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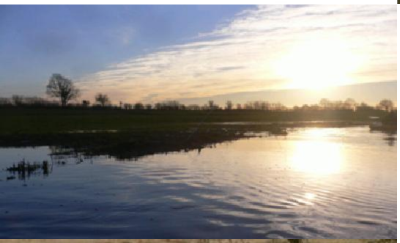






Ecohydrologic and hydraulic stream modelling to describe aquatic habitats

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"In fließendes Wasser zu schauen
wird niemals langweilig!"

Karl Krack

Abstract

Natural catchments, streams and aquatic diversity were globally degraded due to the impacts of industrial and urban development, as well as the intensification of agriculture. Degradation occurred at different spatial scales and rehabilitation measures are required in both streams and catchments, to improve conditions for the aquatic biota. Models, applied for planning restoration measures, are mostly targeting individual components of the complex chain linking the abiotic and biotic environment; e.g., models might be used just for predicting hydrological or hydraulic variables. Hereby, the cause-effect chain is compromised, which links drivers, pressures, state and impacts of the riverine system. There are almost no models considering the overall system catchment-streams-habitat-aquatic biota. Thus, tools are unavailable, with which the effects of measures on the stream ecosystem can be tested, ideally already during the design phase.

It is the scope of this dissertation to build an integrated, Geographic Information System (GIS)-based model system considering the cause-effect chain from the catchment to the stream and aquatic biota. The models require data on climatic and physical catchment properties, and on the geometry and structure of the streams. This enables the assessment of the impact of global change as well as of more regional and local changes on the stream ecosystem. The approach of this thesis is based on the Driver-Pressure-State-Impact-(Response) (DPSI(R)) concept and includes the linkage of one ecohydrologic, two hydraulic and two habitat models:

The ecohydrologic model Soil and Water Assessment Tool (SWAT) was used for depicting the discharge regime and erosion processes controlled by land use and climate on the catchment scale. As part of this, two lowland-specific tools have been developed and have been used for hydrologic simulations: First, a method for incorporating the high surface retention potential of the catchment and second, an estimation model to evaluate the ratios of sediment entry pathways for quantifying sediment input from the field, tile drains and the river bank.

The discharge and sediment time series resulting from the hydrologic modelling were used for hydraulic simulations on the reach scale. Water depth, flow velocity, substrate changes and sediment transport were simulated in variable resolutions with the Hydrologic Engineering Center River Analysis System (HEC-RAS) one-dimensionally and with the Adaptive Hydraulics Modelling system (AdH) two-dimensionally.

Combined with structural river mapping, the temporally and spatially dynamic results of the hydraulic models were used for describing macroinvertebrate habitats. Based on different parameters, two independent simulations were carried out: First, the distribution of a single species, the freshwater clam *Sphaerium corneum* was modelled with the species distribution model (SDM) BIOMOD, based on parameters related to hydraulics and sediment transport. This took place in cooperation with the Senckenberg Institute Gelnhausen. Second, within the scope of this thesis and in cooperation with the Faculty of Biology, Aquatic Ecology, University Duisburg-Essen the Habitat Evaluation Tool (HET) was developed. The HET model was used to simulate the prevailing macroinvertebrate community in the stream based on the river's substrates.

Model results are maps and statistics of the spatial occurrence of species at different points in time which are connected to the prevailing environmental conditions. The model system was developed and successfully applied in the northern German lowland catchment of the Kielstau. Results of the submodels show very good agreement with observed hydrological and hydraulic parameters and good agreement with observed spatio-temporal erosion. Simulated spatial species distributions are realistic when compared to observed distributions derived from sampling campaigns. The methodology is transferrable and has been applied already during the development phase in different catchments.

The developed model system advances integrated modelling, but future improvements are necessary. This particularly concerns the simulation of abiotic parameters, investigation of organism preferences, the combined simulation of numerous organism groups and the simulation of interactions and feedback loops. Such a more comprehensive modelling approach would most effectively be developed by interdisciplinary teams.

Kurzfassung

Naturnahe Einzugsgebiete, Fließgewässer und die aquatische Vielfalt wurden primär aufgrund der Auswirkungen der industriellen und urbanen Entwicklung sowie der Intensivierung der Landwirtschaft weltweit degeneriert. Diese Veränderungen haben auf verschiedenen Skalen stattgefunden und daher sind Rehabilitationsmaßnahmen in Fließgerinnen und Einzugsgebieten notwendig, um die Bedingungen für aquatische Lebewesen zu verbessern. Modelle, die für die Planungen von Renaturierungsmaßnahmen angewendet werden, zielen meist auf einzelne Komponenten der komplexen Kette, die Abiotik und Biotik verbindet; so werden Modelle z.B. für die Prognose von hydrologischen und hydraulischen Zielgrößen verwendet. Dadurch wird die Wirkungskette unterbrochen, die die Antriebskräfte, Belastungen, Zustand und Auswirkungen auf das Gewässersystem verbindet. Es gibt kaum Modelle, die das Gesamtsystem Einzugsgebiet-Fließgewässer-Habitat-aquatische Lebewesen betrachten. Daher fehlt es an Werkzeugen, mit denen die Auswirkungen solcher Maßnahmen auf den aquatischen Lebensraum, möglichst schon während der Planungsphase, getestet werden können.

Ziel dieser Dissertation ist daher die Erstellung eines integrierten, geographischen Informationssystems (GIS)-basierten Modellverbundes, der eine ganzheitliche Betrachtung der Wirkungskette vom Einzugsgebiet über das Fließgerinne zum aquatischen Lebewesen ermöglicht. Der Datenbedarf der Modelle umfasst die klimatischen und physischen Eigenschaften von Einzugsgebieten, sowie die Geometrie und Struktur der Fließgerinne. Dies ermöglicht es, den Einfluss des globalen Wandels sowie regionale und lokale Veränderungen auf den Lebensraum im Fließgewässer zu bewerten. Der Ansatz dieser Arbeit basiert auf dem "Driver-Pressure-State-Impact-(Response)" (DPSI(R)) Konzept, und beinhaltet die Verknüpfung von einem ökohydrologischen-, zwei hydraulischen-, und zwei Habitatmodellen:

Das ökohydrologische Modell "Soil and Water Assessment Tool" (SWAT) wurde genutzt, um das Abflussregime und die Erosionsprozesse auf Einzugsgebietsebene in Abhängigkeit von Landnutzung und Klima abzubilden. Im Rahmen dessen wurden zwei flachlandspezifische Werkzeuge entwickelt und in der hydrologischen Modellierung angewendet: Erstens, eine Methode zur Berücksichtigung des hohen Oberflächenretentionspotentials des Einzugsgebietes und zweitens, ein Abschätzungsmodell für die Bestimmung der Proportionen der Sedimenteintragspfade, um den Sedimenteintrag aus der Fläche, den Drainagen und Ufererosion zu quantifizieren.

Auf Fließgewässerebene wurden dann die Abfluss- und Sedimentzeitreihen aus der hydrologischen Modellierung genutzt, um hydraulische Simulationen durchzuführen. Hierfür wurden mit dem Modell "Hydrologic Engineering Center River Analysis System" (HEC-RAS) eindimensional und mit dem "Adaptive Hydraulics Modelling system" (AdH) zweidimensional Wassertiefe, Fließgeschwindigkeit, Substratveränderungen und Sedimenttransport in variablen Auflösungen simuliert.

Zusammen mit Gewässertrukturkartierungen wurden die zeitlich und räumlich dynamischen hydraulischen Modellergebnisse genutzt, um den Makrozoobenthoslebensraum zu beschreiben. Basierend auf verschiedenen Parametern fanden zwei unabhängige Simulationen statt: Erstens wurde mit dem Habitatmodell BIOMOD die Flussmuschel *Sphaerium corneum* basierend auf verschiedenen Sedimenttransport- und hydraulischen Habitatparametern abgebildet. Dies fand in Zusammenarbeit mit dem Senckenberg Forschungsinstitut Gelnhausen statt. Zweitens wurde im Rahmen dieser Arbeit und in Zusammenarbeit mit der Fakultät für Biologie, Aquatische Ökologie, Universität Duisburg-Essen das "Habitat Evaluation Tool" (HET) entwickelt. Mit dem HET Modell wurde die im Fließgewässer vorhandene Makrozoobenthosgemeinschaft basierend auf dem Gewässersubstrat modelliert.

Die Modellausgabe sind Karten und Statistiken des räumlichen Vorkommens der Arten an unterschiedlichen Zeitpunkten, die mit den vorherrschenden Umweltbedingungen verbunden sind. Das Modellsystem wurde am Beispiel des ländlich geprägten Kielstau Einzugsgebietes im Norddeutschen Tiefland erstellt und erfolgreich angewendet. Die Ergebnisse der Teilmodelle zeigen eine sehr gute Übereinstimmung mit gemessenen hydrologischen und hydraulischen Parametern und eine gute Übereinstimmung mit beobachteten räumlichen- und zeitlichen Erosionsformen. Simulierte räumliche Artenverteilungen sind realistisch im Vergleich zu beobachteten Verteilungen, abgeleitet aus Probenahmekampagnen. Die Methodik ist übertragbar und wurde bereits während der Entwicklung in anderen Einzugsgebieten angewendet.

Die Entwicklung des Modellsystems führt zu einem Voranschreiten der integrierten Modellierung, aber zukünftige Verbesserungen sind notwendig. Dies betrifft vor allem die Simulation von abiotischen Parametern, die Erforschung von Präferenzen der Organismen, die kombinierte Simulation mehrerer Organismengruppen sowie die Simulation von Interaktionen und Rückkopplungseffekten. Solch ein umfassenderer Modellierungsansatz könnte am effektivsten durch interdisziplinäre Teams entwickelt werden kann.

Contents

Abbreviations

1D	One-dimensional
2D	Two-dimensional
ABAG	Allgemeine Bodenabtragsgleichung / German Version of the USLE
AdH	Adaptive Hydraulics Modelling System
ASL	Above Sea Level
AUC	Area Under the receiver operating characteristic Curve
CAU	Christian-Albrechts-University
CHL	Coastal and Hydraulics Laboratory
DAV	Digitales Anlagenverzeichnis / Digital River Structure Database
DEM	Digital Elevation Model
DPSIR	Driver-Pressure-State-Impact-Response
ERDC	Engineering Research and Development Center
ERPL	Estimation of Retention Potential in Lowlands
FFH	Fauna-Flora-Habitat Richtlinie / Fauna-Flora-Habitat directive
GAM	Generalized Additive Model
GBM	Generalized Boosting Model
GIS	Geographic Information System
GLM	Generalized Linear Model
HEC-RAS	Hydraulic Engineering Center River Analysis System
HET	Habitat Evaluation Tool
HRU	Hydrological Response Unit
LANU S.-H.	Landesamt für Natur und Umwelt des Landes Schleswig-Holstein / Schleswig-Holstein State Agency for Nature and Environment
LiDAR	Light Detection And Ranging
LLUR S.-H.	Landesamt für Landwirtschaft, Umwelt und ländliche Räume Schleswig-Holstein (früher LANU) / Schleswig-Holstein State Agency for Agriculture, Environment and Rural Areas (formerly LANU)
MUSLE	Modified Universal Soil Loss Equation
NSE	Nash-Sutcliffe-Efficiency
PSLG	Planar Straight Line Graph
SDM	Species Distribution Model
SWAT	Soil and Water Assessment Tool
SCS-CN	Soil Conservation Service - Curve Number method
SEPAL	Sediment Entry Pathway Assessment in Lowlands
TSS	True Skill Statistic
USACE	United States Army Corps of Engineers
USDA-ARS	United States Department of Agriculture - Agricultural Research Service
USLE	Universal Soil Loss Equation
WFD	Water Framework Directive

Chapter 1 Introduction

1.1 Motivation and objectives

Anthropogenic influences and natural processes impact the current status and the future development of rivers and streams. In many parts of the landscape, human interference with the natural environment has pushed natural processes of riverine development in the background. For instance, urban and agricultural development have led to changes in water regime and water balance, in sediment transport and in river morphology. Where this happened, aquatic freshwater habitats have changed and often degraded considerably (Baron et al., 2002), which is threatening freshwater species (Dudgeon et al., 2006). Increasingly, society and policy demands a rethinking and a rehabilitation of riverine ecosystems (EC, 2000; CWA, 1972). This can be achieved through coordinated changes in both catchment management and enhancing river structure (Poff et al., 2009; Kemp et al., 1999). In this regard, engineers and scientists are faced with an optimization problem on different scales: the incorporation of technical, ecological and climate change boundary conditions into the design of sustainable catchment and river channel rehabilitation measures (Wilby et al., 2011; Grantham et al., 2013).

However, optimizing rehabilitation measures without a clear consideration of the prevailing natural boundary conditions often results in disappointing results. Keeping in mind the system's natural processes is essential for obtaining a near-natural aquatic ecosystem that is sustainable and stable (Beechie et al., 2010; Feld et al., 2011). Natural processes interact in a complex spatio-temporal chain with the aquatic habitat. For instance, climate induces a cascading effect which eventually reaches aquatic habitats: An increase in intensity of rain events due to changing climate causes erosion from fields and a higher instream sediment transport which can potentially cover habitats. At the same time, increased streamflow can also create new habitats through an increase in channel dynamics elsewhere in the river.

Observing the impact of these anthropogenic actions and natural processes is crucial for deriving cause-effect chains such as described in the example above. This process knowledge is essential for building models and for carrying out simulations. Similar to observations, computer models are able to depict the current status of system components and they have an additional, decisive advantage: A reliable and comprehensive model can be used to assess scenarios and to evaluate different choices of action in a time- and resource-efficient manner. This has numerous prospects, e.g. in planning climate change mitigation measures, or improving the chances of success of rehabilitation measures. Ideally, such simulations need to fully integrate anthropogenic actions and natural processes acting on different scales, because only then can the impact of perturbations to natural processes be simulated sufficiently. However, these are often modelled detached from each other, though the catchment and its riverine system act as one natural entity. This lack of a comprehensive simulation of catchment, instream and habitat-defining processes is a well known research deficit in the field of integrated modelling (Newman et al., 2006; Shields et al., 2006; Goethals et al., 2007; Diembeck et al., 2008) and the main motivation for this work.

To fill these research gaps, scientists face four challenges: (1) Individual approaches are mostly used to solve catchment, river and ecological questions, largely independent from each other. A methodological solution to this isolated consideration is thus necessary. Rice et al. (2009) for instance, define the lack of interdisciplinary developed methods as the core deficit for the advancement of river science. Hence, closing the mentioned research gap requires an interdisciplinary view in selecting habitat parameters, at linking catchment, instream and habitat-defining processes, and in choosing and developing suitable models.

(2) After having defined the methodological approach, technical solutions need to be found: The mentioned processes act in different environments from lowlands to mountainous regions. While lowlands are characterised through low hydraulic gradients, high groundwater influence and, possibly, artificial drainage systems, mountainous regions are subject to higher surface runoff fractions, a faster hydrological response and higher erosive force of the water fluxes. The model system must be adapted to the prevailing processes of the study catchment which includes the consideration of the relevant drivers of the system (Beechie et al. 2010; Feld et al., 2011). The processes also act on different scales and are usually addressed by experts from different fields using separate simulation tools: First, on the catchment scale, hydrologic models are used to depict the land phase of the hydrological cycle. These tools require climatic and catchment-related data to calculate the spatial and temporal distribution of discharge and water-transported substances in river networks and channel systems. Hydrologic models are applied since decades (Crawford and Linsley, 1966), mostly within the scope of water-related engineering, economical and environmental problems (BWK, 2002). Second, on the stream channel scale, numerical hydraulic and morphodynamic models are used to simulate fluid dynamics and sediment transport. Physical information of the bathymetry and channel properties are necessary to calculate hydraulic flow parameters like velocity, depth, shear stress as well as sediment deposition and erosion. Since sufficient computational power is commonly available, numerical models are more and more substituting their physical precursors since the 1990s (Ettema, 2000). Third, processes in riverine habitats have been observed by ecologists who gathered extensive knowledge on abiotic-biotic dependencies (Schmidt-Kloiber and Hering, 2011). This information on species habitat preferences can be used for deducing knowledge-based rules (Schmedtje, 1995; Jowett, 2003) and, more recently, to carry out simulations (Gertseva et al., 2004; VanBroekhoven et al. 2006; Kuemmerlen et al. 2014). In summary, the methodologies to depict the above defined processes exist, but the technical requirements for connecting these three components need to be established in a way that the models can be applied in different ecoregions.

(3) Hydrologic and hydraulic processes are interlinked from the catchment's fields down to the instream micro scale. Depicting this interdependence through interdisciplinary developed methods (challenge 1) and suitable technical tools (challenge 2) leads to simulation results which need to be validated against observations. In that context, two factors are important: First, it is challenging to supply measurement results of hydrologic and hydraulic parameters at different scales which have to be sufficiently reproduced both spatially and temporally by the simulations. Second, beyond the simulations, static and anthropogenic habitat components which cannot be modelled, have to be depicted through mapping and have to be merged to the dynamic simulation results.

(4) The last challenge is the assessment of how the integrated, abiotic simulation results (challenge 3) impact aquatic organisms. The European Water Framework Directive (WFD) (EC, 2000) generally defines fish, macroinvertebrates, macrophytes and phytoplankton as target organisms for improving the aquatic habitat. This thesis is focusing on macroinvertebrates, as they are considered the most appropriate organism group for a first application of the model system due to the generally sensitive response to a multitude of stressors and the occur-

rence from small streams to large rivers (Rawer-Jost et al., 2004; Sandin and Hering, 2004; Hering et al., 2006). An assessment needs to include a realistic simulation of the species response to the simulated abiotic parameters. Different approaches exist (Harby et al., 2004; VanBroekhoven et al., 2006; Goethals et al., 2007; Hoang et al., 2010; Schuwirth and Reichert, 2013) of which the most applicable need to be adopted and revised for the model system.

The above described research deficit leads to the main objective of this work: the conceptualisation and application of an integrated model framework that can simulate the impact of catchment and instream changes on the riverine habitat of macroinvertebrate species.

Based on the four challenges defined above, the following subgoals are defined to meet the main objective:

(1) developing the interdisciplinary methodological approach: defining habitat parameters, linking processes and identifying necessary models

(2) establishing the technical requirements: creating tools to apply the model system in a GIS framework and catchments such as the study area of this work

(3) verifying the integrated abiotic simulations: validate the depicted core environmental parameters that define the riverine habitat across different scales

(4) assessing species response: predicting species occurrence and community structure based on the simulated environmental parameters

1.2 Study area

The model framework is developed and applied exemplarily in the Kielstau, a typical northern German lowland catchment located about 10 km south-east of the city of Flensburg in Schleswig-Holstein (Figure 1.1). From the source to the catchment outlet, the Kielstau has a total length of 16.2 km and a mean gradient of 1.2 ‰ (LVA, 1992-2005). About 5 km downstream of its origin it flows through Lake Winderatt, a small lake with a surface area of about 0.24 km^2 (Grudzinski, 2007). The lake's water outflow is artificially ponded through a fixed weir. Downstream of Lake Winderatt (Figure 1.1. Nr. 6) two tributaries, the Moorau (Figure 1.1. Nr. 5) and the Hennebach, and various drainage pipes and open ditches discharge into the Kielstau. Close to the catchment outlet, the gauging station Soltfeld is located (Figure 1.1. Nr. 1a), which is part of the official gauging network of the Federal State Schleswig-Holstein and also a WFD monitoring station. Discharge there is measured since 1985 with an average annual minimum flow of 0.07, average flow of 0.47 and average annual maximum flow of 2.61 $m^3 s^{-1}$ (all statistics from 1985-2012) (LKN, 2013). The Kielstau joins with the River Bondenau and flows shortly after into Lake Treßsee, a small and shallow lake which is increasingly silting up. The outflow from the Treßsee forms the origin of the river Treene, which is part of the river basin Eider.

The mean annual precipitation and temperature are 893 mm and 8.3 °C, respectively (DWD, 2010). The land use of the rural catchment is dominated by arable land (56 %), pasture (26 %), forest (8 %) and small urban settlements (3 %), (DLR, 1995). Typical for the region are the predominant sandy and loamy soils in the catchment with Haplic Luvisols in the eastern and Stagnic Luvisols in the western part, whereas Sapric Histosols are present along the stream and its tributaries (BGR, 1999). Topography of the Kielstau catchment is relatively flat. It ranges from 27 m to 78 m above sea level (ASL) but has rolling hills and numerous natural landscape depressions where water is ponding after intense rain events (Figure 1.1. Nr. 2b). This topography was formed by glacial- and periglacial processes of the late Pleistocene (Lundquist, 1986; Sommerhäuser and Schuhmacher, 2003). Extensive drainage measures have been implemented during the reallocation of land, mainly from the 1950s to the 1980s (MELF, 1980) to enhance agricultural productivity (Riedel and Polensky, 1987). About

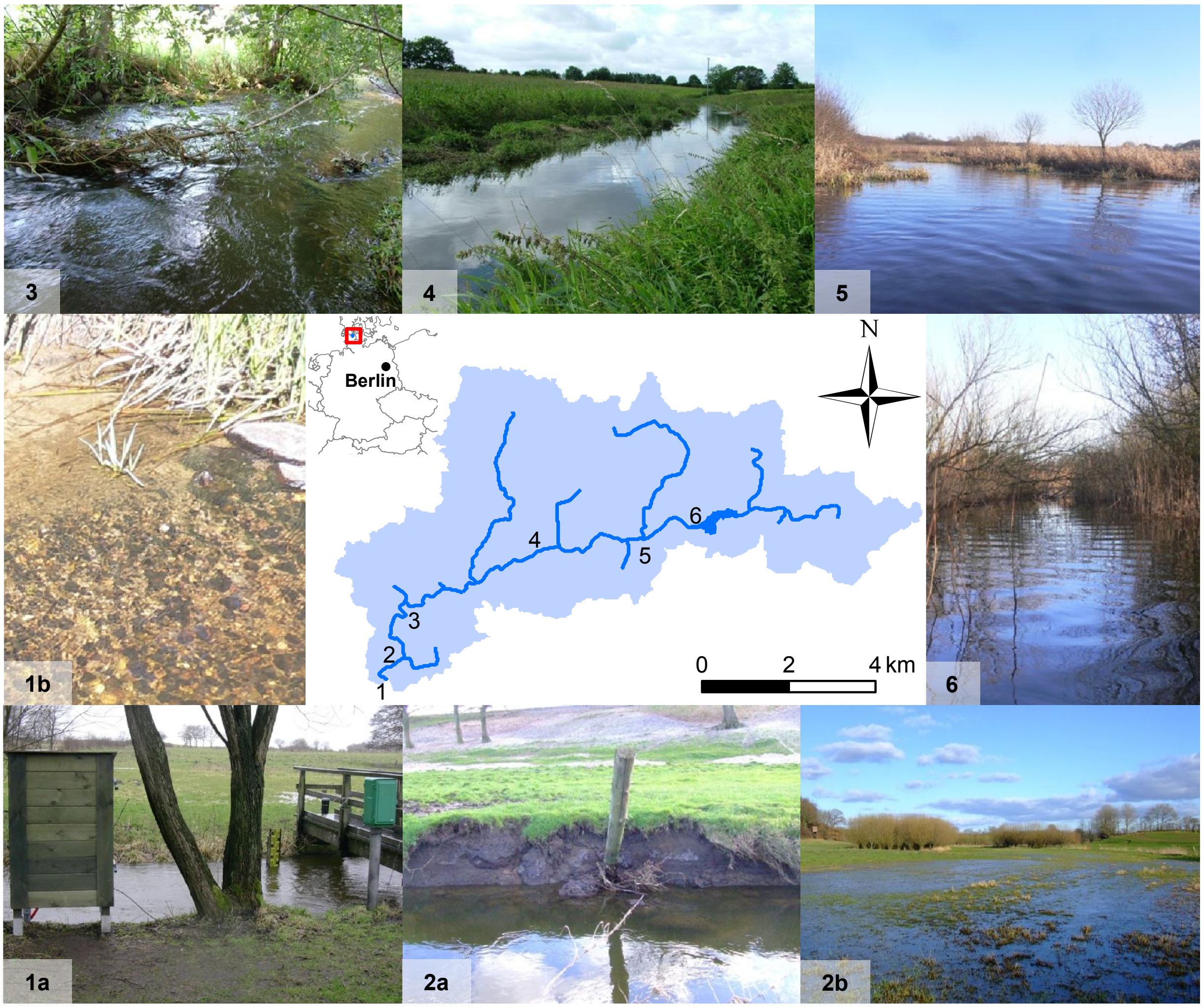

Figure 1.1 Impressions and location of the Kielstau catchment in Germany (red square). 1a: gauge Soltfeld; 1b: substrate upstream the gauge; 2a: bank erosion (foreground), field erosion (background); 2b ponding water on the surface; 3: stream section in western part of catchment; 4: channelized section through agricultural fields; 5: Moorau tributary from north; 6: outflow from Lake Winderatt

31 % of the catchment are artificially drained (Fohrer et al., 2007). The hydrology is thus characterised by agricultural drainage systems, near-surface groundwater, low hydraulic gradients and a high interaction between groundwater and surface water. Water quality is influenced by six wastewater treatment plants with 8,495 person equivalents (Umweltatlas, 2014) and agricultural non-point sources.

The area is subject to intense interest by seven environmental associations, authorities and foundations that are investigating the current state and are discussing rehabilitation measures at all scales. The Kielstau is part of the Fauna-Flora-Habitat (FFH; EC, 1992) protection area and 175 ha of land along the river and around Lake Winderatt are owned by two nature conservation foundations which increases the potential for river rehabilitation measures within and beyond the scope of the WFD. For research purposes, the Institute for Natural Resource Conservation, Department of Hydrology and Water Resources Management Kiel is carrying out continuous measurements and temporal sampling programs on different scales. Investigated parameters are discharge, water quality (oxygen, temperature, electric conductivity, nitrogen and phosphorous fractions, biological oxygen demand, herbicides), hydraulic variables, stream morphology, erosion, sediment transport, climatic variables, phytoplankton and macroinvertebrates. This focus on ecohydrological relevant processes (Schmalz and Fohrer, 2010) recently led to the designation of the Kielstau as Germany's first UNESCO ecohydrology demosite (Fohrer and Schmalz, 2012).

Many parts of the Kielstau channel network have been changed markedly from its natural course during the reallocation of land in the mid-twentieth century. The main stream channel has been straightened, incised and cleared for enhancing drainage (Figure 1.1. Nr. 4). This mostly disconnected the river from its floodplains, decreased flow length and stream roughness which altered not only the hydraulic regime but also sediment processes: flow velocities increased and caused erosion in parts of the streams bed and banks (Figure 1.1. Nr. 2a), while the remaining river sections are very stable. The Kielstau is classified as a lowland gravel bed river (type 16, according to LAWA, 2000), but there are also sections of the stream that are covered with sand layers which show high dynamics over the course of one year. Mostly due to the structural remodelling, the morphological state of the gravel stream is assessed 'poor' to 'moderate' (Olbert et al., 2006) according to the standard hydromorphological river survey method in Germany (LAWA, 2000). This status is typical for many lowland streams in northern Germany. Still, near-natural river sections exist according to the 'Digital River Structure Database' (DAV-WBV/LAND S.-H., 2006) which can act as reference points (Figure 1.1. Nr. 3). A location close to the catchment outlet was chosen for a detailed investigation and simulation within this thesis due to its reasonable structural conditions and its usage as a WFD monitoring site: there, stream width is about 4 m with varying flow depths. Three distinct bends, four riffles and five pools facilitate a diverse current pattern. Mobile substrates, mainly sand and gravel (Figure 1.1. Nr. 1b), are interspersed with large wood debris, alder roots, stones, water plants and coarse particulate organic matter.

The combination of being a well-researched study catchment, having near-natural as well as degraded sections and the public attention to the catchment’s status makes the Kielstau an ideal example for testing the integrated model framework.

1.3 Outline

The following chapters are individual papers submitted to international peer-reviewed journals and structured according to the research objectives defined in Chapter 1.1. In Chapter 2, the development of the interdisciplinary methodological approach is shown. The focus lies on the integration of the modelling framework into the DPSI(R) concept, developed by EEA (1999). The cause-effect chain from the *Driver* to the *Impact* and its relevance for this work are explained, the models for depicting the abiotic environment are introduced, including a general overview of the input data as well as simulated processes and parameters.

After defining the methodological framework, technical adaptations of the models and simulations were carried out. In Chapter 3, Chapter 4 and Chapter 5, the development and application of tools for using the model system in the lowland study catchment and in the ArcGIS environment are presented. In Chapter 3, the hydrologic simulations of the study catchment with SWAT (Arnold et al., 1998) are described. This represents the groundwork of the model framework since the hydrological catchment processes significantly impact riverine habitat properties. As part of this, two important landscape features of lowland catchments that impact hydrologic processes are identified and implemented into the hydrologic simulations: artificial drainages and landscape depressions. In Chapter 4, sediment pathways in lowlands and the long-term sediment budget of the study catchment are identified using a newly developed tool, SEPAL (Sediment Entry Pathway Assessment in Lowlands). The focus lies on the distinction between field, channel and tile drain sediment input into the streams. This is fundamental for correctly depicting temporal and spatial sediment pathways as a basis for developing appropriate mitigation measures. In Chapter 5, the technical stage is set for linking the individual models to a model cascade in ArcGIS. This includes user interface developments between the hydrologic model SWAT and the hydraulic model HEC-RAS (USACE, 2010a) as well as an ArcGIS interface for the two-dimensional hydraulic model AdH (Berger et al., 2011).

The integrated simulation of the core environmental parameters is shown in Chapter 6. There, the hydrologic simulations, explained in detail in Chapter 3, are extended to simulate catchment sediment input to the streams. Catchment hydrology and sediment processes are then linked to the two hydraulic models HEC-RAS and AdH that are used to depict instream water and sediment fluxes in one and two dimensions. Hydraulic flow parameters, substrate changes, channel erositivity and stability are simulated over the course of one year in the main reach of the Kielstau down to the instream micro scale. This hydrological-hydraulic model connection represents the link between catchment processes and small-scale riverine habitat changes.

The depicted abiotic parameters are used in two ways for the simulations of the macroinvertebrate habitats: First, in Chapter 7, the SDM BIOMOD is used in cooperation with the Senckenberg Institute in Gelnhausen to exemplarily simulate the distribution of a freshwater clam along the main channel of the Kielstau. These simulations are based on different hydraulic parameters, depicted by the one-dimensional hydraulic model HEC-RAS (Chapter 6). Second, in Chapter 8, the development and application of a new macroinvertebrate model, HET, is presented. The model is applied on the micro scale to model the changes of the macroinvertebrate community based on the two-dimensional simulation of the riverine substrates (Chapter 6). These two habitat model applications are the final step in the simulations and close the complex cause-effect chain linking the abiotic to the biotic environment.

In Chapter 9, the work is summarized and interpreted. Capabilities and limitations of the developed model system are explained. The achieved objectives are discussed according to their scientific and practical relevance. Finally, suggestions for further work are given.

Chapter 2 A transdisciplinary approach for modelling macroinvertebrate habitats in lowland streams

In: Ecohydrology of Surface and Groundwater Dependent Systems: Concepts, Methods and Recent Developments. Editors: M. Thoms, K. Heal, E. Bøgh, A. Chambel, V. Smakhtin. IAHS Publication Red Book Series 328, 2009, 24-33, ISBN: 978-1-901502-99-2

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Abstract Manifold anthropogenic influences are the main cause of river habitat degradation and extensive regeneration needs to be conducted to achieve the aims of the WFD in Europe. As the outcome of river and stream rehabilitation measures is sometimes difficult to foresee, a GIS-based approach consisting of an ecohydrologic, a hydraulic and a GIS mapping submodel is suggested for creating an integrated catchment and an instream modelling system to dynamically depict the influence of abiotic changes on the habitat quality. The DPSI(R) concept is utilised to depict the complex cause-effect chain of hydromorphological changes on macroinvertebrate habitats in lowland streams. A first application of the three submodels has been conducted in the North German Kielstau catchment and first results of modelling and mapping the impact on selected habitat parameters are displayed. Further work needs to be done in linking the submodels and in assessing the impact of the altered state on the macroinvertebrate fauna by parameter functions derived from a knowledge-based database and sampling schemes.

Keywords DPSI; hydrological model; hydraulic model; GIS; parameter functions; hydromorphology; macroinvertebrate; lowland

2.1 Introduction

Dominant characteristics of flowing waters are the discharge regime, the currents and the interrelated sediment dynamics. Together with the properties of the ecoregion (Omernik, 2004) in which the catchment is located, and the anthropogenic influence, these parameters mainly define hydromorphology and stream biota (Frissell et al., 1986; Lorenz et al., 2004). Discharge regime, water quality and fine sediment input into the river are factors acting on the catchment scale. A variety of (eco-)hydrological models exist that are capable of depicting these processes (Abbott and Refsgaard, 1996; Singh and Woolhiser, 2002; Schmalz et al., 2008a). Current patterns and substrate dynamics in the stream are affected by catchment-scale processes, but are also highly dependent on cross-sectional river features on the micro scale. Nowadays these processes can be successfully depicted with hydraulic computer models (Zanke, 2002; Tate, 2006). This intertwined influence of catchment and stream properties on the aquatic habitat stresses the necessity for an integrated approach where a joint view on catchment and stream processes is indispensable.

Throughout history, cultural development induced continuous changes in attitude towards the environment. This had a decisive influence on how river management is carried out and is perceived in our society (Gregory, 2006). In the late 1980s for example, awareness was focused mainly on water quality while currently the river morphology is a major concern (Umweltbundesamt, 2007). The human influence on catchments and rivers is manifold (Surian and Rinaldi, 2002; James and Marcus, 2006) and can be well described by the DPSI(R)-concept (EEA, 1999). Applying the DPSI(R) conceptual model on flowing waters, the following cause-effect chain can be derived: the general drivers behind river degradation were and still are an increasing industrial and agricultural production caused by population and economic growth. The drivers induce numerous hydrological, water quality and hydromorphological pressures affecting physical conditions of the rivers. The resulting state defines the impact on the aquatic habitat. The response is an external feedback parameter from the society and only occurs if impacts lead to political responses (Kristensen, 2004) and it is thus not possible to be considered for a habitat modelling system. In Europe, a major political response to water quality and habitat degradation resulted in the WFD (EC, 2000) making the current approach on river management operational: the rehabilitation of aquatic habitats in order to restore good ecological status by 2015 (EC, 2000). As anthropogenic influence, which has driven the degradation of aquatic habitats still puts a number of pressures on water bodies, failing of achieving the aims of the WFD within the considered time frame seems inevitable (Moss, 2008). It is thus necessary to optimize rehabilitation measures. Therefore, human constraints have to be discussed together with the ecological demands and aims in order to improve river ecological status. Both degradation and restoration of rivers cause a complex impact chain with positive and negative feedbacks (Wang et al., 2008). The overall impact can not always be directly foreseen (Reichert et al., 2007), as for example changing the drivers can reduce one pressure but increase another pressure, therefore unintentionally causing a degradation of the state (Nedeau et al., 2003) and thus negatively impact habitat quality.

The following points are hence important for successfully modelling aquatic habitats and biota based on the DPSI(R) concept: First, an integrated approach is needed for considering stream and catchment processes so that it is possible to depict the major drivers with the model input data; second, the main pressures on the system need to be defined and be represented in the model algorithms; third, based on the multiple pressures, it then has to be possible to dynamically assess the changes of the state of habitat parameters with the model output; (4) in the final step, the impact of the state on the aquatic habitat and biota needs to be evaluated which closes the complex cause-effect chain from the drivers to the impact.

The WFD generally defines fish, macroinvertebrates, macrophytes and phytoplankton as target organisms for improving the aquatic habitat. This study is restricted to macroinvertebrates, as they are considered the most appropriate organism group due to the generally good response to a multitude of stressors (Rawer-Jost et al., 2004; Sandin and Hering, 2004), and beyond that, lowland streams have relatively species-poor fish and macrophyte community, which narrows the scope even more on macroinvertebrate species (Hering et al., 2006).

The motivation for this work is the lack of an integrated modelling system that is capable of optimizing catchment and instream rehabilitation measures in regard to their influence on the aquatic habitat. The aim of this paper is the description of a methodology to model macroinvertebrate habitats in lowland streams from driver through impact based on the DPSI(R)-concept using a hydrological and a hydraulic model, GIS mapping techniques and the development of parameter functions for selected species.

2.2 Model description

In order to model a dynamic DPSI-system from driver through impact it is important that the drivers are adequately accounted for by the model input data. Table 2.1 shows the representation of the drivers though the corresponding data and the used submodels.

The input data are necessary for applying the following submodels in the ArcGIS (ESRI, 1997) environment: an integrated *hydrologic* and *hydraulic model* and *GIS mapping* techniques. The 2005-version of SWAT (Arnold et al., 1998) is used as the ecohydrologic model. The physically based model can simulate the water balance, nutrients and pesticides, erosion, plant growth cycles, management practices and water bodies on a daily time step for continuous simulations over long time periods (Neitsch et al., 2005a). The SWAT model is applied on the catchment scale and is used to simulate the hydrological cycle and to assess the sediment input from fields and artificial drainages. As displayed in Table 2.1, the model requires spatially distributed data on GIS maps, climate data and physical information for a relational database. For calibration and validation, daily measured discharge, suspended sediment and water quality data are needed.

AdH (Berger and Tate, 2007) is used as the hydraulic model. Model features are the automatic adaptation of the numerical mesh to improve model accuracy and the rapid convergence of flows to steady state solutions. AdH's two-dimensional (2D) shallow water equations and the sediment transport module are applied for stream reaches up to 1 km length for regions of particular interest, e.g. rehabilitation measures. In order to solve the hydrodynamic equations, the model requires flow boundary conditions, surface roughness values and topographic data to construct a numerical mesh with triangular elements. The element size is chosen to be in the range of $< 1\ m^2$ within the stream to be capable of depicting boulders, dead wood and other flow obstructions and $> 1\ m^2$ in the floodplains. As Light Detection and Ranging (LiDAR), which is not penetrating the water surface, is used to depict the floodplain topography and the river course, the stream bathymetry needs to be refined with additional topographic data. For modelling sediment transport and substrate stability, the model requires information on substrate grain size and distribution.

GIS mapping techniques are used to refine the available structural river data that have been recorded on digital maps within the scope of the WFD status report (DAV-WBV/LAND S.-H., 2006). The river section of interest therefore has to be visited in order to capture substrate and small-scale morphological features on digital GIS sketches.

Table 2.1 The drivers of the system, data to depict the drivers and corresponding submodel

Driver	Data	Submodel
Agriculture	Climate data Land use map Soil map Drainage map Topography Management information Physical vegetation and soil parameters	Ecohydrologic model SWAT
Urbanisation	Land use map Soil sealing ratio Point discharge data	Ecohydrologic model SWAT
	Riverbed and bank material	Hydraulic model AdH GIS mapping
Flood control	Channel topography Hydraulic structures River course	Hydraulic model AdH
	Flow obstructions (stones, debris)	Hydraulic model AdH GIS mapping
	Instream vegetation	Hydraulic model AdH GIS mapping

2.3 Methodology

Figure 2.1 schematically explains the structure of the proposed modelling system which is being developed for lowland rivers. The *drivers* of the system are *agriculture*, *urbanisation* and *flood control*. Although *flood control* is necessary and somewhat caused by the needs of productive farming and to maintain the standard of living, it is listed here as an individual driver due to the unique pressures it causes. The three drivers need to be implemented in the modelling system by the input data (Table 2.1).

The *pressures* on the habitat are grouped according to the submodel which has to be capable of depicting the relevant processes. Important pressures on the macroinvertebrate habitat are: *hydrologic stress* (Li et al., 2008) defined as events exceeding a certain threshold in discharge and duration, *fine sediment intake* (Berry et al., 2003), *hydraulic stress* (VanBroekhoven et al., 2006), *profile alteration* and *straightening* (Horsák et al., 2008), *substrate stability* (Lorenz et al., 2004), *substrate degradation* (Hering et al., 2004), *river cleaning* (Aldridge, 2000) and *bank and bed fixation* (Horsák et al., 2008). It is important to note that there has to be a connection between the submodels, emphasised by the wide black arrows in Figure 2.1: the output hydrograph and sediment load graph of the hydrological model are used as input for the hydraulic model, thus linking catchment to instream processes. This is of particular importance for the sediment, as aquatic habitats are affected differently from the fine sediment input from fields and agricultural drains with high carbon content and the desired erosion of river banks which creates new flow patterns and increases river dynamics. The GIS mapping submodel needs to be connected with the hydraulic model as *substrate degradation, river cleaning* and *bank and bed fixation* influence the flow characteristics.

The *state* is the actual condition of the habitat parameters and represented by the output from the submodels. The output time series of the hydrological model is analysed and the *duration of extreme events* like minimum and maximum discharge periods are recorded. The hydraulic habitat parameters *velocity*, *water depth* and information about the substrate *silt and clay, sand* and *gravel* calculated by the hydraulic model are recorded on maps. These maps are dynamic over time, meaning that depending on the hydrological and hydraulic regime the rameters change over time, resulting in one map for one time step. Changes of catchment

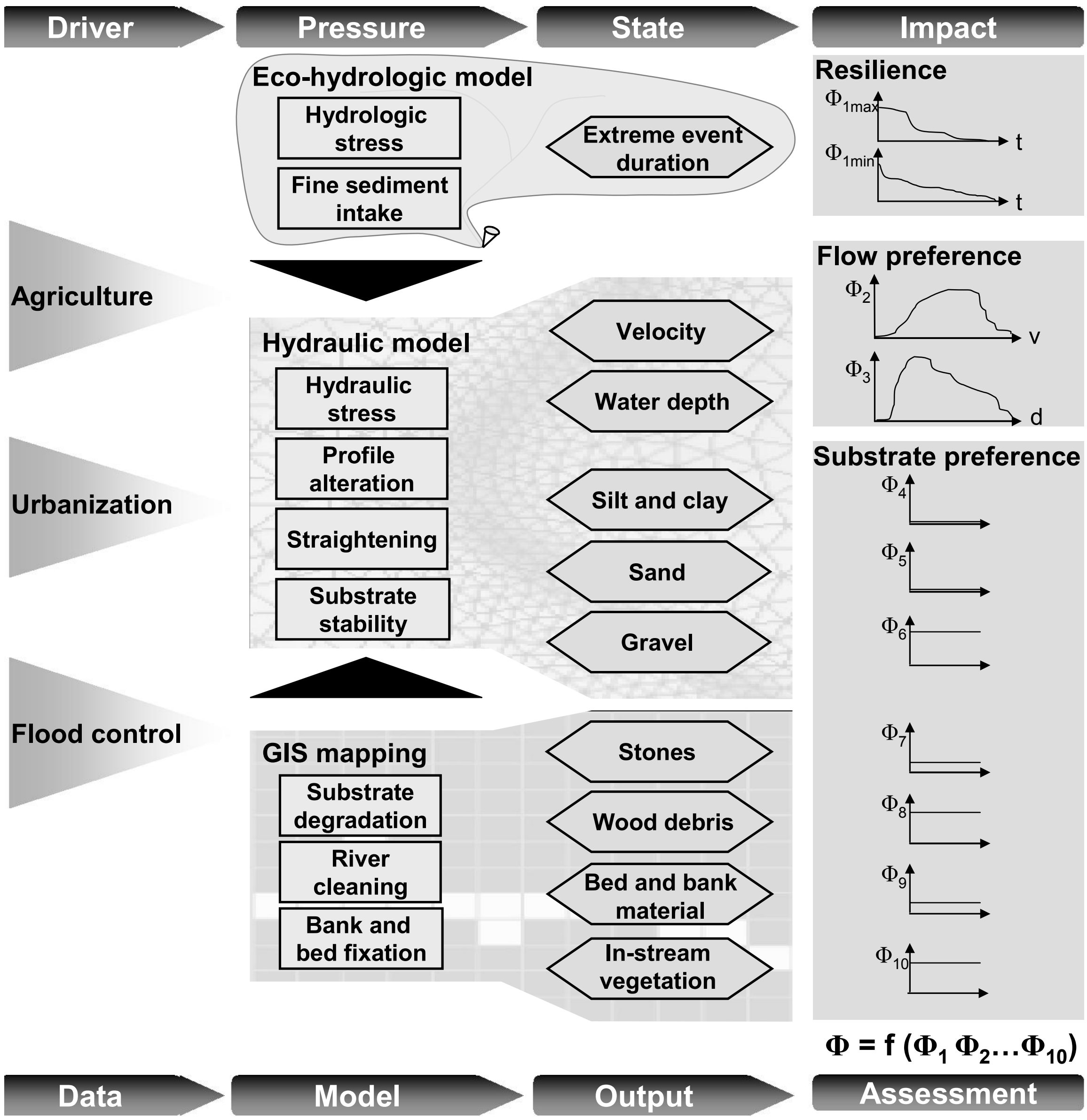

Figure 2.1 Implementing the DPSI concept in the habitat modelling system

properties and stream structure thus are causing an alteration in the hydraulic habitat parameters. Depending on the time period of interest and the type of the rehabilitation project, the output time step can be daily up to yearly. The state of *stones, wood debris, bed and bank material* and *instream vegetation* is directly derived from the GIS maps, thus allowing an easy implementation of potential instream rehabilitation scenarios.

