

Sharma KD (1998) A physically-based sediment delivery model for arid regions. IAHS Publ 249:157–164

Simons DB, Senturk F (1992) Sediment transport technology; water and sediment dynamics. Water Resources Publications, Littleton

Singh VP (1995) Watershed modelling. In: Singh VP (ed) Computer models of watershed hydrology. Water Resources Publ., Highlands Ranch, Colorado, pp 1–22

Singh VP, Regl RR (1983) Analytical solutions of kinematic equations for erosion on a plane. I: rainfall of indefinite duration. Adv Water Resour 6:2–10

Smith RE (1981) A kinematic model for surface mine sediment yield. Trans ASAE, pp 1508–1514

Storm B, Jorgensen GH, Styczen M (1987) Simulation of water flow and soil erosion processes with a distributed physically-based modelling system. IAHS Publ 167:595–608

Tayfur G (2001) Modelling two-dimensional erosion process over infiltrating surfaces. ASCEJ Hydrol Eng 6(3):259–262

Tayfur G (2002) Applicability of sediment transport capacity models for nonsteady state erosion from steep slopes. ASCE J Hydrol Eng 7(3):252–259

Tingsanchali T, Lal NK (1992) A combined deterministic-stochastic model of daily sediment concentrations in a river. In: Proceedings sixth IAHR international symposium stochastic hydraulics. Taipei, pp 221–228

Wallach R, Grigorin G, Rivlin (Byk) J (2001) A comprehensive mathematical model for transport of soil-dissolved chemicals by overland flow. J Hydrol 247:85–99

Wicks JM (1988) Physically-based mathematical modelling of catchment sediment yield. Department of Civil Engineering, University of Newcastle Upon Tyne, Thesis submitted for the degree of doctor of philosophy

Wicks JM, Bathurst JC, Johnson CW, Ward TJ (1988) Application of two physically-based sediment yield models at plot and field scales. IAHS Publ 174:583–591

Wicks JM, Bathurst JC, Johnson CW (1992) Calibrating SHE soil-erosion model for different land covers. ASCE J Irrig Drainage Eng 118(5):708–723

Williams JR (1980) SPNM, a model for predicting sediment, phosphorus, and nitrogen yields from agricultural basins. AWRA Water Resour Bull 16(5):843–848

Wischmeier H, Smith DD (1978) Predicting rainfall erosion losses. Agriculture Handbook no. 537, USDA Science and Education Administration

Yalin MS (1963) An expression for bed-load transportation. ASCE J Hydraul Div 89(HY3):221–250

Yalin MS (1972) Mechanics of sediment transport. Pergamon Press, Braunschweig

Yan M, Kahawita R (1997) Modelling pollutant transport in overland flow with infiltration. In: Proceedings of the 27th congress of the IAHR, vol 1, pp 347–351, San Francisco, 10–15 Aug 1997

Yan M, Kahawita R (2000) Modelling the fate of pollutant in overland flow. Water Res 34 (13):3335–3344

Yang CT (1972) Unit stream power and sediment transport. ASCE J Hydraul Div 98 (HY10):1805–1826

Young RA, Onstad CA, Bosch DD, Anderson WP (1989) AGNPS: a nonpoint-source pollution model for evaluating agricultural watersheds. J Soil Water Conserv 44(2):168–173 (March–April)

Yurtsever Y, Goksu E, Alisik A, Ozenir C (1978) Analysis of average sediment yield of hydrological watersheds in Turkey. In: Proceedings of 1st erosion and sedimentation symposium. Ankara, pp 458–467

Zhang GH, Liu YM, Han YF, Zhang XC (2009) Sediment transport and soil detachment on steep slopes: I. Transport capacity estimation. Soil Sci Soc Am J 73(4):1291–1297

Zhang L, O'Neill AL, Lacey S (1996) Modelling approaches to the prediction of soil erosion in catchments. Environ Softw 11(1–3):123–133

Suspended Sediment Estimation Using an Artificial Intelligence Approach

Mustafa Demirci, Fatih Üneş and Sebahattin Saydemir

Abstract Forecasting of sediment concentration in rivers is a very important process for water resources assignment development and management. In this paper, a neural network approach is proposed to predict suspended sediment concentration from streamflow. A comparison was performed between artificial neural network, sediment rating-curve and multilinear regression models. It was based on a 5 years period of continuous streamflow, suspended sediment concentration and mean water temperature data of West Virginia, Little Coal River, Danville station operated by the United States Geological Survey. Based on comparison of the results, it is found that the artificial neural network model gives better estimates than the sediment rating-curve and multilinear regression techniques.

Keywords Suspended sediment · Forecasting · Neural network · Sediment rating curve · Multi-linear regression

1 Introduction

The assessment of the volume of sediment transported by a river is of vital interest in hydraulic engineering due to its importance in the design and management of water resources projects. The prediction of river sediment load constitutes an important issue in hydraulic and sanitary engineering. The sediment yield is usually calculated from the direct measurement of sediment concentration of river or from sediment transport equations with hydrological stations in basin outlet point. Sediment rating curves are largely used to estimate the sediment transport in river. However, traditional sediment rating curves are not able to provide sufficiently accurate results. A sediment rating curve is a relation between the sediment and

M. Demirci (✉) · F. Üneş · S. Saydemir
Engineering Faculty, Civil Engineering Department, Hydraulics Division, Mustafa Kemal University, 31040 İskenderun, Hatay, Turkey
e-mail: demircimustafa97@gmail.com; mdemirci@mku.edu.tr

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_6

river discharges. Such a relationship is usually established by a regression analysis, and the curves are generally expressed in the form of a power equation. McBean and Al-Nassri (1988), investigated suspended sediment rating curves and the practice of using sediment load versus discharge is shown to be misleading, since the goodness of fit implied by this relation is spurious.

Artificial Neural Network (ANN) is a flexible mathematical structure, having strong similarity to the biological brain and therefore a great deal of the terminology is borrowed from neuroscience. Artificial Neural Networks (ANNs) are gaining popularity, especially over the last few years, in terms of hydrological applications. In the hydrological forecasting context, recent experiments have reported that ANNs may offer a promising alternative for rainfall–runoff modelling (Sudheer et al. 2002; Wilby et al. 2003; Solomatine and Dulal 2003), streamflow prediction (Raman and Sunilkumar 1995; Zealand et al. 1999; Chibanga et al. 2003; Cigizoglu 2003; Kisi 2004a; Cigizoglu and Kisi 2005) and reservoir inflow forecasting (Saad et al. 1996; Jain et al. 1999). Üneş (2010b) developed an ANN model for dam reservoir level estimation. Toprak and Cigizoglu (2008) used ANN for predicting longitudinal dispersion coefficient in natural streams. Üneş (2010a) predicted density flow plunging depth in dam reservoir using the ANN. The last decade has witnessed a few applications of the artificial intelligence techniques in water resources forecasting (Hundecha et al. 2001; Tayfur 2002; Tayfur et al. 2003; Kisi 2004b). To the knowledge of the author, no work has been reported in the literature that addresses the application of the neuro-fuzzy approach for the estimation of suspended sediment. This provided an impetus for the present investigation. Jain et al. (1999) used a single ANN approach to establish sediment-discharge relationship and found that the ANN model could perform better than the rating curve. Tayfur (2002) developed an ANN model for sheet sediment transport and indicated that the ANN could perform as well as, in some cases better than, the physically-based models. Cigizoglu (2004) investigated the accuracy of a single ANN in estimation and forecasting of daily suspended sediment data. Kisi (2004c) used different ANN techniques for daily suspended sediment concentration prediction and estimation and he indicated that multi-layer perceptron could show better performance than the others. Kisi (2005) developed an ANN model for modeling suspended sediment and compared the ANN results with those of the rating curve and multilinear regression. Cigizoglu and Kisi (2006) developed some methods to improve ANN performance in suspended sediment estimation. Lohani et al. (2007), evaluated the performance of the conventional sediment rating curves, neural networks and fuzzy rule-based models using the coefficient of correlation, root mean square error and pooled average relative (underestimation and overestimation) errors (PARE) of sediment concentration. Demirci and Baltaci (2012), proposed a fuzzy logic approach to estimate suspended sediment concentration from streamflow. It was found that the fuzzy logic model gave better estimates than the other techniques.

In Lopes and Ffolliott (1993), data from a 455-acre clear-cut ponderosa pine forest watershed in northern Arizona were used to identify relationships between suspended sediment concentration and streamflow discharge. Scatter about the straight line relationship was found when all available pairs of suspended-sediment-concentration and streamflow measurements were used together. The effect of some of the variation was offset by subdividing the data set on the basis of streamflow generation mechanisms.

The main aim of this study is to analyze the performances of an adaptive ANN computing technique for daily sediment estimation. This study is concerned with the application of neural network for modeling suspended sediment concentration. This logic is used to develop discharge–sediment rating curves. The daily streamflow, temperature and suspended sediment time series data belonging to one station in USA are used.

2 Neural Networks

2.1 Artificial Neural Network

Artificial neural networks (ANNs) are based on the present understanding of the biological nervous system, though much of the biological detail is neglected. ANNs are massively parallel systems composed of many processing elements connected by links of variable weights. Of the many ANN paradigms, the back propagation network is by far the most popular (Lippman 1987). The network consists of layers of parallel processing elements, called neurons, with each layer being fully connected to the proceeding layer by interconnection strengths, or weights (W). Figure 1 illustrates a three-layer neural network consisting of layers i, j and k, with the interconnection weights W_{ij} and W_{jk} between layers of neurons. Initial estimated weight values are progressively corrected during a training process that compares predicted outputs to known outputs, and back propagates any errors (from right to left in Fig. 1) to determine the appropriate weight adjustments necessary to minimize the errors. The methodology used here for adjusting the weights is called “momentum back propagation”, and is based on the “generalized delta rule”, as

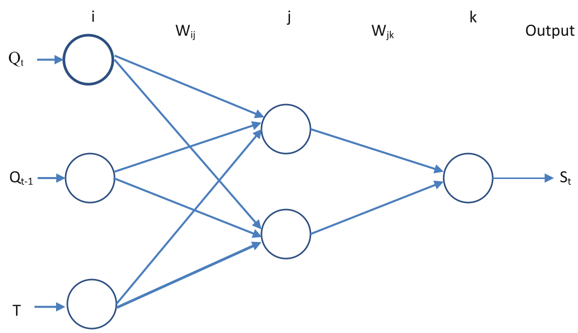


Fig. 1 An ANN architecture used for suspended sediment estimation (ref)

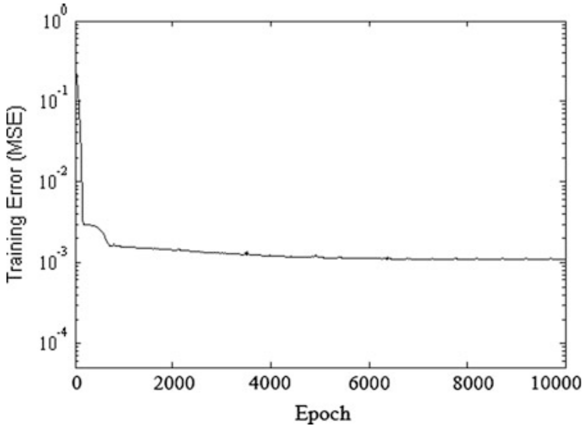


Fig. 2 The training error graph for the ANN model; training error, +: epoch (ref)

presented by Rumelhart et al. (1986). Throughout all ANN simulations, the adaptive learning rates were used for increasing the convergence velocity. The sigmoid and linear functions are used for the activation functions of the hidden and output nodes, respectively. The hidden layer node numbers of each model were determined after trying various network structures, since there is no theory yet to tell how many hidden units are needed to approximate a given function. The training of the ANN networks was stopped after 10,000 cycles, when the variation of error became sufficiently small. The error graph for an ANN model during training is shown in Fig. 2.

2.2 *Sediment Rating Curves (SRC)*

In the absence of manpower or automatic apparatus for frequent sampling, and laboratory facilities for analysis of numerous samples, many workers have utilized the rating-curve technique to estimate suspended sediment loads. A rating curve consists of a graph or equation relating sediment discharge or concentration to discharge, which can be used to estimate sediment loads from the streamflow record. The sediment rating curve generally represents a functional relationship of the form

$$S = aQ^b \tag{1}$$

in which Q is stream discharge and S is either suspended sediment concentration or yield. Values of a and b for a particular stream are determined from data via a linear regression between (log S) and (log Q). Equation (1) is usually combined with a streamflow duration curve to estimate the mean annual yield (Piest and Miller (1975)). A study by Campbell and Bauder (1940) on the Red River in Texas provides an early documented example of the use of sediment rating curves in the USA. They developed a 'silt rating curve' by plotting daily suspended sediment load against daily river flow on logarithmic coordinates.

2.3 Multi-linear Regression (MLR)

If it is assumed that the dependent variable Y is effected by m independent variables X_1, X_2, …, X_m and a linear equation is selected for the relationship between them, the regression equation of Y can be written as:

$$y = a + b_1x_1 + b_2x_2 + \cdots + b_mx_m \tag{2}$$

y in this equation shows the expected value of the variable Y when the independent variables take the values $X_1 = x_1$, $X_2 = x_2$, …, $X_m = x_m$.

The regression coefficients a, b_1, b_2, …, b_m are evaluated, similar to simple regression, by minimizing the sum of the e_{yi} distances of observation points from the plane expressed by the regression equation (Bayazıt and Oguz 1998).

$$\sum_{i=1}^{N} e_{yi}^2 = \sum_{i=1}^{N} (y_1 - a - b_1x_1 - b_2x_2 - \cdots - b_mx_m)^2 \tag{3}$$

In this study, the coefficients a, b_1, b_2, …, b_m are determined using least squares method.

3 Application and Results

The time-series data of Little Coal River, Danville Station located at West Virginia (USGS Station No. 03199000, latitude 38°04′47″, longitude 81°50′11″), operated by the USGS were used in the study. The location of the station is shown in Fig. 3. The drainage area at this site is 697,000 km^2. The gauge datum is 201 m above sea level. For this station, daily time-series of river flow, suspended sediment concentration and mean water temperature were downloaded from the USGS Web server.

The statistical parameters of streamflow, suspended sediment concentration, temperature data of Little Coal River station are shown in Table 1. In this table, S_x, C_{sx}, X_{max}, X_{min}, X_{ort} denote the standard deviation, the skewness coefficient, maximum, minimum and mean values. It can be seen from Table 1 that the S_x and C_{sx} coefficients for both the training and the testing period are very high. This shows the complexity of the streamflow—sediment interaction.

Fig. 3 The location of the Little Coal River, Danville station at West Virginia (USGS station no. 03199000)

Table 1 The statistical parameters of Little Coal River Danville station data

	Variables	T_{Max} (°C)	T_{Min} (°C)	T_{ort} (°C)	Q_s (m^3/s)	S (mg/L)
Training period	x_{max}	34.50	27.00	30.50	365.29	2,990.00
	x_{min}	0.50	0.50	0.50	0.42	0.00
	X_{mean}	15.74	12.92	14.33	9.25	82.62
	s_x	9.16	7.74	8.42	18.23	195.91
	c_{sx}	0.01	−0.04	−0.03	10.92	6.67
Testing period	x_{max}	30.50	26.00	28.25	222.85	2,090.00
	x_{min}	1.00	0.50	0.75	0.91	0.00
	x_{ort}	14.72	12.60	13.66	15.45	127.19
	s_x	7.94	7.19	7.54	21.33	219.20
	c_{sx}	0.06	0.04	0.05	4.58	3.96

T_{Max} Maximum temperature; T_{Min} Minimum temperature; T_{ort} Mean temperature; Q_s Mean streamflow; S Mean suspended sediment concentration

The input combinations used in this application to estimate suspended sediment values for Little Coal River station are (i) Q_t; (ii) Q_{t-1}; (iii) T_{ort}; where Q_t and T_{ort} represent, respectively, the streamflow and mean temperature at day t.

For 5 years data, results of modeling SRC, MLR, and ANN are shown as follows. For each model the minimum mean squared error (MSE), the total squared error (MAE) and correlation coefficients (R) are calculated between model predictions and observed values is calculated. Results are used to compare the performance of the model prediction and observation data. MSE and MAE was determined as follows:

$$\mathrm{MSE} = \frac{1}{\mathrm{N}}\left(\sum_{\mathrm{i}=1}^{\mathrm{N}} \mathrm{Yi}_{\mathrm{observed}} - \mathrm{Yi}_{\mathrm{forecast}}\right)^2 \quad (4)$$

and

$$\mathrm{MAE} = \frac{1}{\mathrm{N}}\sum_{\mathrm{i}=1}^{\mathrm{N}} \left|\mathrm{Yi}_{\mathrm{observed}} - \mathrm{Yi}_{\mathrm{forecast}}\right| \quad (5)$$

where N is the number of data sets and Yi sediment concentration data.

Using the data of mean water temperature, daily real-time streamflow and sediment concentration in the Little Coal River station, the best model was investigated and comparisons were made with the better results. As data for this study, 5 year data belonging to Little Coal River station has been used. With the using the 1827 data between 01 December 1975 until 01 December 1980, models are generated.

3.1 SRC Model Results

In Sediment rating curve (SRC) model, 1096 of 1827 data for the training, 731 data are divided for testing. Sediment rating curve for the training data (SRC) is shown in Fig. 4. The obtained sediment concentration data are compared with testing data and scatter plot is shown in Fig. 5.

In Fig. 5, the correlation coefficient was obtained as R = 0.785. In the test phase, sediment rating curve (SRC) is obtained and suspended sediment concentration scatter graph is shown. Values of sediment rating curve are seen to be spaced out from the actual values. The observed values are shown to be scattered for the results of the SRC for training data in Fig. 6 and for testing data in Fig. 7.

When scatter graphs for training and testing data are analyzed, SRC sediment concentration values show deviations between estimated values and the actual values. SRC values for training data are lower than the values given by the actual testing data values, SRC values for testing data are higher than the estimated values.

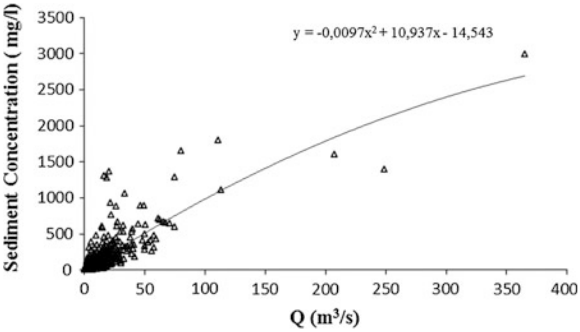


Fig. 4 Sediment rating curve for the training data (SRC)

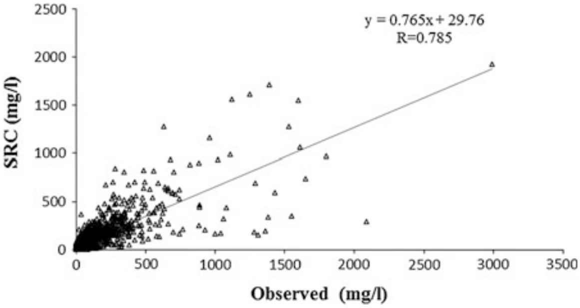


Fig. 5 SRC scatter graph for the observed data

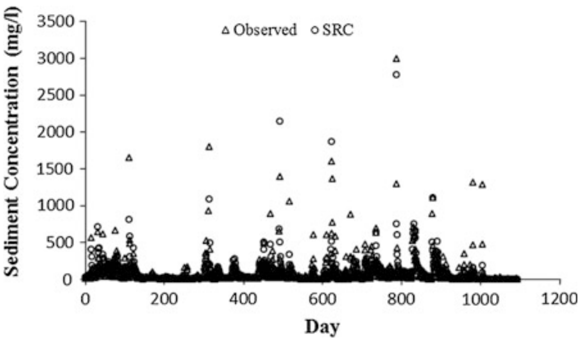


Fig. 6 Observed and SRC distribution graph for the training data

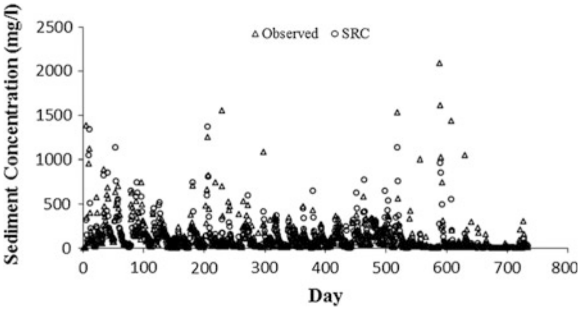


Fig. 7 Observed and SRC distribution graph for the testing data

3.2 MLR Model Results

For multiple linear regression (MLR), 5-years data are evaluated and the results are offered in figures. Distribution and scatter plots are shown for training data in Figs. 8 and 9 and for testing data in Figs. 10 and 11.

The correlation coefficient was obtained as R = 0.862 from Fig. 9. Although daily real-time suspended sediment concentration values are better than SRC values, the estimated results are worse than the observed actual values. In distribution

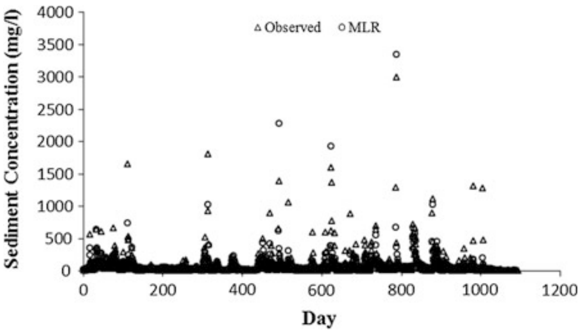


Fig. 8 Observed and MLR distribution graph for the training data

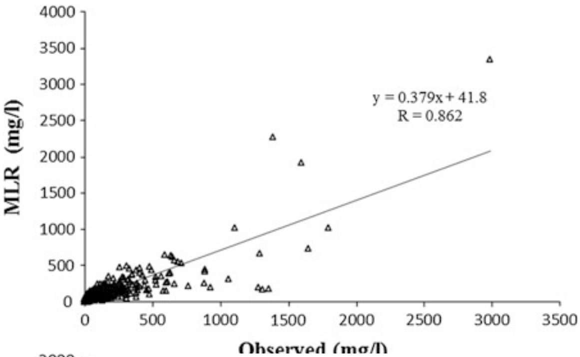


Fig. 9 Observed and MLR scatter graph for the training data

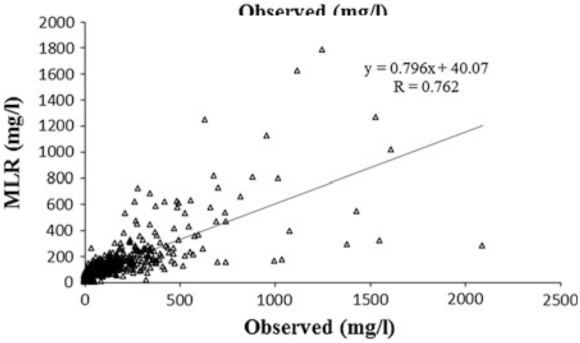


Fig. 10 Observed and MLR scatter graph for the testing data

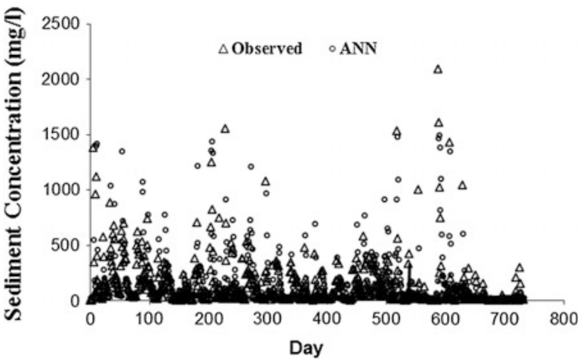


Fig. 11 Observed and ANN distribution graph for the testing data

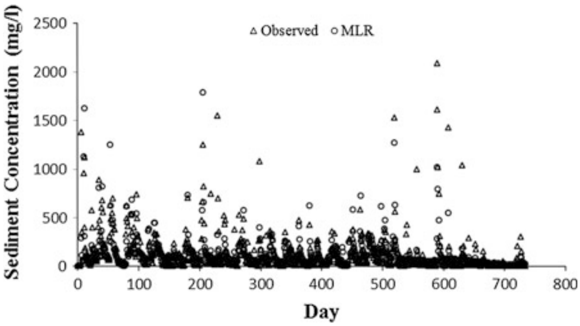


Fig. 12 Observed and MLR distribution graph for the testing data

and scatter charts, MLR values are smaller than the actual values. The following figures are shown in Figs. 10 and 12 for testing data distribution and scatter plots.

The correlation coefficient were obtained as R = 0.762 from the generated graphic. Although daily real-time sediment concentration values is better results than the SRC values, the worst estimated results are observed according to the actual values. In distribution and scatter charts, MLR values are smaller than the actual values. It is observed from figures that MLR estimated test data perform better than the estimated training data.

3.3 Artificial Neural Network (ANN) Model Results

Five-year data were evaluated for ANN model and results are defined as follows. Training and testing data are separated into two parts as three inputs and one output and then entered into Matlab program. The results that created according to the rules are entered. Linguistic relationships between the temperature and flow and suspended sediment concentration rules are created and results are obtained.

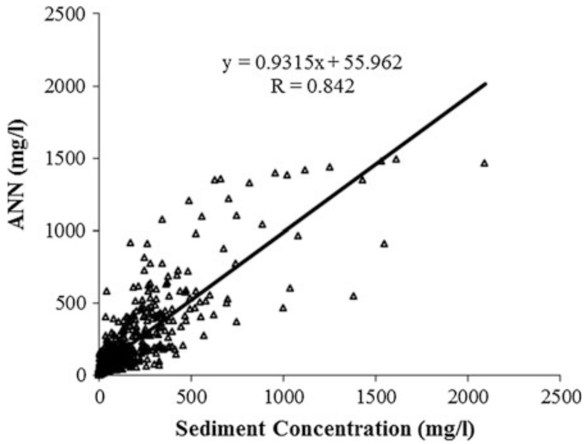


Fig. 13 Observed and ANN scatter graph for the testing data

ANN models are evaluated for 5-year data created in Matlab program. Estimated testing results are shown in Figs. 11 and 13 as respectively the distribution and scatter plots.

The correlation coefficient was obtained as R = 0.842. The ANN estimated values are observed in the test phase and give better results than the SRC and MLR values. As can be seen from figures, the fit line of the ANN is closer to the exact line with a higher R-value than those of the SRC and MLR models. As seen from the scatter plots, the ANN model estimates are less scattered in comparison to the other models.

3.4 General Evaluation

Using daily real-time stream flow, suspended sediment concentration and mean water temperature data from Little Coal River, Danville station, correlation coefficient (R), the lowest mean squared error (MSE), the total squared error (MAE) are calculated for performance evaluation of SRC, MLR, ANN models. Results are used to compare the performance of model prediction and the observation data. Comparing parameters of MSE, MAE and R obtained from testing data are shown in Table 2.

When Table 2 is considered, ANN model gives better results than SRC and MLR models in all performance values.

Table 2 Comparing performances of models created for the Little Coal River, Danville station

Method	MSE ($m^3 s^{-1}$)	MAE ($m^3 s^{-1}$)	R
SRC	4,897.28	45.38	0.785
MLR	4,190.39	45.55	0.762
ANN	3,425.17	34.18	0.842

MSE Mean squared error, *MAE* Mean absolute error, *R* Correlation coefficient

4 Conclusions

In this study, sediment rating curve (SRC), multiple linear regression (MLR), and artificial neural network (ANN) models were investigated in order to improve methods to estimate the suspended sediment concentration. The mean water temperature, daily real-time flow rate, sediment concentration of 5 year data in the Little Coal River, Danville station, West Virginia were analyzed. Model comparisons were made using the research to see which model gave better results. Based on the comparison results, the ANN technique was found to perform better than the other models.

The accuracy of the ANN model in total sediment load estimation was also investigated and results were compared with those of the SRC and MLR models. Comparisons revealed that the ANN model had the best accuracy in total sediment load estimation.

For 5 year data, according to the MSE, MAE and R criteria, the best results were obtained in ANN model. In general, the worst results were obtained in MLR models.

Acknowledgments The data used in this study were downloaded from the web server of the USGS. The author wishes to thank the staff of the USGS who are associated with data observation, processing, and management of USGS Web sites.

References

Bayazıt M, Oguz B (1998) Probability and statistics for engineers. Birsen Publishing House, Istanbul, p 159

Campbell FB, Bauder HA (1940) A rating-curve method for determining silt discharge of streams. Eos Trans Am Geophys Union 21:603–607

Chibanga R, Berlamont J, Vandewalle J (2003) Modelling and forecasting of hydrological variables using artificial neural networks: the Kafue River sub-basin. Hydrol Sci J 48(3):363–379

Cigizoglu HK (2003) Estimation, forecasting and extrapolation of river flows by artificial neural networks. Hydrol Sci J 48(3):349–361

Cigizoglu HK (2004) Estimation and forecasting of daily suspended sediment data by multi layer perceptrons. Adv Water Resour 27:185–195

Cigizoglu HK, Kisi O (2005) Flow prediction by three back propagation techniques using k-fold partitioning of neural network training data. Nordic Hydrol 36(1):49–64

Cigizoglu HK, Kisi O (2006) Methods to improve the neural network performance in suspended sediment estimation. J Hydrol 317:221–238

Demirci M, Baltaci A (2012) Prediction of suspended sediment in river using fuzzy logic and multilinear regression approaches. Neural Comput Appl doi:10.1007/s00521-012-1280-z

Hundecha Y, Bardossy A, Theisen HW (2001) Development of a fuzzy logic based rainfall-runoff model. Hydrol Sci J 46(3):363–377

Jain SK, Das D, Srivastava DK (1999) Application of ANN for reservoir inflow prediction and operation. J Water Resour Plan Manage ASCE 125(5):263–271

Lippman R (1987) An introduction to computing with neural nets. IEEE ASSP Mag 4:4–22

Lohani AK, Goel NK, Bhatia KKS (2007) Deriving stage–discharge–sediment concentration relationships using fuzzy logic. Hydrol Sci J 52(4):793–807
Lopes VL, Ffolliott PF (1993) Sediment rating curves for a clearcut ponderosa pine watershed in northern Arizona. Water Resour Bull 29(3):369–382
McBean EA, Al-Nassri S (1988) Uncertainty in suspended sediment transport curves. J Hydraul Eng ASCE 114(1):63–74
Kisi O (2004a) River flow modeling using artificial neural networks. J Hydrol Eng ASCE 9(1):60–63
Kisi O (2004b) Daily suspended sediment modeling using a fuzzy-differential evolution approach. Hydrol Sci J 49(1):183–197
Kisi O (2004c) Multi-layer perceptrons with Levenberg-Marquardt optimization algorithm for suspended sediment concentration prediction and estimation. Hydrol Sci J 49(6):1025–1040
Kisi O (2005) Suspended sediment estimation using neuro-fuzzy and neural network approaches. Hydrol Sci J 50(4):683–696
Piest RF, Miller CR (1975) Sediment yields and sediment sources. In: Vanoni VA (ed) Sedimentation Engineering. ASCE, New York
Raman H, Sunilkumar N (1995) Multivariate modelling of water resources time series using artificial neural networks. Hydrol Sci J 40(2):145–163
Rumelhart DE, Hinton GE, Williams RJ (1986) Learning internal representation by error propagation. In: Rumelhart DE, McClelland JL (ed) Inf: Parallel Distributed Processing, vol 1, Foundations. MIT Press, Cambridge
Saad M, Bigras P, Turgeon A, Duquette R (1996) Fuzzy learning decomposition for the scheduling of hydroelectric power systems. Water Resour Res 32(1):179–186
Solomatine DP, Dulal KN (2003) Model trees as an alternative to neural networks in rainfall–runoff modelling. Hydrol Sci J 48(3):399–411
Sudheer KP, Gosain AK, Ramasastri KS (2002) A data-driven algorithm for constructing artificial neural network rainfall–runoff models. Hydrol Processes 16:1325–1330
Tayfur G (2002) Artificial neural networks for sheet sediment transport. Hydrol Sci J 47(6):879–892
Tayfur G, Ozdemir S, Singh VP (2003) Fuzzy logic algorithm for runoff-induced sediment transport from bare soil surfaces. Adv Water Resour 26:1249–1256
Toprak ZF, Cigizoglu HK (2008) Predicting longitudinal dispersion coefficient in natural streams by artificial intelligence methods. Hydrol Process 22:4106–4129
Unes F (2010a) Prediction of density flow plunging depth in dam reservoir: an artificial neural network approach. Clean-Soil Air Water 38(3):296–308
Unes F (2010b) Dam reservoir level modeling by neural network approach: a case study. Neural Netw World 4(10):461–474
Wilby RL, Abrahart RJ, Dawson CW (2003) Detection of conceptual model rainfall–runoff processes inside an artificial neural network. Hydrol Sci J 48(2):163–181
Zealand CM, Burn DH, Simonovic SP (1999) Short term stream flow forecasting using artificial neural networks. J Hydrol 214:32–48

Projected Climate Change Impact on Soil Erosion and Sediment Yield in the River Elbe Catchment

Thorsten Pohlert

Abstract The scope of this paper is to analyse the impact and the uncertainty of climate change on soil erosion and consequent sediment yield in the River Elbe catchment by ensemble modelling techniques. The model ensemble comprises five bias-corrected and gridded climate data-sets that origin from coupled runs of global circulation models (GCM) and regional circulation models (RCM) that were driven by both the C20 and the A1B emission scenarios. The data-sets were aggregated for climate normals that are referred to as C20 (1961–1990), 'near future' (2021–2050) and 'far future' (2071–2100). Furthermore, the HYRAS data-set that covers the period 1961–1990 of gridded station data was used as the actual climate data. First, the PESERA-model was chosen as a climate impact model to simulate soil erosion on a 500 × 500 m grid within the entire River Elbe catchment based on relief-data, land cover, soil, crop and the aforementioned climate data. Second, the simulated annual average soil erosion for the actual climate and each projected climate was used to calculate sediment delivery with the approach of spatially distributed sediment delivery ratios (SDR-approach). The actual simulated soil erosion using the actual climate data is in good agreement with other published soil erosion maps for this scale. Furthermore, averaged soil erosion per land use class meets reported data in literature well. Highest soil erosion rates are simulated in the South-East of the catchment and in the range of hills in the central part of the River Elbe catchment. Simulated sediment yield was over predict by a factor of two, that can be attributed because of the methods sensitivity of the underlying river network map and the temporal shift between both periods of actual climate data and reference data on suspended solids load. Sediment delivery slightly drops in the 'near future' and 'far future', which coincidences with decreasing summer rainfalls. However, results of sediment delivery are largely dominated by the chosen GCM-RCM models. It is concluded that the impact of climate change on soil erosion is lower than the impact of potential land cover change.

Keywords Large scale soil erosion · Climate change impact · PESERA model

T. Pohlert (✉)
Federal Institute of Hydrology, Am Mainzer Tor 1, D-56068 Koblenz, Germany
e-mail: Thorsten.Pohlert@bafg.de

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_7

1 Introduction

Sediment delivery that origins from field soil erosion are among river bank erosion and riverbed erosion the dominant sources for suspended solids in rivers. The deposition of these fine grained materials within the rivers mainly govern the demand for dredging activities of the Waterways and Shipping Administration in order to facilitate shipping and construction safety at harbours, impounded rivers, and barrages in interior German waterways, which is entirely financed by tax money. The annually dredged material in interior waterways amounts up to 4×10^6 m^3 according to internal statistics of the Federal Institute of Hydrology. Although a lot of effort has been spend in order to develop both physical and numerical models to simulate in-stream transport and deposition of suspended sediments, little attention has been paid to develop tools for accounting of sediment delivery from fields in the entire catchment and to assess climate change impact on consequent sediment yield.

The scopes of this study are (i) to develop a tool to account for soil erosion and consequent sediment delivery into the streams in the large-scale hydrological domain that is applicable for scenario analysis, (ii) to validate the model with best available data in the River Elbe catchment (124,614 km^2) and (iii) to apply the model to project changes in sediment yield using pre-processed data of climate change projections.

2 Materials and Methods

2.1 *Rationale of the Soil Erosion and Sediment Delivery Models*

The simplified process-based model—Pan-European Soil Erosion Risk Assessment (PESERA; Kirkby et al. 2008)—was used to simulate field soil erosion on a 500×500 m grid within the entire catchment of the River Elbe. The grid size was chosen to be in line with the underlying assumptions of PESERA and for reasons of computational efficiency. The PESERA model is an at-a-point model that accounts for both, infiltration excess overland flow and saturation excess overland flow based on total daily rainfall storms and monthly soil moisture, as well as down-slope sediment transport to provide estimates of annual average soil erosion rates for the field under a given climate.

According to the bucket model approach daily volumes of overland flow r is given by (Eq. 1)

$$r = p(R - R_0), \tag{1}$$

where R is total daily storm rainfall (mm) that is estimated using a Gamma distribution and monthly rainfall statistics, R_0 is the run-off threshold or bucket storage capacity (mm) and p is a dimensionless proportion of subsequent rainfall that runs off. PESERA has implemented the TopModel approach (Beven and Kirkby 1979) to estimate subsurface flow, whereas the average saturated deficit is estimated on a monthly basis in order to give the saturation constraint on the run-off threshold (R_0) that controls overland flow in each storm. Due to these combinations, PESERA has included to mechanisms to account for infiltration excess and saturation excess overland flow (Kirkby et al. 2008). Sediment transport from a field is accounted for using a power-law approach (Eq. 2).

$$C = k\, q^m \Lambda^n \tag{2}$$

The transporting capacity is denoted as C (kg m^{-1} day^{-1}), k is the soil-erodibility, q is the overland flow per unit flow width (L m^{-1} day^{-1}), Λ is the dimensionless local slope gradient, and $m = 2$ and $n = 1$ are empirical exponents. The units of k change with the exponent m. Due to our selected exponent $m = 2$, the unit for k is kg L^{-2} m day. By some substitution, one leads to the final equations that are incorporated in PESERA (Eqs. 3 and 4).

$$E = k\, L\, \Lambda_B \sum r^2. \tag{3}$$

The annual soil erosion E (kg m^{-2} year^{-1}) that is transported to the base of the slope is proportional to the sum of the frequency distribution of daily overland flow events (r), L is the total slope length (m) and Λ_B is the local slope gradient evaluated at the slope base. Furthermore, the term $L\Lambda_B$ can be approximated as the total slope relief in meters H, that leads to

$$E = k\, H \sum r^2. \tag{4}$$

It is well known that annual soil erosion estimates in a catchment are by far higher than observed sediment yield (or unit-area load of suspended solids) of the catchment. This is because a large proportion of eroded soil does not enter the stream network but accumulates on the base of a slope, within hollows (i.e. colluviation) or alluvial fans. A simple way to account for this discrepancy between soil erosion and sediment delivery is to introduce sediment delivery ratios (SDR-approach). In this study a distributed SDR-approach was applied instead of a lump SDR-approach for the entire catchment. An earlier study (Ferro and Minacapilli 1995) connected sediment delivery ratios for a grid cell with travel time of overland flow from the cell to the nearest stream channel. As travel times were found proportional to flow path l_p and inversely related to the square root of the slope of flow path s_p, the model of Ali and De Boer (2010) was used to calculate distributed sediment delivery ratios (*SDR*) for each grid as given by (Eq. 5)

$$SDR_i = \exp\left[-\beta \sum_{p=1}^{m} \frac{l_p}{\sqrt{s_p}}\right], \tag{5}$$

with β a dimensionless factor that is assumed to be unity in our case. An advantage of this approach is that it only bases on connectivity of the grid cells to the stream network that is a constant parameter. Furthermore, this SDR-approach is consistent with the PESERA model, because no parameter is accounted twice, as it is often the case in lumped SDR-approaches that make use of multiple linear regression models. If one multiplies soil erosion E_i (Eq. 4) with sediment delivery ratio SDR_i for the ith grid cell, than sediment delivery SD_i (kg m^{-2} year^{-1}) for the ith cell is

$$SD_i = SDR_i \times E_i. \tag{6}$$

The sediment yield (SY) for a selected catchment is estimated by averaging SD_i within the entire catchment that comprises $N \geq i$ cells: $SY = \sum SD_i/N$. This estimation is taken as the equivalence of observed unit-area load of suspended solids; kg m^{-2} year^{-1}. The inherent assumption of this procedure is that for a given period of time (i.e. 30 years) the rivers sediment budget is in steady state, i.e. there is neither a net deposition (loss) of incoming sediments from the catchment in the river system, nor a net delivery (gain) of fine sediments due to bank erosion or riverbed erosion. The SDR-approach was implemented into the SAGA Geographical Information System (SAGA User Group Association).

2.2 Data-Sets

The PESERA model was set-up using the SAGA-GIS to process the available national and transnational data on relief, soils, land-cover, crop statistics and phenology (Table 1). PESERA's standard input for soil comprises six soil parameters, i.e. crusting, erodibility, effective soil water storage capacity, soil water available to plants in top 300 mm, total soil water available to plants, and scale depth. These parameters were derived with pedotransfer rules as outlined by the PESERA manual. The codes of land use classes of the German national survey on land use (ATKIS Basis DLM) were translated into classes of the Corine Land Cover (CLC2000) data-set using look-up tables. These classes of CLC2000 correspond to the land use class codes given in PESERA. The input grids for the first and second dominant crop on arable land was created using statistical records of crops that were planted on arable land within administrative units (Agrarstrukturerhebung). For the Czech part of the River Elbe catchment the transnational data were used. PESERA standard input for soils could be derived from the European Soil Database (ESDB2) and land use was taken from the CLC2000 data-set. However, first and second dominant crops as well as phenological dates of these crops were estimated using expert guesses, as no data about crop statistics was available for the Czech part.

Table 1 Available data for the PESERA model

PESERA-input	German data-sets	Transnational data-sets
Soil	Nutzungsdifferenzierte Bodenübersichtskarte der Bundesrepublik Deutschland (BÜK1000N)[a]	European soil database (ESDB2); Übersichtskarte der Bodenerosionsgefährdung der Schweiz–K-Faktoren[b]
Terrain	SRTM3	SRTM3
Land use	ATKIS basis DLM	Corine land cover (CLC2000)[c]
Crop cover	Agrarstrukturerhebung 2007[d]	Land use according to NUTS 2[e]
Phenology	Phenological data (1961–2010)[f]	

[a] Bundesanstalt für Geowissenschaften und Rohstoffe, Version 2007
[b] Prasuhn et al. 2007
[c] European Environmental Agency
[d] Statistisches Bundesamt
[e] EUROSTAT
[f] Deutscher Wetterdienst

The required relief parameter, i.e. standard deviation of elevation, was calculated according to the PESERA manual. A pre-processed SRTM3 that was already corrected for forest canopy height and roof height in urban areas was used to calculate the relief parameter.

The data-set for actual climate (1961–1990) was derived from the HYRAS data-set, that is gridded data of daily mean air temperature and daily sums of rainfall derived from climate station data (Rauthe et al. 2013). The HYRAS data-set was further processed to get PESERA standard input for climate of the given period, i.e. mean monthly rainfall, mean rainfall per rainy day by month, monthly coefficient of variation of rain per rainy day, mean monthly temperature, monthly temperature range and mean monthly potential evaporation (PET). The method according to Thornthwaite (1948) was used to calculate PET.

A climate model ensemble of $N = 5$ was chosen that comprises a set of results from runs of global climate models (GCM) that were coupled with regional climate models (RCM) and driven by the C20 and A1B emission scenarios (Table 2). The output of the RCMs were then regionalised to fit on the same grid. Finally, the daily data were bias corrected using the linear scaling approach with the underlaid HYRAS data-set (Imbery et al. 2013). The projected daily data were aggregated for the denoted periods of C20 (1961–1990), ‘near future’ (2021–2050) and ‘far future’ (2071–2100) to yield PESERA standard input for climate and PET was calculated using as well the approach of Thornthwaite (1948). The selected ensemble covers the range between projected ‘dry’ to projected ‘moist’ climate in Central Europe for the A1B emission scenario (Imbery, pers. com., German Weather Service).

Data on suspended solids concentration were available for $N = 8$ gauges that are located along the German river stretch of the River Elbe. Each data-set covers the period 2003–2009. Within the monitoring programme on suspended solids concentration at German waterways the samples are continuously taken every working

Table 2 Climate model ensemble used for this study

Control run/GCM SRES scenario		RCM	Notation
C20/A1B	ECHAM5r3 (MPI-M)[a]	REMO5.7 (MPI-M)	EH5r3_REMO
C20/A1B	ECHAM5r3 (MPI-M)	RACMO2 (KNMI)[b]	EH5r3_RACMO
C20/A1B	HadCM3Q0 (HC)[c]	CLM2.4.6 (ETHZ)[d]	HADCM3Q0_CLM
C20/A1B	BCM2 (NERSC)[e]	RCA3 (SMHI)[f]	BCM2_RCA3
C20/A1B	ECHAM5r1 (MPI-M)	CLM2.4.11 (GKSS)[g]	EH5r3_CLM

Control runs (C20) cover the period 1961–2000, projection runs are for the period 2001–2100 on the SRES-scenario A1B. All projections were regionalised and bias corrected using the linear scaling approach

[a] Max Planck Institute for Meteorology
[b] Royal Netherlands Meteorological Institute
[c] Met Office Hadley Center
[d] Swiss Federal Institute of Technology in Zurich
[e] Nansen Environmental and Remote Sensing Center
[f] Swedish Meteorological and Hydrological Institute
[g] Helmholtz-Zentrum Geesthacht

day by the Waterways and Shipping Administration. The samples are filtered and suspended solids concentration is gravimetrically determined after drying. The data are archived and further processed at the Federal Institute of Hydrology to fill data gaps by means of linear interpolation. Daily loads of suspended solids are computed by multiplying daily suspended solids concentration with daily discharge that is recorded at nearby flow gauges (Vetter 2001).

In this study, the daily records were aggregated to yield average annual load of suspended solids. This measure was then divided by the area of the sub-catchments to yield unit-area loads of suspended solids as an equivalence for the validation of simulated *SY* using the actual climate.

3 Results and Discussion

3.1 Actual Soil Erosion

Prior to an application for *SY* estimation, the PESERA outputs of annual soil erosion (Fig. 1) using the actual climate were visually assessed with existing maps on soil erosion in Germany (Auerswald et al. 2009) and the Czech Republic (Dostál et al. 2007) that both were calculated using the Universal Soil Loss Equation (USLE). Highest soil erosion rates are identified in the South-East of the Elbe catchment (highlands and foothills of Bohemia) and in the range of hills in the central part of the River Elbe catchment that is partly covered with loess. The simulated spatial mean (standard deviation) of soil erosion (t ha^{-1} year^{-1}) per selected land use class within the River Elbe catchment were 3.4 (9.8) for arable land, 0.3 (2.5) for pasture land or grasslands, and 0.1 (2.1) for forested land. This is

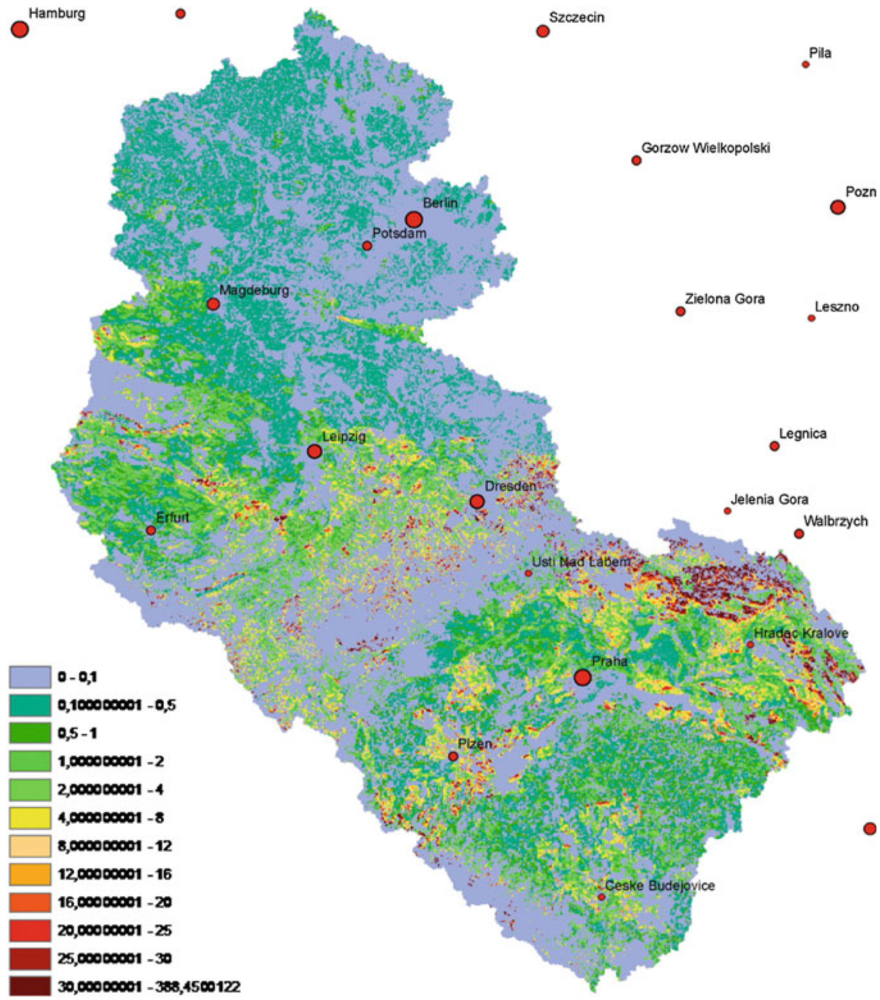


Fig. 1 Simulated annual soil erosion (t ha^{-1} year^{-1}) in the River Elbe catchment as modelled with PESERA using actual climate [i.e. the climatology of the HYRAS data-set (1961–1990)]

within the range of other reports (e.g. Auerswald et al. 2009), that give a spatial mean (standard deviation) of soil erosion (t ha^{-1} year^{-1}) of 5.7 (8.6) on arable land, 0.5 (2.3) on pasture land or grasslands, and 0.2 (2.6) on forested land for Germany. It is worth to note that both studies are in agreement that soil erosion on arable land is about 10 times higher than on pasture land. Dostál et al. (2006) report a simulated average soil erosion of 2.3 t ha^{-1} year^{-1} for agricultural land including vineyards and hop gardens in the Czech Republic using the USLE method and land cover data from 1995. Consequently, the PESERA simulations are within the span of reported values in literature and similar spatial patterns can be identified by map comparison.

The contribution to total actual soil erosion according to the PESERA model within the River Elbe catchment of arable land, pasture land or grasslands, and forested land are 96.1, 2.1, and 1.9 % respectively. Auerswald et al. (2009) reported contributions to total soil erosion in Germany of 92.8, 3.7, and 2.6 % for the aforementioned land-use classes (Vineyards and hop gardens were accounted separately and sum-up to 1.7 %) that is in good agreement with the findings in this study. The differences between the findings can be explained by different spatial domains (i.e. the spatial extent of the River Elbe catchment versus the administrative units of Germany and the Czech Republic), the different conceptualisations of the PESERA model and the USLE approach, the spatial resolution of the underlying digital terrain model, the timeliness of the underlying land-cover/land use maps, and different climate periods used for actual climate.

