Scale appropriate analysis, assessment and management of landscape water and matter dynamics

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FOREWORD

The presented studies in this thesis emerged from my scientific activities of the last 10 years. The work was carried out mainly at the Department Landscape Ecology of the Helmholtz Centre for Environmental Research – UFZ in Leipzig, Germany. Other parts of this work resulted from several stays and collaborations with the teams of the USDA-ARS in Temple, TX, USA and Tifton, GA, USA, and with the Department of Geology of the Baylor University in Waco, Texas, USA.
The motivation for this work goes back to the beginning of the project "Landscape development, landscape balance and multiple land use of the region Bitterfeld-Dessau-Wittenberg", in which I collaborated closely with colleagues who worked in the so-called "Elbe ecology project". We had numerous and sometimes long-winded discussions on how we could achieve an integrated assessment of the impact of land use patterns on soil protection, nutrient fluxes, groundwater recharge, biodiversity, or even on landscape functions in general, in order to achieve sustainable landscape development. Which environmental measure could best help to improve soil protection, water availability, water quality or biodiversity on different scales from the field to the region? Which indicators and other methods would be best suitable to analyse these processes on different scales relevant to both science and environmental management? We discussed several valuable theoretical concepts from geography, landscape ecology, hydrology and systems sciences (such as hierarchy theory), and found it difficult to both transfer and implement these theories into spatial planning and environmental and river basin management, which are nevertheless hierarchically organised. Hence, I felt that there was (and still is) the need to bring together both theoretical and methodological aspects from different scientific disciplines such as landscape ecology, system sciences, hydrology, and soil science with the concepts and procedures of "the practice" such as spatial planning, water and forest management as well as nature conservation in order to achieve sustainable environmental management. Needless to say, that my collaborations with the relevant state and regional authorities were and still are invaluable for this work.
Besides these challenges, the bridge had to be made between valuable theories developed over decades - "traditional" methods of landscape, soil and hydrological analysis – and new and increasingly appearing GIS-, remote sensing and computer-based modelling techniques. By considering these aspects, I was able to develop a framework that included scale levels relevant for spatial planning, environmental and river basin management, and the associated suitable methods and data sets for such integrated analysis.

My work on these topics was spurred by my participation in several projects on landscape development and natural resources protection and economic development but particularly by the implementation of the European Water Framework Directive (WFD) in the year 2000: The implementation of this directive is considered as a paradigm change in river basin management. This new paradigm requires the following factors to support the integrated planning process: the use of river basins and surface water bodies as reference units (instead of administrative units), the consideration of several environmental and socio-economic aspects on different scales; and an emphasis on public participation in the planning process. For planning processes in river basin management, such as the development of a management plan, more transparency and multi-disciplinary strategies were required, which should be supported by GIS, models and spatial decision support systems (SDSS). I am grateful that I had the chance to work in two projects that aimed at the development of methods for integrated ecological-economic river basin analysis and management in order to support the implementation of the WFD. I had the opportunity in the process to complete my work on a methodological framework for (hierarchical) scale appropriate analysis, assessment and management of landscape water and matter dynamics. The procedure suggested in this framework was implemented within the FLUMAGIS project, where authorities responsible for water management, agriculture and nature conservation were also involved.

This is a cumulative thesis comprising 26 publications. However, I structured the work in five Chapters, describing stepwise the development of the methodological framework for scale appropriate analysis, assessment and management of landscape water and matter dynamics. The thesis starts with an introduction and an overview, in which the scope of the work is presented and the project background, from which the paper resulted, is explained. Each of the following four Chapters starts with a short overview of the research techniques, followed by the abstracts of the publications. In such a manner the studies can be better applied to the overall concept.

I would like to thank all the people who were involved directly and indirectly for their support in these studies and projects. Special thanks go to my scientific mentor Prof. Dr. Ralf Seppelt for his support, inspiring and motivating discussions and, finally, for his friendship. Parts of the theoretical considerations (Chapters 2 and 3) resulted from the intensive collaboration with my former colleague Prof. Dr. Uta Steinhardt. I would like to thank her for the great time in Leipzig. Thanks to Dr. Jeffrey Arnold and his team in Temple, TX, USA, Prof. Dr. Peter Allen, Baylor University Waco, TX, USA, Dr. David Bosch (USDA-ARS Tifton, GA, USA), for their support, help and friendship. My friends and former colleagues Prof. Dr. Rudolf Krönert (Leipzig), Dr. Markus Möller (Halle), Dr. Gerd Schmidt (Halle), Stefan Liersch (Berlin), Antje Ullrich (Halle), PD Dr. Carsten Lorz (Dresden), Martin Steinert (Leipzig), Prof. Dr. Karsten Schulz (Munich) and Dr. Leo Lymburner (Canberra, Australia) worked with me on several projects together; I am grateful for their discussions and support in the presented work by means of joint papers. Last but not least I would like to thank Ellen and Cornelius for their support and patience during recent years.

Naumburg, October 2008

CONTENTS

1. INTRODUCTION AND OVERVIEW

Extreme events such as floods, droughts, as well as water scarcity and poor water quality have been increasing globally during recent decades. Global change phenomena, increasing population density in some parts of the world, as well as multiple land use of landscapes such as agricultural management, urbanisation, and industrialization are some of the main reasons for these problems (Falkenmark and Rockström, 2004). Both, the mentioned reasons as well as the resulting environmental consequences, represent some of the world's most pressing problems (Cabrera et al., 2008). In recent decades, integrated river basin and environmental management has been introduced as a potential but challenging instrument to tackle these complex transdisciplinary problems around the world. However, several problems still exist before an effective, integrated river basin and environmental management can be realized (Fohrer, 2005; Sullivan and Meigh, 2007; Brouwer and Hofkes, 2008). The motivation for this work is to contribute to the solution for some of these (methodological but also environmental) problems within a developed "open" methodological framework. In the following, the existing problems and needs of the handled topics are briefly argued,and my work on the solutions to these problems, which provides additionally an overview on the structure of the thesis, is shortly illustrated.

Theory and practice - Definition of scales relevant for environmental planning and management

Environmental problems occur on different spatio-temporal scales, from the field to the region, and even globally (Steinhardt and Volk, 1999; Urban, 2005; Bastian et al., 2006; Hein et al., 2006). The dominance of processes and their controlling factors change with these different scales (Blöschl, 1996; Blöschl, 2004). There are several theoretical considerations and small scale experiments on scale variance and scale invariance of processes in landscapes. Some of these theories are either not proven or were proven only for small scales. They are certainly essential for improving process understanding, but their findings can mostly only be used indirectly for landscape and river basin analysis and management (Bierkens et al., 2000; Herrmann, 2001; Quinn, 2004). For instance, for implementation of measures to improve water quality and water quality in the range of sub-basins to large river basins, more pragmatic approaches are needed (Jessel and Jacobs, 2005). Bierkens et al. (2000), for instance, state that 'philosophical' (very difficult and complex), system approaches have to be simplified for application to environmental planning. Here, a combination with more practical approaches is conceivable and should be aimed at. What is needed is a determination of the minimum scale at which given acts can effectively become part of a balance-seeking act (Yanarella and Levine, 1992; Herrmann, 2001). From the perspective of planning practice, a landscape (with defined ranges) represents the suitable scale for the planning of sustainable development, because it assumes an intermediate position that is close to both the local and the national decision-making level (Briassoulis, 1999). Nevertheless, such implementations should be based on theoretical findings and experiences, such as those gained from hierarchy and ecosystem theory (Müller, 1992; Klijn, 1997; Ravetz, 1998; Wu, 1999).

Hence, at first a compilation and close examination is needed for the most important landscape definitions and scale theories in landscape research, taken from, for example, geography, system sciences, landscape ecology and hydrology, which are presented in Chapter 2. On the basis of these theoretical findings, scale levels relevant for spatial planning and river basin management were defined. The work is based on the assumption that each scale level i) has its own specific dominant processes and controlling factors, ii) needs its scale specific methods and data layer, and finally iii) has its own specific information level. The Chapter includes initial overviews and tests for selected models on defined scales.

Which method and dataset on which scale? The selection of scale appropriate methods and data

Hence, the accurate assessment of the impact of land use and management on water and nutrient fluxes has to comprise a range of scales: from the landscape dependent differentiation of water and nutrient balance in large river basins, to the quantification of these fluxes in medium-sized basins and sub-basins, and finally to detailed measure planning and efficiency control in smaller catchments and assessment units (Wu and Hobbs, 2002; Drewry et al, 2006; Boardman, 2006; Sullivan and Meigh, 2007). Because one method (or model) alone cannot fulfil these requirements, the suitability of different models and data for these levels has to be tested for this procedure (Quinn, 2004). This requires also consideration of the sensitivity of the methods to input data variation as well as accuracy of both the methods and data used for calibration and validation (Beven, 2002; Jha et al., 2004; Romanowicz et al., 2005; White and Chaubey, 2005; Harmel et al., 2006). Jakeman et al. (2006) state that "Best (model) practice entails identifying clearly the clients and objectives of the modelling exercise; documenting the nature (quantity, quality, limitations) of the data used to construct and test the model; providing a strong rationale for the choice of model family and features (encompassing review of alternative approaches); justifying the techniques used to calibrate the model; serious analysis, testing and discussion of model performance; and making a resultant statement of model assumptions, utility, accuracy, limitations, and scope for improvement."

The work that I have done in this field is presented in Chapter 3. I worked intensively on investigating the capabilities of different methods and models that are used to simulate water and nutrient fluxes on defined planning- and management-relevant scales, including the development of a method to quantify the proportion of tile-drained land as a basis for the simulation of related fluxes of water, nitrogen, phosphorus, and pesticides that are transported by the drainage water. A hierarchical approach to analyse the impact of land use on water and matter balance on micro-, meso- and macro-scales by using selected models is suggested. The methodology was implemented in the FLUMAGIS project (cp. Table 1),

where the transfer of information and thus the linkage between the scales was carried out by means of indicators (Sections 3.2.4 and 5.2.2).
With regard to the above mentioned statement of Jakeman (2006), Chapter 3 also comprises my investigations of i) the impact of the uncertainty of monitoring data on model calibration and validation, ii) the sensitivity of models (ABIMO[1] and SWAT[2]) on varying input data and parameters, and iii) the test of the suitability and limitations of different variants of the universal soil loss equation (USLE). Suggestions were made for a simple model to survey groundwater levels by using remote sensing, as well as for an improvement of the SWAT model with regard to better spatial distribution of water and nutrient transport processes and land management options. This work is done in close co-operation with the USDA-ARS laboratories in Temple, TX, USA and Tifton, GA, USA and the Baylor University in Waco, TX, USA.

Water quantity and quality simulations

Land use is the parameter by which society controls the landscape water balance (Calder et al., 1979). The investigation of the effect of land use change and land use patterns on hydrologic processes such as groundwater recharge or surface runoff provides information for the development of sustainable land use concepts and integrated environmental and river basin management as demanded by the landscape programmes or the European Water Framework Directive (WFD; European Commission, 2000). There are numerous examples for studies described in literature where hydrological models are applied successfully for the simulation of the influence of land use changes on potential groundwater recharge or streamflow. Mostly variants or scenarios are investigated that are based on assumptions of climatic change or of the influence of political decisions. When simulating water balances for large areas, only considerable land use changes result in noteworthy shifts of the simulated total runoff. Depending on the selected model type, database and regional site conditions of the study site, land use changes can affect the results (and also the further modelling) to varying degrees. The assumptions of land use changes can be made on the basis of plausibility considerations, or models can be applied (Fohrer et al., 1999). Fohrer et al. (2003) and Hörmann et al. (2005) give an overview of the prospects and limitations of eco-hydrological models for evaluation of land use options in mesoscale catchments. The papers that they analysed are classified into three categories: analysis of consequences of observed (or historical) land use changes on the water cycle, simulation of hydrologic consequences of land use change based on ecological and mainly economic scenarios, and finally optimization of land use according to economic or ecological criteria, where both components, hydrology and land use change, are simulated. The papers presented in Chapter 4 can be assigned to the last two categories. The studies of Section 4.2 include examples for the simulation of the impact of land use on water availability. The work was carried out in different projects for decision-making support in spatial planning and water

[1] ABIMO is an acrynom for **Ab**fluss**bi**ldungs**mo**dell (runoff- simulation model; Glugla and Fürtig, 1997). See also Chapter 2.
[2] SWAT is an acronym for **S**oil and **W**ater **A**ssessment **T**ool (Arnold et al., 1998). See also Chapter 2.

and natural-resource management, which document the suitability of the methods. The projects dealt with recommendations for water protection (priority areas for drinking water abstraction; seeping water quality) in landscape management, and natural-resource protection and economic development.

In much of Europe, the United States and parts of Australia, increased inputs of especially nitrogen and phosphorus to land in the form of fertilisers, manures and biosolids means that agricultural runoff now comprises a greater share of these nutrients in groundwater, rivers and lakes and associated water quality problems (Heathwaite, 2003; Neal and Heathwaite, 2005). Numerous site-specific field studies have quantified the potential export of nutrients in agricultural runoff. But it is clear that to meet the requirements of end-users, the research effort needs to shift towards developing suitable models that are based on expert knowledge to simulate the impact of land use and land management practices on diffuse source pollutant transport. The papers assigned to Sections 4.3.1 and 4.3.2 include examples for the application of two different model systems (the process-based carbon-and-nitrogen-dynamics models CANDY[3] and the (balance-oriented) whole-farm simulation model REPRO[4]) to determine land usage variants which, employing the regional regulation potential, lead to a reduction of nutrient outputs into neighbouring ecosystems. The work shows clearly the influence of the quality and availability of data on the simulation results, whereas the model-structure seemed to have less influence in this case study.

The experiences of different European and national projects dealing with the model-supported implementation of the WFD revealed that the available *integrated* model systems are still far from being suitable for operational applications with regard to water quality simulations (Horn et al., 2004; Payraudeau et al., 2004; Euroharp-Project, 2007). However, complex trans-disciplinary problems have to be tackled through implementation of WFD or other environmental directives. Hence, in order to achieve optimal working efficiency of the models in the management processes, it is required that they contribute information over a wide range of abiotic and biotic aspects of hydrology and water quality demanded by the decision makers, which cannot be achieved by individual groundwater, water quality or erosion models. Consequently, I checked the suitability of the publicly-available river basin model Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998), the available database and the existing water quality monitoring network to adequately represent general trends in water quality changes resulting from various measures based on land use and management change in the intensively used Upper Ems Basin (Germany) (Section 4.3.3)..

[3] CANDY is an acronym for **C**arbon-**a**nd-**N**itrogen-**Dy**namics (Franko et al., 1995; Franko, 1996). See also Chapter 2.
[4] REPRO is an acronym for Reproduction of Soil Fertility (Hülsbergen, 2003). See also Chapter 4.

Scale appropriate application of models for integrated ecological-economic assessments

Integrated river basin and environmental management involve all management objectives related to the use, pollution mitigation, pollution rehabilitation, protection and rehabilitation of water bodies as well as many other impacts on biodiversity, soil protection, water quantity and quality in river basins or administrative units (Berlekamp et al., 2007). An integrated approach implies that relations between the abiotic and the biotic part of the various water systems, between ecological and economic factors and between various stakeholder interests are considered in decision-making processes. It takes into account three often conflicting main dimensions: ecology, economy and equity (Hirschfeld et al., 2005). Directives such as the WFD, the NATURA2000 in Europe or Total Daily Maximum Loads (TDML) in the US call for multidisciplinary approaches of river basin and environmental management. Hence, during recent decades a number of research projects have developed (spatial) decision-making support systems for integrated water resources and environmental management (including socioeconomic analyses) with respect to groundwater management and flood prevention problems (Mödinger et al., 2004; Möltgen and Petry, 2004; Schneck et al., 2004; Feld et al., 2005; Hirschfeld et al., 2005; Giuponni 2007; Berlekamp et al., 2007; Van Delden et al., 2007). Most of them offer the possibility of drawing information from geographical information systems and/or supplying interdisciplinary multi-criteria analyses of the hydrological, ecological and economic consequences of different management strategies, based on either pre-calculated scenarios or model coupling (Lautenbach et al., 2009). Hirschfeld et al. (2005), Giupponi (2007), Van Delden et al. (2007), Burstein et al. (2008) and Lautenbach et al. (2009) give examples and overviews of the development, application and potential of different types of DSS. Giupponi (2007) states that despite the many DSSs developed in the field of environmental management, the risk of Decision Support Systems failing to meet the challenge of real-world problems is reported to be high, and even the criteria for judging whether a DSS has been successful or not are often a matter of discussion (e.g. Newman et al., 1999; Zapatero, 1996, Uran and Janssen, 2003). He emphasizes that there is a widely-recognised need to develop new decision support tools in this field, with greater attention to the needs of potential users and to identification of the application context. Hence, Chapter 5 includes my work on two different Decision Support Systems, in which several research institutions and relevant planning authorities were involved in examining the needs and providing such an application context. Section 5.2.1 presents an example for integrated modelling of nitrogen transport within the 4th order Weisse Elster River (Weisse Elster project, Table 1), which comprises the combined terrestrial and in-stream transport processes. Section 5.2.2 presents the innovative spatial decision support system (SDSS) approach from the FLUMAGIS project (cp. Table 1) which is based on the integration of methods for ecological and socio-economic assessment, scale-specific modelling, knowledge processing and techniques for visualization. In this project, most of the methodology developed in this thesis has been implemented. The project has developed an interactive tool for the assessment and (three-dimensional) visualization of the hydrological and ecological conditions in river basins and economic aspects of river basin management measures. The innovative tool is designed to increase awareness of catchment scale hydrological and ecological issues on different scales.

Finally, Chapter 6 includes the conclusions and fields of the presented topics that will, in my opinion, be of great importance in the near future.

Figure 1 provides an overview over the developed methodological framework. The projects, co-operations, supervised diploma theses and dissertations that resulted in the presented publications are listed in Table 1. The work was carried out in study areas with sizes that range from 0.5 to 23,000 km², in order to cover all scales relevant for environmental and river basin management.

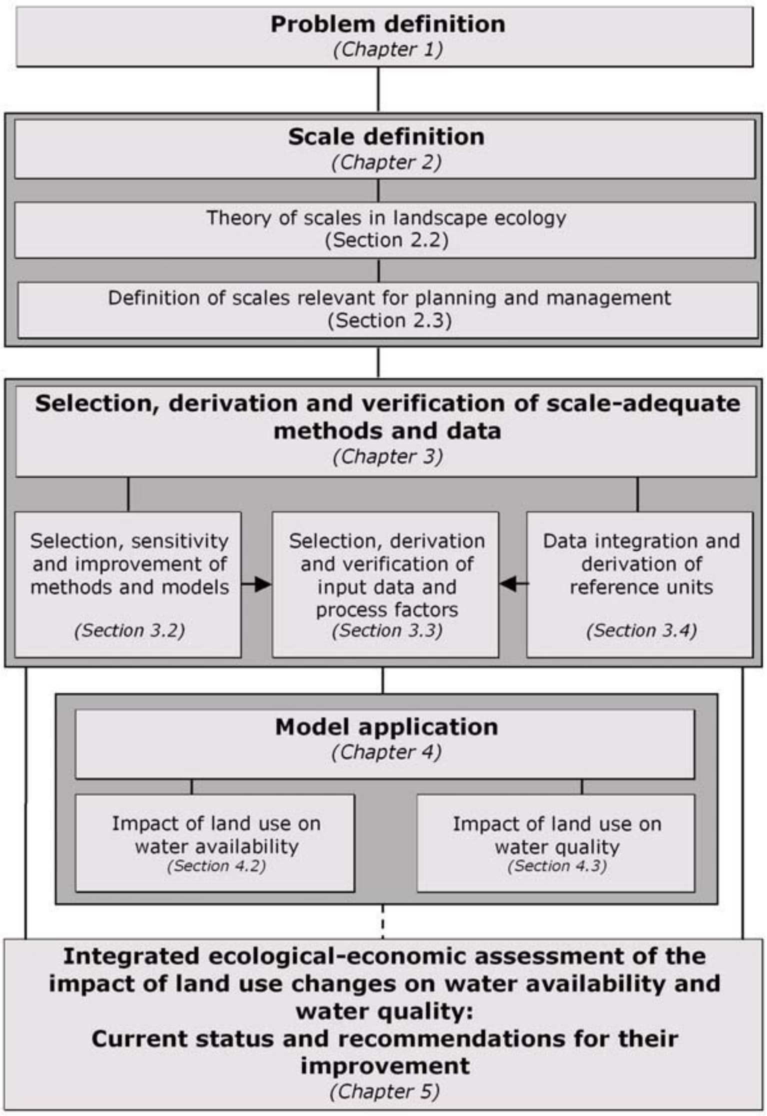

Figure 1. A methodological framework for scale appropriate analysis, assessment and management of landscape water and matter dynamics.