The term *impact* describes the consequences of the altered state, in particular changes in substrate and flow properties on the macroinvertebrate fauna. In order to depict the current state of the macroinvertebrate fauna with the modelling system, parameter functions have to be derived (VanBroekhoven et al., 2006; Li et al., 2008) for species regularly occurring in the stream. Two datasets of macroinvertebrate samples are available (Brinkmann, 2002; LANU, 2006a) listing several species occurring in the study stream. *Parameter functions* (Φ_1 – Φ_{10}) between resilience to extreme discharge events, current patterns, substrate composition and species abundance are derived from a knowledge-based database (Euro-Limpacs, 2009) and from data collected in comparable lowland streams (Kramm, 2002; Wenikajtys, 2004). The parameter functions define the suitability for certain substrates and current velocity classes for each species. The functions will be weighed and interconnected by a decision tree diagram for each species. Each decision tree is then implemented in the GIS and applied on each time step map. The result of each decision tree is the *habitat suitability* (Φ), which leads to a dynamic habitat description for each species.

A validation of the knowledge-based parameter functions and the overall model results is necessary. Based on substrate and flow specific macroinvertebrate samples the model performance will be evaluated by assessing how well the model is capable of reproducing the status quo of the aquatic habitat.

2.4 Study area and data

The modelling system will be tested in the 50-km^2 Kielstau catchment. The catchment is located in northern Germany in the state of Schleswig-Holstein as part of a lowland area (Figure 2.2a). The mean annual precipitation and temperature are 893 mm and 8.3 °C, respectively (DWD, 2010). Land use is dominated by arable land and pasture. There are only few small villages and detached farms (Figure 2.2b). From the source to the catchment outlet, the Kielstau has a total length of 16.2 km and a mean gradient of 1.2 ‰. The topography in the catchment ranges from 78 m to 27 m ASL, is flat but relatively uneven with rolling hills and numerous depressions (Figure 2.2c). The prevailing soils are Haplic and Stagnic Luvisols, while the river valleys are characterised by peat soils (Figure 2.2d). About 5 km downstream of its origin, the Kielstau flows through Lake Winderatt, which has a surface area of 0.24 km^2. Downstream of Lake Winderatt two large tributaries, the Moorau and the Hennebach, and various drainage pipes and open ditches discharge into the Kielstau. The location and extent of drained areas within the catchment has been estimated by Fohrer et al. (2007) using a GIS-based methodology. The fraction of drained area in the catchment is estimated to be approx. 31 % (Fohrer et al., 2007; Figure 2.2d). Close to the catchment outlet the gauging station Soltfeld is located, which is part of the official gauging network of the Federal State Schleswig-Holstein.

The hydrology is characterised by agricultural drainage, near-surface groundwater, low hydraulic gradients and thus a high interaction between groundwater and surface water. Many parts of the Kielstau have been changed markedly during the reallocation of land from its natural course. In these areas, the river has been straightened, incised and thus disconnected from its flood plains. Here, hydromorphological variety and value is relatively low, while near-natural river sections still exist and can act as reference points (DAV-WBV/LAND S.-H., 2006). The overall morphological state of the stream is assessed as 'poor' to 'moderate' (Olbert et al., 2006) according to the standard hydromorphological river survey method in Germany (LAWA, 2000) and is typical for many streams in northern Germany. Nevertheless, the Kielstau is part of the FFH (EC, 1992) protection area and 175 ha of land along the river and around Lake Winderatt are owned by two nature conservation foundations which increases the potential for river rehabilitation measures within and beyond the scope of the WFD.

The macroinvertebrate assemblage of the Kielstau was assessed by Brinkmann (2002) and the Schleswig-Holstein State Agency for Nature and Environment (LANU, 2006a); further data have been generated in the framework of the present study. The macroinvertebrate community is mainly composed of generalists inhabiting lakes and lentic zones of streams, in particular snails (Gastropoda), mayfly larvae (Ephemeroptera), beetles (Coleoptera), caddis larvae (Trichoptera) and midges (Chironomidae). Dominant feeding types are grazers (mayfly larvae and snails), filter feeders (several caddis larvae and mussels) and shredders (Amphipoda). More specialised species include various caddis larvae feeding on dead wood (e.g. *Lype reducta*).

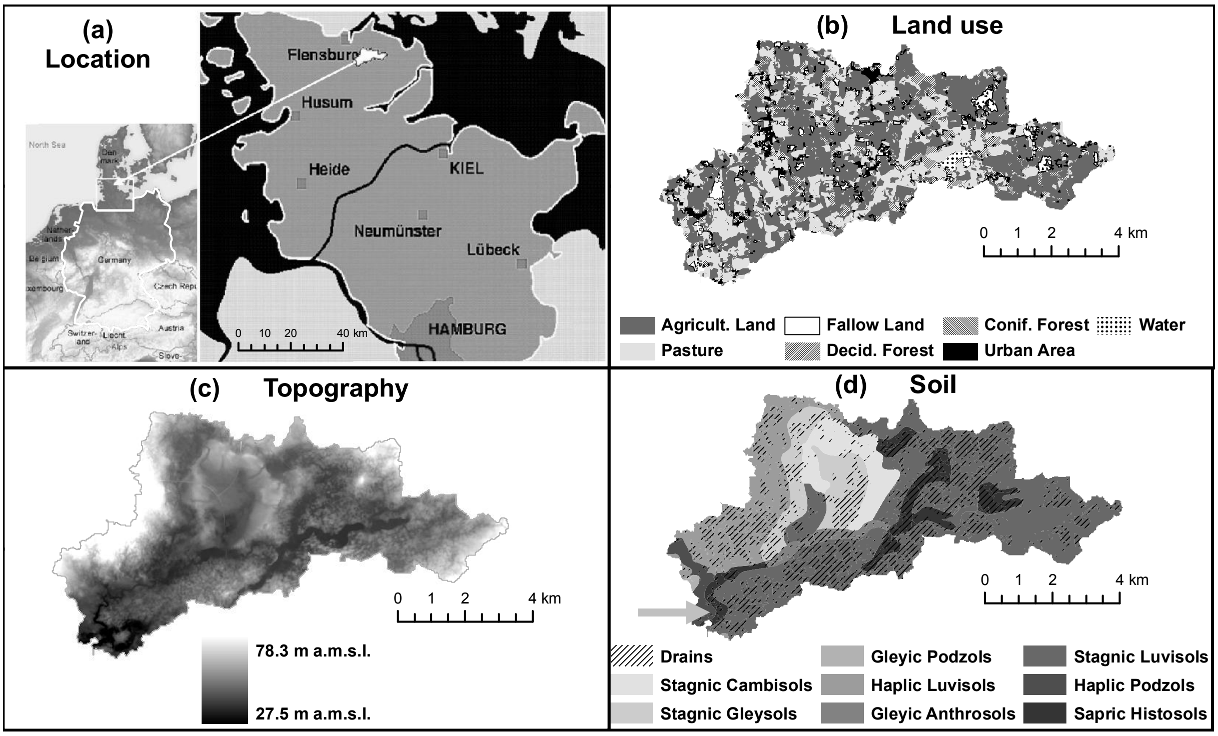

Figure 2.2 Location and properties of the Kielstau catchment: location in Germany (a) (Jose, 2006; LVA 1992-2004); land use (b) (DLR, 1995); topography (c) (LVA 1992-2004); soil (BGR, 1999) with drained areas (Fohrer et al., 2007), the grey arrow marks the location of the hotspot (d).

The data described above are available for the whole catchment. A 230-m-long stream section was chosen where instream measures to improve the aquatic habitat will be tested. For this hotspot, additional data has been gathered in order to apply the modelling system (grey arrow in Figure 2.2d). Here, channel topography data has been surveyed (soilAQUA, 2009), discharge rating curves have been established and a morphological river mapping campaign including a sediment analysis has been conducted (Thiemann, 2008). The following ten substrates have been recorded and their distribution has been digitised on GIS maps: fascines, alder trees, water plants, dead wood, coarse particulate organic matter, clay, sand, gravel, cobbles and stones.

To evaluate model performance, macroinvertebrate sampling was carried out in spring 2008 and spring 2009. Within the first sample campaign, each substrate has been sampled eight times, resulting in 80 sampling sites. Additionally, actual flow velocity and water depth has been measured on all sites. Oxygen concentration, water temperature, electrical conductivity and pH have been recorded on all sampling days and daily continuous measurements of the nitrogen and phosphorous fractions were conducted at the stream section since 2006.

2.5 Results

The data representing the drivers has been gathered and implemented in ArcGIS and the submodels (Table 2.1). The hydrological model SWAT has been applied to depict the pressures *hydrologic stress* and *fine sediment intake* on the catchment scale. The modelled and measured daily discharge hydrographs for the 5-year calibration period show a very good fit. Moriasi et al. (2007) suggested performance tests which are all passed by the model: Nash-Sutcliffe-Efficiency (NSE) of 0.80, Percent bias of 6.34 and a Root Mean Square Error (RMSE) of 0.19 (Figure 2.3a). In order to achieve a good model performance it was of particular importance to consider the two lowland characteristics landscape depressions and agricultural tile drains. Therefore, the surface water storage potential has been derived from high quality topographic data and has been implemented in the model together with the distributed drainage map. The tile drain parameters have been used within plausible ranges for model calibration.

Sediment input in the lowlands mainly consists of three sources: fields, agricultural drains and the river banks. It is possible to depict sediment input from agricultural fields, but so far no model can account for sediment input from tile drains. Due to this, a GIS-based methodology has been developed to estimate the sediment entry pathway share in the Kielstau catchment resulting in sediment input of 15 % from fields, 15 % from drains and 70 % from river banks (Kiesel et al., 2009b). Based on this estimation, a drainage flow sediment concentration has been derived and implemented in SWAT. Suspended sediment data are available since September 2006 and thus the model has been applied to the time period displayed in Figure 2.3b. The graph shows an underestimation of modelled sediment concentration to measured sediment concentration (Figure 2.3b) because the model considers sediment input from fields and drainages only, while the samples are conducted with the total sediment concentration of the stream which comprises the bank entry pathway. Including the separated depiction of the bank erosion processes with the hydraulic model will improve the result.

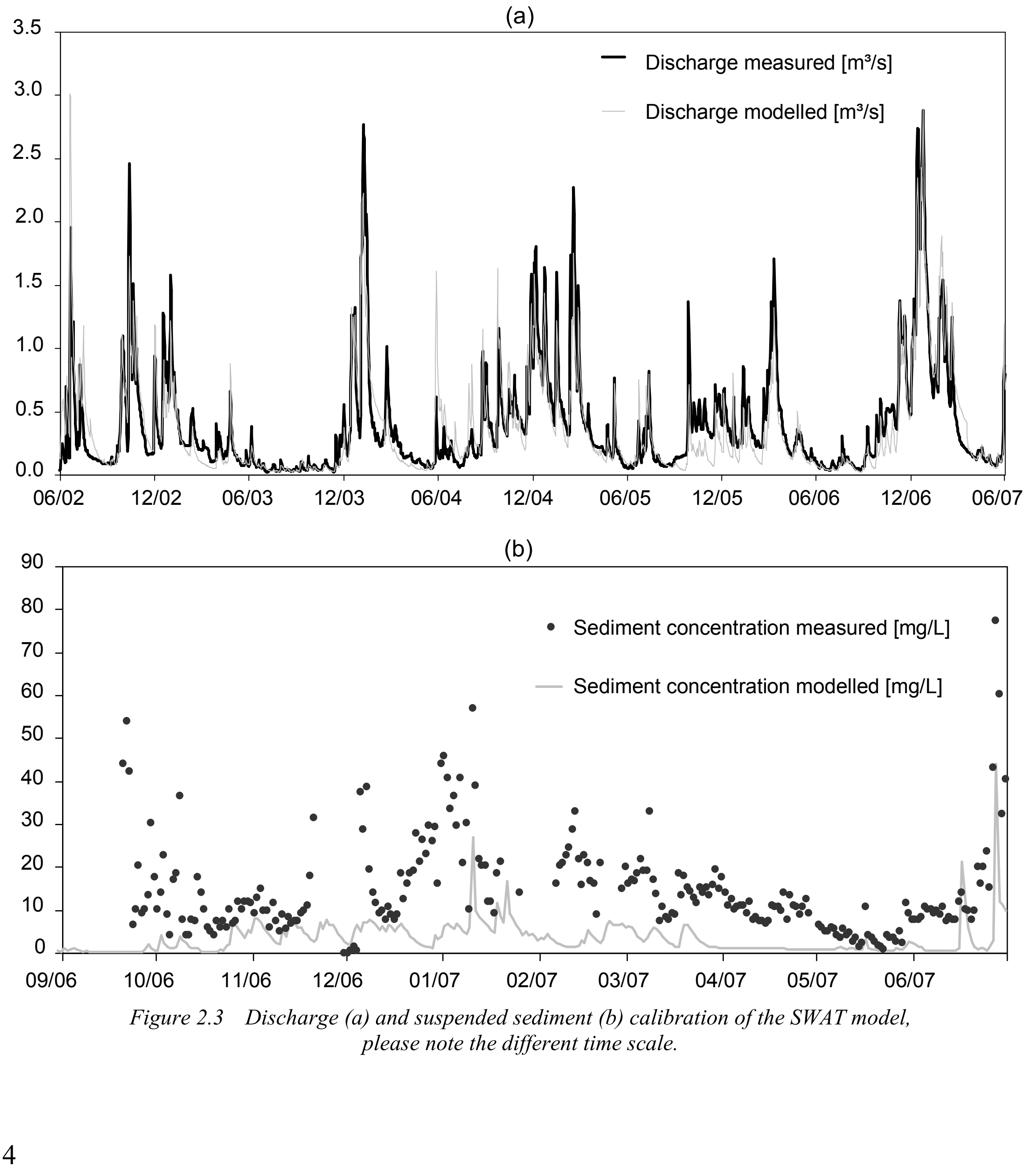

Figure 2.3 Discharge (a) and suspended sediment (b) calibration of the SWAT model, please note the different time scale.

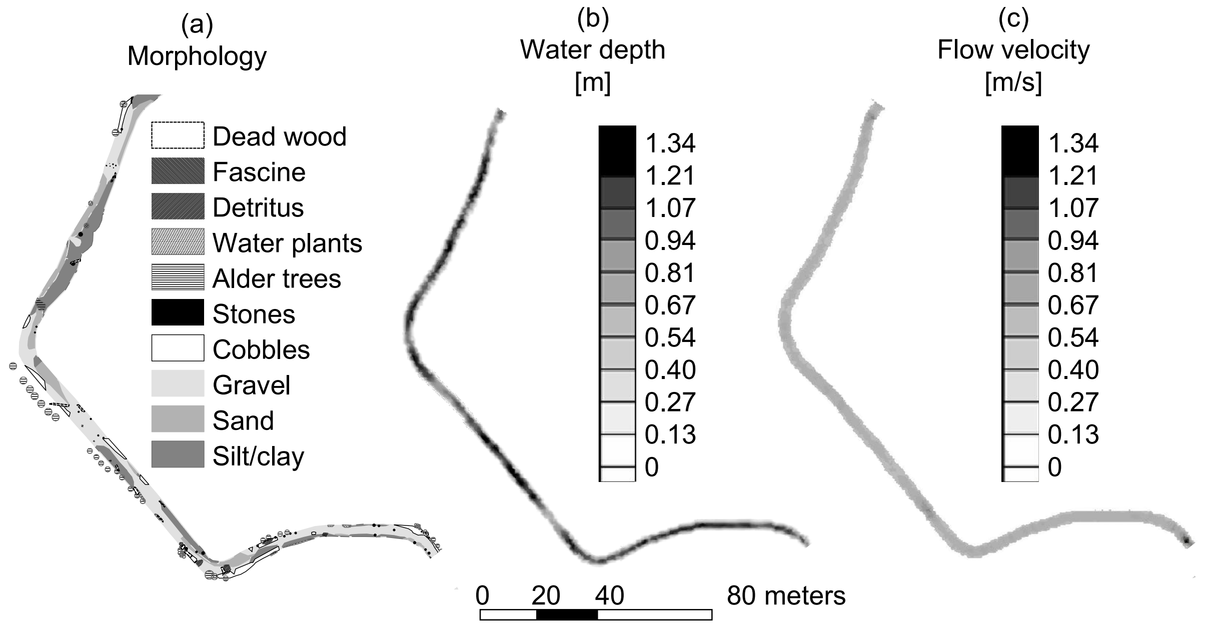

Figure 2.4 GIS mapping of observed substrates (a); AdH model results (from October 2008): water depth (b) and flow velocity (c) distribution.

The GIS mapping took place at a 230-m-long section of the Kielstau where the *substrate degradation* and *bank and bed fixation* have been mapped (Thiemann, 2008). *River cleaning* is not occurring within that stream reach. The resulting digital map shows the distribution of ten different substrates (Figure 2.4a).

The hydraulic model AdH has been applied on this stream section. Figure 4b and c show the resulting depth and velocity distribution of a steady state run for a high discharge value. As the stream reach bathymetry needs to be further refined with the aid of the digital GIS map, and the SWAT hydrograph and sediment load graph has not yet been linked, these are only preliminary results. Especially for the time-dependent simulation of the current patterns and the substrate, it is necessary that the hydrological discharge regime is linked to AdH.

The parameters descriptively shown in Figure 2.4 form the base for the proposed simulation of the macroinvertebrate habitat on the micro scale.

2.6 Discussion

A methodology has been introduced on how the impact of anthropogenic changes of catchment and river properties on the macroinvertebrate habitat can be assessed. The first step is the depiction of the *status quo*, while the strength of the modelling system is the capability of assessing the influence of changes on the catchment scale and also the effect of instream measures on potential indicator species. Therefore, the DPSI concept (EEA, 1999) is utilised within a GIS-based modelling system on the example of a 230-m-long river section located in a German lowland catchment. Three submodels are used to depict the current state of necessary habitat parameters by incorporating the drivers into a hydrological and a hydraulic model and GIS maps. The results show that the hydrological discharge simulation performs well in comparison to the measured data and is capable of depicting the scale and duration of extreme events that can cause hydrological stress. Considering, that the fine sediment input from fields and agricultural drainages accounts for about 30 % of the total sediment contribution, the model results show a reasonable depiction of fine sediment intake. The preliminary results of the steady state hydraulic model application emphasise the need for a dynamic link of catchment and instream processes so that the change in flow characteristics and substrate is based on the hydrological regime. In order to depict the current status of the stream, field mapping of the morphology is necessary. The natural seasonal variation of stream properties, especially the vegetation, can be considered and it is the only possibility to include important small-scale habitats like wood debris, water plants, stones and artificial structures into the modelling system. Furthermore, the influence of habitat upgrading measures can be conveniently assessed by modifying the digital morphological GIS maps.

Concerning the evaluation of the parameter functions, different demands of species during its life cycle have to be extracted from the database and have to be considered in the decision tree for each particular species. The emphasis on constructing the GIS-based modelling system lies: First, on a smooth data handling between the hydrological and the hydraulic model; second, the direct usability of the GIS maps for the hydraulic model by linking roughness values to the surface type; and third, to keep the data transfer within a manageable size by simplifying long-term daily hydrographs to time-dependent flow classes for computing quasi-steady model runs.

2.7 Acknowledgement

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Chapter 3 Incorporating landscape depressions and tile drainages of a northern German lowland catchment into a semi-distributed model

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Abstract Hydrological models need to be adapted to specific hydrological characteristics of the catchment in which they are applied. In the lowland region of northern Germany, tile drains and depressions are prominent features of the landscape though are often neglected in hydrological modelling on the catchment scale. It is shown how these lowland features can be implemented into SWAT. For obtaining the necessary input data, results from a GIS method to derive the location of artificial drainage areas have been used. Another GIS method has been developed to evaluate the spatial distribution and characteristics of landscape depressions. In the study catchment, 31 % of the watershed area is artificially drained, which heavily influences groundwater processes. Landscape depressions are common over the 50-km^2 study area and have considerable retention potential with an estimated surface area of 582 ha. It was the scope of this work to evaluate the extent by which these two processes affect model performance. Accordingly, three hypotheses have been formulated and tested through a stepwise incorporation of drainage and depression processes into an auto calibrated default setup: (1) integration of artificial drainage alone; (2) integration of depressions alone and (3) integration of both processes combined. The results show a strong improvement of model performance for including artificial drainage while the depression setup only induces a slight improvement. The incorporation of the two landscape characteristics combined led to an overall enhancement of model performance and the strongest improvement in r^2, RMSE and NSE of all setups. In particular, summer rainfall events with high intensity, winter flows and the hydrograph's recession limbs are depicted more realistically.

Keywords tile drains; potholes; sinks; DEM; ArcGIS; SWAT

3.1 Introduction

Through the incorporation of hydrometeorologic, geomorphologic, agricultural, pedologic, geologic and hydrological data, watershed models can be sufficiently adapted to almost all regions all over the world (Singh and Woolhiser, 2002). In the lowland region of northern Germany, there are two additional factors that influence hydrological conditions which are difficult to incorporate into models using standard data sources and techniques: First, the comprehensive implementation of tile drainages in the past induced major changes in catchment hydrology and second, depressions which are usually removed from topographic data can be an abundant landscape feature.

Along with the intensification of agriculture during the last century, land use changes and management adaptations led to substantial variations in catchment hydrology (Krause et al., 2007a). Drainages have been implemented to optimize the soil moisture conditions (Eggelsmann, 1981) for widening the crop choice, to become more independent from seasonal weather constraints, to improve the impact of fertiliser (Smedema et al., 2000), for enlarging cultivatable area, for improving trafficability of the fields and agricultural productivity (Ritzema, 1994). Smedema et al. (2000) estimated that 10–14 % of cropland worldwide is artificially drained and Feick et al. (2005) showed that on each 10 km^2 of agricultural land in England, the Netherlands, Denmark, northern Germany, southern Sweden and Poland drainages have been implemented in fractions of 5–100 %. Due to the fast distraction of excessive stored soil water, drainages have a decisive influence on hydrological flow pathways (Stone and Krishnappan, 2002), especially in highly groundwater-influenced lowland regions where tile drain flow primarily forms the fast flow component (Northcott et al., 2002). In such regions, surface runoff plays a minor role and thus drainage flow should be considered in hydrological modelling (Yuan et al., 2000). The impact of drains can be sufficiently modelled on the field scale (Vepraskas et al., 2006) and on the catchment scale with models such as MOdelling Nutrient Emissions in RIver Systems (MONERIS) (Behrendt et al., 2007), ArcEGMO (Klöcking et al., 2009) and SWAT (Arnold et al., 1998). Sogbedji and McIsaac (2002) point out that there is often a lack of information about the location and characteristics of the tile drain system, which can hamper the incorporation of drainages on the catchment scale. Having been aware of this restriction, Fohrer et al. (2007) developed a GIS methodology to derive the spatial distribution of tile-drained areas using the example of a lowland catchment in northern Germany. These results are used within this study as data input for the SWAT drainage algorithms.

The characterisation of lowland depressions at the catchment scale is limited by similar spatial data deficiencies, leading to neglect in spatially distributed hydrological modelling. Depressions, commonly referred to as sinks, pits, potholes, billabongs or pans are landscape features that can be found all over the world (Wentworth, 1944; Covich et al., 1997; Smerdon et al., 2005; Kalettka and Rudat, 2006; Colburn, 2008; Karlen et al., 2008; Neigh et al., 2008). Depending on their historical formation, properties and location, different expressions are used to address these landscape features. For example, in areas affected by the last ice age, many kettle holes have formed (Grube et al., 1986; Sibrava et al., 1986). The only comprehensive term that includes all mentioned depressions is ‘temporary waters’ (Colburn, 2008). These temporary waters have an influence on the hydrology because they intercept and store surface runoff (Hayashi and Van der Kamp, 2000; Antonic et al., 2001), which increases the availability of water for evaporation and infiltration. Thus, modelling hydrological processes in a lowland is often accompanied by extensive hydrological calibration of water retention and evaporation parameters due to a lack of physically based retention data: Schmalz et al. (2008a) used additional ponds and wetlands and Hörmann et al. (2007) used additional wetlands for calibrating streamflow successfully. Digital elevation models (DEMs) are widely used to describe catchment topography (Thompson et al., 2001). However, when using hydrological models and depicting the topography with DEMs, closed depressions are treated as errors and erased during the catchment delineation process (Tarboton et al., 1991; Grimaldi et al., 2007). This filling procedure increases the elevation of depression cells to that of the surrounding cells and eliminates areas that could have had actual water retention potential (Martz and Garbrecht, 1999). This has been justified because of the reasonable assumption that closed depressions are mainly spurious DEM features (Hutchinson, 1988; Martz and Garbrecht, 1998). But this is not supportable for modern, more accurate DEMs and in regions where depressions are present in the landscape (Moore et al., 1991; Lindsay and Creed, 2006). Instead, Du et al. (2005) emphasise that it would rather be desirable to develop a GIS methodology to determine depression parameters for hydrological modelling. One such developed GIS methodology described within this study follows that suggestion. Similar to the mentioned tile drainage approach, these results are used to apply the SWAT pothole algorithms.

It is the aim of this study to provide an assessment of incorporating drainages and depressions derived from readily available model input data into the SWAT model of an agriculturally used lowland catchment. Therefore, three hypotheses are to be evaluated in this paper: (1) that the implementation of artificial drainages alone improves model performance; (2) that the implementation of depressions alone improves model performance and (3) that the combined implementation of artificial drainages and depressions leads to the best model performance.

3.2 Materials and methods

Figure 3.1 gives an overview of how the evaluation of the hypotheses within this paper is carried out. The major assessment is a stepwise incorporation of tile drainages and depression processes into an auto calibrated default setup. Thereafter, each box is described in detail in a corresponding section.

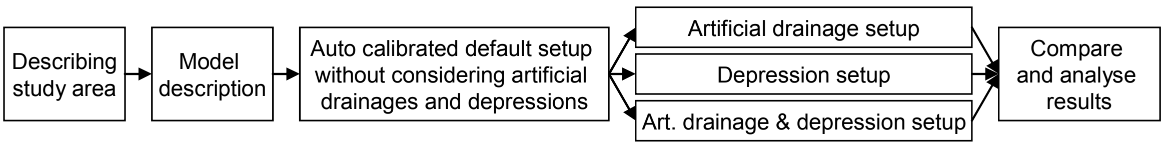

Figure 3.1 Methodology of assessing drainages and depressions

3.2.1 Study area

The study area is the 50-km^2 Kielstau catchment, located in northern Germany (Figure 3.2a). The mean annual precipitation and temperature are 893 mm and 8.3 °C, respectively (DWD, 2010). The land use of the rural catchment is dominated by arable land (56 %), pasture (26 %), forest (8 %) and small urban settlements (3 %), (DLR, 1995) (Figure 3.2b). From the source to the catchment outlet at the gauging station Soltfeld, the stream Kielstau is 16.2 km long, has a mean gradient of 1.2 ‰ and flows through a small lake with a surface area of about 0.2 km^2.

The rolling hills topography of this lowland region (Figure 3.2c) was heavily influenced by glacial and periglacial processes of the late Pleistocene (Lundquist, 1986; Sommerhäuser and Schuhmacher, 2003). The Kielstau flows through a valley that has been eroded by subglacial melt water that discharged under high pressure beneath the ice sheets (Riedel and Polensky, 1987). Landscape depressions have been formed under the once ice-covered lowland region due to isolated ice blocks covered by till, which delayed melting and thus leaving hollow moulds on the surface (Wahnschaffe and Schucht, 1921; Briem, 2003). During field trips to the study area, it has been observed that after heavy rain events depressional areas are filled with water. According to Riedel and Umland (1983) and LANL (1989) at least seven depressional areas are present per 100 ha. Due to this, the region is a characteristic example of

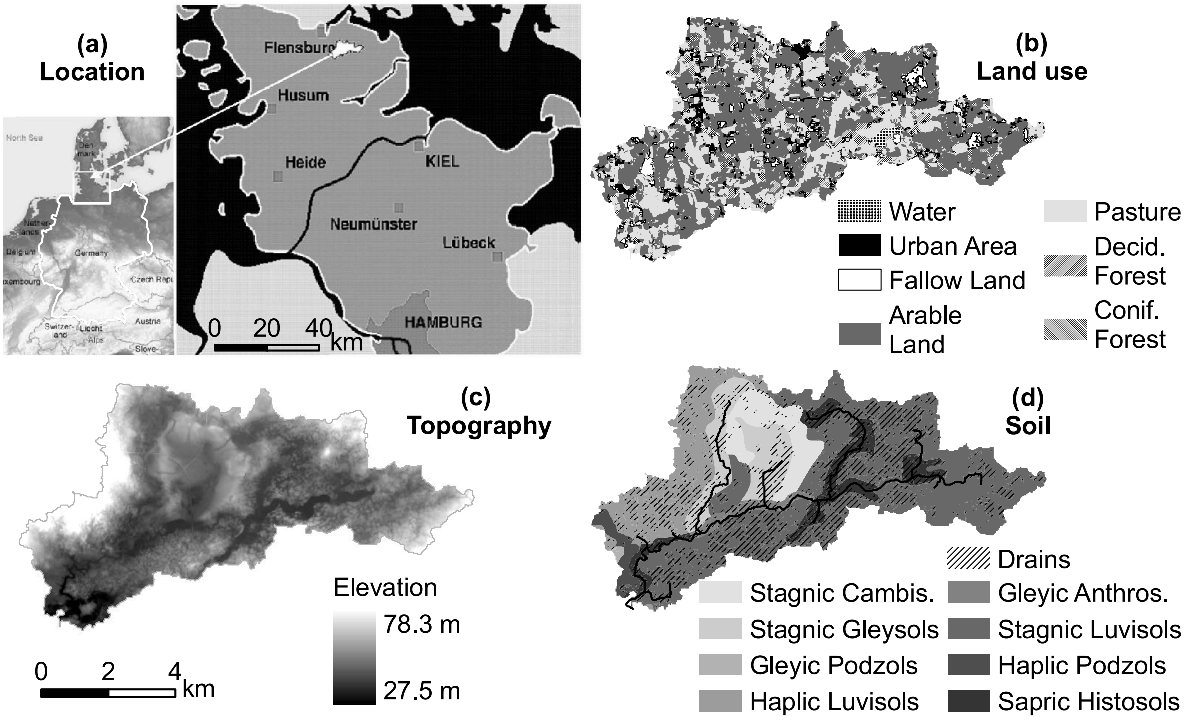

Figure 3.2 Location and properties of the Kielstau catchment: location in Germany (a) (LVA, 1992–2004; Jose, 2006); land use (b) (DLR, 1995); topography depicted on 5-m DEM (c) (LVA, 1992–2004); the river network (DAV-WBV/LAND S.-H., 2006), the soil (BGR, 1999) with drained areas (d) (Fohrer et al., 2007)

young moraine landscapes (Lorentzen, 1938). During wet periods, many depressions are filled with water which evaporates and slowly infiltrates into the soil. Typical for the region are the predominant sandy and loamy soils in the catchment with Haplic Luvisols in the eastern and Stagnic Luvisols in the western part, whereas Sapric Histosols are present along the stream and its tributaries (BGR, 1999) (Figure 3.2d). Theoretically, the landscape would have a high water retention potential due to the peat soils, a fen at the lake and flat floodplains. However, extensive drainage measures have been implemented during the reallocation of land, mainly from the 1950s to the 1980s (MELF, 1980) to secure agricultural productivity (Riedel and Polensky, 1987).

Due to the high density of actual depressions in the study area (Beuck, 1996) as well as the intensively drained agricultural areas (Fohrer et al., 2007), the Kielstau catchment is a suitable test area for applying and testing the developed methodologies.

3.2.2 Model description

The two lowland features, drainages and depressions, are assumed to influence lowland hydrology, and to test the influence the hydrological model has to be capable of depicting the processes on the catchment scale realistically. For this study, version 2005 of SWAT (Arnold et al., 1998) was chosen. For the generation of the input files and the pre-processing of the input data, the GIS interface ArcSWAT (Winchell et al., 2007) was used. SWAT has been applied to various regions and within a multitude of projects all over the world (Arnold and Fohrer, 2005; Gassman et al., 2007). It is suitable for the application in lowland catchments as the implemented processes have been successfully tested in watersheds with flat topography, low hydraulic gradients, shallow groundwater, a high potential for water retention in peatland and lakes (Schmalz et al., 2008b), for tile-drained sites (Ahmad et al., 2002; Fohrer et al., 2007) and potholes in the landscape (Du et al., 2005). The model can simulate the water balance, nutrients and pesticides, field erosion, plant growth cycles, management practices and water bodies on a daily time step for continuous simulations over long time periods (Neitsch et al., 2005a). This paper deals with the integration of spatially distributed watershed features that influence the water balance. Hence, it is important to understand the model's approach to simulate these hydrological phenomena.

Based on the DEM and user-defined stream threshold values ArcSWAT partitions the watershed into subbasins. These are then further divided into hydrological response units (HRUs). Each HRU is a unique combination of soil, land use and slope class within a subbasin. One HRU thus represents a certain area with specific physical and hydrological properties. Consequently, a higher heterogeneity of these properties in a subbasin leads to a larger number of HRUs. Even though HRUs loose their spatial reference within the subbasin, the HRU quantity and the area ratio are affected by incorporating additional regional features. It is therefore necessary to have the physical properties of drainages and depressions available on a spatially distributed map.

For each HRU that is defined as being drained, SWAT calculates the tile drain discharge for each time step according to equation {3.1} (adapted from Neitsch et al., 2005a):

$$q_{tile} = \left(\left(\frac{h_w - h_d}{h_w} \times (SW - FC) \times \left(1 - e^{\left[\frac{-24}{t_d}\right]} \right) \right) + q_{tilestore^{i-1}} \right) \times \left(1 - e^{\left[\frac{-24}{t_l}\right]} \right) \quad \text{if} \quad h_w > h_d \qquad \{3.1\}$$

where q_{tile} (mm H_2O) is the tile drain discharge; h_w (mm) is the height of the water table perched atop the impervious zone which is calculated based on the water table height from the previous time step, the user-defined baseflow recession constant and the recharge to the aquifer at the current time step; h_d (mm) is a user-defined parameter and represents the height of the tile drain above the impervious zone, SW (mm H_2O) is the soil water content, FC (mm H_2O) is the field capacity water content of the soil; t_d (h) is a user-defined parameter to account for the time required to drain the soil to field capacity; $q_{tilestore}^{i-1}$ (mm H_2O) is the amount of tile flow stored from the previous day and t_l (h) is a user-defined parameter to take account for the velocity of the tile discharge. It specifies the lag time between the instant the water enters the drain and the moment the tile discharge enters the river channel. Drain discharge for a given day only occurs if the soil water exceeds field capacity and if the height of the water table exceeds the height of the drain.

Depressions have water storage capability and can retain water. Thus, they can have a significant effect on stream flow and the hydrological balance (Du et al., 2005). SWAT is capable of incorporating water retention based on a volume approach. For defined surface water storages within an HRU or subbasin, SWAT calculates the water balance for each time step according to equation {3.2} (Neitsch et al., 2005a):

$$V = V_{stored} + V_{flowin} - V_{flowout} + V_{pcp} - V_{evap} - V_{seep} \qquad \{3.2\}$$

where V (m^3 H_2O) is the water volume for the current time step, V_{stored} (m^3 H_2O) is the water volume from the last time step, V_{flowin} (m^3 H_2O) is the inflow of water from adjacent areas calculated by the surface runoff and the extend of the user-defined contributing area to the depressional storage element, $V_{flowout}$ (m^3 H_2O) is the overflow occurring when the maximum user-defined storage volume is exceeded, V_{pcp} (m^3 H_2O) is the volume of precipitation falling on the surface area, V_{evap} (m^3 H_2O) is the volume of water extracted by evaporation from the surface area calculated by the Penman–Monteith equation. If the plant leaf area index exceeds a threshold of three, evaporation from the open water surface is restricted. V_{seep} (m^3 H_2O) is the volume of water that infiltrates into the soil depending on the hydraulic conductivity. The internal calculation of the surface area is carried out by assuming that each pothole is cone shaped.

For further information about the calculation of the variables, please refer to Neitsch et al. (2005a). The two equations include all major hydrological processes of drainages and depressions and the usage for this study is assessed as suitable. The suitability of applying equations {3.1} and {3.2} on the HRU-scale in a meso-scale catchment, their parameter sensitivity and degree of uncertainty has been tested by Du et al. (2005) and found to be acceptable. The next step of the methodology is the setup of an optimum base model of the study catchment without drainages and depressions.

3.2.3 Setup and auto calibration description

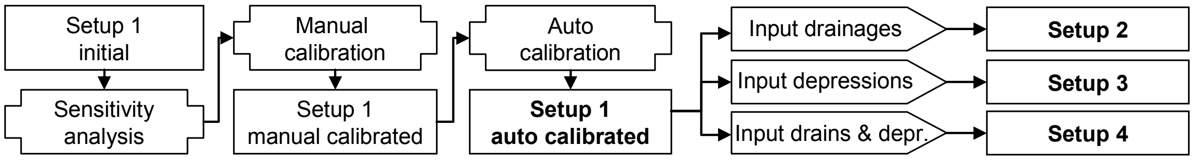

Figure 3.3 Setup and calibration flow chart

This section explains the usage of different model setups for testing the three formulated hypotheses. The influence of drainages and depressions on the model performance is tested through using an auto calibrated model setup. Figure 3.3 demonstrates the methodology for an unbiased assessment: the initial setup is based on the available data. After manual and automatic sensitivity analyses (Table 3.1), *setup 1 manual calibrated* results from a first manual calibration, which is carried out to reach at least positive NSE (Nash and Sutcliffe, 1970). Optimisation of this setup is carried out with the shuffled complex evolution algorithm with the sum of the squares of the residuals as the objective function (VanGriensven et al., 2006) to fit observed to measured daily discharge and to find the optimum parameter set. The parameters, the values and ranges for the auto calibration are displayed in Table 3.1. Additional parameter information is supplied in Neitsch et al. (2005b). The manual calibration, the sensitivity analysis and the auto calibration revealed a strong influence of groundwater parameters, which was also observed by Schmalz and Fohrer (2009). Auto calibration is limited to the eight most sensitive parameters. Less sensitive parameters are either well depicted in the soil borehole data, calculated based on measurements or excluded due to their insensitivity to restrict the number of model runs. It was made sure that the auto calibration routine changes the parameters only in plausible ranges by defining thresholds to obtain an appropriate, unbiased and optimum parameter set. The ranges in Table 3.1 were chosen based on assumptions of the

Table 3.1 Sensitivity analysis results and auto calibration parameters: parameter rank, parameter with a short explanation, manual calibrated values to reach positive NSE which are the start values for the auto calibration, minimum and maximum parameter range for auto calibration and auto calibrated end value

SA rank	Parameter	Parameter explanation	Start value	min	max	AC unit	End value
1	GWQMN	Threshold depth of water in the shallow aquifer for return flow to occur (mm)	0	0	5000	mm	0
2	CN2	Initial SCS runoff curve number for moisture condition II (-)	calculated by slope	-10	+10	%	9.980
3	CH_K2	Effective hydraulic conductivity in main channel alluvium (mm hr^{-1})	0.5	0.1	75	mm h^{-1}	0.1
4	ESCO	Soil evaporation compensation factor (-)	0.95	0	1	Abs	0.852
5	SOL_AWC	Available water capacity of the soil layer (mm mm^{-1})	soil.dat	-25	+25	%	25
6	SURLAG	Surface runoff lag time (-)	1	0	10	Abs	0.277
7	CH_N2	Manning's n-value for the main channel (-)	0.03	0.01	0.05	Abs	0.015
8	GW_REVAP	Groundwater revap coefficient (-)	0.3	0.1	0.3	Abs	0.241

uncertainty of each default value for each input parameter. *Setup 1 auto calibrated* represents the optimum model if drainages and depressions were not considered. Based on the setup explained in the following section, the formulated research hypotheses are tested by three further setups: (1) solely incorporating the drainage parameters (*setup 2*); (2) solely incorporating the depression parameters (*setup 3*); (3) incorporating both drainages and depressions (*setup 4*). No further calibration is carried out for *setup 2*, 3 and 4. The model parameter set is the same as in *setup 1*, except for the drainages and depression parameters. It is assumed that the auto calibration routine tries to compensate for the lacking drainage and depression processes within the defined ranges in *setup 1*. If the obtained model can then be further enhanced by incorporating the actual drainage and depression processes, it is considered not possible to compensate for the lacking processes in a usual model setup and a true model improvement is inferred. Each setup is run for the same time period and compared with the previous setup to evaluate if the implemented measures improve the auto calibrated model (hypotheses 1 and 2) and if the joint depiction of both drainages and depressions lead to the best model (hypothesis 3).

3.2.4 Primary data implementation and calibration—*initial setup* and *setup 1*

The basic model setup is carried out in three consecutive steps: First, the catchment and subbasin delineation is conducted with a 25-m DEM (LVA, 1992–2004) and stream network data (DAV-WBV/LAND S.-H., 2006), resulting in 17 subbasins. Second, HRUs are defined based on slope classes and equal land use and soil areas. Spatial land use information is derived from a 25 x 25-m resolution satellite scan (DLR, 1995; MOBIO, 1999) and linked to the corresponding land use from the SWAT databases. Crop rotations, sowing and harvest dates, fertiliser applications and tillage operations were researched by Bieger (2007). Soil distribution was determined using a 1 : 200,000 digital soil map (BGR, 1999). Selected soil properties are deduced from data on 656 boreholes in and around the catchment (LANU, 2006b). Physical properties of soil including organic carbon content, rooting depth, hydraulic conductivity, grain size fractions and bulk density are taken from Ad-Hoc-AG Boden (2005), Janßen (2006) and Succow and Joosten (2001). The moist soil albedo is calculated according to Baumer (1990) and Post et al. (2000). Third, additional spatial and temporal data affecting the hydrology are included: mean discharge data for six municipal wastewater treatment plants (Andersen, 2006), geometric properties of Lake Winderatt (Grudzinski, 2007), daily climate values from 1993 to 2008 on precipitation, temperature, humidity and wind from the German Weather Service (DWD, 2010) and a yearly solar radiation curve (IFM, 2007). For the calibration and validation of the model, daily discharge data are available for the gauging station Soltfeld at the catchment outlet from 1986 to 2008 (LKN, 2010). Due to limited available climate data, the five-year calibration period was chosen from 1999 to 2004, the five-year validation period from 2004 to 2009. The periods include all typical hydrological patterns, spring flood events with the second highest recorded flow in history, drought periods and short, high-intensity summer storm events.

Based on this initial setup both manual and automatic sensitivity analysis (VanGriensven et al., 2006) have been carried out after the baseflow recession constant has been calculated (Arnold et al., 1995) from observed discharge data and the Soil Conservation Service - Curve Number (SCS-CN) values have been modified according to the slope of the HRU. Within a first manual calibration, groundwater parameters, soil available water capacity and soil hydraulic conductivity have been adjusted. *Setup 1* is then auto calibrated and represents the initial condition for the implementation of the drainages and depressions. To incorporate these two landscape features, the hydrological parameters have to be evaluated. It is difficult to derive plausible spatially distributed information on the catchment scale based on area-wide available data. Two GIS-based methodologies are presented on how to approach this task.

3.2.5 Incorporating drainages—*setup 2*

The approximate area and years in which agricultural tile drainage and ditch draining were implemented in the northeast of Schleswig-Holstein is known (MELF, 1980). However, the spatial distribution and the extent of the drained area within certain catchments are unknown as the drain-measures were not recorded on a land register or other maps in a comprehensive manner. To incorporate drainages into hydrological models, it is necessary to successfully estimate the location and occurrence of drained areas, which has been achieved by Fohrer et al. (2007). Based on detailed soil classification for agricultural land from the mid-20 th century (LANU, 2006b) and topographic data, the drain demand at the time of drainage implementation for each grid cell has been estimated and represented by a certain probability. A threshold value has been used to define the grid cell as drained or not drained. This threshold number is calibrated and validated with drainage maps that show areas for which financial support for the drainage implementation has been applied for (ALR, 1962/1971). For the calibration and validation, the model results overlap the drained areas on the map by 70 and 83 %, respectively. As funding has not been requested for all drainage measures, drainage density is very likely to be higher than as shown on the maps. The calibration and validation results support this reasoning, as the model overestimated the areas by 11 % and 28 %, respectively. Thirty-eight percent of the agriculturally used area, itself 31 % of the total Kielstau catchment area, has been defined as artificially drained. The spatial distribution of the modelled drainages is displayed in Figure 3.2d.

This final drainage map has been used for the implementation of the drains into the SWAT model by overlaying it with the soil map and renaming the soil types. Each soil type marked as not drained inherits the same soil properties as the corresponding soil type marked as drained. Using this map for the HRU definition, the different naming makes it possible to distinguish between drained and undrained HRUs. For the application of equation {3.1} on each drained HRU, four parameters have to be defined and are used as calibration parameters. The values of the parameters are lumped over the whole catchment area as neither detailed spatial information of the physical properties nor flow data of the drains are available. The influence of the parameters is tested within plausible ranges and adjusted to maximize visual fit of modelled to measured daily catchment discharge. Table 3.2 shows the parameters, a short explanation, the calibration ranges and the manually calibrated end value. The drain values obtained during manual calibration were plausible. A DDRAIN of 800 mm is realistic as drainages in the catchment are used to extract the shallow groundwater which is, according to Eggelsmann (1981), in a depth of less than 1.3 m and occurs mainly during late autumn to late spring. According to the results obtained by Schmalz et al. (2008c), the shallow groundwater was met in depths of 0.5–1.5 m in early spring and the depths of several drainage ditches have been found to be between 0.65 m and 0.8 m. A value of 850 mm for DEP_IMP induced ponding of groundwater and seems slightly too shallow, but is within a reasonable range to the data. Borehole data (LANU, 2006b) showed an average jump of up to one order of magnitude towards lower permeability in depths of 1 m in the catchment and Schmalz et al. (2008c) encountered an impermeable layer in depths between 1 m and 2 m. To gain specific information about soil drainage time, it must be measured in the laboratory for different soil types. As those measurements could not be carried out, comparison was only possible with drainage

Table 3.2 Drain parameters with explanation and corresponding values

Parameter	Parameter explanation	Unit	Minimum	Maximum	End value
DDRAIN	depth from the soil surface to the drainage pipe or ditch	mm	400	1000	800
DEP_IMP	depth of the impervious soil layer underlying each drain	mm	450	2000	850
TDRAIN	time it takes to drain the soil to field capacity	h	6	48	24
GDRAIN	time lag until the drain water reaches the stream channel	h	12	48	48

rates roughly estimated based on soil texture by Kays and Patterson (1982), resulting in a drainage time of 24–36 h for the dominating soils in the catchment. Comparing this time span with the calibrated 24 h for the soil drainage time parameter TDRAIN indicates that the calibrated value is on the lower end. However, considering the high uncertainty of the estimated time span, the calibrated value is assessed as plausible. Initially, 48 h for GDRAIN for the 50-km^2-catchment seems long. However, monitoring two agricultural drainage stations and several drainage ditches, backwater has been observed over a few days during spring and autumn 2007. This leads to the assumption that high water levels in the stream channels induce ponding of ground and drainage water due to the low hydraulic gradients and that some drainages might even be blocked. The average transfer time of 48 h is thus plausible for the study catchment. The calibrated drain parameters are implemented for all drained HRUs into the auto calibrated *setup 1* to produce *setup 2*.