3.2 Actual Sediment Yield

The critical source areas for sediment delivery are located along the slopes that drain into the major river-network of the catchment (Fig. 2). Although soil erosion takes place in the entire River Elbe catchment only these critical source areas are relevant for sediment delivery and consequent sediment yield according to the PESERA-SDR conceptualisation.

For the assessment of the model performance the observed *SY* (1993–2001) was compared to the PESERA-SDR simulations using the data for actual climate and data of each C20 run. It should be noted that the PESERA-SDR model was not calibrated, though the SDR-approach was found sensitive to the density of river network. From River Stretch Kilometre $RSK = 0$ that equals the borderline between Germany and the Czech Republic to approx. $RSK = 210$ there is no increase in *SY* present due to the absence of major tributaries (Fig. 3). From the mouths of the tributaries Mulde and Saale ($RSK = 295$) the *SY* rises almost to its final stage of approx. 500,000 t ha^{-1} as given by observed *SY*. There is a decline (or second rise) present at $RSK = 390$ in observed *SY*. It is not clear, whether this decline (or second rise) is due to deposition/remobilisation from the groyne-fields or by systematic under-sampling, because water samples are only taken at one point in the river's cross-section in the suspended solids monitoring programme.

The simulated *SY*s follow the same pattern (Fig. 3), although simulations for the actual climate (1961–1990) are in absolute terms about two times higher as compared to observed *SY*. However, this is still an acceptable model performance given the systemic uncertainty in model conceptualisation and data quality such as the temporal shift between the periods for observed *SY* (1993–2001) and the period for actual climate (1961–1990). Differences between *SY*s that were simulated with data from C20-runs and with actual climate data are only attributable to uncertainties in the output of the C20-runs. This is because the PESERA-SDR model and its model set-up for relief, soils, and land-cover remained the same for each run. As simulated *SY*s of four out of five C20-runs are higher than simulated *SY* using the actual climate data, it

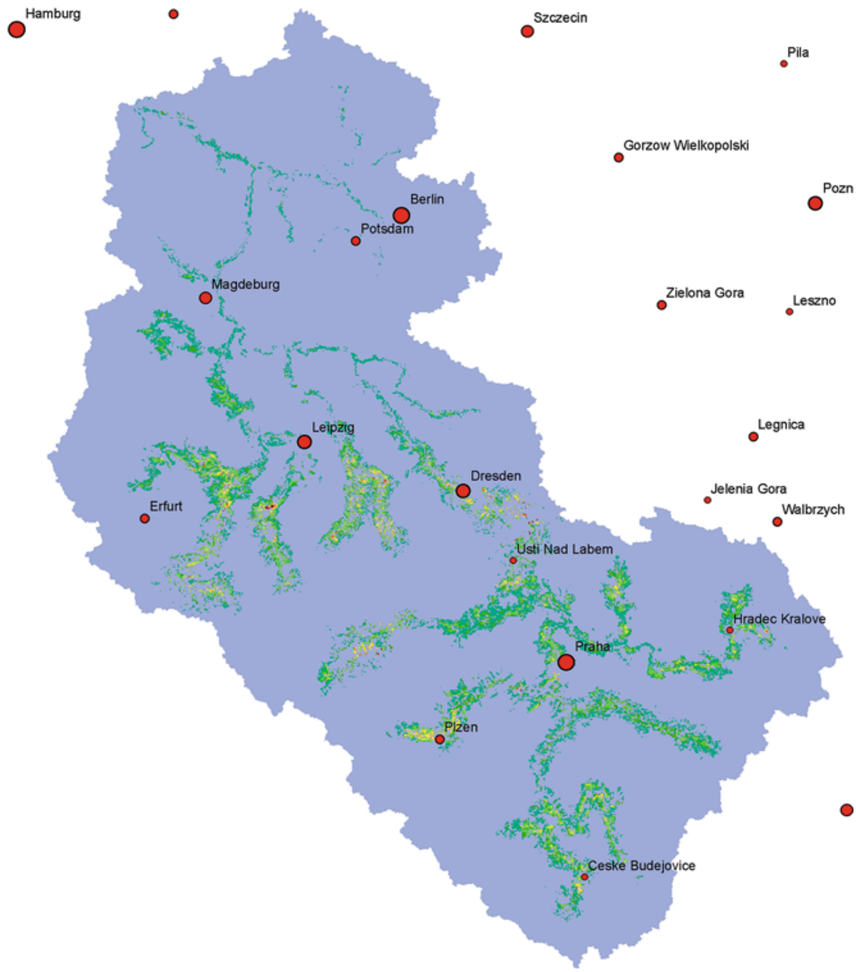


Fig. 2 Simulated annual sediment delivery (t ha^{-1} $year^{-1}$) in the River Elbe catchment as modelled with PESERA using actual climate [i.e. the climatology of the HYRAS data-set (1961–1990)]. Same legend as Fig. 1

is concluded that the coupled GCM-RCM produce a significantly different precipitation climatology that increases sediment delivery in the River Elbe catchment.

3.3 Projected Sediment Yield

It is common practise in climate change impact research to analyse relative changes between projections and the C20-runs in order to detect a bias-free change signal. Two out of five projections,—the runs of EH5r3_REMO and EH5r3_RACMO—,

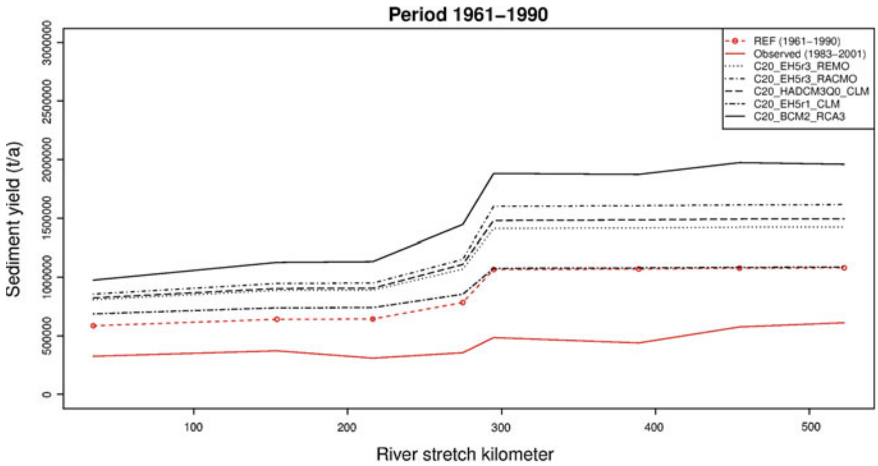


Fig. 3 Longitudinal profile of simulated sediment yield (or load of suspended solids, t year^{-1}) along the stretch of the River Elbe. River stretch kilometre increases from the German/Czech border to the final outlet

for the 'near future' (2021–2050) give a relative increase of sediment yield by approx. +5 to +10 % up to the mouth of the River Saale ($RSK = 295$, Fig. 4 left). Thus, sediment delivery in the upstream areas (Czech Republic) increases according to the precipitation climatology of the aforementioned GCM-RCMs that leads to a positive change signal. As from the mouth of the River Saale ($RSK = 290$) the change signal of projected sediment yield is approx. −10 % (Fig. 4, left), total sediment delivery from the range of hills in the central part of the

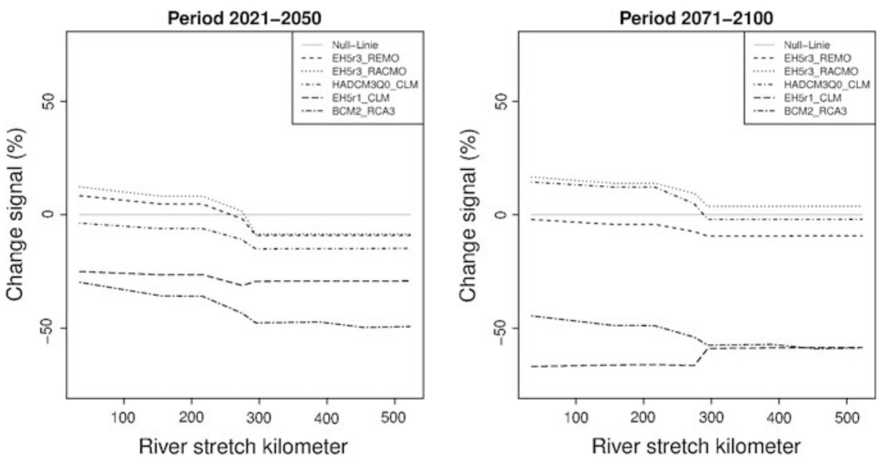


Fig. 4 Longitudinal profile of the projected change signal (%) of sediment yield (or unit-area load) along the stretch of the River Elbe for the 'near future' (2021–2050) and 'far future' (2071–2100)

River Elbe catchment decreases in such a way, that the surplus of sediment yield coming from upstream is not compensated. However, three out of five projections for the 'near future' give a spatially consistent decrease in sediment yield, which results in an average decrease of sediment yield of approx. −25 % at the final outlet Hitzacker ($RSK = 523$). The projections are similar for the 'far future' (2071–2100, Fig. 4, right), although the span of the change signal is higher.

The findings are consistent with detected climate change signals for Germany, as mean projected summer precipitation using a GCM-RCM ensemble of $N = 19$ decreases weakly in the 'near future' (2021–2050) but can decrease to about −20 % in the 'far future' (2071–2100) with some regional variation (Imbery et al. 2013).

4 Conclusion

The study aimed at developing a tool to estimate soil erosion and sediment delivery in large-scale river catchments, the validation of the tool, and the consequent application for climate change impact analysis. The PESERA model estimates soil erosion within the same range as compared to reported soil erosion values in literature for the similar area, when the USLE-approach was applied. It is therefore concluded that PESERA is suitable for large-scale soil erosion modelling. The performance of the spatially distributed SDR-approach was reasonably well, as far as an over-estimation of factor two for sediment yield is concerned. This mismatch can be explained by the temporal shift between both reference periods of climate (1961–1990) and observed sediment yield (1993–2001). Hence, the PESERA-SDR approach is still applicable for scenario analysis.

Projected change signals in sediment yield accounted for a mean reduction of approx. −25 % for both the 'near future' (2021–2050) and 'far future' (2071–2100), though the uncertainty span that is attributable to the GCM-RCM runs is remarkably high. The reduction of sediment yield coincidences with the projected decrease in summer rainfalls for the given climate periods in the 'near future' (2021–2050) and 'far future' (2071–2100). It should be recalled that soil erosion on arable land is in average one order of magnitude higher than soil erosion on pasture land or grasslands. Therefore it is concluded that climate change impact on soil erosion and *SY* in the River Elbe catchment is less important as compared to land use change in critical source areas for sediment delivery as given by the dominating contribution of arable land for soil erosion of up to 96.1 % of total soil erosion in the River Elbe catchment.

An advantage of the spatially distributed PESERA-SDR approach over lumped SDR-approaches is that it can provide maps according to the critical source area concept. Once the critical source areas are identified, further investigations can be undertaken such as meso-scale simulation of soil erosion and sediment yield using process based models, as well as measures can be implemented for efficient on-site reduction of sediment delivery.

Acknowledgments This research has been carried out in the project 5.01 'Climate projections for sediment balances and risks due to cohesive sediments' within the departmental research programme 'KLIWAS—Impacts of climate change on waterways and navigation' that was financed by the German Federal Ministry of Transport, Building and Urban Development. The GIS-processing was conducted by scilands GmbH Göttingen with assistance of Geoflux GbR Halle for soil data processing. Thanks go to our KLIWAS-colleagues of the German Weather Service Offenbach, who prepared the climate data-sets for this study. Thanks go to two anonymous reviewers, whose comments improved this paper.

References

Ali KF, De Boer DH (2010) Spatially distributed erosion and sediment yield modeling in the upper Indus River basin. Water Resour Res 46(W08):504. doi:10.1029/2009WR008762

Auerswald K, Fiener P, Dikau R (2009) Rates of sheet and rill erosion in Germany—a meta-analysis. Geomorphology 111:182–193. doi:10.1016/j.geomorph.2009.04.018

Beven KJ, Kirkby MJ (1979) Towards a simple, physically based, variable contributing area model of catchment hydrology. Hydrol Sci Bull 24:43–69

Dostál T, Janecek M, Kliment Z, Krása J, Langhammer J, Váška J, Vrana K (2006) Czech republic. In: Poesen J, Boardman J (eds) Soil erosion in Europe. Wiley, New York, pp 107–116

Dostál T, Vrána K, Krása J, Jakubíková A, Schwarzová P, David V, Nováková H, Bečář M, Veselá J, Kavk P (2007) Metody a zpusoby predikce povrchověho odtoku, eroze a transportu sedimentu v krajině, vŷzkumná zpráva projektu COST1P04OC634.001. Technical report, ČVUT v Praze, Fakulta stavebnÍ, Katedra hydromelioracÍ a krajinného inženýrstvÍ, Praha

Ferro V, Minacapilli M (1995) Sediment delivery process at basin scale. Hydrol Sci J 40(6):703–717

Imbery F, Plagemann S, Namyslo J (2013) Processing and analysing an ensemble of climate projections for the joint research project KLIWAS. Adv Sci Res 10:91–98. doi:10.5194/asr-10-91-2013

Kirkby MJ, Irvine BJ, Jones RJA, Govers G, The PESERA team (2008) The PESERA coarse scale model for Europe. I.—Model rationale and implementation. Europ J Soil Sci 59:1293–1306

Prasuhn V, Lininger HP, Hurni H, Friedli S (2007) Bodenerosionsgefährdungskarte der Schweiz. Agrarforschung 14(3):120–127

Rauthe T, Steiner H, Riediger U, Mazurkiewicz A, Gratzki A (2013) A Central European precipitation climatology—Part I: generation and validation of high-resolution gridded daily data set (HYRAS). Meteorol Z 22(3):235–256

Thornthwaite CW (1948) An approach toward a rational classification of climate. Geogr Rev 38(1):55–94

Vetter M (2001) Das Schwebstoffdauermessstellennetz an den Bundeswasserstraßen. In: BfG (ed) Schwebstoffe und Schwebstofftransport in Binnenwasserstraßen. Gewässermorphologisches Kolloquium, Koblenz, Veranstaltungen 3/2003, pp 5–18, 8 Nov 2001

Part III
Sediment Quality

Water Quality and Sediment Management in Brahmaputra Basin of India: Impact of Agricultural Land Use

Uttam C. Sharma and Vikas Sharma

Abstract In Brahmaputra basin of India, the social sanctions and belief system maintained a balance between resource potential and their utilization for a long time but due to the increase in the demographic pressure and indiscriminate use of natural resources, imbalance has been created. Socio-economic constraints like shifting cultivation, land tenure system, small size of land holdings, unabated deforestation, free range grazing and undulating terrain have affected the sediment yield and, quantity and quality of available water. The mean annual sediment yield per ha from the Brahmaputra basin constitutes, 23.2 tonnes of soil and, 26.1, 4.2, 19.4, 0.93, 0.58, 2.3 and 1.79 kg of N, P_2O_5, K_2O, Mn, Zn, Ca and Mg, respectively. The Brahmaputra river in India has more than 100 tributaries of which 15 in the north and 10 in the south are fairly large. It was estimated that about 660 $m^3\ km^{-2}$ of sediment load is brought by the northern and 100 $m^3\ km^{-2}$ by the southern tributaries to the main river channel, annually. To evolve eco-friendly and sustainable farming systems to replace sediment encouraging practice of shifting cultivation, a multidisciplinary, long-term study was undertaken with seven land use systems on micro watersheds viz.; livestock based (grasses and fodders), forestry, agro-forestry, agriculture, agri-horti-silvi-pastoral, horticulture and shifting cultivation, to monitor their comparative efficacy with regard to in-situ retention of rain water, water quality and sediment yield. The sediments emanating from the farming systems affected the surface and groundwater quality, the magnitude of which was highly related to the use of amount of fertilizers and other agricultural chemicals in the basin.

U.C. Sharma (✉)
Centre for Natural Resources Management, V. P. O. Tarore, Jammu 181133, Jammu and Kashmir, India
e-mail: ucsharma2@rediffmail.com

V. Sharma
S.K. University of Agricultural Sciences and Technology, Chatha, Jammu 180009, Jammu and Kashmir, India

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_8

1 Introduction

Healthy ecosystems in properly functioning watersheds depend on maintaining soil onsite (Nichols et al. 2002). Vegetation loss is often accompanied by erosion and transport of eroded sediment. In addition to productivity loss on uplands, eroded soil can have significant impacts on downstream water quality, and sediment deposition can reduce reservoir storage capacity. Soil loss and movement in watershed uplands is difficult to measure, and may go unnoticed until it is a severe problem. Sediment yield is the sediment load from the drainage area and is the net result of erosion and deposition processes within a basin. Thus, it is controlled by those factors that control erosion, topography, soil properties, climate, vegetation cover, catchment morphology, drainage network characteristics, and land use (Walling 1994; Hovius 1998). Sediment yield studies are very important for studying linkages between soil erosion and suspended sediment transport in large rivers (Verstraeten and Poesen 2001). Measurements of sediment yield are also key elements for understanding the impacts of past land-use or climate changes (Walling 1997; Verstraeten and Poesen 2001). Scientists have attempted to explain the global pattern of sediment yield in terms of climatic factors (Douglas 1967; Wilson 1973; Walling and Webb 1983), the role of relief and elevation of drainage basins (Milliman and Syvitski 1992; Summerfield and Hulton 1994), vegetation as controlled by climate (Douglas 1967) and land use (Trimble 1975; Verstraeten and Poesen 2001). The Asian rivers are the highest sediment producers among the large river basins worldwide (Milliman and Mead 1983; Walling and Webb 1983; Milliman and Syvistski 1992; Summerfield and Hulton 1994; Ludwig and Probst 1998). Land clearance, land use change and other facets of catchment disturbance, soil conservation and sediment control programmes and dam construction are shown to have resulted in significant recent changes in the sediment loads of many world rivers (Walling 2000, 2008).

The environment of a place determines the habitat, the mode of life and the progress of civilization up to a large extent. In Brahmaputra basin of India, the social sanctions and belief system maintained a balance between resource potential and their utilization for a long time but due to the increase in the demographic pressure and indiscriminate use of natural resources, imbalance has been created. Socio-economic constraints viz. shifting cultivation, land tenure system, small size of land holdings, unabated deforestation, free range grazing and undulating terrain have affected the sediment yield and, quantity and quality of available water (Sharma 1997, 1998, 2003). The fast growing population in the region has pressurized the food production base and to satisfy their needs, the people have mismanaged and misused natural resources of soil, water and vegetation, resulting in soil and environmental degradation and loss of water quality (Sharma 2003). The prevalence of shifting cultivation in the basin has resulted in heavy soil erosion, deforestation and water resources degradation (Sharma and Sharma 2004a). The major problems of facing the harmonious development and management of rainwater in the region are; socioeconomic constraints, paucity of reliable data and lack of human and institutional capacity necessary for confronting the complex interactions of the hydrological cycle with societal needs and the institutional reforms for better management and utilization of water resources and environment. There is annual loss of 83.3 million tonnes of soil and 10.65, 0.37 and 6.05 thousand tonnes of available N, P_2O_5 and K_2O, respectively due to shifting cultivation alone (Sharma and Prasad 1995). A long-term-multidisciplinary study was, therefore, undertaken to assess the soil and nutrient losses from the hill slopes, in situ retention of rainwater as affected by vegetation and, water and soil conservation measures to reduce runoff as well as its impact on the environment.

2 The Study Site and Methodology

The Brahmaputra river basin extends to four northeastern states of India viz. Arunachal Pradesh, Assam, Meghalaya and Nagaland, with an area of 1.94 × 105 km^2 (Fig. 1). The basin is predominantly hilly and inhabited mostly by different tribes.

Fig. 1 Brahmaputra basin in north-eastern region of India

Table 1 Vegetation cover in different land use systems

Land use	Slope (%)	Crops/trees	Livestock	Soil and water conservation measure
Livestock (grasses and fodders)	32.0	Maize, rice-bean, oats, pea, guinea grass, tapioca, broom grass	Cows, pigs, rabbits	Contour bunds, grassed water-ways, trenches
Forestry	38.0	*Alder nepalensis*, *Albziia lebbeck*, *Acacia auriculiformis*	None	None
Agro-forestry	32.2	*Ficus hookerii*, Eucalyptus, guava, pine, pineapple, french bean, pulse crops	Goats, rabbits	Contour bunds
Agriculture	32.4	Beans, radish, maize, paddy, ginger, turmeric, ground-nut, oats, grasses on risers	Cows	Bench terraces, contour bunds, grassed water-ways
Agri-horti-silvi-pastoral	41.8	Beans, vegetables, guava, Citrus, ginger, *Alder nepalensis*, *Ficus hookeri*, *grasses*	Pigs, goats	Bench terraces, contour bunds, grassed water-ways
Horticulture	53.2	Peach, pear, citrus, guava, lemon, vegetables	None	Same as above
Shifting cultivation	45.0	Mixture of crops	None	None

To evolve eco-friendly and sustainable farming systems to replace sediment encouraging practice of shifting cultivation, a multidisciplinary, long-term study was undertaken with seven land use systems on micro watersheds viz.; livestock based (grasses and fodders), forestry, agro-forestry, agriculture, agri-horti-silvi-pastoral (forestry on top of the hill slope, followed by pasture, horticulture and agriculture down the slope), horticulture and shifting cultivation, to monitor their comparative efficacy with regard to in situ retention of rain water, water quality and sediment yield (Table 1). Sediment and water yield was monitored through representative gauges installed at the exit point of each watershed. The watersheds slope varied from 32 to 41 % and, soil and water conservation measures followed were contour bunds, trenches, bench terraces, half-moon terraces and grassed water-ways in all land uses except forestry and shifting cultivation. Slope instability has induced major geo-morphological changes due to landslides and their long term effects, increasing sediment load, causing permanent changes in valleys and plains, and significant changes in Brahmaputra river flow. The influence of anthropogenic factors on natural dynamics of erosion is distinctive in the basin. To study the socio-economic impact on sediment yield, old records were scanned as well as benchmark survey was conducted in selected areas. The meteorological data were collected in the observatory located near the project site. The chemical analysis of soil and water samples was done as per procedures outlined by Jackson (1973).

3 Results and Discussion

3.1 Brahmaputra Basin

The Brahmaputra river drains 194,400 km^2 area with an annual flow of 537.2 km^3 of water at an annual average of 17,040 $m^3\ s^{-1}$, varying from 3,200 to 19,200 $m^3\ s^{-1}$ during lean period and monsoon season, respectively. A maximum of 72,748 $m^3\ s^{-1}$ discharge was recorded in 1963 (Goswami 1985) and minimum of 3,280 $m^3\ s^{-1}$ in 1960. The river has more than 100 tributaries, of which 15 in the north and 10 in the south, are fairly large. The river and its tributaries produce enormous sediment load when they flow through geologically young and unstable terrain and banks being extremely unstable, are subjected to huge soil and nutrient erosion as well as subsidence. The Brahmaputra river basin in the India receives about 450 km^3 of water annually, as rainfall at an annual average of 2,450 mm. Cherrapunji, which is known as the place of highest rainfall in the World, with a record rainfall of 26,461 mm during August 1860 to July 1861, is located in this basin. Srivastava and Mandal (1995) reported that Cherrapunji still holds the record of highest rainfall of 1,036 mm in India in 24 h, during 1876.

3.2 Socio-economic Aspects

The Brahmaputra basin in India is inhabited by various tribes and their economy can conveniently be divided into hunting, nomadism, pastrolism, shifting cultivation and now, settled cultivation up to some extent. The rural economy of the region mainly depends on shifting cultivation. In the past, when the land was in abundance and population sparse, the rotational cycle of shifting cultivation in the region used to be 25–30 years, the land getting enough time for rejuvenation of vegetation. The man as hunter as well as cultivator used to co-exist with forests. The soil fertility was maintained with in situ burning of vegetation of forests and the production was enough to feed the limited population. However, with increase in population, the rotational cycle has come down to 2–10 years and the land does not get enough time for rejuvenation (Table 2). The annual area under shifting cultivation in the region is 3,869 km^2, whereas total area affected is 14,660 km^2 (Anonymous 2000).

Shifting cultivation is not only a set of agricultural practices but implies the whole nexus of people's religious belief, attitude, self image and tribal identity. This kind of inter-connections between different elements and domains of social life restricts the cultivators to leave shifting cultivation. As high as 70.6 and 130.2 t ha^{-1} of annual soil loss has been reported during first and second year of shifting cultivation on a hill having a slope of 70 % (Singh and Singh 1978). The soil fertility is on decline as there is limited material to burn and add to the soil. The results showed that at least 20 years time is necessary for rejuvenation of enough vegetation to get optimum available major nutrients for crop support (Fig. 2) (Sharma 2001).

Table 2 Shifting cultivation and, soil and nutrient loss in the region

State	Shifting	Cultivation		Total soil loss (million tonnes)	Nutrient loss (thousand tonnes)		
	Annual area (km^2)	Fallow period (years)	Soil loss (million tonnes)		N	P	K
Arunachal Pradesh	700	3–10	14.5	178.1	217	36.6	153
Assam	696	2–10	12.3	178.4	201	33.4	155
Manipur	900	4–7	20.4	64.0	76	7.4	63
Meghalaya	530	5–7	14.2	57.7	62	7.0	48
Mizoram	630	3–4	13.0	39.4	60	6.9	40
Nagaland	190	5–8	8.0	41.7	44	5.2	34
Tripura	223	4–9	5.9	15.4	26	2.7	18
Total	3,869	2–10	88.3	601.2	686	99.2	511

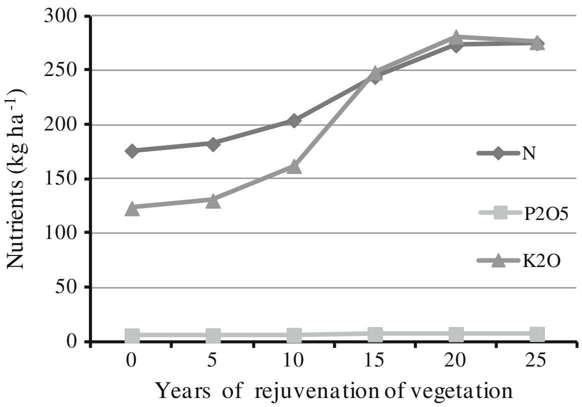


Fig. 2 Available soil nutrients per ha by burning of rejuvenated vegetation

In 20 years cycle, the available N, P_2O_5 and K_2O increased by 55.1, 22.8 and 128.4 %, respectively over their initial status. The low increase in available P_2O_5 compared to other nutrients was due to the reason that soil being strongly acidic in reaction, most of the phosphorus got fixed as aluminium and iron compounds. The land tenure system in the north-eastern region is unique. The land belongs either to (i) village chief, (ii) community or (iii) individuals. In the first two categories, the farmers have usufructuary rights over land and so, have least interest in its development and protection from soil erosion. Free range grazing is responsible for huge sediment yield from hill slopes as well as valley lands. With proper vegetative cover, maximum rainwater could be retained in situ and the soil can retain sufficient moisture for growing winter crops (Sharma 2001, Sharma and Sharma 2009a, b, c).

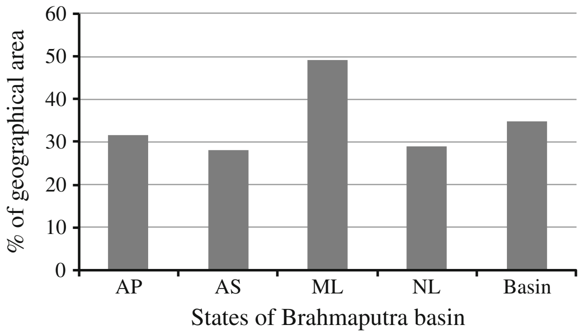


Fig. 3 Land degradation in Brahmaputra states of India (*AP* Arunachal Pradess, *AS* Assam, *ML* Meghalaya, *NL* Nagaland)

This would also help in arresting runoff and soil loss and, better ecological conditions could be assured. The region has a foodgrains deficit of about 2.5 million tonnes and the deficit gap is widening year after year (Sharma 1999). The important issue is to promote the conservation and sustainable use of natural resources which allow long term economic growth and enhancement of productive capacity, along with being equitable and environmentally acceptable (El Bassam 1997). About 34.7 % of the land of the basin has been degraded due to faulty agricultural practices (Fig. 3).

3.3 Sediment Yield

3.3.1 Present Scenario

Erosion and sediment transport are part of the natural evolution of the landscape. They constitute some of the most fundamental problems of the development of agriculture, water management, forestry and for utilization of natural resources (Kostadinov 2004). Soil erosion is the main agent of lateral material transport on anthropogenically affected land in the basin.

A part of the huge sediment load emanating from the catchments of the basin settles down on the bed of Brahmaputra river, thus, reducing the size of the channel and water intake capacity. The rainwater during May to September overflows its banks, resulting in floods of high magnitude. About 31,740 km^2 area is prone to floods in the basin, out of which 3,609 km^2 area is affected by floods every year. The huge amount of nutrient load present in flood water show the amount of nutrients removed from the basin, thus, rendering the soils infertile. Nutrients export from slopes has decreased soil fertility and surface water is polluted by mineral and organic substances held by transported sediment. Particle structure is the major factor in regulating the behaviour of suspended material in aquatic environment (Nicholas and Walling 1996). While the magnitude and duration of sediment storage depends mainly on sediment supply and hydrological conditions, the stored load may comprise a significant portion of a systems' annual sediment budget (Ownes et al. 1999). Unless drastic measures will be undertaken to change the existing land uses, the basin would continue to reel under the loss of valuable soil and vegetation resources and food insecurity. Sediment transport and land-water management in Brahmaputra basin for sustainable development of water resources and food security.

3.3.2 Effect of Land Use and Rainfall on Soil Loss

There was significant variation in the soil loss from the shifting cultivation and other land uses under various rainfall regimes (Fig. 4a). The average sediment yield was only 0.44, 2.68, 1.47, 0.31, 0.73 and 2.27 % in fodder, forestry, agro-forestry, agriculture, agri-horti-silvi-pastoral and horticulture land use systems of that of shifting cultivation. The differences were so large that the soil erosion had to be shown as 1/5th root of the actual values to conveniently accommodate in a figure (Fig. 4b). The rainfall during a particular year significantly affected the sediment yield from the watersheds having different farming systems (Fig. 4c). The annual sediment yield varied from 0.14 to 2.3 t ha^{-2} in new land uses (other than shifting

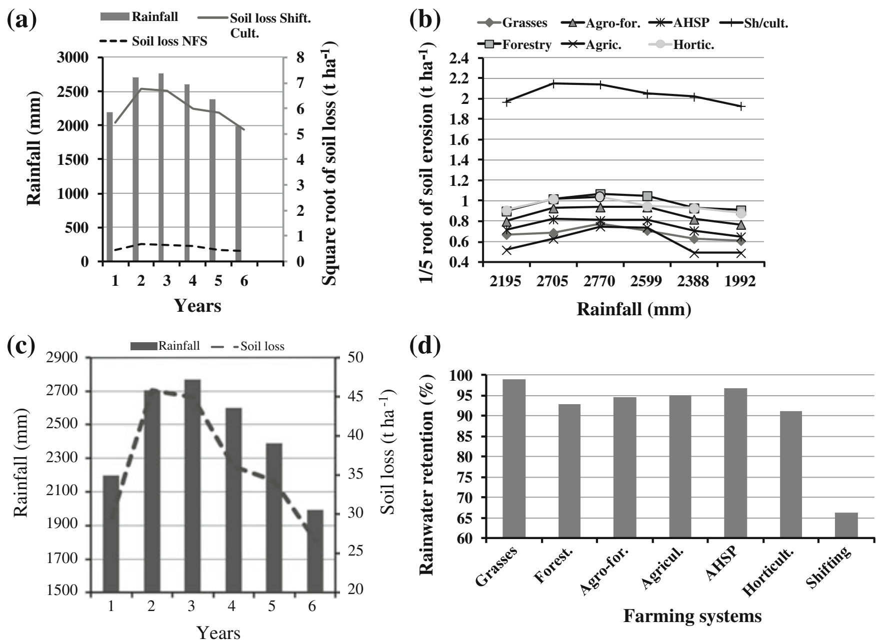


Fig. 4 Impact of rainfall regimes on the soil loss in shifting cultivation and other farming systems combined (**a**), on 1/5th root of soil erosion in different farming systems (**b**), rainfall and soil loss during different years of study (**c**) and, impact of farming systems on in-situ rainwater retention (**d**)

cultivation) as against 36.2 t ha^{-2} in shifting cultivation. The major reasons for variability in sediment yield were due to the amount and intensity of rainfall, slope, amount and nature of vegetation cover, soil texture and management practices, including water and soil conservation measures. While in shifting cultivation 34.1 % of rainwater escaped as runoff, it varied from 0.9 to 7.1 % in the new land use systems (Fig. 4d). Maximum of 99.1 % of rain water was retained in livestock based land use system, followed by agriculture (99.1 %). It was reported earlier also that more than 95 % of rain-water can be retained in situ by following these land use systems (Anonymous 2000).

Annual soil loss due to erosion was very much higher than the permissible limit (10 t ha^{-2}) in the shifting cultivation whereas; it was highly significantly low in the new land use systems. The soil loss was very low in newly tried land use systems due to reduced runoff because of proper vegetation cover and water and soil conservation measures undertaken. These land use systems could be adopted to replace shifting cultivation in the region keeping in view the topography, slope and nearness to the market. The sediment yield from different farming systems was found to be highly significantly related to surface runoff (r = 0.944**), sub-surface runoff (r = 0.760**) and total runoff (r = 0.855**).

3.3.3 Sediment Yield from Brahmaputra Basin

Fluvial erosion, sediment transport and sediment deposition make up the complex of channel processes. As the demand for water increases and the human impact on the landscape broadens, the attention of different agencies turns to balancing the need of humans with those of catchment and riverine ecosystem. Sediment yield in the Brahmaputra basin of India can be divided into formation and transport of sediments, and sediment yield processes and is affected by natural factors like precipitation, steep slopes, soil characteristics and vegetation cover and; anthropogenic factors such as prevalence of shifting cultivation, land tenure system, free range grazing, construction of roads, buildings and other structures as well as ignorance of people about methods of conservation and their relative benefits. Since sediment originates from a combination of different sources, it is useful as a part of monitoring process, to be able to quantify not only the change over time in sediment yield, but also identify the sources of sediment, enabling more conclusive results to be drawn about the effect of conservation practices in reducing sediment yield. The land use has many effects on environmental systems as well as influencing the efficiency of slope-channel transfer. High sediment loads in surface water in the basin have reduced water quality with a negative effect on ecology. The study has shown high sediment concentrations in runoff water draining from the catchments in the Brahmaputra basin. The sediment load per litre of runoff from watersheds ranged from 1,250 to 20,300 mg soil, 5.4–23.6 mg NO_3–N, 2.3–6.5 mg P–PO_4, 17.2–35.8 mg K_2O, 0.4–1.8 mg Zn, 0.9–2.7 mg Mn, 6.5–12.0 mg Mg, 7.1–18.4 mg Fe and 4.0–7.2 mg SO_4. The sediment transport from the catchments showed spatial and temporal variations. Intensification of agriculture has potentially harmful impact on already fragile hydrological system. A major sink for the forest litter and humus transferred from the hill slopes was found valley land adjacent to the hills, which retain about 50–65 % of them, making the soil highly fertile. The annual sediment yield from the Brahmaputra basin, alone constitutes, 438.7 million tonnes of soil and 506, 81, 376, 18.2, 11.4, 45.8 and 34.9 thousand tonnes of N, P, K, Mn, Zn, Ca and Mg, respectively (Sharma and Sharma 2004b). In pursuit of increasing agricultural production, a rapid increase in the use of agricultural chemicals will further degrade the water quality of the freshwater resources, affecting human health and aquatic ecosystems, and alter the carbon cycle and biological and life support ecosystem in the region

The suspended sediment load transported by the Brahmaputra river represent a mixture of sediment derived from different locations and from different sediment sources within the basin. Information on suspended sediment provenance is an important requirement in the examination of sediment routing and delivery and in the construction of sediment budgets (Trimble 1983; Walling 1988). Sediment sources can also exert a fundamental control on the sediment associated transport and contaminants in the river basin, since the source of sediment is likely to influence its physical and chemical properties and its contaminant loading. The sediment transfer through Brahmaputra basin largely depends on the relative influence of erosion by rainfall and flood events and stabilization by vegetation (Sharma and Sharma 2003, 2004b). Vegetation in the riparian zone and in the catchments control erosion through vegetation stability. The slope is extremely important since it influences the velocity of sediment flow and detachment of sediment particles from the soil surface to trigger erosion. Rainfall, intensity and amount are significant as it forms the medium carrier of sediment load. The vegetation cover reduces erosion by adsorption of the impact of raindrops, reducing the velocity and scouring power of the runoff, binding soil with roots and reducing runoff volumes by increasing percolation in the soil. The networking of the Brahmaputra river is shortening due to filling of the river channel by sediments from cultivated slopes, fall of groundwater level due to deforestation and river flow diversions. About 51.4 % of the soil and 57.6 % of the nutrient load was carried out of the basin towards sea. This corroborate the findings of Walling and Webb (1983), who mentioned that the estimates of rates of on-site soil loss do not provide direct indication of the suspended sediment yield from local catchments as much of the sediments mobilized by soil erosion may be deposited prior to reaching a water course or stream and the specific sediment yield of the catchment may be substantially less than the equivalent local rates of soil loss. The annual variability in sediment yield is a reflection of the variability in precipitation and runoff. In the present study, the micro-watershed size varied from 0.9 to 3.5 ha and the total sediment yield extrapolated for the Brahmaputra basin may be higher. The unit rate of sediment yield decreases as drainage area increases. Branson et al. (1981) presented a graph illustrating the relationship between sediment yield and drainage area based on the work of several researchers. The decrease in sediment yield can be explained in part by increases in deposition and sediment storage within the channel network with increasing watershed size. Available evidence suggests that much of the sediment is derived from erosion of agricultural lands. The rates of soil loss provide a useful basis for identifying areas at risk from accelerated soil loss and for assessing patterns of sediment mobilization by soil erosion (Evans 1990; Banasik et al. 2005). The higher sediment yield in the shifting cultivation in the basin can be attributed to sparse vegetation cover and comparatively more slope than other farming systems.

3.3.4 Carbon Displacement in the Basin

Sediment load in the runoff originate from different sources, with the relevant contribution varying over time and space due to various erosion processes in the basin. Agricultural activities significantly influenced the contemporary geomorphic processes and expansion in the cultivated land and mismanagement of rain water has increased the rate of soil erosion in the Brahmaputra basin. Information on the suspended sediments provenance is an important requirement in the examination of sediment routing and delivery, and in the construction of sediment budgets (Trimble 1983; Walling 1988). The total organic carbon displaced was 3.05 Mt, which was 0.67 % of the total sediment load. Interestingly, the carbon load was 51.4 % in the sediments deposited on land as compared to 48.6 % in the sediment carried to the sea. The organic carbon content in the sediments deposited in the river, other sinks combined, on land and sea was, 0.61, 0.75, 0.71 and 0.63 %, respectively. The low organic carbon content in the sediments deposited in the river bed may be due to high velocity of water flow. The higher content of carbon in the temporary storages, flood areas, and lake and reservoirs may be due to stagnation of water for a longer period and its subsequent deposition at the surface. The deposition of sediments in major possible sinks in the Brahmaputra basin, calculated by multiplying the total sediments with the coefficient value, was 65.1, 34.9, 49.9, 44.0, 22.7, 4.6 and 233.9 Mt in the river, flooded area, streams/tributaries, temporary water storages, lakes and reservoirs and the sea, respectively. Ownes et al. (1999) also reported that while the magnitude and duration of sediment storage depends mainly on the sediment supply and hydrological conditions, the stored load may comprise a significant position of a system’s annual sediment budget. One of the most important factors defining the sediment supply to river channels is the area of cultivated land within the drainage basin and its dynamics during the period of intensive agriculture. In Brahmaputra basin, not all the sediments displaced from different location are transported out of the basin, but about 48.6 % of the sediment load is deposited in different sinks in the basin itself.

3.4 Runoff from Experimental Area

The impact of rainfall and farming systems on the surface and base flows (runoff) has been shown in Fig. 5a–d. The runoff from different farming systems varied from

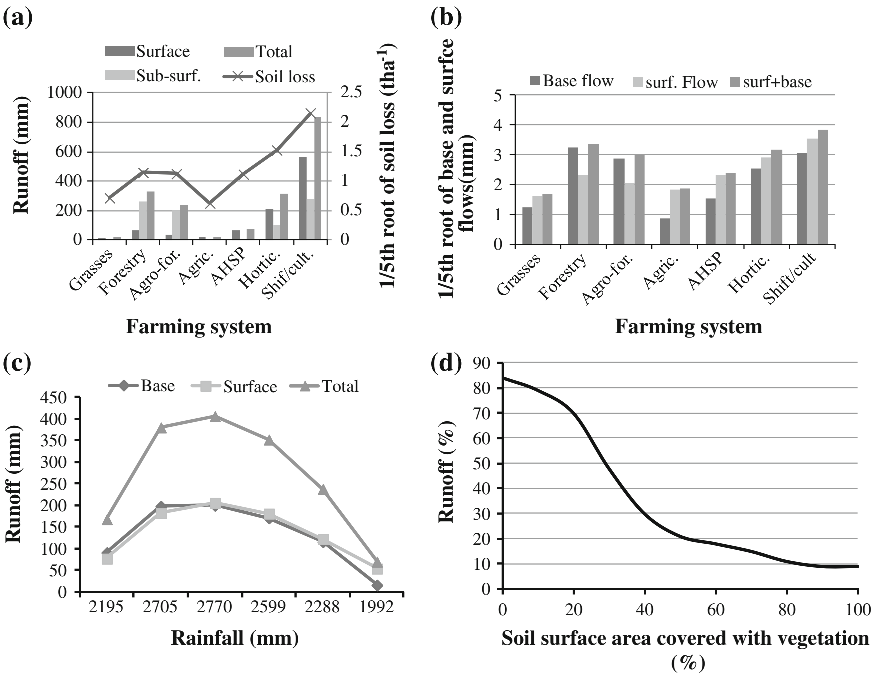


Fig. 5 Impact of farming systems on the runoff (**a**), 1/5th root of surface and base flows (**b**), effect of rainfall amount on the base and surface flows (**c**) and, vegetation covered land surface area and per cent runoff generation (**d**)

15.7 to 416.3 mm in new land use systems as against 835.8 mm in shifting cultivation (Fig. 5a, b). Total mean runoff (surface and sub-surface flows) was found to be 19, 333, 237, 21, 77, 319 mm in grasses/fodders, forestry, agroforestry, agriculture, agri-horti-silvi-pastoral and horticulture based farming system compared to 835 mm from shifting cultivation. Since there were large variations, 1/5th toot of actual values was used to depict the comparison. Besides, the amount of vegetation on the soil surface had a great influence on the runoff generation. The data provided in Fig. 5c, show that amount of rainfall had a great influence on the runoff generation during different years. The runoff was found to be about 87 % of rainfall from bare soil with 70 % slope compared to 9 % of rainfall from soil covered with dense vegetation and having a slope of 30 % (Fig. 5d). Maximum rain water could be retained in situ and the soil can retain sufficient moisture for growing winter crops (Sharma 2001).

Sediment yield and runoff frequency are major measures of geomorphic activity. Runoff frequency is closely associated with the pattern of local precipitation, and changes in frequency can reflect changes in vegetation, land use and climate and, can also be a major determinant of sediment yield. The aim of management strategies in runoff agriculture will obviously be to maximise biomass production per unit of water collected. This goal may be achieved by intercropping annuals into stands of trees grown on stored water (Lövenstein et al. 1991). Higher runoff from the forestry land use in the present study, was mainly due to the fact at initial stages of tree growth; most of the land surface area remained bare, resulting in more runoff and soil erosion. In case of horticulture land use, the slope gradient was also higher compared to other land use systems.

3.5 Water Quality Management in Brahmaputra Basin

3.5.1 Significance of Water Quality

Water quality is an important parameter of the ecosystems as it is an indicator of health of a community, food to be produced, economic activities, ecosystem health and biodiversity. The management of water quality in a river basin require determination of the actual health of the river. Because river pollution is often caused by processes and activities at a watershed scale, watershed assessment is related to the understanding and protecting water quality. Sediment has to be tested to understand their polluting potential. Poorly soluble hydrophobic compounds remain attached to the soil particles and can be detected only in extremely small quantities in the water. Water quality can be affected by sewage, pathogens in waste streams from humans and domesticated animals, agricultural runoff and human wastes loaded with nutrients. There is likelihood that water quality may deteriorate the functioning of ecosystems and in extreme cases, the system may collapse. More nutrients in freshwater and coastal ecosystems, can possibly lead to algal blooms and oxygen-depletion, making most animal life impossible. Climate change and changes in hydrological patterns will affect water quality and exacerbate water pollution from sediments, nutrients, dissolved organic carbon, pathogens, pesticides and salt, as well as thermal pollution. Long-term development of freshwater needs holistic management of resources and a recognition of the interconnectedness of the elements related to freshwater quality. Major problems affecting the water quality arise in variable order of importance depending upon situation, from ill treated domestic sewage, inadequate control on the discharges of industrial waste waters, destruction of catchment area, deforestation, shifting cultivation and poor agricultural practices. Freshwater management should be done and based on the balanced consideration of the needs of the people and the environment. Safe water supplies and the environmental sanitation are important for protecting environment, improving health and alleviating poverty. Sediment free water is necessary for different uses, including drinking. The effect of sediment on the aquatic life has been clearly established as it affects spawning and habitat. The affects include inhibition of photosynthesis and damage to fish (Hoak 1959). Sunlight is essential for the synthesis of organic matter by plants and chlorophyll bearing organisms.

3.5.2 Sediments and Water Quality

The samples of surface and groundwater taken from the areas near the agricultural fields showed that the values of NO_3–N, P–PO_4, chlorides and sulphates ranged from 28.4 to 43.5, 5.9 to 10.6, 41.8 to 77.5 and 17.3 to 25.8 mg l^{-1}, respectively, in samples of surface water and, 37.6 to 55.1, 4.3 to 7.7, 91.4 to 118.2 and 22.3 to 30.4 mg l^{-1} in groundwater (Fig. 6a). In Fig. 6b, the mean values of these nutrients have shown. Except P–PO_4, other nutrients recorded higher concentration in groundwater compared to surface water. The NO_3–N concentration was higher than the prescribed limit for drinking of water (45 mg l^{-1}) in a few samples. Most of the samples were well below the standard prescribed limits. This was due to the reason that the use of fertilizers and agricultural chemicals is less in the Brahmaputra basin. Runoff from improperly managed soils, carry soil particles with it leading to soil erosion and degradation of the land.

Clay may stay in suspension for very long periods, contributing significantly to water turbidity, thereby deteriorating the water quality. Sediment affects water quality physically, chemically, and biologically. Sediment often carried organic matter, animal or industrial wastes, nutrients, and chemicals. Phosphorus is very immobile in most soils and concentrates in the top few inches of soil. Sediment also may carry pesticides that may be toxic to aquatic plants and animals. Certain dissolved nutrients, like nitrate, and pesticides can reach the groundwater by moving down through the soil. The capacity of the soil to use, retain, or reduce the undesirable effects of sediment varies significantly according to the physical, chemical, and biological properties of the soil and the sediment properties. Reducing soil erosion is important in reducing the damaging effects of sedimentation. In the long-term study undertaken, appropriate soil and water conservation measures were used to reduce soil erosion. Crops themselves as well as crop residues were used to hold the soil in place and allow water to move into it rather than to run off the surface. Grassed waterways and terraces were used to provide the

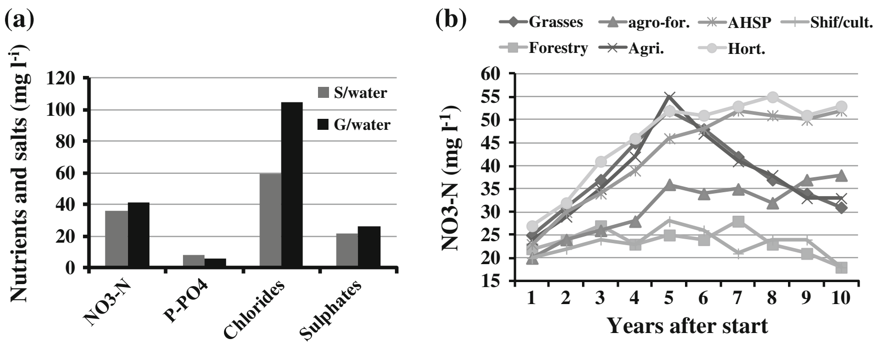


Fig. 6 Mean nutrient and salts concentration in surface and groundwater (mg l^{-1}) (**a**) and, changes in NO_3-concentration in groundwater over a period of time (**b**)

necessary control. As sediments affect the surface and groundwater quality, inputs need to be based on realistic crop yield. Water quality can be improved while protecting the productivity and value of the soil for agricultural and industrial uses.

4 Conclusions

1. Both, natural and anthropogenic factors prevailing in the Brahmaputra basin have attributed to the vulnerability of the basin to floods and, land and environmental degradation.
2. The practice of shifting cultivation as a method of food production is no more economical and sustainable and has become a resource depleting practice causing huge soil erosion from the hill slopes.
3. The Brahmaputra basin has plenty of water resources but their misuse has resulted in large scale erosion of soil and associated elements. Transport of soil and sediment associated nutrients in the runoff and its deposition in various sinks is a matter of concern.
4. Deforestation and denudation of basin has led to water scarcity because the natural water cycle has been upset. There is urgent need to replace the age old practice of shifting cultivation with eco-friendly, sustainable and socially acceptable land use systems. The present land tenure system requires modification by enacting suitable laws by the government and giving ownership rights to the cultivators so that they may feel a sense of belonging and responsibility of judicious management.
5. Judicious management of rainwater is necessary to reduce runoff and associated sediment load. Immediate solution to water problem requires awareness among users through radical government policies as well as relevant institutional reforms.