Table 1. Information about the projects where the work was done that resulted in the presented publications. Detailed information about these projects is provided in appendix A1.

Project	Duration	Objective	Study area	Study area (km²)	Section / Publication
Dessau	1995-2000	Regional landscape development	Dessau district (Germany)	4,300	2.3.1, 2.3.2; 3.2.2, 3.3.1, 3.3.3; 4.2.1
Torgau	1997-2001	Integrated ecological-economic assessment	Torgau region (Germany)	686	4.2.2, 4.3.1, 4.3.2
Elbe	1998-2002	Integrated River basin management	Parthe watershed and Schnellbach sub-basin (Germany)	315 8	2.3.1, 2.3.2; 3.2.2, 3.3.1, 3.3.3
Flumagis	2002-2005	Integrated River basin management (ecological-economical)	Upper Ems River Basin and sub-basins (Germany)	3,740 350 160 14	3.2.5; 4.3.3; 5.2.2;
Weisse Elster	2002-2005	Integrated River basin management (ecological-economical)	Weisse Elster watershed (Germany)	5,300	5.2.1
Co-operation USDA/Baylor (1)	2003-2006	Model development (data-based) (Groundwater level estimation)	Seco and Hondo Creek Watershed (TX, USA)	3,000	3.3.4
Co-operation USDA/Baylor (2)	2003-	Model development (SWAT)	Riesel experimental watershed (TX, USA)	0.5	3.2.9
Co-operation USDA	2003-	Model development (SWAT)	Gibbs farm experimental watershed (GA, USA)	12	3.2.8
Co-operation TU Dresden	2005-	Model development (concept)	-	-	3.2.7
Co-operation IGB	2006-	Method development (data-based) (calculation of the proportion of tile-drained areas)	Saale river basin (Germany)	23,000	3.2.4
PhD thesis M. Möller	2003-2008	Method development (landscape object segmentation)	Könnern study area (map sheet) (Germany)	100	3.4.1, 3.4.2
PhD thesis A. Ullrich	2003-2009	Monitoring data uncertainty / model sensitivity	Parthe watershed (Germany)	315	3.2.6

2. SCALES IN LANDSCAPE ECOLOGY: THEORY UND DEFINITION OF SCALES RELEVANT FOR PLANNING AND MANAGEMENT

2.1 Overview

Chapter 2 covers the theoretical basis for the development of the presented methodology, which comprises the definition of the term “landscape” (Volk and Steinhardt, 2002a; Section 2.2.1) and an overview on scale theory (Steinhardt and Volk, 2001; Section 2.2.2), which leads to the definition of relevant scales and a first grouping of methods and models relevant for spatial planning and management (Volk and Steinhardt, 2001, Section 2.3.1, and Steinhardt and Volk, 2002, Section 2.3.2).

At the beginning of integrated projects that deal with management options for river basins or regions to find suitable measures for soil and water protection in a socio-economic assessment framework, clear definitions of i) basic terms such as “landscape”, which we investigate, and ii) the relevant scales to be considered in the study have to be determined in order to generate reasonable results (due to the objectives of the studies) and to avoid any confusion with the project partners from research and planning practice. The publications assigned to this section reflect parts of my work in the Elbe and Dessau project (cp. Table 1). The work has been published in three book chapters and in Steinhardt and Volk (2002).

Regarding the definition of the study object “landscape”, Volk and Steinhardt (2002a; Section 2.2.1) evaluate the related discussions in landscape research. It was published in a Chapter of the book “*Development and Perspectives in Landscape Ecology*” edited by Bastian and Steinhardt (2002). Most simply, Turner and Gardner (1991) considered a landscape to be a spatially heterogeneous area. In a similar vein to the ideas of Haase et al. (1991), Forman and Godron (1986) suggest three landscape characteristics that are useful to consider when thinking about landscape: structure, function, and change. "**Structure** refers to the spatial relationships between distinctive ecosystems, that is, the distribution of energy, materials, and species in relation to the sizes, shapes, numbers, kinds, and configurations of components. **Function** refers to the interactions between the spatial elements, that is, the flow of energy, materials, and organisms among the component ecosystems. **Change** refers to alteration in the structure and function of the ecological mosaic through time" (Turner and Gardner, 1991). By evaluating the studies of several authors who have worked in this field, Volk and Steinhardt (2002a; Section 2.2.1) conclude with seven statements in order to handle the term "landscape" pragmatically in applied research and land and river basin management.

The contribution of Steinhardt and Volk (2001; Section 2.2.2) contains some of my research on scales and dimensions in landscape ecology, which includes the transformation, aggregation and disaggregation of landscape information. Several studies in the environmental sciences deal with the hierarchical organization of ecosystems (Müller, 1992; Klijn, 1995; Wu, 1999). Unfortunately, in landscape ecology, treatments of hierarchies were mostly limited to consideration of the different spatio-temporal resolution of the basic data. In fact, the problem of scales and transformation with narrow limits is related to the compilation of the data and the choice of the indicators. The scale problem results from the transfer from one hierarchical level to another, which is true for both the "top down" and the "bottom-up" approach. Both approaches have to consider generalizations suitable for the scale, and changes to the criteria of homogeneity. According to Herz (1973) homogeneity can be achieved at each level of consideration by agglomeration or generalization. An increasing hierarchical order is often accompanied by an increase in heterogeneity. Thus, during studies it should be made clear whether mean values or data of dominant conditions and processes have been used, or whether features of heterogeneity (frequency, minima and maxima, variance, etc.) are also considered.
The crucial point is to define scale-specific processes: what processes act at which scale? If this is known, we can derive the data necessary to describe these processes. Of course, the problem still remains of whether these data are actually available (cp. Chapter 3 and Volk et al., 2008; Section 4.3.3).
The choice of hierarchical level depends on the question formulation in science or environmental management. At any hierarchical level, the arrangement and classification of the landscape can be made using its components such as soil, climate and relief. This method can include the whole nature complex abstracting from land use within the order of natural areas or classification. Beside the nature complex, land use is considered as either the feature in landscape classifications or the delineation of landscape units. These spatial units are organized in spatiotemporal hierarchies, which can be approached at micro-, meso- and macroscale levels (Steinhardt, 1999; Steinhardt and Volk, 2000).
Based on these fundamental theories of scales and dimensions in landscape ecology, conceptual, quantitative and evaluative models have to be found to analyse critical anthropogenic and non-anthropogenic processes related to ecosystems on defined scale levels.

Volk and Steinhardt (2001; Section 2.3.1) suggest a hierarchical (scale-specific) approach in landscape research to investigate the water and matter balance on the meso-landscape scale. By continuing the findings of the results in Section 2.2, the publications discuss topics such as i) characterizing processes concerning extension, duration, intensity and continuity, ii) interactions between landscape structures and processes, fluxes matter, energy and information in landscapes, iii) scale-specific and cross scale investigations, etc., to finally conclude with suggestions for the above-mentioned hierarchical approach for water and matter balance investigations on the meso-scale. Integrated models such as SWAT (***S****oil and* ***W****ater* ***A****ssessment* ***T****ool*; Arnold et al., 1998) and ASGi (***A****bfluss und* ***S****tofftransport - integrierte Modellierung unter Nutzung von* ***G****eo****i****nformationssystemen*) which consists of the models AGNPS (***AG****ricultural* ***N****on-****P****oint* ***S****ource Pollution Model*; Young et al, 1987) and the runoff simulation model Wasim-ETH (Schulla, 1997), the runoff simulation model ABIMO (in German: *Abfluss****bi****ldungs****mo****dell*; Glugla and Fürtig, 1997), the carbon and nitrogen simulation model CANDY (***C****arbon-****a****nd-****N****itrogen-****Dy****namics*; Franko et al., 1995, Franko, 1996) and the erosion simulation models EROSION2D/3D (Schmidt, 1991; von Werner, 1995) and USLE (***U****niversal* ***S****oil* ***L****oss* ***E****quation*; Wishmeier and Smith, 1978) are listed here and grouped according to their scale-specific applicability. Initial results of model applications (ABIMO, USLE and EROSION3D) and land use scenarios in the suggested hierarchical framework are presented here: Methodological steps for the hierarchical, scale-specific landscape analysis of the landscape balance are suggested and shown in exemplary studies in the Dessau district in the German State of Saxony-Anhalt and the Parthe watershed in the State of Saxony. Steinhardt and Volk (2002; Section 2.3.2) enhanced this scale-specific approach for landscape balance investigation and evaluation by more specific suggestions for integrating the assessment results into landscape planning. Three (flexible) ranges of scales have been defined (1:10,000 to 1:25,000; 1:25,000 to 1:50,000 and 1:50,000 and larger). Spatial planning levels, assessment units, databases (spatial resolutions), models and information level and application possibilities have been assigned to these three levels. With this suggested approach, the sections 2.3.1 and 2.3.2 form the basis for the following studies and the further development of the methodological framework.

2.2 Theories on scales in landscape ecology and landscape-related research

2.2.1 (Reprint of the paper at Appendix A2.1)

Volk, M., and Steinhardt, U., 2002a. The landscape concept. What is a landscape? In: Bastian, O., Steinhardt, U. (eds.). *Development and Perspectives of Landscape Ecology*. Kluwer Academic Publishers, Dordrecht, 1-9.

Extended Summary[5]

Considering the various answers that have been given to the question: "What is a landscape?", some general statements can made, for all disciplines of landscape ecology (the following statements are in accordance with a compilation of Forman and Godron 1986, Hansen and Di Castri 1992, Klijn 1995, Turner 1987, Urban et al. 1987, Zonneveld and Forman 1990):

[5] There is no summary available for this edited book chapter. Therefore the main findings of the chapter are described shortly here.

- Landscapes are nearly always the result of both natural and man-induced processes during, nearly always, various time-scales. Landscapes can effectively be described as palimpsests, **patterns superimposed on each other**, showing features of different eras. These legacies affect present day and future processes.
- **Landscapes are changing**, but changes occur at different rates, either gradually or suddenly, even catastrophically. Landscapes that are stable for a long period are almost fiction.
- Nevertheless there are **stabilizing forces within landscapes**: disturbances are followed by a return to a former status or by a new equilibrium, both in a physico-chemical and in a biological sense.
- Although landscape dynamics show many unexpected or unexplainable phenomena, there is still a **large portion of predictable change** such as primary or secondary succession or degradation stages.
- Landscapes are mainly **open systems**: open to vertical influences (e.g. radiation, atmosphere), open to influences from their surroundings and internally open (exchange between patches within one landscape). Landscapes can be understood by insight into the **flows of matter, energy and organisms**.
- **Landscapes are heterogeneous**, both in a vertical and horizontal direction. Vertically one can distinguish layers (atmosphere, canopy, soil, groundwater, rock, etc.). Horizontally, landscapes consist of patches (or ecotopes) with repeat themselves in a certain pattern. Between "homogenous" patches are boundaries that can be sharp or gradual. Boundaries are sometimes open to the exchange of matter, energy or organisms; they sometimes act as barriers or membranes.
- Landscapes are perceived as parts of the earth's surface with **a certain size but with uncertain lower and upper limits**. Questions are open concerning the spatio-temporal definition of landscapes, so that it is not possible to derive any standard sizes or scales. The definition depends on the priority of view.

In the authors' opinion, these statements contain all relevant characteristics (and also open questions) of (and about) landscapes and their role in landscape ecology.

2.2.2 (Reprint of the paper at Appendix A2.2)

Steinhardt, U., and Volk, M., 2001. Scales and spatio-temporal dimensions in landscape ecology. In: Krönert, R., Steinhardt, U., Volk, M. (eds.). *Landscape balance and landscape assessment.* Springer, Berlin Heidelberg-New York, 137-162.

Extended summary[6]

The human factor 'land use' affects the interactions between water, soil, geomorphology, vegetation, etc. on several spatial and temporal scales in different manners and intensities. The implementation of strategies for sustainable land use assumes specific research concepts from the local to the global scale (micro-, meso- and macroscale). Therefore, landscape ecology science has

[6] There is no summary available for this edited book chapter. Therefore the main findings of the chapter are described shortly here.

to provide investigation methods for all these different scales. A number of papers from different scientific disciplines deal with the hierarchical organization of nature (Burns et al. 1991, O'Neill et al. 1986). The hierarchical concept was introduced into German landscape ecology by Neef (1963, 1967) and continued by several other landscape ecologists (Leser 1997). An overview of hierarchical concepts in landscape ecology is given by Klijn (1995). These concepts are mainly focused on the hypothesis, that each of the scale levels (micro-, meso- and macroscale) is characterized by specific temporal and spatial ranges. As a consequence, each scale level needs specific investigation methods as well as data layers with suitable spatio-temporal resolution on the one hand, and which provide specific knowledge on the other (Steinhardt and Volk, 2000).

Due to the increased application of GIS over the past few years, this is often reduced to the spatial resolution of the data layers. This paper stresses the necessity of considering scale-specific investigation methods in landscape ecological research. In connection with this, the difficult question of regionalization is treated. Several examples will be given of proposals for considering scales and spatio-temporal dimensions in landscape research, as well as of scale-specific problems within process-oriented or structurally oriented investigations. One of the main topics is the definition of a linkage between the different scales. The authors present a hierarchical approach, their main hypothesis being that the basic components for most landscape-ecological processes are similar at all scale levels. It is only the importance of the factors (and the factors themselfs) which changes for each scale and have to be defined (Helming and Frielinghaus, 1999; Steinhardt and Volk, 2000; Volk 1999).

2.3 Definition of scales relevant for planning and management

2.3.1 (Reprint of the paper at Appendix A2.3)

Volk, M., and Steinhardt, U., 2001. Landscape balance. In: Krönert, R., Steinhardt and Volk, M. (eds.). *Landscape balance and landscape assessment.* Springer, Berlin Heidelberg-New York, 163-202.

Extended Summary[7]

The characteristic distribution of the landscape's components land use, land cover, soil, morphology, hydrology, climate, geology, etc., forms the landscape structure. These components are interrelated by fluxes of water, material, energy and information (landscape-ecological processes), which result in the 'landscape balance'. This term is based on the German concept of the *Landschaftshaushalt*, which describes the associations between the geoecofactors in a geoecosystem due to the laws of nature (Leser 1997, Marks et al. 1992, Troll 1939, Zepp & Müller 1999). Schmithüsen (1973) transferred the theories and considerations of thermodynamics and synergism into geography with the term 'geosynergetic landscape research', which describes the totality of all interactions within a landscape (cf. also Müller 1999). Neef (e.g. 1973) also largely developed the system theories for 'his' landscape research on the basis of such knowledge. The geoecosystem is regarded as an 'open system' characterized by an equilibrium of flows, with input and output interactions with the landscape balances of the adjacent geoecosystems (the

[7] There is no summary available for this edited book chapter. Therefore the main findings of the chapter are described shortly here.

environment). Despite the dimension of the landscape ecosystem, a model of the landscape balance can be created for any order of magnitude. In doing so, methodological extensions or limitations arise for the different dimension steps. Several problems have emerged with the development of scale-specific methods, the improvement of knowledge about the interactions between landscape structure and landscape-ecological processes and the processual interactions and changes within the landscape ecosystem itself at different dimensions and scales. These questions become even more important when considering the impact of land use and its changes on the landscape balance and its assessment as a basis for a sustainable development.
Human impacts - such as land use - affect the interactions within a landscape ecosystem by changing the landscape structure and thus altering conditions for landscape-ecological processes. The human factor 'land use' within the complex ecosystem has a strong impact on the adaptability, regeneration and regulation capability of the landscape balance. It should be mentioned in this context that it is still a problem to assess the adaptability and dynamics of the landscape balance as a reaction to human impacts (feedbacks) within landscape analysis owing to the lack of knowledge about these interactions (Fig. 7.1 and Fig. 7.2). As most of the relevant processes in the landscape depend mainly on the mobile agent water, they have influences ranging from small to large scales. However, understanding of these processes - especially on large scales - is still insufficient, as most of the processes take place on small scales. Concepts for sustainable development have to consider the implementation of information about the landscape balance on all scale levels. Special attention should be paid to larger scales because most of the environmental conflicts and changes become apparent on the landscape scale. However, most of the useful methods for the analysis and assessment of landscape ecological processes and parameters are limited to scales up to 1:25,000, and the importance of the parameters - and the parameters themselves - are limited to changes in a hierarchical spatio-temporal way.
To solve these problems, the following questions should be asked:

- How does the importance of parameters (as well as the parameters themselves)of their landscape balance components (morphology, soil, hydrology, soil, hydrology, land use and cover and climate) change on different scales?
- How does the impact of changes to the landscape structure (especially land use) affect the water, material, energy and information fluxes (horizontal and vertical) on different scales?
- How does the land use influence the quality and quantity of soil and water?

This also requires characterizing the processes concerning extension, duration, intensity and continuity – and improving knowledge about possible feedback. Examples on the complex interactions of the landscape balance within the landscape system and the problems of its investigation and assessment (as well as an example for positive feedback) are presented in this contribution.

In this paper, several national and international approaches and models for these investigations are presented. Finally, our hierarchical approach is described for mesoscale application. In addition, suggestions are made for the verification of large scale calculations. For integrated landscape analysis, we aim to combine both 'top-down' and 'bottom-up' approaches with GIS-coupled model applications and traditional methods (e.g. mapping, measuring, etc). Using traditional methods is an essential part of verifying modelling results, as well as for improving knowledge of how landscape ecosystems function.

2.3.2 (Reprint of the paper at Appendix A2.4)

Steinhardt, U. and Volk, M., 2002. The investigation of water and matter balance on the meso-landscape scale: A hierarchical approach for landscape research. Landscape Ecology 17(1), 1-12.

Summary

The realization of strategies for sustainable land use assumes specific research concepts from the local to the global scale (micro-, meso- and macroscale). Therefore, landscape ecological science has to provide investigation methods for all these different scales. By combining "top-down" and "bottom-up" approaches in addition to coupled GIS-model applications and traditional methods, the investigation of landscape ecological structures and processes seems to be possible. The presented studies show this approach on examples of two study areas in Eastern Germany: A watershed of 400 km² and an administrative district of about 4000 km². The scale-specific applicability of several models and methods were tested for these investigations, and the validation of the calculated results is presented. An important outcome of the project should be the prevention of conflicts between agriculture, water management and soil, and water and nature conservation; based on recommendations for land use variants with decreased pollutant loading within agricultural areas. The scale specific investigations can be considered as a base for establishing sustainable land use.