3.2.6 Incorporating landscape depressions—*setup 3*

The hydrological parameters and the location of depressions are derived from sinks in the DEM. Whether the sinks represent actual depressions in the landscape or DEM artefacts depends on the quality and resolution of the topographic data. Topography in the hydrological model was defined by a 25-m DEM (LVA, 1992–2004) because it is sufficient to depict the slope characteristics of the HRUs and because catchment delineation and model runs are evaluated quickly. This DEM is not sufficient to depict small-scale depressions and thus, two higher quality DEMs were used to identify depressions. The first dataset is a 5-m DEM (LVA, 1992–2004) which was derived from topographic maps, aerial photographs and additional land surveying. The vertical accuracy is 0.5 m, the horizontal accuracy 1 m. Another DEM has been derived from LiDAR data, recorded by aircraft in 2007. Three to four elevation points per square meter were measured and the cell values for a 5-m DEM were derived from the Delaunay-triangulated plane (LVA, 2008). The horizontal and vertical accuracy is 0.3 m and 0.15 m, respectively. The DEM sinks are evaluated by the fill-sink-routine (Jenson and Domingue, 1988; Tianqi et al., 2003) that is implemented in ArcGIS9 (ESRI, 1997). The algorithm increases the elevation of all grid cells within a sink to the height of the pour point of the sink. Subtracting the filled map from the original DEM reveals the extent and depth of the filled sinks. Within a nature protection area of 1.1 km^2, GPS measurements have been carried out to map depressions in the landscape (SN-SH, 2006). In 2006, artificial depressions have been constructed as temporary habitat for amphibians. Table 3.3 compares the areas of the mapped depressions with the area of the sinks derived from the DEMs. It can be seen that 98.96 % of the mapped depressions are covered by the sinks of the 5-m DEM from 2004 but that the depressions are overestimated by almost 140 % in area, probably caused by spurious sinks in the DEM. The DEM derived from the LiDAR data shows a better fit, as 99.83 % of the mapped depressions are found and the overestimation of about 40 % is considerably lower. The overestimation is considered plausible and has been anticipated, as not all depressions and their correct extend can be easily detected visually within field campaigns. The mapped area is fallow land with grass and bush vegetation in a landscape with rolling hills. This makes it difficult to distinguish depressions and some might not be found that actually exist.

Table 3.3 Comparison of DEM datasets with GPS measurements

Database	Year	Depression area before 2006 (ha)	Depression area after 2006 (ha)	Depressions represented on DEM (%)	Overestimation of depressions (%)
GPS measurements	2006	3.19	5.44	-	-
5 m DEM	2004	7.62	-	98.96	138.94
5 m DEM LiDAR	2007	-	7.60	99.83	39.63

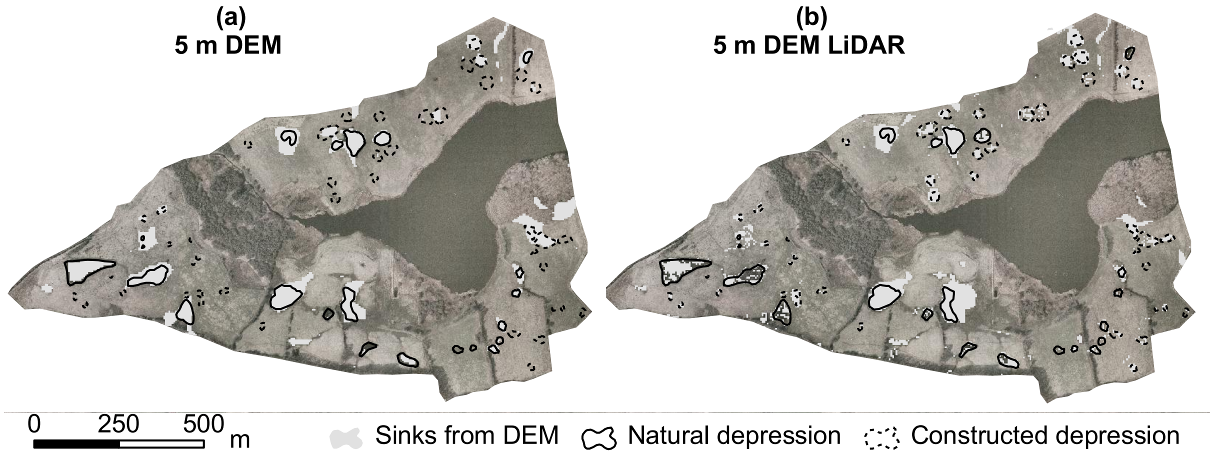

Figure 3.4 Aerial photograph (LVA, 1992–2004) with depressions: natural depressions in solid line, artificial depressions constructed in 2006 in dashed line, GPS-mapped (SN-SH, 2006); (a) the sinks derived from the 5-m DEM (LVA, 1992–2004); (b) the sinks derived from the 5-m LiDAR DEM (LVA, 2008)

Figure 3.4 shows the super-imposed sinks from the DEMs and the GPS measurements of the depressions on an aerial photograph. The spatial and temporal variability of the depressions is well depicted in the dataset. In Figure 3.4a, it can be seen that almost all newly constructed artificial depressions are not depicted on the old 5-m DEM (LVA, 1992–2004), while Figure 3.4b shows that they can be found on the 5-m LiDAR DEM (LVA, 2008). Due to the displayed plausibility test, the quality of the LiDAR-based DEM is considered suitable to depict landscape depressions based on DEM sinks and to incorporate them into the hydrological model.

The identification of the depressions and their parameters on the catchment scale has been carried out in the ArcGIS scripting environment and follows the simplified flow chart depicted in Figure 3.5. Input maps are ovals, calculation processes rectangles, intermediate maps in round-rectangles and the output in hexagons. The output maps are used to check if the results are plausible.

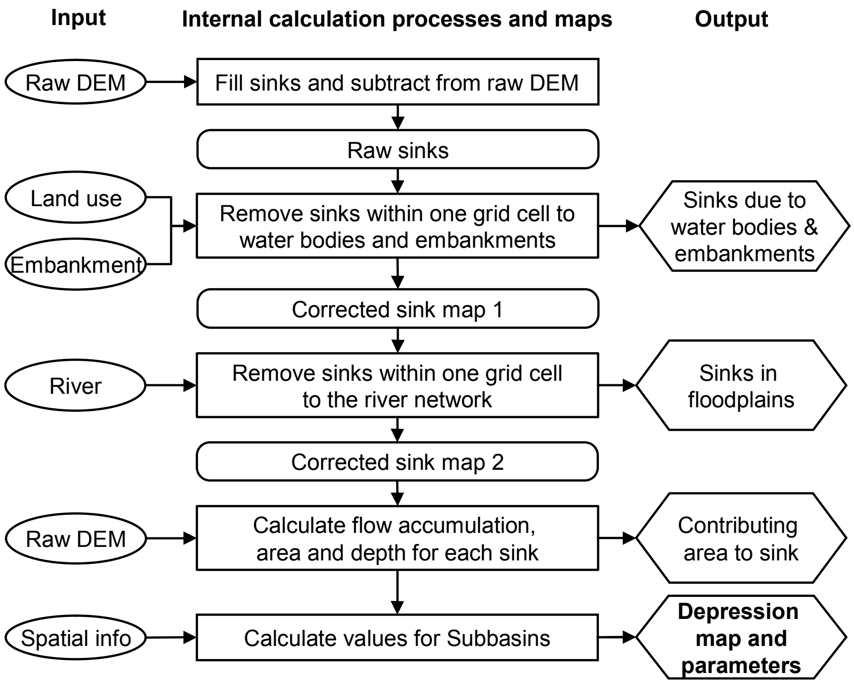

Figure 3.5 Flow chart to derive spatial information and parameters of depressions

The depression information is derived from a high quality DEM and three additional GIS maps: a map containing embankments from roads and railways, a land use and river network map. The first step is the calculation of raw sinks within the whole catchment from the supplied DEM. It is important to note that large flat areas blocked by embankments from roads or railway lines as well as water bodies such as lakes and reservoirs are also derived as sinks (Figure 3.6a) and have to be removed (Figure 3.6b). Road embankments are artificially drained and thus have no storage potential. Water areas have other hydrological characte-

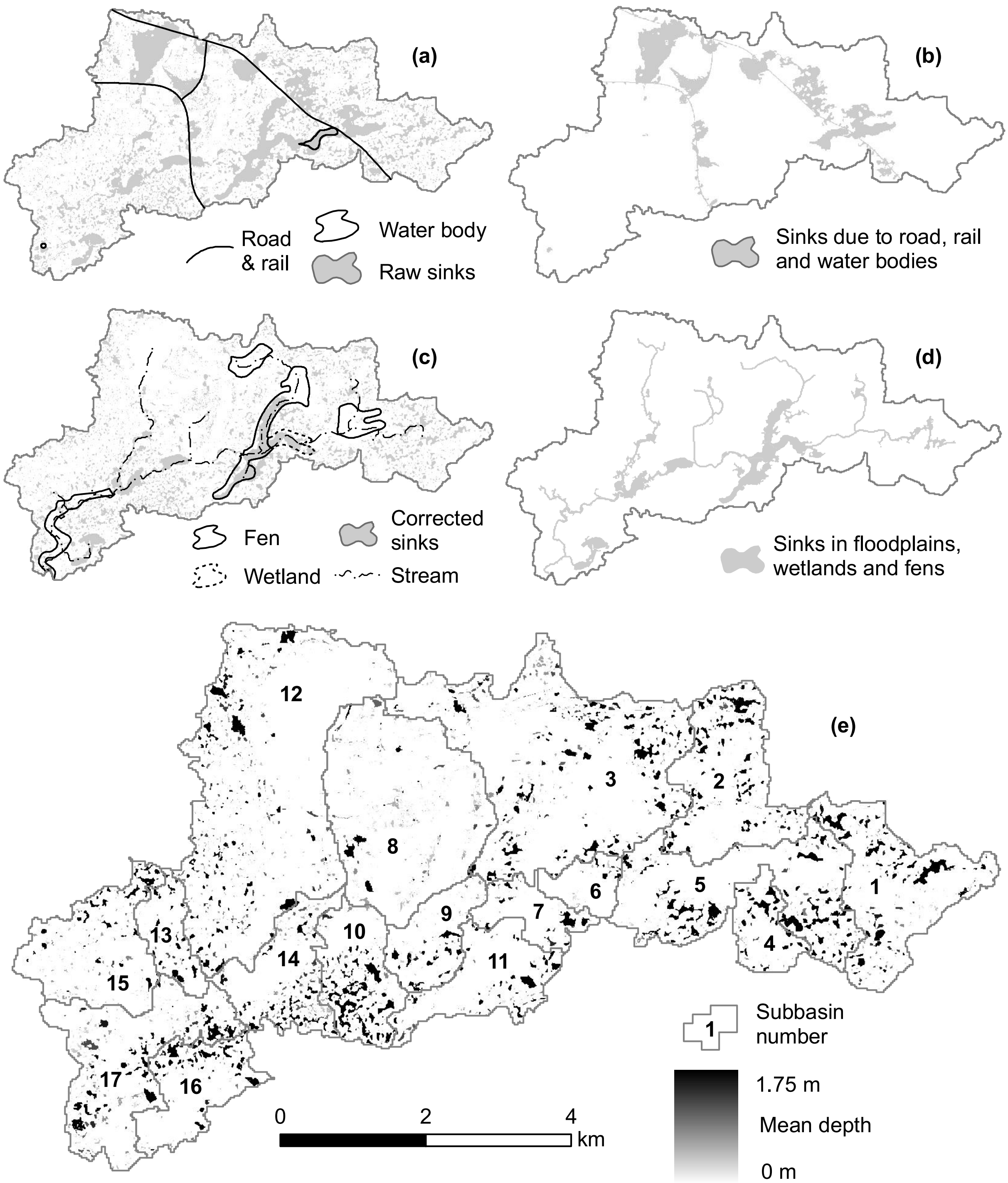

Figure 3.6 Deriving the retention potential in each subbasin: (a) the raw sinks with the streets, railway lines and water bodies; (b) the sinks caused by streets, railway lines and water areas which are extracted from Figure 3.6a; (c) the first correction without sinks from Figure 3.6b, with the river channel, wetlands and fens; (d) the sinks located in floodplains, wetlands and fens which are extracted from Figure 3.6c; (e) the final sinks with the corresponding average depth representing the depressions within each modelled subbasin

ristics as depressions, and permanent water bodies are usually implemented into the model separately. The corrected sink map is shown in Figure 3.6c. Floodplains which have a hydraulic connection to the channel system and areas close to the channel that are defined on the soil map as wetlands and fens may also be derived as depressions (Figure 3.6c). These areas are removed by supplying a river map and erasing all interconnected depressions that are within the vicinity of one grid cell to the river (Figure 3.6d) from which a second corrected sink map is obtained. The remaining sinks are then the depressions in the landscape (Figure 3.6e). To implement the depressions into the model, it is necessary to calculate the area and average depth of each sink and the area which contributes surface runoff to the sink. The area and depth is calculated from the final depression map (Figure 3.6e). The catchment area for each sink is derived by applying the D8 flow direction algorithm on the raw DEM and then calculating the number of cells draining into each sink by the flow accumulation function (Jenson and Domingue, 1988). Taking the maximum cell value of the flow accumulation grid within each depression and multiplying it with the area of one grid cell yields the contributing area to the sink. Summing these areas up for all sinks, 35 % of the catchment area drains into depressions, which have a surface area of 582 ha and a mean depth of 0.29 m resulting in a potential storage volume of 1,687,000 m³. The last step in the flow chart (Figure 3.5) represents the transformation of the derived values to fit the model input requirements. In the case of SWAT, the values are summarized in the map's attribute table for each subbasin according to spatial information of the SWAT project. For each subbasin the area contributing to the depression, the area of the depression and the depth are assigned to the HRUs in the subbasin. As only one HRU per subbasin can be categorized as a depression, the largest HRU is defined as the depressional HRU and the remaining HRUs as contributing to this depressional HRU with the calculated area fraction. The described parameters are then implemented into the corresponding input files of *setup 1* to obtain *setup 3*.

3.2.7 Incorporating drainages and landscape depressions—*setup 4*

The calculated drain parameters (see *setup 2*)and the calculated depression parameters (see *setup 3*) are jointly incorporated in *setup 1* to obtain *setup 4*.

3.3 Results

Figure 3.7 shows the influence on the most affected flow components by comparing *setup 1* with *setup 2* (Figure 3.7c) and *setup 1* with *setup 3* (Figure 3.7d) on the subbasin scale. The values are averaged over the whole modelling period. Figure 3.7a illustrates the percentage of drainage area within each subbasin. Figure 3.7b shows the percentage of the subbasin area that is affected by depressions, i.e. the percentage of area contributing flow to the depressions plus depression surface area. The influence of the drainages on the groundwater flow can be seen in Figure 3.7c. The decrease in average catchment groundwater flow is 36 % (min. 0 %, max. 73 %). Tile drain fractions below 20 % influence groundwater flow

Table 3.4 Model performance (RMSE, r2 and NSE) for the different setups

	RMSE		r^2		NSE	
	Calibration	Verification	Calibration	Verification	Calibration	Verification
Setup 1 (initial)	0.38	--	0.16	--	-0.31	--
Setup 1 (manual calibrated)	0.17	--	0.54	--	0.42	--
Setup 1 (auto calibrated)	0.08	0.07	0.72	0.71	0.72	0.65
Setup 2 (drains)	0.07	0.07	0.78	0.78	0.76	0.66
Setup 3 (depressions)	0.09	0.06	0.74	0.73	0.69	0.70
Setup 4 (drains & depr.)	0.06	0.04	0.82	0.82	0.78	0.78

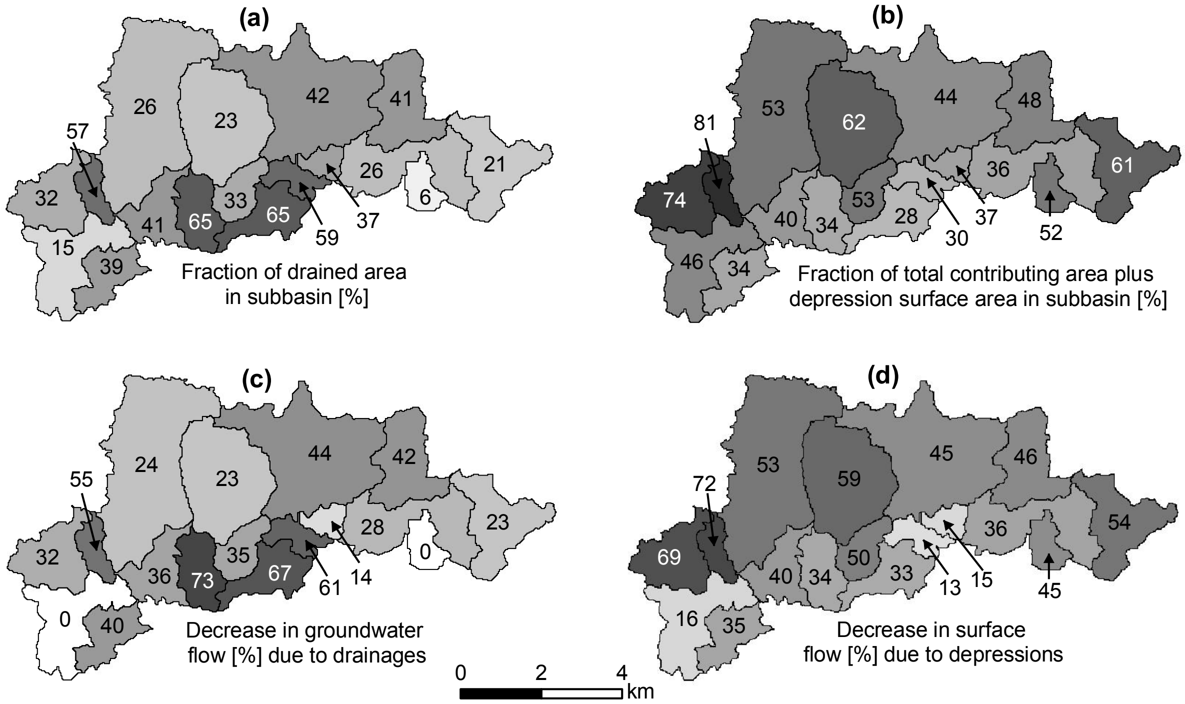

Figure 3.7 Impact of retention on hydrology per subbasin: Percentage of drainage area in each subbasin (a); percentage of contributing area plus depression area in each subbasin (b); percentage decrease in mean groundwater flow from setup 1 to setup 2 in each subbasin (c); and percentage decrease in mean surface flow from setup 1 to setup 3 in each subbasin (d)

only marginally. The influence of the introduction of the depressions on the surface runoff is shown in Figure 3.7d. The surface flow retention in the depressions leads to an average decrease in catchment surface runoff by 46 % (min. 13 %, max. 72 %). The area of the drainages and depressions is proportional to the decrease in groundwater and surface flow, with a coefficient of determination r^2 of 0.90 and 0.72, respectively.

The model results for the different setups are presented in Figures 3.8 to 3.10 by comparing modelled and measured catchment discharges. Standard measures of model performance including RMSE, r^2 and NSE are given in Table 3.4. r^2 is always higher compared to the initial setup if drainages or depressions, or both are considered. The same is true for the other parameters, except the incorporation of depressions alone which increases the RMSE and decreases NSE for the calibration period. The joint input of drainages and depressions leads to the best statistical performance in all cases.

Figure 3.8a shows that *setup 1* (initial) does not depict the discharge dynamics in a sufficient manner. It shows only two flow components: a too high baseflow with incongruous surface runoff and mostly overestimating peak flows, especially in summer 2002, resulting in a NSE of -0.31. The first manual adjustment of the soil and groundwater parameters as well as the calculation of the baseflow recession constant and the CN values to *setup 1* (manual calibrated) achieve a more dynamic discharge and an occurrence of interflow with a NSE of 0.42 (Figure 3.8a). Auto calibrating *setup 1* obtains a maximum NSE of 0.72 for the calibration (0.65 for the validation) period when reaching the optimum parameter set after 4,473 runs. The discharge depiction improved considerably to the manually calibrated setup. The depiction of the low flow periods either fits reasonably well or is underestimated, especially during long, dry summer periods. Larger discrepancies are depicted in the peak flows. Underestimation mainly occurs in the winter periods (1999, 2000, 2002, 2003) and overestimation mainly in the summer periods (2002, 2003, 2004, 2005, 2007). All following assessments are based on the comparison with *setup 1* (auto calibrated). For an easier realisation of the hydrological differences between each setup, the period October 2006 to May 2007 is displayed in Fig-

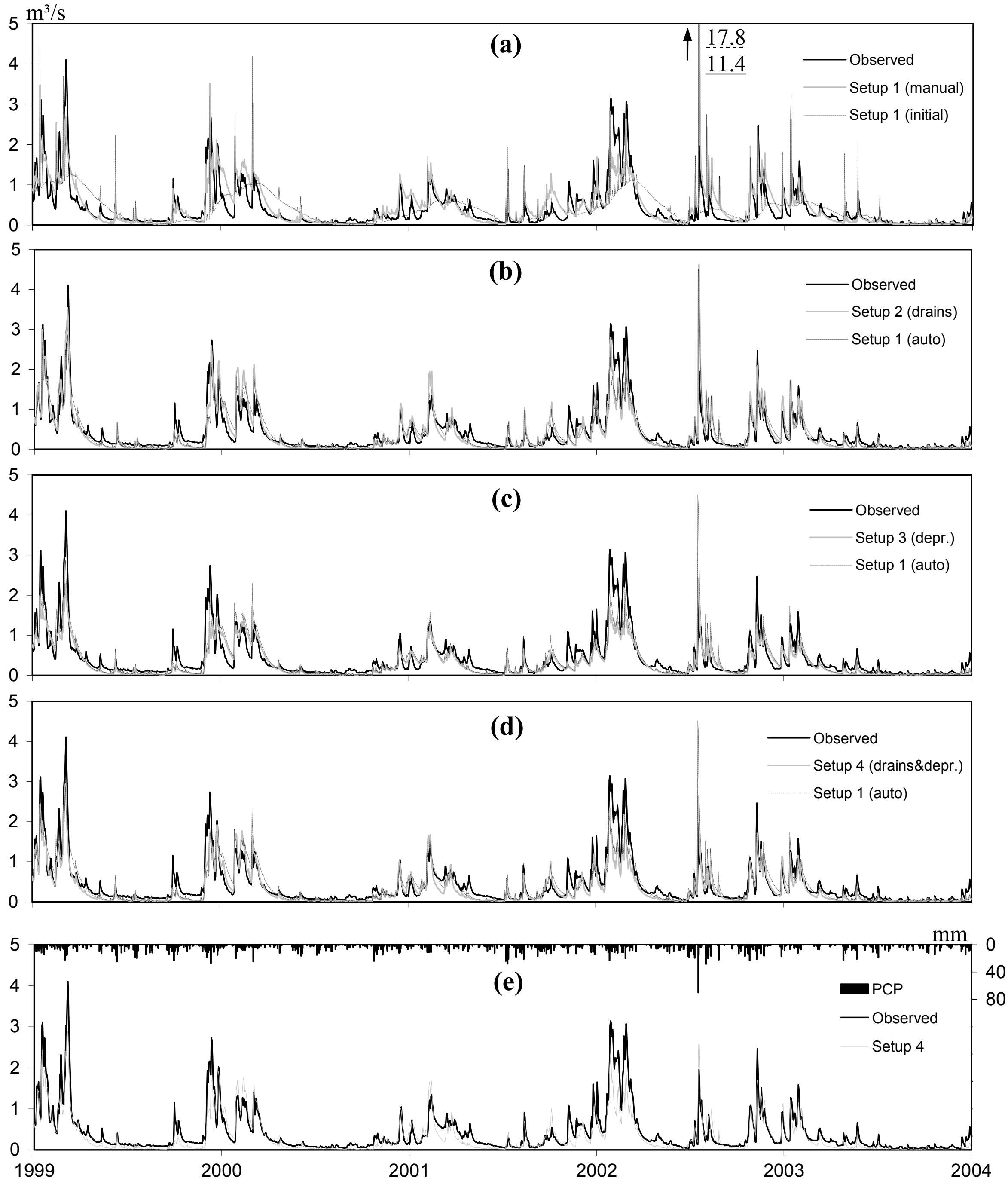

Figure 3.8 Simulation results calibration period 1999-2004: (a) comparison of setup (initial) and setup 1 (manual calibrated); (b) comparison of setup 1 (auto calibrated) and setup 2 (drains); (c) comparison of setup 1 (auto calibrated) and setup 3 (depressions); (d) comparison of setup 1 (auto calibrated) and setup 4 (drains & depressions); (e) comparison final setup 4 (drains & depressions) with observed flow

ure 3.10. Figures 3.8b, 3.9a and 3.10a show that implementing the drain parameters mostly results in higher peak flows and steeper hydrograph recession. An increase of peaks occurs in all winter periods while the summer peak flows remain comparably constant. For the single two largest summer rain events (summer 2002 and summer 2007) the peaks are relatively constant. A larger increase is present at the beginning of the wet periods than at the end (2000, 2001, 2002, 2007, 2008). The incorporation of the calculated depressional storage volume to *setup 3* (Figures 3.8c, 3.9b and 3.10b), results in decreased peak flows. Both summer and winter periods are equally affected, but especially the larger peak flows exhibit greater change. The implementation of both drainages and depressions (*setup 4*) shows both charac-

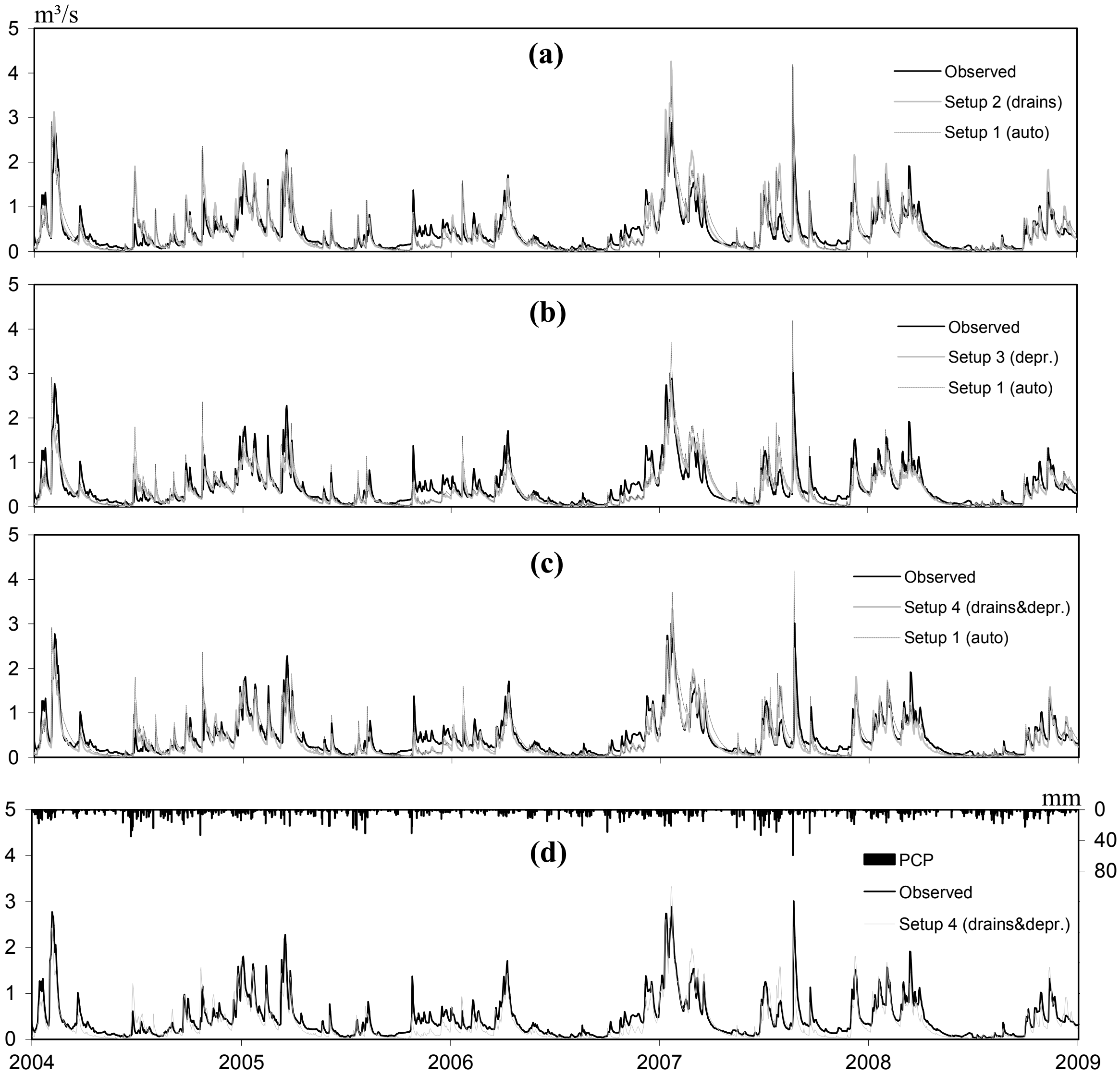

Figure 3.9 Simulation results Validation period 2004-2009: (a) comparison of setup 1 (auto calibrated) and setup 2 (drains); (b) comparison of setup 1 (auto calibrated) and setup 3 (depressions); (c) comparison of setup 1 (auto calibrated) and setup 4 (drains & depressions); (d) comparison final setup 4 (drains & depressions) with observed flow

teristics of setups 2 and 3: A faster recession of the falling hydrograph limb, constant or increased winter peak flows, and a reduction of summer peak flows (Figures 3.8d, 3.9c and 3.10c). For a better overview of the final model, *setup 4* is solely compared with the observed flow in Figures 3.8e and 3.9d.

3.4 Discussion and conclusions

It has been the objective of this study to assess the common assumption of neglecting drainages and depressions in lowland catchment modelling by testing three hypotheses: (1) that the implementation of artificial drainages alone improves model performance; (2) that the implementation of depressions alone improves model performance and (3) that the combined implementation of artificial drainages and depressions shows the best model performance. This assessment was carried out by comparing an auto calibrated default setup (*setup 1 auto*) with three refined setups, for which drainage (*setup 2*), depression (*setup 3*) and drainage and depression parameters (*setup 4*) were implemented into *setup 1 auto* without further calibra-

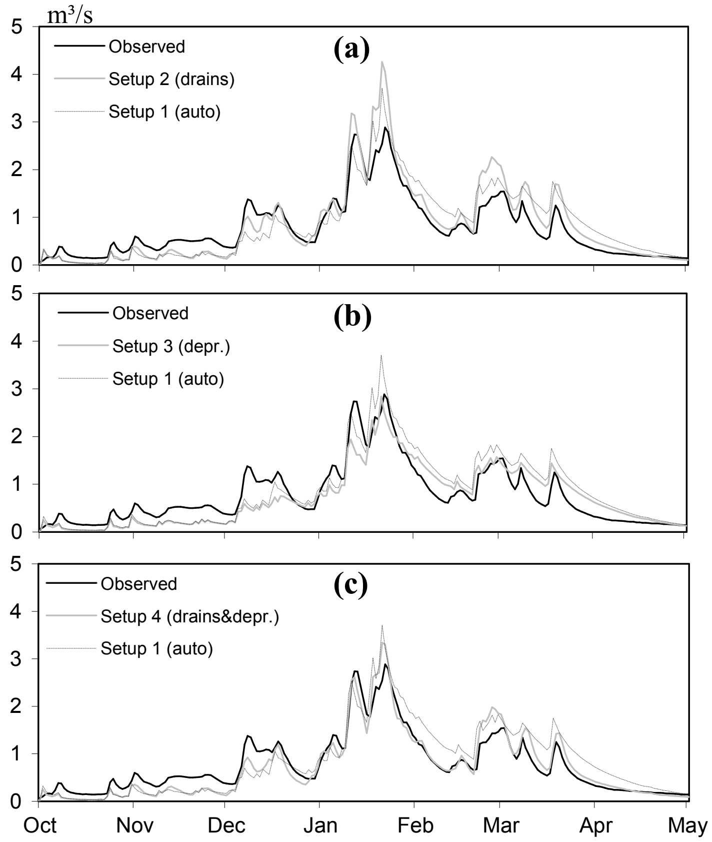

Figure 3.10 Detailed simulation results of the four setups for the period October 2006 to May 2007

tion. The results of the proposed methodology are analysed and it is discussed whether the hypotheses are supported.

The shown reduction in groundwater and surface flow components on the subbasin level and their positive spatial correlation with drainage area as well as area affected by depressions implies that the general model response should be considered plausible. The tile drain implementation induces tile flow which is extracted from groundwater flow. The depressions induce retention of surface flow as a fraction of surface runoff is redirected to the implemented storage volume.

The incorporation of the artificial drainage improved model performance. The primary reason for this improvement is the increase of peak flows in winter periods. This is a plausible model behaviour as shallow groundwater in the region mainly occurs during this season (Eggelsmann, 1981), leaving a high potential for groundwater extraction through the drains. Second, no higher modelled peak discharge occurs for the intense summer rain events. This model response can also be explained as drainages usually have no influence on summer rain events. If rainfall intensity exceeds soil infiltration capacity during those events, surface runoff is relatively high and the amount of infiltrated water is not sufficient to induce ponding of groundwater to the drainage pipe depth, so that no, or only little drain flow occurs. Tile drain investigations support this finding (Kiesel et al., 2009b), as tile drain discharge falls in April and May and no tile drain flow is expected to occur in the summer period as the top soil layers dry due to climate conditions and extraction of soil water by vegetation. This process seems to be realistically depicted by the model. Third, the higher increase of peaks at the beginning of the winter periods compared to the end of the winter periods. An explanation for this particular model response is that the soil moisture at the beginning of wetter periods gradually increases. If the field capacity is reached, groundwater is ponding above the impermeable layer. As soon as the water level reaches the drain pipe, drain discharge occurs. Further infiltration water is then extracted faster than the groundwater flow usually contributes to the runoff, increasing the peak flows. If the modelled soil layers, however, are all completely saturated after long wet periods, further rain events can only cause surface runoff, leaving no additional possibility to flow contribution by the drains. Deriving the drain locations and calibrating the drain parameters on the catchment scale has improved the model performance and process depiction. It can thus be concluded that the first hypothesis is supported by the results.

Model performance during the calibration period after the incorporation of the depressions improved r^2 and reduced RMSE and NSE compared to *setup 1 auto*. For the validation period, model performance improved for all parameters. *Setup 3* shows a general reduction of the peak flows, with a higher impact on summer peak flows. This is plausible, as the modelled depressions induce retention and infiltration of water so that a fraction of the fast surface runoff is transformed to interflow and groundwater flow. Improved model performance is mainly due to the reduction of these summer peak flows as they have been overestimated in the auto calibrated base setup. The reduction of winter peak flows on the contrary leads to a less good fit as those tended to be underestimated in the auto calibrated base setup of the calibration period. Looking at the summer months of the calibration and validation flows separately reveals that within the calibration period fewer peak events occur than within the validation period (5 days and 14 days, respectively above 1 $m^3 s^{-1}$). In addition, as the winter periods of the calibration period are rather underestimated in the auto calibrated base setup, the validation period has improved. This fact has to be considered when assessing the second hypothesis: process depiction has improved for the short and intense summer rain events. For the winter period, however, the introduction of the additional retention volume seems to induce too slow catchment responses to rain events. A universal assessment on model performance is not drawn, as there is a dependence on the pattern of the hydrological events. On the example of the total ten-year modelling period, model performance has slightly improved.

The joint incorporation of drainages and depressions leads to the best model performance. Generally, the implemented hydrological processes could somehow counteract each other: drainages reduce retention while depressional storage increases retention. The implementation of the drainages induces higher peak flows especially after dry periods, while already very strong events are less affected. The depressions induce a decrease of peak flows, especially for short and intense events. This implies that the hydrological impact is possible to be levelled out in certain peak discharge events, but not for high-intensity rains. The model performs better especially for short rain events in summer with a high rainfall intensity. One example for this is a 70-mm rain event at the end of July 2002. The models without depressions overestimate this event by 137 %, *setup 4* only by 35 %. In order to depict the winter flows realistically, it is important that the two processes are incorporated jointly. Furthermore, the drainages enable a more realistic depiction of the hydrograph's recession limbs by inducing a faster recession after wet periods. Hypothesis three is thus supported by the results of this study.

As two of three hypotheses are supported and one is partially supported, the implementation of the drainages and the depressions results in a more plausible and feasible model. Overall, the final model performs well in depicting the modelled daily peak flows with the measured discharge over the ten-year simulation period, considering that only one rain gauge outside the catchment supplies daily aggregated precipitation sums. The process enhancement is of particular importance when simulating the impact of removing drainages (Krause et al., 2007b) by land use changes from arable land to fallow land or the rewetting of wetlands. It is also expected to influence erosion and nutrients for water quality modelling as depressions in the landscape impact surface runoff processes which affect these entry pathways (Deasy et al., 2009).

To further improve model performance, it is advantageous to obtain better rainfall data and use a sub-daily modelling time step due to the relatively small catchment with a sub-daily time of concentration for the first surface runoff. Possible improvements are also seen in the calculation of evapotranspiration in the depression HRUs. Evapotranspiration in depression HRUs was found to be lower than for similar non-depression areas. As different results were expected, it is recommended to revise the model algorithms in limiting evapotranspiration in depressions. If further detail is desired, e.g. to model sediment transport, settling and water quality in the depressions, a more detailed representation of spatial characteristics and an up-hill to downhill routing between the landforms within the model or, alternatively, a coupling with a spatially distributed model is recommended. The methodology should be tested in additional lowland catchments and with further and longer hydrological and climate time series. In its current version, SWAT has the advantage of depicting comprehensive hydrological processes in a user-defined, flexible resolution. The results of this study show that the spatial representation within SWAT is sufficient to depict the influence of drainages and depressions on streamflow on the meso scale. Based on this study, it is concluded that the incorporation of drainages and depressions in lowlands should not be neglected.

3.5 Acknowledgement

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Chapter 4 SEPAL – A simple GIS-based tool to estimate sediment pathways in lowland catchments

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Abstract Even though soil loss in the lowlands imposes not as much a restriction on land use and agricultural productivity as in erosion affected mountainous areas, the input of fine sediment into the rivers and streams is a concern due to water quality issues and substrate siltation. Drains, river banks and agricultural fields are the three main sources of fine sediment in lowland regions. For a successful implementation of measures to decrease sediment input a well-founded knowledge of the individual entry pathways is essential. To assess the importance of possible entry pathways, a GIS-based methodology (SEPAL) has been established combining the ABAG, a river bank erosion formula and a regression approach to include the contributions of drains. SEPAL has been applied on a study catchment in northern Germany. The results show that 15 % of the sediment input into the river comes from agricultural drains, 71 % from river banks and 14 % from adjacent fields. A comparison of the results with field mapping and -sampling shows that the approach is plausible. The calculated total annual sediment input is 616 t yr^{-1}, while the measured suspended sediment load is 636 t yr^{-1}. It can be concluded that the methodology is suitable for estimating sediment entry pathways and annual sediment loads in lowland catchments as a base for modelling projects and further investigations. However, further work is necessary for gaining sound knowledge about uncertainties and especially about the processes forcing sediment input from drains.

4.1 Introduction

Although the emphasis of erosion studies is focused on regions with steep slopes, erosion processes in lowland catchments should not be neglected (Imeson and Ward, 1972). Fine sediment input into waterways has a decisive influence on water quality (Davies-Colley et al., 1992; Ryan, 1991), aquatic life-forms (Berry et al., 2003) and their habitat (Wood and Armitage, 1999), as well as anthropogenic usage like ship traffic and artificial water structures as harbours (Stevens and Ekermo, 2003). Especially in lowland regions, the siltation process of sediment can lead to necessary but harmful measures for the environment like dredging and mud extraction activities (Licursi and Gómez, 2009). Collins and Walling (2004) point out that, though it is difficult to acquire information about sediment entry pathways, it is important to understand sediment sources for deriving management plans and to prevent environmental problems. The main sediment entry pathways in lowlands are the input from agricultural drains, bank erosion and field erosion (Russel et al., 2001; Kronvang et al., 1997; Walling et al., 2002). Variable criteria influence sediment input from these three sources: The sediment contribution from drains is highly variable and rarely researched. It seems to be governed by factors like soil type, groundwater levels, soil moisture, drainage depth and age, land use, irrigation (Walling et al., 2002; Stone and Krishnappan, 2002) as well as the size of the drained area (Smith et al., 2005). While Kronvang et al. (1997) and Stone and Krishnappan (2002) found sediment losses from drains of 20 to 130 kg ha^{-1} yr^{-1}, Ulèn and Persson (1999) and Chapman et al. (2005) investigated drains under loamy and clayey soils with a high susceptibility to form macropores and measured extremely high values of up to 230 and up to 1,000 kg ha^{-1} yr^{-1}, respectively. River bank erosion depends on water flow characteristics like depth and velocity, freezing and thawing processes, the soil type, soil density and moisture, vessel traffic, cattle treading, as well as stream properties like curvature, cross-sectional shape and plant cover (Hooke, 1979; Saynor et al., 2003; Laubel et al., 1999; Bradbury et al., 1995; Wynn et al., 2004). It is generally considered as the main entry pathway for sediments in lowlands (Laubel et al., 1999; Kronvang et al., 1997; Hasholt, 1988). For the fields, besides the influence of soil type, precipitation, topography and land management also drainage, hedgerow density and small-scale field patterns due to anthropogenic influence (Hassenpflug, 1971) affect this entry pathway in lowlands.

The above-named processes of field and bank erosion are successfully implemented in a number of modelling concepts (Merritt et al., 2003; Bärlund et al., 2006; Tate, 2006). However, no modelling approach has so far been established for depicting sediment input from drains. Guidelines for estimating sediment budgets mostly cover only field and river bank erosion (Reid and Dunne, 1996). Though Walling (2005) stresses the importance of drain systems on the sediment source and delivery pathway, assessing the input quantities is rarely carried out because measurements on the catchment scale are tedious and cost intensive. Since neglecting one pathway would lead to a biased model calibration, it is necessary to incorporate all potential sediment pathways into modelling and planning processes for the successful development and assessment of management measures. It is the scope of this study to derive a pragmatic desktop approach for estimating the long-term share of the three sediment entry pathways in lowlands by combining remote sensing and structural river data with knowledge from comparable research studies. To validate the model estimations calculated results are compared with field measurements. The field approach consists of erosion mapping, erosion pin readings and suspended sediment sampling.

4.2 Investigation area

The study area in which the developed methodology has been tested is located in the northern German lowlands (Figure 4.1a). The land use of the 50-km² Kielstau catchment is dominated by arable land and pasture (Figure 4.1b). The mean annual precipitation and temperature are 893 mm and 8.3 °C respectively (DWD, 2010). The relatively flat topography (Figure 4.1c) with rolling hills and numerous depressions in the catchment is typical for the north-eastern Schleswig-Holstein landscape (Lorentzen, 1938) and leads to a low surface runoff fraction and low hydraulic gradients. Predominant soil types are Haplic Luvisols in the eastern and Stagnic Luvisols in the western part while Sapric Histosols are occurring along the stream and its tributaries. Extensive drainage measures have been implemented during the reallocation of land, mainly from the 1950s to the late 1970s (MELF, 1980). The drain location has been estimated by Fohrer et al. (2007) (Figure 4.1d). Due to the typical lowland processes occurring in the Kielstau catchment it is a suitable area for applying and testing the developed tool for assessing sediment entry pathways.