References

Anonymous (2000) Basic statistics of the north-eastern region. North Eastern Council, Shillong

Banasik K, Madeyski M, Mitchell JK, Mori K (2005) An investigation of lag times for rainfall-runoff-sediment yield events in small river basin. Hydrol Sci J 50:857–866

Branson FA, Gifford GF, Renard KG, Hadley RF (1981) Rangeland hydrology, range science series, vol 1. Society for Range Management, Denver

Douglas I (1967) Man, vegetation and the sediment yield of rivers. Nature 215:925–928

El Bassam N (1997) Fundamentals of sustainability in agricultural production systems and global food security. In: El Bassam N, Behl RK, Prochnow B (eds) Sustainable agriculture for food, energy and industry. James and James Publishers, London, pp 1–12

Evans R (1990) Soils at risk of accelerated soil erosion in England and Wales. Soil Use Manage 6:125–131

Goswami DC (1985) Brahmaputra river, Assam, India: physiography, basin denudation and channel aggradation. Water Resour Res 2197:959–978

Hoak RD (1959) Physical and chemical behaviour of suspended solids. Sewage Industrial Wastes 31:1401–1408

Hovius N (1998) Controls on sediment supply by large rivers, relative role of eustasy, climate, and tectonism in continental rocks. Society of sedimentary geology. Special, Publication No. 59. pp 3–16

Jackson ML (1973) Soil chemical analysis. Prentice Hall of India, New Delhi

Kostadinov S (2004) Soil erosion and sediment transport within small torrential drainage basins of Serbia. In: Poster report during symposium on "sediment transport through the fluvial system" held in Moscow, August 2–6, pp 16–21

Lövenstein HM, Berliner PR, van Keulen H (1991) Runoff agroforestry in arid lands. Forest Ecol Manage 45:59–70

Ludwig W, Probst JL (1998) River sediment discharge to the oceans: present-day controls and global budget. Am J Sci 298:265–295

Milliman JD, Meade RH (1983) World-wide delivery of river sediment to the oceans. J Geology 91:1–21

Milliman JD, Syvitski JPM (1992) Geomorphic/tectonic control of sediment discharge to the ocean: the importance of small mountainous rivers. J Geology 100:525–544

Nicholas AP, Walling DE (1996) The significance of particle aggregation in the over bank deposition of suspended sediment on river flood plains. J Hydrol 186:275–293

Nichols MH, Renard KG, Osbor HB (2002) Precipitation changes from 1956–1996 on the walnut gulch experimental watershed. J Am Water Resour Assoc 38:161–172

Ownes PN, Walling DE, Leek GJL (1999) Deposition and storage of fine grained sediment within the main channel system of the river tweed. Scotland, Earth Surf Proc 24:1061–1076

Sharma UC (1997) Some indigenous farming systems of north-eastern India. Asian Agri-Hist 1:267–273

Sharma UC (1998) Water management by the tribes of north-eastern region: the traditional wisdom and future strategies. Fertilizer News 43:43–49

Sharma UC (1999) Food security in the northeast: new paradigms. In: Sundariyal RC, Shankar U, Upreti TC (eds) Persrpective for planning and development in north-eastern India. G.B. Pant Institute of Himalayan Environmental and Development, Almorah, pp 197–212

Sharma UC (2001) Effect of farming system type on in-situ ground water recharge and quality in northeast India. In: Gehrels H, Peters NE, Hoehn E, Jenson K, Leibundgut C, Griffioen J, Webb B, Zaadnoordijk, WJ (eds) Impact of human Activity on Groundwater Dynamics, IAHS Publ No. 269, IAHS Press, Wallingford, pp 167–169

Sharma UC (2003) Impact of population growth and climate change on the quantity and quality of water resources in the northeast region of India. In: Bloschl G, Franks G, Kumagia M, Musiake K, Rosbjerg D (eds) Water resources system—hydrological risks, management and development. IAHS Publ No. 281, IAHS Press, Wallingford, pp 349–357

Sharma UC, Prasad RN (1995) Socio-economic aspects of acid soil management and alternate land use systems for north-eastern states of India. In: Date RA, Grundon NJ, Rayment GE, Probert ME (eds) Plant-soil Interactions at Low pH. Kluwer Academic Press, The Netherlands, pp 689–696

Sharma UC, Sharma V (2003), Mathematical model for predicting soil erosion by flowing water in ungauged watersheds, In: Boer D, Wojciech F, Mizuyama T, Pietroni A (eds) Erosion predictions in ungauged basins: integrating methods and techniques. IAHS Publ No. 279, IAHS Press, Wallingford, pp 79–83

Sharma UC, Sharma V (2004a) Implications of nutrient and soil transfer with runoff in the northeastern region of India. In: Golosov V, Belyaev V, Walling DE (eds) Sediment transfer through the fluvial system. IAHS Publ No. 288, IAHS Press, Wallingford pp 488–493

Sharma UC, Sharma V (2004b) Sediment budgets and sinks in the Brahmaputra basin and their agricultural and ecological impacts. In: Webb B, Acreman M, Maksimovic C, Smithers H,

Kirby C (eds) Hydrology: science and practice for the 21st century, vol 2. British Hydrological Society, London, pp 399–405
Sharma UC, Sharma V (2009a) Plant-animal-soil-hydrology interactions in the northeastern region of India. Biologia 64:460–464
Sharma UC, Sharma V (2009b) Adaptive surface and groundwater management under vulnerable hydro-ecosystem in the northeastern region India. In: Bloschl G, Giesen N, Murlidharan D, Ren L, Seyler F, Sharma UC, Vrba J (eds) Improving integrated surface and groundwater management in a vulnerable and changing world. IAHS Publ No. 330, IAHS Press, Wallingford, pp 69–75
Sharma UC, Sharma V (2009c) Assessment of flood events in data sparse Brahmaputra basin in the northeast India. In: Yilmaz KK, Yucel I, Gupta HV, Wagener T, Yang, Savenije DH, Neale C, Kunstmann H, Pomeroy J (eds) New approaches to hydrological predictions in data sparse regions. IAHS Publ No. 333, IAHS Press, Wallingford, pp 220–225
Singh A, Singh MD (1978) Effect of various stages of shifting cultivation on soil erosion from steep hill slopes. Indian Forester 106:115–121
Srivastva H, Mandal GS (1995) Highest rainfall during 24 hours in India. Mausam 46:99–100
Summerfield MA, Hulton NJ (1994) Natural controls of fluvial denudation rates in major world drainage basins. J Geophys Res 99:13871–13883
Trimble SW (1975) Denudation studies: can we assume steady state? Science 188:1207–1208
Trimble SW (1983) A sediment budget for coon creek basin in the driftless area, wisconsin. 1853–1977. Am J Sci 283:454–474
Verstraeten G, Poesen J (2001) Factors controlling sediment yield from small intensively cultivated catchments in a temperate humid climate. Geomorphology 40:123–144
Walling DE (1988) Measuring sediment yield from river basins. In: Lal R (ed) Soil erosion on agricultural land. Soil and Water Conservative Society, Ankeny, pp 39–73
Walling DE (1994) Measuring sediment yield from river basin. In: Lal R (ed) Soil erosion research methods. Soil and Water Conservation Society, Ankeny, pp 39–80
Walling DE (1997) The response of sediment yields to environmental change. In: Human impact on erosion and sedimentation, proceedings of the rabat symposium S6, July 1997, vol 245. IAHS Publications, pp 77–89
Walling DE (2000) Linking land use, erosion and sediment yields in river basins. Hydrobiol 410:223–240
Walling DE (2008) The changing sediment loads of the world's rivers. Ann Warsaw Univ Life Sci 39:3–20
Walling DE, Webb BW (1983) In: Gregory KJ (ed) Patterns of sediment yield background to palaeohidrology. Wiley, New York, pp 69–99
Wilson L (1973) Variations in mean annual sediment yield as a function of mean annual precipitation. Am J Sci 273:335–349

Contamination of Sediments in the German North Sea Estuaries Elbe, Weser and Ems and Its Sensitivity to Climate Change

Carmen Kleisinger, Holger Haase, Uwe Hentschke and Birgit Schubert

Abstract Projections of climate-induced changes in temperature and precipitation let expect also altered frequencies and intensities of extreme hydrological events such as floods or prolonged periods of low river discharges. Particularly such extremes may, moreover, lead to modified inputs of particulate matter into rivers and estuaries, and may thus affect the quality of estuarine sediments. This study focuses on the assessment of potential climate-induced changes of particle-bound contaminant concentrations in the estuaries of the rivers Elbe, Ems and Weser and the resulting challenges for sediment management in the navigable waterways there. The estimation of climate-induced changes of contaminant concentrations in estuarine particulate matter (PM) was based on results of projections on the fluvial PM input into the Elbe estuary in the near (2021–2050) and far future (2071–2100) and on assumed extreme changes of such inputs. A mixing model using the concentrations of selected contaminants as indicators for marine and fluvial PM was applied. Distinct changes of contaminant concentrations were found only for the far future and with the assumed extreme PM inputs in the inner Elbe estuary. And only for the inner Elbe estuary the worst-case scenario indicated that concentrations of some organochlorine contaminants in the far future exceed the national assessment criteria for the handling of dredged material within coastal waterways more distinct than today. Therefore, adaptations of practices for the management of dredged material to higher particulate matter contaminations should be considered there in the medium or long-term perspective.

1 Introduction

Numerous studies show that particulate matter (PM), i.e. suspended particulate matter (SPM) and sediments, in estuaries often are contaminated (Müller and Förstner 1975; Förstner et al. 1990; Brügmann 1995; Irabien et al. 2008; Graydon

C. Kleisinger (✉) · H. Haase · U. Hentschke · B. Schubert
Federal Institute of Hydrology, Am Mainzer Tor 1, 56068 Koblenz, Germany
e-mail: Kleisinger@bafg.de

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_9

et al. 2009; Hatje and Barros 2012). Contamination of PM plays an important role for the ecological quality of water bodies and may compromise the achievement of the objectives of the Water Framework Directive (WFD) (EU 2000, 2013) and the Marine Strategy Framework Directive (MSFD) (EU 2008). In addition, PM contamination has to be taken into account in dredged material management of navigable waterways. Contaminated sediments can, for example, lead to restrictions for the aquatic depositing of dredged material in estuaries and coastal areas (OSPAR 2014; BfG 2014).

In surface waters, trace metals and some priority organic compounds are preferentially particle-bound and are mainly associated with fine-grained fractions of the PM (e.g. Ackermann et al. 1983; Horowitz 1991; OSPAR 2002). In the estuaries of the rivers Elbe and Weser, a decrease of contaminant concentrations towards the sea can be observed for contaminants having their main sources in the inland reaches of the rivers (e.g. ARGE Elbe 1980; Förstner et al. 1990; Knauth et al. 1993; Prange 1997; Ackermann and Schubert 2007; www.fgg-elbe.de; Banat et al. 1972; Ackermann 1998; BfG 2013a, b). This gradient is attributed to the mixing of low contaminated marine sediments transported upstream and high contaminated fluvial sediments transported downstream the estuaries (Table 1). In the Ems estuary, however, contaminant concentrations are close to the marine level from the river mouth up to the tidal weir, as fine PM of marine origin dominates over more contaminated freshwater sediments throughout the estuary (Ackermann 1998, 2004; Kowalewska et al. 2011).

Furthermore, particularly in the mixing zone of marine and fluvial PM in the Elbe and the Weser estuaries, considerable within-year variations of concentrations have been observed for many contaminants (Ackermann and Schubert 2007; Kowalewska et al. 2011). Monitoring of contaminants in freshly deposited sediments and PM revealed that concentrations mainly vary with inflow of freshwater transporting PM and associated contaminants from the inland reaches of the rivers to the estuaries. With increasing river discharge, and accordingly increasing input of particle-bound contaminant loads, concentrations of contaminants in estuarine PM increase too, and vice versa.

The boundary conditions controlling contaminant concentrations in estuaries may be influenced by several factors, e.g. by reduction measures for contaminant sources in the inland reaches of the rivers, by anthropogenic activities like construction measures, and by climate change.

Primarily, this study investigates the influence of climate change on the contamination of PM in the North Sea estuaries of the rivers Elbe, Weser and Ems. As a result of projected climate-induced changes of temperature and precipitation (IPCC 2007), alterations of the frequency and intensity of extreme events such as floods, storm surges or extended periods with low river discharge and a rise of the sea level are expected. An increase in intensity and occurrence of floods would result in an additional input of contaminated sediments from the inland reaches of the rivers to the estuaries and consequently, a deterioration of the quality of estuarine PM may occur. In case of more frequent low river discharge situations and of sea level rise, the upstream transport of slightly contaminated sediments of marine origin may be intensified, and cause decreasing concentrations of contaminants in PM. As PM in the estuaries are assumed to be a composition of marine and fluvial sediments, a linear mixing model was applied to estimate the sensitivity of contaminant concentrations to climate-induced alterations of the fluvial PM load to the estuaries. Furthermore, the assessment takes into consideration the effects of a reduction of fluvial contaminant concentrations as a result of e.g. restoration measures in the inland reaches of the rivers. Finally, the potential implications of climate-change induced alterations of contaminant concentrations for the maintenance of estuarine waterways and the need to adapt maintenance practices are assessed.

1.1 Estuaries Studied

Although, the estuaries of the rivers Elbe, Weser and Ems (Fig. 1) have some features in common, there are some differences that may influence the sensitivity of contaminant concentrations to climate change (Table 1). Generally, concentrations of many contaminants show a decrease towards the sea and distinct within-year variations in the inner estuaries (e.g. Elbe estuary, Fig. 2).

Differences between the estuaries under consideration exist e.g. in contaminant patterns and concentration levels, the length of the estuaries, the level of river discharges, the PM load at the tidal weir, and the extent of upstream transport of marine, slightly contaminated fine-grained sediments (Table 1).

Table 1 Characteristics of the estuaries

	Elbe	Weser	Ems
Mean discharge (m^3/s) (2003–2012)	695 (Neu-Darchau, km 536.4)	299 (Intschede, km 331.3)	76.5 (Versen, km 171.5)
Frequent range of discharge (m^3/s)	400–900	210–400	45–100
Particulate matter input (t/a) (2003–2012)	600,000 (Hitzacker, km 522.9)	400,000 (Intschede, km 331.3)	65,000 (Lathen, km 253.3)
Extent of marine influence (% of estuary length)[a]	17 (Bunthaus, km 609.77)	36 (Farge, UW[b] km 26.25)	up to the tidal limit (Herbrum, km 212.75)
Main contaminants	Hg, Cd, Zn, Cu, HCB, p,p′-DDX	Cd, Pb, Zn, Cu, PCB	Pb, Cd, Zn, Cu
Contamination level	High	Medium to high	Low to medium

Hg: mercury, Cd: cadmium, Zn: zinc, Cu: copper, Pb: lead, HCB: hexachlorobenzene, p,p′-DDX: p,p′-DDT, p,p′-DDD, p,p′-DDE, PCB: polychlorinated biphenyls

[a] Tidal weir = 0 %

[b] Unterweser

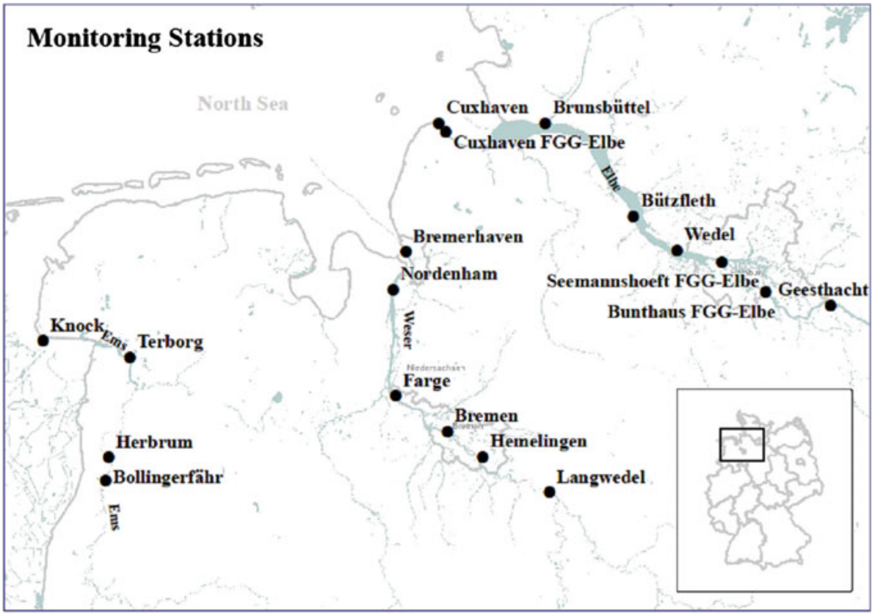


Fig. 1 Monitoring sites at the Elbe, Weser and Ems estuaries, operated by the Federal Institute of Hydrology (BfG) and the River Basin Community Elbe (FGG Elbe)

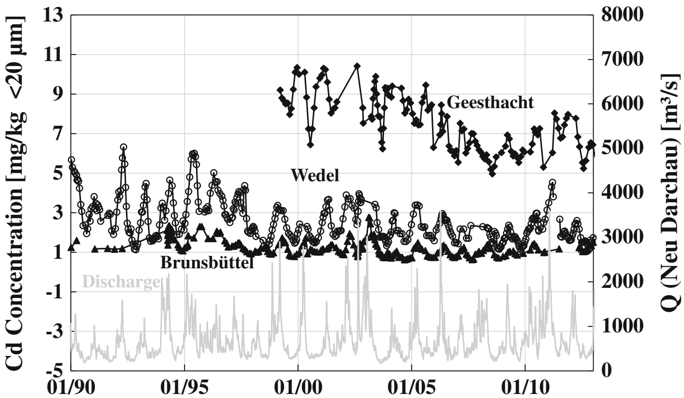


Fig. 2 Cadmium (Cd) concentrations in the fraction <20 µm of particulate matter at three monitoring stations in the Elbe estuary (Geesthacht, km 584.1, Wedel, km 642, Brunsbüttel km 696.3) and river discharge (Q) from Neu-Darchau (km 536.4)

2 Methods

In order to identify potential alterations of estuarine contaminant concentrations resulting from (a) climate-induced changes of PM input to the estuary or (b) a reduction of fluvial contaminant concentrations, sensitivity studies based on a binary mixing model were carried out (e.g. Ackermann and Schubert 2007; Martinez-Carreras et al. 2008; Walling 2005). So simplifying, fine-grained fluvial and marine PM are regarded as main sources for the contamination of estuarine PM.

2.1 Database

The results from a long term monitoring programme on contamination of PM at two to three selected sites per estuary, including the monitoring site upstream of the tidal weir (Fig. 1) were used (unpublished data). For trace metals, monitoring started at most sites in the 1980s; regular data for organic contaminants are available only since 1999. Usually, the sampling frequency is once per month. In case of flood events, the sampling frequency in the Elbe estuary is increased to once in a week or in a fortnight. Investigations comprised concentrations of trace metals and selected organic contaminants. In general, the investigated contaminants are bound to the fine-grained size fractions. In order to compensate for differences in grain-size distribution, trace metals are analysed in the <20 µm fraction, and concentrations of organic contaminants, analysed in the <2,000 µm fraction, are normalised to the <63 µm fraction (Ackermann et al. 1983; Kersten and Smedes 2002; OSPAR 2002; BGBl 2011).

2.2 Reference Status of Contamination

As long-term monitoring data revealed decreasing concentrations over time for many contaminants, trends were tested with the Mann-Kendall test (Gilbert 1987) to identify an appropriate reference period, i.e. a period with no significant temporal trends. In order to reflect general contamination levels excluding extreme concentrations during extreme floods or prolonged periods of low river discharges, reference concentrations in fluvial and estuarine PM as well as their variability were derived from monitoring results of periods with a range of frequent and thus medium river discharges (e.g. 400–900 m^3/s in the river Elbe, cf. Table 1). The defined marine level was based on results from monitoring campaigns in the German Bight since the year 2005 (BfG 2005, 2013a). For estuarine concentrations, average temporal shifts (e.g. Ackermann and Schubert 2007; Kowalewska et al. 2011) observed between the maxima of river discharge and the maxima of contaminant concentrations were taken into consideration. The variability of reference concentrations includes uncertainty of measurements and variations as a result of dynamic boundary conditions.

2.3 Mixing Model

The application of the model required several simplifying assumptions. First of all, it was presumed that the PM within the estuary is a binary mixture of contaminated fluvial PM and low contaminated marine PM. Contaminants suitable for estimating the mixing ratio of the two sediment sources within the estuary should have no local sources in the estuary and their concentrations should add linearly. Concentrations of contaminants potentially suitable as indicators for both, the marine and the fluvial PM as sources for the estuarine PM, should differ significantly. Differences in the levels of contamination were tested with the Welch's test (Welch 1947) that tolerates unequal variances of the datasets. Furthermore, contaminants should behave conservative, i.e. they should not undergo considerable chemical or biological alterations, and remobilisation to the water phase should be negligible (Walling 2005; Ackermann and Schubert 2007). Organic contaminants under consideration were regarded as sufficiently stable for using them as indicators.

(a) **Impact of climate-induced changes of fluvial PM input**

First of all, mixing ratios of marine to fluvial PM (MR_{ref}) were estimated (Eq. 1) for the reference period. Modified fluvial PM input to the estuary will result in changes of mixing ratios of fluvial and marine PM at a location x in the estuary, and accordingly, contaminant concentrations changes, too. Provided the amount of marine PM within the estuary remains constant, a new mixing ratio (MR_{new}) at the location x can be estimated (Eq. 2). Fluvial and marine contaminant concentrations ($C_{fluvial}$, C_{marine}) are assumed to be constant, too. With known annual PM loads ($PM_{inputref}$) and assumptions on modified annual fluvial PM loads ($PM_{inputnew}$), the mixing model allows to estimate resulting concentrations ($C_{estuarynew}$) at a given location x in the estuary (Eq. 3). For the assessment of the impact of climate change, these estimated concentrations were compared with estuarine reference concentrations.

(b) **Impact of a reduction of fluvial contaminant concentrations**

For a given mixing ratio of marine to fluvial sediments, Eq. 3 also allows to estimate the impact of a change of fluvial contaminant concentrations on contaminant concentrations in the estuary.

$$MR_{ref} = \frac{C_{fluvial} - C_{estuary}}{C_{estuary} - C_{marine}} \tag{1}$$

$$MR_{new} = MR_{ref} \frac{PM_{inputref}}{PM_{inputnew}} \tag{2}$$

Equation 3 derived from Eq. 1

$$C_{estuarynew} = \frac{MR_{new} * c_{marine} + c_{fluvial}}{1 + MR_{new}} \tag{3}$$

MR_{ref}, $MR_{\ new}$: mixing ratios of marine to fluvial PM for the reference period and for a modified fluvial PM input to the estuary;
C_{marine}, $C_{fluvial}$, $C_{estuary}$: concentration of contaminants in marine, fluvial sediments and estuarine sediments at location x (mg/kg; µg/kg);
$\mathrm{PM_{inputref}}$: mean annual fluvial PM input to the estuary (t/a) for the reference period;
$\mathrm{PM_{inputnew}}$: modified fluvial PM input (t/a).

2.4 Boundary Conditions for the Mixing Model

(a) **Impact of climate-induced changes of fluvial PM input**

For the Elbe, the mean annual load of fluvial PM into the estuary over the reference period ($\mathrm{PM_{inputref}}$) was derived from annual data of Hitzacker (Elbe-km 522.9 in the inland reach of the Elbe) and projections of the fluvial PM load ($\mathrm{PM_{inputnew}}$) for the near and far future (2021–2050 and 2071–2100) at Hitzacker were provided by Hillebrand et al. (2014). It had been assumed that the PM does not settle on its way to the estuary and that the PM loads measured in Hitzacker represent the PM input to the Elbe estuary in Geesthacht. To cover a wide range of potential changes, the projected PM loads are based on projections for various river discharge conditions (Nilson et al. 2014) that resulted from different model chains (Fig. 3). All model chains started with the emission scenario A1B that expects moderately increased anthropogenic emissions of greenhouse gases (IPCC 2007).

To test the sensitivity of contaminant concentrations in the Elbe estuary towards climate change, the results of five selected projections of the PM load at Hitzacker were used. They include the strongest projected increase of the PM load and accordingly, a worst-case projection for the increase of contaminant concentrations. Additional assumptions of +100 and −50 % PM input were considered to provide indications of the impact of extreme floods associated with high PM input and of

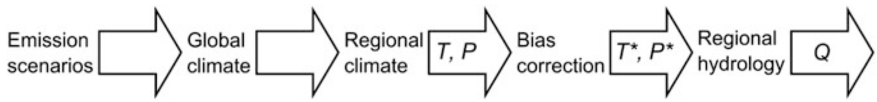


Fig. 3 Scheme of a model chain for the hydrological climate impact research (Lingemann et al. 2013)

periods with extreme low river discharges and small PM input, respectively. From the rivers Weser and Ems, no projections of the PM input to the estuaries exist, so that the same rates of change in PM inputs like in the Elbe estuary were used in the sensitivity studies instead.

For the mixing model, reference concentrations in estuarine PM at given locations ($C_{estuarine}$), in fluvial PM that enters the estuary ($C_{fluvial}$), and in marine PM transported to the upstream direction of the estuary (C_{marine}) were used.

(b) **Impact of a reduction of fluvial contaminant concentrations**

Furthermore, the effect of potential future remediation measures aiming at the reduction of contaminant sources in the inland reaches of the river Elbe on estuarine concentrations was assessed by reducing the fluvial contaminant concentrations by a factor of two. The results should give a first impression on the extent of reduction of contaminant concentrations required to compensate for potential climate-induced increases of contaminant concentrations.

3 Results and Discussion

3.1 Evaluation of Existing Data

3.1.1 Reference Concentrations, Reference Period and Selection of Contaminants

Over the last three decades, monitoring data showed a temporal decrease of contaminant concentrations in SPM and sediments in all three estuaries, with the strongest decrease observed in the Elbe estuary (BfG 2013b, 2014; Heise et al. 2005; FGG Elbe 2013). Basically, this decline can be attributed to restoration and closures of old industries, reduction of contaminant emissions, improved sewage plants and the ban of hazardous substances, like the DDT and PCBs. Particularly, at stations in the inner part of the Elbe estuary, the Mann-Kendall test (Gilbert 1987) revealed significant downward trends for concentrations of some trace metals for time series starting in 1990.

Restricted to the years 2003–2012, significant downward trends were only detected for Hg and Ni at Geesthacht/Elbe, for Hg at Wedel/Elbe, for Pb at Brunsbüttel/Elbe, Hemelingen/Weser and Farge/Weser and for Cd at Bollingerfähr/Ems. Therefore, these trace metals were not considered any further in the selection of indicators for marine and fluvial PM in the estuaries mentioned above. Among the other contaminants under consideration, no or only slightly significant trends were observed.

For several contaminants, concentrations calculated over the period 2003–2012, that was defined as reference period, proved to be much higher at the fluvial monitoring sites close to the tidal weir than those at monitoring sites in the outer estuaries or at sea. Except for Hg at the fluvial station Hemelingen/Weser, the results of the Welch's test (Welch 1947) showed significant differences for all contaminants under consideration. However, for the Elbe estuary, Zn and Cu were rejected as indicators for the marine and fluvial PM, as unexpectedly high concentrations in the longitudinal profile indicated local sources at the monitoring site Wedel in the inner Elbe estuary (km 642). Furthermore, Hg and some of the organochlorine contaminants were not used as indicators, whenever concentrations were close to detection limits or coefficients of variation were high (Table 2). For selected contaminants, reference concentrations and their variability are presented in Table 2, and contaminants that were identified as suitable indicators are emphasized in bold.

3.1.2 Mixing Ratios in the Estuaries

Mixing ratios MR_{ref} of marine and fluvial sediments calculated for the reference period at a given monitoring site in the estuaries are included in Table 2. Generally, the MR_{ref} differed only slightly for the indicator contaminants. For Hg, that showed significantly decreasing concentrations in the Elbe estuary, the MR_{ref} was considerably lower than for other contaminants. For Zn and Cu, the mixing ratios in the Elbe estuary confirmed the interference of the suspected local sources. Mean values of the mixing ratios in the reference period MR_{ref} were calculated for the selected indicator contaminants at each site and they were applied for further assessments.

3.1.3 Projections and Sensitivity Studies

For all three estuaries, concentration changes estimated on the basis of the projections for the fluvial PM input and on the assumptions of +100 and −50 % PM input for selected monitoring sites are presented in Table 3. As projections for the climate-induced changes of PM input vary considerably, also projected changes of contaminant concentrations vary in a wide range.

Elbe Estuary

For the five projected alterations of the annual PM input by −13 to 18 % in the near future, the changes of contaminant concentrations are expected to be similar at both monitoring sites in the inner and the outer Elbe estuary. They vary in the range of −12 to +12 % and are within the natural variability observed in the reference period. For changes of the annual PM load by −31 and +56 % as projected for the far future (projections 1 and 4), more distinct changes in contaminant concentrations in the range of −25 to +34 % are estimated. For the maximum projected change of PM input of +56 %, the estimated changes of contaminant concentrations partially are above the natural variability. Significant changes of contaminant concentrations of −42 and +63 % can only be expected for the assumed extreme changes of PM input by −50 or 100 %, respectively. Both estimates are assumptions of PM inputs that

Table 2 Reference concentration (ref. conc.) with coefficients of variation (CV) of contaminants of the selected monitoring stations (n.d.: not determined; n.a.: not applicable)

			Ref. conc. (mg/kg; µg/kg)[a]	CV (%)	Mixing ratio				Ref. conc. (mg/kg; µg/kg)[a]	CV (%)	Mixing ratio
ELBE	*Geesthacht*	**Cd**	**7.3**	**18**	**n.a.**	Weser	*Hemelingen*	**Cd**	**4.7**	**23**	**n.a.**
		Hg	2.5	30	n.a.			Hg	0.4	23	n.a.
		Zn	1,131	13	n.a.			**Zn**	**854**	**18**	**n.a.**
		Cu	98	27	n.a.			Cu	68	21	n.a.
		p,p′-DDD	**39**	**41**	**n.a.**			**p,p′-DDD**	**1.8**	**30**	**n.a.**
		p,p′-DDE	**14**	**29**	**n.a.**			p,p′-DDE	2.4	34	n.a.
		HCB	**38**	**45**	**n.a.**			HCB	1.8	44	n.a.
	Wedel	**Cd**	**2.1**	**24**	**2.9**		Farge	**Cd**	**3.3**	**18**	**0.5**
		Hg	1.4	22	0.9			Hg	0.4	21	n.d.
		Zn	784	34	0.6			**Zn**	**652**	**13**	**0.4**
		Cu	88	27	0.2			Cu	60	11	0.2
		p,p′-DDD	**11**	**24**	**2.8**			**p,p′-DDD**	**1.2**	**37**	**0.8**
		p,p′-DDE	**3.9**	**26**	**2.9**			p,p′-DDE	1.5	42	n.d.
		HCB	**8.5**	**28**	**3.6**			HCB	1.0	55	0.4
	Brunsbüttel	**Cd**	**1.1**	**27**	**8.2**		Nordenham	**Cd**	**0.9**	**14**	**7.0**
		Hg	1.1	21	1.5			Hg	0.3	30	n.d.
		Zn	366	14	3.6			**Zn**	**298**	**12**	**3.8**
		Cu	47	16	2.2			Cu	35	21	3.0
		p,p′-DDD	**7.4**	**34**	**4.6**			**p,p′-DDD**	**0.9**	**32**	**2.1**
		p,p′-DDE	**2.6**	**31**	**5.5**			p,p′-DDE	0.6	28	n.d.
		HCB	**3.8**	**37**	**9.9**			HCB	0.5	58	20

(continued)

Table 2 (continued)

			Ref. conc. (mg/kg; µg/kg)[a]	CV (%)	Mixing ratio				Ref. conc. (mg/kg; µg/kg)[a]	CV (%)	Mixing ratio
Ems	*Bollingerfähr*	Cd	3.5	13	n.a.	Marine	Inner German Bight	**Cd**	**0.4**	**25**	**n.a.**
		Hg	0.5	28	n.a.			**Hg**	**0.3**	**9**	**n.a.**
		Zn	**845**	**8**	**n.a.**			**Zn**	**153**	**15**	**n.a.**
		Cu	**133**	**15**	**n.a.**			**Cu**	**23**	**23**	**n.a.**
		p,p′-DDD	1.6	77	n.a.			**p,p′-DDD**	**0.4**	**63**	**n.a.**
		p,p′-DDE	**2.1**	**50**	**n.a.**			**p,p′-DDE**	**0.3**	**68**	**n.a.**
		HCB	1.6	76	n.a.			**HCB**	**0.4**	**89**	**n.a.**
	Herbrum	Cd	1.0	36	3.9						
		Hg	0.4	29	n.d.						
		Zn	**241**	**17**	**6.9**						
		Cu	**35**	**30**	**8.1**						
		p,p′-DDD	0.5	38	9.6						
		p,p′-DDE	**0.6**	**26**	**5.7**						
		HCB	1.6	137	0.03						

[a] Trace metals in mg/kg, organic contaminants in µg/kg

Table 3 Projected and assumed changes for the PM input at Hitzacker and resulting changes of contaminant concentrations in the estuaries

	Projection	PM load (%)	Wedel (Elbe)	Brunsbüttel (Elbe)	Farge (Weser)	Nordenham (Weser)	Herbrum (Ems)
			Δ conc. (%)	Δ conc. (%)	Δ conc. (%)	Δ conc. (%)	Δ conc. (%)
Near future	1	17	12	4.7	n.d.	n.d.	n.d.
	2	18	12	12	4.2	0.5	12
	3	−12	−8	−11	n.d.	n.d.	n.d.
	4	−13	−9.2	−12	−3	−9.4	1.1
	5	6	3.8	1.9	n.d.	n.d.	n.d.
Far future	1	56	34	30	9.4	15	22
	2	20	14	11	n.d.	n.d.	n.d.
	3	−12	−8	−11	n.d.	n.d.	n.d.
	4	−31	−23	−25	−9	−21	−5.5
	5	−8	−5.8	−7.1	n.d.	n.d.	n.d.
	Assumed	−50	−40	−42	−17	−27	−12
	Assumed	100	56	63	15	43	36

Δ PM load: projected PM input to the estuary
Δ conc.: resulting change in contaminant concentrations
n.d.: not determined

may occur during extreme events of high or low river discharges. According to projected climate-induced changes of temperature and precipitation, these events may become more frequent (IPCC 2007) in future.

Weser and Ems Estuaries

Generally, contaminant concentrations projected on the basis of the same rates of change in PM input like in the Elbe estuary, are found to be smaller in the estuaries of the Weser and the Ems. They are in the range of natural variability in the near (2021–2050) and the far future (2071–2100). Natural variability is exceeded only at Nordenham/Weser assuming +100 % PM input.

Sea Level Rise

The mixing model did not take into consideration the potential impact of the expected sea level rise on contaminant concentrations. The increase of the sea level will result in an increase of upstream transport of low contaminated PM of marine origin, and accordingly in decreasing contamination by diluting the PM prevailing in the estuaries. Results of a numerical simulation for the development of concentrations of PM and for Cd as representative for contaminants in the Elbe estuary indicated slightly decreasing contaminant concentrations by only a few percent in the estuary (Seiffert et al. 2014). Simulations that assumed a sea level rise of 80 cm were carried out for high and low river discharge conditions (ca. 2,400 and <400 m^3/s). Although a climate-induced sea level rise is expected to mitigate a potential increase of contamination pursuant to a projected increase of PM input, or to enhance a decrease of contaminant concentrations resulting from a projected decrease of PM input, its impact is assumed to be of minor importance.

3.1.4 Potential Consequences for the Management of Dredged Material in North Sea Estuaries

Today, contaminant concentrations in PM in the Elbe estuary are high compared with the contamination of the estuaries of the Weser and Ems. Particularly in the inner Elbe estuary, reference concentrations of some organic contaminants, e.g. HCB and the degradation products of p,p′-DDT (p,p′-DDD and p,p′-DDE) exceed the upper national action levels for the handling of dredged material in coastal waterways (Anonymus 2009) (Fig. 4). Even at the monitoring site Brunsbüttel (Elbe-km 696), that is close to the North Sea, the reference concentration of p,p′-DDD (not shown in Figs. 4 and 5) slightly exceeds the action level. According to the national regulations that implement the OSPAR dredged material guidelines (OSPAR 2014), exceedance of upper action levels triggers a thorough impact assessment prior to the aquatic depositing of dredged material, and depositing might be subject to constraints or not permitted at all.

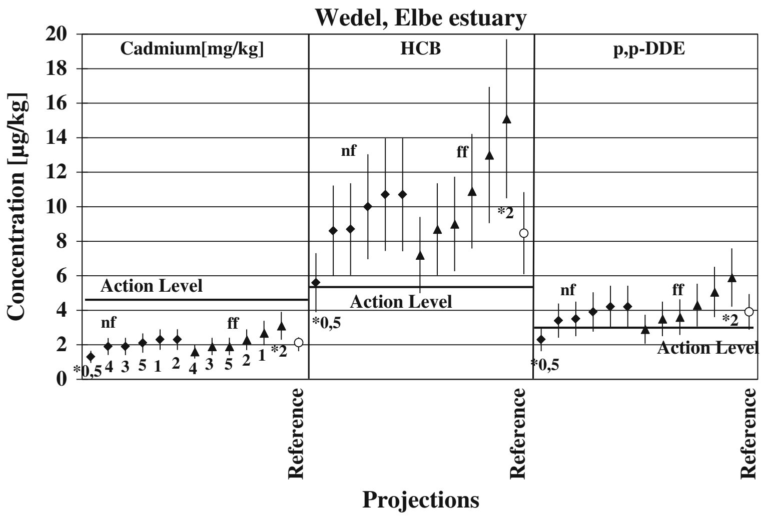


Fig. 4 Contaminant concentrations estimated on the basis of five projected PM loads for the near future (nf; 2021–2050), the far future (ff; 2071–2100) (Table 3), and two assumed PM inputs into the estuary compared with reference concentrations and uncertainties of results; here: monitoring site Wedel in the inner Elbe estuary (km 642)

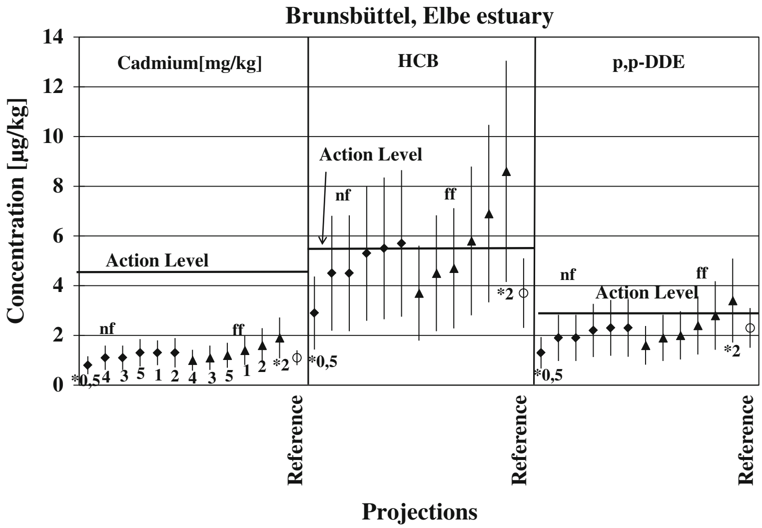


Fig. 5 Contaminant concentrations estimated on the basis of five projected PM loads for the near future (nf; 2021–2050), the far future (ff; 2071–2100) (Table 3), and two assumed PM loads compared with to reference concentrations and uncertainties of results; here: monitoring site Brunsbüttel in the outer Elbe estuary (km 696)

Figures 4 and 5 show the estimated concentrations of selected contaminants at the monitoring sites Wedel and Brunsbüttel in the Elbe estuary for the near and the far future compared with reference concentrations over the period 2003–2012. Additionally, the upper national action levels for the handling of dredged material (Anonymus 2009) are marked.

At both locations, Cd concentrations comply with the upper action level in the reference period as well as in all projections for the near and far future, even with an assumed increase of the PM load by 100 %. For concentrations of p,p′-DDD (not presented in Figs. 4 and 5), p,p′-DDE and HCB at Wedel in the inner Elbe estuary, the projections for the near future indicate that the extent of the predicted exceedance of action levels will not increase significantly. In the far future, however, estimates based on the projected change of the PM input of +56 % and on the assumed increase of PM input by 100 %, indicate that a more distinct exceedance of action levels may occur for these contaminants. The improvement of the contamination estimated for the projected change of PM input by −31 % or for its assumed decrease by 50 % is expected to be small and concentrations of organochlorine contaminants are still in the range of action levels.

At Brunsbüttel in the outer Elbe estuary, concentrations projected for the near future show a clear increase and the projected HCB concentration based on the PM increase by 18 % exceeds the upper action level slightly (Fig. 5). For HCB in the far future, the projected concentration based on the PM increase by 20 and 56 %, exceeds the upper action level as well. Assuming that the PM input increases by

+100 %, also projected concentrations of p,p′-DDD and p,p′-DDE exceed the upper action levels.

In the Weser estuary, particularly the reference concentrations of organic contaminants in PM are generally lower than in the Elbe estuary, and the concentrations of these compounds and of trace metals comply with the upper national action levels for the handling of dredged material (Anonymus 2009). The same applies for the Ems estuary. Also the increase of contaminant concentrations that were estimated on the basis of the PM projections for the Elbe, does not lead to an exceedance of action levels. Even with the extreme assumption of +100 % PM input, expected contaminant concentrations are within the natural variability, and action levels are not exceeded. Results for the Ems estuary are similar (Table 4).

Projected contaminant concentrations vary in a wide range, as the range of PM projections is wide, and results of the projections are associated with large uncertainties. The calculated HCB concentrations do not reflect the slight decrease of the PM input by −8 to −13 %, and instead, a slight increase is projected. Probably, this may be attributed to the large variability of contaminant concentrations and thus the uncertainties in estimating the mean mixing ratio MR.

The Water Framework Directive (EU 2000, 2013) aims at improving the quality of water including sediments. The potential impact of measures to reduce contamination in surface waters was assessed for contaminants exceeding current action levels of the national regulations for the management of dredged material in coastal waters. For this purpose, the mixing model was run with reduced fluvial contaminant concentrations of p,p′-DDE, p,p′-DDD and HCB (Geesthacht) and the impact on concentrations in Wedel was estimated.

Figure 6 shows the calculated contaminant changes based on the five projections for the PM load in the near future for p,p′-DDE. For the projections with the lowest and the highest PM input, in addition the fluvial contaminant concentrations were reduced by 50 %. This reduction resulted in a significant decrease of contaminant concentrations in PM, even in the model run with an increase of PM input. For p,p′-DDE, the estimated concentration was below the action level of 3 µg/kg, and the HCB concentration decreased to meet the action level. This assessment demonstrates that measures to reduce contamination in the freshwater part of the Elbe can achieve a considerable improvement of PM quality in the estuary and a potential climate-induced increase of contaminant concentrations may be mitigated.

Compared with climate-induced changes of PM input to estuaries, anthropogenic activities in the rivers, as for example river engineering measures and remediation measures in the freshwater parts of the rivers, may give rise to more intense effects on sediment transport and quality. For example, construction works including capital dredging in the Elbe estuary in the year 1999/2000 resulted in increasing siltation and a considerable decrease of contaminant concentrations in PM in the inner Elbe estuary (Ackermann and Schubert 2007; BfG 2014). These changes were attributed to an enhanced upstream transport of fine-grained sediments carrying low contaminated PM of marine origin and the decrease of contaminant concentrations

Table 4 Reference concentrations (ref. conc.) of indicator contaminants and the minimum and maximum concentrations estimated for −50 and +100 % PM input to the estuaries compared to the upper action levels (act. lev.) (Anonymus 2009)

			Ref. conc.	Min	Max	Act. lev.				Ref. conc.	Min	Max	Act. lev.
ELBE	Wedel	Cd mg/kg	2.1	1.3	3.1	4.5	Weser	Farge	Cd mg/kg	3.3	2.6	3.9	4.5
		p,p′-DDD µg/kg	11	5.9	16	6.0			Cu mg/kg	60	47	61	90
		p,p′-DDE µg/kg	3.9	2.3	5.9	3.0			Zn mg/kg	652	512	719	900
		HCB µg/kg	8.5	5.6	15	5.5			p,p'-DDD µg/kg	1.2	1.1	1.5	6.0
	Brunsbüttel	Cd mg/kg	1.1	0.8	1.9	4.5		Nordenham	Cd mg/kg	0.9	0.8	2.8	4.5
		p,p′-DDD µg/kg	7.4	3.1	9.3	6.0			Cu mg/kg	35	28	50	90
		p,p′-DDE µg/kg	3.0	1.3	3.5	3.0			Zn mg/kg	298	231	554	900
		HCB µg/kg	3.8	2.9	8.9	5.5			p,p'-DDD µg/kg	0.9	0.6	1.2	6.0
Ems	Herbrum	Zn mg/kg	241	205	323	900							
		Cu mg/kg	35	31	50	90							
		p,p′-DDE µg/kg	0.6	0.5	0.8	3.0							

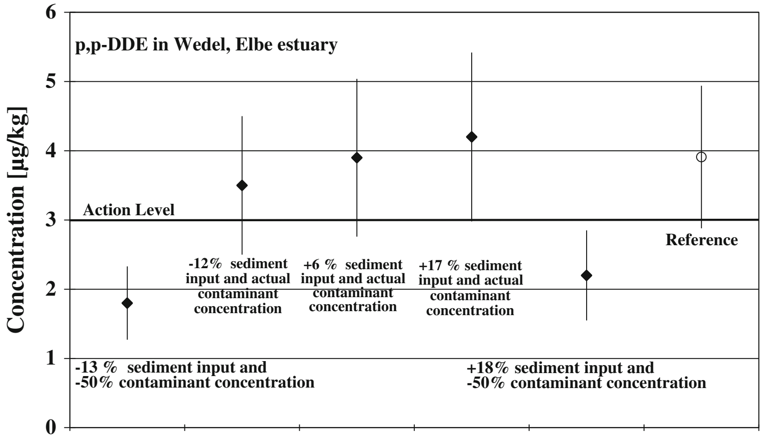


Fig. 6 Calculated p,p′-DDE concentrations with uncertainties in the Elbe estuary on the basis of five projected PM loads for the near future and a reduction of the concentration at the weir Geesthacht by 50 % for the lowest and highest PM input in comparison to a reference, here: monitoring site Wedel

resulted from dilution of contaminated PM with low contaminated PM, while the amount of contaminants within the Elbe estuary did not decrease. Remediation measures, however, may reduce the contamination level, as observed in the past and the amount of contaminants of the estuaries significantly.

4 Summary and Conclusions

This study focuses on the assessment of potential climate-induced changes of particle-bound contaminant concentrations in the North-Sea estuaries of the rivers Elbe, Weser and Ems and on consequences for dredged-material management practices in the navigable waterways there. Dredging activities must take the quality of sediments into account, and pertinent assessment criteria are specified in national regulations for the handling of dredged material in coastal waterways (in Germany: GÜBAK, Anonymus 2009).

The estimation of changes in contaminant concentrations in the Elbe estuary was based on projections of climate-induced changes in inputs of fluvial particulate matter (PM) into the estuary. These projections made a differentiation between the near future (2021–2050) and the far future (2071–2100). From the rivers Weser and Ems no comparable information was available, so that the same rates of changes in PM inputs like in the Elbe estuary were used in the sensitivity studies instead. For particle-bound contaminants that have their main sources in the inland reaches of the rivers, estuarine concentrations predominantly depend on the mixing ratios of marine and fluvial sediments. Therefore, a linear mixing model was applied to derive information on the sensitivity of estuarine contaminant concentrations to changes of fluvial PM inputs. In addition, scenarios of extreme changes in fluvial PM inputs were established to estimate the potential impacts on contaminant concentrations under extremely high or low river discharges. The contribution of a sea-level rise to climate-induced changes in contaminant concentrations is considered to be small compared with the impact of changing PM inputs into estuaries and was not included in this study.

In the Elbe estuary, projected contaminant changes in the near future vary from −12 to +12 %, and thus remain within the natural variability (reference period 2003–2012). However, regarding the far future, two of the five PM projections indicate distinct impacts on sediment quality. The projected rates of change in contaminant concentrations of −25 and +34 %, respectively, slightly exceed the natural variability of the reference contamination. For the inner Elbe estuary (Wedel), where already the reference concentrations of HCB, p,p′-DDD and p,p′-DDE are above the upper action levels of the national regulations, the results of the worst-case projection for the potential climate-induced change of contamination (+34 %) indicate a considerably higher exceedance of the action-triggering levels in the far future. Even in the outer Elbe estuary (Brunsbüttel), where reference concentrations of the above-mentioned organochlorine compounds often comply with the assessment criteria, contaminant concentrations derived from the worst-case projection may slightly exceed the upper action levels in the far future. However, significant deviations of the concentrations from those of the reference period are to be expected only for the assumed changes of PM inputs by −50, and +100 %, respectively.

Reference concentrations of contaminants in the Weser and Ems estuaries do not exceed the upper national action levels for dredged-material management. For these estuaries, concentrations are expected in the near and in the far future to remain within the natural variability of the references, and no future consequences of climate change are expected for sediment-management practices with regard to contamination.

Against the worst-case projection of a potential increase of particle-bound contaminant concentrations in the Elbe estuary in the far future, the study recommends to consider adaptations in sediment management strategies to changed concentration levels in the medium or the long terms. On the one hand, e.g. the practices of depositing dredged- material within the water system might be adapted (BfG 2014). On the other hand, the implementation of remediation measures like those planned under the WFD could mitigate the climate-induced increase of contaminants. However, before the planning of adaption measures begins, the respectively prevailing contamination status should be verified, as climate-induced changes of contaminant concentrations might be superimposed by direct anthropogenic activities, e.g. remediation measures to reduce contamination or construction works in waterways. Particularly, remediation measures may improve sediment quality, and thus mitigate a potential climate-induced increase of contamination levels. In the late 1980s industrial closures and remediation measures in the Elbe river basin led to significant decreases in the concentrations of several contaminants (e.g. Heise et al. 2005; BfG 2014). However, for a number of years now, this downward trend of contamination in estuaries has lessened. In order to assess the effectiveness of potential future remediation measures, the mixing model was run for the inner Elbe estuary with fluvial contaminant concentrations reduced by 50 % and two projected changes in PM. The model estimated a decline of contamination by almost 50 %.

Thus, it can be concluded that the remediation of contaminant sources in the inland reaches of the rivers is still an appropriate measure to improve sediment quality within estuaries and to mitigate the impacts of climate-induced increases of fluvial PM inputs on estuarine contamination levels.

As the bandwidth of the projected changes of PM inputs to the estuaries is wide, and the model uncertainties are large, the results of the five projections on changing contaminant concentrations also vary widely. Further statistical evaluation of the data, including multivariate statistics and modelling of missing data, was initiated to improve the reliability of the model outputs and to deal with local sources in estuaries. However, although results are associated with large uncertainties, the approach presented allows estimating the range of changes in particle-bound contamination levels in North Sea estuaries as a consequence of changing boundary conditions.

Acknowledgments Investigations were carried out within the scope of the project "Impact of climate change on the transport behaviour of contaminated sediments and the maintenance of coastal waterways" that is part of the German research programme KLIWAS "Impacts of climate change on waterways and navigation", initiated by the German Federal Ministry of Transport and Digital Infrastructure (BMVI) (www.kliwas.de). The coastal Federal Waterways and Shipping Authorities kindly collect samples for the monitoring programme since more than 30 years.