3. SELECTION, DERIVATION AND VERIFICATION OF SCALE APPROPRIATE METHODS AND DATA

3.1 Overview

Scale appropriate simulation of nutrient fluxes and balances is necessary, because structures, functions and processes change with scale (Blöschl and Sivapalan, 1995; Quinn, 2004; Jessel and Jacobs, 2005; Hein et al., 2006). The scale appropriate selection and testing of models, datasets and assessment units play an essential role for a sound river basin and environmental management. Water balance and water quality components, as well as soil erosion calculated for meso- and macro-scale river basins or administrative units depend strongly on the accuracy of ground data and on the spatial distribution (and accuracy) of measured input variables (Lahmer *et al.*, 1999). The applicability of the models is restricted by the lack of suitable data for the different scales, which Wu and Hobbs (2002) consider as a main problem in landscape-ecological research and analysis. Input data and simulation results are components of a multi-functional system (Molenaar, 1998; Wielemaker et al., 2001; de Vente and Poesen, 2005). Each functional hierarchy consists of specific reference units that can be of administrative or natural type (e.g. administrative units, river basins, or terrain units). Landscape assessments determine a semantic and geometric integration of the reference units and of the assessment's subject matter respectively, which can lead to uncertainties in the assessment results (Openshaw and Taylor, 1981; Jelinski and Wu, 1996; Malczewski, 1999; Marceau, 1999b; Mysiak et al., 2004).

However, meso- to macro-scale studies often suffer from restrictions induced by the availability of data. For instance, the lack of long time series of water quality data with daily time step and higher spatial resolution limit our capacity to evaluate water quality simulations – which represents a general problem and results in uncertainty. In addition, the existing monitoring programs for water quality in Europe are not suitable yet to deliver a sound database for the simulations (Jarvie et al., 1997; Rekolainen et al., 2003; EEB and WWF, 2005; Allan et al., 2006). But the accurate calibration and evaluation of the models need an appropriate data base in the form of monitoring data. Moreover, uncertainties in the monitoring data such as load estimations influence the calibration and thus the parameter settings which affect the modelling results. Hence, comparisons of different time-based sampling strategies and different load estimation methods for model calibration are needed to optimize water quality simulations and to establish a good calibration base for simulation models.

With regard to the above mentioned topics, Chapter 3 comprises the results on my work on the selection, derivation and verification of scale appropriate methods and data. The section starts with an evaluation and selection of scale appropriate methods and models used in river basin management (Volk and Steinhardt, 2002b; Section 3.2.1) and introduces an enhanced hierarchical approach to analyse the impact of land use on water and matter balance on the meso-scale by using these different models (Steinhardt and Volk, 2003; Section 3.2.2; according to the findings of Chapter 2). Many different aspects of nonpoint- and point-source pollution have to be considered with the model-based analysis of the complex interrelated causes for problems of water availability and poor water quality. These aspects include also questions of biodiversity, nature, and soil protection. The analyses have to comprise also studies on the sensitivity of these models (Ullrich and Volk, submitted; Section 3.2.5; Ullrich et al. 2007; Section 3.2.6) and on the improvement if their spatial distribution of processes (Bosch et al., 2007; Section 3.2.8; Volk et al., 2007; Section 3.2.9). Another problem on the landscape scale is the quantification of nutrient leaching from tile drained land. Nutrient leaching from tile drainage systems combines nonpoint- and point-source pollution problems. Hirt and Volk (under revision; Section 3.2.3) present a solution for this problem by means of a method that is based on using agricultural statistics, and both representative soil physical data and landscape characteristics.

Models, scales and indicators

Integrated models play an increasing role in river basin management of medium to large river basins. They are able to describe the impact of land use and land management on water and nutrient fluxes. Beside the description of land management practices and its impact on nitrogen transport, and soil erosion, it is possible to include also point sources and/or water management aspects. SWAT is one example for such modular models. The model has been used successfully throughout the world (Arnold and Fohrer, 2005; Gassman et al., 2007). It was already identified as one of the potential models for the scale appropriate analysis and assessment of water and matter dynamics in medium to large river basins. Volk and Schmidt (2004; Section 3.2.4) specified the scale levels for river basin management on the basis of the work presented in the first four publications. Models are selected and tested on their suitability for these levels. The transfer of information and thus the linkage is carried out here by means of indicators. This concept was used throughout the FLUMAGIS project

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Model sensitivity and monitoring data uncertainty

Using models in integrated river basin analysis requires the impact of uncertainty of model parametrisation, but also of monitoring data on model calibration and evaluation. This point is tackled in Ullrich and Volk (submitted; Section 3.2.5) and Ullrich et al. (2008; Section 3.2.6) on the example of studies in our experimental watershed, the Parthe watershed in Saxony, Germany. Here, the results of studies of the influence of i) different land management parametrisations on calculated water and nutrient fluxes, and of ii) sampling strategies, and different load estimation techniques on SWAT model calibration and evaluation is presented. Knowledge about the model sensitivity to land management parameters would help models better simulate the effects of land management alternatives. Hence, Ullrich and Volk (submitted; Section 3.2.5) carried out a sensitivity analysis for conservation management parameters (specifically tillage depth, mechanical soil mixing efficiency, biological soil mixing efficiency, curve number, Manning´s roughness co-efficient for overland flow, USLE support practice factor, and filter strip width) in SWAT. Based on the results of our analysis the following sensitivity ranking can be concluded:

1) Duration of vegetation period and soil cover over time with
   - 1a) implementation of catch crop;
   - 1b) dates of planting (winter/spring crop);
   - 1c) date of first tillage operation applied after harvesting (fall tillage/spring tillage)
2) Soil cover characteristics of applied crops (e.g. grains/row crops);
3) Conservation support practices (contouring) and filter stripes
4) Tillage intensity (means applied tillage practice; basic scenarios);

We consider this ranking as a first recommendation for the parameterisation of tillage operations and management practices for SWAT users and for our further studies - always with a view to the initial conditions of input data.

In addition to the management parameter settings, uncertainties in monitoring data influence the calibration and thus the parameter settings which affect the modelling results. Hence, we compared three different time-based sampling strategies and four different load estimation methods and analysed the influence on SWAT model calibration and simulation results. The Nitrate-N load estimation results differ considerably depending on sampling strategy, used load estimation method and period of interest. The load estimation results for the daily composite data set showed the lowest ranges (14% and 2% maximum deviation related to the mean value of all applied methods). Estimation results for the submonthly and the monthly data set vary in greater ranges (between 25% and 52%). We calculated the percentage deviation of mean load estimations of sub-monthly and monthly data sets related to the mean estimation value of composite data set to show differences between sampling strategies: The maximum deviation of 82% occurs for the sub-monthly data set in 2000. This affects the model and leads to different parameter settings in model calibration and evaluation. We recommend both the implementation of optimised monitoring programs and the use of more than one load estimation method to describe the water quality situation in a better way and to establish a good calibration base for simulation models.

Improving spatially distributed simulation of landscape processes

The implementation of environmental measures for water and soil protection in river basins requires spatially explicit simulation of landscape processes. Most of the models used for medium to large basins are lumped models, where landscape positions and characteristics are not sufficiently taken into account. A related problem is that during recent years it has become obvious that the influence of forests on water and nutrient balance has to be taken into account more in river basin management. Hence, Lorz et al.

(2007; Section 3.2.7), Bosch et al. (2007; Section 3.2.8) and Volk et al. (2007; Section 3.2.9) work on the improvement of the SWAT model. Lorz et al. (2007; Section 3.2.7) consider spatial distribution and functionality of forests in a modeling framework for river basin management. In some regions the influence of forests on water and nutrient fluxes within the land use pattern of river basins can play an important role. Because most of the existing models have more of an agricultural or water management background, the forest process is described only moderate to poor. Hence, this work gives some recommendations for how to better integrate forest processes into the models. We state that the most promising approaches in the future are either spatial explicit models or integrated models with both improved forest modules and landscape positioning. The efficiency of these models could be proved by using virtual catchments. As a first conceptual approach towards the base concept of a virtual catchment, we propose a five-unit-model (FUM), representing cross-sections with typical land use sequences. The basic idea of our model is to identify major process units and to implement them in the river basin modeling and management.

The papers of Bosch et al. (2007; Section 3.2.8) and Volk et al. (2007; Section 3.2.9) document my work and co-operation with the USDA-ARS in Temple, TX, and Tifton, GA, as well as with Baylor University Waco, TX, on the integration of landscape positions into the SWAT model. Watershed configuration for SWAT currently consists of: 1) subbasins defined by surface topography and 2) hydrologic response units in each sub-basin to account for heterogeneity in soils and land use. The hydrologic response units do not account for landscape position within the sub-basin. In an attempt to account for landscape position and processes, SWAT was modified to simulate landscape units (divide, hillslope, floodplain) within sub-basins. Simulated daily stream flow at the watershed outlet after routing across the landscape units, compared well to measured flow ($R^2 = 0.7$). Mean annual lateral flows across landscape units were also realistically simulated. Soil moisture (upper 1 m) was compared to measured soil moisture at one monitoring site in each landscape unit with the model predicting early drying in the summer, but following general wetting/drying cycles. The revised version of the model is also tested using data collected from a low-gradient watershed near Tifton, which contains heavily vegetated riparian buffers (Bosch et al., 2007; Section 3.2.8). The modified model provided reasonable simulations of surface and subsurface flow across the landscape positions without calibration. Future planned development includes: 1) additional testing groundwater heights at the Riesel Y2 watershed, 2) further testing the model for the USDA-ARS Gibb's Farm experimental watershed at Tifton with “classic' riparian zones within a sand soil terrain, 3) using the kinematic wave equation for overland and channel routing between landscape units, 4) incorporation of sediment and nutrient routing (and management operation schemes) across the landscape, 5) model testing on larger watersheds with defined flood plains, and 6) testing model outputs, efficiency, and run times when compared to grid based models on large watersheds (cp. our work in Arnold et al., submitted).

Selection, derivation and verification of input data and process factors
Integrated scale appropriate analysis and assessment of water and matter requires a sound database of terrain, soil, land use and land management, climate, river networks, water management, and spatial planning. The development of homogeneous data sets at different scale levels is difficult and costly. The existing data sets of the respective federal states are mostly compiled using different data management methods, which results in possible incompatibilities and hence errors when the data are put together and processed for large river catchment applications. Most of the economic farm data are available only at aggregated levels (municipalities, counties, federal states) due to confidentiality laws. It is therefore nearly impossible to assess management strategy effects on micro-scale economic and ecological conditions. There is a lack of long-term time series on water quality data (daily measurements) and of high spatial resolution, which complicates simulation evaluations. Hence, Section 3.3 includes the publications that document my work on selection, derivation and verification of input data and process factors. Volk and Steinhardt (1998; Section 3.3.1) show methods to integrate data layers from different sources compiled with different data management methods in Geographic Information Systems (GIS) for landscape ecological assessment on the example of Central Germany. The importance of suitable databases has been investigated by Petry et al. (2000; Section 3.3.2). Here, the influence of data quality in meso-scale water balance simulations is documented by using the conceptual runoff model ABIMO in two adjacent test areas with different input data. In the publication the suitability of varying data sets is discussed. Main emphasis is focuses on spatial resolution and heterogeneity, data regionalisation, and validation of modelling results. A comparison of different model versions of the ABIMO model and data sources in the Torgau region (Saxony, Germany) showed that the model version used had an insignificant effect on the result, while the data source played an important role (Herzog et al., 2001). Data quality, as well as different variants for the derivation of the model factors, can lead to faulty results. This has been illustrated by Volk, Steinhardt et al. (2001; Section 3.3.3). The influence of using different methods to derive the factors (and here especially of the rainfall factor R) of different variants of the Universal Soil Loss Equation (USLE) on the calculated results is shown comparatively for meso-scale applications. Recommendations for the application of specific variants are given dependent on the data base (temporal resolution of rainfall data, spatial resolution of the Digital Elevation Model, etc.) and the project goal. The study is a contribution towards optimised application of different, regionally adapted and scale-oriented factor variants in the USLE and integrated models.

As indicated by many studies, there is still a lack of suitable data for integrated analysis and management. Hence, alternative methods have to be found to derive relevant data. An attempt was made for predicting groundwater level by means of using NDVI (Normalized Differenced Vegetation Index) response to plant water content over a watershed located in the Edwards Aquifer of Texas, USA (Chen et al., 2006; Section 3.3.4). We could show that the NDVI values were affected by both temperature and precipitation, and the amount of rainfall was strongly correlated to the stream flow and groundwater level. The study successfully linked satellite data to groundwater levels in the artesian zone of the Seco and Hondo creek watershed, TX, USA, while more studies are required to develop a reliable approach for groundwater level estimation in the recharge zone. This study initiated a unique approach to surveying groundwater level based on satellite information and meteorological data.

Data integration and derivation of reference units

Scale appropriate analysis and assessment of water and matter dynamics is related to medium to large river basins as natural system units. Remote sensing and other digital data are used here either as input data for models or to get information about land use and land management or process-relevant landscape objects. Understandable segmentation methods are needed here that aggregate (classify) remote sensing or other digital data to provide valuable information about the size, distribution and context of landscape objects at a range of scales. However, there is a need for well defined and robust validation tools to assess the reliability of segmentation results. Hence, we developed a validation algorithm that a) enables the localization and quantification of the segmentation inaccuracies; and b) allows the assessment of segmentation results on the whole (Möller et al., 2007; Section 3.4.1).

Landforms and landscape context are particularly import for an understanding the processes of soil genesis and soil formation in the spatial domain, which is a basis for land management strategies to enhance soil and water quality protection. The classification algorithm described in Möller et al. (2008; Section 3.4.2) handles object detection and classification separately. Landscape objects are defined at multiple scales using a region-based segmentation algorithm which allows each object to be placed into a hierarchical landscape context. The classification is carried out using the terrain attribute mass-balance index across a range of scales. Soil genesis and transport processes at established field sites were used to guide the classification process. Both methods were tested in an area in the German State of Saxony-Anhalt that contains heterogeneous land surfaces and soil substrates.

Both papers are related to data integration of existing thematic information and available continuous data sets (DEM and remote sensing data). The data integration is realized by the linkage of functional hierarchies and multi-scalar object structures. The studies are focused on two aspects: The derivation of reference units and comprehensive terrain classification over all scales.

The papers presented in Chapter 3 emerged from my work in four projects, five co-operations and of two supervised theses (Table 1). The contribution of Volk and Steinhardt (2002b; Section 3.2.1) is published as a chapter of the book on "*Development and Perspectives in Landscape Ecology*" edited by Bastian and Steinhardt (2002).

3.2 Methods and models in river basin and environmental management: Selection, sensitivity and improvement

3.2.1 (Reprint of the paper at Appendix A2.5)

Volk, M., and Steinhardt, U., 2002b. Models in landscape ecology. In: Bastian, O., Steinhardt, U. (eds.). *Development and Perspectives of Landscape Ecology*. Kluwer Academic Publishers, Dordrecht, 295-306.

Extended Summary[8]

Society needs a way to handle a landscape as a whole, so that the human manipulative capabilities do not have too much headstart over our knowledge about the impacts of these manipulations (Odum 1969). However, extent and rate of effectuate changes in landscapes still exceeds, to a high degree, the scientific capability to reliably predict long-term impacts of technological developments on natural cycles and processes. Human impact on landscape pattern, material fluxes, habitats for plants and animals, but also on socio-economic situations has in fact reached a degree that may lead to irreversible changes and put at risk the natural systems essential for life support. Thus, landscape ecology and other environmental sciences have to develop suitable and improved methods to assess the impacts of anthropogenic changes in landscapes and to develop a conceptual base for sustainable land use.

During the last few decades it has turned out that models are suitable instruments to improve understanding of natural or economic systems. Additionally, they seem to enable comparison and assessment of results from factors that are assumed to influence these systems. By formalization and generalization of the complex reality, landscape models – like any other kind of model – provide the opportunity to connect detailed knowledge of different disciplines (Leser 1991a). Thus, it becomes possible to assess the related ecological and economic consequences of alternative management strategies or potential impacts of human induced landscape changes. In spite of the recent progress, the evaluation of integrated dynamic landscape models is only at the beginning of a far-reaching development. This shortcoming stands to reason considering the lack of quantified data on some topics, the high complexity of the task, as well as the methodological problems to get data in landscape ecosystems. Wenkel (1999) describes the five steps of development from single models to complete model-GIS-integration, which is characterized by coupling and interactive information exchange between sectoral dynamic process models among each other and with a GIS, as well as interactive handling. This Section deals with the development and application of models for the investigation of several parts of the landscape ecosystem including the state of the art on integrated dynamic landscape models. This includes both technical and theoretical aspects.

There is an obvious trend from the development and application of single models to the development of integrated dynamic landscape models holding a multitude of very different modules in a model bank. Landscape models aim at the analysis and assessment of medium- to long-term ecological and socio-economic consequences of human caused landscape changes. Landscape ecology is understood as an inter- and transdisciplinary scientific branch (Tress and Tress, 2002). That means that an instrument trying to consider the landscape ecosystem from a holistic

[8] There is no summary available for this edited book chapter. Therefore the main findings of the chapter are described shortly here.

perspective and bridge the methodological and technical difference between scientific disciplines can only be developed in a multidisciplinary cooperation of many scientific fields. Due to Wenkel (1999) the future progress in landscape modeling will depend particularly on the success of unite theoretical and experimental ecologists with system analysts, computer scientists, and socioeconomists. Beside many scientific and technical open questions, some complex problems have to be solved in the future (Wenkel 1999; Volk and Steinhardt 2001).

3.2.2 (Reprint of the paper at Appendix A2.6)

Steinhardt, U. and Volk, M., 2003. Meso-scale landscape analysis based on landscape balance investigations: problems and hierarchical approaches for their resolution. Ecological Modelling 168, 251–265.

Summary
Varied utilization demands of society to the landscape are leading to an overlay of interests and thus to land use conflicts. Thereby, essential landscape functions like the regulation function (i.e. run-off regulation, groundwater recharge, groundwater protection, buffer functions of the soil, etc.) may be affected, and result in stresses to our natural resources like soil and water. The land use conflicts become especially obvious in a regional context. The diminution of such land use conflicts in terms of a regional management of environment and natural resources requires the knowledge of the response of the landscape balance to land use changes. The results of integrated landscape analysis enable the calculation of scenarios that allow the derivation of site suitable land use variants with positive effects (decrease) to material out-wash from landscape parts and material inputs into surface water and groundwater. Numerous and complex methodological problems arise with such analysis, as well as with the investigation and assessment of the landscape water balance and water-bound material fluxes on the mesoscale.
As a contribution for the resolution of these problems, the authors present a hierarchical nested approach that interlinks scale-specific methods. Due to the complexity and difficult implementation from purely system-oriented approaches in both applied landscape research and planning, the connection to more pragmatic approaches is herewith striked. Thus, information about the impact of land use changes on the landscape balance, as well as the assessment of landscape functions for both watersheds and administrative units should be enabled. Beside the check of the scale-specific applicability of models (i.e. E2D/3D, ABIMO, ASGi, SWAT, modifications of the USLE), the transferability of parameter- and indicator systems for the assessment of the landscape balance on the concerned scale levels is also investigated. An important objective is thereby the optimization of the validity of landscape information for the spatio-temporal levels of the mesoscale.

3.2.3 (Reprint of the paper at Appendix A2.7)

Hirt, U., and Volk, M.: Quantifying the proportion of tile-drained land in large river basins as a contribution to modelling water and nutrient fluxes. Physics and Chemistry of the Earth (*under revision*).

Summary

A considerable reduction in the nutrient and pesticide inputs into the rivers and lakes of Germany is required in order to meet the "good ecological status" as demanded by the European Water Framework Directive (WFD). Sub-surface tile drainage systems are one of the main pathways for such diffuse nutrient and pesticide inputs. However, the simulation of water and matter fluxes under tile-drained land on the landscape scale is still problematic in many countries, mainly due to a lack of data about the existing drainage systems. The present study examines for the first time whether an existing method to calculate the usually unknown proportions of tile-drained areas could be transferred to a large river basin, for which minimal data about drained areas is available. The study area was the Saale river basin (24,000 km²) in central Germany, with a broad variety of soils and site characteristics. The share of tile-drained areas in the Saale river basin was calculated to be 11% of the agricultural area. Apart from that, the calculated proportion of tile-drained areas corresponded satisfactory with the statistical data of the meliorated areas of the former German Democratic Republic. The successful application of the promising method is considered as an important step towards the calculation of the proportion of tile-drained areas for the whole Germany and Europe

3.2.4 (Reprint of the paper at Appendix A2.8)

Volk, M., and Schmidt, G (2004): The model concept in the project FLUMAGIS: Scales, simulation and integration. In: Srinivasan, R., Jacobs, J.H., Jensen, R. (eds.). 2nd International SWAT Conference Proceedings. – TWRI Technical Reports 266: 236-248.