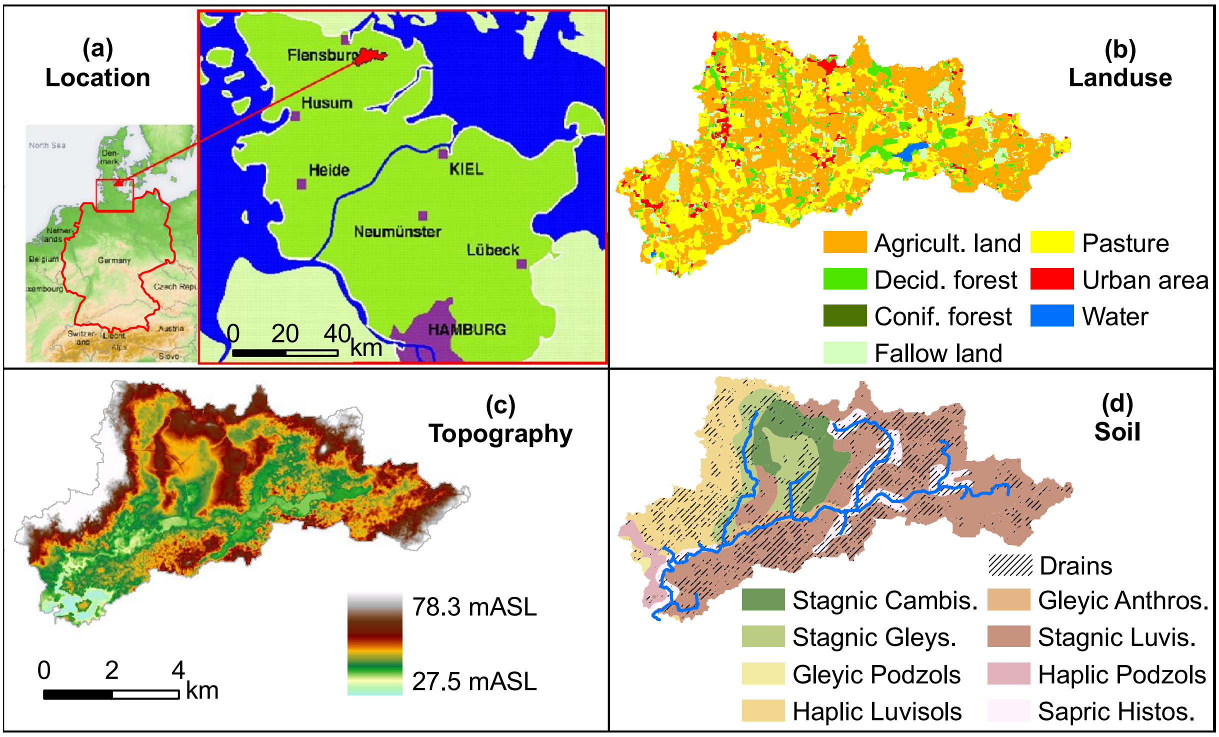

Figure 4.1 Location and properties of the Kielstau catchment: location in Germany (a) (Jose, 2006; LVA 1992-2004); land use (b) (DLR, 1995); topography depicted on the 5-m DEM (c) (LVA 1992-2004); the river network (DAV-WBV/LAND S.-H., 2006), the soil (BGR, 1999) with drained areas (d) (Fohrer et al., 2007)

4.3 Methods

4.3.1 Desktop tool SEPAL

The GIS-based SEPAL approach has been implemented in an ArcGIS 9.2 script. The script consists of one calculation process for each sediment entry pathway. The presented equations can not be used to quantify the sediment input for short time periods or single events. Only long-term yearly average estimations are possible. The simplified flowchart in Figure 4.2 shows the required GIS maps, the user input parameters, the calculation tasks, intermediate data and the output. The values for the necessary input parameters for applying SEPAL in the Kielstau are explained in Table 4.1.

Table 4.1 Input parameters for SEPAL, (D)rain-, (B)ank-, and (F)ield pathway

Parameter		Value	Unit	Reference
Percentage of drained area (D)	A_d	38	%	Fohrer et al., 2007
Soil erosion factor (B)	K	0.061-0.143	--	Williams et al., 1995
Adjacent land use value (B)	A_{gf}	1-16	--	Dickinson et al., 1989
Soil Bulk density (B)	BD	0.60-1.65	$t\ m^{-3}$	AD-HOC-AG, 2005
Bank erosion height (B)	h	1	m	Field inspection
Critical water depth (B)	h_c	0.6	m	Zacharias, 2007
Vicinity threshold (F)	V_s	100	m	Field inspection
Mean precipitation (F)	N_J	893	mm	DWD, 2010
Soil erosion factor (F)	K	0.061-0.143	--	Williams et al., 1995
Management factors (F)	C; P	0.01-0.1; 0.5	--	Schwertmann et al., 1987

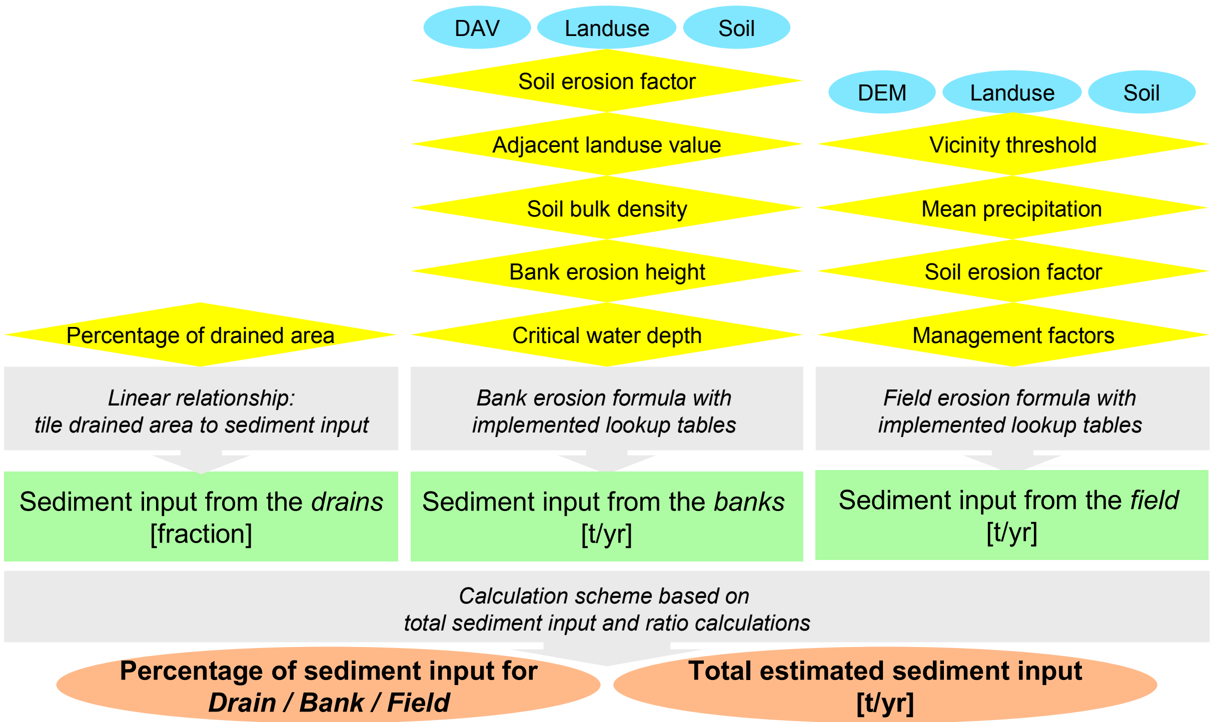

Figure 4.2 SEPAL input, processes and output: GIS maps (blue ovals), user input parameters (yellow diamonds), calculation processes (grey rectangles), intermediate values (green rectangles) and output (red ovals) necessary for calculating the sediment input from the three pathways (drains left, banks middle and fields right)

The results of the three processes are combined in a simple scheme that calculates the percentage of sediment input for each pathway.

The sediment input from the river banks is calculated with the bank retreat equation by Dickinson et al. (1989). Dickinson's formula originally yields the absolute bank retreat only and is thus modified to equation {4.1} by taking the actual river section length and erosive bank height into consideration to gain the total sediment input from the banks:

$$Y_b = \sum_i \left[(2 \cdot 10^{-10} (K_i^{2.5} \cdot A_{gf_i}^{7.2}) + 1.75^{\frac{h_{m_i}}{2 \cdot h_c}}) \cdot BD_i \cdot L_i \cdot h \right] \qquad \{4.1\}$$

where *Yb* [t yr^{-1}] is the sediment input from the two river banks for the whole river length. *i* [-] is the river section number adopted from the river structure mapping (DAV) for sections susceptible to erosion (DAV-WBV/LAND S.-H., 2006). In the DAV database, rivers are partitioned in sections with equal morphological properties. The average segment length is 130 m for the study stream and contains information on substrate, incision depth and cross section geometry. K [-] is the soil erosion factor for each soil type. A_{gf} [-] is a corresponding value for the adjacent land use which can be taken from a look up table in Dickinson et al. (1989). h_m [m] is the maximum possible water depth, taken from the DAV database. h_c [m] is the critical water depth at which bed load transport begins. BD [t m^{-3}] is the bulk density of the soil, L [m] is the river section length from the DAV and h [m] the bank height up to which erosion can occur.

Because processes and factors governing and affecting sediment input from drains are not yet researched, only an empirical regression approach could be implemented. Equation {4.2} is used to derive the sediment input share from drains:

$$F_d = \frac{I_{dm}}{100} \cdot A_d \qquad \{4.2\}$$

where F_d [%] is the estimated percentage of sediment input from the drains. I_{dm} [38.9 %] is the percentage of the sediment input from drains if the catchment would be 100 % drained. This value is calculated in Table 4.2 with [26.0 % + 59.0 % + 31.6 %] / 3 based on three studies carried out in catchments in Denmark and the United Kingdom with similar properties as the Kielstau catchment. A_d [%] is the percentage of the actual drained catchment area on the total catchment area, which is the necessary user input parameter for the investigated catchment.

Except for intensive rain events, the sediment input is expected to come mainly from river banks and drains. Imeson and Ward (1972) state, that the sediment input from fields in lowland catchments is minor because of the small fraction of surface runoff entering the streams directly. Hence, it can be assumed that sediment is entering the stream only from adjacent, sloping areas and according to DVWK (1996) mainly from agriculturally used fields during intensive rain events. Thus, the sediment input from the fields is estimated only for areas within a certain vicinity to the open stream channel. Equation {4.3} is based on the German revision of the Universal Soil Loss Equation (ABAG, Schwertmann et al., 1987; USLE, Wischmeier and Smith, 1978) and is implemented in the GIS script:

$$Y_f = \sum_c \left[R \cdot K_c \cdot L_c \cdot S_c \cdot C_c \cdot P \cdot A_c \right] \text{ while } c \text{ lies within } V_s \text{ to the open stream channel} \qquad \{4.3\}$$

where Y_f [t yr^{-1}] is the sediment loss from the adjacent fields. c is the grid cell. R is the rainfall erosion factor based on the equation for Schleswig-Holstein (Sauerborn, 1994): $R = -21.08 + 0.0905 \cdot N_J$ where N_J [mm] is the mean annual precipitation. K is the soil erodibility factor for each soil type. L is the slope length calculated from the DEM according to Wischmeier and Smith (1978). S is the slope steepness factor calculated from the DEM for slopes < 9 % according to Feldwisch (1995) and for slopes > 9 % according to Renard et al. (1997). C is the cover-management factor. P is the support practice factor. A [ha] is the area

Table 4.2 Percentage of sediment input from drains based on percentage of drained catchment area, bold numbers are the linear extrapolation if the catchment would be 100 % drained

Study and country	Time period	% drained area	% sediment input	Soil	Land use	Precipitation [mm yr^{-1}]
Krovang et al., 1997; Denmark	1993-1996	50 **100**	13.0 **26.0**	Moranic deposits	90 % farmed	720
Walling et al., 2002; UK	1997-1999	90 **100**	53.0 **59.0**	Silty clay loams	Mixed agriculture	660
Walling et al., 2002; UK	1998-1999	90 **100**	28.5 **31.6**	Clayey & also permeable	Mixed agriculture	660

of the DEM grid cell. V_s [m] is a proximity parameter for the GIS calculations which is based on the relief of the catchment. Therefore, field inspections might be useful to estimate the average overland distance that sediment can possibly be transported to the stream channel. Apart from the total accumulated sediment loss, the GIS implementation of the ABAG also allows the spatial regionalization of erosive grid cells in a map.

The results from equation {4.1}, {4.2} and {4.3} are used in equation {4.4} and {4.5} to calculate the fractions of sediment input for all three pathways:

$$F_b = (100 - F_d) \cdot \frac{Y_b}{Y_b + Y_f} \qquad \{4.4\}$$

$$F_f = (100 - F_d) \cdot \frac{Y_f}{Y_b + Y_f} \qquad \{4.5\}$$

where F_b, F_f and F_d [%] are the estimated percentage of sediment input from the banks, fields and drains. Y_b and Y_f [t yr^{-1}] are the sediment inputs from the banks and fields. Now, the average total estimated sediment input (Y_{tot} [t yr^{-1}]) can be calculated in equation {4.6}:

$$Y_{tot} = \frac{Y_b + Y_f}{F_b + F_f} \cdot 100 \qquad \{4.6\}$$

4.3.2 Field measurements

For testing the results of the SEPAL approach, field measurements were carried out: The bank retreat is quantified via erosion pins for three river sections that are marked in the structural river mapping database (DAV-WBV/Land S.-H., 2006) as susceptible to erosion. Hooke (1979), Laubel et al. (1999) and Saynor et al. (2003) describe the general usage of erosion pins from which the methodology for this study has been derived. The used erosion pins are stainless steel rods with a diameter of 5 mm and a length of 0.5 m. A representative river section of 230-m length has been chosen where 21 erosion pins have been placed in the banks at three different sites at low flow conditions on 07th July 2008. The pins are pushed in the banks and the distance from the tip of the pin to the river bank is immediately recorded for each pin. The time interval of erosion pin measurements to quantify the bank retreat generally ranges from certain storm events to years (Hooke et al., 1979). The distance measurements from the pin tip to the banks are carried out with a calliper and are conducted from each side of the pin (left, right, above and below) and the mean value is calculated. The distance from the pin tip to the bank was measured on 01st December 2008 in order to gain information about the bank retreat since 07th July 2008.

The field erosion is assessed by mapping campaigns according to DVWK (1996). First of all, a classification of river sections is obtained with topographic maps (1 : 25,000 scale) and aerial photographs (1 : 5,000 scale) in order to plan the mapping along the main channel of the Kielstau. The mapping took place on 22th April 2008 and 29th April 2008 where erosive patterns have been paced off and captured in sketches on a scale of 1 : 5,000 and on digital photos. The gained information has been digitised in the GIS in order to compare mapped erosive fields with the calculated ABAG map.

As a continuous assessment, aggregated daily suspended sediment samples at the catchment outlet are taken since July 2007 with an automatic sampler and are used to calculate the sediment load of the Kielstau according to DVWK (1999): 1-L samples are taken and filtrated with a 65-µm filter. The filters are dried and weighed with precision scales to obtain the mean daily sediment concentration (mg L^{-1}). To gain the sediment load, the measured sediment concentration is multiplied with the daily mean discharge value (m^3 s^{-1}) for the corresponding day.

4.4 Results

The SEPAL-calculations led to a sediment input share of 15 % from the drains, 71 % from the river banks and 14 % from the fields. The total sediment input, expressed as an annual mean value is estimated to be 616 t yr^{-1}. The calculated bank retreat has a mean value of 2.9 cm yr^{-1} for all erosive sections. This adds up to an average total sediment input by bank erosion of 437 t yr^{-1}. The calculated soil loss from the fields ranges from 0 to 3.9 t ha^{-1} yr^{-1}. Taking the fields into consideration that are within V_s = 100 m vicinity to the streams, this leads to a mean total sediment input by field erosion of 88 t yr^{-1}. The yellow marked areas in Figure 4.3 show the calculated erosive fields with an estimated sediment loss of ≥ 0.5 t yr^{-1} at the main channel. In total 112 locations have been depicted with a total area of 5.5 ha. The sediment input from all agricultural drains in the catchment is then calculated to 91 t yr^{-1}.

The erosion pin readings for assessing the bank retreat are summarized in Table 4.3. Displayed are the measured mean values from the pin tip to the bank. The total bank retreat over the five month period is the difference of the mean values. The reading of pin 21 is not plausible, possible reasons can be either measuring errors or the pin has been pushed further into the bank by a possible collision with floating refuse. On the day of the measurements, seven pins have not been accessible due to high water levels. To compare the measurements with the SEPAL results, it is necessary to derive long-term average values from the measurements. The discharge regime during the measurement period is the main factor governing bank erosion and thus, the mean discharge over the five month period (147 days, 0.30 m^3 s^{-1}) and the long-term average (365 days, 0.43 m^3 s^{-1} based on flow data from 1986 to 2009) is used for a linear extrapolation to gain yearly average values. The bank retreat ranges from 0 to 3.6 cm for the five month period and 0 to 12.8 cm for the long-term average with a mean value of 2.4 cm yr^{-1} and a sediment input of 356 t yr^{-1} from bank erosion.

The field mapping of erosive areas close to the stream is used for a comparison with the ABAG calculations. The fields found to be erosive are displayed in Figure 4.3. In general the field erosion and the sediment input by surface runoff to the stream are assessed to be very low. In total 23 fields have been mapped as eroded, the largest being 1.3 ha, the mean area is 0.16 ha. The mapped total erosive area having potential to contribute sediment to the stream in case of intense rain events is 3.6 ha.

The daily suspended sediment samples can be used to calculate the total annual sediment load. The concentration ranges from 4.2 to 256 mg L^{-1} with a mean value of 22.3 mg L^{-1} over the 1.5-yr period. Multiplying the concentration with discharge data yields a load of 636 t yr^{-1}.

Table 4.3 Erosion pin readings (mean distance from pin tip to river bank); the calculated bank retreat for the fuve-month period and the yearly average; all values in [cm]

Pin number		2	3	4	5	6	7	8	9	16	17	18	19	20	21
Mean distance	07.07.2008	3.9	4.2	3.1	2.6	3.7	3.4	3.6	2.9	3.3	3.8	3.1	3.1	4.3	3.4
	01.12.2008	5.9	7.8	3.3	2.7	4.1	3.7	4.1	3.4	3.9	4.1	3.2	3.4	4.3	3.1
Bank retreat 5 mon		2.1	3.6	0.2	0.1	0.4	0.3	0.4	0.5	0.6	0.3	0.0	0.3	0.0	-0.3
Yearly average		7.4	12.8	0.6	0.3	1.4	1.2	1.6	1.6	2.0	1.2	0.0	0.9	0.1	--

Pin not available on 1st December 2008: 1, 10, 11, 12, 13, 14, 15
Calculation example for pin number 2 to gain the yearly average:
2.1 cm × (147 d / 365 d) × (0.43 m^3 s^{-1} / 0.3 m^3 s^{-1}) = 7.4 cm

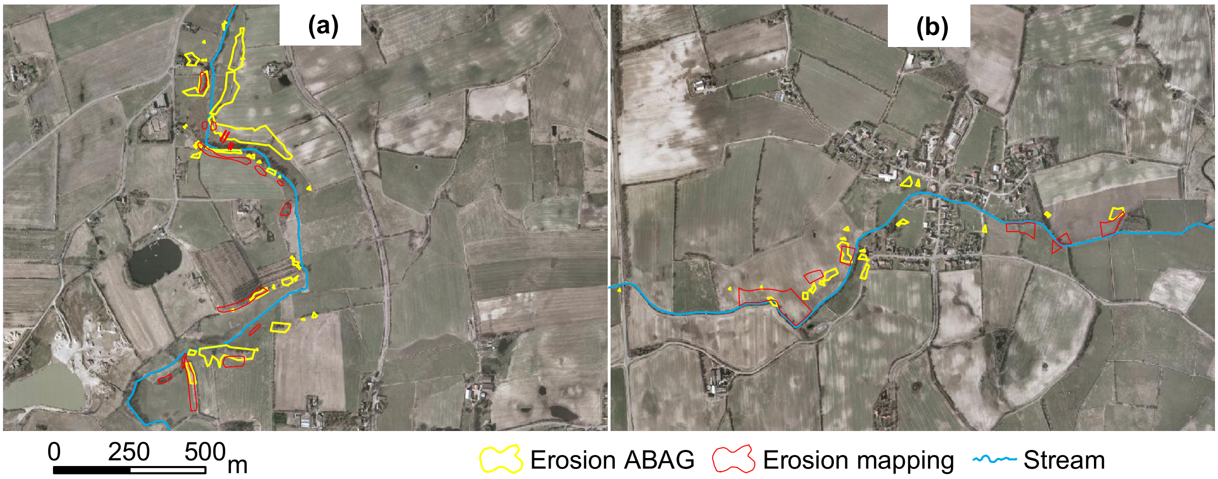

Figure 4.3 Comparison of field erosion from ABAG calculation with field mapping on the downstream (a) and upstream erosive region (b) on aerial photographs (LVA)

4.5 Conclusions

The scope of this paper is the development of a pragmatic methodology to estimate the shares of the drain-, river bank- and field sediment entry pathway in lowlands. The SEPAL approach has been tested in a rural 50-km² catchment in northern Germany. The results show that the bank erosion is predominant with 71 %, followed by the drains with 15 % and fields with 14 %. No studies in other German lowland catchments are available. However, the results are assessed plausible when comparing them with five lowland catchments in Denmark. There, the bank sediment input shares are: 75 % (Laubel et al., 1999), 77.5 % (Kronvang et al., 1997), 44 %, 82 % and 56.7 % (Hasholt, 1988), while the rest is caused by drain and field erosion input.

SEPAL is slightly overestimating the measured bank erosion as the formula calculated a 0.5 cm yr^{-1} higher mean bank retreat and 84 t yr^{-1} higher soil loss from the banks as the erosion pin readings. Considering the simple approach of both measurement and calculation, this is reasonable. Uncertainties can be due to the extrapolation of the five month pin measurements to gain the long-term average bank retreat value with the assumption that the measurement period is representative. The extrapolation takes the mean discharge into consideration, but other factors like peak flow intensity and distribution, freezing and thawing processes or plant cover are neglected. An improvement of the results is expected by extending the pin measurements to longer time periods. The discrepancy can also be due to the simple approach of the Dickinson formula which does not take the flow regime, bank slope and bank vegetation cover into consideration. However, Dickinson's formula appears to be a suitable compromise between exactness and data requirements for this task.

Hempel (1963) and Meyer (1996) stress, that the sediment input from the fields in the Kielstau region is very low. In fact, field erosion only occurs at two 1-km-long river sections. The ABAG does not consider deposition and is thus susceptible to overestimating the actual sediment input into the streams. This is taken into account by restricting the application to adjacent river areas using the threshold value (V_s). Overlaying the part of the ABAG map at the main stream channel with the field mapping (Figure 4.3a and 3b) shows that generally, some mapped erosive fields are also depicted by the ABAG, while many fields that are estimated to be erosive have not been identified in the mapping campaign. Reasons for this can either be the field mapping at the end of April as some winter erosion events might not be visible anymore and also the coarse and generalized soil and land use map used for the ABAG calculations. Probably the main cause for this discrepancy is the low soil loss values from the ABAG in the range of < 1 t ha^{-1} yr^{-1} which can be difficult to find in the field. The fact that the location and spatial extent of the fields show a relatively low agreement can be caused by the temporal difference between the data (early 1990s) and the mapping (2008). This can result in different field erosion locations due to the dynamic process of erosion and deposition. It can be concluded that the main erosive regions along the river have been identified sufficiently, but that the agreement on the smaller scale lacks exactness due to the mentioned objections.

Although the actual sediment input is expected to be higher than the suspended sediment transport due to possible deposition occurring in the stream, the calculated mean sediment input of 616 t yr^{-1} is within reasonable accordance with the load of 636 t yr^{-1} calculated from the suspended sediment samples. Uncertainties and the most likely reason for underestimations are considered to be caused by the drain depiction, as the implemented drain sediment fractions from the literature are relatively low compared to sites with mainly clayey soils. Furthermore it is likely that local factors like the type and age of drain pipe, the existence and type of filter between drain pipe and soil, the number and location of sediment traps in the drain system as well as the common practice of farmers to purge the drain system in early spring influence this process.

SEPAL has proven to be easily and quickly to implement with only little data requirements. The comparison with other lowland studies and the measurements show that the approach is plausible. But as the obtained results have a considerable degree of uncertainty, the method should only be used as a first approach to gain an overview about possible predominant sediment pathways in lowlands. Such information can be valuable especially for integrated catchment modelling approaches that can depict field- and river bank- but not drain-sediment input. The implemented regression equation for the drain input should and can easily be enhanced and extended if further data and research becomes available. In order to gain more reliable data especially for the drain sediment input, suggestions for further work are an intensification and temporal expansion of the measurement campaigns and collaborating with farmers concerning their drain maintenance so that dependencies of the sediment concentration can be derived and a physically based approach can be established.

4.6 Acknowledgement

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Chapter 5 Across the scales: From catchment hydrology to instream hydraulics

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Abstract Assessing the impact of large-scale processes on small scales requires modelling of landscape- and instream processes in an integrated manner. This paper describes the development of a three-step modelling cascade through connecting public domain models in ArcGIS. The models used are the ecohydrological model SWAT, the one-dimensional (1D) hydraulic model HEC-RAS and the two-dimensional (2D) hydraulic model AdH. Dynamic data transfer between the models as well as the development of an ArcGIS interface for AdH was important to apply the model system in two different catchments. The combined models have proven to be a valuable tool for assessing water and sediment fluxes from the catchment down to the reach scale and can be used for environmental assessments on different scales.

5.1 Introduction

The human interaction with the natural environment through catchment management has changed the environment considerably. The need has grown to foresee the impact of human actions on streams and catchments but also vice versa. Altering catchment attributes inevitably impacts instream processes in a cause-effect chain. Integrated modelling tools can depict these dependencies on variable scales and simulate parameters like water quality and quantity, flow velocity distributions, erosion, sediment transport and aquatic habitat suitability. For using the models in academic training, additional model requirements are user friendliness, robustness, future-proof through constant development, availability in the public domain and usability in a GIS.

The SWAT (Arnold et al., 1998) model, developed by the U.S. Department of Agriculture - Agricultural Research Station (USDA-ARS), with its GIS interface ArcSWAT (Winchell et al., 2007) is a widely used and suitable tool for covering these tasks on the catchment dimension. Land use changes, discharge regime, impact of drainages, groundwater levels, urban or rural water quality and climate change can be dealt with on various catchment sizes for time scales of days to decades. It also has a long history of application at the Department of Hydrology and Water Resources Management Kiel (Schmalz and Fohrer, 2009). Clearly, different physical processes are governing water and particle fluxes on the catchment- and reach scale. For depicting spatially explicit instream processes on the reach scale, 1D hydraulic models are suitable tools. They are capable of calculating hydraulic flow properties like velocity, depth and shear stress, sediment erosion and deposition on small streams up to complex river systems in reasonably fast computation times. The U.S. Army Corps of Engineers (USACE) HEC-RAS (USACE, 2010b) model with its GIS interface HEC-GeoRAS (USACE, 2011) is utilised by major water-related administrations, universities and engineers worldwide and fulfils the mentioned requirements. However, the 1D calculation limits the application to linear systems. It is not possible to satisfactorily depict the dominating processes on broad and short river sections or to assess small-scale hydraulic impact on substrates. 2D models are used for such tasks. The USACE Engineer Research and Development Center (ERDC) has developed AdH (Berger et al., 2011), a model that is capable of depicting hydraulic flow, erosion and sedimentation processes in user-defined resolutions. Unlike the previous models, AdH has no ArcGIS interface but the prospect of being linked to HEC-RAS (Brunner, 2011), the automatic adaptation of mesh resolution and time steps during model runs and the fully MPI-parallelised code makes it an ideal tool for the described tasks.

The mentioned differences between the three model types are the main reason why development is not undertaken by one workgroup, resulting in individual programs. The need for model coupling and standardized data transfer has resulted in promising initiatives like the Open Modelling Interface or the Hydrologic Engineering Center Data Storage System. No standard is however so far used by the models described or other models with comparable capabilities. Similar to most environmental modellers, students and scientists at the Department of Hydrology and Water Resources Management Kiel use GIS with their data storage and data modification capabilities. It is thus the aim of this paper to present solutions for a quick and user friendly data transfer between SWAT and HEC-RAS including the development of a pre- and post-processing GIS interface for AdH.

5.2 Methods

Figure 5.1 gives an overview of the model structure. Input data is supplied in GIS databases. ArcGIS user interfaces are available for SWAT and HEC-RAS, while for AdH an

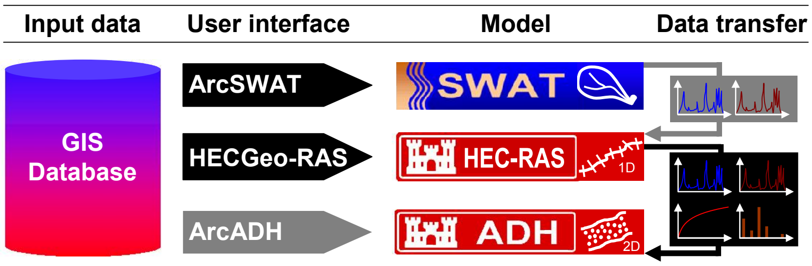

Figure 5.1 Structure of the input data, interfaces, models and data transfer; grey colors indicate the developed tools

ArcGIS interface had to be created. The first subsection below describes the development and operation principle of this interface. Each of the three models is run as a standalone program after the input files have been created. Discharge and sediment load data from the SWAT run have to be transferred to HEC-RAS. This is described in the second section below. Subsequently, the usage of HEC-RAS results within AdH is explained. Flow, sediment load, discharge rating curve and grain size distributions are transferred. The last subchapter gives a brief overview of the application of the models in two different environments, in Germany's northern lowlands and mid-range mountains.

5.2.1 User interface development ArcAdH

The newly developed ArcGIS 9.3 interface for the 2D hydraulic model AdH consists of two modules. ArcAdHin uses GIS and user input data to create the AdH input files. After a model run is successfully carried out, ArcAdHout reads the simulation results so that they can be displayed in the GIS. Three files are needed to run AdH: '.3dm' is a 2D triangular finite element mesh with coordinates, elevation and material information for each triangle node. '.bc' is the boundary condition file which contains physical properties and control parameters. '.hot' is the hotstart file which holds information of the initial flow conditions for each mesh node. For creating the mesh file, the core part of ArcAdHin is the utilisation of the Triangle program developed by Shewchuk (2002). Triangle is a flexible open source, 2D quality mesh generator and Delaunay triangulator. The program is supplied in C code and after compilation can be run as a batch program with control switches. A GIS source shapefile (Figure 5.2a) contains the substrate distribution on the modelling domain. It is transferred to a Planar Straight Line Graph (PSLG) which contains all polygon vertices of the original shapefile (Figure 5.2b) and a point within each closed polygon. For creating high quality meshes with Triangle, a conforming Delaunay triangulation is used on the PSLG with a minimum angle threshold to avoid thin triangles.

ArcAdHin assigns the shapefiles material (substrate) attributes to each area which is then appointed to each triangle element. An area constraint value represents the maximum size any triangle may have for each material group. The triangulation result with the added mesh nodes can be displayed in ArcGIS (Figure 5.2c). Finally, ArcAdHin writes the elevation to each mesh node from a DEM (Figure 5.2d) and writes the '.3dm' file. The boundary condition file is assembled from pre-defined '.dbf' tables in which the user has to supply general simulation parameters, material properties and time controls according to Berger et al. (2011).

Generation of the '.bc' file is coupled to mesh file creation because node numbers from the mesh have to be assigned to edgestrings (influx, outflux locations) which have to be supplied on a polygon shapefile. The hotstart file is created from user supplied flow values and the discharge rating curve that is needed as a boundary condition. All velocities at $t = 0$ are set to $0\ m\ s^{-1}$. The initial water level is taken from the rating curve and assumed to be horizontal over the modelling domain. Each mesh node is assigned the water depth resulting from the water surface elevation minus the mesh point elevation. The AdH input files are copied to the specified locations from which the model is run.

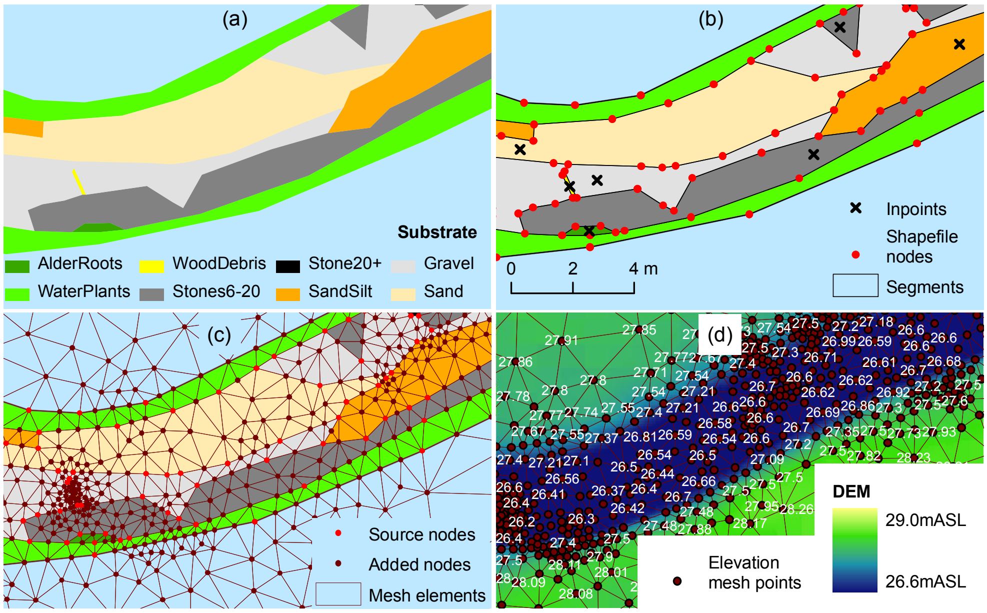

Figure 5.2 ArcAdHin mesh generation: (a) material shapefile with substrate polygons; (b) the PSLG with segments, nodes and inpoints; (c) the triangulated mesh, elements with source- and added nodes; (d) the elevation assigned to each node

ArcAdHout reads the output files generated by AdH. The ASCII files contain the output parameters in subsequent node order and for all output-requested time steps. All or single AdH output files can be handed over to the script, that converts the requested parameters to polygon shapefiles. The result of one time step represents one column in the shapefiles attribute table, which enables a straightforward visualisation and comparison of different time steps. Results in depths (Figure 5.3a), absolute velocity (Figure 5.3b), error thresholds and bed elevation change (Figure 5.3d) are averaged for each triangular element and can directly be compared with available, georeferenced observed data. Velocity vectors (value and angle, Figure 5.3c) are saved in a point shapefile for each mesh node.

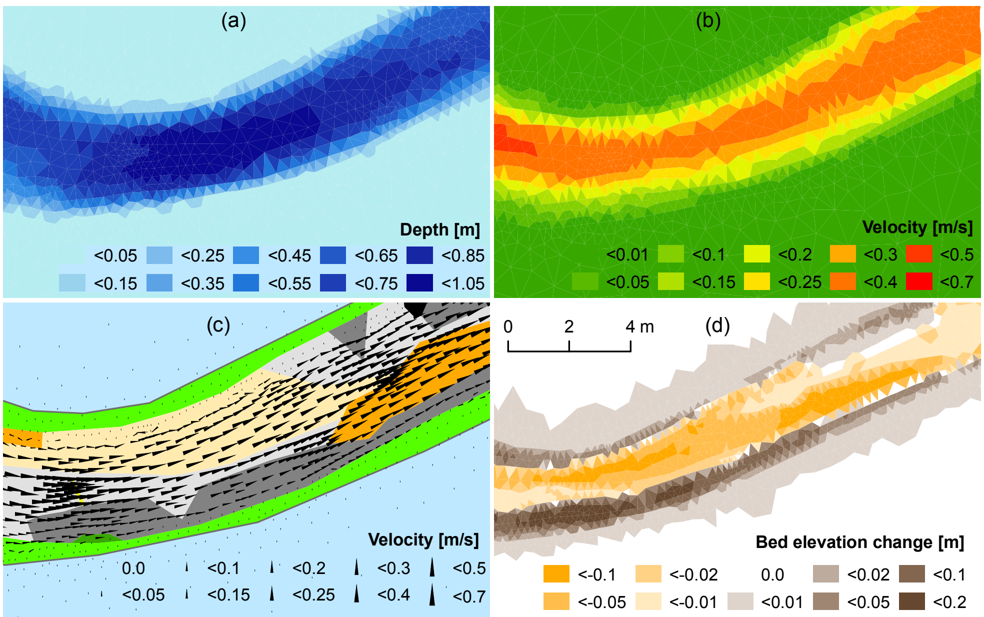

Figure 5.3 ArcAdHout output visualisation: (a) water depth, (b) flow velocity, (c) flow velocity vectors over substrate, (d) bed elevation change over flexible time periods

5.2.2 Data transfer from the catchment model SWAT to the instream 1D model HEC-RAS

To connect SWAT with HEC-RAS, results from the hydrologic model are used as an input to the hydraulic channel model. Depending on the complexity of the watershed, amount of parameters to be handed over and number of scenarios that are to be modelled, data formatting, adjustment and transfer can consume a considerable amount of time. SWAT temporal data time series have to be handed over on the subbasin level to HEC-RAS. To achieve this, SWAT and HEC-RAS model domains are overlaid in the GIS and the subbasin outlets are snapped to the closest HEC-RAS cross section. Figure 5.4 shows a schematic in which HEC-RAS depicts only a part of the SWAT river network, which is a relatively common situation. The interface extracts 'flow series' at upstream HEC-RAS boundaries (Figure 5.4, subbasins 1, 2, 4), 'uniform flow series' are used to depict confluences along the modelled channel (Figure 5.4, subbasins 3, 7, 8) and 'uniform lateral flow series' are groundwater or overland inflows occurring equally distributed along the modelled channel at each HEC-RAS cross section in between confluences (Figure 5.4, subbasins 4, 5, 6, 9). Information required within each time series is 'flow duration', the 'computation increment' during flow duration and the 'flow value', which is all calculated based on the flow value and the SWAT time step.

If sediment modelling is desired, the interface also transfers sediment load values for each time step. SWAT calculates the sediment input from fields to the stream with the Modified Universal Soil Loss Equation (MUSLE, Williams, 1995). The interface reads these values and processes the data series according to a similar methodology as for the water fluxes. The difference to the flow methodology is the fact that HEC-RAS cannot process uniform lateral sediment influxes. Thus, all sediment loads produced in a subbasin containing no tributary (subbasins 4, 5, 6, 9) are distributed to the tributaries. In addition, HEC-RAS requires the grain size distribution of the SWAT sediment load. Therefore, the interface reads the SWAT soil database and calculates the average grain size distribution of the upper soil layer for each subbasin. In low-gradient environments the option is added to exclude grain sizes larger than the coarse sand fraction. Water temperature data also has to be supplied to HEC-RAS in the format 'duration' and 'temperature'. The values are calculated according to Stefan and Preud'homme (1993) from SWAT air temperature data.

If not otherwise stated by the user, all calculations are carried out for each SWAT time step individually. For long-term simulations, the data load to HEC-RAS and computation times may be too high and thus, the user has the possibility to simplify data: By supplying a threshold of digits, each flow, sediment or temperature value is rounded to the entered number of digits. As long as the rounded value does not change, the interface summarizes the data and generates one flow value for multiple SWAT outputs. The resulting time series will have less entries, but increased flow- and input durations for each summarized entry. The user can copy the resulting data directly to the HEC-RAS flow editor.

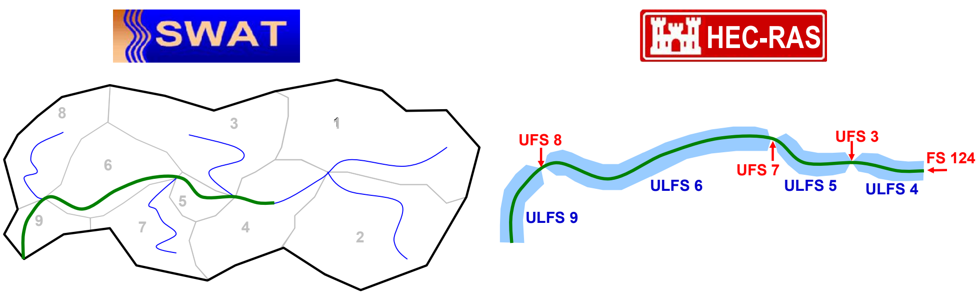

Figure 5.4 Schematic of data transfer SWAT to HEC-RAS: SWAT catchment with subbasins, stream channels and HEC-RAS channel (green); HEC-RAS channel with input locations of flow series (FS), uniform flow series (UFS), uniform lateral flow series (ULFS) with subbasin numbers

5.2.3 Data transfer from the 1D HEC-RAS model to the 2D AdH model

High resolution simulation of river reaches require detailed water surface, discharge and transport data which are rarely part of a general monitoring network. Additionally, if habitat assessments are to be carried out in pristine areas, stage discharge relationships, sediment or flow data is usually not available. Simulation results from HEC-RAS provide important boundary condition data for running AdH in such cases. Data transfer between the models can be achieved without applying additional tools. Discharge values and sediment loads for different grain sizes are retrieved from HEC-RAS at the cross sections marking the upstream boundary of the AdH mesh. At the downstream boundary, the calculated stage discharge relationship from HEC-RAS is copied from the respective cross section. AdH water surface elevations are read from this rating curve at all modelled flow values.

5.2.4 Application of the coupled model cascade

Based on the tools and methodologies explained above, it is possible to carry out GIS-based simulations on the catchment- down to the reach scale. The model cascade was successfully applied in the Kielstau, a 50-km² lowland catchment in northern Germany, and the Kinzig, a 1,065-km² catchment in Germany's mid-range mountains. Figure 5.5 shows the model domains. SWAT comprises the entire catchment areas. HEC-RAS has been applied on a 9-km (Kielstau) and a 32.8-km-long main channel section (Kinzig). The model domain of AdH in the Kielstau is about 230 m long and located at the outlet of the catchment. In the Kinzig,

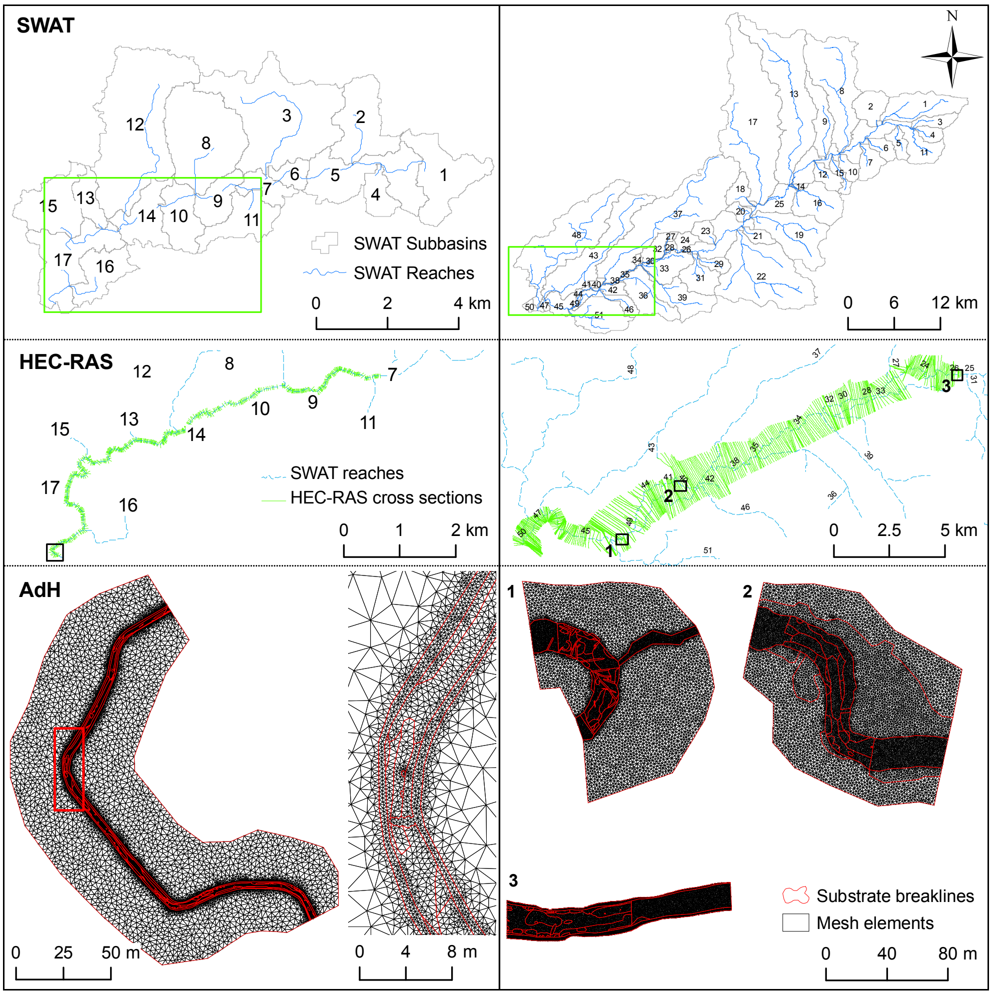

Figure 5.5 SWAT, HEC-RAS, AdH model domains: Left Kielstau, right Kinzig catchment. From top to bottom: SWAT model area, location of HEC-RAS model marked with rectangle; HEC-RAS model area, AdH model area marked with rectangle; AdH model domains with detailed part of the Kielstau mesh

three AdH models have been setup, depicting each about 150-m channel lengths. Catchment water fluxes and sediment erosion from the fields have been modelled in both catchments for a time period of ten years with SWAT. HEC-RAS was run for the same time period to depict hydraulic parameters as well as sedimentation along the Kinzig channel. In the Kielstau, channel bed erosion was modelled additionally. In the Kinzig, AdH was applied to model sediment deposition in the floodplains for a five-day high flow period and fine sediment deposition on instream substrate for a ten-day period at the three locations. AdH has been used in the Kielstau to simulate small-scale substrate changes over one year. The model results will be compared with observed data, ranging from hydraulic parameter measurements, to suspended sediment samples and substrate mappings.