References

Ackermann F, Bergmann H, Schleichert U (1983) Monitoring of heavy metals in coastal and estuarine sediments—a question of grain size: <20 µm versus <60 µm. Environ Technol Lett 4:317–328

Ackermann F (1998) Dynamik der Schwermetallbelastung in feinkörnigen Sedimenten und Schwebstoffen im Tidebereich von Ems, Weser und Elbe. Bundesanstalt für Gewässerkunde, Koblenz BfG-1188 (only available in German)

Ackermann F (2004) Schwermetalle als Tracer für den Feinpartikeltransport in den Nordseeästuaren. 13. Chemisches Kolloquium – Auswirkungen des Ausbaus deutscher Nordsee-Ästuare auf die Gewässergüte und das Baggergutmanagement – 22.06.2004 in Koblenz. Veranstaltungen 2/2004 (only available in German)

Ackermann F, Schubert B (2007) Trace metals as indicators for the dynamics of (suspended) particulate matter in the tidal reach of the River Elbe. In: Förstner U, Westrich B (eds) Sediment dynamics and pollutant mobility in rivers. Springer, Berlin, pp 296–304

Anonymus (2009) GÜBAK—joint transitional arrangements for the handling of dredged material in german federal coastal waterways between the federal authorities and the federal states (Bundesländer), Koblenz 2009, p 39

ARGE Elbe (Arbeitsgemeinschaft für die Reinhaltung der Elbe) (1980) Schwermetalldaten der Elbe. Bericht über die Ergebnisse der Schwermetalluntersuchungen im Elbabschnitt von Schnackenburg bis zur Nordsee 1979/1980

Banat K, Förstner U, Müller G (1972) Schwermetalle in Sedimenten von Donau, Rhein, Ems, Weser und Elbe im Bereich der Bundesrepublik Deutschland. Die Naturwiss 59:525–528

Brügmann L (1995) Metals in sediments and suspended matter of the river Elbe. Sci Total Environ 159:53–65

BfG—Bundesanstalt für Gewässerkunde (2005) Abschätzung der ökologischen Auswirkun-gen der Verbringung von Baggergut aus der Hamburger Delegationsstrecke der Elbe auf die Umlagerungsstelle Tonne E3 nordwestlich von Scharhörn. Koblenz, BfG-Bericht 1472

BfG—Bundesanstalt für Gewässerkunde (2013a) Überprüfung der ökologischen Auswirkungen der Verbringung von Baggergut aus der Hamburger Delegationsstrecke der Elbe auf die Verbringstelle Tonne E3 nordwestlich von Scharhörn. Abschlussbericht 2011, Koblenz, BfG-Bericht 1775

BfG—Bundesanstalt für Gewässerkunde (2013b) Sedimentmanagementkonzept Tideweser. Untersuchung im Auftrag der Wasser- und Schifffahrtsämter Bremen und Bremerhaven. Bundesanstalt für Gewässerkunde, Koblenz, BfG-Bericht 1794

BfG—Bundesanstalt für Gewässerkunde (2014) Sedimentmanagement Tideelbe - Strategien und Potenziale - Systemstudie II. Ökologische Auswirkungen der Unterbringung von Feinmaterial. Band 1, Endbericht. Im Auftrag des Wasser- und Schifffahrtsamtes Hamburg. Bundesanstalt für Gewässerkunde, Koblenz, BfG-Bericht 1763

BGBl—Bundesgesetzblatt (2011) Decree for the protection of surface waters (Verordnung zum Schutz der Oberflächengewässer). BGBl Teil I, Nr. 37, 25.7.2011 Bundesanzeigerverlag: 1429–1470

EU (2000) EU Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000, establishing a framework for Community action in the field of water policy (Water Framework Directive)

EU (2008) Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive)

EU (2013) Directive 2013/39/EU of the European Parliament and of the Council of 12 August 2013 amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy (Water Framework Directive)

FGG Elbe (2013) http://www.elbe-datenportal.de/FisFggElbe/content/start/BesucherUnbekannt.action

Förstner U, Schoer J, Knauth H-D (1990) Metal pollution in the tidal Elbe River. In: Allen RJ et al (eds) The science of the total environment, Elsevier, New York, pp 347–368

Gilbert RO (1987) Statistical methods for environmental pollution monitoring. Van Nostrand Reinhold, New York

Graydon JA, Emmerton CA, Lesack LFW, Kelly EN (2009) Mercury in the Mackenzie river delta and estuary: concentrations and fluxes during open-water conditions. Sci Total Environ 407:2980–2988

Hatje V, Barros F (2012) Overview of the 20th century impact of trace metal contamination in the estuaries of Todosos Santos Bay: past, present and future scenarios. Mar Pollut Bull 64:2603–2614

Heise S, Claus E, Heininger P, Krämer T, Krüger F, Schwartz R, Förstner U (2005) Studie zur Schadstoffbelastung der Sedimente im Elbeeinzugsgebiet – Ursachen und Trends. Study for the Hamburg Port Authority

Hillebrand G, Pohlert T, Breitung V (2014) Klimaprojektionen für den Sedimenthaushalt und Risiken durch kohäsive Sedimente. Schlussbericht KLIWAS. Projekt 5.01. KLIWAS48/2014. doi: 10.5675/Kliwas_48/2014_5.01

Horowitz AJ (1991) A primer on sediment element chemistry. Michigan7Lewis Publishers

IPCC (2007) Climate Change 2007. The physical science basis. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Contribution of working group

i to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, 996 p
Irabien MJ, Cearreta A, Leorri E, Gómez J, Viguri J (2008) A 130 year record of pollution in the Suances estuary (southern Bay of Biscay): implications for environmental management. Mar Pollut Bull 56:1719–1727
Kersten M, Smedes F (2002) Normalization procedures for sediment contaminants in spatial and temporal trend monitoring. J Environ Monit 4:109–115
Knauth H-D, Gandraß J, Sturm R (1993) Vorkommen und Verhalten organischer und anorganischer Mikroverunreinigungen in der mittleren und unteren Elbe. Forschungsbericht des Bundesministeriums für Umwelt, Naturschutz und Reaktorsicherheit, Forschungsbericht 10204363, Erich Schmidt Verlag
Kowalewska G, Belzunce-Segarra MJ, Schubert B, Heininger P, Heise S (2011) The role of sediments in coastal monitoring. In: Quevauviller, P, Roose P, Verreet G (eds) Chemical marine monitoring—policy framework and analytical trends, Chap. 12. Wiley, London, pp 377–395
Lingemann I, Nilson E, Carambia M, Krahe P (2013) Änderungen des Wasserhaushalts der Elbe im 21. Jahrhundert. Die Zukunft des Wasserhaushaltes im Elbeeinzugsge-biet, Kolloquium 29/30.11.2012 Dresden. BfG-Veranstaltungen 6/2013
Martinez-Carreras N, Gallart F, Iffly JF, Pfister L, Walling DE, Krein A (2008) Uncertainty assessment in suspended sediment fingerprinting based on tracer mixing models: a case study from Luxembourg. Sediment dynamics in changing environments, vol 325 (Proceedings of a symposium held in Christchurch, New Zealand), pp 94–104
Müller G, Förstner U (1975) Heavy metals in sediments of the Rhine and Elbe estuaries: mobilization or mixing effect? Environ Geol 1:33–39
Nilson E, Krahe P, Lingemann I, Horsten T, KleinB, Carambia M, Larina M (2014) Auswirkungen des Klimawandels auf das Abflussgeschehen und die Binnenschifffahrt in Deutschland. Schlussbericht KLIWAS-Projekt 4.01. KLIWAS-43/2014. doi: 10.5675/Kliwas_43/2014_4.01
OSPAR (2002) JAMP guidelines for contaminants in sediments. Technical Annex 5—normalisation of contaminant concentrations; Ref no. 2002-16
OSPAR (2014) OSPAR guidelines for the management of dredged material at sea, Agreement 2014-06. OSPAR Commission, London
Prange A (1997) Erfassung und Beurteilung der Belastung der Elbe mit Schadstoffen. Teilprojekt 2: Schwermetalle – Schwermetallspezies. Zusammenfassende Aus- und Bewertung der Längsprofiluntersuchungen in der Elbe. BMBF-Forschungsvorhaben: 02-WT 9355/4, GKSS-Forschungszentrum Geesthacht GmbH, Geesthacht
Seiffert R, Hesser F, Büscher A, Fricke B, Holzwarth I, Rudolph E, Sehili A, Seiß G, Winkel N (2014) Betroffenheit wasserbaulicher Anlagen der Nordseeküste und der Ästuare durch Klimaänderungen, und Anpassungsoptionen für Wasserstraßen und Häfen an der deutschen Küste sowie für den Küstenschutz bei Extremereignissen
Walling DE (2005) Tracing suspended sediment sources in catchments and river systems. Sci Total Environ 344:159–184
Welch BL (1947) The generalization of “Student’s” problem when several different population variances are involved. Biometrika 34(1–2):28–35. doi:10.1093/biomet/34.1-2.28 (MR 19277)

Part IV
Sediment Monitoring

Application of a New Monitoring Strategy and Analysis Concept of Suspended Sediments in Austrian Rivers

Petra Lalk, Marlene Haimann and Helmut Habersack

Abstract In recent years the hydrological service in Austria developed and implemented a new, continuous, long term suspended sediment monitoring system taking the spatial and temporal variability of the suspended sediment transport process into consideration. The new monitoring strategy and analysis concept during the data processing was applied at 28 measurement sites for the period from 2009 to 2011. Furthermore a lot of investigation was done to verify and validate the suspended sediment data in preparation for the annual publication. The results of analysis of the suspended sediment concentrations, transport rates, loads and yields in Austrian Rivers confirm the well done verification and validation and allow conclusions about the respective suspended sediment transport processes.

1 Introduction

Former measurements of suspended sediment transport in Austria were mostly carried out for special-purpose and have been time limited. Along the Danube and in selected measurement sites in Upper Austria measurements of suspended sediment transport have been undertaken for several decades. These involve the collection of single water samples once a day and during flood events up to four times a day (Prazan 1994; Nachtnebel et al. 1998). As in Austrian rivers the highest amount of suspended sediments are transported during short-lived flood events, this approach

P. Lalk (✉)
Division IV/4 Water Balance, Federal Ministry of Agriculture, Forestry, Environment and Water Management, Marxergasse 2, 1030 Vienna, Austria
e-mail: petra.lalk@bmlfuw.gv.at

M. Haimann · H. Habersack
Christian Doppler Laboratory for Advanced Methods in River Monitoring, Modelling and Engineering, Institute of Water Management, Hydrology and Hydraulic Engineering, Department of Water – Atmosphere – Environment, BOKU—University of Natural Resources and Life Sciences, Vienna, Vienna, Austria

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_10

does not cope with the demand for accuracy of the suspended sediment monitoring required to meet the stringent needs of modern water management planning.

Actual problems from a river engineering and ecological point of view concern the imbalanced sediment regime and discontinuity in rivers caused e.g. by disruptions in flow, which include weirs and the reservoir of hydro power plants (Habersack et al. 2013). In order to determine and optimize measures to counteract reservoir sedimentation numerical or physical models are used, which requires certain input data in an adequate quality and accuracy. Further applications of numerical models involve analysing problems of water management, river engineering or different aspects of water quality and the evaluation of river restoration projects.

Thus the hydrographic service in Austria developed and implemented a new, continuous, long term suspended sediment monitoring system. The consideration of the temporal variability of the suspended sediment transport process is achieved by continuous recording to be able to determine also individual events as well as possible. Additionally the spatial variability in a cross section is taken into account.

With this new monitoring strategy the demand was fulfilled to provide comparable basic data evaluated by a standardized method with a sufficient accuracy of the suspended sediment concentration, the suspended sediment transport rate and load. The basic data determined by the conversion of the new monitoring concept are made available and are published since 2008 in the Hydrological Yearbook of Austria and in the internet http://ehyd.gv.at.

2 Monitoring Network

With the amendment to the water law 2003 the legal requirement to measure sediment loads, prescribed in the European Water Framework Directive, was new fulfilled. Details of important regulation according to the measurements of suspended sediment transport and the scale of the governmental basis network, which covers 34 measurement sites, were defined in the hydrological cycle survey by law 2006. The monitoring network in Austria aims to determine the suspended sediment input and output at the Austrian borders, the sediment input in great lakes as well as the erosion rates and sediment yields associated with selected catchments. Furthermore data are provided to assist in analysing problems and needs of water management, river engineering or different aspects of water quality and to evaluate river restoration projects (BMLFUW 2008). The focus of the monitoring program lies on the determination of the long term variability of the sediment budget at selected cross sections for evaluating trends and long-term changes of suspended sediment supply due to e.g. reservoir construction, land use change or land disturbances, including for instance logging or mining (e.g. Walling and Fang 2003).

In the hydrological Yearbook of Austria 2008 suspended sediment data of 20 measurement sites were published for the first time. Since 2009 the number of stations was further increased up to 28 measurement sites (2 in the Rhine area, 26 in

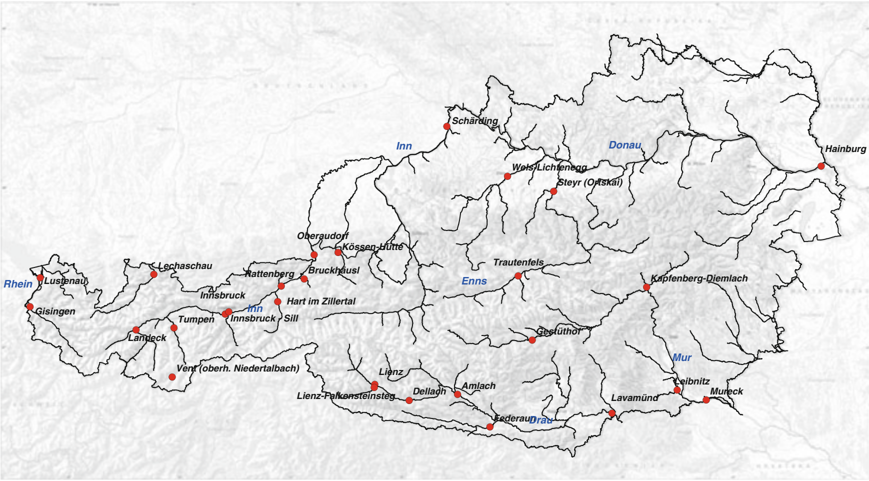


Fig. 1 Monitoring network of suspended sediment data since 2009

Danube area), which are shown in Fig. 1. As these 28 existing measurement sites are situated in different representative catchment areas of Austria the data basis from these stations is used for further investigation. The areas of the monitored catchments range from less than 100 up to more than 100,000 km^2. The situation of the measurement sites varies from unaffected small catchments in high altitudes to affected large catchments by hydro power plants in lower regions.

3 Methods and Monitoring Strategy

To detect the suspended sediment concentration a wide range of technology is available (Wren et al. 2000; Gray and Gartner 2009). All these methods have their advantages and fields of application but are not able to detect the complexity of the suspended sediment transport completely. For this reason the monitoring strategy applied at the Austrian rivers (Table 1) consists of a combination of direct and indirect monitoring methods (BMLFUW 2008; Haimann et al. 2014).

Investigations revealed that the suspended sediment concentration corresponds much better with turbidity than water level or discharge (Lewis 1996). Hence, each measurement site is equipped with an optical turbidity sensor that continuously measures the scatter of the emitted signal and provides information about the turbidity. As the detected signal is influenced by the size, composition and shape of the particles in the river (Gippel 1995; Shoellhamer and Wright 2003) the sensor data have to be calibrated in situ using water samples, which are taken in vicinity of the sensor. The sampling frequency is dependent on water level or suspended

Table 1 Suspended sediment monitoring strategy employed at Austrian Rivers (BMLFUW 2008)

Parameter	Method	Frequency
Turbidity	Turbidity sensor	Continuously
Concentration of suspended sediments close to the sensor	Single point samples close to the sensor	High sediment discharge: at least daily mean sediment discharge: at least 1–2/week low sediment discharge: at rare intervals
Distribution of the concentration of suspended sediments in a cross section	Multi-point sampling, depth-integration method, ADCP + samples	2–4 times/year at different discharges

sediment concentration and varies from once or twice a week at mean conditions up to several times per day during flood events. During low flow conditions the sampling frequency can be further reduced (BMLFUW 2008).

To calibrate the turbidity data two different methods can be applied, which can also be used in combination. The first method calculates a factor (probe factor: $k_s = s_s/s_k$) for each time where concentrations of water samples (s_k) and turbidity data (s_s) are available. By linear interpolation between these time steps the correction factor is calculated for each turbidity value. The second method uses a linear regression between turbidity data and concentrations of water samples as depicted by Gippel (1995), Lewis (1996), Wass and Leeks (1999). Applying these correction methods the turbidity data are converted into a record of suspended sediment concentration close to the sensor.

Additionally the spatial variability of the suspended sediment concentration in a cross section has to be considered. Two to four times a year multi-point measurements (ISO 4363 2002; Edwards and Glysson 1999) are performed at each measurement site. For that matter samples are taken in at least five verticals and different depths (three to five depending on water depth). The sampling is performed using a US-P61-A1 or US-P63 sampler as these samplers can be electronically controlled to open the valve at the correct depth and for the defined time (Edwards and Glysson 1999; Davis 2005). As this method is tedious and very time consuming and ADCP devices are already applied to determine flow velocity and discharge, the ADCP measurements were also analysed regarding suspended sediments. ADCP devices use the Doppler shift to determine the flow velocity from the difference of frequency between emitted and received signal. As the signal is reflected by the particles in the water, the received signal contains information about the suspended sediment in the water. The concentration in the river can be calculated from the backscatter using the sonar equation (e.g. Deines 1999) when water samples are taken simultaneously.

The velocity weighted mean suspended sediment concentrations gained from the cross-sectional measurements (multi-point measurements or ADCP-measurements combined with concentrations of water samples) are related to the suspended sediment concentrations close to the sensor determined at the same time. By multiplying this record with the discharge the suspended sediment transport is calculated. By integrating the transport over time the suspended sediment load is determined and loads for different time intervals (events, months, years) can be provided.

4 Quality Management of Suspended Sediment Data Processing

The first step in the suspended sediment data processing is to identify anomalous values of turbidity data and concentrations of water samples. Reasons for no or anomalous turbidity data could be the result of a series of malfunctioning situations, including that the turbidity sensor is out of range, a biofilm is growing on the lenses of the sensor (e.g. López-Tarazón et al. 2009), covering with bedload or lowering of the water level below the sensor optic, etc. Implausible suspended sediment concentrations of the water samples can result from imprecise and inaccurate sampling or laboratory analysis up to mix-ups by labelling or handling the water bottles. The erroneous values are deleted and if possible corrected under consideration of reliable calibration samples, other turbidity sensors from close measurement sites at the same river or if needs must water level from the same station. For the evaluation of individual measurement sites where no nearby data records are available or downstream of natural river basins or hydro power plants, increased requirements for the safe recording of data are provided. It is recommended to equip these measurement sites with respect to turbidity sensors redundant as this can also help to avoid/fill data gaps.

Water samples close to the sensor (calibration samples) are necessary to calibrate the turbidity data. During flood events, thunderstorms or mudslides the material often have different characteristics whereby the relation between turbidity and suspended sediment concentration can change. The sampling at these events is of special interest, even more as single events can contribute high percentages to annual and monthly loads (maximum daily loads contribute 37 % as an average—from 2009 until 2011 and 28 measurement sites—to the monthly loads and up to 50 %—maximum value from 2009 until 2011 measured at Lechaschau/Lech—to the annual loads).

Most suspended sediment transport in Austria occurs during short-lived flood events that are frequently inadequately sampled by manual instantaneous sampling. Automatic samplers, where the sampling interval cannot only be regulated by time but also by turbidity or water level, are often applied to calibrate turbidity data (e.g. Oeurng et al. 2010).

For each station the choice has to be made whether the near sensor concentration is calculated more accurately by the probe factor or by linear regression between turbidity data and the calibration samples. If a high scatter in the relation between turbidity data and calibration samples is observed due to different sources in the catchments or event types (e.g. snow melt, thunderstorms) and the samples were taken in an adequate interval the calculation of the near sensor suspended sediment concentration is performed using the probe factor. The advantage of the probe factor is the adaption of the hydrograph on the calibration samples taking into consideration the changing grain size distribution from transported material of different sub catchments. Hence taking samples of high suspended sediment concentrations is necessary. The disadvantage of the application of the probe factor is the transfer of errors from the calibration samples to the hydrograph. At homogeneous catchments and/or high correlation between turbidity data and concentrations of water samples the conversion is carried out using a linear regression. The linear regression can also be applied with a less number of calibration samples. In any case implausible samples shall not be taken into account.

A correlation between turbidity data and suspended sediment concentration close to the sensor established for one year may not be valid in the next year as the example of the measurement site Lienz/Isel demonstrates (Fig. 2). Amongst other release of high amounts of material can change the correlation between turbidity data and suspended sediment concentration close to the sensor. This example highlights the necessity to continue sampling.

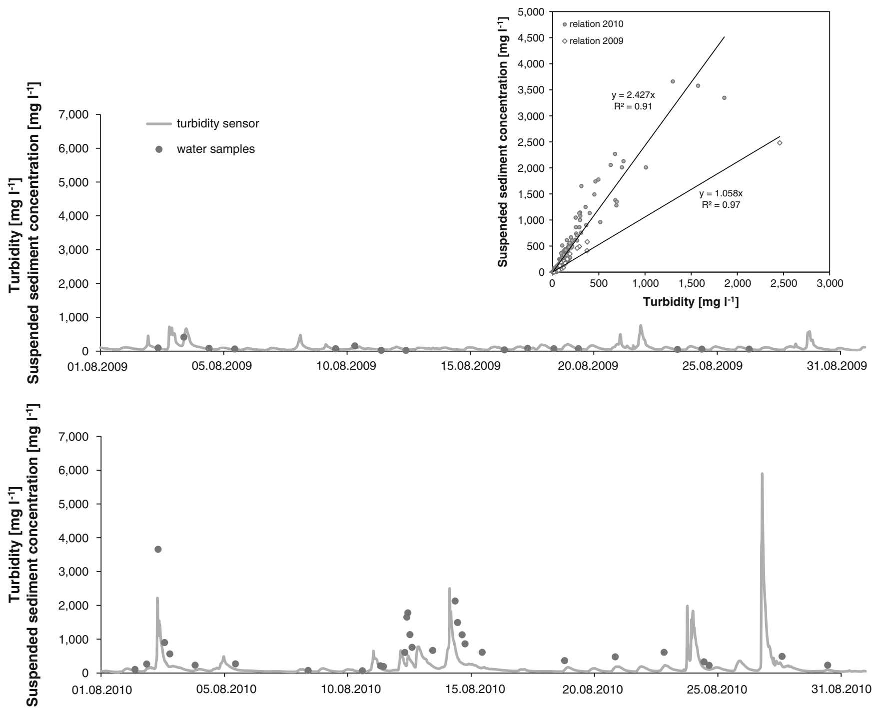


Fig. 2 Turbidity data and calibration samples at the measurement site Lienz at the Isel River August 2009 and 2010 inlet: relation between turbidity data and calibration samples for the years 2009 and 2010

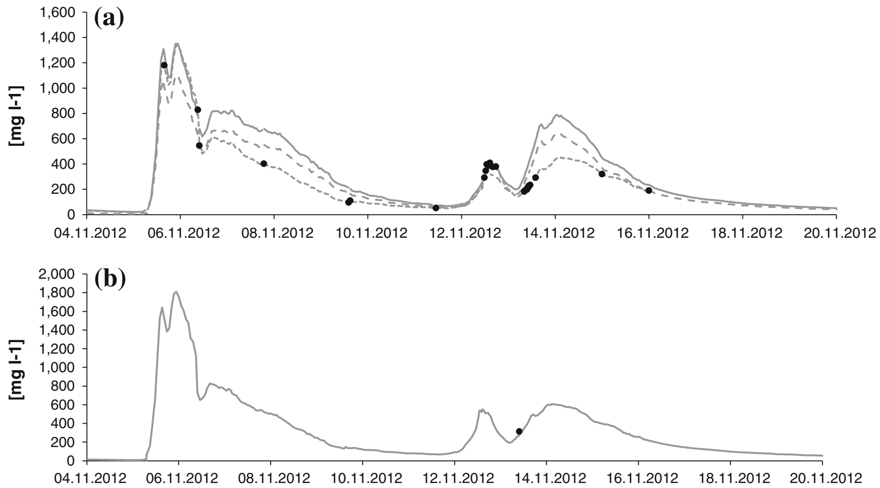


Fig. 3 **a** Comparison between probe factor (*dashed hydrograph*) and linear regression (*dotted hydrograph*) valued at the, **b** measured mean suspended sediment concentration in the cross section at Lavamünd/Drava 2012

Additionally the measured mean suspended sediment concentration in the cross section can be taken into consideration for the validation of the near sensor concentration, as demonstrated in the following example. The continuous mean suspended sediment concentration during the flood wave from 5 to 15. November 2012 in Lavamünd/Drava is evaluated more accurately with the probe factor (Fig. 3a), valued at the measured mean suspended sediment concentration in the cross section (Fig. 3b).

Although all measurement sites in Austria are equipped with the same type of turbidity sensor the relations between turbidity data and the calibration samples valid for 2011 are not similar but increase with increasing altitude (Fig. 4a, b). This seems plausible due to the fact that in higher altitudes coarser material will be removed and is a further evidence of a suitable evaluation of correlation between turbidity data and suspended sediment concentration.

At the Austrian measurement sites the two methods, probe factor or linear regression between turbidity data and the calibration samples, are used in equal parts. An application of both methods at the measurement sites indicate that at some sites the deviation in sediment concentration are marginal but can reach up to ±50 % from the average of both methods in terms of maximum concentrations and up to ±20 % from the average of both methods in terms of mean concentrations and loads (Fig. 5). Furthermore the instant of time when the maximum concentrations or transport rates occur can be shifted between the two methods.

Investigations at Austrian measurement sites but also from other rivers (e.g. Spreafico et al. 2005; Porterfield 1972) showed that it is important to detect the distribution in the cross-section additionally. Generally a linear regression between suspended sediment concentration close to the sensor and mean suspended

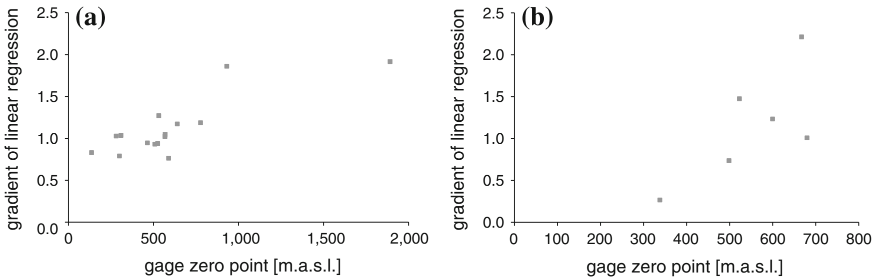


Fig. 4 Gradient of linear regression between turbidity data and suspended sediment concentration close to the sensor for the year 2011 versus gauge zero point in the catchments **a** Danube and **b** Drava

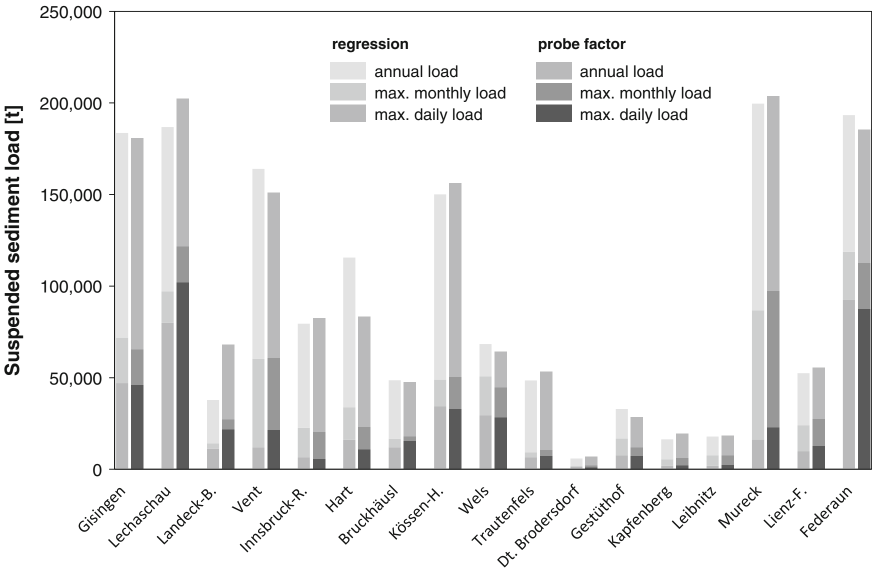


Fig. 5 Suspended sediment loads at Austrian measurement sites for the year 2011 calculated based on linear regression and probe factor

sediment concentration in the cross-section can be applied (Fig. 6a). At the Austrian measurement sites the gradient of the straight line varies between 0.8 (i.e. the mean concentration in the cross-section is lower than the concentration close to the sensor) and 2.1 (i.e. the mean suspended sediment concentration in the cross-section is more than the double of the concentration close to the sensor). The results show that not at all measurement sites a single straight line can be applied. For example at the measurement site Hainburg (Straßenbrücke)/Danube some of the multi-point measurements indicate that the relation between mean suspended sediment concentration in the cross-section and suspended sediment close to the sensor

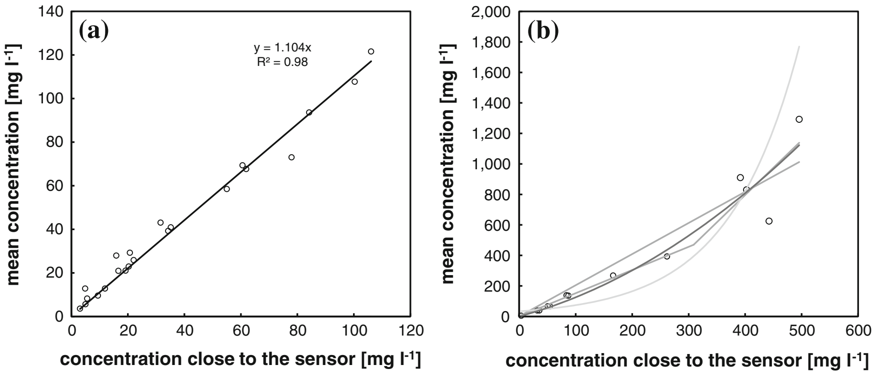


Fig. 6 Correlation between mean suspended sediment concentration and concentration close to the sensor **a** at the measurement site Mureck/Mur, **b** at the measurement site Hainburg (Straßenbrücke) /Danube

increases with high discharges (Fig. 6b). The application of different relation curves (potential, exponential, bilinear) was tested and differences in annual loads between these methods of about ±43 % of the average load of all methods were calculated. In this case the application of a bilinear relation showed the most reasonable results (Haimann et al. 2012). Reasons for changes of relations between mean concentrations and concentrations close to the sensor might be due instantaneous input of material e.g. remobilization of sediments in reservoirs of hydro power plants or mudslides.

After evaluating the suspended sediment data of each measurement site individually all the information available has to be assembled in preparation for the annual publication. Therefore the hydrograph curves within one catchment area are compared, the peaks of the continuous suspended sediment concentration are verified from the event catalogue or hydrological characteristic (BMLFUW 2013) in size and time and the causes of events (thunderstorms, fronts, mudslides, construction works) are analysed and described in the footnotes of the Hydrological Yearbook. Conclusions about the respective processes can be drawn. Wherever applicable, annual loads and the biggest event loads are accounted for all peaks to verify and validate.

The mass balance of annual suspended sediment loads and event loads can be used to verify the accurate application and combination respectively of all the correction factors on the results of load calculations. Where several measurement sites exist at one river system the mass balance of suspended sediment loads can be calculated within the catchments. In this process the annual loads can be compared on the one hand but even event loads can be examined on the other hand. In case of event loads it is important to consider the event type. Heavy rainfalls or thunderstorms limited to smaller sub catchments may lead to high loads in this area but have a decreasing influence on the suspended sediment loads at the downstream measurement sites whereas rainfalls in the whole catchment may add up the

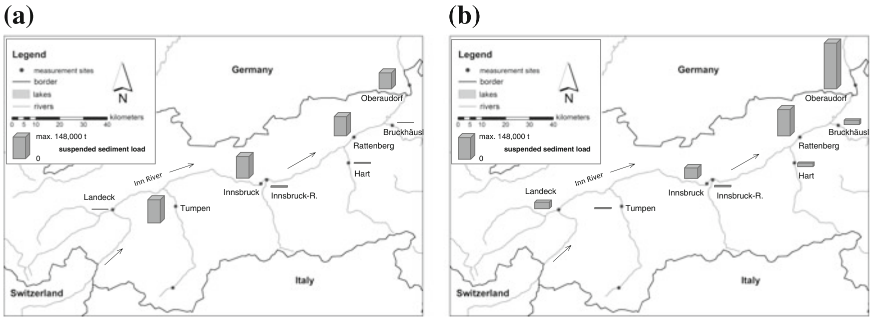


Fig. 7 Suspended sediment loads in the catchment of the Inn River **a** from 04.09.2011 until 07.09.2011, **b** from 10.10.2011 until 13.10.2011

suspended sediment loads with increasing catchment size as described in the following example. On 4th of September 2011 there was a local event in the valley Ötztal. As shown in Fig. 7a there are high daily suspended sediment loads in Tumpen/Ötztaler Ache, which is a tributary of the Inn and are still observed in Innsbruck/Inn. Downstream of Innsbruck/Inn the daily suspended sediment loads decrease up to Rattenberg/Inn and onwards to Oberaudorf/Inn. From 10th to 11th of October 2011 heavy precipitation along the north side of the Alps lead to floods along many rivers in the province Vorarlberg and the northern part of Tyrol (mainly Ill, Lech and in the Inn catchment). As shown in Fig. 7b all tributaries of the Inn (except the Ötztal) add to the amount of daily suspended sediment loads from 10th to 11th of October 2011, hence the highest amount can be found at the most downstream measurement site Oberaudorf/Inn.

5 Analysis of the Suspended Sediment Data in Austrian Rivers

As the time series for the selected 28 measurement sites only exist for 3 years, analysis about long term developments or climate change are not possible, but processes of suspended sediment mobilization and transport can be identified due to their impacts as thunderstorms (particularly heavy rainfall, debris flow), floods, snow melt, power plants or construction works.

At first the mean annual suspended sediment loads and yields from 2009 to 2011 are correlated with the main areal parameters as catchment size, altitude of catchment area, specified as gage zero point [meter above sea level (Adriatic sea)] and percentage of glaciation.

Generally the annual suspended sediment load increases with the size of catchment area (Fig. 8). However, the mean annual suspended sediment loads at measurement sites downstream of lakes or hydro power plants are low in comparison with the unaffected sites due to suspended sediment deposition. The annual

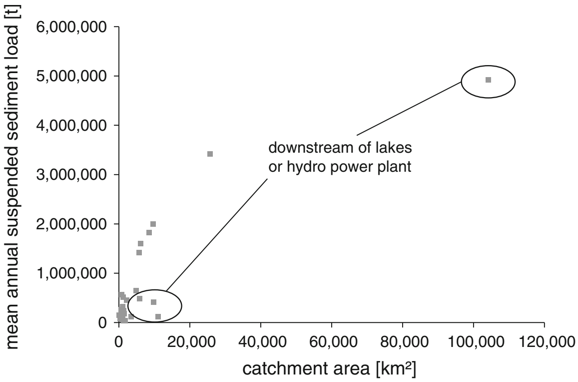


Fig. 8 The mean annual suspended sediment load 2009–2011 versus catchment area

suspended sediment yield increases with the altitude of the catchment (Fig. 9a) and/or with the percentage of glaciation (Fig. 9b), which is significant over ca. 5 % of glaciation, although low values of mean annual sediment yield shows a high scatter.

In Austria the highest mean annual sediment concentration (ca. 250 mg l^{-1} in Vent/Rofenache) come from the western alpine regions, particularly from the glacial zones. In these regions material can be removed by glacial melting or precipitation. With the decreasing mean annual precipitation (Kling et al. 2005) the mean annual sediment concentration decreases from West-to East-Austria. The lowest mean annual sediment concentrations can be found at measurement sites in the alpine upland in the northern part of Austria (e.g. Wels-Lichtenegg/Traun) and in the south-east part of Austria as well as downstream of lakes or hydro power plants (e.g. Lavamünd/Drava). With the increasing discharge and catchment area the mean annual sediment concentration decreases and the mean annual transport rates increases correspondingly. In Austria the highest mean annual transport rate with 156 kg s^{-1} and the highest annual sediment load with 4.9 Mio t a^{-1} respectively can be found in Hainburg (Straßenbrücke)/Danube (Fig. 10).

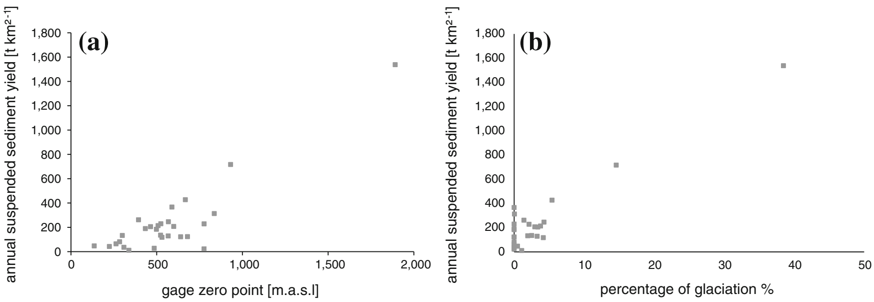


Fig. 9 The mean annual suspended sediment yield 2009–2011 versus **a** gage zero point and **b** percentage of glaciation

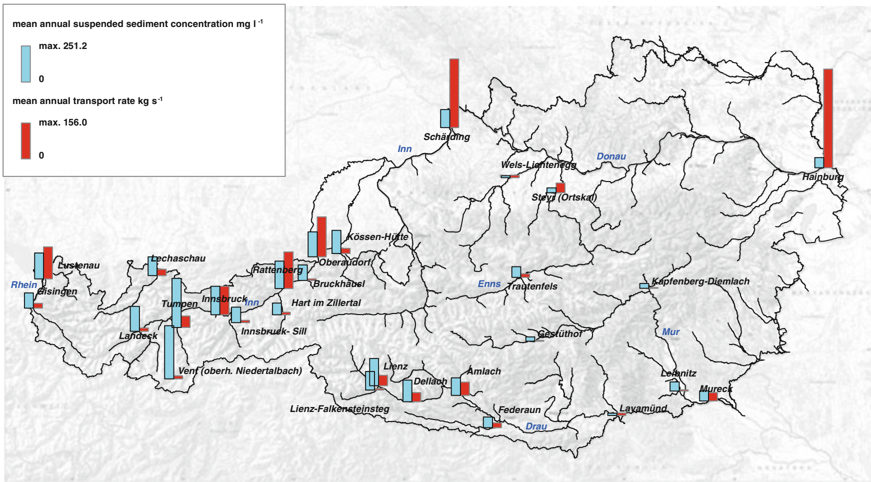


Fig. 10 The mean annual suspended sediment concentrations and transport rates 2009–2011

The maximum suspended sediment concentrations and transport rates are caused by different processes which may occur in different seasons. Debris flow due to thunderstorms and heavy rainfall on a small scale, normally during summer, may lead in alpine regions to high suspended sediment concentrations, whereas the maximum transport rates usually come along with increasing discharge. In alpine regions maximal transport rates are mainly due to orographic precipitation sometimes including thunderstorms in the course of meteorologically fronts. In lower regions (northern and south-east part of Austria) the maximum transport rates, mostly caused by areal soil loss by water (Strauss 2007), usually take place during flood or snowmelt waves in May/June (Fig. 11).

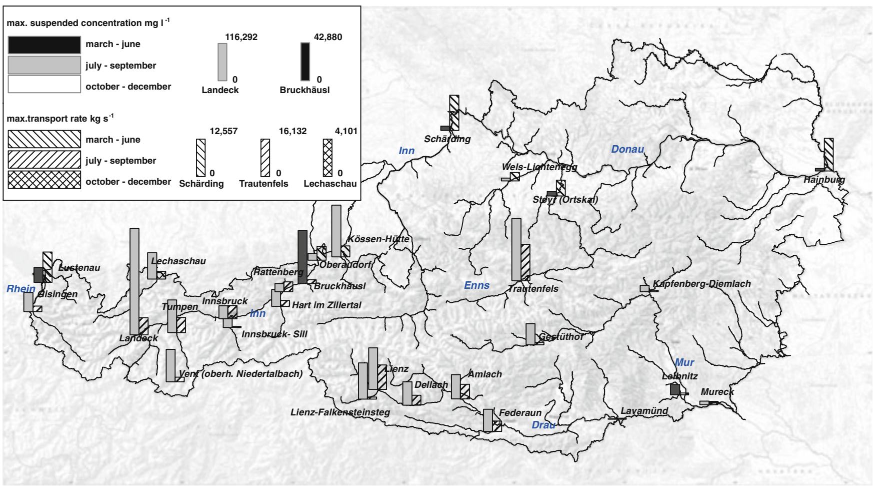


Fig. 11 The maximum suspended sediment concentrations and transport rates 2009–2011

6 Conclusions

In Austria a combination of direct and indirect methods is used to be able to determine the temporal and spatial variability of the suspended sediment transport. Optical sensors continuously record the turbidity at one point of the river bank and are calibrated by concentrations of water samples taken close to the sensor. To validate and verify the data anomalous values of turbidity record and concentrations of water samples have to be identified. Gaps and wrong turbidity records should be filled or corrected under consideration of all available data (calibration samples, water level, records from nearby stations, etc.) whereas unreliable samples shall not be taken into account.

To calculate the hydrograph of near sensor concentrations two approaches (probe factor and linear regression between turbidity data and calibration samples) are used whereby also a combination of the approaches is possible. The selected method to convert the turbidity data into near sensor suspended sediment concentration varies within the catchments but also with the season and has to be selected for each measurement site as it strongly affects the transport and load calculations. Comparing the results of both methods at each measurement site indicates that at some sites the deviation in sediment concentration are marginal but can reach up to 50 % in terms of maximum and up to 20 % in terms of mean concentrations and loads.

Furthermore measurements are performed to detect the distribution of suspended sediments in a cross section (multi-point sampling or bottle samples combined with acoustic devices). To consider these measurements a direct correlation (regression) between near sensor concentration and mean concentration is established. The results document that the consideration of the distribution of suspended sediments in a cross section is crucial to determine the suspended sediment load as the mean suspended sediment concentration in the cross section can reach more than the double of the suspended sediment concentration near the banks. Thus it is possible to estimate the suspended sediment load for certain time periods.

A mass balance of annual suspended sediment loads and event loads within a catchment can be performed to verify the accurate application and combination respectively of all the correction factors on the results of load calculations. In case of event loads it is important to consider the event type as heavy rainfalls or thunderstorms, limited to smaller sub catchments, may lead to high loads in this area but have a decreasing influence on the suspended sediment loads at the downstream measurement sites. Rainfalls in the whole catchment may add up the suspended sediment loads with increasing catchment size.

The analysis of the data 2009–2011 show that in general the annual suspended sediment load increases with the catchment area. The annual suspended sediment yield increases with the altitude of the catchment and/or with the percentage of glaciation, which is significant over ca. 5 % of glaciation. In Austria the western alpine regions, particularly the glacial zones, add to the highest mean annual sediment concentrations which decreases on the one hand with the decreasing mean annual precipitation from West- to East-Austria and on the other hand due to influences of natural river basins or hydro power plants. The mean annual transport rate accumulates with the increasing discharge and catchment size. Debris flow due to thunderstorms and heavy rainfall during summer may lead in alpine regions to high suspended sediment concentrations, whereas in lower regions (northern and south-east part of Austria) the maximum transport rates, mostly caused by areal soil loss by water, usually take place during flood or snowmelt waves in May/June. However, at measurement sites downstream of lakes or hydro power plants the mean annual transport rate or annual suspended sediment load is affected by suspended sediment deposition or flushing during flood waves.

Acknowledgments This work was financially supported by the Austrian Ministry of Agriculture, Forestry, Environment and Water Management. A special thank goes to the Hydrographic Services and via donau for the financial support, the supply of resources and performing the field measurements. The financial support by the Federal Ministry of Economy, Family and Youth and the National Foundation for Research, Technology and Development is gratefully acknowledged. We want to thank our colleagues especially Gabriele Müller and Reinhold Godina for their extremely useful comments and suggestions.

References

BMLFUW (2008) Schwebstoffe im Fließgewässer – Leitfaden zur Erfassung des Schwebstofftransportes. Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, Wien

BMLFUW (2013) Hydrographisches Jahrbuch von Österreich 2011. Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, Wien

Davis BE (2005) A guide to the proper selection and use of federally approved sediment and water-quality samplers. U.S. geological survey open file report, 2005-1087, USGS, Reston, Virginia

Deines KL (1999) Backscatter estimation using broadband acoustic doppler current profilers, oceans 99 MTS. In: IEEE proceedings, San Diego, California

Edwards T, Glysson G (1999) Field methods for measurement of fluvial sediment. In: Techniques of water-resources investigations of the U.S. geological survey: application of hydraulics. U.S. Geological Survey, Virginia (Book 3)

Gippel ChJ (1995) Potential of turbidity monitoring for measuring the transport of suspended solids in streams. Hydrol Process 9:83–97

Gray JR, Gartner JW (2009) Technological advances in suspended-sediment surrogate monitoring. Water Resour Res 45 W00D29. doi:10.1029/2008WR007063

Habersack H, Wagner B, Schoder A, Hauer C (2013) Die Bedeutung von Feststoffhaushalt und Sedimentdurchgängigkeit für eine nachhaltige Nutzung der Wasserkraft. Österreichische Wasser- und Abfallw 65:354–361. doi:10.1007/s00506-013-0108-0

Haimann M, Liedermann M, Naderer A, Lalk P, Habersack H (2012) Integratives Schwebstoffmonitoringkonzept—Innovative Ansätze auf Basis direkter und indirekter Methoden. Österreichische Wasser- und Abfallw 64:535–543. doi:10.1007/s00506-012-0038-2

Haimann M, Liedermann M, Lalk P, Habersack H (2014) Development and evaluation of an integrated suspended sediment transport monitoring and analysis concept. Int J Sediment Res 29(2):135–148. doi:10.1016/S1001-6279(14)60030-5

ISO 4363 (2002) Measurement of liquid flow in open channels—methods for measurement of characteristics of suspended sediment. International standard, Geneva, Switzerland

Kling H, Nachtnebel HP, Fürst J (2005) Mean annual areal precipitation using water balance data. In: BMLFUW (ed) Hydrologischer Atlas Österreichs. 2. Lieferung. Kartentafel 2.3. Wien: Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, ISBN 3-85437-250-7

Lewis J (1996) Turbidity-controlled suspended sediment sampling for runoff-event load estimation. Water Resour Res 32(7):2299–2310

López-Tarazón JA, Batalla RJ, Vericat D, Francke T (2009) Suspended sediment transport in a highly erodible catchment: the river Isábena (Southern Pyrenees). Geomorphology 109:210–221

Nachtnebel H-P, Seidelmann R, Müller HW, Schwaighofer B (1998) Schwebstoffe in der Donau. Forschung im Verbund, Schriftenreihe 45:147

Oeurng C, Sauvage S, Sánchez-Pérez J-M (200) Dynamics of suspended sediment transport and yield in a large agricultural catchment, southwest France. Earth Surf Process Land 35: 1289–1301

Porterfield G (1972) Computation of fluvial-sediment discharge. Techniques of water-resources investigations. United States Geological Survey, Virginia, Chapter C3

Prazan H (1994) Determing suspended sediment transport in Austria. Österreichische Wasser- und Abfallwirtschaft, Heft 7(8):181–194

Schoellhamer DH, Wright SA (2003) Continuous measurement of suspended-sediment discharge in rivers by use of optical backscatterance sensors. In: Erosion and sediment transport measurement in rivers: technological and methodological advances, vol 283. IAHS Publication, pp 28–36

Spreafico M, Lehmann C, Jakob A, Grasso A (2005) Feststoffbeobachtung in der Schweiz – Ein Tätigkeitsbericht der Landeshydrologie, Berichte des BWG, Serie Wasser nr. 8, Bern

Strauss P (2007) Areal soil loss by water. In BMLFUW (ed) Hydrologischer Atlas Österreichs. 3. Lieferung. Kartentafel 8.4. Wien: Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft ISBN 3-85437-250-7

Walling DE, Fang D (2003) Recent trends in the suspended sediment loads of the world's rivers. Global Planet Change 39:111–126

Wass PD, Leeks JL (1999) Suspended sediment fluxes in the Humber catchment, UK. Hydrol Process 13:935–953

Wren DG, Barkdoll BD, Kuhnle RA, Derrow RW (2000) Field Techniques for suspended-sediment measurement. J Hydraul Eng 126: 97–104

Investigation of the Metal Contamination in the Upper Olifants Primary Catchment by Using Stream Sediment Geochemistry, Witbank Coalfield, South Africa

Robert Netshitungulwana, Bisrat Yibas, Christoph Gauert, Danie Vermeulen, Obed Novhe and Tshepa Motlakeng

Abstract The Olifants primary catchment area, consists of nine sub-catchments marked from B1 to B9, extends over the border between South Africa and Mozambique, and has a total area of approximately 87,000 km^2. The B1 catchment, where most of the mining activities surround the major towns of Witbank (Emalahleni) and Middleburg, in turn straddles the provinces of Mpumalanga and Limpopo. Although industrial and agricultural activities are also important, the contribution of contamination from the mining activities within the catchment is significant as the result of intense mining activities of various mineral commodities such as coal and from ferrochrome processing plants located in Emalahleni and Middleburg towns with in the catchment area and yet not fully quantified. This paper investigates the severity of the mining impacts on the water resources and the ecosystem of the Olifants primary catchment area and in particular, the upper reaches of the catchment. The paper discusses the results of research which focused on deciphering the severity and the sources water contamination, and on how to minimise the dispersion of these metals into the streams, and on the relationship of the water quality and metal loadings on the sediments. Stream sediment and water samples have been collected and analysed. The sediments were analysed by Simultaneous X-ray Fluorescence and Inductively Coupled Plasma-Mass Spectrometry techniques for metal loadings. The areas were marked by anomalous level determined at 50th percentile threshold of Fe, Mn, Ni, Cr, Co, V, Pb in Emalahleni

R. Netshitungulwana (✉) · B. Yibas · O. Novhe · T. Motlakeng
Department of Economic Geology, Council for Geoscience, 280 Pretoria Road, Silverton, Pretoria 0184, South Africa
e-mail: robertn@geoscience.org.za

C. Gauert
Department of Geology, University of the Free State, 205 Nelson Mandela Drive, Park West, Bloemfontein 9300, South Africa

D. Vermeulen
Institute for Ground Water Studies, University of the Free State, 205 Nelson Mandela Drive, Park West, Bloemfontein 9300, South Africa

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_11

and Al, Fe, Mn, Cr, As, Zn, Pb and U in Middleburg. The ICP-MS and IC analytical techniques were used in the assessment of water quality data. From the stream sediments regional geochemistry at catchment level and for this investigation, the sediments that were found marked by high levels of Na, K, Mg, Al, Ca, Mn, and Fe signature can be attributed to the coal mines as a probable source. Whereas the sediment quality of the areas like Emalahleni and Middleburg towns, where mining of coal (with many abandoned mines) and ferrochrome processing is happening simultaneously, there are anomalous level of Cr, Ni, V and As, which is a signature of the Bushveld PGE mines material. The SO_4^{2-} concentration of above 500 mg/kg on the water quality, which has exceeded the Department of Water Affairs water quality guideline for domestic and industrial use, is an evidence for contamination. The approach adopted herein suggests that the stream sediment and water quality data can be used in characterizing or fingerprinting impacted areas.

Keywords Stream sediments · Geochemistry · Threshold · Olifants primary catchment area · South Africa

1 Introduction

The Olifants River catchment has an area of approximately 87,000 km^2 and is located in the northern part of South Africa straddling the border with Mozambique to the East (Fig. 1). The catchment consists of 9 subcatchments named B1 to B9. The B1 sub-catchment is the subject of this paper. The major towns within the B1 sub-catchment are Witbank (Emalahleni) and Middleburg located in the northern parts of the catchment. The towns are largely surrounded by a number of active and abandoned coal mines.