Summary

In order to reach the environmental targets of the European Community (EC) water framework directive on different scales, a concept for the scale-specific simulation of water-bound fluxes in the FLUMAGIS project is presented. According to the interdisciplinary relevance, scale levels have been defined which comprise the micro-, meso-, and macroscale. Thus, for the description of the water balance and matter fluxes within the landscape the models NASIM (microscale), ArcEGMO (micro- to macroscale), ABIMO and SWAT (meso- to macroscale) have been selected. The usage of all these models aims to examine the transferability and applicability of the simulation results to the next higher or lower scale, as well as holding the system open for other models. The scale transition and thus the information exchange between the models are scale-specific depending on the application and compilation of existing parameters and indicators. During the first working phase, the behaviour and sensitivity of the models on different frame conditions and factors is checked out and possibly adapted by using artificial areas. Despite the different model concepts and the temporal differentiation of the input parameters, the results show only small differences, whereas the results of ABIMO and NASIM show greater differences. In general, these differences are caused by different temporal resolutions and parametrization options of the models. As a first

step towards the consideration of the whole area, the ABIMO conceptual model was found suitable for estimating the mean runoff for the Upper Ems River Basin.

3.2.5 (Reprint of the paper at Appendix A2.9)

Ullrich, A., and Volk, M.: The use of a SWAT model to predict the impact of tillage on water quality. Agricultural Water Management (*submitted*)

Summary

Alternative land management practices such as conservation or no-tillage, contour farming, terraces, and buffer strips are increasingly used to reduce nonpointsource and water pollution resulting from agricultural activities. Models are useful tools to investigate effects of such management practice alternatives on the watershed level. However there is a lack of knowledge about the sensitivity of such models to parameters used to represent these conservation practices. Knowledge about the sensitivity to these parameters would help models better simulate the effects of land management alternatives. Thus, this paper presents a sensitivity analysis for conservation management parameters (specifically tillage depth, mechanical soil mixing efficiency, biological soil mixing efficiency, curve number, Manning´s roughness coefficient for overland flow, USLE support practice factor, and filter strip width) in the Soil and Water Assessment Tool (SWAT). With this analysis we aimed to improve model parameterization and calibration efficiency. Based on the results we parameterised for examle curve number values in detail in contrast to tillage depth and mixing efficiency.

Finally the analysis consisted varying selected management practices for different crops. Results showed that the model is very sensitive to applied crop rotations and in some cases even to small variations of management practices. But the different settings do not have the same sensitivity. Duration of vegetation period and soil cover over the time with was most sensitive followed by soil cover characteristics of applied crops.

3.2.6 (Reprint of the paper at Appendix A2.10)

Ullrich, A., Volk, M., and Schmidt, G., 2008. Influence of the uncertainties of monitoring data on model calibration and evaluation. In: M. Sànchez-Marrè, J. Béjar, J. Comas, A. Rizzoli and G. Guariso (Eds.): Proceedings of the iEMSs Fourth Biennial Meeting: International Congress on Environmental Modelling and Software (iEMSs 2008). International Environmental Modelling and Software Society, Barcelona, Catalonia, July 2008. Vol. 1: 544-552. ISBN: 978-84-7653-074-0. http://www.iemss.org/iemss2008/uploads/Main/Vol1-iEMSs2008-Proceedings.pdf (peer reviewed).

Summary

The model-based prediction of the impact of different land management on nutrient loading requires measured nutrient flux data. Thereby the accurate calibration and evaluation of the models need an adequate data base in form of monitoring data. Uncertainties in the monitoring data influence the calibration and thus the parameter settings which affect the modelling results. Hence,

we compared three different time-based sampling strategies and four different load estimation methods for model calibration and compared the results. For our study we used the river basin model SWAT (**S**oil and **W**ater **A**ssessment **T**ool). Study area is the intensively used loess-dominated Parthe watershed (315 km²) in Central Germany.
Nitrate-N load estimation results differ considerable depending on sampling strategy, used load estimation method and period of interest. For study period the load estimation results for the daily composite data set have the lowest ranges (14% and 2% maximum deviation related to the mean value of all applied methods). In contrast estimation results for the sub-monthly and the monthly data set vary in greater ranges (between 25% and 52%). To show differences between sampling strategies we calculated the percentage deviation of mean load estimations of sub-monthly and monthly data sets related to the mean estimation value of composite data set. The maximum deviation of 82% occurs for the sub-monthly data set in 2000. This affects the model and leads to different parameter settings in model calibration and evaluation. Therefore we recommend both the implementation of optimised monitoring programs and the use of more than one load estimation method to describe the water quality situation in a better way and to establish a good calibration base for simulation models.

3.2.7 (Reprint of the paper at Appendix A2.11)

Lorz, C, Volk, M. and G. Schmidt, G., 2007. Considering spatial distribution and functionality of forests in a modeling framework for river basin management. Forest Ecology and Management 248, 17-25

Summary
This paper emphasizes the need of an improved consideration of the spatial distribution and functionality of forests in river basin management. The review of relevant papers has shown that forests, despite their frequent occurrence in temperate zones, play presently only a minor role in river basin management. In general, most of the studies highlight the positive effect of forests on water and nutrient fluxes in river basins. But hydrologists have also reported consistently flood events in or originating from forested areas. In context of the discussion on forest ecology and water quality it became obvious, that forest ecosystems can be sources depending on system properties, time and atmogenic pollution.
The simulation of land use changes on water yield in forested river basins has been carried out in a great number of research projects, but mostly without considering the spatial distributed function of forests. The objective of our work is thus to improve the consideration of spatial distribution of forests in river basins and its effect on water yield and water quality. The most promising approaches in the future are either spatial explicit models or integrated models with both improved forest modules and landscape positioning. The efficiency of these models could be proved by using virtual catchments. As a first conceptual approach towards the base concept of a virtual catchment, we propose a five-units-model (FUM), representing cross-sections with typical land use sequences. The basic idea of our model is to identify major process units and to implement them in the river basin modeling and management.

3.2.8 (Reprint of the paper at Appendix A2.12)

Bosch, D.D., Arnold , J.G. and Volk, M., 2007. SWAT Revisions for Simulating Landscape Components and Buffer Systems. 2007 ASABE Annual International Meeting, June 17-20, 2007, Minneapolis, Minnesota, USA. ASABE Meeting presentation. Paper number: 072175.

Summary

Methods for simulating different landscape positions within the SWAT model are being examined. A three component system, consisting of the watershed divide, the hillslope, and the floodplain landscape positions, has been developed to address flow and transport across hydrologic response units prior to concentration in streams. The modified SWAT model is capable of simulating flow and transport from higher landscape positions to lower positions within a single river basin. The revision was developed to address variable source areas within watersheds and stream-side buffer systems which exist alongside many streams. The enhanced model will allow for more accurate simulation of natural transport processes within a hillslope. The revision was tested using data collected from a low-gradient watershed near Tifton, Georgia, USA which contains heavily vegetated riparian buffers. The modified model provided reasonable simulations of surface and subsurface flow across the landscape positions without calibration. The application demonstrates the applicability of the model to simulate filtering of surface runoff, enhanced infiltration, and water quality buffering typically associated with riparian buffer systems.

3.2.9 (Reprint of the paper at Appendix A2.13)

Volk, M., Arnold, J.G., Bosch, D.D., Allen, P.M., and Green, C.H., 2007. Watershed Configuration and Simulation of Landscape Processes with the SWAT Model. In: Oxley, L., and Kulasiri, D. (eds). MODSIM 2007 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2007, p. 2383-2389, ISBN: 978-0-9758400-4-7.

http://www.mssanz.org.au/modsim07/papers/43_s47/Watersheds47_Volk_.pdf

Summary

Recent and future river basin management requires a more spatially distributed description of basin hydrology and nutrient transport processes to enable land use management as a process controlling factor to realize sound river basin management. The spatial description of these processes in the Soil and Water Assessment Tool (SWAT) watershed model is presently realized by aggregating the flows from overlaid soil and land use patches in subbasins with averaged slope angles. Many concepts with different degrees of complexity have been developed in river basin modelling to aggregate units with similar hydrologic behavior (Hydrological Response Units). Watershed configuration for SWAT currently consists of: 1) subbasins defined by surface topography and 2) hydrologic response units in each subbasin to account for heterogeneity in soils and land use. The hydrologic response units do not account for landscape position within the subbasin. Until recently, many existing watershed models did not implicitly account for landscape processes within a subbasin. Other smaller scale models do account for hillslope transfer (e.g. WEPP, REMM, APEX, HYDRUS-2D).

In an attempt to account for landscape position and processes, SWAT was modified to simulate landscape units within subbasins. Surface, lateral vadose zone, and groundwater flows are routed

between landscape units (while allowing for hydrologic response units within each landscape unit). Surface runoff can be overland or channelized when routed from one landscape unit to the next. The model is being tested on the USDA-ARS experimental Y-watershed at Riesel, Texas, USA, using soil moisture and groundwater data. Using GIS techniques, the watershed was divided into three landscape units - valley bottom, hillslope, and upland. Further development will include landscape unit routing of sediment and nutrients and stream interaction with the valley bottom (i.e.; riparian/flood plain landscape unit). Simulated daily stream flow at the watershed outlet after routing across the landscape units, compared well to measured flow (R^2 = 0.7). Mean annual lateral flows across landscape units were also realistically simulated. Soil moisture (upper 1 m) was compared to measured soil moisture at one monitoring site in each landscape unit with the model predicting drying early in the summer but following general wetting/drying cycles. The revised version of the model is also tested using data collected from a low-gradient watershed near Tifton, Georgia, USA which contains heavily vegetated riparian buffers. The modified model provided reasonable simulations of surface and subsurface flow across the landscape positions without calibration. The application demonstrates the applicability of the model to simulate filtering of surface runoff, enhanced infiltration, and water quality buffering typically associated with riparian buffer systems. Future validation will include comparison with: 1) the Riparian Ecosystem Management Model (REMM) and riparian data sets; 2) with data from larger basins with defined floodplains; and 3) watersheds having well defined variable source contributing areas. The concept assumes the controlling factors for hydrological processes and functions must be adequately described at different spatio-temporal scales to accurately delineate such response units. This requires a sound description of the characteristics by using physically based parameters and indicators, but also simplified solutions at larger scales. Presentation of the new model concept and first results of testing simulations of different aspects of catchment-related control of landscape processes, pattern hydrology, and spatially distributed modelling are discussed.

3.3 Selection, derivation and verification of input data and process factors

3.3.1 (Reprint of the paper at Appendix A2.14)

Volk, M. and Steinhardt, U., 1998. Integration unterschiedlich erhobener Datenebenen in GIS für landschaftsökologische Bewertungen im mitteldeutschen Raum. Photogrammetrie-Fernerkundung-GIS 2, 349-362.

Integration of different data layer in GIS for landscape ecological assessments in Central Germany

Summary

Current research and discussions in landscape ecology and related disciplines make the need for integrated assessments on landscape scales obvious. The importance of GIS as an integrative tool and as an issue-based information system is increasingly recognised, especially by those who are concerned with environmental planning and management in science, legal authorities and business.
However, there are still problems like incompatible systems and a lack of appropriate data. The two examples in eastern Germany show how to get all important data layers in cooperation with environmental authorities and agencies in consideration of the problems linked with that bulk of

different data. Additionally the modification of input data for a groundwater recharge model is presented as an example for an understandable aggregation and generalization of different scale information that is required for integrated landscape assessments.

3.3.2 (Reprint of the paper at Appendix A2.15)

Petry, D., Herzog, F., Volk, M., Steinhardt, U. and Erfurth, S., 2000. Auswirkungen unterschiedlicher Datengrundlagen auf mesoskalige Wasserhaushaltsmodellierungen: Beispiele aus dem mitteldeutschen Raum. Z. Kulturtechnik und Landentwicklung 1, 19-26.

Potentials and Limits of Mesoscale Water Balance Modeling with Varying Input Data

Summary
Land use and land use dynamics influence the landscape water balance. Mesoscale models quantify these impacts at the regional scale and form the basis for groundwater protection. The quality of input data determines the model output to a high degree. By using the runoff model ABIMO in two adjacent test areas with different input data the suitability of varying data sets is discussed. Main emphasis is laid on spatial resolution and heterogeneity, data regionalisation, and validation of modelling results.

3.3.3 (Reprint of the paper at Appendix A2.16)

Volk, M., Steinhardt, U., Gränitz, S. und Petry , D., 2001. Probleme und Möglichkeiten der mesoskaligen Abschätzung des Bodenabtrages mit einer Variante der ABAG. Wasser & Boden 53(12), 24-30.

Meso-scale estimation of soil erosion with a variant of the universal soil loss equation (USLE): problems and possibilities

Summary
Modified variants of the universal soil loss equation (USLE) are normally used to calculate the risk of erosion and horizontal material transport within integrated hydrological models. Due to limitations in adapting the relevant factors in the models and the problematic transferability, the calculated results may be faulty. The errors can be compounded if the values are in other components of the model. The influence of different variants on the calculated results for the derivation of the USLE factors is shown comparatively for meso-scale applications. Recommendations for the application of variants are given dependent on the data base and the project goal. The present study is a contribution towards optimised application of different, regionally adapted and scale-oriented factor variants in the USLE and integrated models. It also aims at optimisation of a scale-specific interpretation of the results (accuracy evaluation).

3.3.4 (Reprint of the paper at Appendix A2.17)

Chen, P.-Y., Arnold, J.G, Srinivasan, R., Volk, M., and Allen, P.M., 2006. Surveying Groundwater Level Using Remote Sensing: An Example over the Seco and Hondo Creek Watershed in Texas. Groundwater Monitoring & Remediation 26(2), 94-102.

Summary

The Normalized Difference Vegetation Index (NDVI) derived from satellite data has been applied to various vegetation studies. The objective of this study was to assess the feasibility of using the NDVI response to plant water content to predict groundwater level over a watershed located in the Edwards Aquifer of Texas, USA. Results showed that the precipitation data collected inside the watershed were not highly correlated to groundwater depth within 10 d of the event, though a 60-foot sinkhole in the study site was expected to collect rainfall and recharge groundwater in a short time. Alternatively, the NDVI derived from SPOT-VEGETATION satellite data and potential evapotranspiration (PET) based on the Hargreaves PET model were significantly correlated to groundwater depth. Moreover, the stream flow measurements were correlated to groundwater level as well. Two simple models were developed for estimating groundwater levels in the artesian and recharge zones. Independent validations were performed to verify both models. All three variables (NDVI, PET, and stream flow) were directly or indirectly related to the precipitation. The PET was mainly controlled by air temperature, and the temperature was negatively related to precipitation. The NDVI values were affected by both temperature and precipitation, and the amount of rainfall was strongly correlated to the stream flow. This study initiated a unique approach to surveying groundwater level based on satellite information and meteorological data.

3.4 Data integration and derivation of reference units

3.4.1 (Reprint of the paper at Appendix A2.18)

Möller, M., Lymburner, L., and Volk, M., 2007. The comparison index: A tool for assessing the accuracy of image segmentation. International Journal of Applied Earth Observation and Geoinformation 9, 311–321.

Summary

Segmentation algorithms applied to remote sensing data provide valuable information about the size, distribution and context of landscape objects at a range of scales. However, there is a need for well defined and robust validation tools to assessing the reliability of segmentation results. Such tools are required to assess whether image segments are based on 'real' objects, such as field boundaries, or on artefacts of the image segmentation algorithm. These tools can be used to improve the reliability of any land use/land-cover classifications or landscape analyses that is based on the image segments.

The validation algorithm developed in this paper aims to: (a) localize and quantify segmentation inaccuracies; and (b) allow the assessment of segmentation results on the whole. The first aim is achieved using object metrics that enable the quantification of topological and geometric object differences. The second aim is achieved by combining these object metrics into a 'Comparison Index', which allows a relative comparison of different segmentation results. The approach demonstrates how the Comparison Index *CI* can be used to guide trial-and-error techniques,

enabling the identification of a segmentation scale *H* that is close to optimal. Once this scale has been identified a more detailed examination of the *CI–H-* diagrams can be used to identify precisely what H value and associated parameter settings will yield the most accurate image segmentation results. The procedure is applied to segmented Landsat scenes in an agricultural area in Saxony-Anhalt, Germany. The segmentations were generated using the 'Fractal Net Evolution Approach', which is implemented in the eCognition software.

3.4.2 (Reprint of the paper at Appendix A2.19)

Möller, M., Volk, M., Friedrich, K., Lymburner, L., 2008. Placing soil genesis and transport processes into a landscape context: A multi-scale terrain analysis approach. Journal of Plant Nutrition and Soil Science 171, 1-12.

Summary
Landforms and landscape context are of particular importance in understanding the processes of soil genesis and soil formation in the spatial domain. Consequently, many approaches for soil generation are based on classifications of commonly available digital elevation models (DEM). However, their application is often restricted by the lack of transferability to other, more heterogeneous landscapes. Part of the problem is the lack of broadly accepted definitions of topographic location based on landscape context. These issues arise because of: (1) the scale dependencies of landscape pattern and processes, (2) different DEM qualities, and (3) different expert perceptions. To address these problems, we suggest a hierarchical terrain-classification procedure for defining landscape context. The classification algorithm described in this paper handles object detection and classification separately. Landscape objects are defined at multiple scales using a region-based segmentation algorithm which allows each object to be placed into a hierarchical landscape context. The classification is carried out using the terrain attribute mass-balance index across a range of scales. Soil genesis and transport processes at established field sites were used to guide the classification process. The method was tested in Saxony-Anhalt (Germany), an area that contains heterogeneous land surfaces and soil substrates. The resulting maps represent adaptation degrees between classifications and 191 semantically identified random samples. The map with the best adaptation has an overall accuracy of 89%.

4. Examples For Simulating the Impact of Land Use on Water availability and Water Quality in Medium to Large-Sized Administrative Districts and River Basins

4.1 Overview

Modelling tools, which take into account possible land use and management scenarios, can be helpful in determining measures to achieve a target ecological status (Krysanova and Haberlandt, 2002; Kersebaum et al., 2003; Seppelt, 2003; Chaplot et al., 2004; Krause et al., 2008). Rekolainen et al. (2003) state that "Successful implementation of the WFD requires appropriate mathematical models and other tools to manage different phases of the planning procedure and to support decision making in various steps of the implementation process."
One outcome of the studies presented in Chapter 3 was the selection of suitable models that can be used for working on such topics.

Hence, the main objectives of the studies assigned to Chapter 4 was to develop land use and land management scenarios that would result in

i) ensuring a safe and sustainable water supply (Section 4.2), and
ii) a reduction of total nitrogen in groundwater and rivers to achieve the target of the drinking water directive or LAWA's[9] water quality class II (Section 4.3).