5.3 Discussion

The aim of the project was to select modelling tools for application in a wide range of water management related projects and connect them in the ArcGIS environment. SWAT is able to simulate catchment processes in a quick, robust and user friendly way but is also limited to those applications as the implemented algorithms are not sufficient to depict hydraulic instream processes. HEC-RAS, on the other hand, is predestined for depicting 1D-hydraulic processes in long channels but depends on hydrologic input data and cannot be used to simulate seamless, small-scale hydraulics in broad channels. AdH can model such processes two-dimensionally and only the available computational power limits the detail and size of the model domain. It can thus be concluded, that the weaknesses of one model can be complemented by the other models which makes them an ideal combination within a modelling system. To jointly use the models with GIS databases, data transfer tools and an interface for AdH had to be developed. The described programs are capable of quickly transferring and supplying data for the models. The first results from the application of the model cascade have proven its applicability and usefulness in water management projects. The selected models can be applied both in the lowlands and the mid-range mountain environments in different landscapes. The model system proves to be a valuable tool for assessing water- and sediment fluxes from the catchment down to the reach scale. The next step is to use validated model results for aquatic habitat assessments on different scales.

5.4 Acknowledgement

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Chapter 6 Application of a hydrological-hydraulic modelling cascade in lowlands for investigating water and sediment fluxes in catchment, channel and reach

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Abstract: This study shows a comprehensive simulation of water and sediment fluxes from the catchment to the reach scale. We describe the application of a modelling cascade in a well-researched study catchment through connecting state-of-the-art public domain models in ArcGIS. Three models are used consecutively: First, the hydrological model SWAT to evaluate water balances, sediment input from fields and tile drains as a function of catchment characteristics; second, the 1D hydraulic model HEC-RAS to depict channel erosion and sedimentation along a 9-km channel one-dimensionally; and third, the 2D hydraulic model AdH for simulating detailed substrate changes in a 230-m-long reach section over the course of one year. Model performance for the water fluxes is very good, sediment fluxes and substrate changes are simulated with good agreement to observed data. Improvement of tile drain sediment load, simulation of different substrate deposition events and carrying out data sensitivity tests are suggested as future work. Main advantages that can be deduced from this study are separate representation of field, drain and bank erosion processes; shown adaptability to lowland catchments and transferability to other catchments; and the usability of the model's output for habitat assessments.

Keywords: SWAT; HEC-RAS; AdH; SEDLIB; Hydrology; Sediment transport; Multiple scales.

6.1 Introduction

The simulation of river- and aquatic habitat changes, based on environmental and anthropogenic forcing, is an ongoing topic in river research (Jähnig et al., 2012; Kiesel at al., 2009a). The movement and characteristics of water and sediment are pivotal for the functioning of riverine ecosystems (Baron et al., 2002). Water and sediment fluxes are interlinked from the catchment fields down to the instream micro scale: Water erosion on agriculturally used fields directly affects soil fertility (Uri, 2000). Depending on the geomorphology, high proportions of these eroded, mostly nutrient-rich, fine sediments can enter the streams. In lowland areas and artificially drained wetlands, an additional pathway is the tile drains that contribute sediment to the streams (Kiesel at al., 2009b; Russell et al., 2001). Sediment gets stored, re-entrained, transported or deposited in the streams and becomes part of the instream processes. These processes along the rivers flow paths have various effects on stream properties and habitats (Veihe et al., 2011). They change conveyance, can cause siltation, and can damage waterways, hydraulic structures and adjacent land property. But instream erosion and sedimentation processes are also desired and important characteristics of functioning aquatic ecosystems (Florsheim et al., 2008). On the one hand, this interconnectedness between landscape- and instream processes requires a combined depiction when investigating sediment movement across scales (Deasy et al., 2011; Jarritt and Lawrence, 2007). On the other hand, a quantitative distinction between field and instream erosion is important, for example, for developing target-oriented best management practices for sediment management or when aiming for natural environmental conditions where nutrient loaded fine sediment inputs are less desired than sediment input from banks. In any case, when investigating water induced movement of sediment, it is important that the characteristics of the main driver, the water fluxes, are known (Merritt et al., 2003).

Mathematical modelling of the main processes governing water and sediment transport in a complex environment is a useful and well accepted approach to investigate the impacts on different scales. Hydrologic and hydraulic models can be used in conjunction to depict landscape and instream processes in an interconnected, yet distinct manner to obtain quantitatively discrete results. Numerous studies are available that focus on parts of the integrated hydrological and hydraulic chain, e.g. on catchment hydrology and field erosion (Borah and Bera, 2004), instream hydraulic, sediment transport and delivery processes (Etemad-Shahidi et al., 2010), and micro-scale substrate assessments (Hauer et al., 2011; Pasternack, 2011). However, an integrated and continuous examination of water and sediment fluxes from the catchment down to the micro-reach scale could not be found in the literature. This paper shows such an integrated assessment through the application of a three-step modelling cascade. In order to achieve seamless results, three models need to be run for obtaining model output on all scales, which is in our view the simplest approach. Still, it requires an extensive database and modelling efforts, but the benefits are three-fold: First, temporal and spatial process knowledge on water and sediment fluxes are obtained, second, results are generated seamlessly from the catchment down to the river-reach scale and third, through the extensive data input, the model system is potentially able to depict the influence of global change, modifications of catchment properties and channel alterations on different scales up to instream substrates. To fulfil this aim, intermediate objectives have to be defined: (1) the realistic depiction of water fluxes which act as the driving forces in particle transport and (2) the ability to simulate the three main sediment entry pathways in lowlands: field erosion, tile drain sediment input and channel erosion.

6.2 Material and methods

6.2.1 Study area

The model cascade is applied in the Kielstau, a northern German lowland catchment, about 10 km south-east from the city of Flensburg. The low relief of the catchment, its rolling hills topography, high number of landscape depressions and a poorly developed overland drainage system cause low surface runoff fraction, low hydraulic gradients and a significant groundwater influence on the catchment hydrology (Kiesel et al., 2010a). Most parts of the catchment are agriculturally used, which is the main reason why 31 % of the catchment area is artificially drained through tile drains, constructed during the second half of the last century. Lake Winderatt, with a surface area of 2 ha, is located in the upper third of the Kielstau River. The lake's water outflow is artificially ponded through a fixed weir. A summary of catchment characteristics is supplied in Table 6.1. In the mid-twentieth century, the river channel was straightened and incised, which decreased flow length and stream roughness. Channel slope and flow velocities increased as a result, altering not only the hydraulic regime but also sediment processes. The Kielstau is classified as a lowland gravel bed river, but there are also sections of the stream that are covered with sand layers which show high dynamics over the course of one year. A WFD monitoring station is located within the Kielstau just upstream of the catchment outlet at gauge Soltfeld. The catchment was chosen to be Germany's UNESCO ecohydrology demo site in 2010 (Fohrer and Schmalz, 2012), also due to the available database collected and research done during the last decade (Schmalz and Fohrer, 2010). The combination of being a well researched study area and the public attention to the catchment's status makes the Kielstau an ideal example for testing new modelling methodologies.

6.2.2 Description of the model cascade

We propose the consecutive application of three models: a hydrologic model, a one-dimensional hydraulic model and a two-dimensional hydraulic model. Figure 6.1 shows the application range of each individual model within the three-step model cascade. The maps on the left hand side visualise the scale on which each model is applied. The flowchart on the right hand side describes the impacts (white on black) that are depicted with each model and the results (black on grey) which are used as an input to the next model on the lower scale. The flowchart illustrates that this consecutive application leads to a consideration of large-scale impacts on small scales. It is important to note that this consideration can only be successful if a continuous temporal and spatial connection between the models is established and if the same time period is simulated in the three models. For each model, the application scale is summarized in Table 6.2, as well as the time for which the models are run and the parameters which are transferred to the next model.

First, the SWAT model (Arnold et al., 1998), in the version 2005, is applied on the whole catchment area of the Kielstau (Figure 6.1a). The model can be used for simulating the impact

Table 6.1 Kielstau catchment characteristics

Elevation	28–78 m ASL (LVA, 1992–2004)	Soils	Haplic and Stagnic Luvisols, Sapric Histosols (BGR, 1999)
Size	50 km² (LVA, 1992–2004)	Land use	arable land (56 %), pasture (26 %), forest (8 %), urban (3 %) (DLR, 1995; MOBIO, 1999)
Population	4,450 (Golon, 2009)	Tile drains	31 % of catchment area (Fohrer et al., 2007)
Longest flow path	16.2 km (LVA, 1992–2004)	Climate	mean: 8.2 °C, 893 mm precipitation (DWD, 2010)
Mean slope	1.2 ‰ (LVA, 1992–2004)	Runoff	mean: 0.42 $m^3\ s^{-1}$ at gauge Soltfeld (LKN, 2010)

of catchment characteristics, climate and land use management on catchment water balance and sediment. The SWAT model and its source code are freely available. It has been and is applied in various EU WFD related projects (Arnold and Fohrer, 2005) and by the US Environmental Protection Agency, the USDA as well as by universities and consultants around the world (Gassman et al., 2007). The application ranges from the field scale (Maharjan et al., 2013) to simulations of continents (Schuol et al., 2008) in hourly to yearly time steps. Within

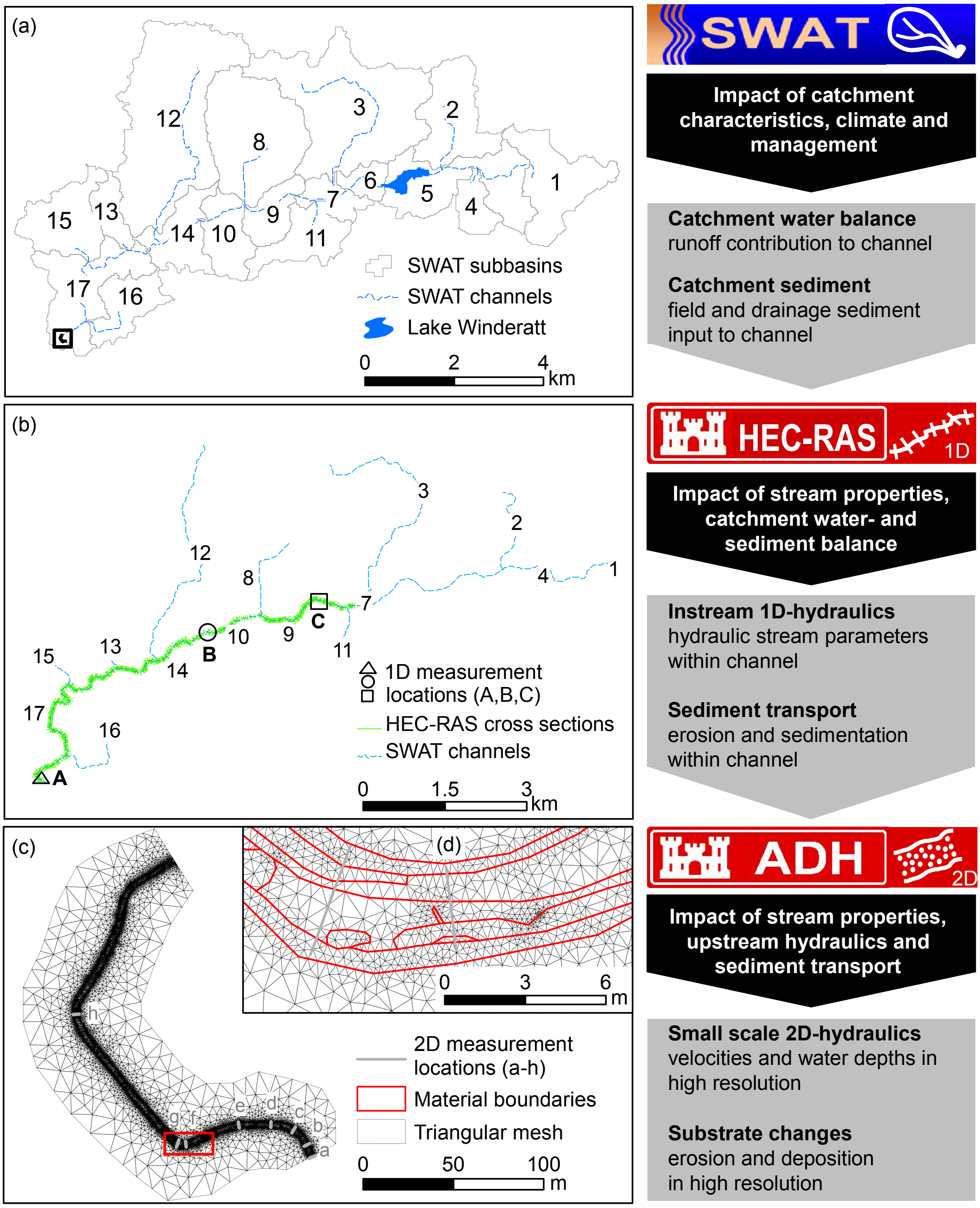

Figure 6.1 SWAT, HEC-RAS and AdH application in a modelling cascade: SWAT model domain (a) with Lake Winderatt in subbasin five and location of the outlet at gauge Soltfeld marked with rectangle, HEC-RAS model domain (b) with location of measurement locations (A, B, C), AdH model domain upstream of gauge Soltfeld including location of measured cross sections a–h (c) and a detailed part of the triangular element mesh with material boundaries (d)

Table 6.2 Information about model application within the model cascade

Model	Scale	Time	Parameters transferred from model output to next model
SWAT	Catchment: 50 km²	(1999–2009) 04/2008–04/2009	flow from groundwater and fields sediment load from fields, drains with additional model SEPAL
HEC-RAS	Reach: 9 km	(2007–2009) 04/2008–04/2009	stream discharge total sediment load
AdH	Reach: 230 m	(04/2008–04/2009) 04/2008–04/2009	–

the model cascade, SWAT is used to simulate runoff contribution and sediment input from the catchment to the reach. Although SWAT's channel erodibility processes have shown to give comparable degradation results to a HEC-RAS model (Allen et al., 2008), SWAT's spatial representation through subbasins is disadvantageous for obtaining differentiated instream results along a stream channel since the same result value is given for each reach, which can be many kilometres long.

The decision to use a separate instream model for depicting processes in the main channel is thus driven by the need for high resolution results with output parameters that the SWAT model is not able to supply, e.g. velocity distributions along the rivers flow path. The 1D hydraulic model HEC-RAS (USACE, 2010a) is used to simulate the 9-km main channel downstream of Lake Winderatt to gauge Soltfeld at the catchment outlet (Figure 6.1b). HEC-RAS is a well tested and widely applied model which has been developed by the USACE-Hydrologic Engineering Centre and is also available in the public domain. The model is utilised by major US water-related administrations, universities and engineers worldwide. The application ranges from small-scale drainage systems to large river networks, comprising subhourly peak flow calculations as well as simulations for years (SWWRP, 2011). Within the model cascade, the HEC-RAS model is used to simulate hydraulic stream parameters as well as erosion and sedimentation within the channel.

As it is not possible to satisfactorily depict detailed processes on broad and short river sections or to assess seamless spatial coverage, small-scale hydraulic impact on substrates with the HEC-RAS model, the model AdH (Berger et al., 2011) linked to the SEDLIB sediment transport library (Brown et al., 2012) is used to simulate the 230 m long river section upstream of gauge Soltfeld. AdH is developed at the ERDC from the USACE. It is capable of simulating the impact of stream properties, upstream hydraulics and sediment transport on small-scale hydraulics and substrates. AdH can describe both saturated and unsaturated groundwater, overland flow, three-dimensional Navier-Stokes and three-dimensional shallow water problems, in addition to the 2D shallow water module applied here (Berger et al., 2011). The software is also available in the public domain. AdH runs on both Windows and UNIX based multi processor machines and is fully parallelised. In the near future, the model will be dynamically linked to HEC-RAS (Brunner, 2011). The user can set thresholds which define the accuracy of the calculated result. The model meets these thresholds by an automatic adaptation of mesh resolution and time steps during model runs. Within the model cascade, AdH is used to simulate velocities and water depths as well as erosion and deposition of sediments in high resolution and two-dimensionally.

6.2.3 Data transfer within the model cascade

SWAT daily flow and sediment load output time series for every subbasin are necessary boundary condition input data to HEC-RAS. A SWAT-HEC-RAS interface was developed (Kiesel et al., 2012) that overlays the SWAT catchment map with the HEC-RAS model domain. It assembles quasi-unsteady flow values, which are steady flow values over defined time increments, for every HEC-RAS cross section. The SWAT model's sediment load values for every time step are allocated to the tributaries draining into the channel modelled with the HEC-RAS model.

Data transfer from the HEC-RAS model to the AdH model can be achieved without the usage of additional tools. Flow and sediment load time series from the HEC-RAS cross section upstream of the AdH model boundary can be directly copied to the AdH input file.

6.2.4 Model algorithms for water processes

SWAT depicts the land phase of the hydrological cycle and its impacts by natural and anthropogenic processes on any hydrologically relevant area. For the present study, the Penman-Monteith equation for evapotranspiration, the SCS-CN method for modelling surface runoff, and the kinematic storage model for interflow are used. The SWAT model calculates the water balance of two groundwater aquifers. The first aquifer enables return flow to surface water or can be tapped through plants, while groundwater entering the second aquifer is lost from the system. A variable storage coefficient method is used to route the flow components across user-defined subbasins to the catchment outlet. Spatially explicit streamflow values are available at each subbasin outlets which can be used to depict tributary flows to main channels. (Neitsch et al., 2009)

The HEC-RAS model is used to simulate 1D open channel hydraulics in river networks at user-defined cross sections. Within this study, steady state simulations are used for each individual SWAT daily time step. Hydraulic parameters are calculated through the energy equation which is solved with the standard step method in case of basic flow problems. For mixed flow regimes and for hydraulic structures the momentum equation is applied within HEC-RAS. For each simulated cross section location, depth- and width averaged parameter values are calculated for the channel. (USACE, 2010b)

For the present study, AdH is used to simulate 2D shallow water flow in a natural, open channel. Therefore, the depth-averaged Navier-Stokes equation is solved for the triangular finite element mesh. The numerical solvers available in AdH are UMFPACK (Davis, 2004) or ParMETIS (Karypis and Kumar, 1998), of which the first was applied in this study.

6.2.5 Model algorithms for sediment processes

SWAT employs the MUSLE (Williams, 1995) to calculate field erosion. Through the exchange of the original rainfall erositivity factor of the USLE against a runoff factor, the MUSLE is assessed more applicable to single events and to consider delivery ratios (Williams, 1995). Erosion types that can be depicted with the MUSLE are sheet and rill erosion, which we refer to as field erosion. Besides field erosion, tile drains are another source of sediment from lowland catchments to the water bodies (Chapman et al., 2005; Kiesel et al., 2009b). The SWAT model is currently not able to depict sediment input from tile drains to the stream. Other modelling concepts to depict this pathway are not available either. A methodology was developed to model daily tile drain sediment loads. The impact of tile drains on the catchments sediment load has been assessed with a GIS-based tool together with field measurements (SEPAL, Kiesel et al., 2009b). In this study of the Kielstau catchment, the long-term, basin-wide average tile drain sediment input fraction was found to be 15 % and field sediment input was 14 % which is in coherence with studies carried out in catchments with similar characteristics (Kronvang et al., 1997). Based on these fractions, the yearly tile drain sediment load is calculated for each subbasin individually:

$$ST_{yi} = SF_{yi} \cdot \frac{fr_T}{fr_F} \qquad \{6.1\}$$

where i is the subbasin, ST_y is the sediment load from tile drains for the current year [kg], SF_y is the yearly sediment load from field erosion calculated by SWAT [kg], fr_T is the tile drain [%] and fr_F is the field sediment input fraction [%] supplied by SEPAL. Together with SWAT's daily modelled tile drain flow, ST_y is used to calculate daily sediment load from fields and drains for each subbasin:

$$S_{TOTdi} = SF_{di} + \frac{QT_{di}}{QT_{yi}} \cdot ST_{yi} \qquad \{6.2\}$$

where S_{TOTd} is the total daily sediment load from fields and tile drains [kg], SF_d is the daily sediment load from field erosion calculated by SWAT [kg], QT_d is the daily tile drain flow [m^3], QT_y is the yearly tile drain flow [m^3]. The equations presented here are not implemented in SWAT, but applied on SWAT's MUSLE and tile drain flow output to obtain the total daily sediment load to the catchments streams.

For each time increment and each cross section, HEC-RAS solves the sediment continuity equation to compute the change in sediment volume based on the sediment transport capacity of the water. Bed elevation change and grain size distribution are then calculated at each node of all cross sections, which makes a spatially explicit depiction of erosion and sedimentation over time possible (USACE, 2010b).

Similarly to the HEC-RAS model, the AdH model requires substrate information, sediment influx and discharge time series. For each time step, an active layer is calculated within the AdH model which acts as a source of sediment to the bed layers in case of depositing sediment and as a sink of sediment from the bed layers in case of eroding sediment. Sediment transport capacity is computed for suspended transport and for bed load transport individually. Grain size distributions, bed layer properties and bed elevation changes are available across and along the river bed at every node of the triangular surface mesh.

6.2.6 Setup and calibration for modelling water fluxes

The data presented in Table 6.3 are used for driving the models' hydrological and hydraulic algorithms. All necessary data for the simulations are summarized there individually for each model. Data are obtained mainly from official sources, reports and own measurement campaigns (Table 6.3). The type of data is given through the information in brackets. The ArcGIS interface ArcSWAT (Winchell et al., 2007) is used to prepare SWAT model input data from spatial datasets. During the SWAT model setup and calibration process it is of importance to consider the hydrologic impact of tile drainages and landscape depressions of the Kielstau catchment (Kiesel et al., 2010a). Therefore, tools are used to obtain a spatially distributed drainage map (DRAINdist, Fohrer et al., 2007) and to estimate the surface water retention potential in lowlands (ERPL, Kiesel et al., 2010a) based on the consideration of closed sinks in a high quality DEM. The catchment is divided in 17 subbasins, so that all tributaries to the main channel of the Kielstau River are represented (Figure. 6.1a). The model is run with a ten-year climate dataset from 1999–2009 (calibration 1999–2003, validation 2004–2009) and discharge is calibrated first manually and then automatically on the catchment outlet. Most sensitive are groundwater (return flow threshold), surface water (curve number) and routing parameters (channel conductivity). The detailed SWAT model setup and calibration is described in Kiesel et al. (2010a).

HEC-RAS geometry data are derived from LiDAR data (LVA, 2008) of the floodplains and instream cross-sectional measurements. Bathymetry is interpolated in between the cross sections with a GIS tool (Merwade et al., 2008). The resulting instream grid is merged to the LiDAR floodplain DEM with spline interpolation. Cross sections for the conveyance calculations are extracted from this surface DEM to HEC-RAS in an average distance of 17 m spac-

Table 6.3 Data for depicting water fluxes in the Kielstau catchment, data type in brackets

	Used data	Data source and format
SWAT	DEM	25x25 m and 5x5 m DEM (LVA, 1992–2004; LVA, 2008) | (GIS)
	Stream network	DAV-WBV/LAND S.-H. (2006) | (GIS)
	Soil map	BÜK 1 : 200.000 (BGR, 1999) | (GIS)
	Drain map	Fohrer et al., (2007) | (GIS)
	Physical soil parameters	Borehole profile (LANU, 2006b); Ad-Hoc-AG Boden (2005); Baumer (1990); Janßen (2006); Post et al., (2000); Succow and Joosten (2001) | (GIS, text)
	Land use map	DLR, 1995; MOBIO, 1999 | (GIS)
	Vegetation parameters	SWAT database (Neitsch et al., 2009) | (table)
	Crop rotations and management	Bieger (2007) | (text)
	Lake properties	Grudzinski (2007) | (text)
	Climate data (precipitation, temperature, humidity, wind, solar)	DWD (2010) station Meierwik; IFM (2007); dew point calculation according to Sonntag and Heinze (1982) | (table)
	Discharge data for calibration	Discharge gauge Soltfeld (LKN, 2010) | (table)
HEC-RAS	Stream network	DAV-WBV/LAND S.-H. (2006) | (GIS)
	Cross sections	soilAQUA (2009) | (table)
	1 m DEM of floodplains	LVA (2008) | (GIS)
	Hydraulic roughness of stream bed and banks (Manning's n values)	Vegetation and physical properties from DAV-WBV/LAND S.-H. (2006); calculation according to Chow (1959) | (GIS, text)
	Hydraulic roughness of floodplains	Field mapping; Chow (1959) | (GIS, text)
	Hydraulic structures (bridges)	1 m DEM (LVA, 2008); aerial photos (LVA, 1992–2004) | (GIS)
	Discharge hydrographs	from SWAT simulation | (table)
	Discharge rating curve	LKN (2010) | (table)
	Water level & flow velocities	Tavares (2006) | (table)
AdH	Discharge hydrograph	from HEC-RAS simulation | (table)
	Substrate distribution	Thiemann (2008); field mapping | (GIS)
	Grain size distribution of substrates	Thiemann (2008); Labadi (2009) | (table)
	Detailed data for hydraulic roughness of bed and banks	Field mapping; Thiemann (2008); Chow (1959) | (GIS, Text)
	Detailed topography	Surveying; LVA (2008) | (GIS)

ing depending on the curvature of the stream using the interface HEC-GeoRAS (USACE, 2011). Information about channel characteristics is available from the state-wide river mapping scheme (DAV-WBV/LAND S.-H., 2006) divided into stream sections of about 10–200 m length depending on stream variability. Manning's n values for the channel are derived from these data using the roughness formula first proposed by Chow (1959). The formula incorporates substrate material, surface irregularities, channel cross section variation, obstructions, vegetation and meandering properties of the channel. Daily flow values are supplied from the SWAT model via the SWAT-HEC-RAS interface. The HEC-RAS model output can be compared at three locations (Figure 6.1b) along the main channel against measured water depths and flow velocities for 24 flow events (0.06–1.26 $m^3 s^{-1}$, Tavares, 2006). Manning's n values are adapted within plausible ranges to match observed data. The calibrated channel Manning's n values range between 0.02–0.06 with a medium value of 0.04.

For setting up the AdH model at the detailed 230-m-long river section, it is not possible to use commonly area-wide available data. Field surveys have been carried out with differential GPS and water depth measurements to record 22 cross sections for interpolating the stream´s bathymetry which is merged to the LiDAR-derived floodplain. Due to the small stream width of mainly 4 m, extensive care had to be taken to obtain measurements with a high accuracy, especially close to the stream banks. Additionally, a morphological mapping campaign has been conducted to obtain a shapefile of substrate distributions (Thiemann,

Table 6.4 Additional data to Table 6.3 for depicting sediment fluxes, data type in brackets

	Used data	Data source and format
SWAT	Soil erositivity and coarse fragment factor, cover and management factor, support practice factor	LANU (2006b); Ad-Hoc-AG Boden (2005); Neitsch et al. (2009); Williams et al. (1995); Dickinson et al. (1989); Schwertmann et al. (1987) | (GIS, table, text)
SWAT	Suspended sediment concentration in sewage plant discharges	Andersen (2006) | (table)
SWAT	Long-term average sediment input ratio from field- and tile drains	Modelling with SEPAL (Kiesel et al., 2009b) | (text)
SWAT	Suspended sediment concentration	Sampling and analysis | (table)
HEC-RAS	Sediment graphs with grain size distribution	from SWAT simulation and physical soil parameters | (table)
HEC-RAS	Grain size distribution river bank and bed	DAV-WBV/LAND S.-H. (2006) | (table)
HEC-RAS	Suspended sediment concentration	Sampling and analysis | (table)
AdH	Sediment graphs with grain size distribution	from HEC-RAS simulation | (table)
AdH	Physical substrate parameters	Ad-Hoc-AG Boden (2005) | (text)
AdH	Bed load transport	Field measurements; Labadi (2009) | (table)
AdH	Substrate changes over time	Substrate mapping (Thiemann, 2008, field mapping in 2009) | (GIS)

2008). The shapefile and surface DEM are used to create the triangular computation mesh with ArcAdH, an ArcGIS interface for AdH (Kiesel et al., 2012). Higher mesh resolution is assigned to regions in bends and highly variable substrates. Element sizes are between 0.08 and 0.2 m^2 in the channel and 5 m^2 in the floodplains. Manning's n values are defined for each mapped substrate. Higher roughness values are assigned to boulders, vegetation and dead wood as the thickness of these structures induce additional energy losses. Estimated eddy viscosities and Manning's n values are calibrated to match measured water surface slope and velocities at eight cross sections within the modelling domain (Figure 6.1c).

6.2.7 Setup and calibration for modelling sediment fluxes

On the basis of the calibrated water fluxes model cascade, the erosion and sediment transport algorithms of SWAT, HEC-RAS and AdH are parameterised and calibrated with data shown in Table 6.4, which need to be available in addition to the data presented in Table 6.3. Characteristics and structure of Table 6.4 is similar to Table 6.3. Daily sediment input from fields is simulated with the SWAT model. Daily tile drain sediment input to the stream is depicted with equation {6.1} and {6.2}. Calibration of the SWAT model is carried out by comparing modelled to measured long-term yearly average sediment loads and by comparing simulated and observed daily sediment load dynamics. Adjusted model parameters are slope lengths, support practice factor and widths of vegetated buffer strips.

The resulting total sediment time series are handed over to the HEC-RAS model using a SWAT-HEC-RAS interface (Kiesel et al., 2012). Sediment grain size distributions of the SWAT time series are calculated from topsoil parameters, which are weighted according to the areal soil type distribution within each subbasin. Substrate grain size distribution of the river bed and banks complete the input data. Subbasins and channels upstream of Lake Winderatt are excluded from the sediment calculations since the lake acts as a sediment sink. The HEC-RAS model is calibrated to the measured total sediment load by fitting the most suitable sediment transport equation, sorting, armouring and fall velocity methods and by adjusting the density of sediments. Six sediment transport equations are implemented in the HEC-RAS model: Ackers and White, Copeland's form of Laursen, Meyer-Peter-Müller, Toffaleti, Yang, Wilcock; (USACE, 2010b). The user has to choose the most appropriate one during the calibration process. The Toffaleti formula (Toffaleti, 1968) is the most suitable for calculating the sediment transport potential for the local conditions. This total load function has been developed for conditions with a significant amount of suspended load and sand transport (USACE, 2010b) which are both typical for the sediment transport in the Kielstau River (Labadi, 2009). The decision is furthermore supported by Yang and Wan (1991), who found good performance and accuracy of the formula in natural rivers compared to other formulas. Sediment transport calculations are carried out with the Exner 5 sorting routine and Toffaleti fall velocity method.

Sediment and water fluxes time series with a daily time step are transferred to the AdH model at the respective cross section that defines the upstream AdH model domain boundary. Besides the influx time series, sediment input data necessary for the AdH model is comprised of properties for distinct grain size classes and bed layers. These are taken from a morphological mapping campaign where besides the movable sediment, wood debris, vegetation and boulders have been recorded (Thiemann, 2008). Eighteen evenly distributed substrate samples have been taken in the stream section from which grain sizes have been analysed in the laboratory. The target parameter for the calibration is the change of the d90 for the upper 5 cm of the river bed between April 2008 and April 2009 for which observed substrate changes are available (Table 6.4). Similar to the HEC-RAS model, the model AdH is calibrated by fitting the most suitable sediment transport equations out of three suspended load formulas (Garcia-Parker, Wright-Parker and VanRijn) and out of three bed load formulas (VanRijn, Meyer-Peter-Müller, Meyer-Peter-Müller with Wong-Parker correction) and by adjusting the density of sediments. The sediment transport equations that yielded the best results for the application in the Kielstau River are Wright-Parker (Wright and Parker, 2004) for the suspended entrainment and Meyer-Peter Mueller with Wong-Parker correction (Wong and Parker, 2006) for the bedload entrainment.

6.3 Results

6.3.1 Results - water fluxes

In order to assess model performance on the three scales, simulation results are compared to temporally and spatially distributed measurements. The SWAT model shows good agreement with observed daily discharge at the catchment outlet for the five year calibration and five year validation period (r^2 = 0.82, NSE = 0.78, Figure 6.2). Most problematic is the depiction of peak flows (Figure 6.2) which is most likely due to first, the availability of only one climate station 5 km outside the watershed and second, the availability of daily aggregated precipitation data, which is too coarse, since the time of concentration of the Kielstau, calculated using the Kirpich (1940) formula, is about 7 h. Additional results and final calibrated parameters can be found in Kiesel et al. (2010a).

The HEC-RAS model, driven with flow data from SWAT, generally matches measured water depths (r^2 = 0.90, Figure 6.3a) and flow velocities (r^2 = 0.88, Figure 6.3b) at the three locations well. The scatter plots enable a direct comparison between measured and modelled hydraulic parameters for different discharge events. At site B, located in the middle section of the stream, the model underestimates highest measured water depths at three occasions while flow velocities are overestimated. At site A, both velocity and depth are underestimated for the highest depth and flow event, which is probably due to a measurement error.

The two-dimensionality of the AdH simulations makes a spatially distributed comparison in x- and y-direction necessary, which is especially important for the calibration process. In addition, by comparing simulated with observed hydraulic parameters along and across the stream section, strengths or weaknesses in the sediment transport simulations can be explained. The AdH model results at the eight cross sections match measured water depths very

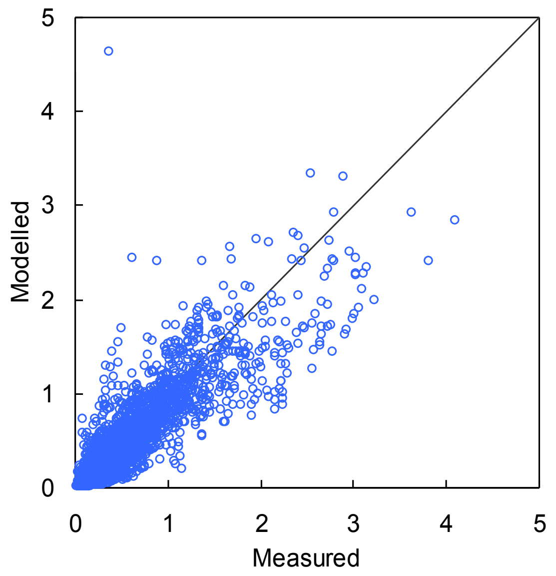

Figure 6.2 Comparison of daily calculated and measured discharges (SWAT) for the modelling period 1999-2009

well (Figure 6.4). At some locations, the model underestimates water depths which are predominantly close to the river banks (a, c, f, g, h). Flow velocity distributions are simulated sufficiently well but are depicted less accurate than water depth. The locations at the banks show highest deviations and the return flow, as observed at cross sections d and g, could not be simulated. The three models are fit as thoroughly as possible to measured data, as good model performance in hydrology and hydraulics is paramount for realistically depicting erosion, sediment transport and deposition.

6.3.2 Results - sediment fluxes

Agreement of modelled with observed sediment load leaving the watershed is an important indicator for the plausibility of the SWAT and HEC-RAS models. At the catchment outlet, modelled and measured sediment load is compared. The modelled sediment pathways are distinguished in Figure 6.5 where the daily distribution of the field, drain and total sediment load is plotted against measured values (r^2 = 0.56, NSE = 0.26). Modelled sediment load leav-

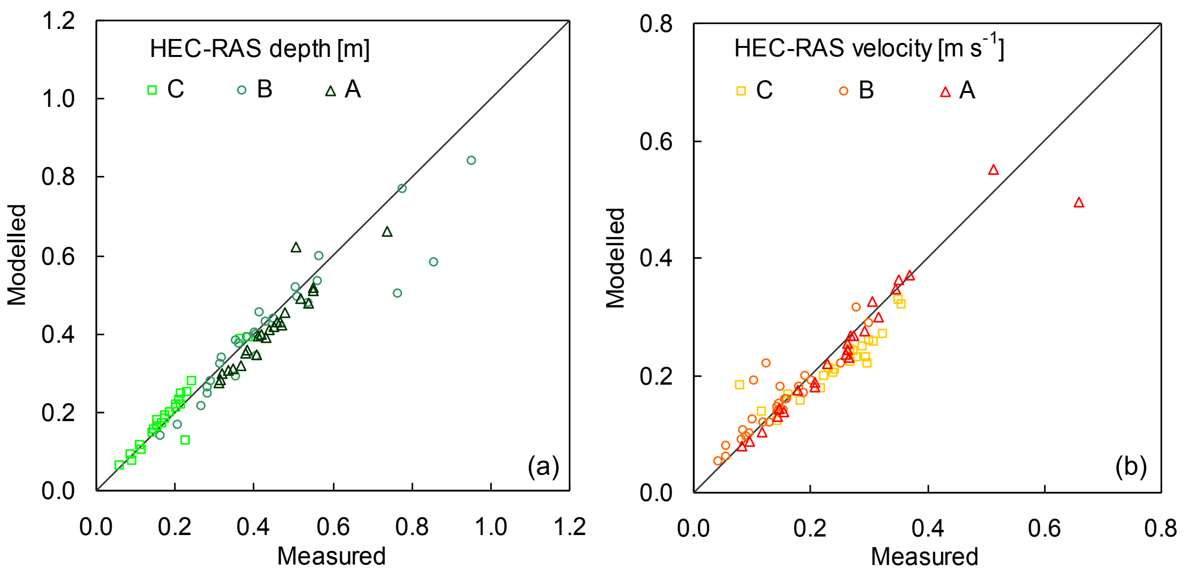

Figure 6.3 Comparison of calculated and measured hydraulic parameters (1D) at locations A, B, C (see Figure 6.1b): (a) flow depth and (b) flow velocity

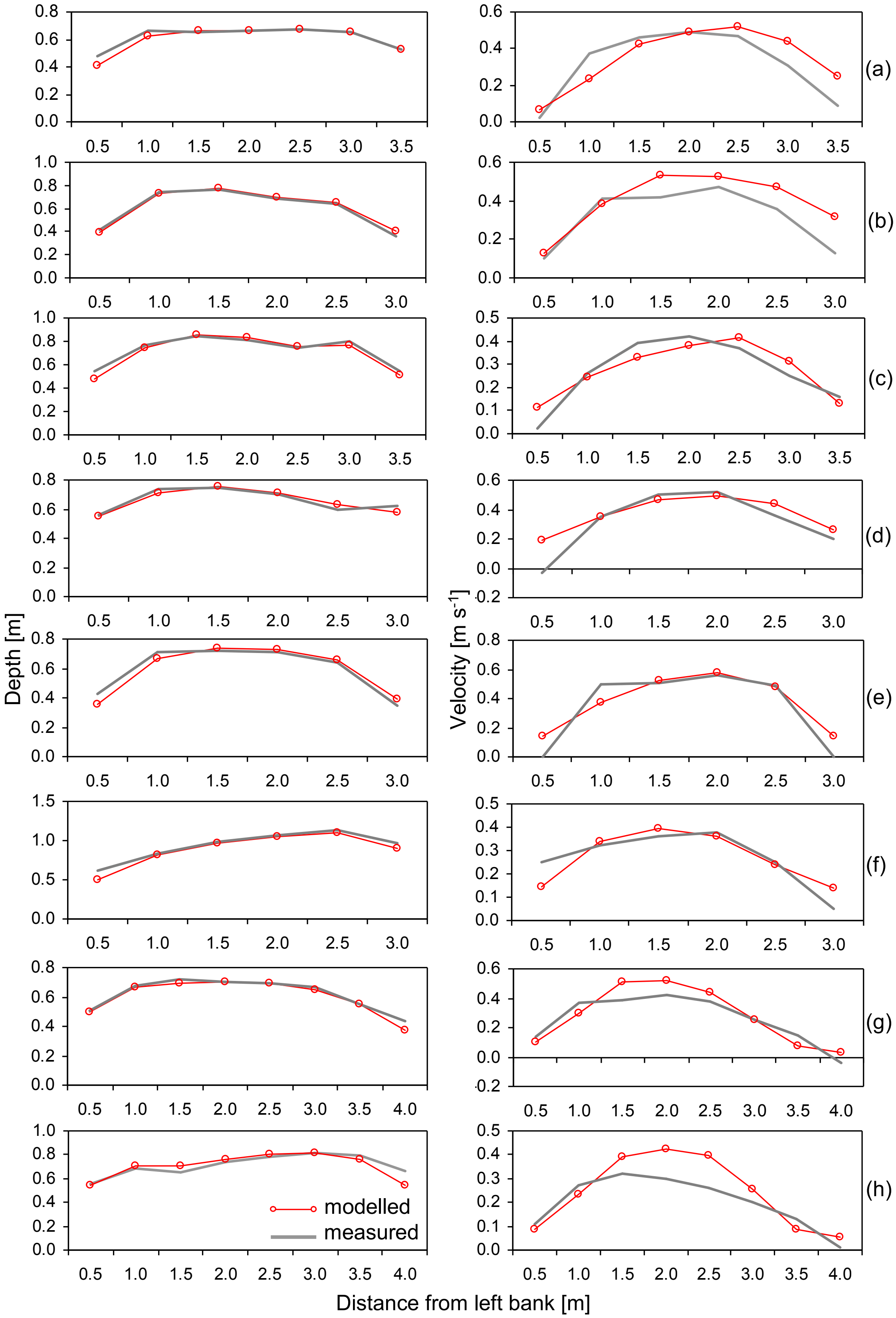

Figure 6.4 Comparison of calculated and measured hydraulic parameters (2D) at cross sections a-h (see Figure 6.1c): (left) flow depth; (right) flow velocity

ing the fields, drains and the channel bed and banks account for 1.6 %, 18.0 % and 80.4 % respectively during the target time period of April 2008 to April 2009. The modelled ratios between the three pathways are governed by the flow components surface runoff, tile flow and stream flow and thus represent the hydrologic situation during the April 2008 to April 2009 time period: The ten most extreme storm events in this period are 40 % lower than in other years, a situation which seems to have caused unusually low field erosion. Calculating the sediment transport ratios over three years, from 2007 to 2009, yields a ratio of 17.1 % from fields, 18.3 % from drains and 64.5 % from the channel, which is comparable to other studies in the same or similar environments (Kiesel et al., 2009b; Kronvang et al., 1997). In comparison to field and drains, the channel banks and bed are more constantly contributing sediment. The model depicts the pattern and magnitude reasonably well, but the single highest measured sediment load during the modelling period (February 2009, 8.4 t d^{-1}) could not be reproduced. It is unclear if the discrepancy refers to a measurement error or a physical explanation like a bank collapse. Measured values have some gaps due to malfunctioning of the automatic sampler.

The combined SWAT and HEC-RAS model supplies flow and sediment time series to the AdH model. Validation of the AdH model is carried out by comparing simulated with mapped d90 of the upper substrate (Figure 6.6) within the modelled reach section. Non-mobile substrates (large wood debris, stones and water plants) have been mapped in the field and are superimposed on the substrate maps. Over the course of the year, most sand fractions have been eroded and transported out of the study reach. In most areas, the model can depict this situation well. In the north-western bend, the model overestimates the d90, which is most likely due to too high flow velocities in the center of the channel (cross section h, Figure 6.4). Further upstream (north) lower simulated d90 values are present at the left bank while this is vice versa on the recorded substrate map. The narrow, long streak of sand that formed can not be reproduced by the model. The southern, steep curve is simulated well, while further downstream in the straight section, the model AdH overestimates d90 at the banks. At that location, the AdH model also already overestimated flow velocities at the banks (cross sections d and e, Figure 6.4).

The strength of the model cascade is not only the distinction between different sediment pathways on the temporal scale, but also the spatial distribution of sediment loss and change.