The B1 catchment drained by the upper reaches of the Olifants, Little Olifants and Riet rivers and their tributary streams, down to the point where the Olifants River joins the Wilge River at the Loskop Dam (Ashton et al. 2001). The rivers drain the coal-mining towns of Witbank, Middelburg, Arnot, Hendrina, and Kriel. The sub-catchment also receives additional water via three inter-basin transfer schemes from the Vaal and Crocodile/Komati catchments. All rivers and streams in this sub-catchment are perennial. The sub-catchment also contains a number of small wetlands located next to every stream and river (Marneweck et al. 2001).

2 Geological Setting

The Witbank coalfield in the Mpumalanga Province of South Africa is located in the northern sector of the main Karoo Basin and intruded by the dolerite dykes and sills during the initial stage of Gondwana fragmentation. In the Witbank coalfields,

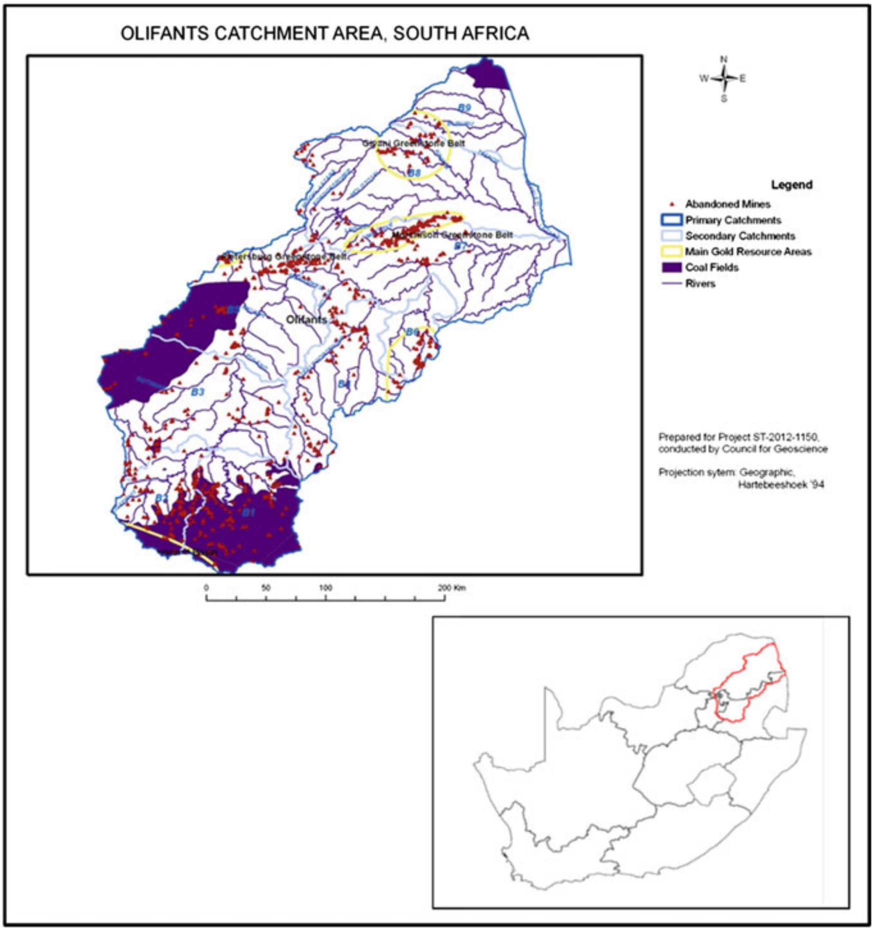


Fig. 1 Locality map of the B1 sub-catchment study area, within the Olifants Primary Catchment. *Note* The sub-catchments are numbered from B1 to B9

the sediments occurring above sills were displaced and uplifted and sediments occurring underneath the sills remain undisturbed.

There is minimal vertical displacement on the Dolerite dykes of the Witbank area (Smith and Whittaker 2005). The heat emanating from dolerite dykes and sills accelerated metamorphism and depleted the volatile constituents of the coal seams. Coal deposits in the Witbank coal-fields occur within Vryheid formation, Ecca Group, Karoo Supergroup. The seams that are normally found in the Witbank Coal Basin are numbered, seam No 1, No 2, No 3, No 4 and No 5 (Fig. 2). Coal seams normally thin out towards the smaller palaeo-ridges and eventually pinch out against the main palaeo highs (Snyman 1998). The seams are flat lying to gently undulating; seals (15–50 m) transgress seams; dykes (0–1 m) common (trends east,

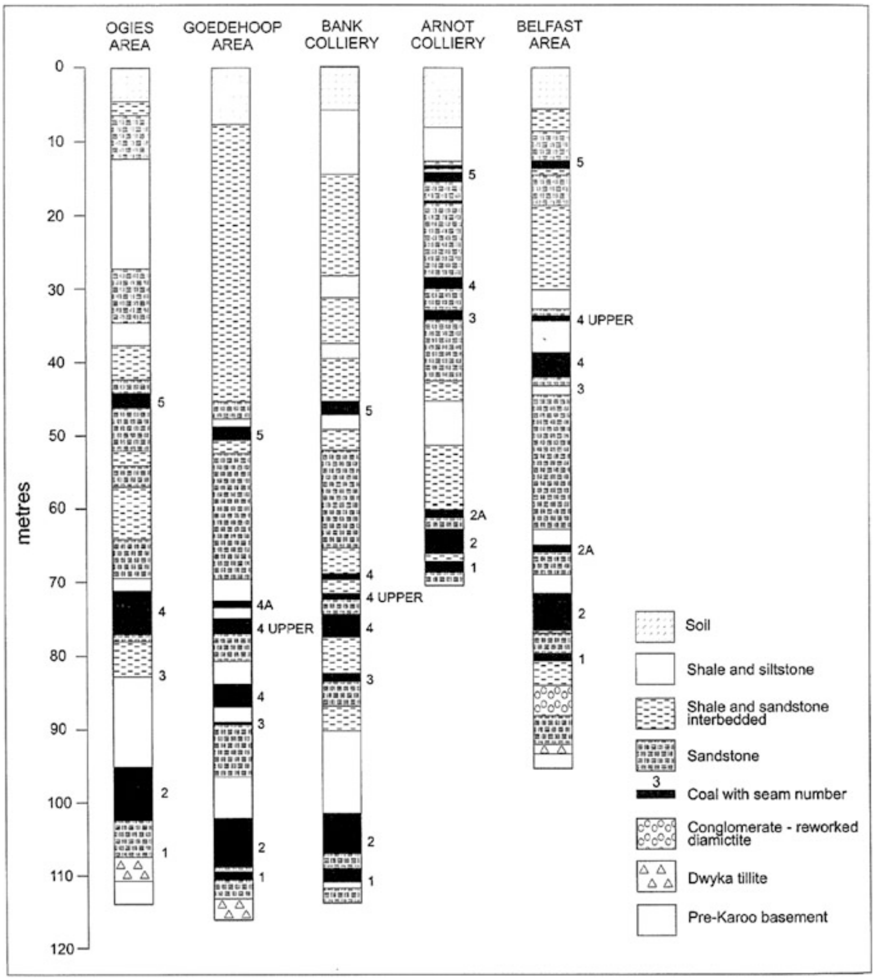


Fig. 2 Generalised stratigraphic columns of the Witbank Coalfield (Smith and Whittaker 1986)

north east, north); most prominent dyke: Ogies dyke (15 m thick, 100 km long and strikes east-west) transgressive seals caused tilting and displacements of seams-mining blocks at different elevations, causing major problems with mining. The degree and extent of coal of burning associated with intrusions poses a serious problem to the mining and to the resource estimation (Smith and Whittaker 2005).

3 Research Background

The Council for Geoscience in collaboration with the Department of Mineral Resources launched a project to investigate the severity of the mining impacts on the water resources and the ecosystem of the country on catchment by catchment based approach. The Olifants primary catchment area has been identified to be studied priory due to the density of mining activity within its territory.

Sediments have been used for mineral exploration and in assessing the environmental concern areas internationally (see Salminen et al. 2005). The surface area of the sediment particle affects the nature and number of binding sites for metal and organic contaminants. Fine sediments often have the highest concentration of contaminants on a dry weight basis as they have a higher relative surface area and thus increased density of sorption sites. Stream sediments are usually considered as a sink for trace metals, but they can also act as a source of metals depending on the change of environmental conditions (Segura et al. 2006). Stream sediment analysis can also be used to estimate the point source of contamination that, upon being discharged to surface waters, are rapidly absorbed by particulate matter, thereby escaping detection by water monitoring (Forstner 2004).

The screening study revealed anomalous values of Fe, Mn, Cr, Pb, Zn, U and Al in sediments in B1 catchment areas A and C, which forms part of the drainage area of the Emalahleni. The sediments from the screening level study revealed the sediments had a potential to generate acid (Netshitungulwana and Yibas 2012; Netshitungulwana et al. 2013) the threshold was mean plus two standard deviation. The following environmental concern areas are associated with the mining activities: area A in Witbank show elevated concentrations of Al, Fe, Mn, As, Cr, Pb, U; area B is located in Middleburg and show elevated concentrations of Al, Fe, Mn, As, Cr, Pb, Zn and U; area C is located south of the Loskop Dam and show elevated concentrations of Fe, Mn, Ni, Zn, Cr, Co, V and Pb.

This paper presents the findings of the study of the severity of the mining impacts on the water resources and the ecosystem of the Olifants catchment area and in particular around Emalahleni areas. The question that this research attempts to answer is the sources of these metals, and to minimise the dispersion of these metals into the streams, and the relationship of the water quality and metal loadings on the sediments. The environmental issues of water quality in the area have been the subject of intense public scrutiny recently. The attention is focused on coal mining area and in particular the coalfields of Mpumalanga.

4 Methodology

A total of 96 stream sediment samples were collected for various analyses and tests which include X-ray Fluorescence Spectrometer (SXRF) for chemical composition and Acid base Accounting (ABA) for Acid mine drainage (AMD) potential assessment. The sample analysis was done for bulk sample <75 µm fraction or grain size. Samples were analysed on a Philips PW 1606 Simultaneous XRF for the following elements: TiO_2 MnO and Fe_2O_3T in %; Sc, V, Cr, Co, Ni, Cu, Zn, As, Rb, Sr, Y, Zr, Nb, Sn, Sb, Ba, W, Pb, Th and U in mg/kg. The steps to calibrate the instrument accuracy and precision obtained, and the reference materials used, were described by Elsenbroek (1996). The resulting stability of analyses was monitored on a daily analysis, per batch of 300 samples analysed. For major elements analysis, the milled sample was roasted at 1,000 °C for at least 3 h to oxidise Fe^{2+} and S, and to determine the loss of ignition (L.O.I). Glass disks were prepared by fusing 2 g roasted sample and 8 g flux consisting of 35 % $LiBO_2$ and 64.71 % $Li_2B_4O_7$ at 1,050 °C. For trace elements analysis, 12 g milled sample and 3 g Hoechst wax was mixed and pressed into a powder briquette by a hydraulic press with the applied pressure at 12 tons. The glass disks and wax pallets were analysed on a P Analytical Axios X-ray Fluorescence spectrometer equipped with a 4 KW Rh tube (Elsenbroek 1996).

The assessment on the elevated concentrations attributable to AMD background concentration was done by means of exceeding probability from the data-set itself and not by comparison of the results to statutory regulations because the South African sediments standards is not yet developed, or geochemical background concentrations as can be found in literature. Considering multiple populations in the data, the mean plus two times standard deviation was not used, and only data above the 50th percentile or the median value was considered anomalous. The anomalous metal areas are represented by the red bubble (50th percentile) and a purple star for 75th percentile (Figs. 3, 4, 5, 6, 7, 8, 9 and 10).

5 Results

The elevated concentrations of major oxides and trace metals presented in this section are located downstream of the coal mining activities (Figs. 3, 4, 5, 6, 7, 8, 9 and 10). The anomalies are mainly located within the Karoo Supergroup geological units (see geological setting section). Al_2O_3 in wt% is symmetrically distributed and the anomalous concentrations are from the 50th percentile value of 6.7 wt% to the maximum value of 18 wt% values (Fig. 3). Figure 4 also indicates asymmetrically distributed of Fe_2O_3T anomalies, which are from the 50th percentile value of 4.41 wt% to the maximum value of 14.3 wt%. The anomalous values of total iron and aluminium are located on the Olifants River at the outflow of the Witbank Dam (Figs. 4 and 5) and also in the contaminated streams of Brugspruit and Klipspruit (Fig. 12). There are also significant anomalies of aluminium and total iron located in Middleburg and South of Witbank areas (Fig. 4).

Cr concentrations (Fig. 5) indicates the 50th percentile value of 129 mg/kg is below the average value of the basaltic rocks 200 mg/kg, there are samples above the basaltic rocks average but lower than that of the ultra-mafic rocks of 2,000 mg/kg (Levinson 1974). The Cr value is above the South African Soil Quality Standard

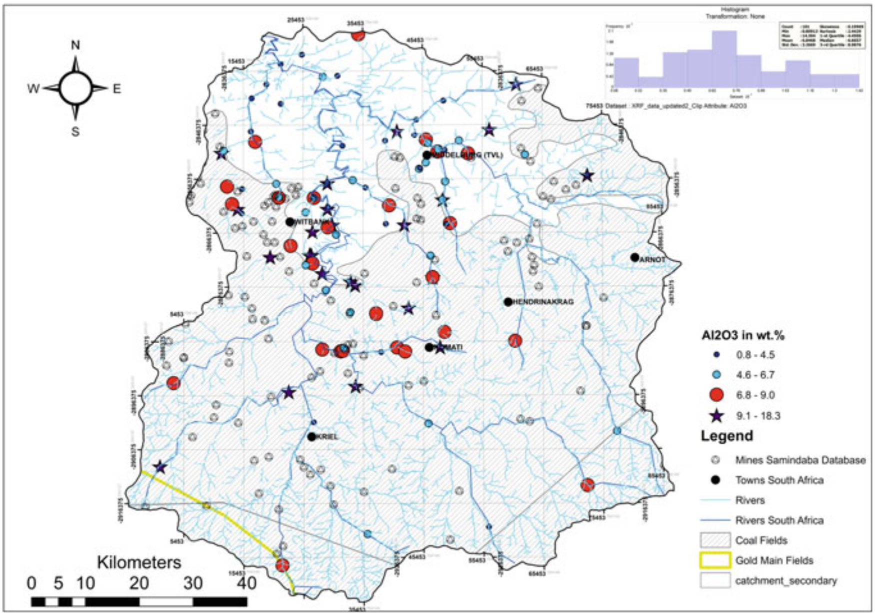


Fig. 3 The distribution of Al_2O_3 concentrations (wt%) in the sediments of the B1 sub-catchment, Olifants catchment

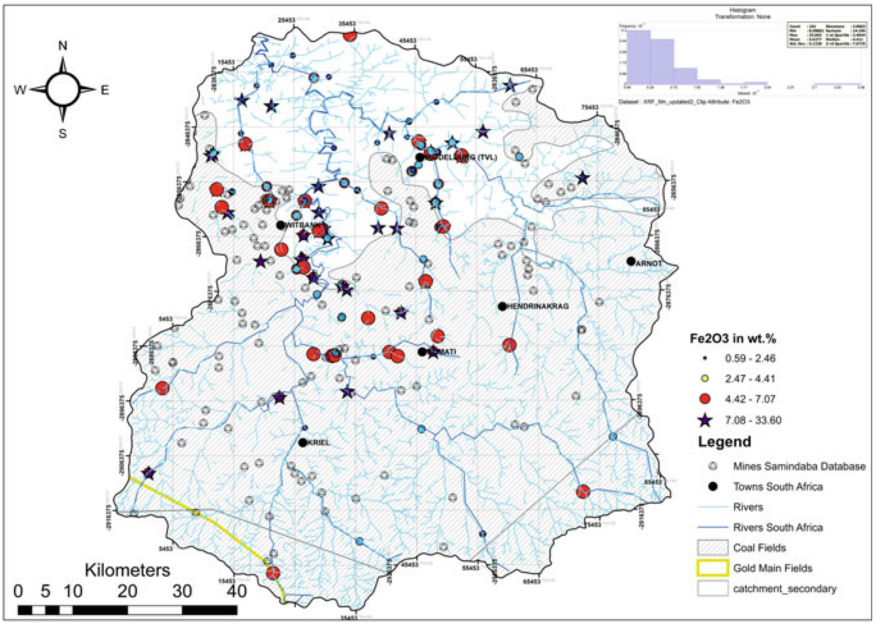


Fig. 4 The distribution of Fe_2O_3T concentrations (wt%) on the sediments of the B1 sub-catchment, Olifants catchment

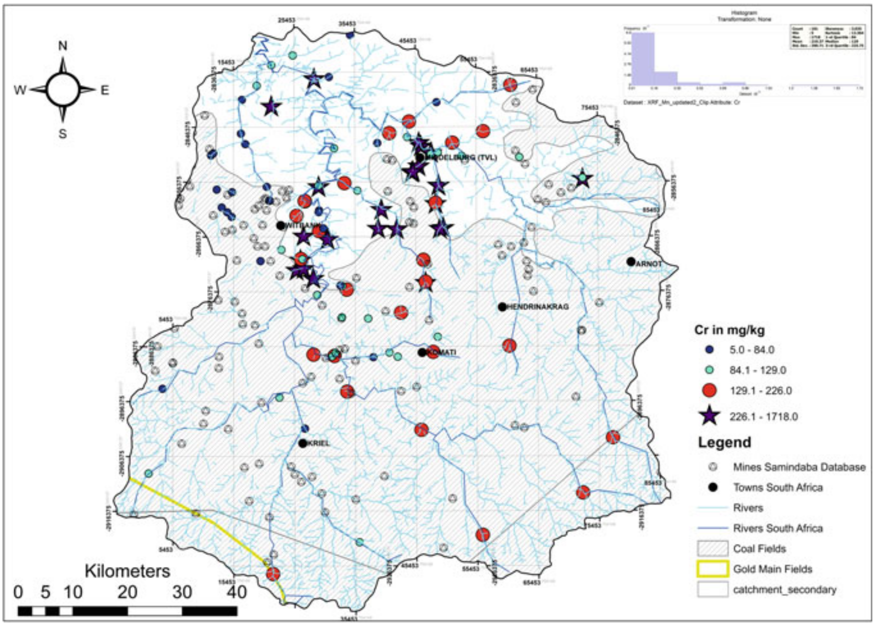


Fig. 5 The distribution of Cr concentrations in mg/kg on the sediments of the B1 sub-catchment, Olifants catchment

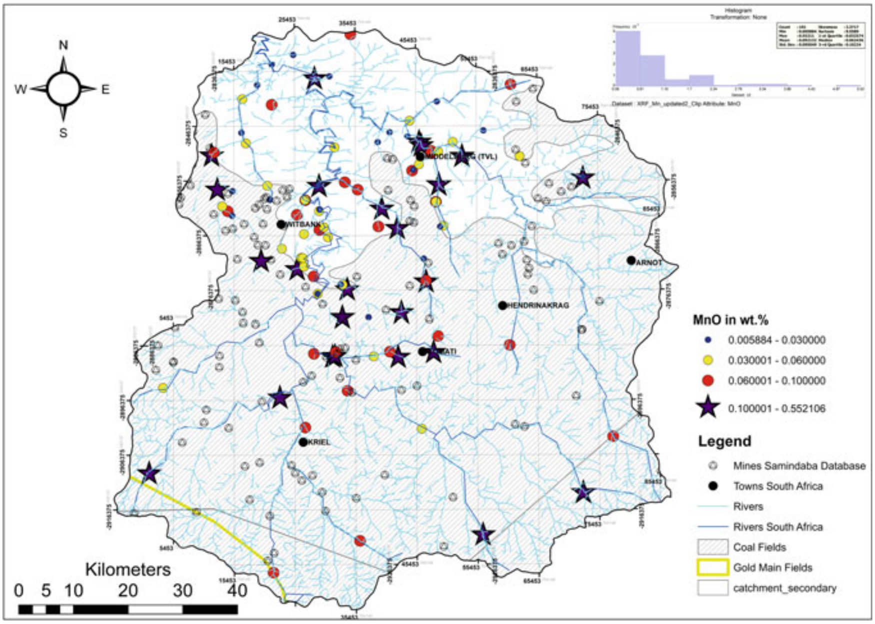


Fig. 6 The distribution of MnO concentrations (wt%) on the sediments of the B1 sub-catchment, Olifants catchment

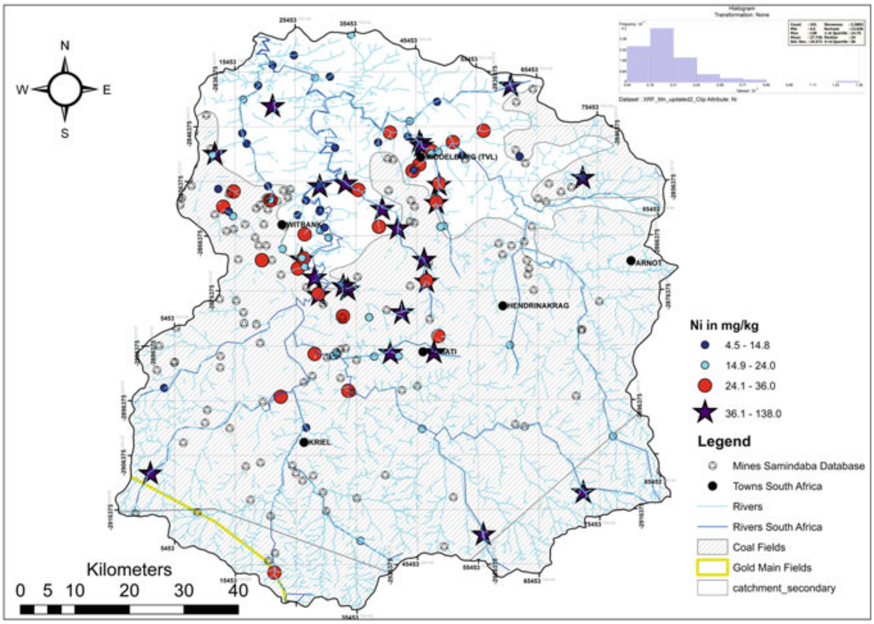


Fig. 7 The distribution of Ni concentrations in mg/kg on the sediments of the B1 sub-catchment, Olifants catchment

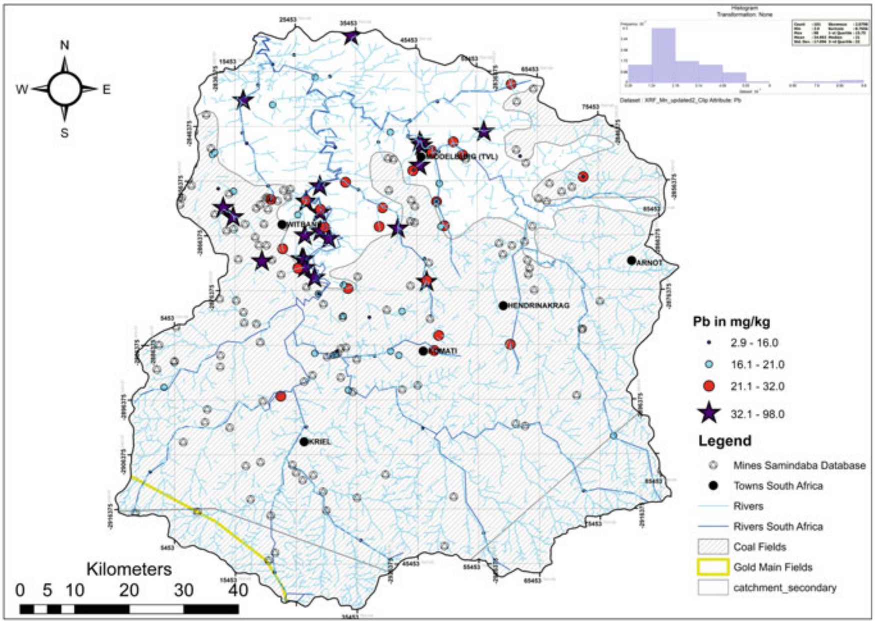


Fig. 8 The distribution of Pb concentrations in mg/kg on the sediments of the B1 sub-catchment, Olifants catchment

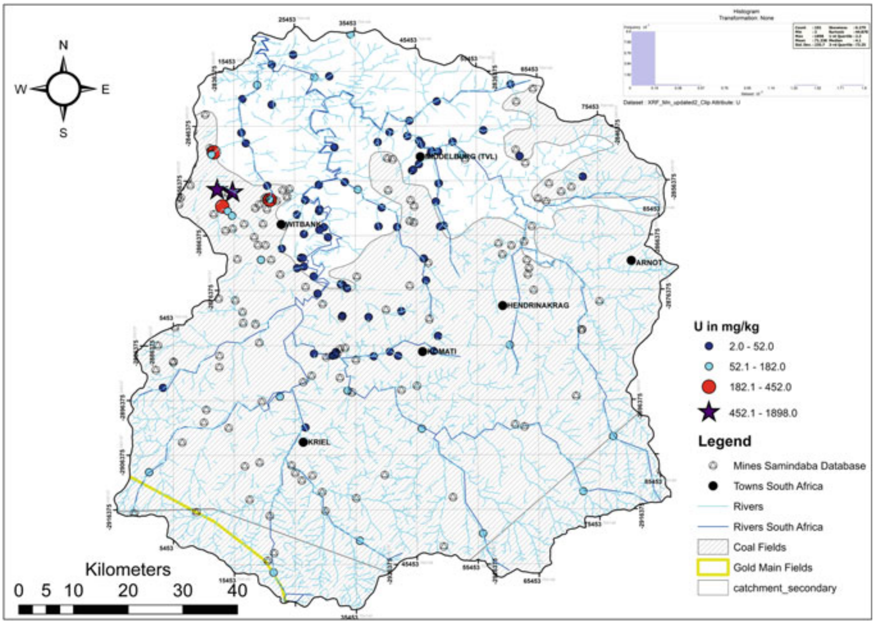


Fig. 9 The distribution of U concentrations in mg/kg on the sediments of the B1 sub-catchment, Olifants catchment

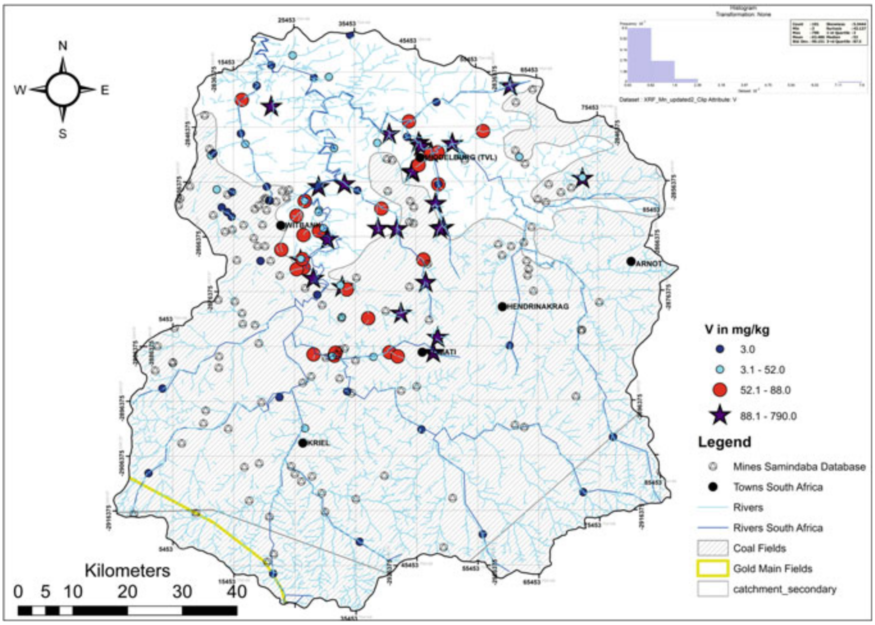


Fig. 10 The distribution of V concentrations in mg/kg on the sediments of the B1 sub-catchment, Olifants catchment

(Table 1). The Cr anomalies supports the presents of mafic rocks within the catchment (Fig. 11), however the chrome anomaly distribution needs further investigations.

MnO in wt% has the standard deviation value of 0.09 wt% which is less than 0.14 wt% of the primary catchment and the highest is in the ultramafic rocks of the Bushveld Complex and the greenstones. The statistical observation suggests the MnO is heterogeneously distributed in the sediments, with a positively skewed data causing the median value to be less than the mean value. The anomalous values of manganese are above on the 50th percentile of value of 0.06 wt% and the maximum value is 0.552 wt% (Fig. 6).

Ni concentrations (Fig. 7) have the standard deviation value of 19.3 mg/kg which is less than the primary catchment value of 73 mg/kg. The mean value of 27 mg/kg is found higher than the median value of 24 mg/kg. These statistical observations suggest that the distribution of the nickel in the sediments is heterogeneous and the data is positively skewed to the right. The median nickel value of 24 mg/kg is below the average of the basaltic rocks value of 200 mg/kg and the ultra-mafic rocks value of 2,000 mg/kg (Levinson 1974). The nickel anomalous values range from 24 mg/kg to the maximum value of 138 mg/kg and therefore generally below the basaltic rock average, however the anomalies are distributed near similar to the Cr anomaly.

The Pb concentration (Fig. 8) has the mean value of 25 mg/kg which is above the median value of 21 mg/kg. The Pb data is heterogeneous distributed and positively skewed. The median value of 21 mg/kg and the maximum value of 98 are way above the average earth's crust value of 12.5 mg/kg (Levinson 1974). The lead concentration is above the South African Soil Quality Standard (Table 1).

Table 1 South African Soil screening standards (Department of Environmental Affairs 2012) versus stream sediments metal concentrations

Metals	Units	All land uses protective of the water resource	Olifants River stream sediments 50th	Olifants River stream sediments 75th
Chromium (Ill)	mg/kg	46	121 Cr total	226 Cr total
Chromium (VI)	mg/kg	6.5	122 Cr total	227 Cr total
Cobalt mg/kg	mg/kg	300	19	37
Copper	mg/kg	16	19	37
Lead	mg/kg	20	21	32
Nickel mg/kg	mg/kg	91	28	36
Vanadium mg/kg	mg/kg	150	52	88
Zinc mg/kg	mg/kg	240	64	175
Uranium	mg/kg	n/a	182	452
Cadmium	mg/kg	7.5	–	–
Manganese	mg/kg	740	–	–
Mercury	mg/kg	0.93	–	–

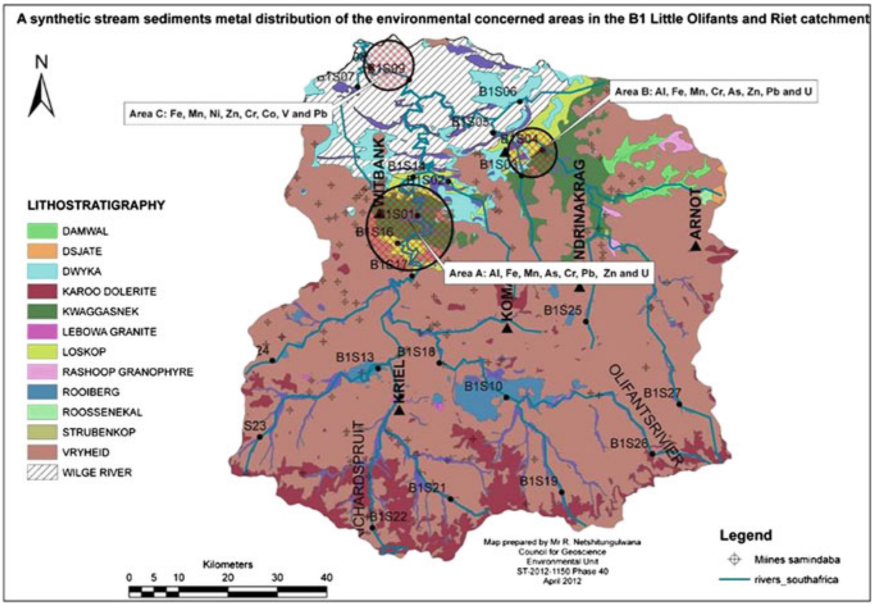


Fig. 11 A stream sediment metal distributions map for the environmental concern areas in the B1 Little Olifants and Riet catchment (Netshitungulwana and Yibas 2012)

The U concentrations in mg/kg presented in Fig. 9 has the mean value of 71 mg/kg which is above the median value of 4 mg/kg and the positively skewness value of 6, suggest that the data is heterogeneously distributed, meaning is probably from the external source. The uranium anomaly of between 452 and 1,898 mg/kg located in Area A of the B1 catchment area, may suggest the sediments are probably drained from the uranium rich bedrock.

The V concentrations presented in Fig. 10 has the mean value of 63 mg/kg which is above the median value of 52 mg/kg. Vanadium is heterogeneous distributed and the data is positively skewed.

6 Discussions

The main objective of this investigation was to assess the contamination sources and the metal distribution in the sediments of the B1 secondary catchment. The area was considered significant because the screening phase revealed geochemical signature identified three major concern areas with high levels of certain metals (Al, Fe, Mn, As, Cr, Ni, Cr, Pb, Zn, V and U) at various level (Fig. 11). The main three impacted areas were identified and discussed below, that is Area A (the area around Witbank town), Area B (the area around Middleburg town) and Area C (the area North to Northwest of the Witbank town) (Fig. 11). Furthermore principal component analysis performed identified four areas. The four areas were identified using the Principal Analysis Factor 1 metals of Al, Cr, Cu, Fe, Mn and As. The PCA areas identified are discussed within the main three impacted areas identified during the screening phase of investigations. Area A is marked by PCA2 and PCA4, Area B is marked by PCA3 and Area C is marked by PCA1 (Fig. 12).

6.1 Area A (PCA2 and PCA4)

From the screening phase, Area A located on the Witbank area has a metal signature of Al, Fe, Mn, As, Cr, Pb, Zn and U. Elevated concentration of Al, Fe and Mn (Figs. 6, 8 and 9), may be associated with the activities of coal mining upstream of the Olifants River, and most of the samples are located within the Witbank Dam. ABA test shows the samples that make up Area A as potentially acid producing, with negative Net Neutralisation Potential (NNP); paste pH of <5.8 (Netshitungulwana and Yibas 2012). The Cr, Pb, U, As and Zn signatures may be associated with the mafic rocks (Fig. 11) dominant in the catchment (Netshitungulwana et al. 2013) and classifies the area as PCA3 (Fig. 12) where metal loadings on the sediments include Fe, Al, Mn, As, Cr, Cu and Mg, and the SO_4^{2-} concentration range from 242 to 860 mg/l and were found above the South African water quality drinking and industrial guidelines.

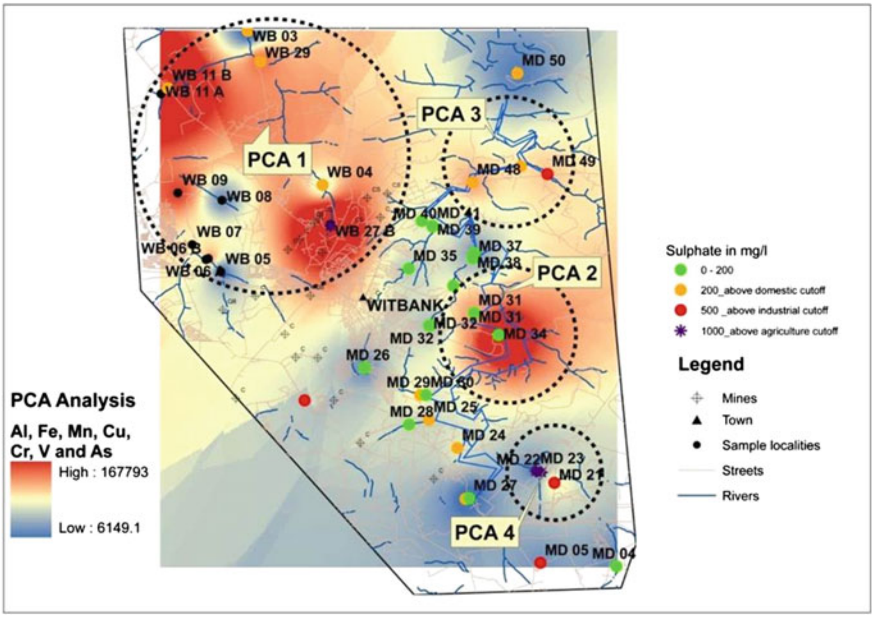


Fig. 12 Areas PCA1 to PCA4 discriminated by the principal component analysis, the metals entered are Al, Fe, Mn, Cu, Cr, V and As for PCA Factor 1 (Netshitungulwana et al. 2013)

6.2 Area B (PCA3)

Area B, located on the Middleburg town has a metal signature of Al, Fe, Mn, Cr, As, Pb, Zn and U. The metal signatures of Al, Fe and Mn may be associated with the coal mining upstream. The samples that make up Area B are potentially acid producing with a negative NNP values. The Ferrochrome Plant/s and around the Middleburg may attribute to the signature of Cr, total iron and V. Arsenic is considered the most possible adverse effect element even at concentration greater than 3 mg/kg in sediment (Irwin et al. 1997 and references therein). The results of arsenic concentration of 8, which is anomalous is above the threshold effect level (TEL), lowest effect level (LEL), minimal effect threshold (MET) and the maximum value of 55 mg/kg is above, effects range low (ERL), probable effects level (PEL), toxic effect threshold (TET), severe effect level (SEL) which could be the possible adverse effects on the sediments (Burton 2002). The other metals which range to the possible adverse include Cr, Ni and Pb.

6.3 Area C (PCA1)

Area C is located in the northern part of Middleburg and Witbank within the Klipsruit River and the tributaries are Brugspruit and Blesbokspruit, south of the Loskop dam approximately 40 km north of mining sites. The area is characterized by elevated concentration of Fe, Mn, Ni, Zn, Cr, Co, V and Pb. The elevated metal signatures may be associated with the catchment geology, the ferrochrome processing plant and abandoned coal mines upstream.

The samples in the Blesbokspruit show average paste pH value of 4.86, paste Electrical conductivity (EC) of 2,613 µs/cm and 1.09 % of sulphur content (Netshitungulwana et al. 2013). Paste pH and EC indicates the low quality of the pore water. The high concentration of Mn, Pb, Mg, Na, Cd and SO_4^{2-} in the water samples exceeded the South African drinking water quality guidelines (Netshitungulwana et al. 2013; Bell et al. 2002). The chemistry of the sediments and associated pore water show AMD formation and release of metals (Netshitungulwana et al. 2013).

7 Conclusions

The study shows that the stream water and streambed sediments are impacted by operating mines, abandoned mines and the unattended mine residue deposits upstream. The sediments with anomalous concentration of Al, Mn and Fe are probably sourced from coal whereas Cr, Ni and V are indicative of mafic rocks sources. The water samples form the most streams in the catchment has pH that is ranging from near neutral to alkaline conditions and at this condition most metals get adsorbed onto the sediments. Streams like Brugspruit and Klipspruit show low levels of Mn in the sediments, which may suggest most of the Mn is still in concentrated in water. The uranium anomaly of between 452 and 1,898 mg/kg located in Area A of the B1 catchment area may be derived from the processed Bushveld material upstream.

References

Ashton P, Love D, Mahachi H, Dirks P (2001) An overview of the impact of mining and mineral processing operations on water resources and water quality in the Zambezi, Limpopo and Olifants catchments in southern Africa, Contract Report to the Mining, Minerals and Sustainable Development (Southern Africa) Project, by CSIREnvironmentek, Pretoria, South Africa and Geology Department, University of Zimbabwe, Harare, Zimbabwe. Report No. ENV-P-C 2001-042. xvi p 336

Bell FG, Bullock SET, Hälbich TFJ, Lindsay P (2001) Environmental impacts associated with an abandoned mine in the Witbank Coalfield, South Africa. Int J Coal Geol 45:195–217

Burton GA (2002) Sediment quality criteria in use around the world. The Japanese Society of Limnology. Limnology 3:65–75

Department of Environmental Affairs (2012) Draft national norms and standards for the remediation of contaminated land and soil quality. Environmental Management: Waste Act, 2008 (Act No. 59 of 2008), South Africa, General Notice 233 of 2012

Elsenbroek JH (1996) Instrumentation and analytical technique for the analysis of the analysis of regional geochemical samples used at the South African Council for Geoscience. Analyst 120:1535–1541

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

Irwin RJ, van Mouwerik M, Stevens L, Seese LD, Basham W (1997) Environmental contaminants encyclopedia. National Park Service Water Resources Divisions, Water Operations Branch, Colorado

Levinson AA (1974) Introduction to exploration geochemistry. Applied Publishing Ltd. Wilmette, Illinois, Department of Geology and Calgary, USA

Marneweck G, Batchelor AL, Turpie J (2001) Ecological and economic valuation of wetlands in the upper Olifants catchment. Water Research Commission Project Report No. K5/1162. (In progress)

Netshitungulwana R, Yibas B (2012) Stream sediment geochemistry of the Olifants catchment, South Africa: implication for acid mine drainage. In: McCullough CD, Lund MA, Wyse L (eds) International mine water association symposium, Bunbury, Western Australia, pp 257–264

Netshitungulwana R, Yibas B, Novhe ON, Motlakeng T (2013) Stream sediment geochemistry of the areas impacted by mining around Emalahleni (formerly known as Witbank), South Africa: fingerprinting AMD potential point sources. In: Brown A, Figueroa L, Wolkersdorfer Ch (eds) Reliable mine water technology. Denver, Colorado, USA (Publication Printers), vol I, pp 1–778

Salminen R, Batista MJ, Bidovec M, Demetriades A, De Vivo B, De Vos W, Duris M, Gilucis A, Gregorauskiene V, Halamic J, Heitzmann P, Lima A, Jordan G, Klaver G, Klein P, Lis J, Locutura J, Marsina K, Mazreku A, O'Connor PJ, Olsson S, Ottesen RT, Petersell V, Plant JA, Reeder S, Salpeteur I, Sandström H, Siewers U, Steenfeldt A, Tarvainen T (2005) FOREGS geochemical Atlas of Europe, Part 1—background information, methodology and maps: geological survey of Finland, Espoo, p 525. http://www.gtk.fi/publ/foregsatlas/

Segura-Munoz SI, Adasilva Oliviery M, Nikaido T, Trelliato MB, Bocio A, Takayanagui AMA, Damingo JL (2006) Metals levels in sugarcane samples from an area under the influence of a municipal landfill and a medical waste treatment system in Brazil. Environ Int 32:52–57

Smith DAM, Whittaker RLG (1986) The coalfields of southern Africa. In: Anhaeusser CR, Maske S (eds) The mineral deposits of South Africa. Geological Society of South Africa, Johannesburg, pp 1875–1878

Smith DAM, Whittaker RLG (2005) The Springs-Witbank Coalfield. In: Anhaeusser CR, Maske S (eds) Mineral deposits of Southern Africa, vol 2. Geological Society of South Africa, Johannesburg, pp 1969–1984

Snyman CP (1998) Coal. In: Wilson MGC, Anhaeusser CR (eds) The mineral deposits of South Africa. Handbook, Council for Geoscience, pp 136–205

Part V
Sediment Managing in River Basins

An Approach to Simulating Sediment Management in the Mekong River Basin

Thomas B. Wild and Daniel P. Loucks

Abstract The Mekong/Lancang River Basin in Southeast Asia is undergoing a period of rapid hydropower development. Newly constructed dams will trap ecologically valuable sediments, which transport nutrients and maintain the river's morphology. Sediments trapped behind hydropower dams could significantly impact the basin's exceptional biodiversity and food production that support many of those living in the basin. This paper introduces an approach for estimating the potential impact of reservoirs on the basin's sediment regime, as well as the potential for various forms of reservoir sediment management to improve sediment passage through and around dams. Our sediment simulation model, *SedSim*, predicts in relative terms the spatial and temporal accumulation and depletion of sediment in river reaches and in reservoirs under different reservoir siting, design and operating policies. The model identifies the relative tradeoffs between hydropower production, and flow and sediment regime alteration, associated with reservoir sediment management techniques, including flushing, sluicing, bypassing, density current venting and dredging. While developed for and applied in the Mekong River basin, this approach may be of interest to those facing similar sediment management challenges in other data-scarce regions.

1 Introduction

The Mekong/Lancang River (Fig. 1) flows from the Upper Mekong Basin in China (where it is called the *Lancang Jiang*) and Myanmar to the Lower Mekong Basin (LMB) in Laos, Thailand, Cambodia, and Vietnam, and empties into the South China Sea.

T.B. Wild (✉) · D.P. Loucks
School of Civil and Environmental Engineering, Cornell University, Ithaca, NY 14853, USA
e-mail: tbw32@cornell.edu

D.P. Loucks
e-mail: DPL3@cornell.edu

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_12

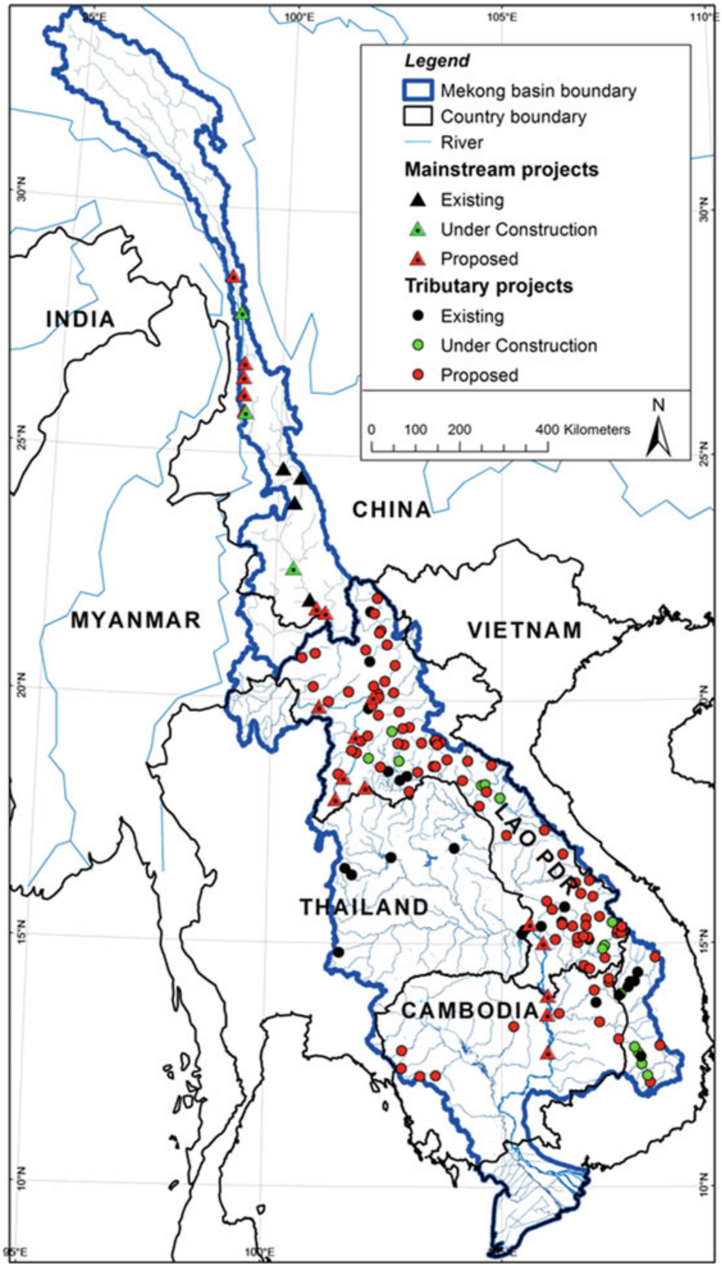


Fig. 1 Map of the Mekong River Basin, showing dams that are existing, proposed or under construction [map borrowed from Cochrane (2010), based on data from MRC (2010)]

Prior to reservoir development, the river's 800,000 km^2 watershed delivered approximately 160 million metric tons (Mt) of suspended sediment per year into the South China Sea (Milliman and Meade 1983). Half of the basin's annual sediment load is estimated to be generated in the Upper Mekong Basin (China), thus the remaining half is generated in the LMB (Clift et al. 2004). The construction of dams in China on the Lancang River is expected to trap much of the 80 Mt generated annually there (Lu and Siew 2006; Fu and He 2007; Kummu and Varis 2007). Much of the 80 Mt/year of sediment yield in the LMB could be trapped in dams that are now being built or are planned (Kummu et al. 2010; Kondolf et al. 2014). Data from recent LMB sediment monitoring efforts from 2009 to 2013 suggest the river is currently transporting only 72.5 Mt/year on average rather than 160 Mt/year (Koehnken 2014).

While the river has remained largely unaltered for much of its history, 42 dams have recently been built, 27 are under construction, and 77 more are currently planned (Mekong River Commission (MRC) 2014). The extent of river basin development planned to occur warrants an evaluation of (1) the potential impact of the planned dam development on the temporal and spatial distribution of ecologically important water and sediment, and (2) the potential for reservoir sediment management techniques to improve sediment passage at dams, as well as the losses in hydropower production that may accompany any measures taken to increase sediment passage.

Given all this infrastructure development activity, the Mekong basin is currently a hotbed of water resources modeling and analysis. Since 1990, many analysts have modeled the hydrology of the Mekong basin to assess potential water-related impacts of dam locations, storage capacities and operating policies (Johnston and Kummu 2012). Conversely, development and application of sediment production, transport, trapping and management models to consider dam-related sediment impacts and management options have been less common. This is primarily because accurately simulating these processes is difficult, and typically requires reliable and extensive data. Even with such data, producing accurate estimates of sediment fate and transport in a river basin over time is difficult. In some parts of the LMB (e.g., the Se San, Sre Pok and Se Kong tributaries area), such data either do not exist, are in the process of being collected (or have only been collected in very recent years). What little data do exist are often not of adequate quality, or are not adequately detailed (e.g., lack of information about grain size distribution) (Walling 2005, 2008; Wang et al. 2011; Irvine et al. 2011; Kummu et al. 2010; Kondolf et al. 2014; Wild and Loucks 2014). Recent sediment monitoring on the mainstream Mekong River should improve modeling efforts in the future (Koehnken 2014). There is also potential for remote sensing to contribute to monitoring of suspended sediment loads in the basin (Heege et al. 2014). Even if more data become available in the future, there still exists the need for a planning tool that can be used despite current data limitations (e.g., only estimates of average annual sediment load), capable of quickly assessing the potential for reservoir sediment management in large networks of reservoirs and river channels.

To satisfy this need for a way to assess various sediment and water management options, we developed a daily simulation screening model, *SedSim*. It was specifically built for use in the Mekong River Basin, though it is generic (data driven) and thus can be applied elsewhere. The model performs a daily time-step mass-balance simulation of flow and sediment that is intended to predict in relative terms the spatial and temporal accumulation and depletion of sediment in multiple river reaches (channels) and reservoirs under different operating and sediment management policies (Wild and Loucks 2012). The model simulates specific sediment management techniques, such as flushing, sluicing, density current venting, bypassing, and dredging. The model is one-dimensional, deterministic, and simulates only one (generic) composite of sediment grain sizes. We have applied it using only an estimate of average annual sediment load. Our application of *SedSim* to the existing and planned reservoirs in the LMB provides water and energy managers in government ministries with a Microsoft Excel-based screening tool that is easily used and modified as desired. The purpose of *SedSim* is to identify the more promising management alternatives that can then be evaluated in more detail with more sophisticated and data-intensive models, as desired.