The environmental problems, as well as the landscape characteristics of the study areas Dessau district, Torgau District, and Upper Ems River Basins are very different. The application possibilities of a runoff-simulation model were examined on the landscape scale by scenarios of land use changes. In Volk and Bannholzer (1999; Section 4.2.1) I present my work on using the runoff-model ABIMO for simulating the impact of land use changes on the water balance in the Dessau district. In cooperation with environmental and governmental authorities, test areas with specific land use conflicts (between forestry, agriculture, nature conservation and water resources management) were selected. Based on the model output, those land use variants could be highlighted which showed positive effects on the quality and quantity of the water resources. Besides other scenarios, 10% and 20%, respectively, of the agricultural land of two different study areas were converted to forest land. At one of the two study areas, which is heavily influenced by intensive agricultural use, this would lead to a decrease of 5 to 9 mm/yr of groundwater recharge (caused by higher evapotranspiration values). This can be considered as tolerable. This

[9] Germany's Working Group of the Federal States on Water Problems Issues (LAWA) requires 3 mg/l of total nitrogen as limit value for surface waters (water quality class II) (LAWA, 1998).

area shows land use conflicts between land use, groundwater protection, soil and biodiversity protection. The results would justify the conversion of areas for environmental and natural resources protection. The other study area is used on the one hand for forestry, but on the other hand, for groundwater extraction. We could show here, for instance, that afforestation in this area only slightly affects the groundwater recharge rate but could improve the groundwater protection by a decrease in the agricultural land. Although the potential lowering of the groundwater table caused by increased water extraction would only entail minor changes, its main ecological impact would be experienced by forest land (owing to dryness effects). The study shows a useful application of a runoff model serving as an instrument for landscape planning and an integral part of landscape-ecological assessment. In the study presented in Volk, Herzog et al. (2001; Section 4.2.2), we used the same model for land use scenarios in the Torgau district. In this part of the River Elbe basin there is a pressing land use conflict between groundwater protection and economic development. The debate on reducing wellhead protection zones in eastern Germany resulted from the sharp decline in the demand for water after German reunification in 1990. The main reasons for this lower demand were far-reaching deindustrialization and reduced water losses in the distribution network following extensive repair work. The land use conflicts were especially pronounced in the Torgau region in Saxony. Following an inquiry of the municipal environmental Department at the Helmholtz Centre for Environmental Research – UFZ to support the decision-making process scientifically, an interdisciplinary project was started to examine the conflict and to evaluate the alternatives to resolve it. From an ecological viewpoint, the main effects to be expected by the relevant land use changes in the Torgau case study were changes in the water balance and in the level of groundwater nitrate pollution. The hydrological effects were simulated with ABIMO, which was used to model data for natural groundwater recharge. This was then taken as a basis in conjunction with area-related nutrient balances to estimate nitrogen discharge by using the model CANDY (Franko et al., 2001; Section 4.3.1. See also in Volk, Franko et al., 2001). With regard to the modelling and estimation of the scenario effects, extensive sensitivity analyses were carried out. As a result not only point estimates for the criteria values were generated but also probability distributions reflecting various kinds of data and model uncertainties (Klauer et al., 2006). The results of the studies presented by Volk, Herzog et al (2001; Sections 4.2.2) and Franko et al. (2001; Section 4.3.1) were used as a basis for multi-criteria ecological-socioeconomic assessments described in Horsch et al. (2001) and Klauer et al. (2006). An alternative model combination with ABIMO and the whole-farm simulation model REPRO (***Repro**duction of Soil Fertility*; Hülsbergen, 2003) was used in this study area presented in Neubert et al. (2003; Section 4.3.2). Models such as REPRO provide efficient tools for evaluating both environmental and economic performance of farming systems. The results indicate an explicit dependency of the nitrate leaching on groundwater discharge and nitrate balance in relation to the variants of cultivation practice like organic, integrated and conventional farming.

In spite of these promising results, there are still numerous uncertainties and problems with the simulation of the impact of land use and management on ground and river-water quality (and its dynamics). Considering the lack of available water quality and model input data (such as land management data), the partly–very-high nitrogen concentrations in rivers of intensively cropped regions, the operational deficits of water quality models, and the rather tight time for achieving the environmental objectives of the WFD (2015), we should attempt to answer the question: How realistic is the achievement of the WFD water quality targets in such river basins dominated by agriculture? The results of our work on this topic are presented in Volk et al. (2008; Section 4.3.3). The objective was to find a land use and land management scenario that would reduce the total nitrogen concentration to meet the WFD requirements for good ecological and chemical status. Study area was the Upper Ems River Basin, which is situated in one of the most intensive agricultural regions in Europe. Consecutive land use and management scenarios were developed on the basis of policy instruments such as the support of agro-environmental measures by Common Agricultural Policy and regional landscape development programs. The results have shown that SWAT is able to adequately represent general trends of water quality changes resulting from measures based on land use and management scenarios. But the results showed also clearly that drastic measures, which are unrealistic from a socioeconomic point of view, would be needed to achieve the water quality target in the basin. It became obvious that the achievement of the WFD targets is only possible with a consideration of regional landscape and land use distinctions. The strategies for water quality monitoring have to be improved, and data accessibility must be established.

4.2 Impact of land use on water availability

4.2.1 (Reprint of the paper at Appendix A2.20)

Volk, M. and Bannholzer M., 1999. Auswirkungen von Landnutzungsänderungen auf den Gebietswasserhaushalt: Anwendungsmöglichkeiten des Modells „ABIMO" für regionale Szenarien. Geoökodynamik 20(3), 193-210.

Impacts of land use changes on the landscape water balance – application possibilities of the runoff simulation model ABIMO.

Summary

The application possibilities of a runoff-simulation model were examined on the landscape scale by scenarios on land use changes. At first, the different and heterogeneous input data were modified (aggregation of data, etc.) to enable the calculation processes of the simulation model. For the definition of the scale-dependent application possibilities of the model, the calculations were carried out at different scales. In cooperation with environmental and governmental authorities, test areas with specific land use conflicts (between forestry, agriculture, nature conservation and water resources management) were selected. Based on the model output, those land use variants can be highlighted which show positive effects on the quality and quantity of the water resources. The

study shows a useful application of runoff model serving as an instrument for landscape planning and an integral part of landscape-ecological assessments.

4.2.2 (Reprint of the paper at Appendix A2.21)

Volk, M., Herzog, F., Schmidt, T., and Geyler, S., 2001. Der Einfluss von Landnutzungs-änderungen auf die Grundwasserneubildung. In: Horsch H., Ring, I., and Herzog, F. (eds.). Nachhaltige Wasserbewirtschaftung und Landnutzung. Methoden und Instrumente der Entscheidungsfindung und –umsetzung. Metropolis, 147-164, Marburg.

The impact of land use changes on groundwater recharge.

Extended Summary[10]

As well as playing a host of ecological functions, natural groundwater recharge is enormously important within the renewal of drinking water resources. Land use (and changes thereto) affect the evapotranspiration of soil and plants, and hence also significantly influence natural groundwater recharge via the landscape water balance. The different interests of farming, forestry, and the water industry as well as nature conservation and landscape protection can generate conflicts which can only be solved by taking an integrated approach to evaluating landscape and socioeconomic components (Horsch and Ring, 2001; cf. also O'Callaghan, 1996; Dabbert et al., 1999). This article uses the example of the Torgau district to present a way of modelling how land use changes influence natural groundwater recharge. Although the study area is mainly used for agriculture, it also contains extensive drinking water protection zones as well as landscape protection areas and nature reserves. Our aim here is to quantitatively assess how land use changes affect natural groundwater recharge. Moreover, by using the assessment criterion 'natural groundwater recharge minus groundwater extraction', we can also roughly determine the sustainability of the land use developments considered with respect to quantitative groundwater resources. These findings can then be considered in the multicriteria analysis of the action alternatives, and also provide a basis for investigations into leachate quality. Apart from these objectives, this examination of model algorithms, the modification of the input data and sensitivity analyses is designed to help optimize usage of the run-off formation model ABIMO.

The results enable regional differentiation of the natural groundwater recharge taking into account the prevailing natural conditions and land use types. The influence of land use changes on groundwater recharge can be simulated for the land use scenarios and – in connection with other information levels – both qualitative and quantitative hazard potentials can be pinpointed. The changes to the mean groundwater recharge rate for the entire area remain within a similar order of magnitude to comparable studies. They appear relatively low, although significant differences may occur locally. In this connection, it should be pointed out that the effects caused by simulated land use changes (scenarios) on the natural groundwater recharge closely depend on the selection of the conversion areas and their natural conditions. In a nutshell, although the influence of the simulated land use changes on groundwater recharge throughout the entire district can be classified as minor owing to compensation effects, pronounced differences certainly occur locally.

[10] There is no summary available for this edited book chapter. Therefore the main findings of the chapter are described shortly here.

When evaluating the findings, it should be noted that long-term means were used which should be regarded as 'most likely values'. Although the calculation results are expressed in absolute figures, since we are dealing with a model and given the low spatiotemporal resolution of the input data, they can only indicate orders of magnitude (Volk and Bannholzer, 1999). More detailed investigations at greater scales would entail using different model systems and sets of data with a higher spatiotemporal resolution. Future investigations must increasingly concentrate on optimizing the application of water balance models at different scales (defining their predictive accuracy, comparing the calculation algorithms of different models). One step in this direction was taken in this article by determining the ranges of fluctuation of the results.
Mesoscale calculations designed to predict the effects of land use changes on the water resources in a landscape are always hypothetical for the reasons listed above, as well as because of the long forecasting period. Nevertheless, the spatially related influences and their impact on the regional and local water balance can be roughly shown. This provides planning authorities with a decision-support tool which can be used to avoid negative consequences for the water balance. All in all, the groundwater recharge rates calculated can be regarded as suitable for further usage in calculating the mean nitrate concentrations in leachate. The assessment criterion 'groundwater recharge minus groundwater extraction' is especially significant as an indicator of the sustainability of the land use development in question from the angle of water resources. However, given the low differences, this criterion is irrelevant for assessing action options within the framework of multicriteria analysis.

4.3 Impact of land use on water quality

4.3.1 (Reprint of the paper at Appendix A2.22)

Franko, U., Schmidt, T., and Volk M., 2001. Modellierung des Einflusses von Landnutzungsänderungen auf die Nitratkonzentration im Sickerwasser. In: Horsch H., Ring, I., and Herzog, F. (eds.). Nachhaltige Wasserbewirtschaftung und Landnutzung. Methoden und Instrumente der Entscheidungsfindung und –umsetzung. Metropolis, 165-186, Marburg.

Modelling the impact of land use changes on nitrate concentration of seepage water.

Extended Summary[11]
In recent decades, the increasing intensification of agricultural production has led to more and more environmental resources being consumed. Nitrogen (N), one of the main nutrients of plants, is one of the most important factors of intensification. Since agricultural production is closely related to the weather, exactly planning nutrient usage to make sure they are completely used up by the crops is practically impossible. The surplus nitrogen can usually only be briefly stored in the ground, resulting in nitrogen entering the atmosphere and the leaching of nitrate (NO_3) on a scale which accelerates with the degree of intensification. However, nitrogen is also output by natural and semi-natural ecosystems. In a state of equilibrium, N outputs exactly match the various N inputs from the atmosphere, which total around 60 kg/ha annually (Isermann, 1990; Russow and Weigel, 2000).

[11] There is no summary available for this edited book chapter. Therefore the main findings of the chapter are described shortly here.

Agricultural land has a positive effect on the landscape-related nitrogen balance if the output into the atmosphere and the groundwater is considerably lower than the input from various sources.

Simulation models have increasingly been used in recent years to study and evaluate the water and nitrogen balances. These can be used as a basis to determine land usage variants which, employing the regional regulation potential, lead to nutrient outputs into neighbouring ecosystems being reduced (Franko et al., 1997; Volk and Bannholzer, 1999). The findings presented here covering the Torgau district were achieved using the CANDY simulation system for N leaching beneath farmland. Land use scenarios were worked out for various economic development frameworks (cf. Messner et al., 2001) in order to study their impact on groundwater quality. Data concerning N leaching beneath forest and grassland were taken from the literature.

The procedure for modelling the influence of land use changes on leachate nitrate concentration largely depends on the availability of data and the simulation models which can be used on this basis. The existing stock of data for the Torgau district meant that groundwater recharge was calculated using the ABIMO run-off formation model, while the nitrogen leaching rates beneath farmland were calculated using the CANDY simulation system. The leaching rates from the grassland and forest land use types were taken from the literature. The percentage of the areas in combination with simulation results resp. literature values allows general statements, which are based on statistics and maps. The simulation of the mean nitrate-concentration of seepage water depends mainly on the quality and resolution of the spatial input data and the statistical data. These relatively fuzzy data have to cover the whole study area and they must be available continuously. In addition, problems with generating simulation objects exist still with the realisation of the distribution of organic fertilizers (communities) and how a differentiation of forested areas and the related nitrate discharge rates should be carried out. The regionalized analysis showed that N leaching beneath forests with N saturated soil is an especially sensitive parameter which, given the higher proportion of woodlands (28.5% of the Torgau district) will in the medium term have a highly negative impact on area-weighted leachate quality. At present, our knowledge of the behaviour of woodland soils is still too limited to provide more accurate information. The results presented here hence describe a forecast trend and are beset by large uncertainty which in future will have to be examined using comparative measurements. By contrast, the agricultural areas were easier to evaluate more accurately. The reliability of the information depends above all on the realistic disaggregation of agricultural statistics.

The simulated nitrate concentration of 106 mg/l lies far over the requested limit value for groundwater. Comparison of the arable farming systems investigated shows that the range of measures available to promote environmentally sustainable agriculture will have a lastingly positive effect on leachate quality. In the global context, in addition to nitrate concentration in leachate, the total nitrate output is of particular importance, which in the Torgau district is about 35 kg ha^{-1} a^{-1} – far less than the German average of over 100 kg ha^{-1} a^{-1} (Wendland et al. 1993; Kolbe 2000).

One crucial problem when evaluating future land use results from the increasing buffering of anthropogenic N inputs from the atmosphere into forest ecosystems. By obtaining a steady state of nitrogen in soil, nitrate leaching rates in the magnitude of agriculture have to be expected also under forest: A long-term protection of the groundwater resources requires a prospectively decrease of the inputs, because the irreversible consumption of the nitrate-reducing substances can lead to a "nitrate breakthrough" after their exhaustion. This results in the forecast of the highest nitrate concentrations for the scenarios. Thus, more research needs to be carried out in order to conclude regional and global strategies to stabilize the buffering capacity of forest soils as well as to reduce N flows into the atmosphere in order to safeguard leachate quality in the long term and to bring the nitrogen cycle into an ecologically sustainable balance.

4.3.2 (Reprint of the paper at Appendix A2.23)

Neubert, M., Volk, M. and Herzog, F., 2003. Modellierung und Bewertung des Einflusses von Landnutzung und Bewirtschaftungsintensität auf den potenziellen Nitrataustrag in einem mesoskaligen Einzugsgebiet. Landnutzung und Landentwicklung 44(1), 1-8

Modelling and Assessment of the Impact of Land Use and Variants of Cultivation Practices on the Nitrate Leaching in a Meso-scale Watershed.

Summary
The authors present a method for analysing the impact of land use and variants of cultivation practices on the seeping water quality on the example of the Torgau region (North Saxony, Germany). The investigation is based on the application of both a runoff simulation model and a model simulating the nitrate surplus caused by agricultural land use. Potential fluxes of nitrate have been calculated by using a Geographic Information System. The results are indicating an explicit dependency of the nitrate leaching on the parameters groundwater discharge and nitrate balance in relation to the variants of cultivation practice like organic, integrated and conventional farming.

4.3.3 (Reprint of the paper at Appendix A2.24)

Volk, M., Liersch, S., and Schmidt, G., 2008: Towards the Implementation of the European Water Framework Directive? Lessons learned from water quality simulations in an agricultural watershed. Land Use Policy (article in press: http://dx.doi.org/10.1016/j.landusepol.2008.08.005)

Summary
The main objective of the European Water Framework Directive (WFD) is the achievement of a good ecological and chemical status of the water environment (water bodies). This status corresponds to the limit value of Germany's Working Group of the Federal States on Water Problems Issues (LAWA) for water quality class II (3 mg/l total nitrogen). The rivers in the intensively cropped Upper Ems River basin (northwestern Germany) show total nitrogen concentrations in excess of 5 to 10 mg/l. Hence, the objective of our study was to find a land use and land management scenario that would reduce the total nitrogen concentration to meet the WFD requirements for good ecological and chemical status. We developed consecutive land use and management scenarios on the basis of policy instruments such as the support of agro-environmental measures by Common Agricultural Policy and regional landscape development programs. The model simulations were done by using the Soil and Water Assessment Tool (SWAT) model. Results of SWAT scenario calculations showed that drastic measures, which are unrealistic from a socioeconomic point of view, would be needed to achieve the water quality target in the basin (reduction of arable land from 77.2% to 46% [13% organic farming], increase of pasture from 4%

to 15%, afforestation from 10% to 21%, increase of protected wetlands from 0% to 9%, etc.]. The example shows additionally that the achievement of the WFD targets is only possible with a consideration of regional landscape and land use distinctions. A related problem yet to be addressed is the general lack of measured water quality data with which to calibrate and validate water quality models such as SWAT. This adds considerable uncertainty to already complicated and uncertainty situations. Thus, improved strategies for water quality monitoring, and data accessibility must be established.

5. INTEGRATED ECOLOGICAL-ECONOMIC ASSESSMENT OF THE IMPACT OF LAND USE CHANGES ON WATER AVAILABILITY AND WATER QUALITY: CURRENT STATUS AND RECOMMENDATIONS FOR MEASURES FOR THEIR IMPROVEMENT

5.1 Overview

An integrated approach to catchments, i.e. administrative units, is essential that would include a necessary basis for balancing and reconciling conflicting interests. Therefore we need further development of integrated natural resources management to incorporate land use, pollution leads, and vital ecological goods and services (Falkenmark and Rockström, 2004). Planners and policy-makers have the difficult task to intervene in complex human-natural systems. They can not focus only on individual processes; rather it is necessary to address the system as a complex integral whole and think about economic values. To fulfil these requirements, tools such as Decision Support Systems (DSSs) that integrate environmental, social and economic concerns and that facilitate the involvement of interested parties in the formulation of strategies may be useful. Especially water is considered more and more as an economic good due to competing water use resulting in resource scarcity (e.g. Briscoe, 2005; Young, 2005; Brouwer and Hofkes, 2008). Decision makers need information about the economic value of water and the economic consequences of water management. The complexity of interactions between water, ecology and the economy can be captured through formal, mathematical models. These models link relevant hydrological and biogeochemical processes to economic 'laws' of supply and demand underlying the provision of scarce water services. Brouwer and Hofkes (2008) edited a special issue on "Integrated hydro-economic modelling: Approaches, key issues and future research directions" in the journal of "Ecological Economics". They distinguish between main approaches: modular, holistic and computable general equilibrium models. The latter top-down models counterbalance the traditional emphasis on bottom-up water engineering approaches. Key issues and future research directions in integrated hydro-economic modelling are discussed and illustrated through a variety of case study applications worldwide. Although the interaction works both ways, feedback effects of water changes on the economy and changes in the economy on the water system are often missing in practice.

The studies assigned to Chapter 5 address these problems. Methods of integrated ecological-economic assessment of the impact of land use changes on water availability and water quality are presented here. The work includes hereby all aspects of the methodological framework described in the previous sections: Scale appropriate analysis and ecological-economic assessment of water and matter dynamics in medium to large river basins were applied and transferred from theory to application. The presented studies were carried out in the Weiße Elster and in the FLUMAGIS project (Table 1). The study presented by Rode et al. (2008; Section 5.2.1) includes the work on integrated nutrient transport modelling with respect to the implementation of the WFD which was done in the Weisse Elster Case Study, Germany. The implementation of the WFD can only be tackled by comprehensive environmental and economic assessments based on an integrated methodology and decision support system. The methodology was developed within an interdisciplinary research project on the highly polluted Weisse Elster River basin, a large subcatchment of the Saale basin which is part of the UNESCO-IHP HELP program. The project focuses on nutrient management, river basin management, and decision-making to achieve good ecological status of surface waters. From the modelling study using SWAT it can be concluded that the investigated organic farming scenarios do not ensure a considerable reduction in high nitrogen loads from agricultural land of the studied catchment. Only the scenario on liberalisation of the agricultural market leads to a considerable reduction in nitrogen loads due to large reduction of agricultural land use of 42.6%. The scenario analysis shows that sufficient reductions in nitrogen loads with respect to the ambitious goals of the European WFD can only be achieved with a considerable change in agricultural land use. With regard to the river water quality modelling study it can be concluded that the impact of the most feasible measures on the concentration of inorganic nitrogen is quite low. Little effect on the yearly mean of inorganic phosphorus is also expected. The reason for this is that the autotrophic assimilation is low and the substance regimes between sediments and the water column are on average balanced, since the seasonal dependence of fixation through sorption or mobilisation by desorption or erosion is preponderant. The parameter uncertainties are high and sometimes larger than the effect of the investigated river restoration management scenarios. The case study shows that easily applicable measures for the reduction in diffuse nutrient (especially nitrogen) loads may not be sufficient to reach the goal of good water quality status requested by the WFD.