Figure 6.7 shows the spatial distribution of field, drain and channel sediment origin, simulated with SWAT and HEC-RAS models from April 2008 to April 2009. The spatial dis-

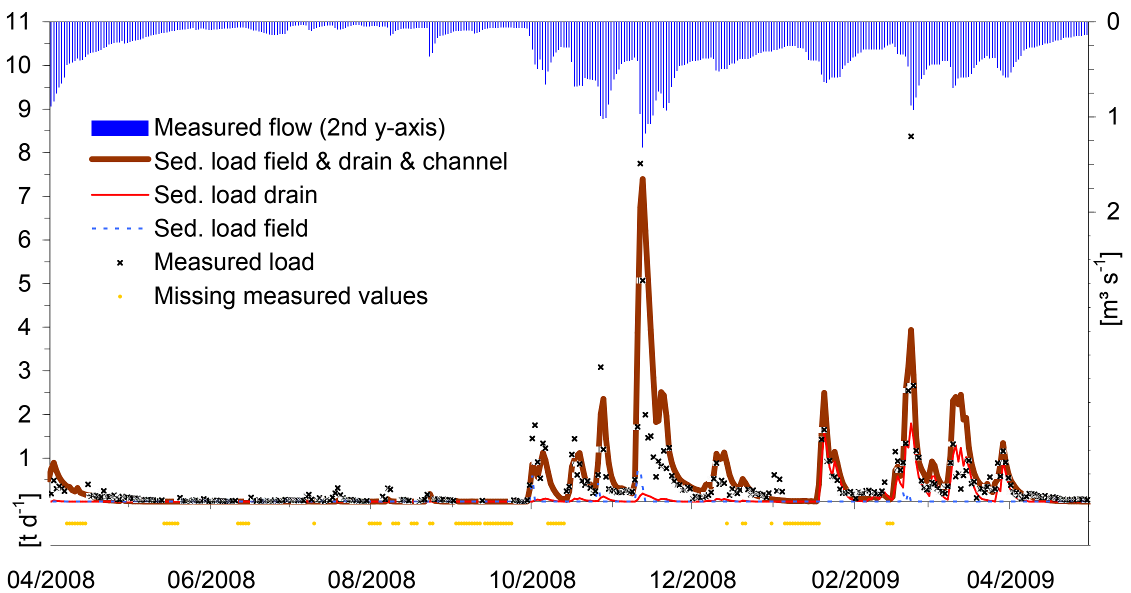

Figure 6.5 Modelled sediment load from field simulated with SWAT, tile drain depicted with SEPAL (Kiesel et al., 2009b) and equations {6.1} and {6.2}, modelled sediment load from field and drain and channel in combination with HEC-RAS; all compared to measured values; note missing measured values below x-axis

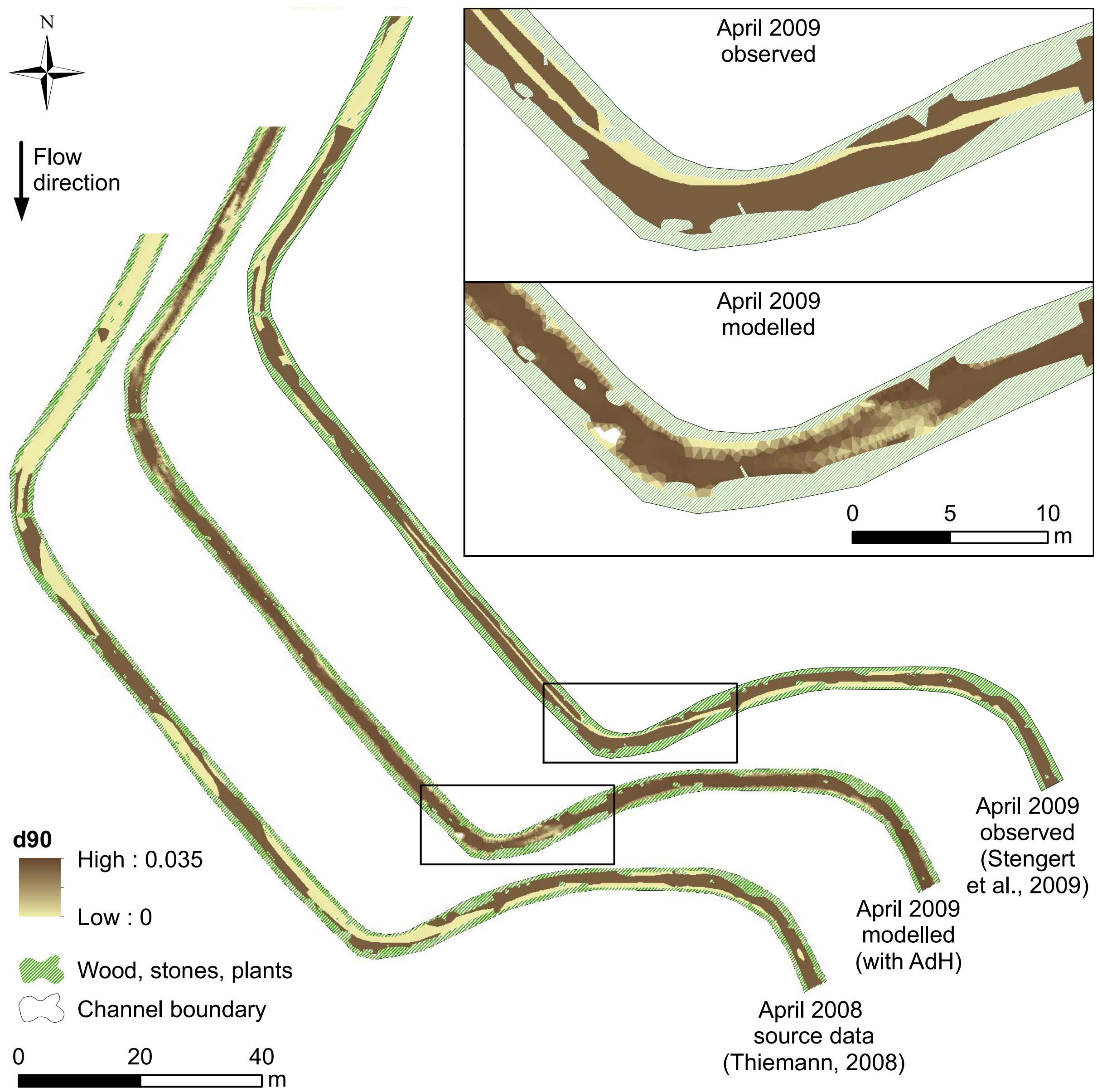

Figure 6.6 d90 of the upper 5 cm of the river bed, April 2008 source data and comparison of modelled AdH results with observed, both April 2009; insets showing the bend in detail. d90 of the upper 5 cm of the river bed, April 2008 source data and comparison of modelled AdH results with observed, both April 2009; insets showing the bend in detail

tribution of tile drains, evaluated by Fohrer et al. (2007), are shown as hatched areas in Figure 6.7. At these locations tile drain sediment is generated. The transported sediment enters the stream at defined locations where the tile drain pipes join the river network. Field erosion input into the stream occurs at more erratic locations depending on the overland flowpaths

These detailed spatial input patterns are lumped over each SWAT subbasin. The shaded area in the eastern part of the catchment feeds into Lake Winderatt that acts as a sediment trap.

Erosion and sedimentation modelled with the HEC-RAS model is shown on the map through lines with alternating thickness within the main stream channel (white), and for a better overview in a separate longitudinal channel change profile. According to field investigations, the spatial channel erosion is plausible. For example, the highest modelled erosive location coincides with a spot where farmers had to move their fences due to the channel bank retreat.

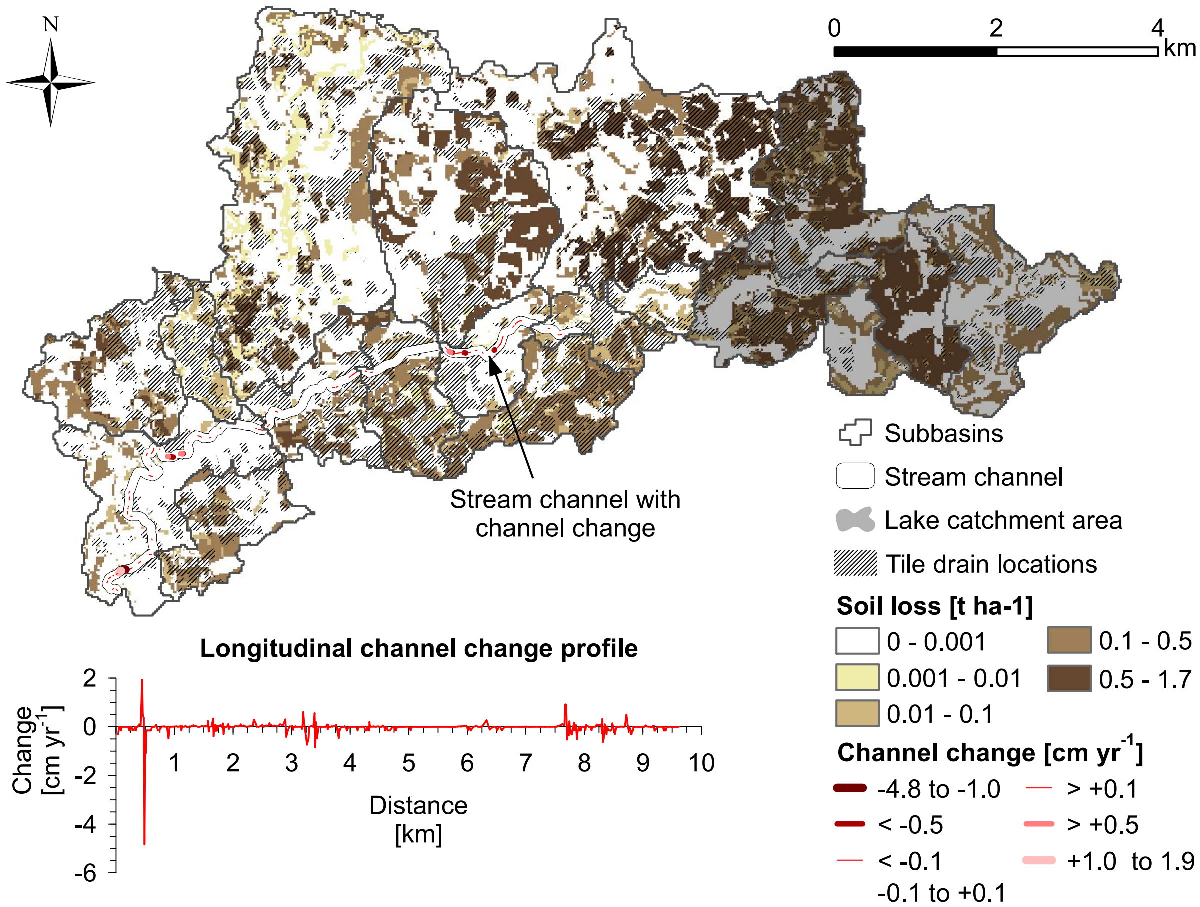

Figure 6.7 SWAT and HEC-RAS spatial erosion results: Spatially distributed erosion from fields and drains modelled with SWAT and channel change modelled with HEC-RAS including a longitudinal channel change profile, average values over the April 2008–April 2009 time period

Figure 6.8 displays the spatial distribution and temporal change of d90 over the course of the observation year modelled with AdH. The stream bed is relatively stable during most months of the year, for which results are omitted. Major erosive events occur in the winter months November, December and January which coincides with the highest discharge events (Figure 6.5).

6.4 Discussion and conclusion

The first objective of this study was the realistic depiction of water fluxes as a solid basis for erosion and sediment transport simulations. The hydrological model SWAT, applied on the catchment scale, is fitted well to the ten year simulation period. The hydraulic models HEC-RAS and AdH both show good agreement of modelled to measured water depths and flow velocities along the modelled stream channel sections. The HEC-RAS simulation has weaknesses in the depiction of highest flows. This is potentially caused by difficult measure ments of hydraulic parameters during those events, which thus contain higher uncertainties. The steep slope of the banks combined with difficulties in referencing the location of the measurements within a few centimetres accuracy cause less accurate simulations with AdH. This is especially visible for the return flow cross sections, where in addition, more detailed bathymetry data should have been available upstream and downstream of the cross section. In summary, the quality of the depiction of water fluxes, according to statistical measures re-searched by Moriasi et al. (2007), is sufficient for simulating erosion and sediment transport processes.

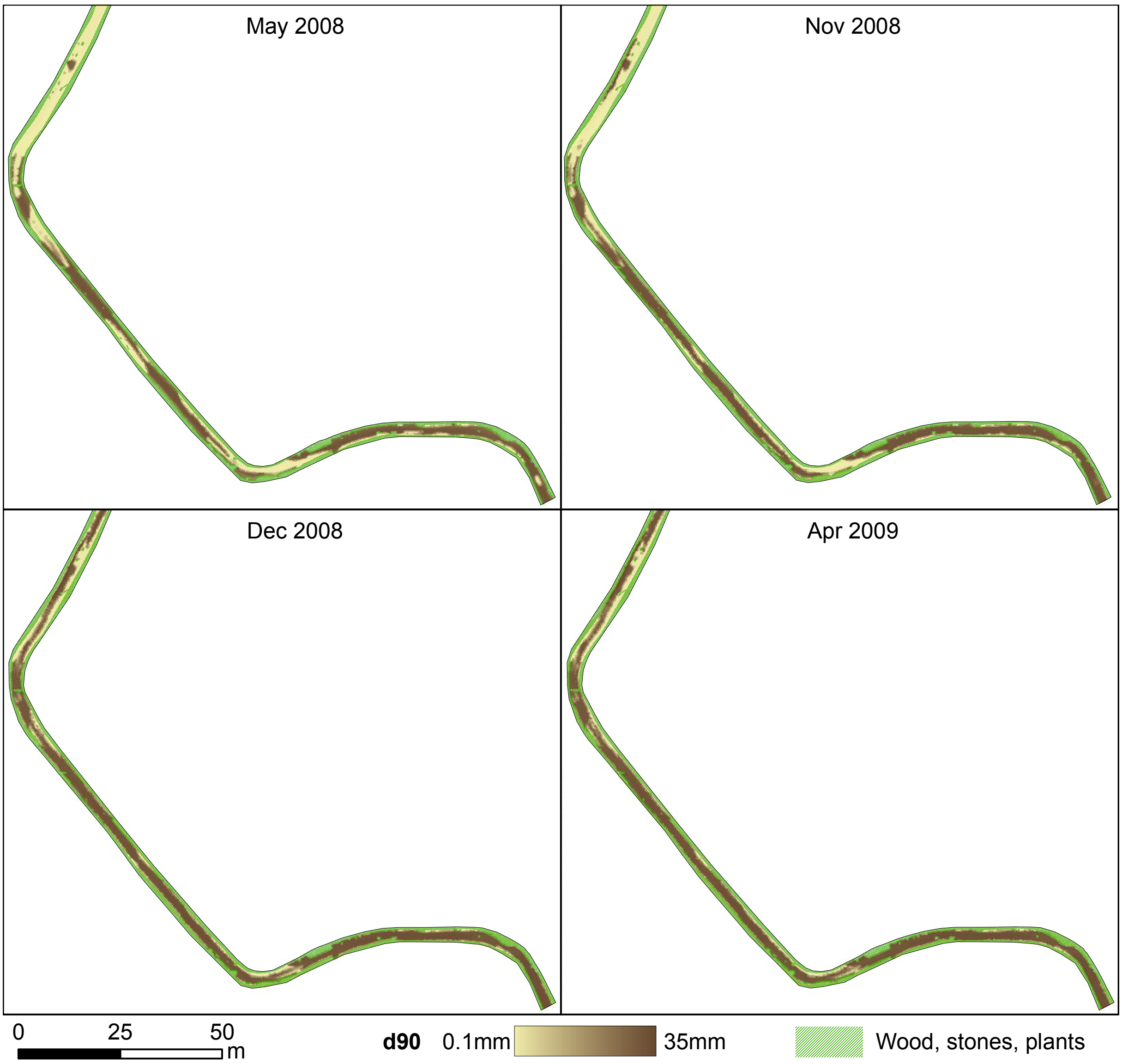

Figure 6.8 AdH spatial erosion results: Spatial distribution and temporal change of d90 over the course of the observation year modelled with AdH, omitted months have a stable river bed where changes would be only marginally visible.

The second objective was the simulation of the three main sediment entry pathways, field-, tile drain-, and channel sediment input in lowlands. The combined SWAT and HEC-RAS sediment simulation has successfully been adapted to the measured sediment load, yielding plausible results for the sediment entry pathway modelling.

The main objective of this study was the application of a three-step modelling cascade that is capable of considering impacts of environmental changes on any scale on water and sediment fluxes on the catchment-, channel-, and reach scale. For achieving good simulation results it is necessary to supply comprehensive input data to the model cascade: information is required about the physical environment, from climate, land use and management, to instream characteristics like channel bathymetry, substrate, vegetation, boulders and wood debris. These data requirements and the presented application show that the model cascade is potentially capable to depict global environmental changes as well as anthropogenic stream alterations.

Further research and improvements are suggested in the following points:

(1) the depiction of tile drain sediment load within this study is dependent on empirical relations and is linearly correlated to tile flow. A physically based model of sediment transport in drain flow is still lacking.

(2) The AdH model, driven with data from the combined SWAT and HEC-RAS simulation, is successfully adapted to observed one-year substrate changes. It would be desirable to generate an even better bathymetric database. Also, the changes are a coarsening of the d90 in most areas of the 230 m long stream segment. Additional validation of the model cascade for sedimentation events would thus be useful.

(3) A sensitivity test of the input data which is consecutively passed through the total model cascade is desirable. Schmalz et al. (2012b) have shown sensitivity evaluations of the SWAT and HEC-RAS combination. A more comprehensive sensitivity evaluation can potentially be deduced from the IWRM-NET project IMPACT (Guse and Fohrer, 2011; Kail and Wolter, 2012), which currently works towards that objective.

The main advantages of the presented model cascade which can be derived from this study are the following:

(1) the technological status of the individual models is good and will likely remain as such because they are constantly improved and developed. In addition, data preparation and results visualisation as well as data transfer methodologies can be achieved in the flexible GIS environment.

(2) The detached representation and results visualisation of interdependent processes on variable temporal and spatial resolutions is useful. Nested approaches for instream erosion, as described by Piégay et al. (2005), can be depicted by the shown methodology. Also, the separate output of sediment input from field, drains and the river can be utilised for assessing sediment pathways. Depicting these three pathways is especially beneficial in agriculturally used lowland areas since the temporal distinction on a daily time step and the spatially distributed sediment map of the catchment both enable a more detailed investigation and management of sediment input. For the correct assessment of the impact of potential best management practices and their implementation, this detailed analysis of sediment sources is indispensable.

(3) The complementation of one's model's weakness through the previous or next model in the series is valuable: the SWAT model can be applied on very flexible spatial resolutions in the catchment, but high spatial instream resolutions and hydraulic parameters have to be depicted with a hydraulic model. The HEC-RAS model has proven to cover the stream and multiple hydraulic processes well in case hydrological and sediment time series are supplied at all tributaries. However, results are too coarse for micro-scale substrate assessments which are necessary in habitat related studies. The AdH model made it possible to simulate these processes successfully on seamless surfaces in flexible resolutions. Through the automatic mesh- and time step refinement, the model is stable and user friendly, but the complex flow and sediment transport calculations demand excessive computer power, especially when, as shown, running long-term sediment transport simulations.

(4) Applying the modelling system in different catchments and environments is possible: As shown, the models could be adapted to hydrologic lowland characteristics such as drainages and landscape depressions as well as specific hydraulic conditions of the small, low gradient stream. Beyond that, parts of the presented methodology have recently successfully been utilised in the Kinzig, a meso-scale catchment in Germany's low mountain range (Schmalz et al., 2012a).

(5) The comprehensive consideration of climate, natural and anthropogenic changes, as well as catchment and stream properties makes the model cascade an ideal tool for habitat assessments. The developed methodology was successfully applied by Jähnig et al. (2012) and Schmalz et al. (2012b).

6.5 Acknowledgement

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Chapter 7 Modelling of riverine ecosystems by integrating models: conceptual approach, a case study and research agenda

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Abstract:
Aim: Highly complex interactions between the hydrosphere and biosphere, as well as multi-factorial relationships, characterise the interconnecting role of streams and rivers between different elements of a landscape. Applying SDMs in these ecosystems requires special attention because rivers are linear systems and their abiotic and biotic conditions are structured in a linear fashion with significant influences from upstream/downstream or lateral influences from adjacent areas. Our aim is the development of a modelling framework for benthic invertebrates in riverine ecosystems and to test our approach in a data-rich study catchment.
Location: We provide a local case study of a 9-km section of the lowland Kielstau River located in northern Germany.
Methods: We linked a hydrologic, a hydraulic, and SDMs to predict the habitat suitability of the bivalve *Sphaerium corneum* in a riverine system. The results generated by the hydrological model served as inputs into the hydraulic model, which was used to simulate the resulting water levels, velocities and sediment discharge within the stream channel.

Results: The ensemble model obtained good evaluation values (AUC: 0.96, Kappa: 0.86; TSS: 0.95). Sensitivity (86.14) and specificity (85.75) measures also indicate a good allocation of true positive and true negative predictions. Mean values for variables at the sampling sites are not significantly different from the values at the predicted distribution (Mann-Whitney U-test $P > 0.05$). High occurrence probabilities are predicted in the downstream half of the 9-km section of the Kielstau. The most important variable for the model was sediment discharge, contributing 40 %, followed by water depth (30 %), flow velocity (19 %), and stream power (11 %).
Main conclusions: The hydrologic and hydraulic models are able to produce predictors, acting at different spatial scales, which are known to influence riverine organisms, which, in turn, are used by the SDMs as input. Our case study yielded good results corresponding to principal ecological knowledge of the studied clam. Although this method is feasible for making projections of habitat suitability on a local scale (here: a reach in a small catchment), several challenges remain for future modelling approaches and large-scale application.

Keywords: Benthic invertebrates; BIOMOD; Germany; HEC-RAS; hydraulics; hydrology; Kielstau; species distribution modelling; streams; SWAT

7.1 Introduction

Freshwater ecosystems, particularly rivers, are under severe pressure from multiple sources. Most rivers are in a state of progressive deterioration due to anthropogenic pollution, bank fixation, disengagement of floodplains, or alterations in hydrology, resulting in severe loss of aquatic and riparian biodiversity. Additionally, they are among those ecosystems most severely affected by climate change (Vörösmarty et al., 2010). To avert further decline in the health of aquatic ecosystems, measures for sustainable use should be implemented. Such measures could be based on integrated models that deliver a sound understanding of ecosystem functions, their interactions, and feedback mechanisms across different spatial and temporal scales. However, the highly complex interactions between the hydrosphere and biosphere, as well as multi-factorial relationships, are a challenge to represent in models, with first attempts focusing on the terrestrial phase (Weber et al., 2001; Fohrer et al., 2002); on pollutants connected to agricultural activities (Pohlert et al., 2005; Lam et al., 2010); or the transport of pesticides (Holvoet et al., 2007; Dietrich et al., 2011). Ecological models predict, for example, the occurrence of aquatic organisms in relation to land use or anthropogenic stressors, provide approaches to assess the effects of spatial processes across various scales or take into account management options (Statzner and Borchardt, 1994; Harby et al., 2004; Adriaenssens et al., 2007; Goethals et al., 2007). Biotic aspects are included less often in integrated modelling studies, but see Statzner and Borchardt (1994), Dedecker et al. (2004), or Holguin and Goethals (2010).

These models are often set for particular river systems or river segments (Bovee et al., 1998), but for the evaluation of the impacts of climate and/or land use changes on aquatic ecosystems at larger scales, there is still a lack of models that are capable of fully describing links within the environment and between the environment and the organisms within it (Kiesel et al., 2009a). SDMs are useful for predicting ecological responses to changing environmental conditions that can be applied to any scale, provided that suitable predictors are available (Elith and Leathwick, 2009). They are more commonly applied to terrestrial organisms and have proven to be valuable tools in the context of vegetation ecology and conservation management. In streams, large-scale and predictive modelling, as applied in climate change impact studies is limited (e.g. Castella et al. 2001; Statzner et al., 2008), but recently modelling studies have embraced extensive regions of riverine environments, especially for fish and invertebrates (Domínguez-Domínguez et al., 2006; Buisson et al., 2008; Depraz et al., 2008; Cordellier and Pfenninger, 2009; Mouton et al., 2010; Balint et al., 2011; Domisch et al., 2011).

We consider benthic invertebrates to be ideal as a study group; they live on and within the substrate of the river bottom (the benthos), and comprise numerous groups such as crustaceans, molluscs, worms, turbellaria and insects. The occurrence of benthic communities is dependent on the characteristics of the catchment and on the suitable aquatic habitats available at the section or site scale (Molnar et al., 2002; Kiesel et al., 2010b). Relevant catchment parameters include seasonal discharge patterns, flood frequency, elevation, geology, or land use (Vinson and Hawkins, 1998; Kiesel et al., 2009a). Hydromorphological conditions are the controlling factors on a reach scale, including stream width, substrate roughness or riparian land use, longitudinal (along the upstream–downstream axis of the river, e.g. blocking by weirs or dams), lateral (characteristics of the riverbanks, the extent of the functioning floodplain and riparian habitats), and vertical continuity (connection to the hyporheic zone and the groundwater) (Brosse et al., 2003; Arscott et al., 2005; Boulton, 2007). On the site scale, relevant habitat parameters include shear stress, water depth, substrate, sediment stability, shading, and physicochemical water parameters (Allen and Vaughn, 2010). Riverine ecosystems and their benthic invertebrate communities are thus shaped by a wide variety of processes and conditions, which render them very heterogeneous, even on a local scale.

Our general aim was the development of a suitable integrated modelling framework for benthic invertebrates taking this complexity into account. As mentioned, there are other modelling approaches available, but our integrated method differs in that it allows for full control of the models' design and linkage, especially related to hydrological and hydraulic modelling in ungauged catchments (Caspar et al., 2011); performs sensitivity analysis for the separate models, pursues an ensemble approach to account for different model outcomes and uncertainty; and most importantly is capable of upscaling in space and time. Although not all these advantages have been implemented so far, we can present a case study of our approach in a data-rich study catchment.

7.2 Materials and methods

7.2.1 General approach

The integrating modelling technique developed uses different models to provide the environmental data, and to describe the relationship between organisms and the environment. The model approach considers the hierarchy of environmental variables at different scales in river ecosystems. It links the catchment to instream processes and then to the biota by following the DPSI framework (Figure 7.1). The modelling system can potentially analyse changes of climate, land use, and river morphology and their effects on the hydrosphere, instream processes, and aquatic habitats down to ecosystem responses. It facilitates evaluation of both landscape and instream measures aimed to improve aquatic habitats.

The model system consists of the ecohydrological SWAT model in version 2005 (Arnold et al., 1998), the 1D hydraulic model HEC-RAS version 4.1.0 (USACE, 2010a), and SDMs

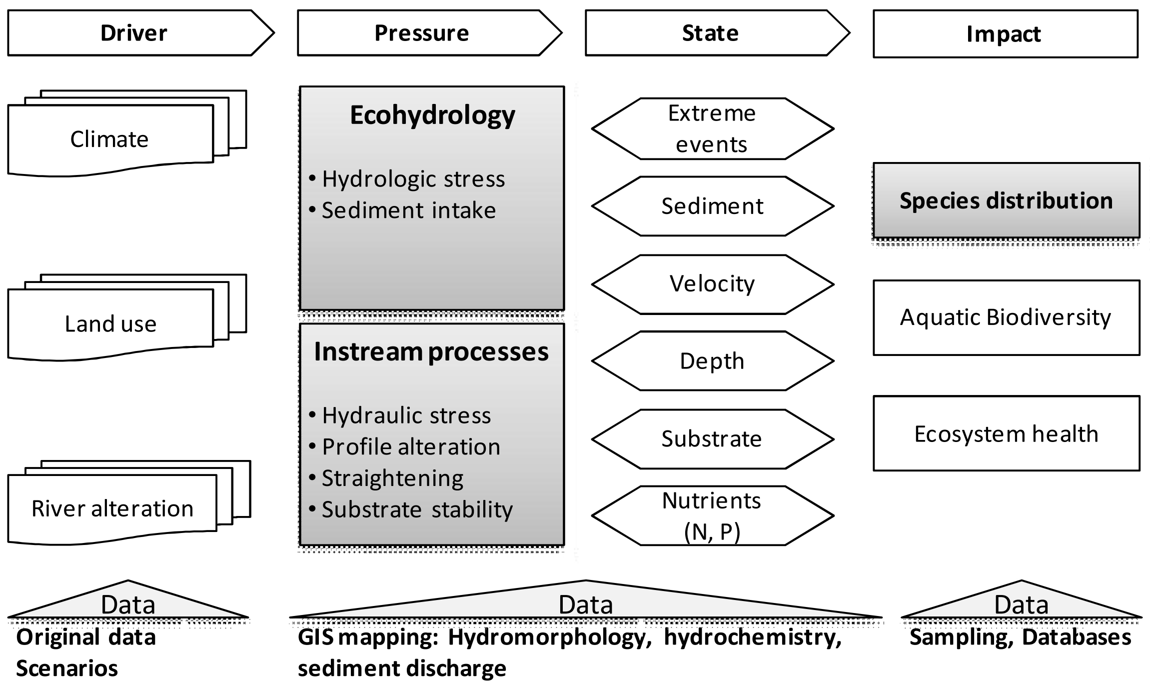

Figure 7.1 Integrated approach to model aquatic ecosystems following the DPSI concept. (1) Major drivers as the model input data are depicted by jointly considering stream and catchment processes. (2) The main pressures on the aquatic ecosystem are defined and represented in the model algorithms. (3) Based on the multiple pressures, it is possible to dynamically assess the changes of the states of habitat parameters in the model output. (4) Finally, the impacts of the states on the aquatic ecosystems can be evaluated, closing the complex cause-effect chain from the drivers to the impact

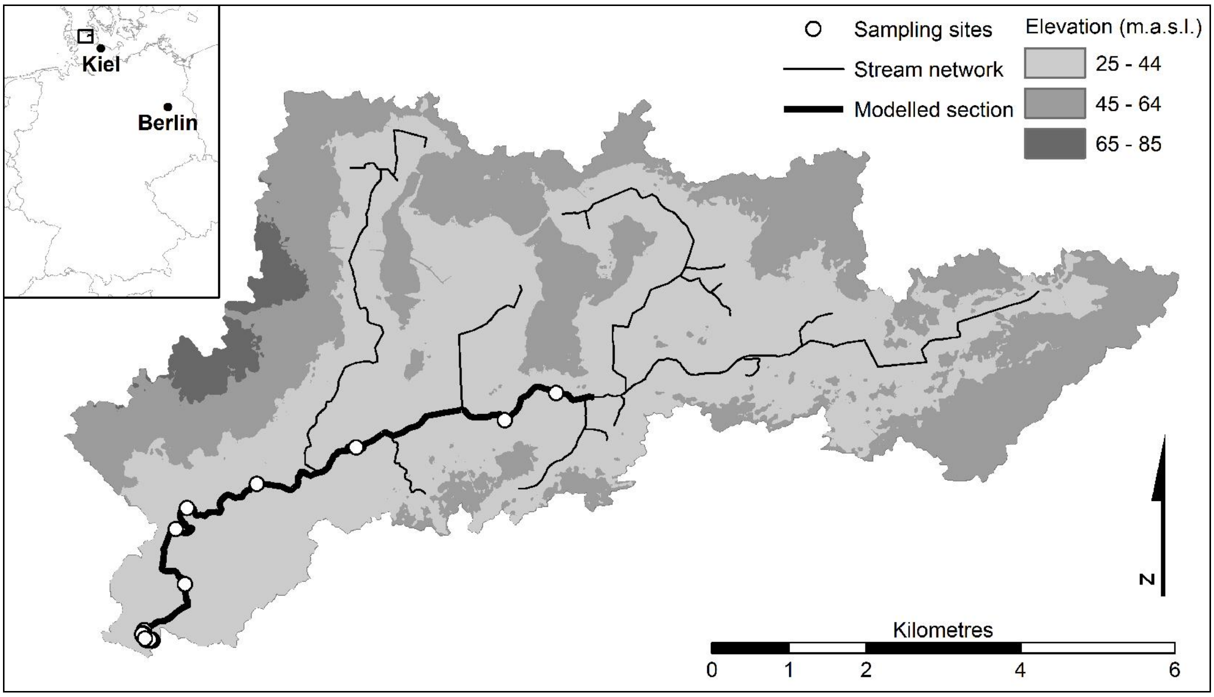

Figure 7.2 Kielstau catchment in northern Germany, with the modelled stream section (map according to LVA (2008))

as provided in the package BIOMOD 1.1-6.9 in R (Thuiller et al., 2009; R Development Core Team, 2011).

Integrated model environments according to this methodology are currently being developed in three German catchments, namely the Kielstau (50 km²; Kiesel et al., 2009a; 2010b), the upper Treene (530 km^2; Guse and Fohrer, 2011), and the Kinzig (a site of the long-term ecological research network (LTER), 1,500 km^2; Schmalz et al. (2012a)), and in the southern Chinese catchment of the Changjiang (1,700 km^2; Kuemmerlen et al., 2012; Schmalz et al., 2012b), each covering different key aspects. Further advancement is planned by realising a hydrology-based model system with European spatial coverage, based on the WaterGAP Global Hydrology Model (WGHM) by Döll et al. (2009). Of these studies, the Kielstau catchment, which serves as a UNESCO demonstration site for ecohydrology, has a very good database and is most advanced in terms of model integration (Schmalz and Fohrer, 2010). It is thus presented below as a case study, exemplarily predicting a suitable habitat area for the bivalve *Sphaerium corneum* (Linnaeus, 1758), the European fingernail clam.

The Kielstau subcatchment has an approximate area of 50 km² and is located in the northern German lowlands (Figure 7.2). The Kielstau Stream is one of the headstreams of the Treene River, which is part of the Eider catchment. The integrated modelling approach was applied to the 9-km section of the Kielstau Stream, downstream of Lake Winderatt.

7.2.2 Models and integration steps

Hydrological models

Abiotic environmental properties on the catchment scale are known to affect riverine communities (Quinn and Hickey, 1990). Hydrologic models use these properties as input data to simulate the hydrological cycle and can, for example, depict runoff from a watershed, calculate the nutrient loads (Horn et al., 2004; Hörmann et al., 2005), or predict droughts or floods. They are based on equations describing the hydrological cycle both in space and time and can thus give a detailed description of the hydrological processes in the catchment. Furthermore, they are used for evaluation, planning and simulating the implementation of management measures, such as the improvement water quality at the watershed level (Lam et al., 2010, 2011). The effects of climate or land use change on the watershed responses can be predicted; thus, they are useful for environmental impact assessment studies (Fohrer et al., 2005) or for integrated water management (Singh and Woolhiser, 2002).

The joint application of hydraulic and biological models requires hydrological information on specific locations, such as stream sections or species occurrence points; for this purpose, a (semi-)distributed, physically based hydrological model is required. In such a distributed model, the spatial variation of input parameters and variables is considered, and the watershed is divided into spatially distinct areas of similar physical conditions.

SWAT is a physically based, semi-distributed model and has been proven to produce reliable results in various studies for integrated water management and has gained international acceptance as a robust interdisciplinary watershed-modelling tool (Arnold and Fohrer, 2005; Gassmann et al., 2007; Kiesel et al., 2010a; Lam et al., 2011). It can simulate water balance, nutrients and pesticides, erosion, plant growth cycles, management practices, and water bodies on a daily time step for continuous simulations over long time periods using spatially distributed data on GIS maps, climate data, and physical information from a relational database. Inputs include spatial information, such as topography, soil, and land use data; additionally, management inputs include crop rotations, tillage operations, planting and harvest dates, irrigation, fertiliser use, and pesticide application rates. Climatic variables are required for simulating water flow, sediment transport, crop growth, and nutrient cycling (see Neitsch et al., 2005a for details). It links the advantages of being an integrated model (e.g. describing the water balance and water-coupled fluxes of matter) and being applicable in a wide spatial range (i.e. from small to very large watersheds).

The first step in the integration process is to obtain water and sediment fluxes for the Kielstau catchment from the SWAT model (Figure. 7.3). Evaporation is simulated with the Penman–Monteith equation, surface runoff with the SCS-CN method, interflow with a kinematic storage model, and baseflow is calculated through the water balance of two groundwater aquifers. Channel flow values are obtained by routing the received water with a variable storage coefficient method. The MUSLE (Williams, 1995) is utilised to simulate field erosion. ArcSWAT (Winchell et al., 2007) is used to prepare the input files from land use (DLR, 1995), soil (BGR, 1999; LANU, 2006b), topographic (LVA, 1992–2004), and climate (DWD, 2010) data in ArcGIS 9.2 (www.esri.com). The model setup, application, and performance are explained in detail in Kiesel et al. (2010a).

Hydraulic models

At a reach- to site scale, the organisms are affected by instream qualities such as flow velocity, depth, or substrate size and type (Vinson and Hawkins, 1998), thus hydraulic models are required to describe these parameters. Hydraulic models combine the morphological conditions of the river reach with discharge ranges into a set of hydraulic parameters that are of major importance to the physical appearance of the aquatic habitat (Steuer et al., 2008). Furthermore, fine sediment delivery to, and storage in, stream channel reaches can be considered as it may disrupt aquatic habitats, impact river hydromorphology, and transfer adsorbed nutrients and pollutants from catchment slopes to the fluvial system (Jarritt and Lawrence, 2007). Models for simulating open channel flows can depict these variables both temporally and spatially. In general, one-dimensional (depth and width averaged) or two-dimensional (depth-averaged) simulation codes are applied in aquatic habitat modelling (Harby et al., 2004). Besides flow velocity, depth, or sediment discharge, the state-of-the-art hydraulic modelling systems describe substrate conditions (USACE, 2010a; Berger et al., 2011), which are important factors for species occurrence (e.g. Hauer et al., 2011). However, applications are rare where substrate properties are simulated continuously for years: the reasons for this are the difficult validation, substantial input data requirements and high computational demand.

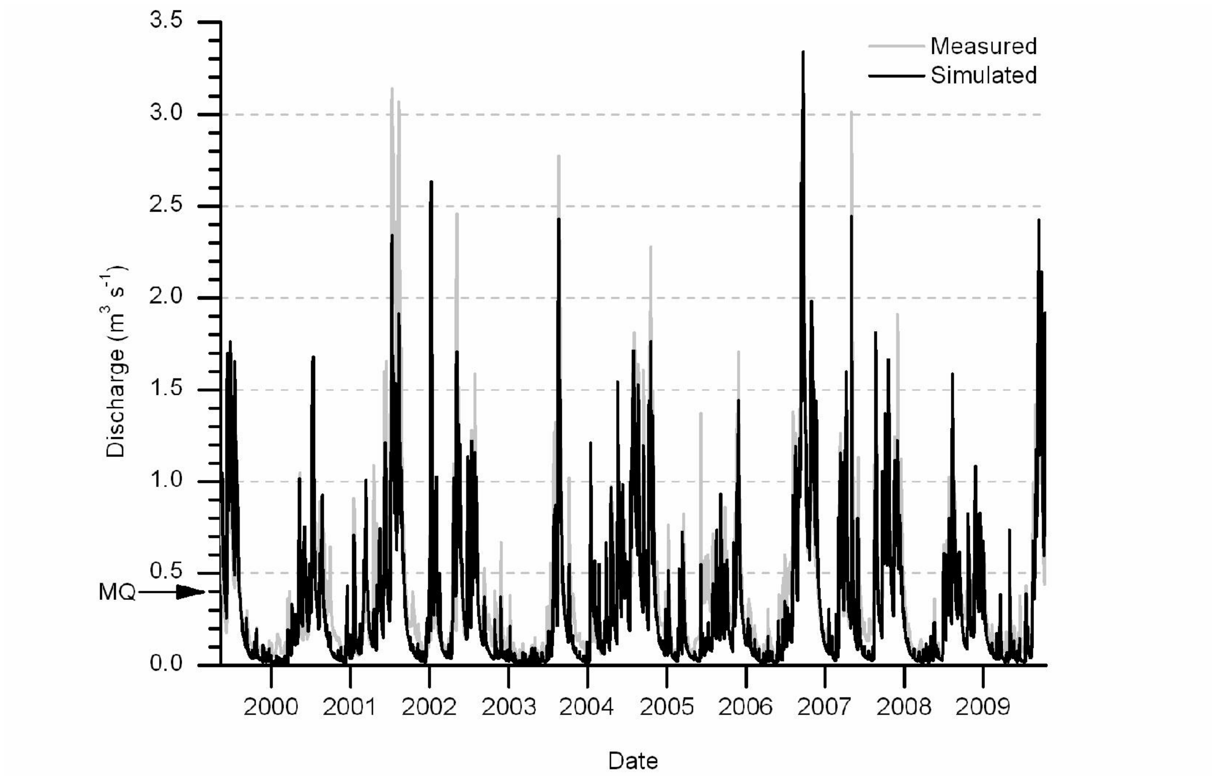

Figure 7.3 Measured (grey) and simulated (black) SWAT discharge line. Variables to be derived could include the mean discharge (MQ, as indicated by the arrow), maximum / minimum discharge in a defined period or mean seasonal flow values

The results from the hydrological SWAT model serve as input for the hydraulic HEC-RAS model, which simulates 1D open channel hydraulics and sediment transport processes in river networks. It contains a number of sediment transport formulas to calculate instream sedimentation and erosion, and can perform steady flow, unsteady flow, sediment transport/mobile bed computations, water temperature modelling, and water quality analysis (USACE, 2010a). It utilises the momentum equation in the case of supercritical flow and on hydraulic structures, and solves the energy equation for basic profile calculations with the standard step method. HEC-GeoRAS (USACE, 2005) is used to prepare HEC-RAS input files from river geometry (soilAQUA, 2009) and morphology data (DAV-WBV/LAND S.-H., 2006) in ArcGIS.

An ArcGIS interface is used to couple SWAT and HEC-RAS (Kiesel et al., 2012). Flows and sediment loads from each SWAT tributary are transferred to the respective HEC-RAS cross sections for each daily time step. Hydraulic and substrate-specific parameters were extracted from HEC-RAS at the 544 cross sections along the 9-km-long river section from 2006 to 2009, and annual mean values were calculated based on daily parameters. All HEC-RAS cross sections were then linearly interpolated to obtain a total of 1,590 continuous hydraulic parameter ASCII maps with a 5-m grid size (1,730 cells), which were then used to select the appropriate predictors for SDMs.

Species distribution models

BIOMOD is used for modelling the geographic distributions of species and their environmental requirements. Occurrence data are statistically correlated with environmental data at each site to describe an environmental niche. Distributions are later projected to other areas where similar suitable conditions are found, and occurrence probabilities are computed. The modelling procedure within BIOMOD employs several individual algorithms, and provides an ensemble forecasting to reduce uncertainties in predictions derived from different modelling algorithms (Thuiller et al., 2009).

An ensemble model was created for *Sphaerium corneum*, based on a generalized linear model (GLM), a generalized additive model (GAM) and a generalized boosting model (GBM) at a spatial resolution of 5 m. Occurrence data were derived from the following unpublished surveys conducted between 2002 and 2010: a 2002 survey by R. Brinkmann, Schlesen, Germany (freelance biologist; contact details available from S.C.J); a 2006 survey by the LANU, Flintbek; 2008 and 2009 surveys by Stengert et al. (2008 and 2009) University of Duisburg-Essen; and a 2010 survey by the Schleswig-Holstein State Agency for Agriculture, Environment and Rural Areas (LLUR), Flintbek. Clam occurrence data at 34 known occurrence locations were split into a training set (70 %) and a testing set (30 %) by applying a random partition as described in Araújo et al. (2005), which allows a validation analysis to be performed based on one occurrence dataset. Each algorithm used 500 pseudo-absences, following the recommendation of Barbet-Massin et al. (2012) to use a relatively large amount of pseudo-absences and ten-fold cross validation for model calibration, resulting in a total of 34 models including consensus models. Because of the small size of the case study catchment and the available amount and distribution of sampling data of organisms, we used a hydraulically oriented subset of available data, omitting data on hydrology, water quality, temperature or land use from an original set of 20 variables by pair-wise correlations ($-0.7 < r < 0.7$) and expert knowledge. However, some variables, for example land use, are still indirectly considered via the implementation in the SWAT model. Four environmental predictors were used for each grid cell: water depth [m], flow velocity [m s^{-1}], stream power [kg m^{-1} s^{-1}] and sediment discharge [metric t d^{-1}]. The variable 'stream power' represents the energy dissipation against the streambed and banks, a combination of shear stress and velocity, while the variable 'sediment discharge' measures the transport of sediment. Because organism data was spatially and temporally heterogeneous, we decided to use annual means for each grid cell. The final model results from a weighted average consensus procedure to minimise uncertainties derived from different algo- rithms, known as ensemble model. For this purpose single algorithm results (10 repetitions per algorithm) were averaged by multiplying their AUC (area under the receiver operating characteristic curve) scores with a decay weight of 1.6. The use of weighted averages has been proven to be superior in creating consensus models (Marmion et al., 2009). We finally transformed the model output into a binary presence-absence map by applying a cut-off value which minimises the difference between sensitivity and specificity (Liu et al.,

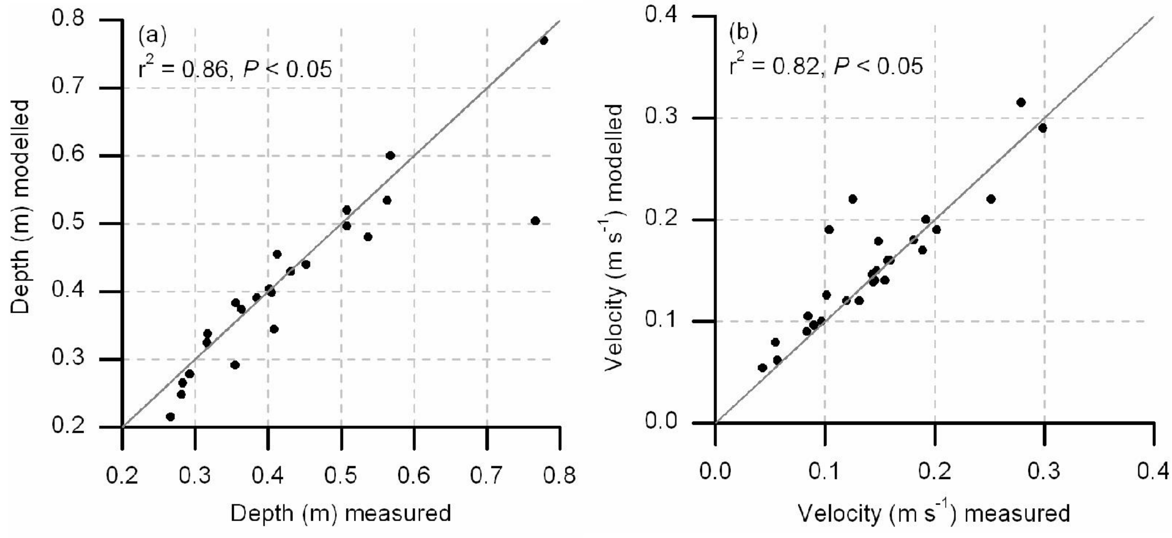

Figure 7.4 HEC-RAS model values and comparison to measured values (Tavares, 2006) for depth (a) and velocity (b). The grey line represents a 1:1 line. Variables to be derived could include the mean or maximum / minimum parameter values for a defined period of time

Table 7.1 Mean values of the modelled variables at the riverine sampling sites and the predicted occurrence of Sphaerium corneum, and variable ranges in the whole 9-km study area along the Kielstau River (min. – max.). Mann–Whitney U-test between grids of sampling sites and predicted occurrence was non-significant (P > 0.05) for all variables

Variable	Sampling sites	Predicted occurrence	Study area
Sediment discharge (metric tonne day^{-1})	2.64 (±1.55)	2.96 (±1.92)	0.07 – 19.12
Water depth (m)	0.29 (±0.04)	0.29 (±0.04)	0.11 – 0.51
Flow velocity ($m\ s^{-1}$)	0.21 (±0.05)	0.22 (±0.06)	0.04 – 0.95
Stream power ($kg\ m^{-1}\ s^{-1}$)	0.81 (±1.12)	1.01 (±1.58)	0 – 67.28

2005). We extracted the ranges of the modelled variables at the sampling sites to describe the preferred habitat and compared it with variable values at the modelled sites. The contribution of each variable in the final ensemble model was assessed by giving each variable used by the GLM, GAM and GBM the same weighting factor that was used for building the consensus projection.