Other papers have reported on our applications of *SedSim* (Wild and Loucks 2014, 2015), but here we focus more on the details of its methodology. More detail than is given here is available in Wild and Loucks (2012).

2 Model Development: Simulating Flow and Sediment

Numerous studies have indicated a strong correlation between water flow and suspended sediment concentration (SSC) in both large and small, and gaged and ungaged rivers (Milliman and Meade 1983; Morehead et al. 2003). Factors such as relief and lithology may also play important roles in sediment production (Vörösmarty et al. 2003). In keeping with this commonly observed watershed characteristic, *SedSim* assumes that sediment enters the modeled network of reaches and reservoirs at the same exact locations at which water flows enter. The rating curve, based on the power regression of *SSC*, C_s (kg/m^3), on discharge, Q (m^3/s), is given by Eq. 1.

$$C_s = kQ^x \tag{1}$$

In Eq. 1, the 'k' and 'x' are parameters, the determination of which will be discussed shortly. The relationship in Eq. 1 was used for two separate purposes: (1) to generate daily incremental sediment loads at locations in the modeled system at which incremental flows were generated by an external hydrologic model (e.g., SWAT) or other means; and (2) to generate daily sediment loads to be discharged from river reaches (channels), in keeping with the concept that each reach has a 'carrying capacity' to produce suspended sediment as a function of reach discharge. The parameters 'k' and 'x' in Eq. 1 will be referred to as 'c' and 'd', respectively, when discussing the application of this general equation to incremental sediment load generation (see Eqs. 2 and 3). Conversely, the parameters ‘k’ and ‘x’ in Eq. 1 will be referred to as ‘a’ and ‘b’, respectively, when discussing the application of this general equation to sediment routing, or discharge from reaches (see Eqs. 4 and 5).

Ideally, the parameters in Eq. 1 would be established using gage station measurements. Unfortunately, such data are not available in many parts of the Mekong basin. In these circumstances, we propose the following methodology. As was discussed previously, about 80 Mt/year of the Mekong’s sediment load are generated in the LMB. Kondolf et al. (2014) partitioned this 80 Mt/year among nine geomorphic regions, which were delineated based on climatic, geologic, topographic, and tectonic features. Sediment yields (t/km^2-year) were determined by Kondolf et al. (2014) for each region. For example, the Se San, Sre Pok and Se Kong tributary basins lie within two geomorphic provinces: the Kon Tum Massif and the Tertiary Volcanic Plateau, which have estimated yields of 280 and 290 t/km^2-year, respectively. While this annual sediment yield information is useful, *SedSim* simulates using a daily time step. Thus, daily sediment load inputs are required. To accomplish this, sediment is generated on a daily basis with a version of Eq. 1 that has been uniquely calibrated for each incremental input location.

At each such location, *SedSim* calibrates the coefficient value k in Eq. 1 given a specification of an exponent value x. For example, an x value of 1.2 would reflect that proportionally more sediment is transported during higher discharge events, as is often observed in practice (Walling 2009). Referring now to Eq. 2, the model calibrates a c_i value for each incremental input location i such that the mean annual sum of daily incremental sediment loads generated in the unregulated system equals the product of the watershed area that contributes to the incremental flows and the annual sediment yield per unit area (described above) for the input location. In symbolic form, the generated yields will satisfy Eq. 2 for all incremental inflow locations i, as given below.

$$\frac{1}{N}\sum_{t=1}^{T} c_i\left(Q_i^{inc}(t)\right)^{d_i} Q_i^{inc}(t)\Delta t = A_i^{inc} Y_i^{inc} \quad \forall\, i \tag{2}$$

In Eq. 2, T is the simulation duration (in days), N is the average number of simulation years (=T/365), c_i is the parameter being calibrated for location i, d_i is a specified parameter for location i, $Q_i^{inc}(t)$ is the daily incremental flow at location i, Δt is the time step (number of seconds in a one day time step), A_i^{inc} is the watershed area (km^2) that incrementally contributes to location i, and Y_i^{inc} is the average annual sediment yield (Mt/year-km^2) per square km of the incremental watershed. The c_i value for each incremental inflow location i can then be determined with Eq. 3.

$$c_i = \frac{A_i^{inc} Y_i^{inc}}{\frac{1}{N}\sum_{t=1}^{T}\left(Q_i^{inc}(t)\right)^{d_i+1}\Delta t} \quad \forall\, i \tag{3}$$

The model currently assumes that there are no limitations to the sediment supply from the watershed, in that sediment is continually generated as a function of flow without exhausting sediment supply. However, sediment availability in river reaches can be optionally limited. The approach given by Eq. 1 does not account for the possibility that watersheds may exhibit seasonal differences in sediment rating curve parameters.

All sediment that exists within the modeled system, including sediment deposits that existed within the system prior to the start of simulation and the incremental loads that enter the system during simulation, are subject to several transport processes. These transport processes are different for reaches and reservoirs.

For reaches, during a one-day time period, any sediment entering a reach element can either settle (with the possibility of being eroded at a later time), or can be discharged from the reach as the model attempts to satisfy the 'carrying capacity' of the reach. Discussion of Eqs. 1, 2 and 3 focused on incremental sediment load generation. However, *SedSim* permits sediment to be generated from within the system as well. Thus, if no sediment incrementally enters the system from watershed runoff, quantities of sediment could be scoured from reaches to compensate for this input of sediment-deprived water. (We assume sediment can only be generated through scouring processes in reaches, not in reservoirs). To calibrate a separate set of power regression functions (in the form of Eq. 1) to represent the 'carrying capacity' of each reach, a separate calibration is conducted, as given by Eq. 4. In this case, the a_j value (similar to the c_i value in Eqs. 2 and 3) is selected such that the mean annual sum of daily sediment loads discharged from each reach j in the unregulated system is equal to the sum of the mean annual sediment loads generated upstream. Applications of *SedSim* in the literature (e.g., Wild and Loucks 2014, 2015) assume that there are no limitations to the sediment supply in the river bed, but this assumption can be modified if such site-specific information becomes available.

A *SedSim* model of a regulated river consists of both reaches and reservoirs, while an unregulated system consists only of reaches. The unregulated parameter values given by Eqs. 4 and 5 are determined for the locations in the network where reservoirs are sited, treating the unregulated reservoir sites as reaches. These same parameters for reaches in the unregulated system are then stored in the model and are used to determine flow-based sediment discharge from each reach in the regulated system. Thus, we assume the river basin is in relative balance in its unregulated state, exporting approximately what is eroded on an average annual basis (Kondolf et al. 2014). However, because the unregulated system coefficients are maintained for the reaches in the regulated system, alterations of reach flow rates by reservoirs and reduction of sediment availability due to reservoir sediment trapping can both result in significantly altered sediment discharge characteristics as given by Eq. 3.

$$\frac{1}{N}\sum_{t=1}^{T} a_j\left(Q_j^{out}(t)\right)^{b_j} Q_j^{out}(t)\Delta t = \sum_{i\in U}\left(A_i^{inc}Y_i^{inc}\right) \quad \forall j \tag{4}$$

In Eq. 4, T is the simulation duration (in days), N is the average number of simulation years (=T/365), a_j and b_j are the parameters being calibrated for reach j, U is the group of all upstream incremental flow locations i that contribute to the outflow at the outlet of reach j, $Q_j^{out}(t)$ is the daily outflow from reach j, Δt is the time step (one day), A_i^{inc} is the watershed area (km^2) that incrementally contributes to location i, and Y_i^{inc} is the average annual sediment yield (Mt/year-km^2) for the incremental watershed area. The a_j value for each reach at location j can then be determined by Eq. 5.

$$a_j = \frac{\sum_{i\in U}\left(A_i^{inc}Y_i^{inc}\right)}{\frac{1}{N}\sum_{t=1}^{T}\left(Q_j^{out}(t)\right)^{b_j+1}\Delta t} \tag{5}$$

In reservoirs, the inflowing sediment concentration is diminished due to the trapping or settling of sediment in the reduced flow behind the dam. Some fraction of the sediment entering a reservoir is trapped. Sediment that has previously settled in a reservoir can only be removed by simulating a sediment management practice (e.g., flushing or dredging). The trapped fraction, $TE(t, r)$, for each reservoir r in each day t, is determined using the Brune (1953) method. The Brune (1953) method uses data from reservoirs in the United States to predict trapping efficiency as a function of the reservoir's residence time and sediment size. Residence time for each simulation day is determined in *SedSim* using the average total water storage in the reservoir divided by the outflow or release of water from the reservoir. Residence time can change throughout the simulation, as declining storage capacity resulting from sedimentation decreases residence time and therefore trapping efficiency over time. This is an important but often neglected feedback process (Minear and Kondolf 2009). Within the reservoir's storage capacity, the volume occupied by settled sediment mass depends on its bulk density, which *SedSim* users can specify as input data.

While there are other methods for estimating trapping efficiencies (e.g., Churchill 1948), the Brune method has been shown to provide reasonable long-term reservoir trapping efficiency estimates for ponded reservoirs throughout the world (Morris and Fan 1998), and has been applied with success by other researchers in the Mekong basin (Fu and He 2007; Kummu and Varis 2007; Kummu et al. 2010; Kondolf et al. 2014).

Sediment that is not trapped can be discharged in the water that is released during future time periods, or can be removed via one or more sediment management techniques. The volume of sediment deposition in *SedSim* is computed as the ratio of trapped sediment mass to the average bulk density of deposited sediment. The model assumes that sediment that settles in the reservoir is stable. Its volume does not change due to compaction processes.

3 Data Requirements and Suggested External Models

Average daily incremental flow and sediment inputs to reservoirs or river channels can be generated from a separate model with rainfall-runoff modeling capabilities, or by using streamflow and sediment concentration data at a gage site. *SedSim* is then used to simulate reservoir operations, channel routing, sediment production and sediment management. To conduct simulations of reservoir operations with *SedSim*, reservoir and dam characteristics are required, including storage-volume-area relationships for reservoirs, elevation-discharge capacity curves for outlet works, installed power plant capacity, dam height, and tail water level. To simulate daily incremental sediment loads, if no calibration is to be conducted in *SedSim* using Eqs. 2 and 3, input data should include a time series of externally generated daily sediment load inputs. If only average annual incremental sediment load estimates are available, then input data should include average annual sediment load, and an exponent coefficient in Eqs. 2 and 3.

Figure 2 presents a conceptual diagram of the modeling tools that we used in conjunction with *SedSim* in Mekong applications. The Soil and Water Assessment Tool (*SWAT*), which was calibrated for the LMB by the MRC, was used to generate local watershed flows (or incremental flows). Note that any model capable of producing watershed flows and routing them to reservoir sites can be used in conjunction with *SedSim*. Reaches, reservoirs and diversions are connected by junction nodes. Runoff from the watershed, which is generated by *SWAT*, enters the *SedSim* model at select junction nodes (typically the upstream ends of reservoir sites), after which the water instantaneously enters the reach or reservoir that is immediately downstream of the junction. *SedSim* conducts reservoir operations and reach routing procedures, and tracks the accumulation and depletion of sediment in reservoirs and reaches, independently of SWAT. The *RESCON* model is a tool that aids in assessing the feasibility of applying particular reservoir sediment management techniques at particular reservoir sites (Palmieri et al. 2003; Kawashima et al. 2003). While this figure references the specific models that we have used to conduct simulations in the Mekong basin, other models performing similar functions could just as easily be used instead.

If available, *SedSim* users could make use of a separate model (e.g., SWAT) capable of generating daily sediment load inflows to the same locations in the modeled system at which incremental flows are generated. Using this approach, *SedSim* would only be used to route sediment between reservoirs and reaches, and to predict sediment trapping in reservoirs. If such modeled output is not available, *SedSim* offers an alternative approach (using sediment rating curves) that we outlined previously.

When simulating the complexity of sediment management processes and practices, the tendency is to develop and use complex models. We chose to see how effective a relatively simple screening approach could be in situations similar to what we faced in the LMB. To carry out this screening of multiple hydropower reservoirs and sediment management practices motivated our development of

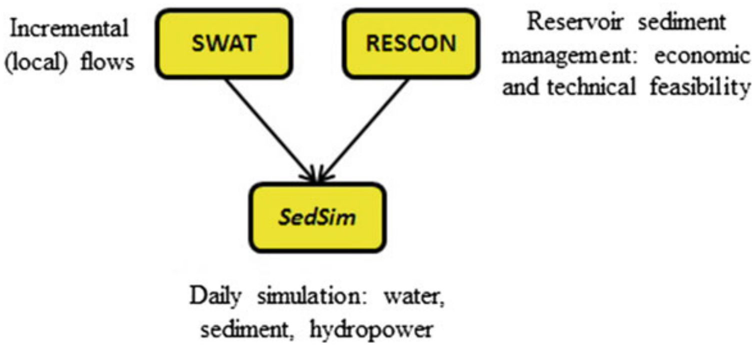


Fig. 2 A schematic demonstrating an example of the modeling tools suggested to be used in conjunction with *SedSim* to assess reservoir sediment management alternatives for a particular dam, including the Soil and Water Assessment Tool (SWAT) for incremental (local) flows and sediment loads if available (though any rainfall-runoff model or streamflow gage will suffice); and the REServoir CONservation (RESCON) model for assessing the technical and economic feasibility of specific sediment management techniques (e.g., flushing, sluicing, bypassing, density current venting, and dredging)

SedSim. There is, of course, a reason why more detailed models may be desired by sediment modelers. Sediment transport during management events like flushing and sluicing is typically very complex, so there is a desire to reproduce as much of that detail as possible in conducting simulations. Such detailed analysis is useful at the level of design of sediment management facilities and reservoir operational strategies. Conversely, *SedSim* is designed for a pre-feasibility, screening level of planning and evaluation of a variety of sediment management options. This is a particularly appropriate level of detail for modeling sediment management in the Mekong basin, because (1) roughly identifying the tradeoffs among hydropower, sediment and flow is an important step before more detailed modeling is conducted; (2) the data required to conduct detailed modeling are not available for many proposed dam sites; and (3) the level of detail offered by *SedSim* can quickly generate information about the tradeoffs associated with sediment management techniques, which complements the pace and level at which development decisions are currently being made.

As an alternative to *SedSim* for simulating reservoir sediment management practices, there exist a variety of detailed sediment transport models, most of which are one-dimensional (1D), just as *SedSim* is, because the elongated geometry of reservoirs are conducive to consideration of only one dimension, and because models that consider more than one dimension require more extensive data and can ultimately be less robust. Within the 1D category, the more detailed, data-intensive models are typically movable boundary models, examples of which include HEC RAS (formerly HEC-6), developed by the U.S. Army Corps of Engineers (1991), and GSTARS, developed by the U.S. Bureau of Reclamation (Molinas and Yang 1986). There are two primary differences between such models and *SedSim*. First, detailed 1-D models (e.g., HEC RAS) assume an interrelationship between channel hydraulics and sediment transport, so there is feedback between the water and sediment components during channel transport. In contrast, *SedSim* ignores such feedbacks. Second, more detailed models have the capability to conduct the approach outlined above for multiple sediment size classes, whereas *SedSim* does not.

4 Predicting Morphologic and Ecosystem Impacts

Ultimately, the assessments of sediment trapping and reservoir sediment management made using *SedSim* can only serve as a rough surrogate for the real metric of interest in many flood-pulse driven river basins such as the Mekong: ecosystem productivity. Sediments, and the nutrients and organics they transport, are responsible for the fertility of the Mekong floodplains, which are responsible for the production of the majority of riverine biomass (Junk et al. 1989). Nutrients drive primary production, including plants such as phytoplankton and periphyton, which in turn drive productivity at higher trophic levels, including fish.

Ideally, it would be possible to directly extend the analysis of sediment management impacts on sediment and hydrology into the ecological domain. Unfortunately, the nature of the relationship between flow, sediment, nutrients and ecosystem health (and ultimately fisheries production) is exceptionally complex in most river basins. In the Lower Mekong Basin a lack of high-quality data, and the complexity of modeling such systems, has hindered the development of models that can explore such linkages (e.g., Arias et al. 2014). Most ecological modeling has relied on geo-spatial analysis and qualitative frameworks (i.e., not numerically-based models) to assess impacts (Johnston and Kummu 2012). More quantitative modeling efforts are needed, but models will be limited until more fundamental information about the functioning of the Mekong system is available.

5 Summary

More than 50 % of the sediment flux in regulated river basins may be getting trapped in reservoirs or other artificial impoundments (Vörösmarty et al. 2003), resulting in a loss in water storage space at a worldwide average rate of 0.5–1 % per year (Mahmood 1987; White 2001). Dams are being rapidly constructed in numerous river basins throughout the world where sediment data are at best sparse, including the Mekong River Basin, yet its management is important. A screening-level sediment modeling approach such as *SedSim* offers the ability to quickly simulate within one simple model the system-wide impact (i.e. systems of many reaches and reservoirs) of sediment management strategies (flushing, sluicing, bypassing, density current venting and dredging) on the mass balances of both sediment and water, as well as hydropower production, within the limitations of existing data.

SedSim permits simulations with limited data. The model can be executed with just an estimate of average annual sediment load instead of requiring a grain size distribution, which is appropriate in the Mekong basin where sediment data are often missing or lacking in quality and detail. *SedSim* is a decision support tool, designed for a pre-feasibility, screening level of water resources systems assessment. Its lack of accuracy compared to more sophisticated and data-intensive models should not detract from its ability to screen alternative operating and sediment management policies. Such analyses are an important part of the decision-making process, useful for identifying the more promising alternatives that can be evaluated in more detail with more sophisticated models.

SedSim was developed specifically to be used by individuals in the water and energy ministries of the Lower Mekong Basin countries, and on computing platforms available to them. It was developed to provide the appropriate ministries of those governments with a simulation tool that is relatively easy to use and modify, using widely available software. This led to its development in Microsoft Excel using the Visual Basic language. We did this to encourage its use by individuals from a variety of backgrounds, as (1) the software is free as long as Excel is available, and (2) Excel is a tool with which many people are familiar.

Acknowledgments This work was performed in conjunction with the Natural Heritage Institute (NHI). The work was funded in part by a grant from the U.S. Agency for International Development (USAID), Cooperative Agreement No. AID-486-A-11-00002, and in part by Cornell University's David R. Atkinson Center for a Sustainable Future (ACSF). The authors wish to thank the Mekong River Commission for providing data and advice throughout our studies, including reservoir and dam characteristics data and streamflow data. We extent our gratitude to George Annandale for helping to improve the *SedSim* model; Matt Kondolf and Zan Rubin for providing us annual Mekong basin sediment load predictions; and to Mauricio Arias, Tom Cochrane and Thanapon Piman for help simulating flows using a SWAT model of the Sre Pok, Se San and Se Kong tributary basins.

References

Arias ME, Cochrane TA, Kummu M, Lauri H, Holtgrieve GW, Koponen J, Piman T (2014) Impacts of hydropower and climate change on drivers of ecological productivity of Southeast Asia's most important wetland. Ecol Model 272:252–263

Brune GM (1953) Trap efficiency of reservoirs. Trans Am Geophys Union 34:407–418

Cochrane T (2010) Mekong Flows Website: Hydropower. http://mekongriver.info/hydropower

Churchill MA (1948) Discussion of "analysis of use of reservoir sedimentation data." In: Gottschalk LC (ed) Proceedings of the federal inter-agency sedimentation conference, Denver, CO, USA, pp 139–140

Clift PD, Graham DL, Blusztajn J (2004) Marine sedimentary evidence for monsoon strengthening, Tibetan uplift and drainage evolution in East Asia. In: Continent and ocean interactions with East Asian Marginal Seas. Geophysical monograph series no. 149, American Geophysical Union, pp 259–282

Fu KD, He DM (2007) Analysis and prediction of sediment trapping efficiencies of the reservoirs in the mainstream of the Lancang River. Chin Sci Bull 52(Supp 2):134–140

Heege T, Kiselev M, Wettle M, Hung NN (2014) Operational multi-sensor monitoring of turbidity for the entire Mekong Delta. Int J Remote Sens 35:2908–2924

Irvine KN, Richey JE, Holtgrieve GW, Sarkkula J, Sampson M (2011) Spatial and temporal variability of turbidity dissolved oxygen, conductivity, temperature, and fluorescence in the lower Mekong River-tonle sap system identified using continuous monitoring. Int J River Basin Manage 9(2):151–168. doi:10.1080/15715124.2011.621430

Johnston R, Kummu M (2012) Water resource models in the Mekong Basin: a review. Water Resour Manage 26:429–455

Junk WJ, Bayley PB, Sparks RE (1989) The flood pulse concept in river-floodplain systems. In: Dodge DP (ed) Proceedings of the international large river symposium (LARS). Canadian Special Publication of Fisheries and Aquatic Sciences, vol 106, pp 110–127

Kawashima S, Johndrow TB, Annandale GW, Shah F (2003) Reservoir conservation volume I: the RESCON approach, economic and engineering evaluation of alternative strategies for managing sedimentation in storage reservoirs. A contribution to promote conservation of water storage assets worldwide. The International Bank for Reconstruction and Development, The World Bank, Washington, DC, USA 52 p

Koehnken L (2014) Discharge sediment monitoring project (DSMP) 2009–2013 summary and analysis of results (final report) MRC/GIZ. Phnom Penh, Cambodia

Kondolf GM, Rubin ZK, Minear JT (2014) Dams on the Mekong: cumulative sediment starvation. Water Resour Res 50. doi:10.1002/2013WR014651

Kummu M, Varis O (2007) Sediment-related impacts due to upstream reservoir trapping, the Lower Mekong River. Geomorphology 85:275–293

Kummu M, Lu XX, Wang JJ, Varis O (2010) Basin-wide sediment trapping efficiency of emerging reservoirs along the Mekong. Geomorphology 119:181–197

Lu XX, Siew RY (2006) Water discharge and sediment flux changes over the past decades in the Lower Mekong River: possible impacts of the Chinese dams. Hydrol Earth Syst Sci 10:181–195

Mahmood K (1987) Reservoir sedimentation: impact, extent, mitigation. World Bank technical report no. 71, Washington, DC

Mekong River Commission (MRC) (2010) State of the basin report. Mekong River Commission, Vientiane, Lao PDR

Mekong River Commission (MRC) (2014) Hydropower project database. Basin development plan programme. Mekong River Commission, Vientiane, Lao PDR

Milliman JD, Meade RH (1983) World-wide delivery of river sediment to the oceans. J Geol 91:1–21

Minear T, Kondolf GM (2009) Estimating reservoir sedimentation rates at large spatial- and temporal-scales: a case study of California. Water Resour Res 45. doi:10.1029/2007WR006703

Molinas A, Yang CT (1986) Computer program user's manual for GSTARS (Generalized Stream Tube model for Alluvial River Simulation). U.S. Bureau of Reclamation, Denver

Morehead MD, Syvitski JP, Hutton EWH, Peckham SD (2003) Modeling the temporal variability in the flux of sediment from ungaged river basins. Global Planet Change 39(1–2):95–110

Morris GL, Fan J (1998) Reservoir sedimentation handbook. McGraw Hill, New York

Palmieri A, Shah F, Annandale GW, Dinar A (2003) Reservoir conservation volume I: the RESCON approach, economic and engineering evaluation of alternative strategies for managing sedimentation in storage reservoirs. A contribution to promote conservation of water storage assets worldwide. International Bank for Reconstruction and Development, The World Bank, Washington, DC, 101002E

U.S. Army Corps of Engineers (1991) HEC-6 Scour and deposition in rivers and reservoirs, user's manual. Hydrologic Engineering Center, Davis, California, USA

Vörösmarty CJ, Meybeck M, Fekete B, Sharma K, Green P, Syvitski JPM (2003) Anthropogenic sediment retention: major global impact from registered river impoundments. Global Planet Change 39(1–2):169–190

Walling DE (2005) Analysis and evaluation of sediment data from the Lower Mekong River. Report submitted to the Mekong River Commission, Department of Geography, University of Exeter

Walling DE (2008) The changing sediment load of the Mekong River. Ambio 37(3):150–157

Walling DE (2009) The sediment load of the Mekong River. In: Campbell IC (ed) The Mekong: biophysical environment of an international river basin. Elsevier, Amsterdam, pp 113–142

Wang JJ, Lu XX, Kummu M (2011) Sediment load estimates and variations in the Lower Mekong River. River Res Appl 27:33–46

White WR (2001) Evacuation of sediments from reservoirs. Thomas Telford, London

Wild TB, Loucks DP (2012) SedSim model: a simulation model for the preliminary screening of sediment transport and management in river basins, version 3.0: documentation and users manual. Department of Civil and Environmental Engineering, Cornell University, Ithaca, NY USA

Wild TB, Loucks DP (2014) Managing flow, sediment and hydropower regimes in the Sre Pok, Se San and Se Kong rivers of the Mekong Basin. Water Resour Res 50:5141–5157. doi:10.1002/2014WR015457

Wild TB, Loucks DP (2015) Mitigating dam conflicts in the Mekong River Basin. In: Hipel KW, Fang L, Cullman J, Bristow M (eds) Conflict resolution in water resources and environmental management. Springer, Heidelberg

Sediment Management on River-Basinscale: The River Elbe

Peter Heininger, Ilka Keller, Ina Quick, René Schwartz and Stefan Vollmer

Abstract All over the world, river basins are under pressure from human activities that affect their chemical and ecological statuses and exhaust available natural resources. Sediment is an essential, integral and dynamic part of the river basins and sediment issues may affect various environmental, social and legal objectives pursued there. Sediment management becomes necessary if the intensity of anthropogenic interventions in the sediment status overwhelms the resilience of ecologic endpoints of the river system or if sediment dynamics and/or sediment status strongly affect human uses. Despite the progress that has been made in the knowledge of sediment management during the last 20 years, practical examples of comprehensive river-basin-scale sediment management concepts are by no means state-of-the-art, and even concepts that focus on only one of the sediment issues are sparse. In Europe, approaches to the management of waters have been radically altered with the introduction of the European Water Framework Directive (WFD). The International Commission for the Protection of the Elbe (ICPER) had declared good sediment quality as one of its key targets. The first Elbe management plan prepared under the WFD (2010–2015) highlights contamination and insufficient hydromorphological conditions as two of the most important supra-regional issues in water resources management. The plan underlines that contaminated sediments and unbalanced sediment conditions are among the main reasons for the failure to meet the WFD management objectives. As a consequence, the member states in the ICPER decided to develop a sediment management concept in preparation for the management cycle from 2016 to 2021. For the first time, an integrated sediment management concept was developed in support of management planning in a large international river basin. The concept is related to the river basin, i.e. it considers cause-effect relations in the entire river basin district Elbe. It combines the issues of

P. Heininger (✉) · I. Quick · S. Vollmer
Federal Institute of Hydrology, Am Mainzer Tor 1, 56068 Koblenz, Germany
e-mail: heininger@bafg.de

I. Keller · R. Schwartz
Free and Hanseatic City of Hamburg, Ministry of Urban Development and Environment, Department of Environmental Protection, ELSA—Remediation of Contaminated Elbe-Sediments, Neuenfelder Straße 19, 21109 Hamburg, Germany

P. Heininger and J. Cullmann (eds.), *Sediment Matters*,
DOI 10.1007/978-3-319-14696-6_13

sediment quantity, hydromorphology, and sediment quality as well as ecological and use-oriented sediment aspects in one concept. The conclusions rely on analyses of risks resulting from an insufficient status of the sediment budget, ecological functions, ecosystem services, and sediment-dependent uses of the river.

1 Introduction

All over the world, river basins are under pressure from human activities that affect their chemical and ecological statuses and exhaust available natural resources. Sediment is an essential, integral and dynamic part of the river basins and sediment issues may affect various environmental, social and legal objectives pursued there. Sediment management becomes necessary if the intensity of anthropogenic interventions in the sediment status overwhelms the resilience of ecologic endpoints of the river system or if sediment dynamics and/or sediment status strongly affect human uses, like flood control, navigation or floodplain agriculture (Apitz 2012). Effective and sustainable management strategies must focus on the entire sediment cycle, rather than on one unit of sediment at a time and/or location. Historically, sediment management was driven by quantity issues (Owens et al. 2005). Sediments were dredged to maintain waterways, or were extracted as a resource (sand, gravel, etc.). Currently, much of the thinking on sediment management and sediment risk assessment is concentrated on sediment quality and on the role of sediments in hydromorphology and ecology. It is the interdependence between the management of sediment quantity and quality that has not been effectively addressed in most assessment and management frameworks (Heise 2009; Heise and Förstner 2007; Owens 2005; SedNet 2003, 2007). While most guidance documents have been generated for specific aspects of sediment management, e.g. for dredged-material management, a basin-scale approach must integrate various sediment goals and provide a universal framework. Different actors (nations, organizations, stakeholders) have different objectives when they address sediments, and a framework must be devised that allows goals and priorities to be balanced in a transparent way (Apitz et al. 2007; Apitz and White 2003). Sustainable sediment management requires careful prioritization of available resources and focuses on efforts to optimize decisions that consider environmental, economic, and societal aspects simultaneously. This may be achieved by combining different analytical approaches such as risk analysis and economic valuation methods (Sparrevik et al. 2011a, b; Von Stackelberg et al. 2002). The inherent uncertainty in predicting ecosystem evolution and response to different management policies requires shifting from optimization-based management to an adaptive management paradigm (Linkov et al. 2006). The objectives of sustainable management of sediment resources at river basin scale are described in detail in two books edited by Heise (2007) and Owens (2008). Effective sediment management requires a holistic approach that takes into account (1) system understanding both in terms of quality and quantity, (2) the integrated management of soil, water, and sediment, (3) upstream-downstream relationships, and (4) supra-regional and transboundary collaboration, (5) an adaptive management approach in order to deal with the always remaining uncertainty and (6) a participatory approach, i.e. involving of stakeholders. While hitherto existing sectorial approaches often tend to think of each sediment issue in relative isolation, and manage these accordingly, each sediment function or use is both dependent on other functions in time and space, and in turn influences many other sediment functions and uses. Sustainable sediment management has to account for this complexity. Thus, if we are to manage sediment for the needs of the environment (e.g. for maintaining habitats) and/or society (e.g. dredging for maintaining navigation), then this always needs to be undertaken with the full awareness of management impacts on nature and society within the river basin. A coherent concept on river basin scale would be the best basis for considering the various functions and uses of sediment operating at different spatial locations within a river basin and operating at different time scales.

In Europe, approaches to the management of waters have been radically altered with the introduction of the European Water Framework Directive (EC 2000). The WFD promotes the integrated management of water resources based on the natural geographical and hydrological units rather than administrative or political boundaries with key objectives to enhance the status of aquatic ecosystems and associated wetlands to a good ecological and chemical status and to prevent any further deterioration of water bodies. It is probably the most significant legislative instrument in the water field that was introduced on an international basis for many years. The WFD represents an enormous opportunity and stimulus to come up with guidance for sustainable sediment management. Reasons for linking sediment management to the WFD are: (1) the 'philosophy' of the WFD provides a platform for river-basin management in terms of policy, institutions, and practical management, (2) sediment concerns the basic WFD objectives, (3) sediments are essential but just a part of the system, (4) sediment management does not stand alone; 'classical' sediment-management actions, e.g. for navigation, flood protection should be evaluated in concert with other management objectives, e.g. for a good environmental status and vice versa, and (5) like WFD issues generally, sediment issues interrelate with many other legislative fields besides the water legislation (Apitz 2012; Apitz and White 2003; Brils et al. 2010; SedNet 2007). The first River Basin Management Plans (RBMPs) prepared under the EU Water Framework Directive have been in process since 2010 and the preparation of the second management cycle (2016–2021) towards a good ecological and chemical status of all European rivers is in progress.

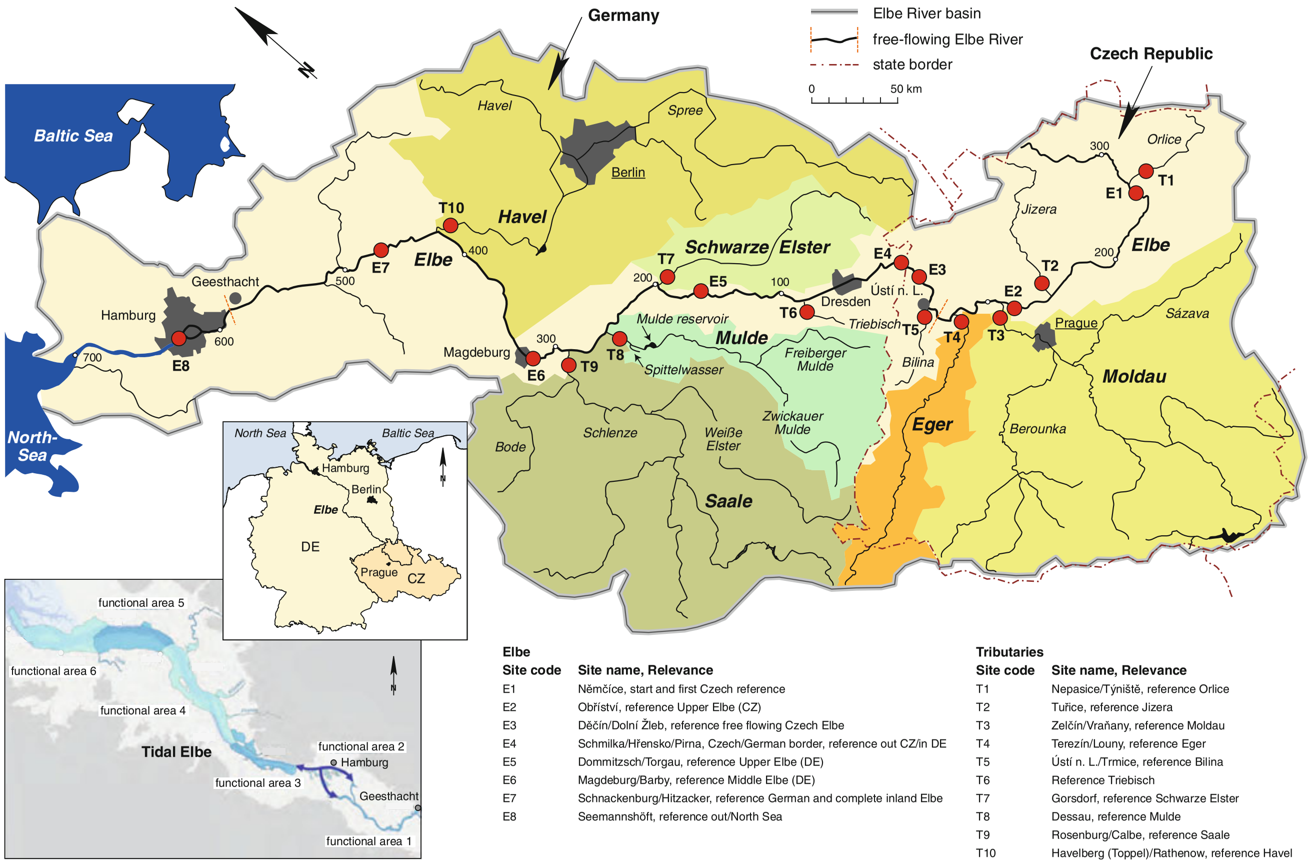
Despite the progress that has been made in the knowledge of sediment management during the last 20 years, practical examples of comprehensive river-basin-scale sediment management concepts are by no means state-of-the-art, and even concepts that focus on only one of the sediment issues are sparse. There are studies focusing on quantitative and hydromorphological aspects for the rivers Rhine and Danube (BMV 1997; Habersack et al. 2010), and the International Sava commission published a practical guidance for the estimation of the sediment balance in the

Sava basin (ISRBC 2013). Strategies for the management of contaminated sediments within the Meuse river system were elaborated by Hakstege et al. (1998). Two studies on the impact of historical contaminated sediments, on the rivers Rhine and Elbe, were initiated by ports and required a practical approach to river-basin management (Heise et al. 2004, 2005, 2008). The International Commission for the Protection of the Rhine adopted a plan which is focused on the risk assessment and management of hot spots of contaminated sediments (ICPR 2009). For the River Elbe, there has not been such a plan on river basin scale yet. The International Commission for the Protection of the Elbe (ICPER) had early declared good sediment quality as one of its key targets (IKSE 1995). In Germany, in the year 2009 the environmental ministers of the Federal States in the Elbe basin declared good sediment quality as a target to be achieved by the year 2027. The first Elbe management plan prepared under the WFD (2010–2015) highlights contamination and insufficient hydromorphological conditions as two of the most important supra-regional issues in water resources management (IKSE 2009). The plan underlines that contaminated sediments and unbalanced sediment conditions are among the main reasons for the failure to meet the WFD management objectives. As a consequence, the member states in the ICPER decided to develop a sediment management concept in preparation for the second management cycle (IKSE 2014). For the first time, a comprehensive sediment management concept was developed in support of management planning in a large international river basin.

2 The International River Basin of the Elbe

2.1 Brief Description of the Catchment

The Elbe is the third largest river in Central Europe, with a length of 1,094 km and a total basin area of 148,268 km^2 (Simon et al. 2005). Figure 1 gives a first general idea of the catchment. The river springs in the Giant Mountains (Krkonoše) in the Czech Republic. Approximately 2/3 of the course of the river lies in Germany and 1/3 in the Czech Republic. Mean annual streamflow rates of the Elbe (Simon et al. 2005) are 313.8 m^3/s at the Czech-German border profile, and 877.3 m^3/s at the last inland gauge before the Elbe is emptying into the North Sea (E4 and E7 in Fig. 1, respectively). While the mean specific river discharge at the last inland gauge of 5.4 l/s km^2 is very low compared with those of other Central European catchments, the ratio of 34.4 between HQ (4,400 m^3/s) and NQ (128 m^3/s) is relatively high. Between 2002 and 2013, the Elbe experienced four extreme floods with maximum river discharges of 3,500 m^3/s or more at the last inland gauge (E7 in Fig. 1). The mean total sediment load of the inland Elbe amounts to 625,000 t/yr at E7 and 200,000 t/yr at E4, respectively. Important tributaries to the Elbe include the rivers Moldau, Mulde, as well as Saale and Havel. Larger parts of the catchment lie in low-mountain regions, and about 60 % of the basin area spans the Middle-German



Elbe

Site code	Site name, Relevance
E1	Němčice, start and first Czech reference
E2	Obříství, reference Upper Elbe (CZ)
E3	Děčín/Dolní Žleb, reference free flowing Czech Elbe
E4	Schmilka/Hřensko/Pirna, Czech/German border, reference out CZ/in DE
E5	Dommitzsch/Torgau, reference Upper Elbe (DE)
E6	Magdeburg/Barby, reference Middle Elbe (DE)
E7	Schnackenburg/Hitzacker, reference German and complete inland Elbe
E8	Seemannshöft, reference out/North Sea

Tributaries

Site code	Site name, Relevance
T1	Nepasice/Týniště, reference Orlice
T2	Tuřice, reference Jizera
T3	Zelčín/Vraňany, reference Moldau
T4	Terezín/Louny, reference Eger
T5	Ústí n. L./Trmice, reference Bilina
T6	Reference Triebisch
T7	Gorsdorf, reference Schwarze Elster
T8	Dessau, reference Mulde
T9	Rosenburg/Calbe, reference Saale
T10	Havelberg (Toppel)/Rathenow, reference Havel

Fig. 1 The Elbe catchment

and the North-German lowlands. Over centuries, the Elbe was morphologically remodelled in wide parts. The streamflow in the Czech section is regulated by reservoirs and lock-and-weir systems, with the latter extending over a stretch of about 200 km. Downstream of the last Czech barrage at Ústí n. L. (Fig. 1), nearly 600 km of the total length of the river are free of barrages but trained by groynes, bank coverings, guide banks, and flood protection dams. This free flowing part of the Elbe ends with another barrage and lock at Geesthacht. The rest of the Elbe of approximately 140 km between Geesthacht and the North Sea is subject to the tides. Also the tidal Elbe has been subject to basic morphological alterations made on behalf of flood protection and navigation. On the entire Elbe, the area of active floodplains and marshes has been dramatically reduced through dyke construction. For example, in the German Middle Elbe the loss amounts to about 75 % (Simon 1996). Today, the Elbe basin comprises a highly developed transboundary European region with a strong economy and very long and intensive industrial and mining traditions. More than 25 million people live in the catchment area, including the major cities of Berlin, Hamburg, Prague, Leipzig, and Dresden. At the same time, around 56 % of the entire catchment area is used intensively for agriculture. However, despite the intensive changes and uses over the centuries, the Elbe stands out in a comparison with other large Central European rivers for its natural resources including its wetland and floodplain forest habitats.

2.2 Sediment Challenges

The sediment budget of a river provides an important measure of its morphodynamics, of the hydrology of its drainage basin, and of erosion and sediment delivery processes operating within its basin. The German reach of the River Elbe suffers each year a deficit of sediment in the order of magnitude of 0.45 million tonnes (IKSE 2014). This corresponds to roughly 100 % of the whole long-term mean annual sediment transport (gravel, sand) that is recorded at the last balancing station on the inland reach (E7 in Fig. 1). On the one hand, this immense deficit is caused by the numerous storage reservoirs, dams, and other flow control structures in the river. Altogether, there are 292 reservoirs in the Elbe basin with a total storage capacity of about 4,000 million m^3. No other large European river basin has such a density of storage basins (Simon et al. 2005). Furthermore, 21 weirs with navigation locks are operated only over the 200 km of the Elbe upstream of Ústí n. L., and consequently the sediment transport in the Czech upper part of the Elbe is completely controlled by these constructions. The total number of smaller and larger flow control structures in the catchment counts by the thousands. Figure 2 gives an impression of this density with the examples of the tributaries Mulde and Saale.

On the other hand, also river training with the resulting enhanced flow intensity and sediment-transport capacity accounts for the sediment deficit. The free-flowing river between Ústí n. L. and Geesthacht first takes course through a low-mountain region with a stable rocky riverbed over about 100 km. Afterwards, when the Elbe

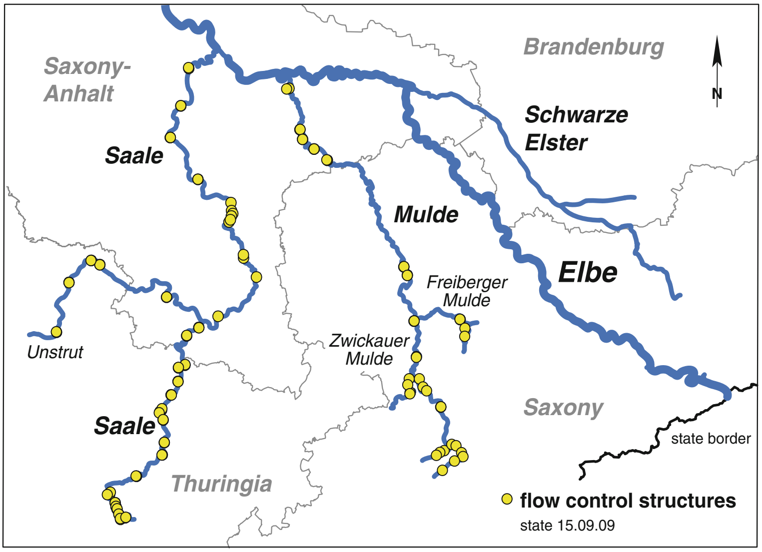


Fig. 2 Flow control structures in the sub-basins of the rivers Mulde and Saale (FGG 2009)

enters the German lowlands, the river is characterized by approximately 6,500 groynes that line the banks. In the lowlands, the bed is loose, with grain sizes of 0.5–2.0 mm. As a consequence of river training, inputs of energy can act only vertically in the direction of the river bed. This encourages depth erosion of the bed. In addition, trapping of coarse-grained sediments that are transported as bedload within groyne fields can significantly reduce the volume of the sediments that are forming the fluvial system (Kesel 2003). The aerial view shown in Fig. 3 exemplifies the relationship between river engineering and the sedimentological and hydromorphological conditions in the free flowing inland reach of the Elbe.

The sediments within the Elbe estuary are a complex mix of particles that have been brought downstream by the river and marine material transported upstream by tidal pumping. The total amount of mobile sediment in the estuary greatly exceeds the annual inputs (Kappenberg and Fanger 2007). The range of the turbidity maximum of about 50 km length illustrates the permanent transport of fine sediment and sediment trapping at the head of the estuary. Within this zone, the total quantity of suspended sediment varies dramatically both during the tidal cycle and between the seasons. Often, layers of high material concentration form near the bottom, within which the most of sediment transport takes place. Like in other tidal rivers, the sediment transport rates over longer terms (months or years) can only be deduced from changes in riverbed bathymetry or from bed erosion. Even then, the validity of the conclusions is limited by the complex topography (Dyer et al. 2000; Jay et al. 1997).

At the 3rd International North Sea Protection Conference in 1990, the Elbe was declared the major sanitation case for the North Sea (Ruchay 1993). What prompted this by no means flattering title was the massive contamination caused above all by

Fig. 3 Aerial survey of the Elbe (DE-km 430–437). *Source* Federal Agency for Cartography and Geodesy (BKG) and Federal Institute of Hydrology (BfG). *Red spots* groyne fields with contaminated fine sediments

inadequately treated industrial and municipal wastewaters and intensive industrial agriculture (Simon 1991). Studies on the Middle Elbe confirmed that in the 1970s and 1980s more than 100 km of the river were almost completely devastated, particularly under low-flow conditions (Guhr et al. 1993). Since 1989, water quality of the Elbe has improved significantly (Adams et al. 2008; Heininger et al. 2003; IKSE 2010). The Elbe is again a "living river" (IKSE 2010) providing habitat, for example, to 112 fish species. This development coincides with the construction of numerous sewage-treatment works and the decline of industries and/or farms. Despite this development, historical pollution from industrial and mining activities as well as present-day point- and non-point emissions are sources of sediment contamination.

Contaminated sediments in the Elbe and in its tributaries act, in turn, as sources of contaminants and may adversely affect the environmental conditions and the uses of the river downstream even to the North Sea. The port of Hamburg is the largest German seaport with an important role for the global trade. As in many other North Sea estuaries, regular maintenance dredging is necessary to maintain safe water depths for navigation to the Port. In the tidal Elbe, 15–20 million m^3 of mainly sandy material need to be dredged annually. Concentrations of several contaminants entering the tidal Elbe from upstream, like pp′-DDT and HCB, are considered as unacceptable risks to the marine environment. As a consequence, a portion of up to 1 million m^3 of dredged material cannot be relocated in the estuary and has to be safely

Fig. 4 Sediment treatment in the City of Hamburg. *Upper section left* reclamation dredger, *right* the 'METHA' plant (MEchanical Treatment of HArbour sediments). *Lower section left* stockpiling of dewatered separated contaminated sediment, *right* silt disposal site Francop/Hamburg. *Sources* Hamburg Ministry of Urban Development and Environment and Hamburg Port Authority

disposed of on land after an industrially organized separation process in the METHA plant (Fig. 4). The additional costs can reach nearly 50 million euros per year.

It is well known that high river discharges and floods make distinct contributions to the downstream transport of suspended matter. This phenomenon is well documented in the River Elbe too (Baborowski et al. 2012; Spott and Guhr 1996). Fine sediments accumulate during low and moderate flows in backwaters of the Elbe and its tributaries such as groyne fields (Schwartz and Kozerski 2003) and lateral structures such as oxbows, floodplain lakes, and harbours (Zachmann et al. 2013). From there, fine sediments together with the associated contaminants are flushed at floodflow (Heininger et al. 1998; Stachel et al. 2006). Particularly the role played by the groynes in the River Elbe in the sediment transport and sediment transfer between the channel and the floodplains, as habitats, and as source of contaminated fine-grained sediments was subject of several studies (Baborowski et al. 2012; Henning and Hentschel 2013; Hillebrand et al. 2012; Ockenfeld and Guhr 2003; Prohaska et al. 2008; Schwartz and Kozerski 2003). The active flood plains are flooded more or less regularly during high water levels. They are considered to be sinks rather than sources of sediment (Krüger et al. 2011, 2014; Vácha et al. 2003). Sedimentation rates of 500–1,500 kg/m^2 year were calculated for areas of medium elevation in front of the dykes in the Middle Elbe River (Schwartz and Kozerski

2003). In a recent study evidence was found for an ongoing relocation of highly contaminated sediments into the Elbe floodplains (Zachmann et al. 2013). The areas along the lower Middle Elbe (downstream of Magdeburg in Fig. 1) have long been used for grazing cattle and sheep. However, due to regular flooding, the grassland has become contaminated by dioxins from the river sediments (Götz et al. 2007), so that restrictions on agricultural use had to be imposed (Kamphues et al. 2011). In order to avoid that dioxin levels exceed the admissible thresholds in animal feed, in milk and in meat, a specific management regime had to be introduced that is causing additional costs.

An opinion poll in the context of efforts towards an adaptive management among 618 stakeholders in the German and Czech parts of the Elbe resulted in the statements that (1) diffuse pollution is among the three greatest management problems and that (2) the estimation of flood risks and the improvement of water quality are among the three most urgent research needs (Hesse et al. 2007). The correlation with sediment quality issues is obvious.

2.3 The View on the Elbe Catchment in the Context of River Basin Sediment Management

The sediment-management concept was compiled to facilitate management planning in the international river-basin district Elbe on a methodologically consistent basis. It is focussed on aspects of the sediment-quality and the sediment budget of supra-regional relevance and omits phenomena of merely local or regional occurrence. Accordingly, the following five components were defined to analyse the Elbe system in terms of river-basin sediment management (cf. Fig. 1), a detailed documentation can be found in (IKSE 2014):

- the impounded inland reach of the Elbe between Němčice and Ústí n. L.;
- the free-flowing inland reach of the Elbe from Ústí n. L. to the impoundment weir at Geesthacht;
- the tidal reach of the Elbe between the weir Geesthacht and the mouth into the North Sea;
- relevant tributaries;
- Reference monitoring sites.

Reference monitoring sites characterize a sub-basin that is relevant for the inter-regional sediment management in qualitative and/or quantitative terms. These stations usually provide long-term time series of data from quality-assured monitoring programmes (IKSE 2014). The risk analysis should rely in any case on the best available data basis. This may mean in certain cases that reference data for quantity and quality are taken from sites slightly differing in their geographical position from each other. A typical example is the reference station E4 in Fig. 1. A station of the international Elbe monitoring programme is situated immediately at the Czech-

German border, where measurements are made in high frequency for suspended sediment quality but not for suspended-solids transport. That is measured about 20 km downstream once per work day in the context of another programme. The data from these two stations are then combined to calculate the loads of particle-bound contaminants. Generally, the sediment management concept was developed using data of the years 2003–2011. During this time dry and wet years occurred with two extreme floods (2006, 2010), one extreme drought (2003) as well as medium discharge conditions. In 2005, nearly average conditions were observed both in terms of discharge and sediment transport. Thus, data of 2005 were occasionally used as a reference.