Volk et al. (2008; Section 5.2.2) present my work on integrated ecological-economic modelling of water pollution abatement management options in the intensively-cropped Upper Ems River basin, which was carried out within the framework of the FLUMAGIS project. A spatial decision support system (SDSS) was developed to support implementation of the WFD. The modelling approach is based on the integration of ecological and socio-economic assessment methods, scale-specific and GIS-based data and knowledge modelling and visualization techniques. A method was developed that enables the transfer of scale-specific data and information. Analyses were performed for baseline conditions and specific management and planning scenarios to improve water quantity and quality at micro-, meso- and macro-scale. The link between water and ecology, which Brouwer and Hofkes (2008) see as another important direction for future research, is also

included in the Spatial Decision Support System (SDSS) presented in Volk et al. (2008; Section 5.2.2).
The results of the study indicate that substantial, expensive water and land management changes at different scales would be necessary to achieve the WFD water quality targets in this basin. Ecological-economic analysis of cost-effectiveness reveals that the costs of achieving certain goals of the WFD can vary more than tenfold depending on which measure is chosen out of the pool of management alternatives. Moreover, the study shows that the differentiation between landscapes and other regional characteristics, although considered essential to the successful implementation of WFD measures, is very data intensive.

5.2 Examples for Integrated ecological-economic assessment approaches in River Basin Management

5.2.1 (Reprint of the paper at Appendix A2.25)

Rode, M., Klauer, B., Petry, D., Volk, M., Wenk, G. and Wagenschein, D., 2008. Integrated Nutrient Transport Modelling with respect to the Implementation of the European WFD: The Weisse Elster Case Study, Germany. - Water SA 34(4), 490-496.

Summary
The goal of the European Water Framework Directive (WFD) is to protect and enhance the status of aquatic and terrestrial ecosystems. To reach this objective an integrated methodology for the implementation of the WFD is essential. The methodology presented was developed within an interdisciplinary research project on the highly polluted 4th order Weiße Elster River basin, a large subcatchment of the Saale basin (Germany), which is part of the UNESCO-IHP HELP program. The project focuses on nutrient management in order to achieve a good ecological status of surface waters. The paper focuses on an integrated modelling of nitrogen transport and comprises combined terrestrial and in-stream transport processes. The mitigation of diffuse and point sources pollution is thereby essential to meet the environmental objectives. Land use scenarios on both organic farming systems and best management practices were analysed and compared with different strategies to reduce point source. The results show that the possible reduction of nitrogen inputs from point sources is much lower compared to the reduction of diffuse inputs from agricultural land use. The results on in-stream nitrogen transformation show that different morphological factors influence the nitrogen retention considerably. The potential of management measures to reduce nitrogen loads by river restoration measures seems to be limited. This is caused by infrastructural facilities that restrict attaining a natural state of river morphology.

5.2.2 (Reprint of the paper at Appendix A2.26)

Volk, M., Hirschfeld, J., Dehnhardt, A., Schmidt, G., Bohn, C., Liersch, S. and Gassman, P.W., 2008. Integrated Ecological-Economic Modelling of Water Pollution Abatement Management Options in the Upper Ems River. Ecological Economics 66, 66-76.

Summary
This paper presents the results of the FLUMAGIS project, in which we developed a spatial decision support system (SDSS) to support the implementation of the European Water Framework Directive (WFD). The modelling approach is based on the integration of ecological and socio-economic assessment methods, scale-specific and GIS-based data and knowledge modelling and visualization techniques. The project study area is the intensively cropped Upper Ems River Basin in north-western Germany. A method was developed that enables the transfer of scale-specific data and information. Analyses were performed for baseline conditions and specific management and planning scenarios to improve water quantity and quality at micro-, meso- and macro-scale. The results of the study indicate that substantial, expensive water and land management changes at different scales would be necessary to achieve the WFD water quality targets in this basin. Ecological-economic analysis of cost-effectiveness reveals that the costs of achieving certain goals of the WFD can vary more than tenfold depending on which measure is chosen out of the pool of management alternatives. Moreover, the study shows that the differentiation between landscapes and other regional characteristics although considered essential to the successful implementation of WFD measures is very data intensive.

6. CONCLUSIONS AND OUTLOOK

In this work I have presented an "open" methodological framework that enables a scale appropriate analysis and assessment of water and matter dynamics in medium to large-sized river basins and management units. "Open" means that new methods can be implemented and assigned to the related scale levels. It was important to me that I used the key findings from the existing scale concepts of disciplines such as geography, system sciences, landscape ecology, and hydrology to define the scale ranges that are relevant to river basin and environmental management because both landscape processes and management and planning procedures follow hierarchical principles. This congruency is the key that opens the door from theory to application.

At first, the development of such a framework needs to cover a wide range of different aspects. The first step is the definition of scales relevant to river basin and environmental management, and the development of a procedure for how the transfer between the scales could be realized (for instance, by using indicators). The second step consists of testing and assigning the related models to the determined scales, as well as of selection and retrieval of appropriate data sets. This is related to uncertainty considerations of models, and input data, as well as investigating in how far we can achieve a rule-based, understandable delineation of assessment units. The question if and how landscape processes should be described and evaluated depends on the related aim of research or management, the related scales and the involved processes.

Hence, several examples presented in this work have demonstrated that models can be reasonably used to investigate the impact of land use and land management changes on water availability and water quality. The examples comprise model applications for questions in the field of designation of suitable priority areas for drinking water protection (among other things the impact of various land use and management scenarios representing different economic developments), and land and water management scenarios to achieve the target value for nitrogen concentration in river water. Apart from these "single" model applications, examples for approaches for the integrated ecological-economic assessment of the impact of land use changes on water availability and water quality were presented, that contain different model concepts and scales to link relevant hydrological and biogeochemical processes to economic 'laws' of supply and demand.

Although we are able to describe and (ecologically-economically) partially evaluate the impact of land use and land management on water availability and water quality in medium to large sized river basins and management units (which is necessary for the recommendation of either measures or land use patterns to improve any environmental situation), several questions remain open here that have to be addressed in the future. In my opinion, the following fields are of great importance to answer these questions. Further and future research should focus on these fields:

i) Improvement of the spatial explicit description of land management and processes in models

Hydrological changes made by human action upstream may have strong repercussions downstream, both on aquatic and riparian ecosystems. Hence, efficient river basin models and management approaches have to link at least upstream and downstream parts of a river basin (Falkenmark and Rockstroem, 2004). But most models to date have not been able to satisfactorily represent measures based on linear structures (such as riparian zones) or spatially explicit measures, and need to be improved. Hence, I am working with the SWAT developers team on a proposed revision to the SWAT model to assess the runoff processes on a representative divide, hillslope and valley bottom at the subwatershed scale. The model structure attempts to reflect the complex controls on infiltration, runoff generation, run-on, and subsurface flow without requiring large computational resources or detailed parameterization. The advantage of the semi-distributed model is that the impact of spatial changes in land use and BMP's on the hillslope valley continuum can now be more realistically assessed. The first steps of that work were published in the papers presented in Bosch et al. (2007; Sections 3.2.8) and Volk et al. (2007; Section 3.2.9). A more advanced version is presented in Arnold et al. (submitted).

ii) Water quality modelling

Experiences of different European and national projects dealing with the model-supported implementation of the WFD revealed that the available models - and here especially integrated model systems - are still far from being suitable for operational applications. This is especially the case for water quality (Euroharp-Project, 2007). For optimum working efficiency of the models in the management processes it is required that they contribute information of a wide range of abiotic and biotic aspects of hydrology and water quality demanded by the decision makers, which cannot be achieved by single groundwater, water quality or erosion models. But there is a need for an improved description of the nutrient dynamics in integrated models.

iii) Remote sensing methods, model parameters and the improvement of water quality monitoring strategies

As shown in Volk, Liersch and Schmidt (2008; Section 4.3.3) and Volk et al. (2008; Section 5.2.2), the lack of a long-term series of water quality data with daily time step measurements and higher spatial resolution has limited our capacity to evaluate water quality simulations – which represents a general problem and results in uncertainty. In addition, the existing monitoring programs for water quality in Europe are not yet suitable to deliver a sound database for the simulations (Jarvie et al., 1997; EEB and WWF, 2005; Allan et al., 2006). On the one hand, modellers can help to reduce uncertainty for the design of monitoring and sampling strategies (as shown in Ullrich et al., 2008; Section 3.2.6). On the other hand, in the future, remote sensing has the potential to become a useful tool to provide information about water quality distribution in water bodies in order to overcome the lack of water quality data. Several authors have studied how space-borne remote sensing can be used for the mapping of water quality in lakes; although little attention has been paid to rivers yet (see for instance Onderka and Pekárová, 2008). My working group on „Integrated Modelling, Remote Sensing and Data assimilation" at the Helmholtz Centre for Enviromental Research – UFZ in Leipzig is currently working on the possibilities of remote sensing methods (hyperspectral, radar and optical data) to derive soil moisture patterns, vegetation parameters and vitality, and other model-relevant parameters (Pause et al., 2008). We will enhance our studies on water quality, which should be possible by using a range of spectral bands.

iv) **The further development and application of parsimonious models in river basin analysis and management**

Several studies have shown that it is very problematic to parametrize such complex river basin models as SWAT. This becomes obvious with several parameters describing soil-, land cover-, groundwater-, river-channel characteristics, or land management and tillage operations. We have to ask here: How detailed do we have to parameterize management operations in such models with large area applications? We have given a few recommendations in Ullrich et al., submitted; Section 3.2.5. In general, parameters with limited availability, which are estimated or derived from literature, increase uncertainty. This becomes even more problematic in regions with very poor data availability, such as in countries with less population density or in developing countries. Hence, several studies and authors suggest the use of so-called parsimonious models. Although they currently do not have the options of models such as SWAT, I believe that there is great potential for their future application in flood risk management, (integrated) river basin and environmental management (Newham et al., 2004). For instance, we used the IHACRES model for both simulation of the streamflow and creation of a rainfall-runoff database to support flood risk assessment (Liersch and Volk, 2008). The IHACRES metric conceptual rainfall-runoff model (Jakeman et al., 1990; Jakeman and Hornberger, 1993) has a parsimonious approach to model parameterisation. The version used in this paper has six free calibration parameters. IHACRES has been applied to catchments with a wide range of climatologies and sizes (Croke et al., 2004). It has been used to predict streamflow in ungauged catchments (Kokkonen et al., 2003; Post and Jakeman, 1999; Post et al., 1998) to study land cover effects on hydrological processes (Croke et al., 2004; Kokkonen and Jakeman, 2002), and to investigate dynamic response characteristics and physical catchment descriptors (Kokkonen et al., 2003; Sefton and Howarth, 1998). Especially the relationship between streamflow, dynamic response characteristics and physical catchment descriptors is of special significance for my future work.

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   Journal impact factor of Agricultural Water Management (2007): 1.388
10) Ullrich, A., **Volk, M.**, and Schmidt, G., 2008. Influence of the uncertainties of monitoring data on model calibration and evaluation. In: M. Sànchez-Marrè, J.

Béjar, J. Comas, A. Rizzoli and G. Guariso (Eds.): Proceedings of the iEMSs Fourth Biennial Meeting: International Congress on Environmental Modelling and Software (iEMSs 2008). International Environmental Modelling and Software Society, Barcelona, Catalonia, July 2008. Vol. 1: 544-552. ISBN: 978-84-7653-074-0. http://www.iemss.org/iemss2008/uploads/Main/Vol1-iEMSs2008-Proceedings.pdf (peer reviewed).

11) Lorz, C, **Volk, M.** and G. Schmidt, G., 2007. Considering spatial distribution and functionality of forests in a modeling framework for river basin management. Forest Ecology and Management 248, 17-25.
Journal impact factor of Forest Ecology and Management (2007): 1.579

12) Bosch, D.D., Arnold , J.G. and **Volk, M.**, 2007. SWAT Revisions for Simulating Landscape Components and Buffer Systems. 2007 ASABE Annual International Meeting, June 17-20, 2007, Minneapolis, Minnesota, USA. ASABE Meeting presentation. Paper number: 072175.

13) **Volk, M.**, Arnold, J.G., Bosch, D.D., Allen, P.M., and Green, C.H., 2007. Watershed Configuration and Simulation of Landscape Processes with the SWAT Model. In Oxley, L. and Kulasiri, D. (eds). MODSIM 2007 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2007, p. 2383-2389, ISBN: 978-0-9758400-4-7. http://www.mssanz.org.au/modsim07/papers/43_s47/Watersheds47_Volk_.pdf (peer reviewed).

14) **Volk, M.** and Steinhardt, U., 1998. Integration unterschiedlich erhobener Datenebenen in GIS für landschaftsökologische Bewertungen im mitteldeutschen Raum. Photogrammetrie-Fernerkundung-GIS 2, 349-362.

15) Petry, D., Herzog, F., **Volk, M.**, Steinhardt, U. and Erfurth, S., 2000. Auswirkungen unterschiedlicher Datengrundlagen auf mesoskalige Wasserhaushaltsmodellierungen: Beispiele aus dem mitteldeutschen Raum. Z. Kulturtechnik und Landentwicklung 1, 19-26.

16) **Volk, M.**, Steinhardt, U., Gränitz, S. und Petry , D., 2001. Probleme und Möglichkeiten der mesoskaligen Abschätzung des Bodenabtrages mit einer Variante der ABAG. Wasser & Boden 53(12), 24-30.

17) Chen, P.-Y., Arnold, J.G, Srinivasan, R., **Volk, M.**, and Allen, P.M., 2006. Surveying Groundwater Level Using Remote Sensing: An Example over the Seco and Hondo Creek Watershed in Texas. Groundwater Monitoring & Remediation 26(2), 94-102. Journal impact factor of Groundwater Monitoring & Remediation (2007): 1.194

18) Möller, M., Lymburner, L., and **Volk, M.**, 2007. The comparison index: A tool for assessing the accuracy of image segmentation. International Journal of Applied Earth Observation and Geoinformation 9, 311–321.

Journal impact factor of the International Journal of Applied Earth Observation and Geoinformation (2007): 1.534

19) Möller, M., **Volk, M.**, Friedrich, K., and Lymburner, L., 2008. Placing soil genesis and transport processes into a landscape context: A multi-scale terrain analysis approach. Journal of Plant Nutrition and Soil Science 171, 1-12.
Journal impact factor of the Journal of Plant Nutrition and Soil Science (2007): 1.082

20) **Volk, M.** and Bannholzer M., 1999. Auswirkungen von Landnutzungsänderungen auf den Gebietswasserhaushalt: Anwendungsmöglichkeiten des Modells „ABIMO" für regionale Szenarien. Geoökodynamik 20(3), 193-210.

21) **Volk, M.**, Herzog, F., Schmidt, T., and Geyler, S., 2001. Der Einfluss von Landnutzungsänderungen auf die Grundwasserneubildung. In: Horsch H., Ring, I., and Herzog, F. (eds.). Nachhaltige Wasserbewirtschaftung und Landnutzung. Methoden und Instrumente der Entscheidungsfindung und –umsetzung. Metropolis, 147-164, Marburg.

22) Franko, U., Schmidt, T., and **Volk M.**, 2001. Modellierung des Einflusses von Landnutzungsänderungen auf die Nitratkonzentration im Sickerwasser. In: Horsch H., Ring, I., and Herzog, F. (eds.). Nachhaltige Wasserbewirtschaftung und Landnutzung. Methoden und Instrumente der Entscheidungsfindung und –umsetzung. Metropolis, 165-186, Marburg.

23) Neubert, M., **Volk, M.** and Herzog, F., 2003. Modellierung und Bewertung des Einflusses von Landnutzung und Bewirtschaftungsintensität auf den potenziellen Nitrataustrag in einem mesoskaligen Einzugsgebiet. Landnutzung und Landentwicklung 44(1), 1-8.

24) **Volk, M.**, Liersch, S., and Schmidt, G., 2008: Towards the Implementation of the European Water Framework Directive? Lessons learned from water quality simulations in an agricultural watershed. Land Use Policy (article in press: http://dx.doi.org/10.1016/j.landusepol.2008.08.005).
Journal impact factor of Land Use Policy (2007): 1.213

25) Rode, M., Klauer, B., Petry, D., **Volk, M.**, Wenk, G. and Wagenschein, D., 2008. Integrated Nutrient Transport Modelling with respect to the Implementation of the European WFD: The Weisse Elster Case Study, Germany. - Water SA 34(4), 490-496.
Journal impact factor of Water SA (2007): 1.12

26) **Volk, M.**, Hirschfeld, J., Dehnhardt, A., Schmidt, G., Bohn, C., Liersch, S., and Gassman, P.W., 2008. Integrated Ecological-Economic Modelling of Water Pollution Abatement Management Options in the Upper Ems River. Ecological Economics 66. 66-76.
Journal impact factor of Ecological Economics (2007): 1.549

DESCRIPTION OF OWN CONTRIBUTION TO THE SUBMITTED PUBLICATIONS

In the following, for each submitted publication of the habilitation thesis my own contribution to the work is differentiated by idea, concept, and realization of the manuscript.

Publication 1:

__Volk, M.__, and Steinhardt, U., 2002. The landscape concept. What is a landscape? In: Bastian, O., and Steinhardt, U. (eds.). Development and Perspectives of Landscape Ecology. Kluwer Academic Publishers, Dordrecht, 1-9.

I had the original idea for the publication, was responsible for the structure and concept of the manuscript and wrote the first version. I also revised the second and third version after intensive discussions with U. Steinhardt.

Publication 2

Steinhardt, U., and __Volk, M.__, 2001. Scales and spatio-temporal dimensions in landscape ecology. In: Krönert, R., Steinhardt, U., and Volk, M. (eds.). Landscape balance and landscape assessment. Springer, Berlin Heidelberg-New York, 137-162.

U. Steinhardt had the idea for the publication, and was responsible for the initial structure and concept of the manuscript, while I was involved in the discussion about the structure. She wrote the first version. I wrote with her together on the second version (revision). I contributed to the sections 6.1, 6.2.4, 6.3, 6.4, 6.5, 6.6, and 6.7, as well as to Tables 6.1, 6.4 and 6.5 and to Figures 6.3.

Publication 3:

__Volk, M.__, and Steinhardt, U., 2001. Landscape balance. In: Krönert, R., Steinhardt, and Volk, M. (eds.). Landscape balance and landscape assessment. Springer, Berlin Heidelberg-New York, 163-202.

I had the idea for the publication, was responsible for the first structure and concept for the manuscript and wrote the first version. I also revised the second and third version after intensive discussions with U. Steinhardt. She also contributed to the "scales part" of the publication.

Publication 4:

Steinhardt, U. and ***Volk, M.****, 2002. The investigation of water and matter balance on the meso-landscape scale: A hierarchical approach for landscape research. Landscape Ecology 17(1), 1-12.*

The idea for the paper came out during a discussion between U. Steinhardt and myself. I was responsible for the structure of the manuscript. I contributed to all sections, figures (except from Figures 7) and tables. The introduction and conclusions were written in equal parts by U. Steinhardt and myself. U. Steinhardt was the lead author for the section on "Landscape hierarchies", while I was responsible for all other sections.