7.3 Results

The SWAT model showed a good model performance (RMSE = 0.06, r^2 = 0.82, and NSE = 0.78; for details see Kiesel et al., 2010a). Likewise, the linked SWAT-HEC-RAS model simulates the hydrological and hydraulic regime from 2006 to 2009 in very good agreement with measured data (Figure 7.4a,b). Sediment simulations were validated with suspended sediment measurements, leading to an agreement in monthly sediment loads of r^2 = 0.68 (data not shown).

Sphaerium corneum is predicted to occur in 232 raster cells according to our results, i.e.

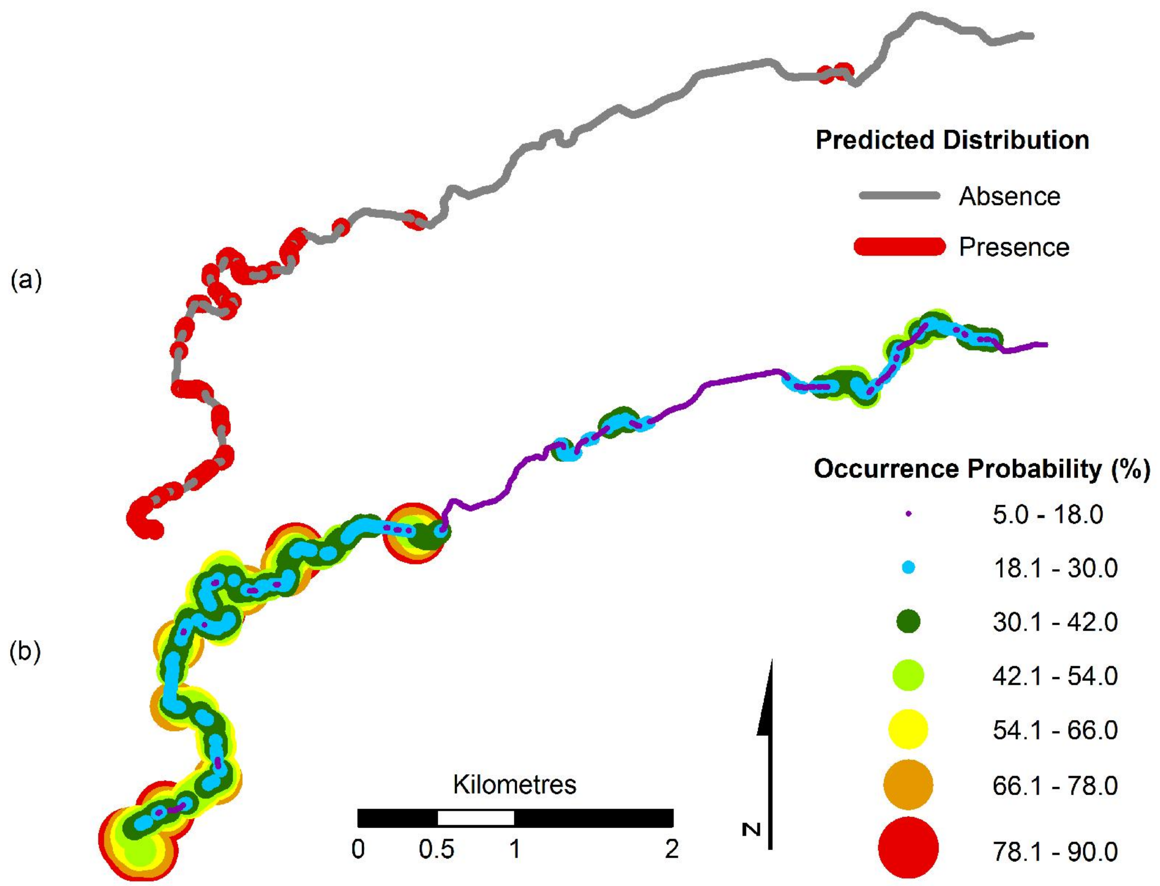

Figure 7.5 Spatial predictions of Sphaerium corneum as (a) presence / absence and (b) occurrence probabilities along the modelled 9-km section of the Kielstau Stream

in about 13.4 % of the modelled area. The ensemble model (Figure 7.5) obtained good evaluation scores (AUC: 0.96, kappa: 0.86; True Skill Statistic, TSS: 0.95; sensitivity: 86.14; specificity: 85.75). Mean values for the modelled variables at the sampling sites of *Sphaerium corneum* are very similar to the values of the predicted distribution (Table 7.1, Mann–Whitney U-test not significant for all variables, $P > 0.05$). High occurrence probabilities were predicted in the downstream half of the 9-km section of the Kielstau. The most important variable for the model was sediment discharge, contributing 40 %, followed by water depth (30 %), flow velocity (19 %), and stream power (11 %).

7.4 Discussion

7.4.1 Integrated modelling of *Sphaerium corneum* in the Kielstau catchment

To model invertebrate occurrences in a catchment framework, actual flow and sediment boundary conditions of the hydraulic modelling domain have to be known for all tributaries during the entire modelling period. This dynamic link poses a challenge for modelling and was solved by using a hydrological model to depict flow and sediment contributions. These data, influenced by catchment management practices and the natural climate, serve as inputs into the hydraulic model: this is then used to simulate the resulting water levels, velocities and sediment processes depending on stream channel characteristics. By considering these abiotic parameters, a major part of the functional chain influencing the occurrence of *S. corneum* can be depicted.

The result in this case study correspond to the known basic ecological requirements of *S. corneum*, which is described from a range of habitats, from wells below springs (Metarhithral) to lentic sites and ponds (littoral) (Nesemann and Reischütz, 2002; Schmidt-Kloiber and Hering, 2011). It is plausible that a freshwater clam, such as *S. corneum*, is dependent on slowly flowing water for the provision of organic sediment to filter and feed upon. A strong current would either erode the fine sediment it burrows in or may even dislodge the clam and transport it downstream. A certain depth in the water column is necessary to withstand temporal fluctuations of the river discharge (Dussart, 1979). In this model, predictions of occurrence seem to cluster at river bends, where sediment discharge, flow velocity, and stream power tend to be reduced, while water depth tends to increase in contrast to straight sections. In this small-scale case study, the data produced proved to be sufficient to model the distribution of *S. corneum* successfully.

7.4.2 Challenges and outlook: integrated modelling of river ecosystems

Aquatic invertebrate SDMs have not been used extensively for large-scale analysis, despite promising first attempts (Balint et al., 2011; Domisch et al., 2011). Typically SDMs rely on terrestrial based bioclimatic data. However, the abiotic factors that structure riverine communities are different than those that influence communities in the terrestrial realm. These particular factors in riverine ecosystems call for integrated model approaches to provide habitat suitability predictions of aquatic organisms using adequate predictors.

Several challenges related to the particular environmental conditions in rivers remain and a full model for riverine invertebrates would have to include the following variables and dependencies.

(1) Hydrological time series are required to derive the low and high flow extent and dates or other seasonally dependent variables. The correct depiction of peak and low flows for single events can be very exact when using small modelling time steps with sufficient data. However, over long time periods, the depiction of extremes lacks accuracy due to data constraints because topography, artificial drainage pathways, soils, and land use data are usually not available dynamically. Sediment and water quality modelling inherits high uncertainties;

thus, the reliable generation of such hydrological model output in ungauged basins is still a challenge.

(2) A full model should also include variables related to hydraulic conditions on the reach or site scales, such as shear stress, sediment availability or distribution, current velocity, water depth, and river bed morphology (e.g. riffle-pool sections, shoreline shape, and other similar variables). For hydraulic models, modelling sediment transport and substrate changes on local scales with reasonably small error margins is a challenge due to temporal, spatial, and physical substrate data availability and computation time.

(3) Other abiotic predictors in stream environments that are not provided by hydrological/hydraulic models have either scarce data or data that are collected independently from biological data and it is not always easy to combine these. Such data include, for example, stream temperature, oxygen content, and nutrient availability, the latter two both being dependent on the first: temperature. Although stream temperatures may be estimated from air temperatures (Caissie, 2006), it imposes the challenge of including factors that are directly and indirectly linked to the stream and that affect stream temperature patterns, e.g. riparian vegetation, geography, and urban settlement (Caissie, 2006). Furthermore, it is important to bear in mind that water provides a buffering solution, and that the lotic state causes a spatial (by the linear structure) and hence temporal lag compared to the outside.

(4) Catchment-related variables, including riverine vegetation and different land use types (most prominent is the proportion of urban land use), are rather easy to obtain. Additionally, in many parts of the world, virtually all rivers show impact from past anthropogenic influence. This 'ghost of land use past' (Harding et al., 1998) is considered one of the major predictors for current communities, but is rarely considered in an adequate way in either ecological studies or modelling approaches. Eventually, it is unclear how significant influences from upstream areas or certain lateral influence from directly adjacent areas (Kail and Hering, 2009; Kappes et al., 2011) could be considered in stream SDMs.

(5) While for some issues, an improved data basis might help (e.g. stream temperatures, nutrients, past and current land use data, etc.), other challenges may be addressed by integrating further models, either directly or by coupling of model output. For instance, coupling a vegetation model (Hickler et al., 2004) with a hydrological model could further improve data accuracy in terms of temperature predictions, shading, or organic material input. Guisan and Thuiller (2005), Elith and Leathwick (2009) or Schurr et al. (2012) mention that there are attempts to integrate SDMs with dynamic and other kinds of models to better represent ecological processes and to allow the inclusion of mechanistic, population, and landscape change effects, but none of these attempts consider riverine ecosystems.

(6) In addition to abiotic drivers, biotic factors also restrict the availability of suitable habitat for species. One special challenge is posed by the different life stages of stream macroinvertebrates. Insects have different larval and adult live stages, which should be considered differently in the models, by life-stage specific habitat requirements or even more pronounced when aquatic and terrestrial life stages are passed. A classic full dispersal assumption, which is often applied, probably falls short when considering major relevant barriers to both aquatic life stages (dams) or aerial life stages (land use, light pollution).

(7) In addition, several of the aquatic organism groups show large natural dynamism (e.g. macrophyte growth and subsequent ecological effects). They might also show highly complex behaviour, such as migration, compensation flights, or drift, which are not fully understood and thus are difficult to consider in a model. Because of the linear structure and lateral influences, communities are highly dependent on distance, size, and conditions of source populations in the surroundings or remaining catchment (Brederveld et al., 2011). Interactions between organisms themselves are not yet taken into account; however, this is a problem shared by most biotic models. To develop a common concept of how barriers, source populations, and interactions could be considered in a SDM would set a new benchmark for niche model-

ling, and approaches are being presented by Kissling et al. (2011), Marion et al. (2012) and Schurr et al., (2012).

(8) Similar to other SDM applications, the 'presence–absence challenge' is still unsolved, and it seems that dispensing of pseudo-absences is particularly tricky in river ecosystems. An absence at a river sampling site might be caused by different processes, including true absence, seasonal absence (aquatic/terrestrial life stage), the rather obscure sampling habitat for humans, the short-term removal of organisms by flood, or other drift causing events. The help of an observational model (Marion et al., 2012) could assist in overcoming the use of pseudo-absences.

7.5 Conclusions

From this and other studies (Kuemmerlen et al., 2012; Schmalz et al., 2012b), we conclude that the proposed model integration between hydrological, hydraulic, and SDMs is a feasible approach to gain further insights into the distributions of stream organisms. The presented model approach is in principle transferable to other catchments or taxa of interest. Yet, we acknowledge the shortcomings of our approach, being data intense by e.g. requiring hydrological and hydraulic models to be elaborated beforehand for a specific catchment or region, requiring extended biological datasets, and relevant abiotic data. Furthermore, several challenges remain for future modelling approaches, such as the difficulties that arise from considering the environmental parameters required in continental to global studies (i.e. large-scale studies).

One of the most evident advantages of our approach is the use of public domain (open source) models at all levels, control of input data in models and their linkages among each other, hence the chance to improve calibration and projections of different spatial and temporal scales within riverine environments, and the use of (biological) ensemble models to allow for uncertainty analysis. Such models can provide useful information for environmental management of the stream channel or the landscape. If there is sufficient knowledge of a catchment, predictions could be made of, for example, how planned changes in land use might alter the composition of the community in a stream. Furthermore, response curves might be useful to select indicator taxa (Dedecker et al., 2004) or determine most influential environmental variables on communities.

7.6 Acknowledgement

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Chapter 8 A new model linking macroinvertebrate assemblages to habitat composition in rivers: development, sensitivity and univariate application

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Abstract: Habitat models are frequently used to simulate species occurrence and abundance in rivers and lakes. While most of these models target individual species, there is yet no macroinvertebrate community model whose output can be directly linked to ecological stream assessments. These assessments are usually based on metrics calculated with a list of occurring taxa and their abundances. Such a model would allow simulating the effect of environmental changes, e.g. due to climate or land use change, on macroinvertebrate assemblages.

This paper describes the development, sensitivity analysis, calibration and application of the empirical, statistical macroinvertebrate community model HET. The model requires three types of input data: First, generating data (or reference data): Habitat-specific, quantitative macroinvertebrate lists of taxa for reference sites, composed of species abundance information in different habitats. These samples are grouped into Habitat Sensitivity Classes (HSCs), which we define as a combination of environmental variables. Second, environmental data on test site: For the test site, where the macroinvertebrate assemblage is simulated, spatial distribution of HSCs needs to be known, either from field sampling or from habitat models. Third, biotic data on test site: Data on macroinvertebrate abundances for the HSCs are required to calibrate the model.

The model is applied in two steps: First, the species abundances in each HSC of the test site are calculated. Transformation formulas are used to find an optimum relation between generating and test data. Suitability values are calculated for each HSC and each species, from which species master lists per HSC are derived. These master lists are then randomly 'sampled' using bootstrapping to mimic the randomness of real-world sampling with the option to reproduce model results through high bootstrapping repetitions. Second, the species assemblage within the targeted stream section is calculated, applying an electronic Multi-Habitat-Sampling (MHS), which is frequently used in river assessment. The resulting species lists can be used to calculate metrics and ecological status.

A first application of the tool was carried out using a lowland dataset of 454 generating- and 162 test samples, the latter from a small lowland stream in Schleswig-Holstein (Germany). The generating samples were filtered and grouped in five subdatasets using the HSC 'substrate', which results in a univariate simulation since only one abiotic variable is used. A sensitivity analysis showed that the transformation formula settings have the highest influence, followed by quality and filtering of the input data. Model simulations reach a Renkonen Index of 56 between simulated and observed species abundance within the HSCs of the test site. A Renkonen Index (RI) value higher 50 indicates a high agreement of the species communities. Based on these results we carried out two MHS, based on a substrate maps from 2008 and 2009. For each MHS application, we ran 1,000 repetitions to evaluate the impact of random sampling on the overall MHS result. Random sampling variance impacted ecological status classes and metrics, but does not mask the influence of the substrate changes from 2008 to 2009. We thus conclude that the model is useful for assessing the impact of substrate changes on macroinvertebrate assemblages and ecological status.

Keywords: macroinvertebrate; substrate; habitat model; species density; multi-habitat sampling

8.1 Introduction

Benthic invertebrates are frequently used for assessing the status of river ecosystems. Besides reflecting the current status of streams, invertebrates are also supposed to respond to changes in both bottom habitats and water quality. While the status of a river can be assessed by comparing the current invertebrate assemblage to a reference condition, the prediction of future assemblages, e.g. following river restoration, can be investigated by simulations.

Species simulation approaches usually yield probabilities of occurrence or a more general index reflecting habitat suitability for a species. Despite the difficulties involved in simulating species abundances, a simulation of entire assemblages has considerable advantages (Ferrier and Guisan, 2006): The simulation results could be used to assess assemblage responses to stress or to define ecological status classes applying assessment schemes (Hering et al., 2004; Furse et al., 2006).

For simulating species communities in aquatic habitats, two main types of models are used: 'process based' and 'empirical models' (Harby et al., 2004). In complex 'process based' models, the whole life cycle of species and species communities can be simulated, and knowledge about the species functioning and interaction is essential and must often be drawn from observations (Schuwirth and Reichert, 2013). 'Empirical models' directly relate environmental variables to a species' habitat and occurrence, resulting in 'knowledge rules' that can be used to make spatial occurrence predictions (Lehmann et al. 2002). These models are usually based on standardized sampling schemes, which are filtered, grouped, classified and summarized to deduce species dependencies on their environment (Feio and Poquet, 2011). The applicability of these models are controversially discussed (Lancaster and Downes, 2009; Lamouroux et al., 2010, Lancaster and Downes, 2010; Ahmadi-Nedushan et al., 2006). The general consensus is, however, that the model has to fit to both the underlying database as well as the application purpose (Caron-Lormier et al., 2009; Ferrier and Guisan, 2006; Guisan and Zimmermann, 2000).

Empirical habitat models are either multivariate or univariate. Multivariate models consider multiple variables and their interaction. Examples are multivariate regression techniques (Chessman, 1999), artificial neural networks (Goethals et al., 2007), fuzzy rule-based functions (VanBroekhoven et al., 2006), or decision trees and support vector machines (Hoang et al., 2010). The complexity of 'process based' and multivariate models has no theoretical limit. However, process knowledge and data availability restrict practical and area-wide applicability of very detailed habitat models (Statzner, 2008). Also, Sickle at al. (2006) suggest minimising the number of variables to avoid the risk of overfitting. Many multivariate macroinvertebrate modelling approaches consider a combination of substrate and hydraulic flow properties for the spatially and temporally explicit simulation of microhabitats (Mérigoux and Dolédec, 2004; Brooks et al., 2005; Schwendel et al., 2010). While these approaches yielded successful applications, critics stress that similar hydraulic conditions cause different suitability depending on river size, slope and type of substrate (Jowett, 2003), which can lead to unpredictable and implausible results (Gore et al., 2001). Univariate models instead rely on a single variable and relate this to species occurrence, absence or abundance. Univariate models have the advantage that empirical data are easier to gather, more widely available and thus lead to a broader application range (Barry and Elith, 2006). Schlossberg and King (2009) summarize their disadvantages, which are mainly oversimplification and high errors in the results. Obviously, the maximum possible explanation of a univariate model is constrained to the temporal and spatial resolution of the chosen variable. This certainly needs to be considered before applying a univariate model, since it is well known that habitat suitability for macroinvertebrates depends on a complex interaction of numerous abiotic and biotic factors

(Statzner et al., 2008; Lancaster and Downes, 2009). To disentangle these dependencies, statistical analyses of large datasets have been performed (Furse et al., 1984; Hering et al., 2006; Statzner et al., 2007). Substrate is seen as the single most important factor for macroinvertebrates on microhabitats in numerous analyses (Furse et al., 1984; Lammert and Allan, 1999; Poepperl, 2000; Hering et al., 2006; Duan et al., 2009; Gibbins et al., 2010; Schröder et al., 2013). Macroinvertebrate assessment schemes, e.g. for the implementation of the European WFD (EC, 2000), are often based on substrate-selective sampling of the river bed through the standardized MHS (Hering et al., 2003; Barbour et al., 1999). Its output is used to obtain representative species lists for a given river reach as an input for ecological assessment. Integrating the MHS into a simulation tool would enable a widespread model application in ecological assessments of streams based on spatio-temporal substrate changes.

Against this background, we developed an empirical, stochastic habitat model for the simulation of macroinvertebrate communities in a lowland stream, called Habitat Evaluation Tool (HET). To enable a link to metric calculations (e.g. for applying assessment methods or calculating 'feeding type' composition), model output includes species abundances and their spatial distribution. In this paper we (1) describe the model, (2) evaluate model sensitivity, (3) calibrate the model against different input datasets, data transformation methods and variable settings and (4) apply the model as a univariate simulation just using substrates as habitat variables. To ensure a wide application range of the developed model, we included options for multi-and univariate simulations, depending on the input data availability. The MHS application of the model is used to evaluate the impact of randomness of samples and the impact of spatio-temporal changes of substrate distributions on the macroinvertebrate assemblage.

8.2 Material and methods

Figure 8.1 shows the HET modelling approach. It is divided in three main modules: (1) dataset analysis to process the input data; (2) simulation of species abundances; (3) evaluation

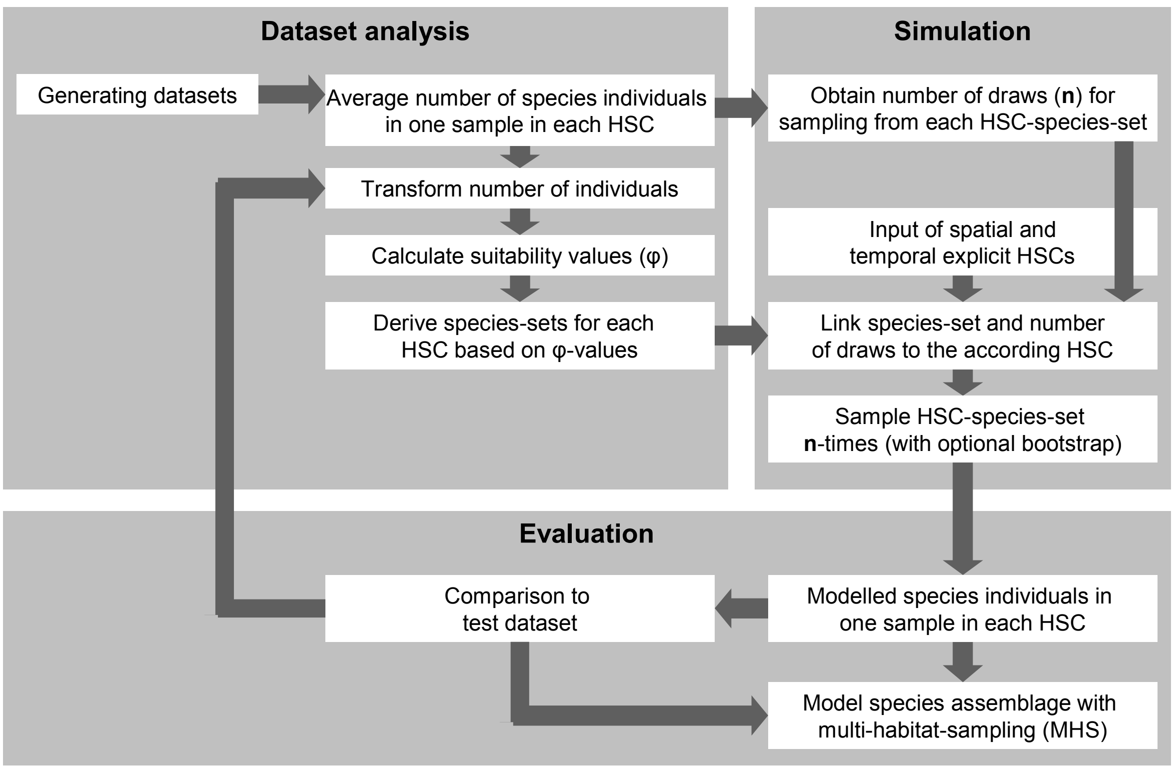

Figure 8.1 Overview of the HET model approach

to test model performance and to apply MHS. Each simulation component is explained in further detail in the following subsections.

In general, we define a HSC, herein referred to as *h*, as a combination of environmental conditions. In this paper we consider only substrate as a habitat, but theoretically it can be made of combination of variables, e.g. one HSC could be 'dead wood' with a flow velocity of 0-0.1 m s^{-1} and a nutrient level of 1-3 mg l^{-1} nitrate. It can thus be any combination of variables like substrate (*u*), flow velocity (*v*), and water quality (*q*) classes: *h(u,v,q)*. In the case of a univariate model that considers substrate as the main input variable, *h* is only dependent on *u:h(u)*. For clarity, the dependencies of *u* are not noted in the model equations.

8.2.1 Dataset analysis

Generating database

For the model species abundance data are required. Those data are obtained from species sampling programs which are structured according to the HSCs; e.g., if HSCs are based on substrates and depths, samples for different substrate and depth classes are required. For each HSC and species combination, the average number of individuals in one sample is calculated through:

$$\left\{\left\{\bar{I}_{sh} = \frac{\sum_{c=1}^{c=C_h} I_{sh\_c}}{C_h}\right\}_{s=1}^{s=S}\right\}_{h=1}^{h=H} \qquad \{8.1\}$$

where *s* is the current species id; *h* is the current HSC. This implies that the variables without subscript are the same for all combinations, with subscript *h* are the same for all species, with subscript *s* are the same for all HSCs and with subscript *sh* are individual for each species and HSC. Variables with subscript *sh* thus occur in a matrix with *s* times *h* fields. *S* is the last species id; *H* is the last HSC id; *c* is the current field sample; C_h is the total number of samples on the HSC *h*; I_{sh_c} is the number of individuals of species *s* on habitat *h* in sample *c*; $\bar{I}_{sh}$ is the average number of individuals of species *s* in HSC *h* in one field sample.

Transformation of the number of individuals

Different transformation techniques are tested for varying the ratios between species individuals of the generating dataset. Field biological data are commonly transformed prior to calculations to account for outliers; macroinvertebrates are often patchily distributed even in well-suited habitats, so there is always the chance to sample a spot with an unusually high abundance. The choice of transformation and its variables are possible calibration options of the model. One of two different data transformation functions is implemented on {8.1}. The first one is the power-root function (P-R):

$$\left\{\left\{I_{T\_sh} = \left(\bar{I}_{sh} + a\right)^b\right\}_{s=1}^{S}\right\}_{h=1}^{h=H} \quad \text{with } a > -\min\left\{\left\{\bar{I}_{sh}\right\}_{s=1}^{s=S}\right\}_{h=1}^{h=H} \quad \text{and } b > 0 \qquad \{8.2\}$$

where I_{T_sh} is the transformed number of individuals of species *s* on HSC *h*; *a* is the calibration factor that is added to the number of individuals; *b* is the calibration factor that defines the strength of the power transformation (if $b > 1$) or the root transformation (if $b < 1$). If

$a = 0$ and $b = 1$, data are not transformed. Values for $b < 0$ are not allowed since it would result in an inversion of species abundance data.

The second transformation formula for altering the ratios within the generating dataset is the log function (LOG):

$$\left\{\left\{I_{T\_sh} = \log_b\left(\bar{I}_{sh} + a\right)\right\}_{s=1}^{s=S}\right\}_{h=1}^{h=H} \quad \text{with } a\text{>1-min}\left\{\left\{\bar{I}_{sh}\right\}_{s=1}^{s=S}\right\}_{h=1}^{h=H} \tag{8.3}$$

where I_{T_sh} is the transformed number of individuals of species s on HSC h; a is the calibration factor that is added to the number of individuals. The logical expression for a is an internal model condition and depends on the logarithmic function since the logarithm of 0 is not defined and values between 0 and 1 would yield negative transformation values. During the calculations, it is thus checked if the current entry of a would result in a logarithm smaller than 1. b is the calibration factor that defines the base of the logarithm. Values for b only lead to different transformations in case $b > 1$ or $b < 1$. For example, $b = 1.1$ or $b = 2.3$ yield the same transformation result since the ratios of the individual numbers are not changed. Similarly, additional transformations that do not alter the ratio between species individuals and yield the same ratio between transformed and untransformed data, like the Hellinger or Chord transformation (Borcard et al., 2011), are not useful since the following suitability calculations lead to the same result.

Calculation of suitability values

Based on {8.2} or {8.3}, suitability values (φ) are calculated for each species within each HSC to standardize the transformed abundance data (Borcard et al., 2011):

$$\left\{\left\{\varphi_{sh} = \frac{I_{T\_sh}}{\sum_{s=1}^{s=S} I_{T\_sh}}\right\}_{s=1}^{s=S}\right\}_{h=1}^{h=H} \quad \text{with } \sum_{s=1}^{s=S} I_{T\_sh} > 0 \tag{8.4}$$

where φ_{sh} is the suitability value for species s in HSC h. A value of $\varphi_{sh} = 0$ indicates no suitability for the species on HSC h, a value of $\varphi_{sh} = 1$ indicates that the HSC is only suitable for one species, a value of $(0 \geq \varphi_{sh} \leq 1)$ indicates that the habitat is suitable for multiple species, with the higher φ_{sh}, the higher the suitability. The sum $\left\{\sum_{s=1}^{s=S} \varphi_{sh}\right\}_{h=1}^{h=H}$ will yield 1 for each HSC h. If all φ-values for a HSC are zero, the HSC is not suitable to any species and the abundance is always zero. The smallest φ-value different from zero is arbitrarily set to 0.00001 to avoid unrealistically high individual numbers in the following species-set calculation.

Derivation of species-set for each habitat

The standardized suitability values are used to generate internal species-sets:

$$\left\{\left\{I_{I\_sh} = \text{int}\left(\frac{\varphi_{sh}}{\min\left\{\varphi_{sh}\right\}_{s=1}^{s=S}} \cdot k\right)\right\}_{s=1}^{s=S}\right\}_{h=1}^{h=H} \quad \text{with } \min\left\{\varphi_{sh}\right\}_{s=1}^{s=S} \geq 10^{-5} \text{ and } k > 0 \tag{8.5}$$

where I_{I_sh} is the internal number of individuals for each species *s* in each HSC *h*; *k* is a positive integer that defines the lowest number of individuals in the species-set, which corresponds to the smallest suitability value within HSC *h*. *k* should be chosen sufficiently high, so that the desired accuracy (= number of digits from φ_{sh}) is reached for the species-set.

8.2.2 Simulation

Obtaining number of draws for sampling from each habitat-species-set

For the simulation, the derived species-sets for each HSC from {8.5} will be sampled n_h -times, which is defined as:

$$\left\{ n_h = \sum_{s=1}^{s=S} \overline{I}_{sh} \right\}_{h=1}^{h=H} \qquad \{8.6\}$$

where $\overline{I}_{sh}$ is taken from {8.1}. The number of draws n_h from each modelled HSC species list is thus equal to the total number of individuals given in one sample of the source data HSC.

Input of spatial and temporal explicit habitats

Information about the test site need to be available in the format of maps or tables that define all modelled HSCs in space and, if relevant, in time. For instance, if the HSCs are solely based on substrates, the relevant information is a map of substrate distribution on the stream bottom. Mapping programs, statistically or arbitrarily derived HSC distributions, or results from environmental modelling can be used to generate such data.

Linking species-set and number of draws to the according habitat

Results from {8.5} and {8.6} are linked to the data described in section 2.2.2. Each HSC on the map or table thus has an assigned species-set ($\left\{ I_{I_{sh}} \right\}_{s=1}^{s=S}$) and an according number of draws (n_h).

Sampling substrate-species-set

For each habitat from section 2.2.2, sampling with replacement is carried out n_h -times to obtain a simulated species distribution:

$$\left\{ I_{M\_h(s)} = \left(s_1, s_4, s_2, s_1, \ldots s_S, s_S \cdot f \right) \right\}_{h=1}^{h=H} \text{ with } f = n_h - \lfloor n_h \rfloor \qquad \{8.7\}$$

where $I_{M\_h(s)}$ is all modelled individuals of different species on HSC *h*; s is one individual of a random species (1 to *S*) with total number of occurrence $s = n_h$. The last entry only exists, if n_h is a fractional number with fraction *f*.

Summarizing all individuals of the same species yields:

$$\left\{ \left\{ I_{M\_sh} = \sum_s I_{M\_h(s)} \right\}_{s=1}^{s=S} \right\}_{h=1}^{h=H} \qquad \{8.8\}$$

where I_{M_sh} is the modelled number of individuals of species *s* in HSC *h* in one simulated sample.

The actual value of I_{M_sh} depends on the random draws and is thus most likely different for each model run. While random sampling is important for the application of the model, randomness is disadvantageous for testing the model hypothesis and for optimizing the calibration variables described in 2.1.2. For calibration, we can thus use the bootstrapping method (Efron, 1979) for forcing similar results for different model runs:

$$\left\{\left\{\bar{I}_{M\_sh} = \frac{1}{R}\sum_{r=1}^{r=R} I_{M\_sh\_r}\right\}_{s=1}^{s=S}\right\}_{h=1}^{h=H} \qquad \{8.9\}$$

where $\bar{I}_{M\_sh}$ is the sample mean of the modelled number of individuals of species *s* in HSC *h* in one simulated sample; *r* is the current sample; *R* is the total number of samples or bootstrapping repetitions, and $I_{M\_sh\_r}$ is the modelled number of individuals for the sample *r*.

8.2.3 Evaluation and results

Comparing simulation results to test dataset

For assessing simulation performance, the modelled number of individuals per species (modelled species list) can be compared with observed number of species individuals (observed species list) for each HSC. As the statistical measure of agreement between the two species lists, we chose the RI (Renkonen, 1938) due to its wide application in dissimilarity classifications. RI is calculated based on {8.9} and a test dataset:

$$RI = \sum\left\{\min\left(\frac{\bar{I}_{M\_sh}}{\sum_{s=1}^{s=S}\sum_{h=1}^{h=H}\bar{I}_{M\_sh}}, \frac{\bar{I}_{V\_sh}}{\sum_{s=1}^{s=S}\sum_{h=1}^{h=H}\bar{I}_{V\_sh}}\right)\right\}_{s=1}^{s=S} \cdot 100 \qquad \{8.10\}$$

where $\bar{I}_{V\_sh}$ is the average number of individuals of species *s* in habitat *h* in one test sample. *RI* is 0 if two species lists do not have any species in common and 100 if the lists are identical. The commonly used coefficient of determination (r^2) is not suitable as a measure of agreement between the modelled and observed taxa lists, since it can be biased in case the data is non-uniformly distributed, i.e. mass occurrences of one or a few species and low occurrence of other species.

Modelling species assemblage through Multi-Habitat-Sampling

Based on spatial information about the distribution of substrates in the test site and equation {8.1} to {8.8}, a digital macroinvertebrate MHS can be performed. MHS is a method where 20 macroinvertebrate samples are taken in a representative stream section and widely used for monitoring streams. The total sum of each substrate-area in the stream section is divided by the total section area which yields the substrate proportion. The 20 samples are then distributed according to these proportions. For each 5-% threshold, one sample is taken from the according substrate, while substrates covering less than 5 % are not sampled. The individuals from all 20 samples are grouped to form one community.

Due to the possible dependence of habitat *h* on environmental variables *h(u,v,q)* other than substrate, the result of {8.8}needs to be averaged over the area of each substrate *u*:

$$\left\{ I_{M\_sh(u)} = \frac{\sum_{v}\left(\sum_{d}\left(\sum_{q} I_{M\_sh(u,v,q)} \cdot A_{h(u,v,q)}\right)\right)}{A_{h(u)}} \right\}_{u=1}^{u=U} \quad \{8.11\}$$

where $I_{M\_sh(u)}$ is the modelled number of individuals of species *s* on the substrate $h(u)$ in one simulated sample; $I_{M\_sh(u,v,q)}$ is the modelled number of individuals of species *s* in HSC $h(u,v,q)$ in one simulated sample (from {8.8}); $A_{h(u,v,q)}$ is the area of habitat $h(u,v,q)$ and $A_{h(u)}$ is the area of the substrate $h(u)$.

The implementation of MHS in the model is accomplished through equation {8.12} - {8.15}.

$$A_{t0} = \sum_{u=1}^{u=U} A_{h(u)} \quad \{8.12\}$$

$$A_t = A_{t0} - \sum_{u=1}^{u=U} A_{h(u\_<5\%)} \quad \{8.13\}$$

where *u* is used as the notation for substrate; $A_{h(u)}$ is the area of the substrate; A_{t0} is the original total domain area; A_t is the total corrected domain area; $A_{h(u\_<5\%)}$ is the area of substrates that have an area ratio of less than 5 % ($\frac{A_{h(u)}}{A_{t0}} < 0.05$).

The number of samples on each substrate *u* are then calculated through:

$$\left\{ n_{MHS\_u} = \left(round\left(\frac{A_{h(u)}}{A_t} \cdot \frac{1}{5} \cdot 100 \right) \right) \right\}_{u=1}^{u=U} \text{ if } \frac{A_{h(u)}}{A_t} \geq 0.05 \quad \{8.14\}$$

where n_{MHS_u} is the number of MHS samples on substrate *u*. Applying {8.8} and {8.11} and summarizing all individuals according to the species yields the MHS-species list:

$$\left\{ I_{MHS\_s} = \sum_{u=1}^{u=U} \left(\sum_{r=1}^{r=n_{MHS\_u}} I_{M\_sh(u)\_r} \right) \right\}_{s=1}^{s=S} \quad \{8.15\}$$

where I_{MHS_s} is the number of individuals of species *s* on all MHS-sampled substrates; $I_{M\_sh(u)\_r}$ is the number of individuals of species *s* on substrate *h* in MHS-sample *r* (from {8.11}). The resulting species list $\{I_{MHS\_s}\}_{s=1}^{s=S}$ can then be used to calculate metrics and ecological status classes, e.g. using software like ASTERICS (2013).

8.3 Model application

8.3.1 Calculation example

To illustrate the model algorithms described above, Table 8.1a-n shows an example calculation. As described earlier, the model has no theoretical limitation in terms of number of

Table 8.1 HET calculation example: (a) number of individuals of species s per sample c on HSC h; (b) average number of individuals per sample; (c) transformed data using power-root function with a = 0 and b = 0.5; (d) phi-values; (e) internal species lists using k = 100 for the least abundant species; (f) number of draws from (b); (g) randomly drawn species list; (h) modelled number of individuals in one digital sample; (i) observed number of individuals in one sample of the test dataset; (j) calculated RI; (k) summarized number of individuals for each substrate u; (l) areas of substrates for MHS, uncorrected and corrected domain area; (m) number of digital MHS samples; (n) summarized species lists sampled with MHS

(a) generating data

I_{sh_c}	c1	c2	c3	c4	c5	c6	c7	c8
s=1, h=1	1.0	0.0	1.0	0.0	1.0	0.0	0.0	0.0
s=2, h=1	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
s=3, h=1	0.0	0.0	0.0	1.0	0.0	3.0	1.0	0.0
s=1, h=2	0.0	0.0	0.0	0.0				
s=2, h=2	1.0	2.0	0.0	0.0				
s=3, h=2	2.0	0.0	3.0	1.0				
s=1, h=3	0.0	1.0	2.0	0.0	0.0	0.0	2.0	
s=2, h=3	1.0	2.0	1.0	3.0	0.0	0.0	1.0	
s=3, h=3	4.0	4.0	3.0	6.0	1.0	2.0	5.0	

(b)

$\overline{I_{sh}}$ {8.1}	h1	h2	h3
s1	0.375	0.000	0.714
s2	0.125	0.750	1.143
s3	0.625	1.500	3.571

P-R a=0, b=0.5

(c)

I_{T_sh} {8.2}	h1	h2	h3
s1	0.612	0.000	0.845
s2	0.354	0.866	1.069
s3	0.791	1.225	1.890

(d)

φ_{sh} {8.4}	h1	h2	h3
s1	0.349	0.000	0.222
s2	0.201	0.414	0.281
s3	0.450	0.586	0.497

k=100

(e)

I_{I_sh} {8.5}	h1	h2	h3
s1	173	0	100
s2	100	100	127
s3	224	141	224

(f)

n_h {8.6}	h1	h2	h3
	1.125	2.250	5.428

(g)

$I_{M\_h(s)}$ {8.7}	h1	h2	h3
	s3	s2	s3
	s1*0.13	s3	s2
		s3*0.25	s2
			s3
			s3
			s1*0.428

(h)

$\overline{I_{M\_sh}}$ {8.8} {8.9}	h1	h2	h3
s1	0.125	0.000	0.428
s2	0.000	1.000	2.000
s3	1.000	1.250	3.000

R=10000

(j)

RI {8.10}			
			77

(i) test data

$\overline{I_{V\_sh}}$	h1	h2	h3
s1	0.765	0.125	0.875
s2	0.294	0.563	1.188
s3	1.147	1.938	6.000

(l) substrate areas [m²] for MHS

	u1	u2	u3
A_h	235.0	15.0	189.0
A_{t0} {8.12}			439.0
A_t {8.13}			424.0

(k)

$\overline{I_{M\_sh(u)}}$ {8.11}	u1	u2	u3
s1	0.125	0.000	0.428
s2	0.000	1.000	2.000
s3	1.000	1.250	3.000

(m)

n_{MHS_h} {8.14}	h1	h2	h3
	11	0	9

(n)

I_{MHS_s} {8.15}	
s1	4.7
s2	17.8
s3	37.8

HSCs *h* and species *s*, but for a comprehensive illustration, we restrict the example to three HSCs and three species. The HSCs in the example are three different substrates. The bold variables in the table headings refer to the result of the associated equation indicated as the numbers in brackets (see Chapter 8.2 for an explanation). The arrows show the workflow of the calculations. Additional variables that are necessary for the indicated equation are supplied on the respective arrows. In Table 8.1a we have listed an example of a raw generating dataset: the number of individuals of each species *s* on each HSC *h* in each sample *c*. For example, sample c7 on HSC 3 contains eight individuals. In HSC 2, only four samples have been taken. Table 8.1i corresponds to Table 8.1b and contains the average number of individuals for each species and HSC in one sample of the test site. This represents the test dataset. Table 8.1k is equal to Table 8.1h since the HSCs are already substrates and do not need to be aggregated as shown in equation {8.11}. The areas of the substrates *u* for the MHS are shown in Table 8.1l.

8.3.2 Database

The generating dataset is composed of 454 samples taken between 2001 and 2011 within the scope of various research projects, carried out in nine streams in Germany (DE) and the Netherlands (NL) (for detailed information see Schröder et al., 2013). The investigated lowland streams are located at an altitude lower than 200 m ASL. Catchment size ranges between 10-100 km^2 for small streams and 100-1,000 km^2 for mid-sized streams, respectively. In terms of catchment geology, streams are impacted by the last ice age. Running in ground and terminal moraines and sandy deposits, the channel substrates are predominantly characterised by varying proportions of sand and gravel. Further, parts of the stream bottoms are covered by organic matter, composed of fine and coarse detritus, wood or macrophytes, with a subordinate amount of mineralic substrates. An overview of the datasets is given in Table 8.2. The available generating datasets are filtered according to different criteria to test if subsets of the dataset yield better simulation results: Subsets are created based on environmental variable similarity between generating and test study sites (Table 8.2). Table 8.3 shows the filter criteria and the number of resulting samples in each subset. The test dataset (see section 8.3.3) against which all model results are tested, is also used as one input dataset (TD) for assessing general model plausibility.

Within all studies considered, a standardized substrate-specific kick-sampling procedure was applied, using a 25 x 25-cm frame shovel sampler (500-µm mesh). Sampling was predominantly performed during spring and summer. Substrates sampled are shown in Table 8.4. Each sample included one type of substrate only. This fact limits the application of the model

Table 8.2 Dataset description and subsets; for subset abbreviations see Table 8.3; DE Germany, NL Netherlands

Country	Year	Ecoregion	Sample season	Streams with number of samples	Included in subsets	Citation
DE	2005	Central lowlands, 14	summer	Gartroper Mühlenbach (22) Schwalm (35)	GD, SGD, TGD, UGD GD, UGD	Lorenz et al. (2009)
DE	2005-2007	Central lowlands, 14	summer	Niers (217)	GD, TGD, UGD	Schattmann (2013)
DE	2002	Central lowlands, 14	spring summer	Elting Mühlenbach (31) Bever (32)	GD, SGD, TGD, UGD GD, SGD, TGD, UGD	Kramm (2002)
NL	2002	Western lowlands, 13	summer autumn	Heelsume Beek (20) Oude Beek (19) Tongerensche Beek (18)	GD, SGD, UGD GD, SGD, UGD GD, SGD	Vlek et al. (2006)
DE	2011	Central lowlands, 14	spring	Treene (60)	GD, BGD, UGD	Schröder et al. (2013)
DE	2008-2009	Central lowlands, 14	spring	Kielstau (only for validation) (162)	TD	Stengert et al. (2008, 2009)

Table 8.3 Available generating datasets with number of samples

Abbreviation	Filter description	Number of samples
GD	all generating samples	454
BGD	same basin	60
SGD	similar stream size	142
TGD	same river type	302
UGD	similar river status	282
TD	test dataset	162

to univariate simulations, since each HSC must be defined as one substrate type, but cannot be split into combinations of other abiotic variables. The samples were stored separately and preserved with 96 % ethanol in the field. Each sample was subsequently processed in the lab and all organisms were removed from the sample. Identification of the organisms was carried out to the lowest possible taxonomic level (usually species).

8.3.3 Test area

The model was tested in a 230-m-long reach of the Kielstau. The Kielstau is a small lowland stream with a catchment area of 50 km² in northern Schleswig-Holstein, Germany. Mean temperature and precipitation are 8.2 °C and 893 mm, respectively (DWD, 2010). Catchment land use is dominated by arable land and pasture (82 %) (DLR, 1995). Average discharge at the catchment outlet at gauge Soltfeld is 0.42 $m^3 s^{-1}$ (LKN, 2010).