When relevant tributaries are selected, one distinguishes two categories. The significance of the influence of Category 1 tributaries stems from their basic characteristics. Criteria for the selection are the share in the catchment area (A), streamflow (Q), and the suspended sediment load (S_s). Principally, the criterion of significance is a minimum portion of 10 % in the mean suspended solids load (2003–2008) of the respective reference station downstream of their inflows into the River Elbe. Relevant tributaries of this category are the rivers Orlice, Jizera, Moldau (Vltava), Eger (Ohře), Schwarze Elster, Mulde, Saale, and Havel (Fig. 1). Strictly speaking the Schwarze Elster does not meet the 10 % criterion, but it is counted in this category because it is an important tributary into the Elbe just where the sediment deficit is most severe. Tributaries of Category 2 themselves do not significantly influence the balances of water and solids in the Elbe, but due to their load of (at least one of) the relevant contaminants they contribute significantly to the supra-regional contamination balance. The quantitative criterion for this selection was fixed at a minimum 10 % share in the total load of a contaminant measured at the respective reference station. Relevant rivers of Category 2 are either direct tributaries to the Elbe (Bílina and Triebisch in Fig. 1) or tributaries to rivers of Category 1. For clarity the latter are not explicitly marked in Fig. 1. Totally nine of them were identified, two on the Czech side and seven in Germany. In Fig. 5 the fluxes of suspended sediment and cadmium are shown. The picture illustrates well the specific role that small tributaries of Category 2 can have. While the contribution of the river Triebisch to the sediment load is negligible it contributes significantly to the cadmium balance of the Elbe.

3 Conceptual Framework of the Elbe Basin-Scale Sediment Management

3.1 General Approach to Risk Prioritization

Figure 6 illustrates the main steps towards the sediment-management concept. After the management goals have been defined significant indicators are selected in order to evaluate the status of the system in terms of quantity, hydromorphology, and

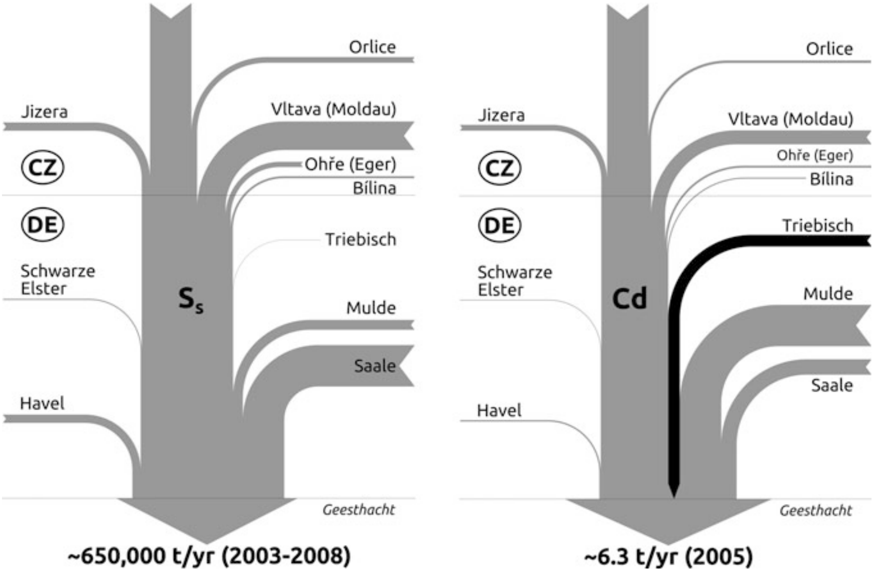


Fig. 5 Material fluxes in the Elbe catchment. S_S suspended sediment, *Cd* cadmium

quality. The risks that arise from the insufficient sediment status for the attainment of the main objectives of the ICPER are analysed. Finally, the significance of these risks is weighted, and recommendations for river basin management planning are derived. Navigation is a use that permanently requires controlling interventions into the sediment regime of a river in order to maintain or restore defined conditions for navigability. That is why this aspect was included in the formulation of the concept from its beginning and can also serve as a model for the integration of other uses of the river. Summarized, the concept complies with the following criteria:

- It is related to the river basin, i.e. it considers cause-effect relations in the river basin district Elbe.
- It is integral because it combines spatial, functional (quantity, hydromorphology, quality) as well as environmental and use-oriented (navigation) sediment aspects in one concept.
- It is risk-based, i.e. its conclusions rely on analyses of risks resulting from an insufficient status of the sediment budget, ecological functions, ecosystem services, and sediment-dependent uses of the river.
- It has a practical orientation, i.e. it was developed in support of river-basin management planning. A collection of proven management practices and technical examples from the basins of the River Elbe and other rivers was compiled in addition to the concept in order to encourage managers to proceed.

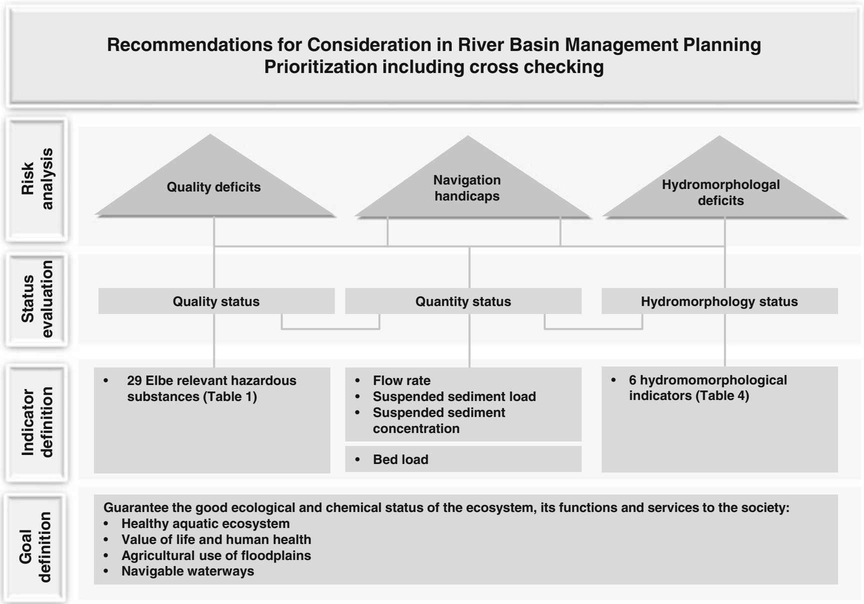


Fig. 6 Conceptual framework

3.2 Quantitative Sediment Conditions

In general, one distinguishes clastic (coarse; >63 µm) and cohesive (fine; <63 µm) sediments. Such a differentiation considers the fact that the fine sediment fraction and the coarse one are subject to different transport processes. Furthermore, the contamination issue is preferentially linked with the fine sediment fraction. The quantitative indicators in this concept were streamflow (Q), the concentration of suspended solids (C_S), and suspended sediment load (S_S). In the inland reach of the River Elbe these are the decisive criteria for the selection of the relevant Category 1 tributaries, and they constitute basic parameters for the risk analyses under the aspects of quality (contaminant transport estimation), hydromorphology, and navigation. Along the inland reach downstream to the weir of Geesthacht they are measured at the reference stations (E and T in Fig. 1) or at the associated reference streamflow gauges (Q). Figure 7 illustrates the variation of the annual suspended sediment load in the course of the River Elbe to the tidal limit at Geesthacht by showing the mean load (2003–2008) and the minima and maxima of this period. In the impounded reach to Ústí n. L., the deficits in transported suspended solids reach in some river stretches, e.g. between Němčice and Obříství, averages in the order of 1,000 and even 10,000 t/yr. Among the tributaries, the rivers Moldau (90,000 t/yr) and Saale (130,000 t/yr) make by far the greatest contribution. For the desired river-basin evaluation it was particularly essential to check the consistency of the

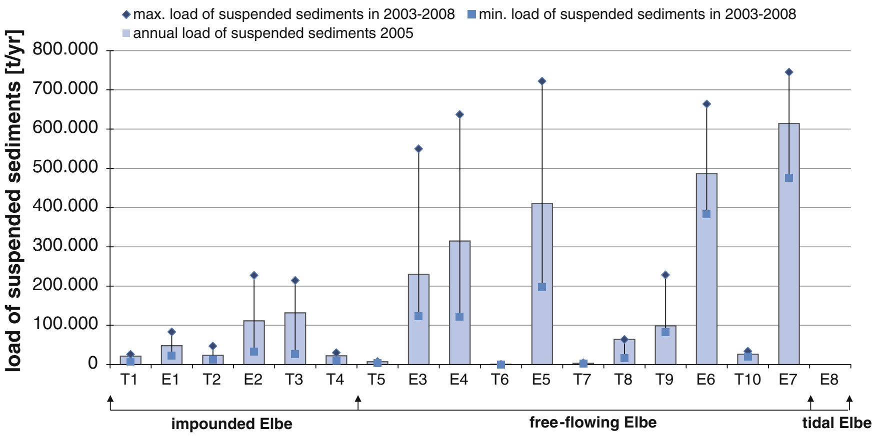


Fig. 7 Suspended sediment loads in the longitudinal section of the Elbe

transboundary sediment transport data. The comparison of the data from each the closest to the common border Czech and German measuring stations revealed a good agreement. The import from the Czech side into the German Elbe is on average around 250,000 t/yr. In the further course of the river, the suspended sediment load increases on average by nearly 400.000 t/yr, so that one can expect that approximately 650,000 t are introduced every year from the inland into the tidal reach of the Elbe. Along the whole 600 km of the free-flowing inland Elbe between Ústí n. L. and Geesthacht, an almost steady increase in the suspended-solids transport in close proportionality with the increase in streamflow is observed (Fig. 7).

Fine sediments that are transported in suspension play a minor role in the total sediment mass balance. The decisive portion is rather the coarser material that is transported along the river bed. Furthermore, there is a close interrelation between sediment balance and riverbed variation in height. The river bed downstream of Ústí n. L. at the first 100 km of the free-flowing inland reach has a stable rock-material bed surface and significant river bed erosion has not been observed in the course of the last more than 100 years (Simon 1996). In contrast to this, studies showed that further downstream the mean bed level has dropped since 1880/1900 with regional variations by maxima of around 2 m (e.g. around Elbe river-km 155 in the German part). Typically, mean erosion rates between 1.0 and 1.25 cm/yr occur in wider parts of the lowland reach. This degradation tendency is continuing on a large-scale and in a long-term perspective. The focus of the erosion regime has shifted in the past decades into the reaches downstream of the inflow of the Schwarze Elster (T7 in Fig. 1). Extreme floods such as those of 2002, 2006, and 2010 may locally contribute significantly to the erosion and profound restructuring of the riverbed.

Solid matter imports into the tidal Elbe come from upstream via the weir of Geesthacht as well as from downstream with the flood tide from the North Sea. The tides periodically change the flow direction of the tidal river reach, and the marine solids that are transported upstream mix with the limnic material coming from the inland reach of the Elbe. The upstream transport of marine fine sediments has significantly increased in the recent past. In the emerging turbidity zone between Elbe river-km 650 and 700, suspended-solids concentrations may be 300 mg/l and more. The absolute amount in the range of maximum turbidity is around 80,000–100,000 t and corresponds to about 15 % of the annual suspended solids import from the river catchment (Kappenberg and Fanger 2007). The marine imports could not be quantified yet. An indicator may be the volumes of sediments dredged. As a rough estimate it was calculated that at the centre of dredging activities downstream of Hamburg, the portion of suspended solids originating from the German Bight of the North Sea makes up 50–80 %, in dependence on streamflow (Ackermann and Schubert 2007; BfG 2008). Upstream of the port of Hamburg (Elbe river-km 610) the marine portion is still between 10 and 40 % (BfG 2008). These complex conditions of sediment quantities in the tidal Elbe are also reflected in high and varying volumes dredged. As for the dry matter of fine sediment, the dredged material amounts to about the 2.5 fold of the mean suspended-solids imported from the inland reach into the tidal Elbe of roughly 650,000 t.

3.3 Qualitative Sediment Conditions and Related Risks

Qualitative indicators are the contaminants that are relevant in the context of sediment management. They are selected in a two-step procedure. According to the general approach to risk prioritization (cf. Sect. 3.1 and Fig. 6), in Step 1 the management goals in the river basin of the Elbe are tested for their sensitivity regarding sediment contamination. The following issues were identified in the context of sediment quality:

- good chemical and ecological status of the waters and integrity of the aquatic community;
- protection of floodplain soils against contamination;
- protection of humans against contaminant uptake.

Then the existing German and Czech regulations (laws, ordinances, guidelines) and pertinent international agreements such as the OSPAR convention were reviewed for their chemical risk requirements. From the resulting pool of chemicals all those were included in the further consideration that are persistent, toxic, bioaccumulative, and adsorptive. Thus, from Step 1 resulted the selection of a set of *potentially* relevant substances. In Step 2, the potentially relevant substances were examined for their relevance in the Elbe basin. This selection was based on data from the reference monitoring stations on the Elbe and the relevant tributaries of Category 1 (Fig. 1) from the period 2003–2008. Finally, 29 contaminants or groups of contaminants were identified, as listed in Table 1. A classification scheme with three classes was established for the gradual application of the indicators in risk analyses: (green) below the lower threshold value C1, (yellow) between the lower and the upper threshold values, and (red) above the upper threshold value C2. The lower threshold is a contaminant-specific formal limit below which—according to the present knowledge and legislation—all management objectives that depend on a good sediment status may be attained without temporal restriction and irrespective of the site. C1 is the lowest value in the row of all in Step 1 identified sediment-quality requirements that are all considered to be equal in rank. The upper threshold C2 is defined by the particular Environmental Quality Standard (EQS; EC 2008) of the respective contaminant following the national implementation of the European WFD. The respective Czech (Sb 2011) and German (OGewV 2011) national regulations are considered equal in rank in the context of the sediment-management concept. They supplement each other and do not have any competing regulations on any of the contaminants. However, there are some contaminants for which none of the two regulations does establish an EQS. In these cases, data from scientific publications (De Deckere et al. 2011; Evers et al. 1996) or national regulations (GÜBAK 2009; RHmV 2009) are applied (cf. Table 1). A detailed description of the approach to selecting the threshold values is given in the concept (IKSE 2014).

The risk analysis (cf. Fig. 8) is made for each of the 29 contaminants that are relevant in sediment management. It was performed on the impounded and in free-flowing reaches of the Elbe, the tidal Elbe, and for the relevant tributaries of categories 1 and 2 in two stages:

1. Evaluation at the sub-basin level to identify the main source areas of particle-bound contaminants. As a result, the qualitative sediment conditions and the particulate contaminant fluxes in the catchment are described.
2. Source-related evaluation within the source areas identified under Stage 1. As a result, the relevant sources in the basin districts are described and ranked.

Stage 1 begins with the annual classification (2003–2011) of suspended sediments at the reference monitoring stations on the River Elbe and the tributaries of Category 1. The resulting large-scale overview on the occurrence of each contaminant allows also to draw conclusions regarding the temporal variations between 2003 and 2011. Figure 9 shows the result with the examples of cadmium (Cd) and hexachlorobenzene (HCB). This qualitative consideration is indispensable for the assessment of the supra-regional relevance and the prioritization of the contaminant fluxes, but it is not sufficient on its own. That is why an evaluation of the load along the course of the River Elbe follows for each relevant contaminant. The result is shown also with the examples of Cd and HCB in Fig. 10. Further, for all contaminants per sub-basin for which C2 was exceeded between 2003 and 2011 at least once, an estimate of the portions of the total annual load at the reference station for the whole inland reach of the Elbe at Schnackenburg was made (% L_{E7}). In order to use the best available data set, in this evaluation loads for heavy metals and arsenic were calculated from the total concentrations (dissolved and particulate fraction) at the reference sites by means of the Eq. (1) and for organic contaminants from the particulate concentrations according to Eq. (2), respectively. A sub-basin was classified as relevant in respect to a certain contaminant for further evaluation in

Table 1 Elbe-relevant contaminants in the context of river-basin sediment management

No.	Contaminant	Unit	Lower threshold (C1)	Upper threshold (C2)	Source of C2
1	Hg	mg/kg	0.15	0.47	Sb (2011)
2	Cd	mg/kg	0.22	2.3	Sb (2011)
3	Pb	mg/kg	25	53	Sb (2011)
4	Zn	mg/kg	200	800	OGewV (2011)
5	Cu	mg/kg	14	160	OGewV (2011)
6	Ni[a]	mg/kg	–	*3*	Sb (2011)
7	As	mg/kg	7.9	40	OGewV (2011)
8	Cr	mg/kg	26	640	OGewV (2011)
9	α-HCH	µg/kg	0.5	1.5	GÜBAK (2009)
10	β-HCH[a]	µg/kg	–	*5*	RHmV (2009)
11	γ-HCH	µg/kg	0.5	1.5	GÜBAK (2009)
12	p,p′DDT	µg/kg	1	3	GÜBAK (2009)
13	p,p′DDE	µg/kg	0.31	6.8	De Deckere et al. (2011)
14	p,p′DDD	µg/kg	0.06	3.2	De Deckere et al. (2011)
15	PCB-28	µg/kg	0.04	20	OGewV (2011)
16	PCB-52	µg/kg	0.1	20	OGewV (2011)
17	PCB-101	µg/kg	0.54	20	OGewV (2011)
18	PCB-118	µg/kg	0.43	20	OGewV (2011)
19	PCB-138	µg/kg	1	20	OGewV (2011)
20	PCB-153	µg/kg	1.5	20	OGewV (2011)
21	PCB-180	µg/kg	0.44	20	OGewV (2011)
22	Pentachlorobenzene	µg/kg	1	400	Sb (2011)
23	Hexachlorobenzene (HCB)	µg/kg	0.0004	17	Sb (2011)
24	Benzo(a)pyrene (BaP)	mg/kg	0.01	0.6	De Deckere et al. (2011)
25	Anthracene	mg/kg	0.03	0.31	Sb (2011)

(continued)

Table 1 (continued)

No.	Contaminant	Unit	Lower threshold (C1)	Upper threshold (C2)	Source of C2
26	Fluoranthene[a]	mg/kg	–	*0.18*	Sb (2011)
27	Sum of five PAH (Σ5PAH)[b]	mg/kg	0.6	2.5	Sb (2011)
28	Tributyltin-Cation[a](TBT)	µg/kg	–	*0.02*	Sb (2011)
29	Sum of dioxins/ furanes (PCDD/F)	ng TEQ/ kg	5	20	Evers et al. (1996)

[a] Only upper threshold value C2 defined yet due to recent legislation specifics
[b] 5 PAH: benzo(a)pyrene, benzo(b)fluoroanthene, benzo(k)fluoroanthene, benzo(g,h,i)perylene, indeno(1,2,3)pyrene

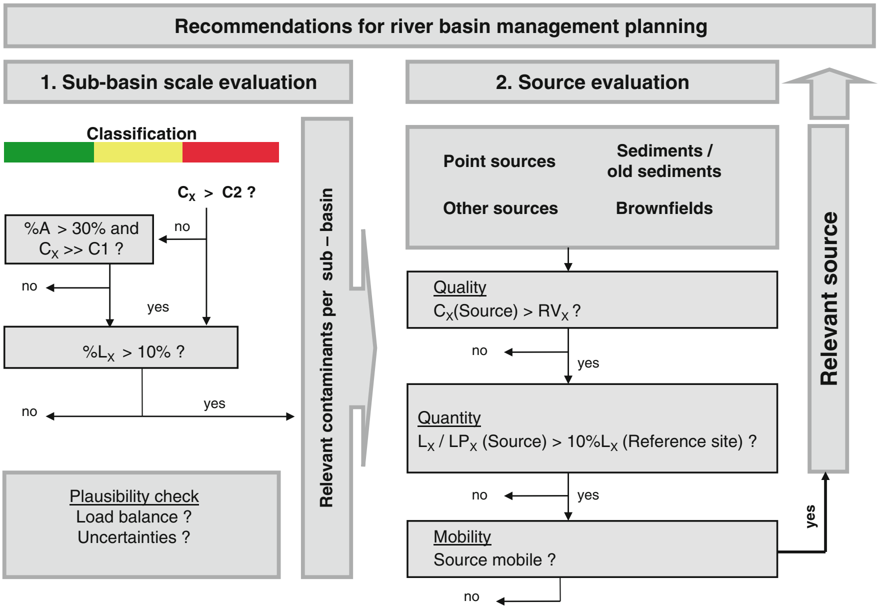


Fig. 8 Scheme of the risk analysis—sediment quality aspect. *A* area of the catchment, *C* concentration, *C1*/*C2* lower/upper threshold value, *L* load, *LP* load potential, *RV* reference value, *X* relevant contaminant

Stage 2 if the portion of L_{E7} at an average (2003–2011) exceeded 10 %. For the tidal Elbe no complete balance of the contaminant transport to the North Sea could be established to date with the available data and models. However, partial fluxes, such as the removal with dredged material or discharges from point sources, can be

(a)

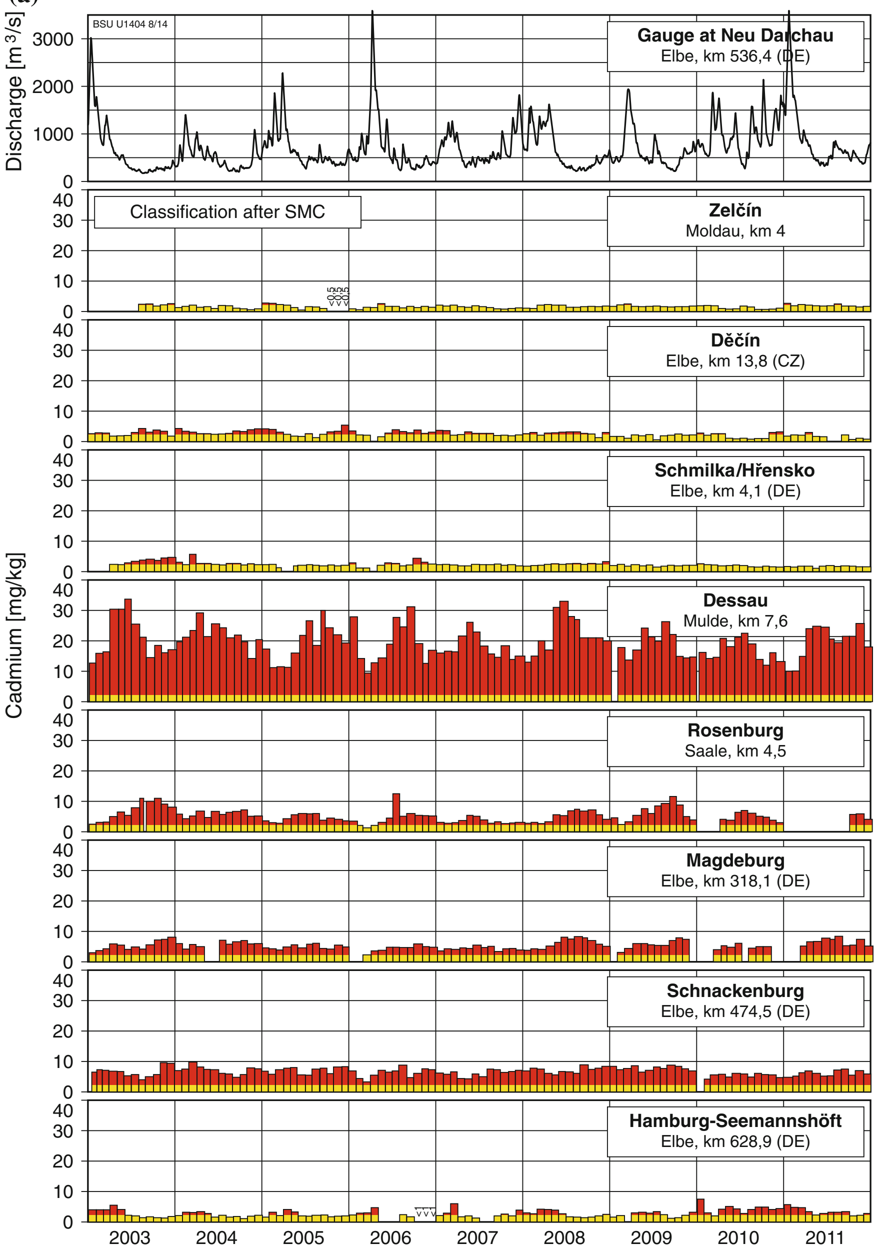


Fig. 9 **a** Space-time diagram of suspended sediment quality, example cadmium. *Red bars* $C_{Cd} > C2_{Cd}$; *Yellow bars* $C1_{Cd} < C_{Cd} < C2_{Cd}$. **b** Space-time diagram of suspended sediment quality, example HCB. *Red bars* $C_{HCB} > C2_{HCB}$; *Yellow bars* $C1_{HCB} < C_{HCB} < C2_{HCB}$. Locations see Fig. 1, threshold values C1, C2 see Table 1

(b)

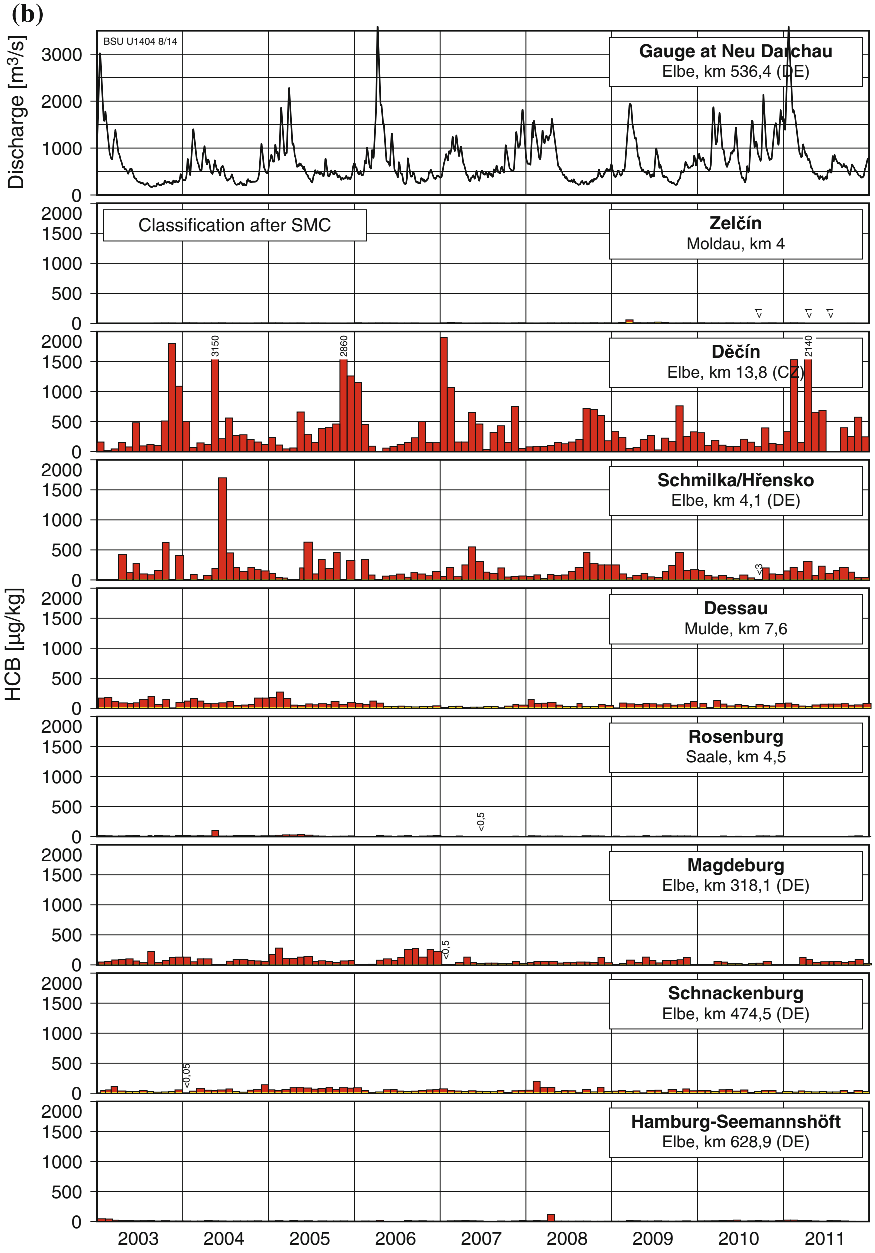


Fig. 9 (continued)

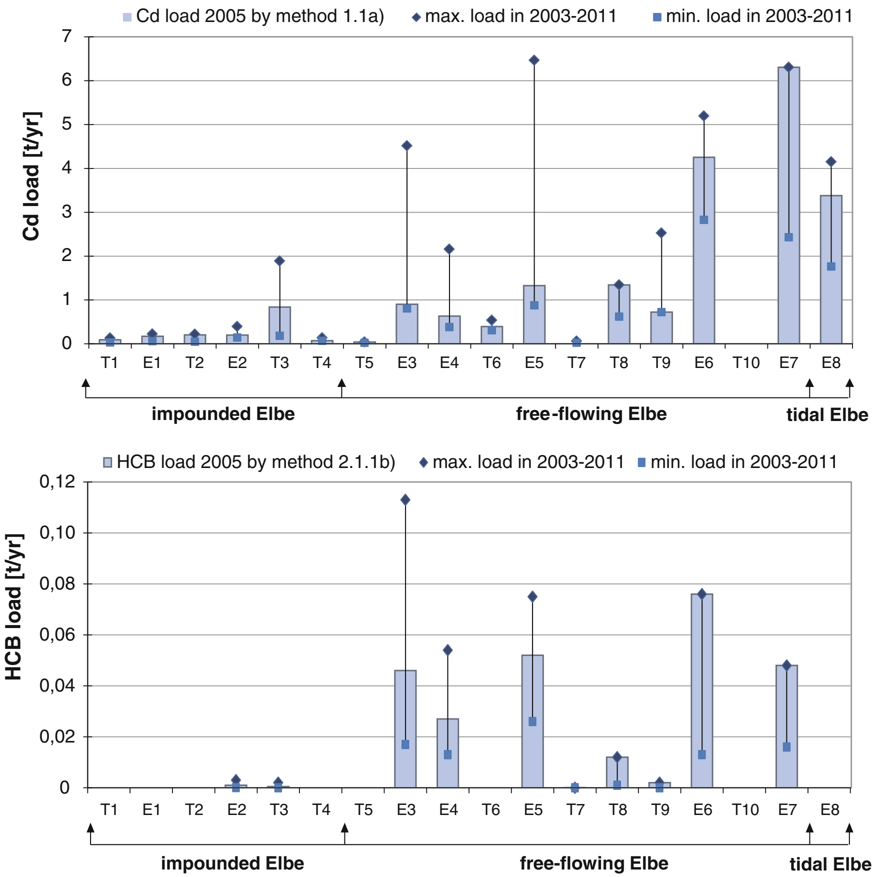


Fig. 10 Contaminant loads in the longitudinal section of the Elbe (*top* Cd, *down* HCB)

quantified without problem. Based on the monitoring data, in the inland river reach from Obříství to Schnackenburg (E2 and E7 in Fig. 1) for a selection of relevant contaminants an annual transport balance was established by Eq. (3). In this balance the inputs from the tributaries of Category 1 were included as well as direct sewage discharges into the River Elbe. Therefrom, conclusions can be drawn on the retention or mobilization of particle-bound contaminants. At ΔL (%) > 0 mobilization dominated, while at ΔL (%) < 0 retention prevailed. It turned out that, for instance, in the German reach of the Elbe between Schmilka and Schnackenburg (E4 and E7 in Fig. 1) heavy retention was observed exclusively during the flood-year 2006, while in all other years between 2003 and 2011 there was a tendency towards mobilization that reached in the maximum between 15 and 50 % depending on the contaminants.

$$L = \frac{MQ_{year} \sum_{i=1}^{n} (C_i \cdot Q_i)}{\sum_{i=1}^{n} (Q_i)} \cdot 0.0864 \cdot 365.25 \quad (1)$$

With: L—load, C_i—total contaminant concentration in mg/l, MQ—mean annual streamflow (calendar year), Q_i—mean daily streamflow

$$L = \sum_{i=1}^{n} C_{i(S)} \cdot L_{i(S,t)} \quad (2)$$

With: L—load, $C_{i(S)}$—contaminant concentration in the suspended sediment collected over a certain time t in mg/kg, $L_{i(S,t)}$—suspended sediment load over the collection time

$$\Delta L(\%) = \left(L_{final} - \sum L_{T+SW}\right) \Big/ L_{E7} \quad (3)$$

With: L—load, final—end of the balanced reach, T—tributary, SW—sewage water, E7—reference site of the whole inland Elbe.

Stage 2 contains the contaminant-specific, source-related risk analysis within the main areas of origin identified in Stage 1. The following types of sources were considered:

- Point sources (sewage water and point discharges from historical mining).
- Sediments/historical sediments. Sediments may be sources or sinks of contaminants. Here, the source function is relevant (mainly induced by floods).
- Historical contaminations such as brownfields or old mining sites in an adjacent zone to the river from which sediment-relevant contaminants are emitted regularly or may be emitted episodically, e.g. due to enhanced streamflow. Potential sources are such contaminated sites within the inundation areas of the River Elbe and its relevant tributaries.
- Other sources such as emissions from urban systems.

Three criteria are applied to estimate the relevance of a source, which must all be met:

1. Minimum concentration. The concentration of at least one relevant contaminant exceeds a threshold that is defined in the context of the respective type of source. In the case of sediments at least one relevant contaminant must exceed the upper threshold (data from the period 2003–2011).
2. Minimum amount of at least one relevant contaminant. The relevance is determined by YES/NO decisions in the context of an expert assessment.
3. The sensitivity to mobilization of the relevant contaminant(s) from that source. The relevance is determined by YES/NO decisions in the context of an expert assessment.

In order to set up the sediment management concept of the River Elbe it was necessary at several points to develop methodologies or to adapt existing ones (FGG 2013; IKSE 2014). This applies particularly to the criteria 2 and 3 above. All the details of the source-related evaluation can hardly be described here but the basic principles will be exemplified in the following. For example, the sensitivity to mobilization is a function of both the material characteristics and the local exposure to hydrological stress. Thus, a methodology was elaborated to identify among the more than 6,000 groyne fields in the German Middle Elbe those with the highest probability of accumulating fine sediments sensitive to mobilization (Hillebrand et al. 2012, 2014). With view to the minimum amount, the criterion is a portion of at least 10 % of the current or potential loads at the respective reference sites. The decision is based either on the determined emission in the case of sewage water or a potential load for the other types of sources (L or LP in Fig. 8). LP is the total amount of a contaminant (in kg or t) per source. In the case of sediments/historical sediments, it can be appropriate from a management point of view to estimate the potential load in terms of spatial units such as sequences of groyne fields in defined river reaches. Figure 3 illustrates this with the example of a group of groyne fields in the Middle Elbe (Hillebrand et al. 2014). The sensitivity to mobilization is assessed in the case of sediments on the basis of laboratory and field measurements of the erosion shear stress and of other parameters that determine the cohesiveness of the material in combination with an estimation of the flood-induced remobilization by means of monitoring data (Schwandt et al. 2014). In the case of historical contaminations in the adjacent zone to the river that are potential sources due to their estimated critical LP, mobilization scenarios and existing documentations are used. They are, on the one hand, based on mandatory assessment procedures which are well established in the Czech Republic and in Germany. On the other hand, these procedures do not account yet for the pathway "sediments in surface waters" and it was necessary to extend them in this respect (FGG 2013).

The evaluation at the river-basin level (Stage 1) found that in the sub-basins of the rivers Jizera, Orlice, Schwarze Elster, and Havel there is no need to perform a source-related risk analysis (Stage 2). Table 2 summarizes the results of the risk analysis from Stage 1 regarding the sub-basins where a source-related analysis had to be performed.

Altogether 38 source-related recommendations for management actions in Germany and in the Czech Republic were derived from the risk analysis and can be found in the concept (IKSE 2014). These final conclusions of Stage 2 are based to a high extent on studies that were launched in support of the elaboration of the concept. Particular issues therein were the function and significance of temporary sinks of cohesive sediments in the rivers Elbe (Heise 2013; Hillebrand et al. 2014; Medek et al. 2014) and Saale (Claus et al. 2014), the role of Elbe floodplains as sinks (Krüger et al. 2014) as well as the relevance and manageability of major sources in the sub-basins of the rivers Mulde (Greif 2013; Jacobs et al. 2013) and Saale (Plejades 2013a, b), and of the sink "Mulde reservoir" (Junge 2013). The conclusions on urban areas as sources of sediment contamination in the River Elbe

Table 2 Results of the risk analysis of suspended sediments—sub-basin-scale evaluation[a]

Sub-basin (reference monitoring site)[b]	%A	Sub-basin-relevant contaminants[c] (criteria: $C > C2$ and $L > 10\ \%$ of L_{E7})
Orlice (T1)	1.4	no
Jizera (T2)	1.4	no
Czech upper Elbe (E2)	9	I: Hg, Cd, Pb, α-HCH, γ-HCH, HCB, BaP, ∑5PAH II: Ni, p,p′DDT, p,p′DDE, 7 PCBs, fluoroanthene
Moldau/Vltava (T3)	19	I: Hg, Pb, BaP, anthracene, ∑5PAH II: Ni, p,p′DDT, p,p′DDE, PCBs, fluoroanthene
Eger/Ohre (T4)	4	I: As II: Ni
Bílina (T5)	0.7	I: As
Czech-German border profile (E4)	35	I: Hg, HCB, BaP, anthracene, ∑ 5PAH, TBT II: Zn, Ni, p,p′DDT, p,p′DDD, p,p′DDE, 7 PCBs, fluoroanthene
Triebisch (T6)	0.1	I: Cd II: Zn
Schwarze Elster (T7)	4	no
Mulde (T8)	5	I: Cd, Pb, As, α-HCH, β-HCH,γ-HCH, HCB, TBT, PCDD/F II: Zn, Ni, p,p′DDT, p,p′DDD, p,p′DDE
Saale (T9)	16	I: Hg, Cd, Pb, α-HCH, γ-HCH, BaP, anthracene, ∑5PAH, TBT, PCDD/F II: Zn, Ni, p,p′DDE, p,p′DDT, fluoroanthene
Havel (T10)	16	no
German *and* International Inland Elbe (E7)[e]	82.4[d]	I: Hg, Cd, Pb, As, α-HCH, β-HCH,γ-HCH, HCB, TBT, PCDD/F II: Zn, Ni, p,p′DDT, p,p′DDD, p,p′DDE, fluoroanthene
Tidal Elbe (E8)[e]	10	I: Hg, Cd, Pb, α-HCH, HCB, ∑5PAH, TBT II: Ni, p,p′DDT, p,p′DDD, fluoroanthene

%A per cent of the total catchment area, *C* mean annual concentration in the suspended sediment at a certain reference monitoring site, *C2* upper threshold value, *L* load, L_{E7} load at reference site E7 (inland Elbe), *I* dangerous priority substances according to the WFD (EC 2008) and substances for which legal regulations had been explicitly set up for the protection of human health, *II* other relevant contaminants

[a] *Data* 2003–2011

[b] Codification see Fig. 1

[c] Contaminants see Table 1

[d] the rest of 7.6 % of A between E7 and the tidal border at Geesthacht is not considered due to lacking specific data

[e] Load criterion not applicable

were drawn from Fuchs et al. (2002, 2010, 2013). Table 3 summarizes important results from Stage 2.

Figure 11 illustrates the whole process of the risk analysis under the aspect of quality once again with the example of Cd. At Stage 1 of the risk analysis (Fig. 8), cadmium was identified to be a relevant contaminant in six sub-basins (cf. Table 2).

Table 3 Relevance of contaminant sources and examples of prioritization and recommendations for action

Source		Summary	Example			
Main type	Sub-type		Source in a sub-basin	Contaminant (I: category 1, II: category 2)	Priority criteria survey (criterion of load >10 % of the reference is met unless the opposite is stated)	Recommendation for action
Point source	Industrial and municipal	Not a specific issue for sediment management	12 largest discharger into the Elbe between E1 and E4	I: Hg, Pb, As, Σ5PAH, II: Zn, Cu	• Load criterion is not met (max. 4.9 % in the sum of 12 dischargers)	Not necessary in terms of sediment management
	Old mining	Relevant issue (major inputs to the Elbe via Mulde, Saale, Triebisch)	historical duct “RothschönbergerStolln” draining into the Triebisch (T6)	I: Cd II: Zn	• Primary source • Levels of difficulty and uncertainty very high • Complete closing impossible	Take measures for minimization of inputs
Brownfield	Industry	Relevant (totally seven major sites with significant inputs directly into the Elbe or via the Mulde and Saale)	SPOLCHEMIE a.s., Usti n.L. discharging to the Czech lower Elbe (E3)	I: Hg, Pb, As, Σ5PAH, II: Zn, Cu	• Close to historical primary source • Levels of difficulty/ uncertainty very high/medium • Complete restoration possible	Take measures for complete closure

(continued)

Table 3 (continued)

Source		Summary	Example			
Main type	Sub-type		Source in a sub-basin	Contaminant (I: category 1, II: category 2)	Priority criteria survey (criterion of load >10 % of the reference is met unless the opposite is stated)	Recommendation for action
	Old mining	Relevant (two major old mining districts in the Mulde)	Old mining district "Freiberg" (T8)	I: Cd, Pb, As II: Cu, Zn	• Close to historical primary source • Levels of difficulty and uncertainty very high • Complete restoration impossible	Take measures for minimization of inputs
Contaminated sediment sites	Groyne field/training structure	Relevant (large depots in exposed training structures in the Czech lower Elbe and in more than 6,000 groyne fields in the German Middle Elbe that are sensitive to flood remobilization)	Training structures in the Elbe downstream T5 until E4	I: Hg, Pb, As, HCB, Σ5PAH, fluoroanthene II: Cu, Ni, DDX, PCBs	• For some contaminants close to the historical primary source in the Bílina (e.g. HCB) for others not (e.g. PCBs) • Medium levels of difficulty and uncertainty • High effect for the free flowing and tidal Elbe	Remove old sediments; verify the conclusions on remobilization potential; verify if a systematic management might be reasonable

(continued)

Table 3 (continued)

Source		Summary	Example			
Main type	Sub-type		Source in a sub-basin	Contaminant (I: category 1, II: category 2)	Priority criteria survey (criterion of load >10 % of the reference is met unless the opposite is stated)	Recommendation for action
	Side structure (backwater, oxbow, harbor)	Relevant (thousands of side structures in the Czech and German parts with huge amounts of contaminated fine sediments sensitive to flood remobilization)	Side structures of the German middle Elbe downstream km 300 (E6 to E7)	I: Hg, Cd, Pb, As, HCHs, HCB, benzo(a) pyrene, Σ5PAH, PCDD/F II: Zn, Cu, DDX, fluoroanthene	• Far from the historical sources • High level of difficulty due to large number • High effect for the tidal reach of the Elbe	Remove old sediments; verify the conclusions on remobilization potential; verify if a systematic management might be reasonable
	Barrage	Relevant (numerous barrages in the Czech part of the Elbe and in the major tributaries Moldau (Vltava) and Saale sensitive to flood remobilization)	Four largest barrages in the Saale river, comprising 80 % of the area of totally twelve (T9)	I: Hg, Cd, Pb, HCHs, TBT, Σ5PAH, B(a) P, anthracene, PCDD/F II: Zn, Cu, DDX, fluoroanthene	• Relatively far from the historical sources • High level of difficulty due to large number • High effect for the middle and tidal reaches of the Elbe	Continue dredging for navigation; verify if the dredging regime for navigation could be extended to a broader fine sediment management

(continued)

Table 3 (continued)

Source		Summary	Example			
Main type	Sub-type		Source in a sub-basin	Contaminant (I: category 1, II: category 2)	Priority criteria survey (criterion of load >10 % of the reference is met unless the opposite is stated)	Recommendation for action
Other sources	From emission-based mass balances in German rivers (Fuchs et al. 2010): diffuse urban emissions, atmospheric deposition, erosion/agriculture, groundwater	Potentially relevant due to high proportions of the overall emissions depending on the pathway and on the contaminant	Diffuse urban emissions (combined sewer overflows and storm sewer outlets) account for about 20 % (Pb), 30 % (Zn) and 40 % (Cu, PAH) of the overall emissions in the German part of the Elbe system	I: Pb, B(a)P, anthracene, Σ5PAH II: Cu, Zn, fluoroanthene	• Primary sources • High level of difficulty and uncertainty, particularly for de-centralized systems • Potential high effect in all parts of the Elbe catchment	Extend knowledge about fine sediment inputs from diffuse urban sources and verify the respective minimization and management opportunities, e.g. by storm water management

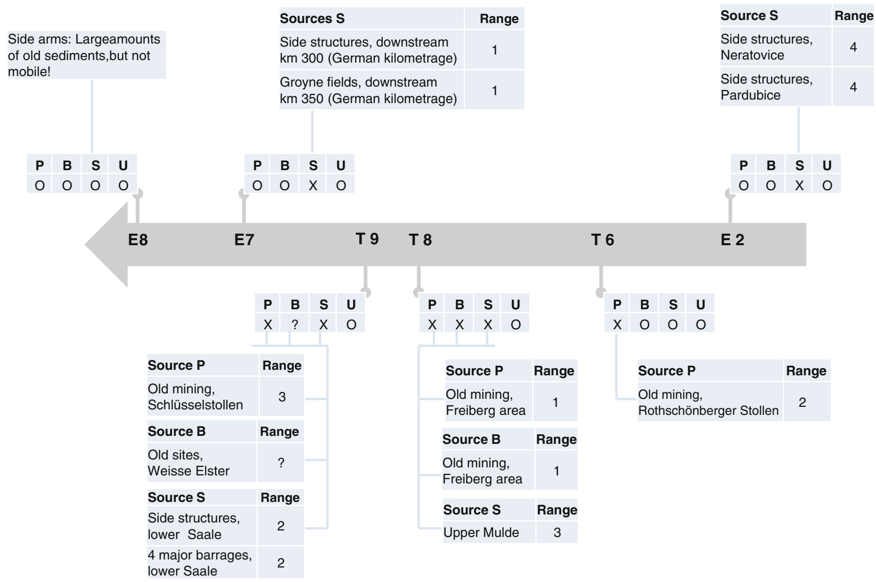


Fig. 11 Source evaluation (example Cd). Source types: *B* brownfield, contaminated site, *P* point source, *S* sediments/old sediments, *U* urban area, *X* relevant, *O* not relevant. Location *E* Elbe monitoring site, *T* tributary monitoring site, *E2* Obříství, *E7* Schnackenburg, *E8* Seemannshöft, *T6* Triebisch, *T8* Mulde, *T9* Saale

These are displayed by E2, E7, and E8 (Elbe) and T6, T8 and T9 (tributaries). The results of Stage 2 are summarized with the small tables in Fig. 11 that are assigned the individual sub-basins. For example, at T8 three types of sources were identified as relevant (X), namely old mining point sources (P), old mining contaminated sites (B), and old sediments (S). The source type 'urban area' (U) does not apply to Cd. In a final step, the specific sources are indicated and ranked by their risks (potential load sensitive to mobilization) within the source type. Thus, the old mining point source of the Freiberg area ranks highest within the three identified relevant sources of this type. Table 3 summarizes in its left hand part the conclusions regarding the different types of sources.

The risk analysis covered not only the sources but also contaminant sinks of supra-regional relevance. Particularly the flood plains play an essential role in the contaminant transport in rivers. This becomes obvious during flood events when river water overtops the banks and flushes suspended solids into the floodplains. The reduced flow velocity in the foreland causes a significant portion of suspended solids to settle and be retained. In this perspective, floodplains are areas of sediment management. For the sediment-management scheme detailed observations and investigations were performed in the German sub-basin (Krüger et al. 2014). It was shown for the period from 2003 to 2008 that, depending on the intensity of the flooding, between 8 % (2004: 2 MQ) and 57 % (2006: extreme event) of the annual mercury transport from the inland Elbe into the tidal reach were deposited into the floodplains of the Middle Elbe downstream of the River Saale. Next to floodplains, natural and artificial river lakes, storage reservoirs, and harbour basins constitute sinks of sediment and thus also of contaminants. For the management concept the potentially most important ones were identified, and the function of the large storage reservoir 'Mulde reservoir' on the middle course of the River Mulde (Fig. 1) was scrutinized more closely (Junge 2013). It could be shown that the retained portion of the mainly particle-bound metals Pb, Cr, Cd, and Cu is under normal flow conditions between 87 and 71 %, and of those with higher portions in solution (Zn, As, Ni) between 50 and 39 %.

3.4 Hydromorphological Conditions and Sediment-Related Risks

The sediment budget of a river is closely connected with its hydromorphology. Weakly developed hydromorphological features are indicators of a disturbed sediment budget. Vice versa, the hydromorphological characteristics of the river have influence on the prevailing sediment conditions (König et al. 2012; Langhammer 2010). On the Czech side, five representative river reaches of altogether 119 km length were subject to pilot studies. On the German inland reach, the whole course from the Czech-German border to the weir of Geesthacht was examined. Furthermore, on the German side the inflows of the tributaries of Category 1 were taken into consideration from the inflow points into the River Elbe upstream to the first weir in the tributary over a cumulative reach of some 95 km as well as the entire tidal reach. One of the Czech pilot stretches was that between Děčín (E3 in Fig. 1) and the Czech-German border, so that the compatibility of the two national approaches could be shown along the entire reach from Děčín to Dresden. The indicators for the assessment of the sediment status under the aspect of the hydromorphology are listed and briefly characterized in Table 4. They are selected in accordance with the approach of the European WFD and the pertinent national regulations.