Publication 5:

Volk, M.*, and Steinhardt, U., 2002. Models in landscape ecology. In: Bastian, O., and Steinhardt, U. (eds.). Development and Perspectives of Landscape Ecology. Kluwer Academic Publishers, Dordrecht, 295-306.*

I had the idea for the publication, was responsible for the first structure and concept for the manuscript, wrote the first version and carried out all necessary revisions. The structure of the manuscript and the revisions were discussed with U. Steinhardt.

Publication 6:

Steinhardt, U. and ***Volk, M.****, 2003. Meso-scale landscape analysis based on landscape balance investigations: problems and hierarchical approaches for their resolution. Ecological Modelling 168, 251–265.*

The idea for the paper came out during a discussion between U. Steinhardt and myself. As a result, U. Steinhardt made the structure of the first version of the manuscript. I contributed to all sections, figures (except from Figures 3, 6 and 8) and tables. U. Steinhardt was the lead author of section 2 (theoretical background: scales and dimension), while I was responsible for section 4. The other sections were written in equal parts by U. Steinhardt and myself.

Publication 7:

Hirt, U., and ***Volk, M.****: Quantifying the proportion of tile-drained land in large river basins as a contribution to modelling water and nutrient fluxes. Physics and Chemistry of the Earth (under revision).*

The work for this publication was done within a co-operation with Dr. Ulrike Hirt from the IGB-Leibniz-Institute for Freshwater Ecology and Inland Fisheries in Berlin, Germany.

The first version of the publication was written by U. Hirt. I worked intensively on the re-organisation, the revisions needed for several versions of the manuscript, and the letter of response to the editor of the journal. I contributed also with a terrain analysis of the Saale River Basin as well as with statistical calculations on the distribution of certain soil types relevant to the estimation of the proportion of tile-drained land.

Publication 8:

__Volk, M.__, Schmidt, G (2004). The model concept in the project FLUMAGIS: Scales, simulation and integration. In: Srinivasan, R., Jacobs, J.H., Jensen, R. (eds.). 2nd International SWAT Conference Proceedings. – TWRI Technical Reports 266, 236-248.

The paper was prepared within the FLUMAGIS project, where I lead the project part of scales, landscape water and nutrient modelling. The paper presents the results of our first simulations with SWAT, ABIMO and NASIM and the procedure for the scale transition that was implemented in the project, using at first virtual and semi-virtual catchments. I did most of the SWAT simulations, G. Schmidt worked together with me on the ABIMO simulations, and the NASIM calculations were done by O. Gretzschel. The idea for the paper was a product of an intensive discussion with G. Schmidt. As a result, I made the initial structure for the paper, and wrote the first version of the manuscript (in discussion with G. Schmidt). G. Schmidt contributed to the revision of the paper (second version).

Publication 9:

Ullrich, A., and __Volk, M.__, 2009. The use of a SWAT model to predict the impact of tillage on water quality. Agricultural Water Management (submitted)

The publication is part of the dissertation of A. Ullrich. I am her supervisor at the Helmholtz Centre for Environmental research – UFZ in Leipzig. A. Ullrich and I worked intensively on the first structure and the concept of the manuscript. Most of the publication was written by A. Ullrich, while I did most of the revision of several versions of the manuscript. I contributed to all Chapters and Sections.

Publication 10:

Ullrich, A., __Volk, M.__, and Schmidt, G., 2008. Influence of the uncertainties of monitoring data on model calibration and evaluation. In: M. Sànchez-Marrè, J. Béjar, J. Comas, A. Rizzoli and G. Guariso (Eds.): Proceedings of the iEMSs Fourth Biennial Meeting: International Congress on Environmental Modelling and Software (iEMSs 2008). International Environmental Modelling and Software Society, Barcelona, Catalonia, July 2008. Vol. 1: 544-552. ISBN: 978-84-7653-074-0.

http://www.iemss.org/iemss2008/uploads/Main/Vol1-iEMSs2008-Proceedings.pdf (peer reviewed).

The publication is part of the dissertation of A. Ullrich. I am her supervisor at the Helmholtz Centre for Environmental research – UFZ in Leipzig. A. Ullrich and I worked intensively on the first structure and the concept for the manuscript. Most of the publication was written by A. Ullrich, while I did most of the revision of several versions of the manuscript. I contributed to all Chapters and Sections.

Publication 11:

Lorz, C, ***Volk, M.*** *and G. Schmidt, G., 2007. Considering spatial distribution and functionality of forests in a modeling framework for river basin management. Forest Ecology and Management 248, 17-25*

The work for this publication was done within a co-operation with Dr. C. Lorz from the Dresden University of Technology and Dr. G. Schmidt from the Martin-Luther-University of Halle-Wittenberg. The idea for the paper emerged during a discussion between C. Lorz, G. Schmidt and myself. As a result, C. Lorz made the structure of the first version of the manuscript. I contributed to section 1, 2 and 6, and I wrote most parts of section 2.2) and 3. I completely wrote section 4 myself. The work on the mentioned sections was discussed with G. Schmidt. The paper includes some results from the Torgau and Flumagis projects. Most of the revisions were done by C. Lorz, after discussions with G. Schmidt and myself.

Publication 12:

Bosch, D.D., Arnold , J.G. and ***Volk, M.****, 2007. SWAT Revisions for Simulating Landscape Components and Buffer Systems. 2007 ASABE Annual International Meeting, June 17-20, 2007, Minneapolis, Minnesota, USA. ASABE Meeting presentation. Paper number: 072175.*

The paper presents the first results of model simulations carried out with the new SWAT version in the experimental watershed Gibbs Farm of the USDA-ARS in Georgia, USA. I am working together with the SWAT developer's team of the USDA on a new version of the SWAT model, where the spatial distribution of the processes and land management is structured in landscape units (valley floor, hillslope, divide). I developed a GIS method that allows the designation of these landscape units and I am working also with the colleagues on the algorithms describing water and nutrient transport through the landscape (and between the landscape units). Most of the paper was written by D. Bosch, I contributed to the first version and revision of the manuscript. J. Arnold and I were involved in all discussions of structure and content of the paper.

Publication 13:

***Volk, M.**, Arnold, J.G., Bosch, D.D., Allen, P.M., and Green, C.H., 2007. Watershed Configuration and Simulation of Landscape Processes with the SWAT Model. In Oxley, L. and Kulasiri, D. (eds). MODSIM 2007 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2007, p. 2383-2389, ISBN: 978-0-9758400-4-7. http://www.mssanz.org.au/modsim07/papers/43_s47/Watersheds47_Volk_.pdf*

The paper presents the initial results from model simulations carried out with the new SWAT version in the experimental watershed Riesel of the USDA-ARS in Temple, TX, USA (see above regarding my contribution to the work on the development of the new SWAT version). I discussed the structure of the paper with all co-authors (but especially with J. Arnold and P. Allen), I wrote the complete text of the paper myself. Linguistic and contextual improvements of the manuscript were made by P. Allen and D. Bosch. J. Arnold, P. Allen and D. Bosch contributed to the second and third revision of the manuscript.

Publication 14:

***Volk, M.** and Steinhardt, U., 1998. Integration unterschiedlich erhobener Datenebenen in GIS für landschaftsökologische Bewertungen im mitteldeutschen Raum. Photogrammetrie-Fernerkundung-GIS 2, 349-362.*

I had the idea for the publication and I was responsible for the initial structure and concept for the manuscript, wrote the first version and carried out most of the necessary revisions. The structure of the manuscript and the revisions were discussed with U. Steinhardt. The paper presents our experience with (and solutions for) the integration of different data layer in GIS in the Dessau and Elbe ecology project, which was needed for modelling purposes and landscape analyis (assessment of landscape functions).

Publication 15:

*Petry, D., Herzog, F., **Volk, M.**, Steinhardt, U. and Erfurth, S., 2000. Auswirkungen unterschiedlicher Datengrundlagen auf mesoskalige Wasserhaushaltsmodellierungen: Beispiele aus dem mitteldeutschen Raum. Z. Kulturtechnik und Landentwicklung 1, 19-26.*

The paper presents our work on the potentials and limits of mesoscale water balance modelling (using ABIMO) with varying input data from the Dessau, Torgau and Elbe ecology projects. The idea and structure for the paper emerged during discussions among all authors. The first version of the manuscript was written mainly by D. Petry, who was working at this time in my group. I contributed to all sections. The revisions of the second version of the manuscript were discussed with all co-authors, but were mainly written by D. Petry.

Publication 16:

__Volk, M.__, Steinhardt, U., Gränitz, S. und Petry , D., 2001. Probleme und Möglichkeiten der mesoskaligen Abschätzung des Bodenabtrages mit einer Variante der ABAG. Wasser & Boden 53(12), 24-30.

I had the idea for the publication and I was responsible for the initial structure and concept of the manuscript and wrote the first version and carried out most of the necessary revisions. The results of the investigations carried out in the Rossel catchment, which is located in the Dessau district, were achieved in the diploma thesis of S. Gränitz, which I supervised. The results of the studies in the Parthe watershed were part of the work of U. Steinhardt. D. Petry worked for his PhD thesis in the Dessau district. He was part of my working group in the project. The structure for the manuscript and the revisions were discussed with U. Steinhardt and D. Petry.

Publication 17:

Chen, P.-Y., Arnold, J.G, Srinivasan, R., __Volk, M.__, and Allen, P.M., 2006. Surveying Groundwater Level Using Remote Sensing: An Example over the Seco and Hondo Creek Watershed in Texas. Groundwater Monitoring & Remediation 26(2), 94-102.

The work for this publication was done during my stay at the USDA-ARS Grassland, Soil and Water Research Laboratory in Temple, TX, USA as well as at the Blackland Research and Extension Centre of the Texas A. & M. University in Temple, TX, USA. The idea for this paper emerged from intensive discussions between Dr. P.-Y. Chen, Dr. J. Arnold and myself. We searched for methods that could provide information about groundwater level changes (as input for hydrological models) without the need for groundwater well data (which is always difficult to get). So we had the idea to check if it was possible to use remote sensing data.

P.-Y. Chen wrote most of the paper and did most of the revisions of former versions. I contributed to the sections “introduction”, “methodology”, “results”, “discussion” and “conclusion”. The regression models were developed by P.-Y. Chen, but discussed intensively with J. Arnold, P. Allen and myself.

Publication 18:

Möller, M., Lymburner, L., and __Volk, M.__, 2007. The comparison index: A tool for assessing the accuracy of image segmentation. International Journal of Applied Earth Observation and Geoinformation 9, 311–321.

The publication is part of the dissertation of M. Möller. I was his supervisor at the Helmholtz Centre for Environmental research – UFZ in Leipzig. M. Möller and I worked intensively on the initial structure and the concept for the manuscript. Most of the publication was written by M. Möller, while I worked intensively on the revision of several versions of the manuscript and contributed to section 1 (introduction), 3 (study area), 4 (results) and 5 (conclusions). L. Lymburner contributed with ideas for the methodology, interpretation of the results, and with the linguistic improvement of the paper.

Publication 19:

Möller, M., ***Volk, M.****, Friedrich, K., and Lymburner, L., 2008. Placing soil genesis and transport processes into a landscape context: A multi-scale terrain analysis approach. Journal of Plant Nutrition and Soil Science 171, 1-12.*

The publication is also a part of the dissertation of M. Möller. The idea of the manuscript emerged during a discussion between M. Möller and myself. M. Möller and I worked intensively on the structure and the first conception for the manuscript. Most of the publication was written by M. Möller and myself. I worked intensively on the revisions of two versions of the manuscript and contributed to section 1 (introduction), 2 1 (site description), 2.2 (data base and preparation), 2.4 (landform classification) and 4 (conclusions). K. Friedrich and L. Lymburner contributed with ideas for the methodology and the interpretation of the results. L. Lymburner contributed to the linguistic improvement of the paper.

Publication 20:

Volk, M. *and Bannholzer M., 1999. Auswirkungen von Landnutzungsänderungen auf den Gebietswasserhaushalt: Anwendungsmöglichkeiten des Modells „ABIMO“ für regionale Szenarien. Geoökodynamik 20(3), 193-210.*

The publication presents the results of the diploma thesis of M. Bannholzer which I supervised. The thesis was prepared within the Dessau project, which I lead together with Prof. Dr. R. Krönert. I was responsible for the idea and concept of the thesis.
I had the idea for the publication and wrote the first concept for the manuscript. M. Bannholzer wrote the first version of the manuscript. I wrote the second version of the manuscript as well as I made all necessary revisions. The revisions were discussed with M. Bannholzer.

Publication 21:

Volk, M.*, Herzog, F., Schmidt, T., and Geyler, S., 2001. Der Einfluss von Landnutzungsänderungen auf die Grundwasserneubildung. In: Horsch H., Ring, I., and*

Herzog, F. (eds.). Nachhaltige Wasserbewirtschaftung und Landnutzung. Methoden und Instrumente der Entscheidungsfindung und –umsetzung. Metropolis, 147-164, Marburg.

The publication presents the results of our work in the Torgau project. I was responsible for the idea and concept of the book chapter. I wrote the manuscript myself, but with intensive discussions with all co-authors. I wrote all revisions of the manuscript, which were also discussed with the co-authors.

Publication 22:

Franko, U., Schmidt, T., and ***Volk M.****, 2001. Modellierung des Einflusses von Landnutzungsänderungen auf die Nitratkonzentration im Sickerwasser. In: Horsch H., Ring, I., and Herzog, F. (eds.). Nachhaltige Wasserbewirtschaftung und Landnutzung. Methoden und Instrumente der Entscheidungsfindung und –umsetzung. Metropolis, 165-186, Marburg.*

The publication presents the results of our work in the Torgau project. U. Franko and T. Schmidt were responsible for the idea and concept of the book chapter. I was involved in the discussion for the structure for the book chapter. I contributed to sections 1, 3.2.1, 4.3 and 5. I also contributed to the first revision of the manuscript.

Publication 23:

Neubert, M., ***Volk, M.*** *and Herzog, F., 2003. Modellierung und Bewertung des Einflusses von Landnutzung und Bewirtschaftungsintensität auf den potenziellen Nitrataustrag in einem mesoskaligen Einzugsgebiet. Landnutzung und Landentwicklung 44(1), 1-8.*

The publication presents the results of the diploma thesis of M. Neubert. The thesis was prepared within the Torgau project, where I lead the section "Impact of land use on water and nitrogen balance". Dr. Felix Herzog and I were responsible for the idea and concept for the thesis. I was the supervisor for the thesis.
M. Neubert and I developed together the idea for the publication. I made an initial concept for the manuscript, which was then written by M. Neubert (I wrote parts of the introduction, methods and the section 2.1.1 on groundwater recharge. I also contributed to section 3.1 on calculation of groundwater recharge). I also worked intensively on the revision of the first version of the manuscript.

Publication 24:

Volk, M.*, Liersch, S., and Schmidt, G., 2008: Towards the Implementation of the European Water Framework Directive? Lessons learned from water quality* *simulations in an agricultural watershed. "Land Use Policy" (article in press: http://dx.doi.org/10.1016/j.landusepol.2008.08.005).*

The publication presents the results of the diploma thesis of S. Liersch, which I supervised. The thesis was prepared within the FLUMAGIS project, where I lead the project section "Impact of land use on water and nutrient fluxes on different scales". I was responsible for the idea and concept of the diploma thesis, but it was intensively discussed with Dr. Gerd Schmidt.
I also had the idea for the publication and wrote the manuscript and the necessary revisions completely myself. Some sections were discussed with S. Liersch and G. Schmidt.

Publication 25:

*Rode, M., Klauer, B., Petry, D., **Volk, M.**, Wenk, G. and Wagenschein, D., 2008. Integrated Nutrient Transport Modelling with respect to the Implementation of the European WFD: The Weisse Elster Case Study, Germany. - Water SA 34(4), 490-496.*

The publication presents the results of the Weisse Elster project. M. Rode was responsible for the original idea and concept for the paper. I contributed to the sections 1 and 2.1, as well as to the SWAT-related topics of sections 2.2 and 2.3, 3.1 and 3.2. I also contributed to the revisions of the manuscript.

Publication 26:

***Volk, M.**, Hirschfeld, J., Dehnhardt, A., Schmidt, G., Bohn, C., Liersch, S., and Gassman, P.W., 2008. Integrated Ecological-Economic Modelling of Water Pollution Abatement Management Options in the Upper Ems River. Ecological Economics 66. 66-76.*

The publication presents the results of our work in the FLUMAGIS project. I was responsible for the idea and concept for the paper. I wrote the complete manuscript myself, except for section 3.2 "Ecological assessment" (written by C. Bohn), and section 3.3 "Socio-economic analysis" which were written by Dr. J. Hirschfeld and A. Dehnhardt. I did the necessary revisions of the second and third version, but with intensive discussions with all co-authors. P. Gassman contributed to the socio-economic section and also to linguistic improvement of the manuscript.

APPENDIX

A 1. Projects and Supervised Thesis
A 2. Reprints of the Publications
A 3. Statutory Declaration
A 4. Curriculum vitae

A 1. PROJECTS AND SUPERVISED THESIS

Dessau project. The title of the project was “Landschaftsentwicklung, Landschaftshaushalt und Mehrfachnutzung in der Landschaft der Region Bitterfeld-Dessau-Wittenberg” (Landscape development, landscape balance and multiple land use of the region Bitterfeld-Dessau-Wittenberg). The project was funded by the Helmholtz Centre for Environmental Research – UFZ.

Torgau project. The title of the project was „Naturressourcenschutz und wirtschaftliche Entwicklung am Beispiel des Elbeeinzugsgebietes“ (Nature resources protection and economic development on the example of the Elbe River Basin). The project was funded by the Helmholtz Centre for Environmental Research – UFZ.

Elbe project. The title of the project was “Ökologische Forschung in der Stromlandschaft Elbe“ (Ecological research in the riverine landscape of the river Elbe). Our department „Landscape Ecology“ worked in the partial project „Gebietswasserhaushalt und Stoffhaushalt in der Lößregion als Grundlage für die Durchsetzung einer nachhaltigen Landnutzung“ (Landscape water and matter balance in the loess region as a basis for sustainable land use). The project was funded by the Federal Ministry of Education and Research (BMBF). FKZ: 0339586.

FLUMAGIS project. FLUMAGIS is an acronym for “Interdisziplinäre Entwicklung von Methoden und Werkzeugen für das Flusseinzugsgebietsmanagement mit Geoinformationssystemen” (Interdisciplinary development of methods and tools for the planning process and measurement control for river basin management with geo-information systems) (see http://www.flumagis.de/english/e_index.htm). The project was funded by the Federal Ministry of Education and Research (BMBF). FKZ: 03300226.

Weisse Elster project. The title of the project was „Entscheidungshilfen für integriertes Flussgebietsmanagement – Konfliktbewertung und mögliche Lösungsansätze am Beispiel der Weißen Elster” (Decision Support for Integrated River Basin Management - Conflict Assessment and Solutions at the Example of the Weisse Elster). The project was funded by the Federal Ministry of Education and Research (BMBF). FKZ: 0330228.

Dissertation on “Object-relationships and functional hierarchies in landscapes”, by Markus Möller (2008). Supervision with Prof. Dr. Volker Hochschild, Eberhardt-Karls-University Tübingen.

Dissertation on “Influence of land management parameterisation and uncertainty of monitoring data on model calibration and evaluation on the example of SWAT”, by Antje Ullrich (in process).

A 2. REPRINTS OF THE PUBLICATIONS

Appendix A.2 contains black-and-white reprints of the original papers and book chapters and sections. Coloured figures can be found in the original documents (PDFs and books).

A2.1

Volk, M., and Steinhardt, U., 2002. The landscape concept. What is a landscape? In: Bastian, O., and Steinhardt, U. (eds.). *Development and Perspectives of Landscape Ecology*. Kluwer Academic Publishers, Dordrecht, 1-9.