The stream channel of the Kielstau has been subject to straightening and clearing for enhancing drainage. Consequently, flow velocities increased and caused erosion in parts of the streams bed and banks, while the remaining sections are very stable (Kiesel et al., 2013). Mostly due to the structural remodelling, the morphological state of the gravel stream is assessed 'poor' to 'moderate' (Olbert et al., 2006). However, the area is subject to intense interest by seven environmental associations, authorities and foundations that are investigating the current state and are discussing rehabilitation measures at all scales. Continuous measurements and sampling programs are carried out on different scales concerning water quantity and water quality, hydraulic variables, stream morphology, erosion, sediment transport, climatic variables, phytoplankton and macroinvertebrates. The focus on ecohydrological relevant processes recently led to the designation of the Kielstau as Germany's first UNESCO ecohydrology demosite (Fohrer and Schmalz, 2012).

The sample reach is located in a near-natural part of the river close to the catchment outlet. Stream width is about 4 m. Water depths range between 0 m to a maximum of 1.3 m at gauge Soltfeld. Three distinct bends, four riffles and five pools facilitate a diverse current pattern. Mobile substrates, mainly sand and gravel, are interspersed with large wood debris, alder roots, stones, water plants and coarse particulate organic matter.

Table 8.4 Substrate description with number of samples

Substrate class	Substrate class description	Number of samples
akal	fine to medium sized gravel (2 cm - 0,2 cm)	70
lithal 1	blocks and large boulders (> 40 cm - 20 cm)	17
lithal 2	cobbles and coarse gravel (20 cm - 2 cm)	47
psammal	sand (< 2 mm)	122
pelal	mineralic mud (< 6 µm)	40
coarse particulate org. matter	deposits of coarse particulate organic matter	30
living parts of terr. plants	fine roots, floating riparian vegetation	16
phytal	submerged and emergent macrophytes	76
xylal	dead wood, roots, twigs, branches	36

8.3.4 Simulation runs

The model was calibrated and tested using the datasets described above (Table 8.2) and applying the univariate simulation procedure explained below. The calibrated model was then used for electronic MHS sampling with simulated substrate distribution data from the Kielstau for two different substrate distributions in 2008 and 2009.

Model sensitivity and calibration

Model sensitivity runs are carried out varying the input subdatasets (Chapter 8.3.2) as well as the transformation functions (equation {8.2}and {8.3}) and their parameter settings. For each run, the RI as described in equation {8.10} is calculated using simulated and observed species data. The resulting spread of the RI indicates the sensitivity of the subdataset, transformation function or parameter. The runs are limited to the species occurring in both the Kielstau and the respective generating dataset. These were 12 to 17 species depending on the subdataset. The substrates, which are used in the simulations, are given in Table 4. During the sensitivity evaluation, the number of bootstrapping repetitions must be set to an adequately high number so that the model runs are reproducible. The number of repetitions of 800 was found through setting initial bootstrapping repetitions to 50 and doubling these until no difference in the resulting RI of the current bootstrapping repetition-run was observed to the last repetition-number simulation. The finally used minimum and maximum values for *a* and *b* are defined based on multiple model runs where high variable ranges were used. From these runs it became clear that increased variable values yielded equal worse model performance. For instance, a transformation with higher powers than two resulted in unrealistically high abundance for the most common species. Figure 8.2 shows the performance diagrams of the models with the six different input subdatasets (rows) and the two transformations (power-root (P-R) function left, logarithm function (LOG) right). The x-axis shows variable *a*, the y-axis variable *b* and the z-axis RI. The arrow marks the best RI for each dataset and transformation function. The three numbers above each diagram are *a* / *b* / RI of the best model run. Generally, the LOG transformation results in lower RI values than the P-R transformation. For the LOG transformation, setting variable $b < 1$ yielded worse results due to a negation of species abundance data. The maximum agreement between modelled and simulated dataset is yielded for the test dataset (TD) with an RI of 80. However, an optimum model should be able to produce a perfect agreement (RI = 100) when the test dataset is used as model input. For any variable setting of the used input data, the LOG function transformation is thus not a plausible transformation option. It is however kept in the model since the log transformation is a valid and applied data transformation method which may yield plausible results for other generating datasets. The P-R function obtained an RI value of 100 for the validating dataset and $a = 0$ and $b = 1$, which is 'no transformation' of data. This is an important model plausibility test, since the untransformed test input data must equal the calculation results. It also clarifies why the LOG transformation cannot reach an RI of 100: any value for *a*- and *b* in equation {8.3} always results in a data transformation. For further assessment of the runs, the test dataset (TD) is excluded from the analysis. The sensitivity analysis shows, that the choice of transformation function influences the best RI values within a maximum of six points (GD, P-R = 48 vs. LOG = 42), but as mentioned earlier, the LOG transformation is in our case not considered a valid transformation option. The influence of the transformation variables for the P-R function within the chosen ranges is quite distinct with a range of 24 RI-points (GD dataset). The different types of generating datasets lead to a range of 19 for RI. In summary, the source dataset and the transformation function parameter setting are most important for the model performance, which combined can influence the result in our case by almost a factor of three (RI of 16 vs. 56).

The sensitivity analysis results can directly be used for choosing the optimum calibration setting. As can be seen for the remaining P-R dataset's a- and b values, which are all different from 0 and 1, the highest RI is never reached through exactly 'no transformation'. The highest RI of 56 is reached through subdataset BGD, the species data from the same basin, and transformation variables $a = 0.3$ and $b = 0.7$, which are relatively close to the 'no transformation'

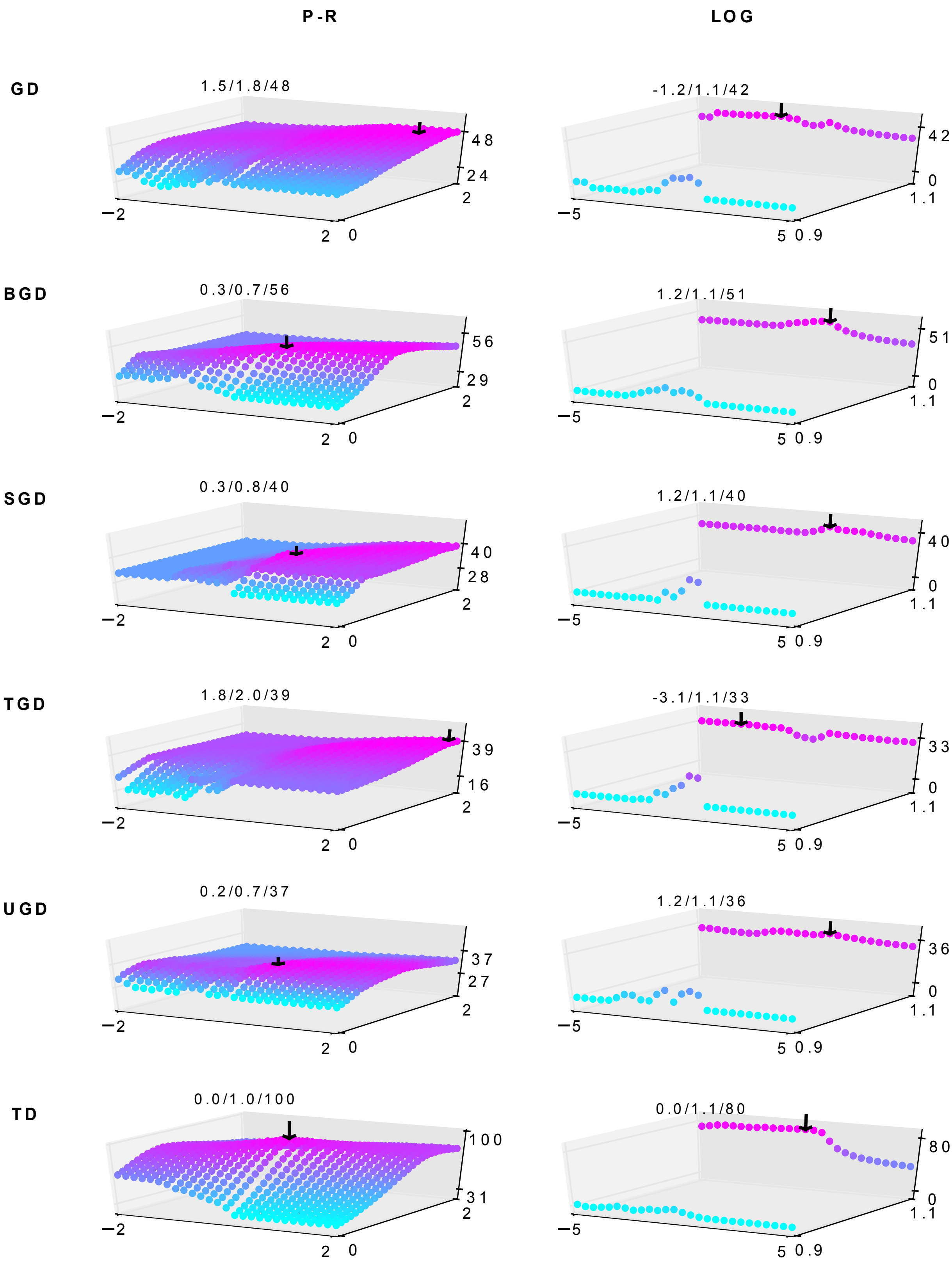

Figure 8.2 Performance diagrams of models with different input subdatasets (rows) and the two transformations (power-root function left, logarithm function right). The x-axis (-2 to 2) shows variable a, the y-axis variable b and the z-axis the RI

setting. If $a > 0$, the generating dataset contains less individuals on certain substrates than there are present in the test dataset for this substrate. Similarly, if $b > 1$, abundant species in the validating dataset have higher number of individuals than abundant species in the generating dataset, and if $b < 1$, abundant species in the validating dataset have a lower number of individuals than abundant species in the generating dataset. Based on the variable setting of the best simulation, it can thus be deduced that in BGD low abundance species are slightly underrepresented while the high abundance species are overrepresented compared to the test dataset. All other subdatasets yield lower RI values, of which GD (all available samples) performs second best and UGD (similar river status) shows the lowest agreement (RI = 37).

According to Zerbe and Wiegleb (2008), a RI value higher than 50 already shows a high agreement of the species communities. Occurrences and abundances of benthic invertebrates are influenced by many variables acting at the microhabitat scale (e.g. substrate composition, substrate stability influencing biofilms, current velocity) along with variables acting at larger scales (e.g. water quality, water temperature). Each model can only cover a restricted fraction of the species' environment. In particular, this concerns models which reflect abundances rather than species occurrences, as abundances are also influenced by chance: particularly extremely high abundances of single species might be a result of attractance (i.e. individuals select spots which are already colonized by other individuals of the same species and thus reflecting suited conditions). For many species the factors governing occurrences and abundances are unknown or only partly known. Given these facts, a univariate modelling approach just using a single, coarsely defined variable and yielding a maximum coherence with field conditions of 56 % is a promising result.

MHS application

We used the best performing subdataset (BGD), transformation function (P-R), and the calibrated variable setting ($a = 0.3$, $b = 0.7$) to simulate the species occurring in both the generating and the test dataset (n = 15). An electronic MHS was carried out based on substrate distributions in the river reach of the Kielstau. The temporal and spatial distribution of substrates was observed and simulated through a hydrologic and hydraulic modelling cascade (Kiesel et al., 2013). The change in substrate composition from April 2008 to April 2009 is shown in Table 8.5. Generally, fine sediments were eroded and substituted by coarser sediments. These two substrate maps are taken as the basis for an electronic MHS. The MHS is carried out with the above described datasets and model settings, resulting in a species list for each model application. Since it was one aim of our study to test the impact of random sampling on the composition of the species list, we used only one bootstrapping repetition. This is equivalent to a real-world MHS where, naturally, each sample is taken only once. The repetitions of the MHS are set to 1,000, which means that the MHS at the stream section is carried out 1,000 times to obtain a spread of MHS results based on random sampling of the substrates. In real-world this cannot be done successively, since a previous MHS will influence the following MHS at the same spot in terms of species composition. The resulting 1,000 spe-

Table 8.5 Substrate changes from 2008 to 2009 (Kiesel et al., 2013)

Substrate	Area [m²] 2008	Area [m²] 2009	% change
xylal	9.0	12.1	35 %
living parts of terr. plants	16.7	16.7	0 %
phytal	3.8	5.5	44 %
coarse particulate org. matter	8.3	10.4	26 %
pelal	175.5	96.1	-45 %
psammal	181.3	74.1	-59 %
akal	308.9	476.0	54 %
lithal 2	48.6	55.5	14 %
lithal 1	6.8	8.7	28 %

cies lists were then analysed first, according to the species abundance difference between 2008 and 2009 and second, according to the spread in abundance of each species based on the MHS repetitions.

8.4 Results and discussion

The observation that generating data from the same basin (BGD) yields the best results is in agreement with Allouche et al. (2008), who found that incorporating distance constraints in

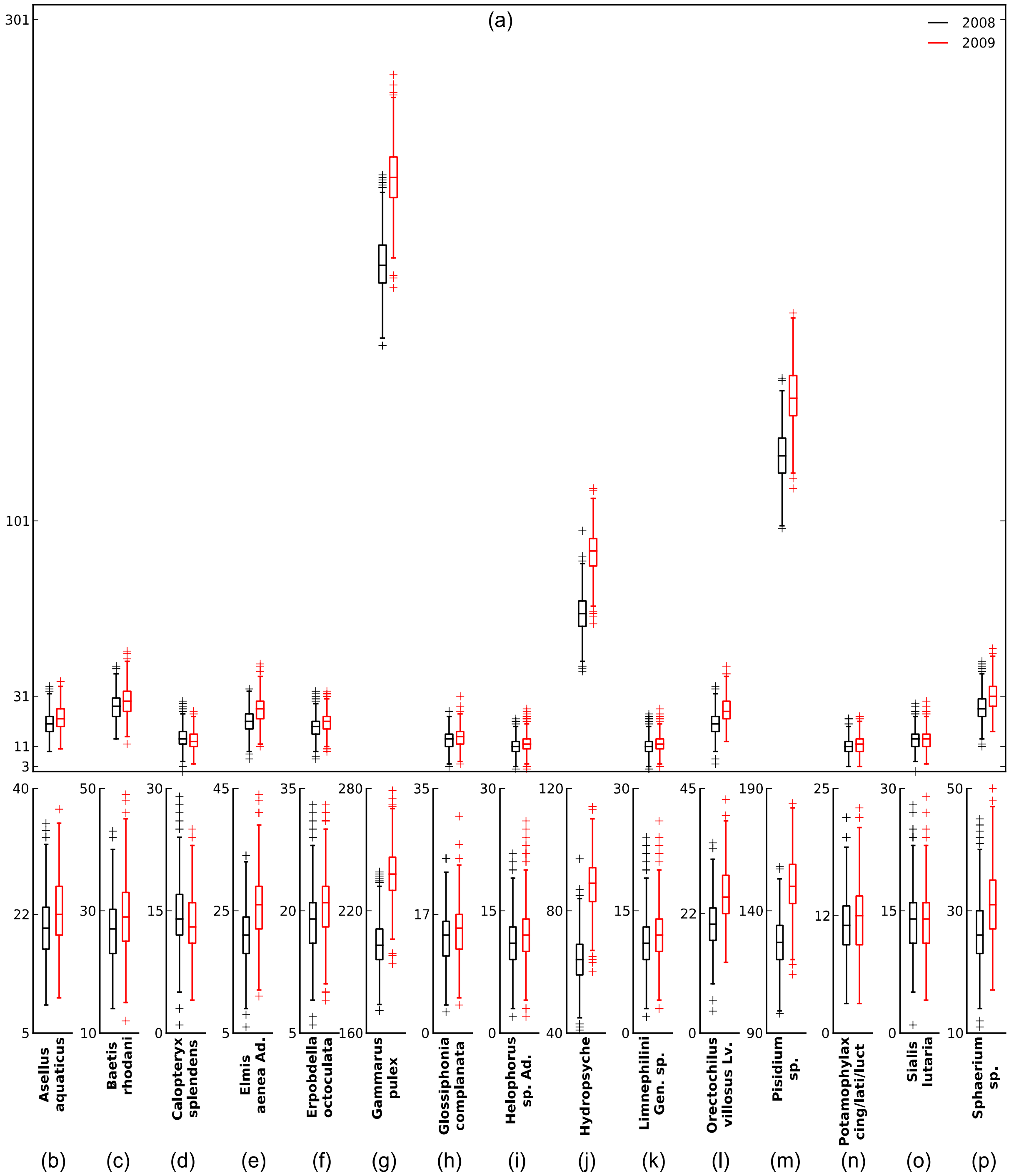

Figure 8.3 Box-whisker plots of the 1,000 species lists of the MHS repetitions for the 2008 and 2009 substrate distribution, y-axis unit is number of individuals per species, which are distributed along the x-axis, the center line of the box plots mark the median, the upper and lower extent of the box the third and first quartile, and the whiskers the 1.5-times range of the box' quartiles; crosses mark the outliers

their model lead to improved simulation results. The 1,000 modelled abundances for each species are shown in Figure 8.3. In the upper diagram, the plots for each species can be compared to each other. The box-whisker plots are grouped in pairs for each species (x-axis), The left ones indicate the modelled abundances for 2008, the right ones for 2009. The y-axis unit is the number of individuals per 0.0625 m². The five y-axis values in Figure 8.3a represent the class break points as defined for the WFD macroinvertebrate assessment in Germany (Meier et al., 2006). The location within the diagram shows the general abundance of the species: *Gammarus pulex*, *Pisidium* sp. and the genus *Hydropsyche* are most abundant. All other medians, apart from *Spaerium* sp. in 2009, are located between 11 and 31 specimens.

In the first step of the results analysis we are discussing the changes between 2008 and 2009 (Figure 8.3b-3p): For all species but *Calopteryx splendens*, individual numbers increased from 2008 to 2009. These changes can be explained through the substrate associations of the species in the generating dataset (BGD): For example, *Elmis aenea* Ad., *Hydropsyche* sp., *Orectochilus villosus* Lv., *Pisidium* sp., and *Spaerium* sp. are most abundant on 'akal' (gravel). Thus, the area increase of this substrate has led to higher abundances in the simulated reach. In contrast, the damselfly *Calopteryx splendens* does not occur on 'akal' in the generating dataset; consequently, abundances decreased from 2008 to 2009. Concerning the magnitude of changes from 2008 to 2009, the median abundances of *Sphaerium* sp. were in the same classes in both years. Whiskers or outliers from *Pisidium* sp., *Hydropsyche* sp., *Calopteryx splendens, Elmis aenea* Ad., *Glossiphonia complanata, Limnephilini* gen. sp., *Orectochilus villosus* Lv., *Sialis lutaria* and *Spaerium* sp. differ in classes. This lends support to the conjecture that changes in substrate composition within one year can significantly influence the abundance of individual species and thus community composition.

In the second step, we analysed the variance from the random repetitions of the MHS: The length, or y-extent, of the box-whisker plots show the spread of the individual numbers for the 1,000 MHS model runs. The full range of most single box plots, including the whiskers, pass one or two class breaks. The 25-75 percentiles are mostly located within one class, but fill out up to a quarter of the class range. Naturally, the proximity of a species' median abundance to a class break influences the likelihood that the box plot passes the break. Species with low numbers show a smaller absolute spread than species with many individuals. This spread is due to the randomness inherited in the MHS sampling scheme when repeated multiple times. The smaller the spread, the more robust is the MHS application and the more representative is a single repetition of the MHS. When looking at all species, the number of class breaks can indicate the robustness of the electronic MHS sampling. In Figure 8.4, we summarized the number of MHS runs that yield the same individual class. Most species (63 %) end up with more than 800 out of 1,000 runs in the same class. For these species, the chance is higher than 80 % that the MHS repetition yield individual numbers that end up in the same class. The lowest value is 51.7 % for the 2009 runs of *Sphaerium* sp., which indicates that almost every MHS repetition yields a different classification. This is due to the fact that median abundance of *Sphaerium* sp. is close to the 31 individuals class break.

The results lead to the question if the variance of the MHS would mask a possible change of species assemblage due to different substrates. Therefore, a Kolmogorov-Smirnov (KS)-test was conducted to assess if the change from 2008 to 2009 is statistically significant. We chose the KS-test because only three modelled datasets follow the normal distribution and the KS-test can be used with non-normally distributed datasets. The KS-test revealed that only the p-value for *Sialis lutaria* is higher than 1 % (45.9 %), indicating that both datasets for *Sialis lutaria* are similar and all other species abundance changes from 2008 to 2009 are significant and thus not masked through the variance inherited in the MHS repetitions.

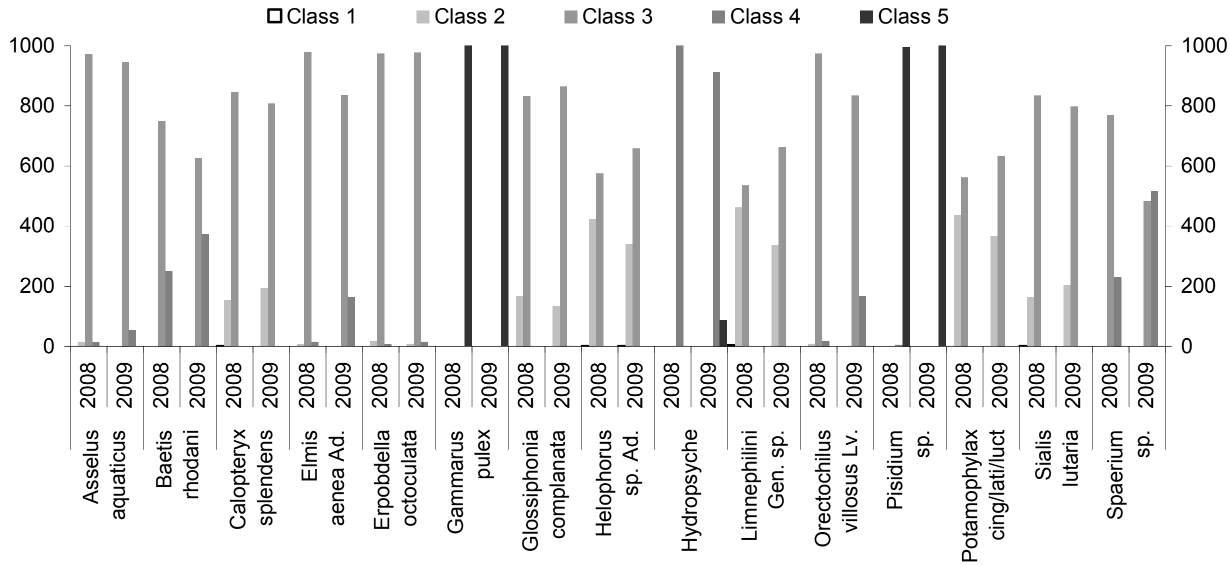

Figure 8.4 Number of run-repetitions that yielded a specific MHS results class (as defined on y-axis of Figure 8.3)

8.5 Conclusions

The objective of this study was the development of an empirical, stochastic macroinvertebrate habitat model whose output can be used for the assessment of the ecological status of streams and which has a wide potential application range through univariate or multivariate simulations.

We recognize that our univariate model application using substrates shows not an optimum statistical performance, because many other variables affecting the occurrence of macroinvertebrate species are not considered. We know that choosing only one predictor variable is a compromise between reaching high performance and being a generally applicable model (Guisan and Zimmermann, 2000). Thus, we base our conclusions not solely on the statistical performance. From the development, calibration, sensitivity testing and application of the model, we draw the following conclusions:

(1) The implemented model equations enable a flexible usage of species sampling data as long as the datasets of references and test sites are sampled with the same methodology and have the same HSC definitions. The number of variables that define a HSC (e.g. substrate, velocity and water quality class) are restricted by first, the variables included in the generating dataset and second, by the temporal and spatial knowledge of the variable distribution in the targeted stream reach. The second restriction is not a severe constraint since mapping, measurements and modern simulation tools can depict a variety of variables in a high spatial and temporal resolution at a target site. The first restriction, however, is due to macroinvertebrate sampling schemes, which are mostly not accompanied by a mapping of numerous environmental and instream variables. The maximum possible explanation of the model results are thus limited by the knowledge about the link between multiple variables and macroinvertebrate species absence, occurrence and abundance. We see three options to increase the possible model explanation: First, simulating species for which these links are known; second, analysing vast empirical datasets which are linked to multiple variables and third, researching species' preferences based on the features and characteristics of the species itself.

(2) Calibration of the model is carried out through input data transformations. Obviously, for that, a test dataset needs to be available. In case the model is applied in an unsampled stream, calibration is not possible and it is suggested to use no data transformation and a filtered input dataset with the samples taken from the same catchment or a dataset that includes samples from the same river type.

(3) The sensitivity tests revealed that the quality of the input dataset, respectively the agreement of the generating data with the test data, is of greatest importance for the model performance, followed by the variable setting of the transformation functions. The impact of repeated random MHS sampling on resulting species abundance lists is visible, but does not seem to influence ecological status classes to a great extent. We did not test the sensitivity on ecological status classes and metrics since we used a restricted species list due to the required agreement of species in the generating and test dataset. It is suggested to investigate this in more detail through applying the model with a full species list, e.g. a generating dataset from the test stream, and using the model results for the calculation of metrics. The model is sensitive to real-world changes in the predictor variable substrate, since it showed a statistically significant shift in species composition for two substrate maps that depict a realistic change from 2008 to 2009.

(4) In its current state the model can be used for predicting the impact of stream renaturation measures, where a change in substrate composition is a major rehabilitation factor. Therefore, the output can be used to calculate ecological status classes through a number of aquatic assessment schemes (Rolauffs et al., 2003; Hering et al., 2004; Furse et al., 2006). Testing the impact of substrate changes due to integrated catchment processes, e.g. through land use and managing the input of fine sediments, is another field of application. The model can be a valuable tool to quantify the shift in species composition due to such changing substrates. This species shift is represented through an increase in species abundance or reduction, up to the absence of certain species. The practical application of the tool is thus especially meaningful in cases where source populations in the stream are missing and other stream properties remain relatively stable. Currently, HET is applied in the IMPACT project (Kail and Wolter, 2012) where the dependence of the macroinvertebrate assemblage on environmental changes due to integrated catchment processes is evaluated.

8.6 Acknowledgement

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Chapter 9 Conclusion

9.1 Summary of key results

The main objective of the thesis was the conceptualization and application of an integrated model framework that can simulate the impact of catchment and instream changes on the riverine habitat of macroinvertebrate species. The key results, relevant for achieving this objective are summarized and grouped according to the four subgoals defined in Chapter 1.1.

(1) The developed interdisciplinary modelling approach is based on the DPSI(R) concept, which is utilised to depict parts of the complex cause-effect chain from the catchment and river to the aquatic habitat. *Drivers* are identified as: agriculture, urbanisation and flood control, which need to be properly represented in the model input data. The chosen models are an ecohydrologic catchment model (SWAT), two hydraulic models (HEC-RAS 1D and AdH 2D), which are all available in the public domain, and a GIS mapping submodel. The widely used ArcGIS environment is chosen as the platform to enable a wide application range. *Pressures* on the biotic system are defined as: hydrologic stress, fine sediment intake, hydraulic stress, channel profile alteration, substrate stability, substrate degradation, river cleaning and bank and bed fixation. The *State* of the habitat is defined through the parameters: duration of extreme events, velocity, water depth, clay, silt, sand, gravel, stones, wood debris, coarse particulate organic matter and instream vegetation. The *Impact* on the macroinvertebrates is assessed, using empirically derived relationships to link species preferences to the abiotic conditions or through species distribution models that directly correlate environmental conditions to species occurrence. A social and policy *Response* towards the impact is not considered in the modelling system. Performance of the modelling framework is tested on reproducing the status quo of the main *Pressures* (hydrologic, hydraulic and sediment transport) as well as the *Impact* on macroinvertebrate assemblages in the study area. Interdisciplinarity is ensured through the involvement of experts from the fields of agriculture, biology, engineering, environmental protection, soil science, hydrologic and hydraulic model development.

(2) Besides parameterizing the individual models, technical solutions for linking the models in the ArcGIS environment and applying them in lowland regions have been established: The models, applied in a cascade, are linked through complementary data utilisation and the usage of one model's output as an input to the next model which was realised through interface programs. SWAT was used to depict the 50-km²-catchment area of the Kielstau, HEC-RAS is used to simulate the 9-km-long main channel, furthermore referred to as 'main channel', of the Kielstau downstream of Lake Winderatt one-dimensionally, and AdH is applied on a 230-m-long reach segment, furthermore referred to as 'hotspot' at the catchment outlet. An ArcGIS pre- and post-processing user interface was developed for the 2D-hydraulic model AdH so that it can be used in the GIS environment. For the application in the Kielstau lowland catchment, characteristics of drainages and depressions have been implemented in the simulations. This more realistic process depiction improved hydrologic model perform-

ance, especially for intense summer rain events, discharges during the winter season and the hydrograph recession periods. Due to the lack of appropriate simulation tools for sediment input from agricultural drains and lacking long-term observations, a method was developed to assess sediment entry pathways in lowlands. In the Kielstau catchment, long-term sediment input shares are: channel (71 %), agricultural drains (15 %) and fields (14 %).

(3) The simulations of water and sediment fluxes across different scales were compared to observations and complemented by mapping campaigns. For the catchment hydrology, simulated discharge with SWAT at gauge Soltfeld shows very good agreement with observed data ($r^2 = 0.82$) for the 10-yr-modelling period. For the 1D hydraulics, simulated water levels and flow velocities with HEC-RAS at three locations along the main channel show very good agreement with observed data ($r^2 = 0.90$ and $r^2 = 0.88$ respectively) for 24 discharge events. For the 2D hydraulics, simulated water levels and flow velocities with AdH at eight river cross sections in the hotspot show very good agreement with observed data ($r^2 = 0.94$ and $r^2 = 0.70$ respectively) for one discharge event. The simulation of the temporal dynamics of sediment fluxes at the catchment outlet, using the combined catchment (SWAT) and instream (HEC-RAS) model, is plausible ($r^2 = 0.56$) but shows some weaknesses in the depiction of observed peak sediment loads. The simulation of the spatial distribution of instream erosion along the main channel (HEC-RAS) show realistic results. Simulated changes in mobile river bed substrates (clay, silt, sand, gravel) at the hotspot (AdH) from April 2008 to April 2009 also fit well to observed data. Simulation of certain substrates is not possible (stones, wood debris, coarse particulate organic matter, vegetation). These substrates were mapped in the hotspot and merged to the simulated mobile substrates.

(4) Species response to the changes in abiotic parameters have been evaluated on two scales using two different approaches: First, based on the combined SWAT-HEC-RAS model results for the main channel, the SDM BIOMOD was used to simulate habitat suitability for the freshwater clam *Sphaerium corneum*. BIOMOD requires species sample data within the model domain to run. Four parameters were evaluated which define the habitat suitability. 'Sediment load' had the highest explanation with 40 %, followed by 'water depth' (30 %), 'flow velocity' (19 %), and 'stream power' (11 %). By considering these abiotic parameters, the distribution of *S. corneum* showed good agreement to spatially distributed sampling campaigns. Second, an empirical habitat model (HET) was developed and connected to the combined SWAT-HEC-RAS-AdH model results for the hotspot. The HET model can theoretically be used without species data in the model domain, but then no calibration of the model is possible. It mimics real-world random macroinvertebrate sampling and was used to simulate the abundance of 15 macroinvertebrate species. Based on the parameter 'instream substrate', the model calculated the spatial abundance of species with good agreement to observed data (RI = 56). The final results of the simulations can be presented as dynamic species distribution maps which visualise the change in species composition over time in a comprehensible manner. A sensitivity test revealed that the quality of the input dataset describing the species habitat preferences is of greatest importance for the model performance. The successful applications of the two habitat models show that it is possible to link different habitat simulation approaches to the integrated simulation of the abiotic aquatic environment.

9.2 Capabilities and limitations

From the described key results, the following practical and scientific benefits can be deduced for each of the four subgoals:

(1) The integrated methodology can be used in different catchments since it incorporates the driving forces acting in complex watersheds. For instance, the applicability and transferability to other scales and also ecoregions (lowland, mid-range mountain and mountainous

environments), was already shown during the development of the model system: the methodology was successfully applied in a German low mountain range basin (Schmalz et al., 2012a), a mountainous Chinese catchment (Schmalz et al., 2012b; Kuemmerlen et al., 2012 and 2014), and on the example of a further lowland area, the IMPACT project (Kail and Wolter, 2012; Guse and Fohrer, 2011) was built on the described approach.

(2) The chosen tools are widely applied and constantly developed public domain models which makes the model framework applicable and future-proof. The hydraulic model chain, consisting of individual models acting on different scales has the advantage that the HEC-RAS model's lacking spatial resolution is complemented by the AdH model, or that AdH's longer computation time is complemented by the HEC-RAS model in case less spatial resolution is required. This makes it possible to choose the desired and necessary level of detail, also based on the data availability. The SWAT model enables the extension of simulated habitat parameters to chemical water quality components and the GIS mapping submodel enables the depiction of additional physical parameters, which can thus potentially be included into habitat assessments. Depending on the desired habitat simulation, the HET model (simulating species communities) and the BIOMOD model (simulating single species) can theoretically be linked to either the HEC-RAS or AdH model output since the hydraulic models are capable to depict similar hydraulic and sediment transport properties.

(3) The development and application of the model framework has enhanced process understanding, particularly in lowlands: With the integrated model chain, nested simulations of catchment and instream erosion processes can be used to investigate spatially distributed sediment sources. Furthermore, the separate representation of field, drain and bank erosion leads to an improved understanding of sediment entry pathways which is important for targeting mitigation measures. The consideration of the combined hydrological catchment characteristics drains and depressions, has led to an improved depiction of lowland hydrological processes especially during intense summer rain events and to an improved depiction of discharge recession after flood peaks.

(4) The key capability of the model framework is that global environmental changes as well as planned rehabilitation measures that influence water and sediment balance and channel hydraulics can be tested for their influence on the aquatic macroinvertebrate habitat. Examples of such influences are climate and land use change, renaturation of reach segments where a change in substrate composition is a major rehabilitation factor, managing the input of fine sediments, but also channel maintenance. The model is a first step to simulate the shift in species composition due to such changing environmental parameters. This species shift is represented through an increase or reduction in species abundance or habitat suitability, up to the total absence of certain species.

Models are simplifications of the natural system and thus, naturally, have limitations. It is important that the limitations are known and that the results obtained from the simulations are used and interpreted accordingly. For each subgoal, the identified limitations are:

(1) The abstraction of the aquatic habitat for macroinvertebrate species in the DPSI(R) framework resulted in the depiction of important parts of the abiotic-biotic cause-effect chain but it must be stressed that it remains a simplified depiction of the complex natural dependencies. Beechie et al. (2010) lists natural processes which would need to be considered for a more complete depiction of the riverine environment: "plant growth and successional processes, input of nutrients and thermal energy, and nutrient cycling in the aquatic food web." Also, the model framework does not consider the existence of source populations, migration and recolonisation as well as predator-prey impacts. The practical application of the tool is thus meaningful only when source populations in the stream are missing. This means that no new species can become part of the community over time.

(2) While a number of technical issues in the application of the model framework have been solved, some challenges remain: The connected models are individual, complex simula-

tion tools which depict different physical processes and require a broad experience in model setup, calibration and results interpretation and an in-depth knowledge of hydrological, hydraulic and aquatic habitat processes.

While the characteristics of a model cascade have the above defined advantages, model sensitivity and uncertainty also have a cascading effect. Their investigation and assessment requires passing data-, parameter- and algorithm uncertainty and sensitivity through the whole model chain, which was not possible to address within the scope of this thesis. Computational demand, particularly of the AdH morphodynamic calculations, restrict the simulation of yearly time periods to parallelised processor clusters.

Due to these restrictions, the complete model framework is not directly transferrable from the current research status to usage in administrations or engineering offices.

(3) The current depiction of the abiotic habitat conditions governed by the water and sediment fluxes has the following limitations: In simulations over long time periods, especially in the depiction of extremes, accuracy is decreasing due to the general lack of dynamically available input data in topography, bathymetry and in stream vegetation as well as the neglect of time period and time step-dependent model parameterisation (Ostrowski et al., 2010). However, while the simulation results of hydrologic and hydraulic parameters is acceptable, modelling sediment transport and substrate changes on different scales with reasonably small error margins remains a challenge.

Input and validation data need to be available on different points in time and spatial scales. The mapping campaigns carried out for this thesis were important to verify the simulations, but may be too cumbersome for future applications. Observed substrate changes were only available for erosion events, thus, while the HEC-RAS and AdH models calculate deposition, validation of sedimentation was not possible.

A weakness in the SWAT model's MUSLE approach for soil erosion, particularly in lowland applications, is the fact that erosive areas identified by the SWAT model may not be connected to the stream network in reality and thus may not contribute sediment to the streams as predicted by the SWAT model, but only reallocate sediment on the fields. Also, the depiction of tile drain sediment fluxes in the model system is suboptimal since it is only based on empirical relationships with a small sample size and thus needs to be checked individually when applied to other catchments. Both the HEC-RAS and AdH models are to date unable to simulate bank collapses which may cause a significant sediment contribution in single events.

(4) For applying the full model framework in its presented form, data requirements are high in both amount and quality. For instance, especially in small lowland streams, position- and height accuracy of river banks and bed need to be high for obtaining reliable hydraulic simulations, but such data is generally not available. Also species distribution data, linked to the abiotic environment needs to be available for verifying the simulations. However, species data collected for stream assessments is mostly aggregated for each site and the connection to the original habitat properties from which it was sampled is lost. A setup of the system with commonly available datasets which were gathered for the WFD is possible. However, the system must then be restricted to connecting the HET or BIOMOD model to the SWAT and HEC-RAS models only. In such cases, model results should not be used in an absolute, but rather in a relative manner. That means that the impact of changes should be evaluated in terms of a positive or negative effect rather than depicting a certain habitat status.

Due to sampling data availability, the BIOMOD model results could not be compared to temporal species distribution maps of *Sphaerium corneum*. Also the simulation with the HET model was only verified spatially: The observed species distribution in 2008 and 2009 showed a matching spatial occurrence of the species to the substrate, but showed a different temporal abundance on the substrates. This indicates that a habitat parameter different from the substrate has probably impacted species abundance from 2008 to 2009. Thus, the model could only be validated on the average species density of 2008 and 2009 on each substrate.

The model system is thus limited to temporal simulations where other stream properties besides the depicted ones remain temporally relatively stable.

These observed limitations are important findings for the advancement of integrated modelling.

9.3 Outlook

Within the process of working on this thesis, additional research deficits and the need for model extensions arose, which could not be completed within the scope of this work, but are listed below as a motivation for further studies. As mentioned in the limitations (Chapter 9.2), the presented model framework is a first step but not a fully comprehensive modelling system for simulating macroinvertebrate habitats. Some relevant components that are neglected and which would improve future simulation approaches have been identified. Again, the suggestions are grouped according to the four subgoals of the thesis.

(1) An improved connection between the simulated species and the abiotic environment is necessary. It can potentially be achieved through: First, simulating only species for which exact preferences during their life stages are known; second, analysing vast empirical datasets which are linked to multiple variables to refine or find new abiotic-biotic relationships and third, researching species preferences based on the features and characteristics of the species itself.

One advantage in using species for an assessment of riverine ecosystems is, that they 'accumulate' environmental conditions, i.e. in measurements on a monthly interval, problematic events can be missed, while the species 'monitor' their aquatic environment constantly and respond to it. However, macroinvertebrates alone do not allow a comprehensive assessment of the riverine status. The WFD, for instance, defines fish, macroinvertebrates, aquatic flora and phytoplankton as target organisms. Thus, the simulation should be extended to other species groups in the long run.

(2) In the introduction, the capability to simulate scenarios was mentioned as one major advantage of models. However, providing a scenario application within the scope of this work was not possible. An example of a scenario application of a comprehensive model chain that built on this work was carried out by Kuemmerlen et al. (2014 and submitted), Schmalz et al. (submitted) and is currently prepared in the IWRM-NET project IMPACT (Kail and Wolter, 2012; Guse and Fohrer, 2011).

A global sensitivity test and uncertainty analysis would enable an assessment which parts of the model chain and especially which input data and ranges are most important and require special focus. Such an analysis would give an important insight in the processes affecting riverine habitats.

(3) Since hydrological extremes as well as long-term hydrological trends over time are important for the riverine environment, hydrological research should aim for a systematic improvement in depicting and reducing uncertainty of low- and peak flows in long-term simulations.

Erosion and sediment transport-related suggestions include the adaptation of novel observation techniques for sediment entry pathway assessments, e.g. less labour intense than the presented methodology, but more expensive, are tracer investigations like the sediment fingerprinting (Walling, 2005; Russel et al., 2001).

An improved depiction of field erosion delivery ratios in the SWAT model, especially in lowlands, would be beneficial for the identification of spatially distributed sediment sources.

Further work is necessary for gaining sound knowledge about the physical processes forcing sediment input from agricultural tile drains.

Including algorithms to depict bank stability and collapses in the hydraulic models would improve the capability of the model system to simulate single-peak events of sediment load, especially in lowland rivers where channel erosion can be a major contributor of sediment.

(4) Since the general applicability and transferability of the model framework is mostly limited by data availability, gathering more sophisticated and area-wide data would be desirable, though it is expensive and labour intensive. Examples include water penetrating high resolution LiDAR data that produces a seamless surface topography and bathymetry as well as the standardised collection of a wide range of habitat properties of the sampling point at the time of sampling.

To improve the temporal accuracy of the model predictions, methods need to be found to widen the simulation to other abiotic parameters in stream environments beyond the parameters that were considered in this thesis, e.g. water quality, vegetation, shading, temperature and availability of food sources which are impacted through the riparian environment. Such parameters are separate models but can be potentially depicted in the GIS environment and included in a model framework through a GIS submodel which can be merged to the simulations similar to the described GIS mapping in this study.

But also biotic preferences need to be improved: macroinvertebrates have different larval and adult life stages, which should be considered in future model approaches, e.g. by life stage-specific habitat requirements for which different demands of species during its life cycle have to be known. For instance, this may include the definition of threshold parameters like 'hydrologic resilience' of species, which may be a unique combination of discharge and its duration combined with physical characteristics of the stream where species can seek refuge during certain stages of their life cycle. Especially dispersal through migration, drift, recolonisation, reproduction are influenced by barriers to both aquatic and aerial- or land-phase stages which would need to be considered for long-term simulations.

If these challenges would be solved and different organism groups could be depicted in one simulation tool, a final, major step forward would then be the simulation of interactions: Interactions between biota, like competition or predator-prey relationships as well as feedback loops of the biota with their environment.

Considering these recommendations, it becomes obvious that an enormous task, or rather numerous opportunities, still lie ahead for the different disciplines involved in optimizing riverine modelling. Lancaster and Downes (2009) and Rice et al. (2009) stress that interdisciplinary teams have the highest prospect of advancing towards these objectives.

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Summary of peer-reviewed publications from the author

1. Kiesel J, Schmalz B, Fohrer N. 2009. SEPAL – a simple GIS-based tool to estimate sediment pathways in lowland catchments. *Advances in Geosciences* 21(3), 25-32.
2. Kiesel J, Hering D, Schmalz B, Fohrer N. 2009. A transdisciplinary approach for modelling macroinvertebrate habitats in lowland streams. In: *Ecohydrology of Surface and Groundwater Dependent Systems: Concepts, Methods and Recent Developments*. Editors: M. Thoms, K. Heal, E. Bøgh, A. Chambel, V. Smakhtin. IAHS Publication Red Book Series 328, 24-33.
3. Liu H, Fohrer N, Hörmann G, Kiesel J. 2009. Suitability of S factor algorithms for soil loss estimation at gently sloped landscapes. *Catena* 77, 248-255.
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7. Jähnig SC, Kuemmerlen M, Kiesel J, Domisch S, Cai Q, Schmalz B, Fohrer N. 2012. Modelling of riverine ecosystems by integrating models: conceptual approach, a case study and research agenda. *Journal of Biogeography* 39(12), 2253–2263.
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10. Schröder M, Kiesel J, Schattmann A, Jähnig SC, Lorenz AW, Kramm S, Keizer-Vlek H, Rolauffs P, Graf W, Leitner P, Hering D. 2013. Substratum associations of benthic invertebrates in lowland and mountain streams. *Ecological Indicators* 30, 178-189.
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12. Kiesel J, Schröder M, Hering D, Schmalz B, Hörmann G, Jähnig SC, Fohrer N. 2014. A new model linking macroinvertebrate assemblages to habitat composition in rivers: development, sensitivity and univariate application. *Fundamental and Applied Limnology / Archiv für Hydrobiologie, Special Issue: Modelling in Freshwater Ecosystems*, accepted.
13. Schmalz B, Kuemmerlen M, Kiesel J, Cai Q, Jähnig SC, Fohrer N. submitted. Impacts of land use changes on hydrological components and macroinvertebrate distributions in the Poyang lake area. *Ecohydrology*, submitted 02/2014.

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Eidesstattliche Erklärung

Hiermit erkläre ich, dass ich die vorliegende Arbeit "Ecohydrologic and hydraulic stream modelling to describe aquatic habitats", selbstständig verfasst habe und keine weiteren Quellen und Hilfsmittel als die hier angegebenen verwendet habe.

Die Arbeit hat weder ganz noch in Teilen bereits an anderer Stelle im Rahmen eines Prüfungsverfahrens vorgelegen. Die Artikel aus Kapitel 2, 3, 4, 5, 6, 7 und 8 wurden in den dort angegebenen Fachzeitschriften veröffentlicht.

Die Arbeit ist unter Einhaltung der Regeln zur Sicherung guter wissenschaftlicher Praxis der Deutschen Forschungsgemeinschaft erstellt worden.

Kiel, 30.06.2014 Jens Kiesel