In the assessment of the individual indicators in the Czech and the German parts of the Elbe catchment a five-level system was used, where in conformity with the WFD the value "1" stands for the best and "5" for the worst assessment level. The assessment methodologies as well as the data and historical data sources are documented in detail in (IKSE 2014) and in the underlying national literature (CZ: Langhammer 2008, 2013; DE: BfG 2011; Quick et al. 2012; Rosenzweig et al. 2012). Of the six selected indicators, four correspond to the hydromorphological component groups 'sediment continuity' and 'river morphology' pursuant to the WFD (EC 2000). In the context of sediment management, the focussing on the sediment budget means, on the one hand, a restriction of the study under the heading "hydromorphology". On the other hand, a necessary extension is performed with the two parameters "sediment balance" and "ratio active floodplain/marsh to

Table 4 Hydromorphological indicators

Indicator	Explanation
Impact on the morphological regime (CZ)	Measure of anthropogenic influences on the natural flow dynamics in the riverbed and the fluvial processes in the riverbed and the floodplain with direct influence on the values of the other indicator-parameters
Mean riverbed changes/sediment balance (DE)	Measure of sedimentation- and erosion processes, characterizes a river system by the development of the sediment budget in the course of time as “sediment deficit”, “sediment excess”, or “balanced” and is dominating regarding the hydraulic coupling or un-coupling of river and floodplain (i.e. active and inactive floodplain)
Sediment continuity (CZ/DE)	Is basically ruled by the presence or absence of flow control structures in the river. Their barrier effect for sediment transport results in upstream backwater effects with sediment accumulation and downstream erosion of the riverbed. Other consequences are modified composition of the bed substrate and changed structural conditions both upstream and downstream of a cross building
Width variation/depth variation (CZ/DE)	Measures of the diversity of habitats. The width variation expresses the ratio between the greatest and the smallest width of the river at a defined streamflow (e.g. bankfull streamflow). The depth variation describes the frequency and the extent of the spatial variation of water depths at mean streamflow in the longitudinal course of the river
Grain-size distribution of the riverbed substrate (CZ/DE)	Fundamental characteristic of the sediment with essential influence on the habitat suitability for vegetation and fauna. Characteristic is the means grain diameter (D_m). In deficit river systems like the Elbe with prevailing erosion there is a tendency towards coarseness, D_m increases in the course of time
Bank stability (CZ)/bank structure (DE)	Measure of the bank enforcement. The indicator “Bank structure” represents the percentage of natural banks along a river. In contrast to engineered banks, pristine or near-natural banks are sources and/or sinks in the sediment budget
Ratio of recent to morphological floodplain/marsh (CZ/DE)	Describes the ratio between the area that can be inundated at present and the area that was originally available for inundation (Holocene floodflow bed). The indicator has high informative power regarding the continuity of water currents and sediment movements in the floodplain and its connection to the fluvial processes

morphological floodplain/marsh” (cf. Table 4). A grave difference regarding indicator parameters between the German and the Czech sides consists in the determination of the indicator “sediment balance” due to the differences in the flow regimes on both sides of the border. While the inland Elbe is free-flowing on the German

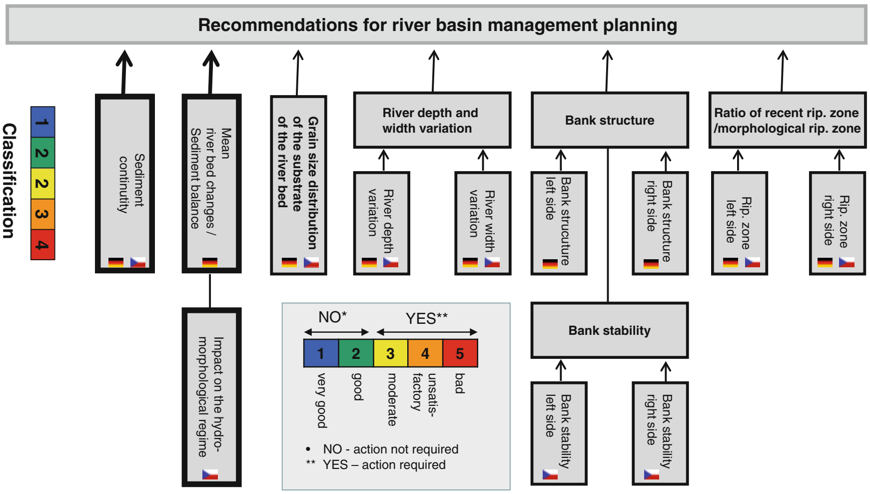


Fig. 12 Scheme of the hydromorphological risk analysis (Rosenzweig et al. 2012; Langhammer 2008)

side, it is widely impounded on the Czech side and the dynamics of the fluvial processes is essentially determined by these impoundments. Therefore, on the German side the indicator "Mean change of the riverbed level/Sediment balance" is used and on the Czech side the "Impact on the hydromorphological regime".

The principle of the risk analysis under the aspect of hydromorphology is illustrated in Fig. 12. The analysis results in a coupling of the assessment of the sediment budget as a part of the hydromorphological status with recommendations to improve the hydromorphological status of the river. Each individual indicator is classified and enters separately into the assessment. No aggregation is performed. Classes 3, 4, and 5 stand for areas that are addressed by the recommendations for actions to improve the sediment budget and the hydromorphological conditions. In Fig. 12, the indicators 'Sediment continuity' and 'Mean river bed changes/Sediment balance (DE)/Impact on the hydromorphological regime (CZ)' are framed bold. These indicators have a key function for the sediment budget. Lacking continuity of sediment and sediment deficits adversely affect also the other hydromorphological indicator-parameters. The two key indicators enter into a first step when recommendations for action are derived. A second step is a check of the other hydromorphological indicator-parameters for synergies which may exist in a combination with Step 1 and whether specific recommendations must be given.

In contrast to the inland reach of the Elbe, the tidal Elbe between Geesthacht and the mouth in the North Sea (Elbe-km 585.9–727.0) does not have the Class 1. Over centuries, the estuary has been subject to basic morphological changes and is today in the terminology of the WFD (EC 2000) designated as a "heavily modified water body" (HMWB). The gradual application of the hydromorphological indicators in the tidal reach follows the principle of the present-day natural potential of water bodies in Germany (*Leitbild*) in four classes in form of expert assessments (FGG 2013). The fundament is the "Integrated management plan for the Elbe estuary" (IMP 2012). It works with functional areas of the Elbe of 20–30 km length each (cf. Fig. 1). Six of them had to be taken into consideration for the sediment-management concept. In the hydromorphological assessment of the functional areas four zones were distinguished and evaluated; if the evaluations of the zones differ, the most unfavourable was the decisive one. The results of the individual indicators were not aggregated. The navigation fairway was considered as the "pelagic zone" of the river (1). The basis was the topographic situation of the present river-training. The "subhydric zone" was considered as the shallow-water zone (2), i.e. the range between the mean tidal low water and the mean tidal low water minus two metres. The "semiaquatic zone" was considered as the intertidal flat (in German: *Watt*), i.e. the area between the mean tidal high water and the mean tidal low water (3). The "semi-terrestrial zone" was the foreland (4) or active marshland, i.e. the height range between the mean tidal high water and the storm-surge/high-tide defence dyke. "Morphological marsh" (relictic marsh) denotes the area within the Pleistocene water course and consists of Holocene alluvial (tidal mud) deposits.

Figure 13 shows for the entire Elbe the result of the risk analysis of the key indicator "Sediment balance/mean river bed changes" or "Impact on the hydromorphological regime" (IKSE 2014). The reaches of less than good quality can be easily identified by the respective colours. Figure 14 presents an overview on the complete results of the hydromorphological assessment of the inland Elbe and points out the role of the two key indicators. Geographically, the diagram starts with the upper of the five Czech pilot reaches and ends at the weir of Geesthacht. The picture is in the very right column completed by the assessment results for the indicator 'sediment continuity' of the four German category 1-tributaries. The causal relationship between riverbed degradation over long reaches as reflected in the assessment results for the indicator 'Sediment balance/mean river bed changes' (Fig. 13) and the critical situation with respect to bank structure, variation in width and depth and of the riparian zone is emphasized by the two blue outlines. It is particularly these reaches of the Middle Elbe which are threatened by an advancing disconnection between river and floodplains due to river bed incision. The black outlines refer to the adverse consequences of the bad sediment continuity. The sediment retention in the whole impounded river section, at the weir Geesthacht on the entrance to the tidal Elbe, and in the major tributaries causes risks in terms of other indicators both above and below the flow control structures.

3.5 The Example of Navigation

Sediment management is an integral part of the maintenance of the River Elbe for navigability. The inland reach of the Elbe is over more than 800 km an important inland waterway, and the tidal Elbe constitutes from the port of Hamburg to the North Sea a well-developed marine waterway for seagoing vessels. The port of Hamburg is

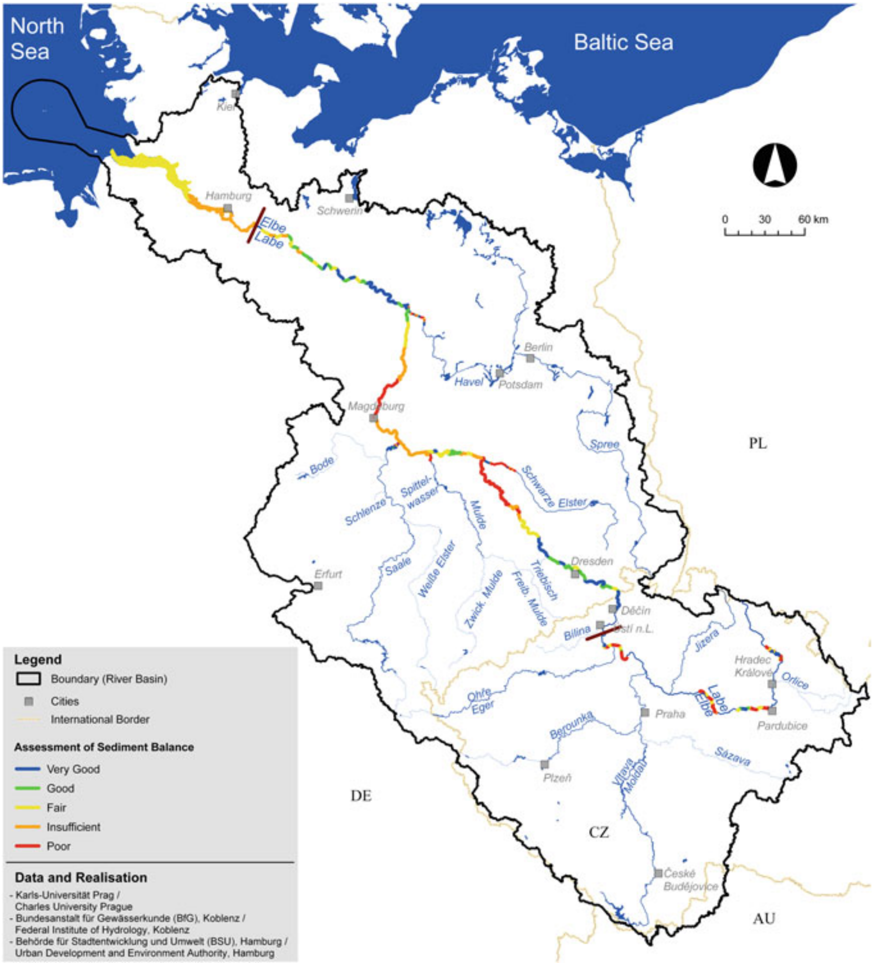


Fig. 13 Risk classification of the key indicator "Mean river bed changes/sediment balance" respectively "Impact on the hydromorphological regime" (IKSE 2014)

Germany's most important sea port and one of the three largest ports in Europe. That is why the aspect of navigation has been integrated into the sediment-management concept from the very beginning. Figure 15 gives an overview on navigation-related sediment-management options for the inland reach of the Elbe as well as on the river sections where these measures are typically applied. Both active measures such as sediment supply and dredging and passive measures such as the construction of groynes and training walls are common and influence the sediment regime of the river. In the tidal Elbe comparable activities take place with a stronger focus on dredging.

The risk analysis from the point of view of navigation consists in a comparison of the situation that actually exists in the river and the maintenance objectives as they are defined in terms of water depths and widths of the navigation channel.

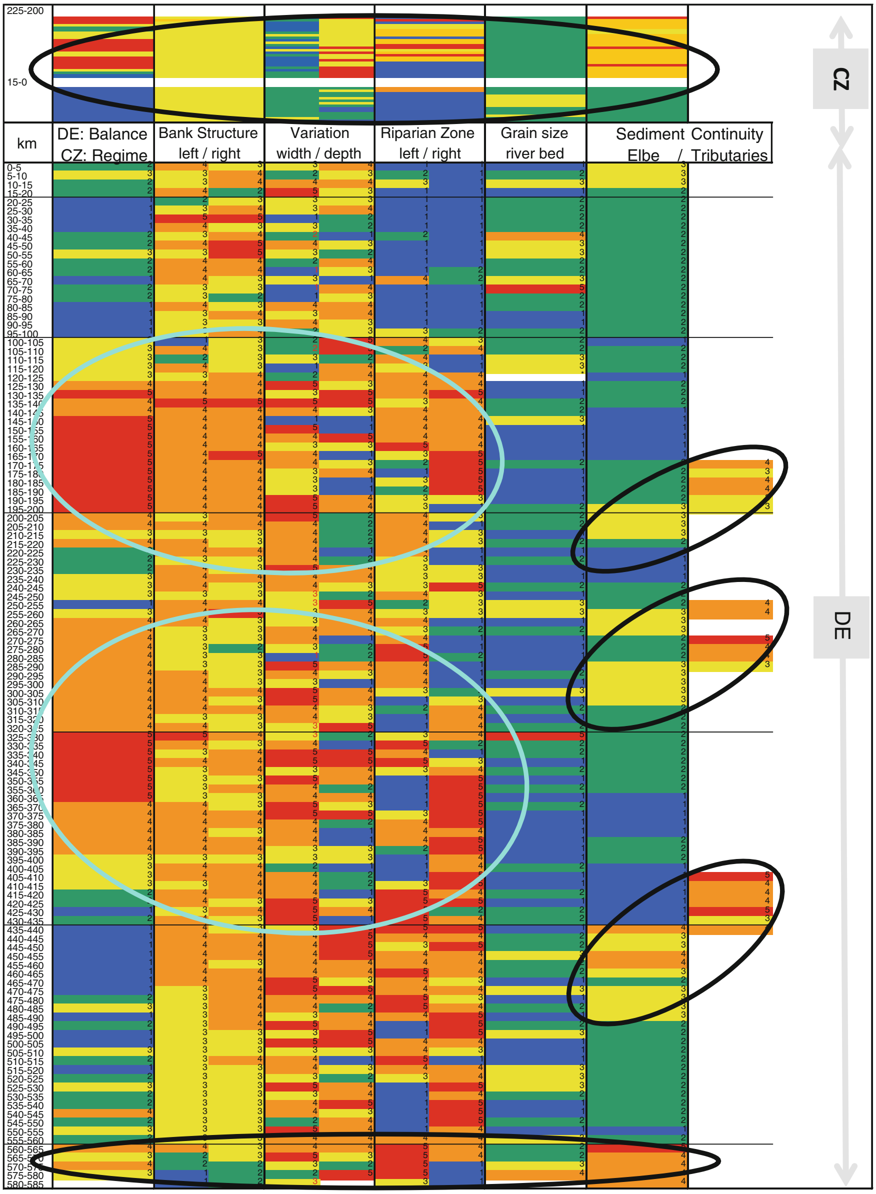


Fig. 14 Hydromorphological risk analysis—overview of the results for the inland Elbe reach. *Blue outlines* reaches of the middle Elbe which are particularly threatened by an advancing disconnection between river and floodplains due to river bed incision. *Black outlines* impact of bad sediment continuity

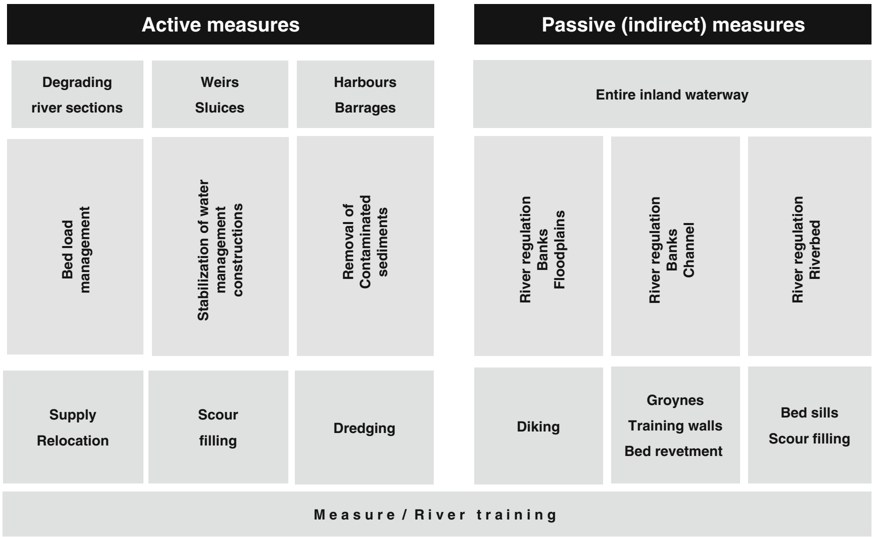


Fig. 15 Sediment management options of navigation in the Elbe inland waters

Depending on the character of the river reach, the main risk from the point of view of navigation differs. As for the impounded reaches of the Elbe and the navigable tributaries Moldau, Saale, and Havel increased sedimentation in zones of reduced flow such as impoundments or forebays of ship locks the stability of waterway constructions may be threatened by massive sediment relocations due to flood events. Accumulated sediments have to be removed immediately what may be a problem both for quantity and quality reasons. In the free-flowing inland Elbe reach the conditions needed for navigation are achieved by means of a regulating system of groynes and training walls that should ensure defined fairway conditions up to mean flow. To this end, the regulating system has to be maintained on a defined level. If necessary, additional active sediment and bedload management is practiced in certain river reaches. Risks for navigation arise, on the one hand, from deficits in the maintenance of the regulating system that does not fulfil its function any more to the required extent. Besides the lack of money also environmental regulations may be important for the dysfunction. Along major reaches of the Elbe, advancing bed incision has consequences for the navigation fairways and the stability of river training constructions. In other reaches, sediment deposits in the fairway reduce the water depth available for navigation. Here, lasting relocation of bedload material is necessary. Finally, in the whole inland reach of the Elbe, it often happens that fine sediments that have to be dredged cannot be relocated within the water, but have to be disposed of on land at higher costs due to their bad quality.

The 15–20 million m^3 of material that has to be dredged annually in the tidal Elbe to preserve the navigable water depth is not an extraordinarily large amount among the estuaries in the North Sea. One can distinguish between wide areas of fine-sediment deposition with mainly (fine) sands drifting in from outside the Elbe channel and local shoals in form of ripples and dunes of medium and coarse sand in the Elbe fairway in the inner estuary (Entelmann and Gätje 2012). A particular challenge has been posed since the year 2000 by the increasing volume of fine sediments in the region of Hamburg with contaminant levels that are too high as that the material could be easily relocated. As described in Sect. 2.2, a costly upland disposal of contaminated sediments has to be practiced that cannot be expanded and is hardly acceptable in a long-term view for economic reasons.

4 Recommendations for the River-Basin Management

4.1 Criteria for Prioritization

Recommendations for the river basin management make up the final step in the sediment-management concept (Fig. 6). Table 5 gives an overview of the criteria for prioritization which had to be defined in this context. While in the upper section aspect-specific criteria are listed, the lower section comprises such of general character. Recommendations given under each of the three aspects have to be assessed also for their effects on the two other criteria ("resonance effect") what is reflected by the general criterion No. 3. The general criteria 1–4 have an upgrading effect, while the criteria 5–7 have a downgrading tendency for the relevance of a management recommendation. The technical conclusion "Absence of appropriate options for solution" is taken only in exceptional cases when the level of knowledge is very well based/substantiated. The economic feasibility of an potential action is not a subject of this concept but it has to be checked in the context of the general management planning, e.g. in the course of establishing plans for the second and the third management cycles pursuant to the WFD (2016–2021 and 2022–2027).

The prioritization under the aspect of quality considers the quantitative importance of a source of sediment pollution (load or potential load, cf. Sect. 3.3) as well as the spectrum of substances concerned. For the purpose of risk prioritization the totally 29 relevant contaminants (cf. Table 1) were differentiated into two groups. Group 1 comprises the priority substances according to the WFD (EC 2008) and substances for which legal regulations had been explicitly set up for the protection of human health, while all others belong into Group 2. Group 1 substances are arsenic, cadmium, mercury, lead, α-, β-, γ-hexachlorocyclohexane, hexachlorobenzene, pentachlorobenzene, benzo(a)pyrene, anthracene, Σ5 PAH, tributyltin, and dioxins/furans.

From the point of view of hydromorphology, it is mainly necessary to identify within the complex system of interactions the dominating active mechanisms and to put them into the focus of the recommendations for actions. The main attention has to be paid to the recovery of natural dynamics of the fluvial processes by reducing anthropogenic interventions into them. The disturbed sediment budget of the Elbe

Table 5 Criteria of prioritization of recommendations for action

Aspect		
Quality	Hydromorphology	Navigation
1. Quantitative significance of a source (load resp. potential load) 2. Number of relevant contaminants of Group I (cf. Table 2) per source 3. Total number of relevant contaminants per source	1. Positive influence on one or both key indicators 2. Positive influence on further indicator-parameters 3. Effect potential for long river reaches 4. Orientation at areas of classes 3, 4, and 5	*Inland Elbe* 1. Maintain, optimize, adapt the regulating system in the free-flowing reaches) and stabilize the riverbed in the longitudinal section and/or the river constructions in the impounded reaches 2. Relocate or add sediment 3. Dredge *Tidal Elbe* 1. Reduce the contaminant import from upstream 2. Establish an adaptive dredged material management
General criteria 1. Solving a problem at source or elimination of the underlying cause 2. If the underlying cause (source) does not exist anymore, the problem should be solved possibly near to the original source 3. The recommendation has positive effect on one or both of the other aspects 4. A single investment causes lower follow-up costs in the long run 5. Degree of difficulty/costs of implementation 6. Safety/uncertainty in the assessment of success, e.g. because of variability of the system		

has to be counteracted rather than symptomatic consequences such as local deficits in the river structure or diversity. Consequently, changes in the key indicators and changes that influence the fluvial processes on a wider scale have highest priority. The two indicator parameters “Sediment continuity” and “Change of the riverbed level/sediment balance” (DE) resp. “Impact on the hydromorphological regime” (CZ) have outstanding importance (cf. Sect. 3.4).

Seen in the perspective of navigation, the priorities from Table 5 differ from river reach to river reach. In the free-flowing parts, an effective system of river training constructions has the highest priority. In the impounded reaches, the stabilization of the riverbed in its longitudinal section and that of engineered river structures rank first. In the tidal Elbe reach, the navigational perspective sees the quality aspects as most relevant in the river-basin context.

4.2 *Options of Risk Management, their Synergies and Conflicts*

In the framework of the sediment-management concept, recommendations for actions are given from each of the three perspectives—quality, hydromorphology, and navigation. From the qualitative perspective, in the concept (IKSE 2014) altogether 38 source-related recommendations are given in the fields of (1) Reduction/restoration of point sources, (2) Reduction/restoration of historical contaminations, (3) Removal of historical sediment deposits sensitive to remobilization, (4) Management of fine sediments in the river combined with the optimization of maintenance strategies, (5) Reduction of imports of contaminated fine sediment from urban areas, and (6) Utilization and management of contamination sinks. In the right part of Table 3 examples are presented from each of the six fields together with a survey of the applied criteria of prioritization and the respective recommendations for action.

As justified before, recommendations for actions in the hydromorphological perspective are primarily directed at the dominating causes of the unsatisfactory situation and thus at the key factors “Sediment continuity” or “Sediment balance/ Hydromorphological regime”. The trends of reduced sediment supply as a result either of retention in the entire river basin, of river training by bank stabilization and of sealing as well as a result of an increased transport capacity of the river due to river-training or dyke construction must be stopped and reversed. In the tidal Elbe, hydromorphologically effective river-training measures should have primary influence on the tidal characteristic with the aim of reducing the “tidal pumping” and thus the upstream transports of fine sediments in the estuary.

From the navigational perspective, actions for the long-term monitoring and stabilization of the riverbed longitudinal section have priority in the impounded inland reach. This requires regular measurements and a continuous active sediment management. In the free-flowing reaches, the regulating system has to be adapted in its regulation parameters in order to ensure again a mostly regulated sediment transport (passive measures, cf. Fig. 15). An active sediment management practice is advisable wherever there are navigation-hindering deposits in the defined fairway channel, e.g. after flood events or as a consequence of a regulation system with restricted functionality. In the tidal Elbe, sediment management for waterway maintenance rests upon three pillars (HPA and WSV 2008). These are (1) an adaptive management of the sediment budget according to the upstream flow conditions, (2) a significant reduction of the contaminant load in sediments from upstream what can only be reached by the entire river basin community, and (3) to take river-training measures.

On the whole, in IKSE (2014) 22 types of recommendations for action are discussed. Table 6 provides two examples from each of the three perspectives (quality, hydromorphology, navigation). The recommendations for action developed from one specific perspective are assessed with view to mutual synergies or conflicts with the two others (Criterion No. 3 in Table 5). The approach is demonstrated in Table 6 as well. Recommendations with positive response of the two other aspects have very high synergy, and those with positive response of one of the two other aspects have high synergy. They are ranked as ‘neutral’ if their implementation would be without grave impacts on the two other aspects. Otherwise, the required evaluation and the possible conflicts are addressed.

Table 6 Management options and their mutual impacts—examples

Recommendation versus response			Conclusion (synergy—conflict)
Quality	Hydromorphology	Navigation	
Recommendation: removal and management of fine contaminated sediments in Elbe side structures downstream E6	*Response*: relief to the river bed particularly with higher discharges; improvement of the river morphology	*Response*: reduced contaminant loads into the reaches downstream and into the tidal Elbe	• Very high synergy due to double positive response
Recommendation: enhancement of fine sediment retention in Elbe floodplains downstream T9	*Response*: deposition may apply not only to fine sediment but also to gravel and sand, thus increasing the sediment deficit downstream. Prognosis difficult due to poor knowledge of underlying processes	*Response*: reduced contaminant loads into the reaches downstream and into the tidal Elbe	• Potential conflict with hydromorphology • Synergy with navigation •Detailed evaluation required
Response: potential conflicts may be avoided if restricted (1) to natural substrates and (2) to predominantly uncontaminated source areas	*Recommendation*: increasing the sediment supply by bed load feeding and dredging and dumping, e.g. by reactivating from the riparian zone	*Response*: may be beneficial for maintaining the navigation channel by steering the measures in terms of location, time, amount	• Synergy with navigation may be reached • Conflict with quality may be avoided
Response: potential conflicts due to expected enhanced mobilization of contaminated fine sediments at least in a transition time; conflicts may be avoided by removal of hot spots before	*Recommendation*: improvement of the sediment continuity (flow control structures; tributaries)	*Response*: potential conflict with maintenance of the navigation channel due to uncontrolled sediment supply by tributaries	• Potential conflict with quality • Potential conflict with navigation • Detailed evaluation required
Response: neutral, potential benefit if measures are combined with the removal of contaminated fine sediments, e.g. from groyne fields	*Response*: potential decrease of the transport capacity	*Recommendation*: maintenance and restoration of the river training system in the free flowing reach of the inland waterway Elbe (E3 to E7)	• Neutral Synergy both with hydromorphology and quality may be reached
Response: potential conflict due to an accelerated transport of contaminated fine sediments to the sea	*Response*: neutral when restricted to the fine fraction	*Recommendation*: scenario-oriented fine sediment management in the tidal Elbe	• Potential conflict with quality • Detailed evaluation required

5 Outlook

The sediment-management concept was developed on the basis of available data and knowledge. Necessarily, some uncertainties will remain in such a complex work. Besides technical- methodical issues that cannot be treated here in detail, the uncertainties refer to very different subject areas such as data, system and process understanding, efficiency control of measures against the background of system variability, and interrelation with regulation areas apart from water management.

Existing monitoring programmes do not yet grant sufficient consideration to the specific issues of sediment management. Flood situations have a special role for sediment transport; especially they are not well covered in their dynamics by regular measuring programmes that are usually tailored to the normal case. The improvement of the databases should be pursued in the course of the adapted monitoring both under normal and extreme conditions.

The knowledge of the interrelations of the system should be developed in the further working process, by special measuring programmes, pilot projects, and applied research projects. This applies to the quantitative and hydromorphological aspects just as well as to the qualitative ones. A better mathematical description of the transport-flow relations, in particular those for floodflow conditions, is required for the better understanding of the sediment transport. The modelling of particle-bound contaminant transport must be linked hereto. Sources and sinks, that have been treated insufficiently or not at all so far, must be included into the overall balance. For the better understanding of the source type "Sediments/old sediments" estimates of masses (potential load) and sensitivity to mobilization must be developed further.

The focuses of the recommendations for actions in terms of quality are (1) on the removal of recent external sources of imports and the improvement of the data situation regarding suspicious areas and (2) on the restoration of deposits of historical sediments and the management of temporary sinks for fine sediments possibly close to the (historical) source. If measures of Category 2 are implemented their efficiency should be reviewed by means of a targeted monitoring on the status of these deposits after their removal and recent external sources should either be closed already or should be closed as soon as possible. Due to the natural variability of the system of sedimentation/remobilisation (cf. Sect. 3.3), an efficiency review of the measures towards better sediment quality can probably be defined only by means of the trend variations over several years.

From the quantitative and hydromorphological perspectives the recommendations for actions focus on the key factors of "Sediment continuity" and "Mean riverbed changes/sediment balance" (DE) resp. "Impact on the hydromorphological regime" (CZ). The efficiency of such measures also with view to other indicators may appear on a short-term, medium-term, and long-term basis. The prospects for success of local measures, e.g. to improve the habitat structure, have to be checked in any case, in a comparison with the key indicators as well as in the supra-regional context and possible combinations of measures. Regarding a spatial assignment of the recommendations for actions on river reaches with similar deficits, a systematic analysis of the impacts and interactions with each other should be performed.

Problematic for integrated sediment management is the current regulatory situation. There is no specific sediment legislative framework, but rather the sediment issue is explicitly or implicitly included in various regulatory contexts. Beside the regulations on water areas such as nature protection, soil, and waste are relevant. Several sediment management goals can be reached only due to measures taken outside of water management. This concerns for example certain diffuse pollution pathways or the import of fine sediment and silting of the river. Other forms of using and shaping the river than navigation like flood defence, the management of floodplains and arable land in general are also touched on by the topic of sediment. These technical and legal aspects have to be accounted for in the further process of management planning, prospects definition and efficiency control.

References

Ackermann F, Schubert B (2007) Trace metals as indicators for the investigation of the dynamics of (suspended) particulate matter in the tidal reach of the river Elbe. In: Förstner U, Westrich B (eds) Sediment dynamics and pollutant mobility in rivers. Springer, Berlin, pp 296–304

Adams MS, Hale B, Gaumert T, Bergemann M, Ohm B, D`Elia C (2008) Environmental history of the German Elbe river-interactions among the ecological, economic, and political environments. Trans Atlantic Institutional Environmental Policy in Relation to Environmental Education (TIES). Madison, Wisconsin

Apitz SE (2012) Conceptualizing the role of sediment in sustaining ecosystem services: sediment-ecosystem regional assessment (secora). Sci Total Environ 415:9–30

Apitz SE, Carlton C, Oen AM, White S (2007) Strategic framework for managing sediment risk at the basin and site-specific scale. In: Heise S (ed) Sustainable management of sediment ressources. Sediment risk management and communication, vol 3. Elsevier, Amsterdam, pp 77–106

Apitz SE, White S (2003) A conceptual framework for river-basin-scale sediment management. J Soils Sed 3(3):132–138

Baborowski M, Büttner O, Morgenstern P, Jancke T, Westrich B (2012) Spatial variability of metal pollution in groyne fields of the middle Elbe—implications for sediment monitoring. Environ Pollut 167:115–123

BfG (2008) WSV-Sedimentmanagement Tideelbe—Strategien und Potenziale—eine Systemstudie. Ökologische Auswirkungen der Umlagerung von Wedeler Baggergut, Koblenz

BfG (2011) Ökologische Modellierungen für die Wasser- und Schiffahrtsverwaltung—das integrierte Flussauenmodell Inform in seiner neuesten Fassung (Version 3). Bundesanstalt für Gewässerkunde, Koblenz

BMV (1997) Sohlengleichgewicht am Rhein. Bestandsaufnahmen und Maßnahmenprogramm. Textband Rheinsohlenerosion Münster/Mainz. Bundesministerium für Verkehr, Bonn

Brils J, Quevauviller P, Slob A, Blind M, Davy T, Carere M, Amorsi N, Brack W, Borchers U, Thompson C, Villesot D (2010) The European water framework directive beyond 2010: let action speak louder than words. J Environ Monit 12:2204–2206

Claus E, Hillebrand G, Becker B, Möhlenkamp C, Heininger P (2014) Die Staustufen der Saale. In: BfG (ed) Fachbeiträge der BfG zum Sedimentmanagementkonzept Elbe. BfG- Mitteilungen Nr. 30. Bundesanstalt für Gewässerkunde, Koblenz. URL: http://doi.bafg.de/BfG/2014/BfG_Mitteilungen_30.2014.pdf

De Deckere E, De Cooman W, Leloup V, Meire P, Schmitt C, von der Ohe PC (2011) Development of sediment quality guidelines for freshwater ecosystems. J Soils Sed 11:504–517
Dyer KR, Robinson MC, Huntley DA (2000) Suspended sediment transport in the Humber estuary. In: Huntley DA, Leeks GJL, Walling DE (eds) Land-ocean interaction. IWA Publishing, Measuring and modelling fluxes from river basins to coastal seas
EC (2000) Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for community action in the field of water policy. European Commission
EC (2008) Directive 2008/105/EC of the European Parliament and of the Council of 16 December on environmental standards in the field of water policy amending and subsequently repealing Council directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council. European Commission
Entelmann I, Gätje B (2012) Sedimentmanagement in der Tideelbe—optimierung von Unterhaltungsstrategien. Magdeburger Gewässerschutzseminar, Hamburg
Evers EHG, Laane RWPM, Groeneveld GJJ (1996) Levels, temporal trends and risk of dioxins and related compounds in the Dutch aquatic environment. Organohalogen Compd 28:117–122
FGG (2009) Ermittlung überregionaler Vorranggewässer im Hinblick auf die Herstellung der Durchgängigkeit für Fische und Rundmäuler im Bereich der FGG Elbe sowie Erarbeitung einer Entscheidungshilfe für die Priorisierung von Maßnahmen. Flussgebietsgemeinschaft Elbe (Ed), Magdeburg, 57 p
FGG (2013) Sedimentmanagementkonzept der FGG Elbe. Vorschläge für eine gute Sedimentmanagementpraxis im Elbegebiet zur Erreichung überregionaler Handlungsziele (Sediment management concept of the German river basin community Elbe. Suggestions for a good sediment management practice in the (German) Elbe catchment to achieve supra-regional management goals). Flussgebietsgemeinschaft Elbe (Ed), Magdeburg, 85 p
Fuchs S, Kiemle L and Kittlaus S (2013) Modelling of fine solids at the river basin scale. In: Proceedings of the Karlsruher Flussgebietstage 2013—international conference on solids in river basins. KIT, Karlsruhe, pp 115–117
Fuchs S, Scherer U, Hillenbrand T, Marscheider-Weidemann F, Behrendt H, Opitz D (2002) Schwermetalleinträge in die Oberflächengewässer Deutschlands. Umweltbundesamt, Berlin, 125 p
Fuchs S, Scherer U, Wander R, Behrendt H, Venohr M, Opitz D, Hillenbrand T, Marscheider-Weidemann F, Götz T (2010) Berechnung von Stoffeinträgen in die Fließgewässer Deutschlands mit dem Modell Moneris. Nährstoffe, Schwermetalle und polyzyklische aromatische Kohlenwasserstoffe. Umweltbundesamt Dessau-Roßlau, 187 p
Götz R, Bauer OH, Friesel P, Herrmann T, Jantzen E, Kutzke M, Lauer R, Paepke O, Roch K, Rohweder U, Schwartz R, Sievers S, Stachel B (2007) Vertical profile of pcdd/fs, dioxin-like pcbs, other pcbs, pahs, chlorobenzenes, ddx, hchs, organotin compounds and chlorinated ethers in dated sediment/soil cores from flood-plains of the river Elbe, Germany. Chemosphere 67 (3):592–603
Greif A (2013) Studie zur Charakterisierung der Schadstoffeinträge aus den Erzbergbaurevieren der Mulde in die Elbe. TU Bergakademie Freiberg. Report for BSU/ELSA Hamburg, 164 p. URL: http://www.elsa-elbe.de/assets/download/fachstudien/Fachstudie-Mulde_ELSA_Greif.pdf
GÜBAK (2009) Preliminary regulation for the dredged material management in coastal waters. Agreement closed between the Federal Republic of Germany and the Federal States of Bremen, Hamburg, Mecklenburg-Western Pomerania, Lower Saxony and Schleswig-Holstein. Federal Ministry of Transport (Ed), Berlin
Guhr H, Büttner O, Dreyer U, Krebs D, Spott D, Suhr U, Weber E (1993) Zusammenstellung, Auswertung und Bewertung des vorhandenen Datenmaterials über die stoffliche Belastung der Gewässergüte der Mittelelbe nach einheitlichen gemeinsamen Kriterien (Vorstudie)—Report for GKSS—Research Centre Geesthacht GmbH, Geesthacht
Habersack H, Jäger E, Hauer C, Schwarz U, Zinke A, Vogel B, Simoner M (2010) Integrative study on hydromorphological alterations on the Danube. Report for EU PLATINA Project swp 5.3 ICPDR

Hakstege AL, Heynen JJM, Eenhoorn JK, Versteeg HP (1998) Strategies for management of contaminated sediments within the Meuse river-system. Wat Sci Technol 37(6–7):419–424

HPA, WSV (2008) River engineering and sediment management concept for the tidal river Elbe. Hamburg Port Authority and Federal Waterways Administration (eds), Hamburg, 39 p

Heininger P, Pelzer J, Claus E, Pfitzner S (2003) Results of long-term sediment quality studies on the river Elbe. Acta Hydrochim Hydrobiol 31(4–5):356–367

Heininger P, Pelzer J, Claus E, Tippmann P (1998) Contamination and toxicity trends for sediments—case of the Elbe river. Wat Sci Technol 37(6–7):95–102

Heise S (ed) (2007) Sustainable management of sediment resources. Sediment risk management and communication, vol 3. Elsevier, Amsterdam

Heise S (2009) Sediments in river basins from the perspective of SedNet, the European sediment network. J Soils Sed 9:393–399

Heise S (2013) Durchführung einer Sondierungsuntersuchung zum Risiko durch eine Schadstoffmobilisierung aus Seitenstrukturen der Elbe. HAW Hamburg. Report for BSU/ELSA Hamburg, 40 p. URL: http://elsa-elbe.de/assets/download/fachstudien/Fachstudie_Seitenstrukturen_ELSA_Heise.pdf

Heise S, Claus E, Heininger P, Krämer T, Krüger F, Schwartz R, Förstner U (2005) Studie zur Schadstoffbelastung der Sedimente im Elbeeinzugsgebiet. Ursachen und Trends. Beratungszentrum für Integriertes Sedimentmanagement, Hamburg, 170 p

Heise S, Förstner U (2007) Risk assessment of contaminated sediments in river basins—theoretical considerations and pragmatic approach. J Environ Monit 9:943–952

Heise S, Krüger F, Baborowski M, Stachel B, Götz R, Förstner U (2008) Bewertung der Risiken durch feststoffgebundene Schadstoffe im Elbeeinzugsgebiet. Beratungszentrum für Integriertes Sedimentmanagement, Hamburg, 349 p

Heise S, Förstner U, Westrich B, Salomons W, Schönberger H (2004) Inventory of historical contaminated sediment in Rhine basin and its tributaries. Final report. Report for Technical University Hamburg in cooperation with the University Stuttgart, Hamburg, 217 p

Henning M, Hentschel B (2013) Sedimentation and flow patterns induced by regular and modified groynes on the river Elbe, Germany. Ecohydrology 6(4):598–610

Hesse C, Krysanova V, Blazkova S, Martinkova M, Koskova R, Möllenkamp S, Gräfe P, Wechsung F (2007) Probleme und Forschungsbedarf im Elbegebiet: Ergebnisse einer Stakeholder-Befragung. Hydrol Wasserbewirtsch 51(2):82–88

Hillebrand G, Becker B, Heininger P, Krämer T, Möhlenkamp C, Pelzer J, Schwandt D, Claus E (2012) Cohesive sediments in groyne fields along the Elbe river. Geophysical Research Abstracts, 14(EGU2012-1803)

Hillebrand G, Claus E, Schwandt D, Becker B, Möhlenkamp C, Heininger P (2014) Die Buhnenfelder der Elbe. In: BfG (ed) Fachbeiträge der BfG zum Sedimentmanagementkonzept Elbe. BfG- Mitteilungen Nr. 30. Bundesanstalt für Gewässerkunde, Koblenz. URL: http://doi.bafg.de/BfG/2014/BfG_Mitteilungen_30.2014.pdf

ICPR (2009) Sediment management plan Rhine. URL: http://www.iksr.org/fileadmin/user_upload/Dokumente_en/Reports/Bericht_175e.pdf)

IKSE (1995) Aktionsprogramm Elbe. International Commission for the Protection of the River Elbe (Ed), Magdeburg

IKSE (2009) Internationaler Bewirtschaftungsplan für die Flussgebietseinheit Elbe nach Artikel 13 der Richtlinie 2000/60/EG. International Commission for the Protection of the River Elbe (Ed), Magdeburg

IKSE (2010) Abschlussbericht Aktionsprogramm Elbe 1996–2010 “Die Elbe ist wieder ein lebendiger Fluss”. International Commission for the Protection of the River Elbe (Ed), Magdeburg

IKSE (2014) Sedimentmanagementkonzept der IKSE. Vorschläge für eine gute Sedimentmanagementpraxis im Elbegebiet (sediment management concept of the ICPER. Suggestions for a good sediment management practice in the Elbe catchment). International Commission for the Protection of the River Elbe (Ed), Magdeburg, 200 p

IMP (2012) Elbe estuary working group. Integrated management plan for the Elbe estuary. URL: http://www.natura2000-unterelbe.de/english.php
ISRBC (2013) Towards practical guidance for sustainable sediment management using the Sava river basin as a showcase—estimation of sediment balance for the Sava river. Report for international Sava river basin commission. http://www.savacommission.org/
Jacobs P, Krautter N, Diesner K (2013) Frachtreduzierung Spittelwasser—Abschlussbericht Oberflächenwasser-, Sediment- und Auenuntersuchungen; Bewertung und Ableitung von Maßnahmeerfordernissen. Report for Landesanstalt für Altlastenfreistellung Magdeburg, 232 p. URL: http://www.laf-lsa.de/fileadmin/Bibliothek/Politik_und_Verwaltung/MLU/LAF/WRRL/Fachbericht_Frachtreduzierung_Spittelwasser.pdf
Jay DA, Uncles RJ, Largier J, Geye WR, Vallino J, Boynton WR (1997) A review of recent developments in estuarine scalar flux estimation. Estuaries 20(2):262–280
Junge FW (2013) Abschlußbericht zum Projekt "Schadstoffsenke Muldestausee—aktuelles Potenzial und jüngste Entwicklung seit 2002". Report for BSU/ELSA Hamburg, 65 p. URL: http://www.elsa-elbe.de/assets/download/fachstudien/Fachstudie_Muldestausee.pdf
Kamphues J, Schulz AJ, Gude K, Bruns-Weller E, Severin K, Appuhn H, Taube V (2011) Investigations concerning the use of dioxin exposed grassland along rivers by food producing ruminants with special regard to food safety. Organohalogen Compd 73:2040–2041
Kappenberg J, Fanger H-U (2007) Sedimenttransportgeschehen in der tidebeeinflussten Elbe, der Deutschen Bucht und in der Nordsee. Report for GKSS Forschungszentrum
Kesel RH (2003) Human modifications to the sediment regime of the lower Mississippi river floodplain. Geomorphology 56:325–334
König F, Quick I, Vollmer S (2012) Defining quantitative morphological changes in large rivers for a sustainable and effective sediment management applied to the river Elbe, Germany. In: Proceedings of the tenth international conference on hydroscience and engineering. Orlando, USA
Krüger F, Scholz M, Kreibich M, Baborowski M (2014) Sedimentrückhalt in den Elbtalauen. Studie zur Erarbeitung des Sedimentrückhaltes in Auen als Teilfunktion des Sedimenttransportgeschehens an der Elbe. Report for BSU/ELSA Hamburg, 52 p. URL: http://www.elsa-elbe.de/assets/download/fachstudien/Fachstudie-Elbauen.pdf
Krüger F, Weniger T, Haensch M, Urban B (2011) Identifizierung von Schadstoff Hot Spots unterschiedlicher Belastungsgeschichte in Auen der Mittelelbe. Paper presented at the annual meeting of DeutscheBodenkundliche Gesellschaft (DBG)
Langhammer J (2008) Hem—metodika pro monitoring hydromorfologických ukazatelů ekologické kvality vodních toků. Report for Přírodovědecká fakulta Univerzity Karlovy v Praze (PřF UK)
Langhammer J (2010) Analysis of the relationship between the stream regulations and the geomorphologic effects of floods. Nat Hazards 54(1):121–139
Langhammer J (2013) Hem-s metodika hodnocení hydromorfologického stavu toku ve vazbě na nakládání se sedimenty. Report for Přírodovědecká fakulta. Univerzity Karlovy v Praze (PřF UK)
Linkov I, Satterstrom FK, Kiker G, Batchelor C, Bridges T, Ferguson E (2006) From comparative risk assessment to multi-criteria decision analysis and adaptive management: recent developments and applications. Environ Int 32:1072–1093
Medek J, Háje, P, Král S, Skořepa J (2014) Sedbila—Bedeutung der Bílina als historische und aktuelle Schadstoffquelle für das Sedimentmanagement im Einzugsgebiet der Elbe. Report for BSU/ELSA Hamburg, 80 p. URL: http://www.elsa-elbe.de/assets/download/fachstudien/Fachstudie_Sedbila_DE.pdf
Ockenfeld K, Guhr H (2003) Groyne fields—sink and source functions of "flow-reduced zones" for water content in the river Elbe (Germany). Water Sci Technol 48(7):17–24
OGewV (2011) Verordnung zum Schutz der Oberflächengewässer (Oberflächengewässerverordnung—OGewV) vom 20. Juli 2011. Bundesgesetzblatt Nr. 37, p 1429. In: German Federal Government (Ed), Bonn
Owens PN (2005) Conceptual models and budgets for sediment management at the river basin scale. J Soils Sed 5(4):201–212

Owens PN (ed) (2008) Sustainable management of sediment resources. Sediment management at the river basin scale, vol 3. Elsevier, Amsterdam, Boston

Owens PN, Batalla RJ, Collins AJ, Gomez B, Hicks DM, Horowitz AJ, Kondolf GM, Marden M, Page MJ, Peacock DH, Petticrew EL, Salomons W, Trustrum NA (2005) Fine-grained sediment in a river system: environmental significance and management issues. River Res Appl 21:693–717

Plejades (2013a) Frachtreduzierung Schlüsselstollen. Ermittlung der Auswirkungen des Schlüsselstollens auf den partikelgebundene Schadstofftransport in der Saale/Elbe. Report for Landesanstalt für Lastenfreistellung Magdeburg, 71 p. URL: http://www.laf-lsa.de/fileadmin/Bibliothek/Politik_und_Verwaltung/MLU/LAF/WRRL/Fachbericht_Frachtreduzierung_Schluesselstollen__Teil_A.pdf

Plejades (2013b) Frachtreduzierung Schlüsselstollen. Bewertung von technisch realisierbaren und verhälnismäßigen Maßnahmen. Report for Landesanstalt für Lastenfreistellung Magdeburg, 28 p. URL: http://www.laf-lsa.de/fileadmin/Bibliothek/Politik_und_Verwaltung/MLU/LAF/WRRL/Fachbericht_Frachtreduzierung_Schluesselstollen__Teil_B.pdf

Prohaska S, Jancke T, Westrich B (2008) Model based estimation of sediment erosion in groyne fields along the river Elbe. IOP conference series: earth and environmental science, vol 4(1), p 012042

Quick I, König F, Svenson C, Cron N, Schriever S, Vollmer S (2012) Hydromorphologische Bewertung und Praxisprojekte mit Schnittstelle zur Gewässerökologie. Hydro-ökologsiche Modellierungen und Ihre Anwendungen—1. Ökologisches Kolloquium. Bundesanstalt für Gewässerkunde, Koblenz

RHmV (2009) Rückstandshöchstmengenverordnung in der Fassung der Bekanntmachung vom 21. Oktober 1999 (BGBl. S. 2082; 2002 I S. 1004), zuletzt geändert durch Artikel 1 der Verordnung vom 2. Oktober 2009 (BGBl. I S. 3230) (Residues and maximum residue levels of plant protection products in the version of October 2009. German Federal Gov. (Ed)

Rosenzweig S, Quick I, Cron N, König F, Schriever S, Vollmer S, Svenson C, Grätz D (2012) Hydromorphologische Komponenten im Flussauenmodell INFORM—Entwicklung und Anwendung der morphologischen Systemkomponene MORPHO und des Bewertungsmoduls VALmorph zur quantitativen Erfassung und Bewertung hydromorphologischer Veränderungen in Fluss und Aue. Report for Bundesanstalt für Gewässerkunde, Koblenz

Ruchay D (1993) Die Elbe—ein Fluß im Wandel. Umwelt 23(5):33–39

Sb (2011) Nařízení vlády ze dne 22. Prosince 2010, kterým se mění nařízení vlády č. 61/2003 sb., o ukazatelích a hodnotách přípustného znečištění povrchových vod a od-padních vod, náležitostech povolení k vypouštění odpadních vod do vod povrchových a do kanalizací a o citlivých oblastech, ve znění nařízení vlády č. 229/2007 sb. (Czech regulation on the admissible pollution of surface water and waste water)

Schwandt D, Hübner G, Claus E, Heininger P (2014) Die hochwasserinduzierte Remobilisierung von Altsedimenten. In: BfG (ed) Fachbeiträge der BfG zum Sedimentmanagementkonzept Elbe. BfG- Mitteilungen Nr. 30. Bundesanstalt für Gewässerkunde, Koblenz. URL: http://doi.bafg.de/BfG/2014/BfG_Mitteilungen_30.2014.pdf

Schwartz R, Kozerski H-P (2003) Entry and deposits of suspended particulate matter in groyne fields of the middle Elbe and its ecological relevance. Acta Hydrochim Hydrobiol 31(4–5):391–399

SedNet (2003) The SedNet strategy paper. In: Brils J (ed) TNO-MEP, The Netherlands, 22 p

SedNet (2007) Sediment management—an essential element of river basin management plans report on the SedNet round table discussion. Venice, 22–23 Nov 2007. www.sednet.org

Simon M (1991) Die Belastung der Elbe und ihrer Hauptnebenflüsse auf dem Gebiet der ehemaligen DDR. wwt Wasserwirtschaft Wassertechnik 1:23–27

Simon M (1996) Anthropogene Einflüsse auf das Hochwasserabflußverhalten im Einzugsgebiet der Elbe. Wasser Boden 48(2):19–23

Simon M, Bekele V, Kulasová B, Maul C, Oppermann R, Řehák P (2005) Die Elbe und ihr Einzugsgebiet. Ein geographisch-hydrologischer und wasserwirtschaftlicher Überblick. Report for IKSE Internationale Kommission zum Schutz der Elbe, Magdeburg, 254 p

Sparrevik M, Barton DN, Bates ME, Linkov I (2011a) Use of stochastic multi-criteria decision analysis to support sustainable management of contaminated sediments. Environ Sci Technol 46:1326–1334

Sparrevik M, Barton DN, Oen AMP, Sehkar NU, Linkov I (2011b) Use of multicriteria involvement processes to enhance transparency and stakeholder participation at Bergen Harbor, Norway. Integr Environ Assess Manage 7(3):414–425

Spott D, Guhr H (1996) The dynamics of suspended solids in the tidally unaffected area of the river Elbe as a function of flow and shipping Arch. Hydrobiol Spec Issues Adv Limnol 47:127–133

Stachel B, Christoph EH, Götz R, Herrmann T, Krüger F, Kuhn T, Lay J, Löffler J, Papke O, Reincke H, Schröter-Kermani C, Schwartz R, Steeg E, Stehr D, Uhlig S, Umlauf G (2006) Contamination of the alluvial plain, feeding-stuffs and foodstuffs with polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans (PCDD/Fs), dioxin-like polychlorinated biphenyls (dl-PCBs) and mercury from the river Elbe in the light of the flood event in August 2002. Sci Total Environ 364(1–3):96–112

Vácha R, Poláček O, Horváthová V (2003) State of contamination of agricultural soils after floods in August 2002. Plant Soil Environ 49(7):307–313

Von Stackelberg KE, Burmistrov D, Vorhees DJ, Bridges TS, Linkov I (2002) Importance of uncertainty and variability to predicted risks from trophic transfer of PCBs in dredged sediments. Risk Anal 22(3):166–174

Zachmann DW, Van Der Veen A, Friese K (2013) Floodplain lakes as an archive for the metal pollution in the river Elbe (Germany) during the 20th century. Appl Geochem 35:14–27