Chapter 1

Landscape and landscape ecology

H.-J. Klink, M. Potschin, B. Tress & G. Tress, M. Volk & U. Steinhardt

1.1 The landscape concept (What is a landscape?)

The question: "What is a landscape?" is problematic. The difficulty associated with the question has its roots in the "normality" of the term "landscape" , because it is part of the colloquial speech. This situation is comparable with those we face when dealing with the words "environment" or "recreation" - everybody "knows" what the words mean but they have their own special definitions and opinions about the concepts. We find the same difficulty in the scientific community when they deal with landscape related research topics. If we consider "landscape ecology" which consists of several different disciplines, we find several different definitions for the term "landscape" in the literature. The definition often depends on the "working scale" of the sub-discipline or the particular focus. We therefore consider here the historical development of the term "landscape" in the context of European Landscape Ecology.

From the beginning, the understanding of the term "landscape" is related to the perception, observation and view of the environment or living space of man. Asking a seven-year old boy-child about his definition, he listed: "...a lot of pasture, a couple of trees, forest, plants, animals, farmland, NO (!) towns, a river and a lake", which shows also this mentioned perceptional-aesthetic view. Naveh and Lieberman (1994) noted the first "visual-aesthetic connotation" of landscape in the book of Psalms (48.2) as, perhaps, the earliest reference to "landscape" in world literature.

In spite of the changes in meaning that the term "landscape" has undergone this "original visual-perceptual and aesthetic" theme has been adopted both in literature and art, and is still used by many people involved in landscape planning and design, and by gardeners" (Naveh and Lieberman 1994). In contrast to North American approaches, in European Landscape Ecology, "landscape" is mostly treated as a system, as a holistic concept that takes in the interrelations between biotic and abiotic components, as well as the human impact upon them. As a result, the analysis of landscape requires an integrated approach (Figure 1.1-1).

Figure 1.1-1: Landscapes comprehend both the abiotic and the biotic components, as well as land use: View of Cres (Croatia) (Photo: O. Bastian 2001)

A. v. Humboldt, the great German geo-scientist, defined "landscape" in the early 19th ^{c}entury as "the total impression of a[n] earth region". Most of the landscape ecologists within geography believe that this definition is related to the landscape as a whole. With the development and specialization of the branches of geo-sciences during more recent times, this view has been seen as more and more "narrow".

Russian geographers, for example, have approached given a much broader interpretation of the concept of landscape, including both biotic and abiotic components. Troll (1970) himself defined landscape as "the total spatial and visual entity of human living space, integrating the geosphere with the biosphere and its noospheric man-made artifacts" (Naveh and Lieberman 1994). In 1939, Troll coined the term "landscape ecology", using the idea to stimulate co-operation between geographers and biologists using aerial photographic interpretation of landscapes (Troll 1939). In doing so, Troll hoped to fulfill his vision of a unified field of earth and life research, a new branch of "ecoscience". In Germany, the geographers who took up "Troll's" Landscape Ecology developed the idea of an integrated landscape view further, both theoretically and philosophically. Discussions about the definition of landscape were closely related to the discussion about the definition of geography itself (Turba-Jurczyk 1990). It should be noted in this context that in Germany "ecological" landscape research was carried out before Troll's time (1939). Studies such as those of Penck (1924, 1941), for instance, had already posed questions at the beginning of the 20th century about the carrying capacity of the earth, and Passarge (1912) talked about **landscape physiology** (Finke 1994).

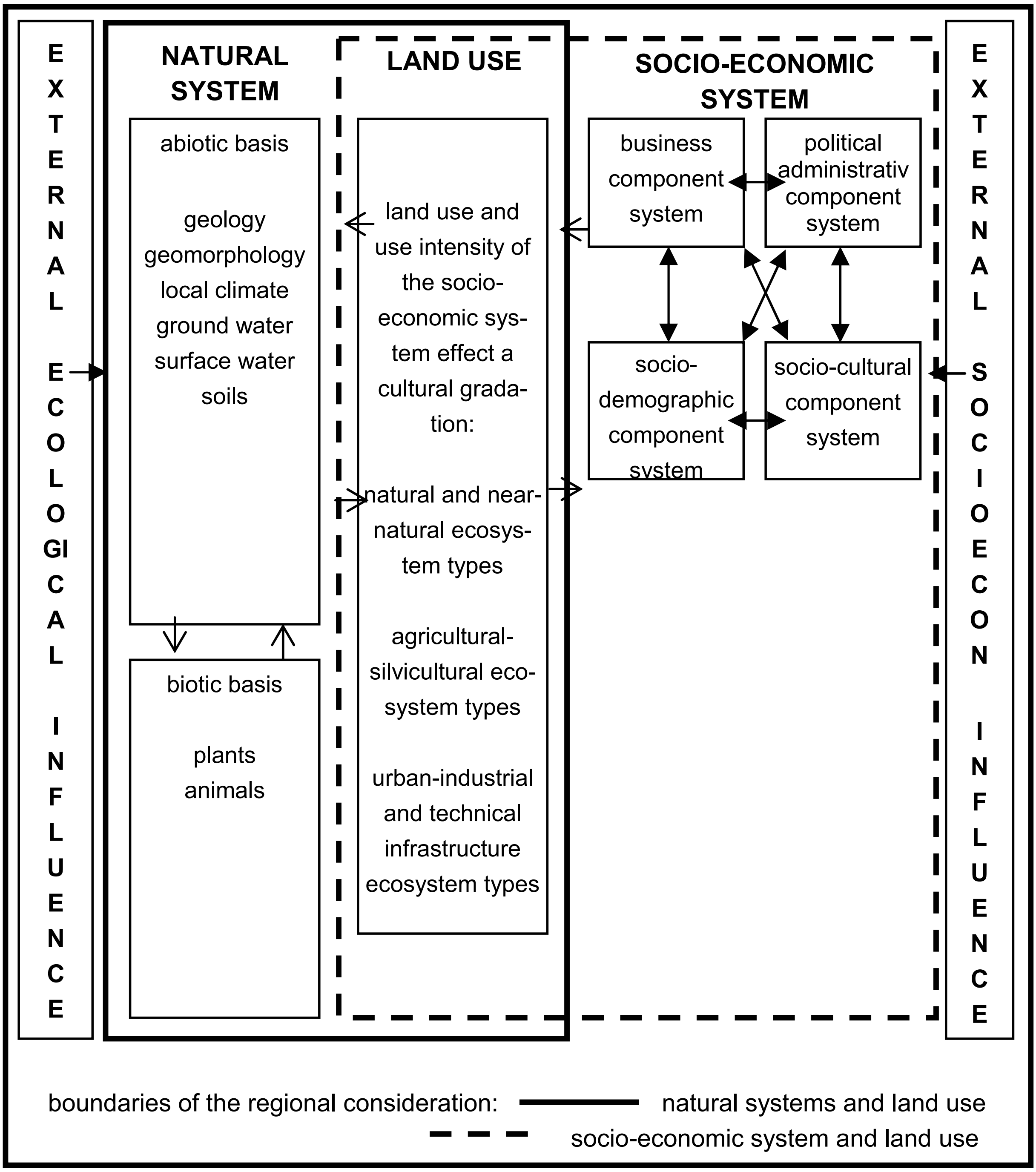

Figure 1.1-2: Schematic presentation of a regional socio-economic ecological system (according to Messerli and Messerli 1979)

Figure 1.1-3: Landscapes are parts of the earth surface with an uniform structure and functional pattern: Cereal fields in the Moritzburg Small Hill Landscape (Saxony, Germany) after the harvest (Photo: O. Bastian 2001)

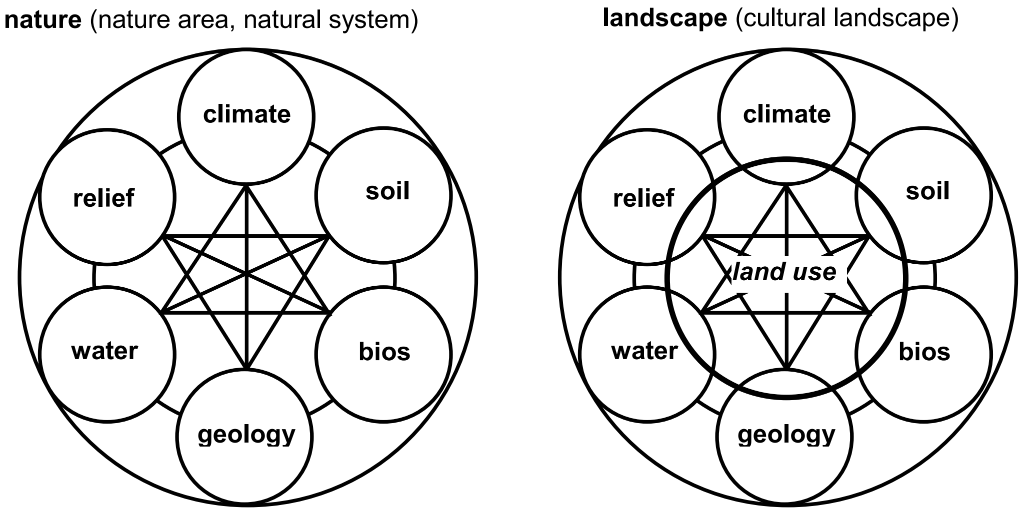

Figure 1.1-4: The landscape concept: According to Neef 1967 and Haase et al. 1991 landscape can be defined as a part of the earth's surface signed by the natural configuration and superimposed by human intervention

The development of Landscape Ecology within geography depends also directly on the discussion about the definition of the term landscape (Bartels 1968, Bobek and Schmithüsen 1949, Neef 1967, Schmithüsen 1963, Turba-Jurczyk 1990). Neef (1967) defined landscape as "... an integrative structure and identic process texture characterized special part of the earth surface", which can be counted as still valid today (Bastian and Schreiber 1999, Figure 1.1-3). Hence landscapes comprehend both the **abiotic and the biotic component**, as well as **land use** (Figure 1.1-2 and 1.1-4). Land use acts as an interface between natural- and socio-economic systems. Landscapes are subject to permanent changes and development due to the natural processes taking place in them and their human use. This use of landscapes results from the **working and living activities of people** (Figure 1.1-5).

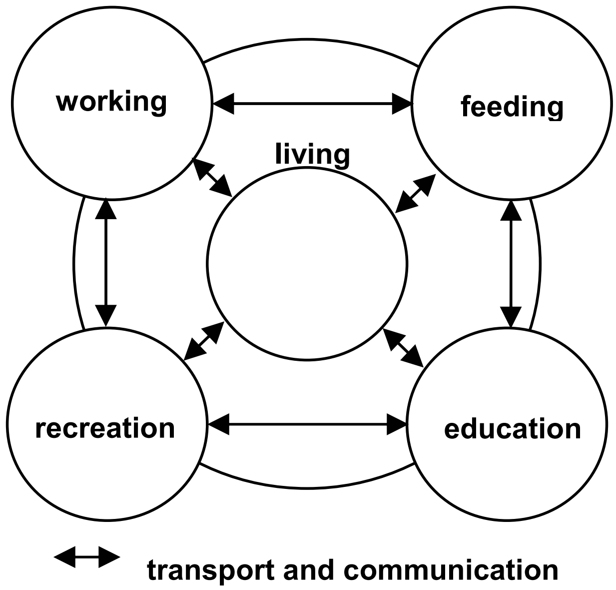

Figure 1.1-5: Basic human needs: fundamental human activities that are immanent in all social ranks and that can be measured temporally and spatially. The number of basic human needs depends on the cultural group as well as the epoch. Our basic human needs are living, working, feeding, recreation and education within communities. Transport and communication are not considered as basic human needs, although they are essential activities for their realization

The overlay of social demands on nature results from the aspirations of people and complex socio-economic interactions. As a result, the production-oriented use of landscapes leads to or contributes to very different environmental stresses. These include the greenhouse effect and depletion of the ozone layer, eutrophication, acidification, toxic contamination, the loss of biodiversity, pollution and consumption of soil, water, forest and marine resources, waste dumping, the consumption and destruction of land, the decrease in environmental quality in urban areas stemming from air, water and soil pollution, noise, and the sealing of land. The change of both land use and cultivation practices, such as ploughing, fertilization, draining, sealing of soil, is one of the most visible features of landscape change and its far-reaching ecological consequences (see Chapter 4.1). Due to natural changes and in view of the history of human impact, on the environment, landscape changes can occur over time scales ranging from thousands of years (e.g. climatic change since the last ice age), centuries (e.g. the cultivation of arable land, settlement, etc.), decades (change of agricultural cultivation practices, sub-urbanization, open cast mining, changes of the weather sequences and water balance, etc.) and years (e.g. crop rotations) to single years (e.g. seasons, phenology and land cover), or even individual (short-term) events (volcanic eruption, earthquakes, flooding). Thus, landscapes have a history (genesis), a current condition or state, and a developmental pathway, as well as a potential natural condition or state (as an abstraction of the current or real landscape). They include renewable and unrenewable **natural resources and potential** use or value. Against this complex background, people aspire to particular conditions of the landscape corresponding to their system of values and demands. But landscapes, like other systems, can exhibit fluctuations around an equilibrium state or in the face of interference a certain resilience to change. Anthropogenic objects and influences activities also have a more or less capacious potential of persistence against change. Thus, cultural landscapes develop as the result of an interplay between the forces of persistence and change (see Chapter 5.1).

A central point of discussions that have ranged over the last decade about the term landscape was the question of whether landscapes are unique or whether types can be identified (Paffen 1953, Schmithüsen 1964). Landscape physiologists developed a theory that the landscape is a synthesis of a multitude of single elements. Later on, this theory became important in landscape ecology.

Another important question explored in discussions about the term landscape was that about spatial dimensions. Thus, Troll (1950) refused to accept the smallest units of nature areas (**physiotopes**, **ecotopes**) as landscapes. In his definition, the term landscape is suitable only up to a typical spatial composition or distribution (**mosaic** of physiotopes or ecotopes). On the other hand, Carol (1957) and Neef (1967) held the view that the size of an area and the direct related exclusion of "wholes" cannot be used as a definition criterion for landscapes. In the disciplines related to landscape ecology, discussion about the central term of geography had also led to confusion rather than to clarification (Finke 1994, Trepl 1987). In these disciplines, especially in the planning branches, a more "unworried" handling with the term landscape can be observed.

Nevertheless, at the beginning of each study dealing with landscape and environment related problems, a definition should be given of what is meant by the term landscape and in which sense it is being used. The definition can depend, for example, on the dominant view, namely whether it is geographical, cultural, functional, and aesthetic or whether other aspects are of interest (Wenkel 1999). A definition given by Haase et al. (1991) in the context of landscape modeling which emphasizes the steps involved when translating from a real landscape to a corresponding landscape model illustrates the process more transparently. According to their definition, landscape is a part of a region that is pre-formed by the natural conditions and more or less shaped and influenced by cultivation and land use. Landscape forms a spatio-temporal structure with interactions between nature and society in it.

From a structural view, a landscape is a mosaic of smallest homogenous spatial units (the topes), from a more functional view it can be described as an ensemble of ecosystems. More simply, Turner and Gardner (1991) considered a landscape to be a spatially heterogeneous area. In a similar vein to the ideas of Haase et al. (1991), Forman and Godron (1986) suggest three landscape characteristics that are useful to consider when thinking about landscape: structure, function, and change. "**Structure** refers to the spatial relationships between distinctive ecosystems, that is, the distribution of energy, materials, and species in relation to the sizes, shapes, numbers, kinds, and configurations of components. **Function** refers to the interactions between the spatial elements, that is, the flow of energy, materials, and organisms among the component ecosystems (but pay attention to the different meaning of landscape function in Chapter 5.2!). **Change** refers to alteration in the structure and function of the ecological mosaic through time" (Turner and Gardner 1991a).

In spite of the discussion between the different disciplines and groups of landscape ecology about the definition of the term landscape, new discussions and questions about the concept arise as the result of the increasing cooperation between of landscape ecologists with economists and socio-economists, in relation to debates about sustainable development. Dabbert et al. (1999) explore this problem in their book about a integrated ecological-economic method for landscape modeling. They point out that in the landscape related sciences, terms like "region" and "regionalization" are used and develop over long periods. Dabbert et al. (1999) cite Grisebach (1872) and Schimper (1898) as examples of writers talking about "biogeographic regions" as early as the end of the 19^{th} Century. These terms have held up and can be found also in he more recent literature (Müller 1980). From the view of plant ecology, "regions" are the subdivisions of "floristic realms", or "bio-realms", in which the differences of macroclimatic conditions become obvious. Considering discussions about landscape, a broader ecological definition of "region" would refer to landscape areas, consisting of similar geological-morphological complexes defined by traditionally similar land use mosaics. These land use mosaics again are reflecting these environmental complexes. In modern agricultural cultivation systems and in the landscape structure, these traditional structures of primary production are often visible today – in spite of variegated changes.

Dabbert et al. (1999) used the term "landscape" in place of "region", making clear their interest in content of spatial ecological and agriculture related economic effects. From a spatial point of view, they identify landscapes as units corresponding approximately to the German **classification of nature areas** ("Naturräumliche Gliederung", Meynen and Schmithüsen 1953-1962, see Chapters 1.2.3 and 2.4.5). With this, they prefer a more pragmatic definition for their integrated approach, but nevertheless, a well grounded one. It seems to be difficult to find a correct translation of the German term "Naturraum": sometimes "natural area" is used, other authors choose "natural sphere" or "natural unit of landscape" and some colleagues prefer "physical region". With respect to content all terms refer to the entirety of natural elements presented in the left part of Figure 1.1.-4. For this book we tried to harmonize this confusion and use the term "nature area".

Considering the various answers that have been given to the question: "What is a landscape?", some general statements can made, for all disciplines of landscape ecology (the following statements are in accordance with a compilation of Forman and Godron 1986, Hansen and Di Castri 1992, Klijn 1995, Turner 1987, Urban et al. 1987, Zonneveld and Forman 1990):

- Landscapes are nearly always the result of both natural and man-induced processes during, nearly always, various time-scales. Landscapes can effectively be described as palimpsests, **patterns superimposed on each other**, showing features of different eras. These legacies affect present-day and future processes.
- **Landscapes are changing**, but changes occur at different rates, either gradually or suddenly, even catastrophically. Landscapes that are stable for a long period are almost fiction.
- Nevertheless there are **stabilizing forces within landscapes**: disturbances are followed by a return to a former status or by a new equilibrium, both in a physico-chemical and in a biological sense.
- Although landscape dynamics show many unexpected or unexplainable phenomena, there is still a **large portion of predictable change** such as primary or secondary succession or degradation stages.
- Landscape are mainly **open systems**: open to vertical influences (e.g. radiation, atmosphere), open to influences from their surroundings and internally open (exchange between patches within one landscape). Landscapes can be understood by insight into the **flows of matter, energy and organisms**.
- **Landscapes are heterogeneous**, both in a vertical and horizontal direction. Vertically one can distinguish layers (atmosphere, canopy, soil, groundwater, rock, etc.). Horizontally, landscapes consist of patches (or ecotopes) with repeat themselves in a certain pattern. Between "homogenous" patches are boundaries that can be sharp or gradual. Boundaries are sometimes open to the exchange of matter, energy or organisms; they sometimes act as barriers or membranes.
- Landscapes are perceived as parts of the earth's surface with **a certain size but with uncertain lower and upper limits**. Questions are open concerning the spatio-temporal definition of landscapes, so that it is not

possible to derive any standard sizes or scales. The definition depends on the priority of view.

In the authors' opinion, these statements contain all relevant characteristics (and also open questions) of (and about) landscapes and their role in landscape ecology.

Discussions about the **landscape concept** are closely related to those surrounding the **ecosystem concept**. Northern American (landscape) ecology has deep roots in biology. So their perception of landscape and landscape ecology differs more or less from the European (see Chapter 1.4, for a discussion of the "ecosystem" term see Chapter 1.2). Following the definition of Chapin (2001) ecosystem ecology links the study of organisms and the physical environment with the functioning of the Earth System. An **ecosystem** is defined as consisting of all the organisms and the abiotic pools with which they interact, and **ecosystem processes** are defined as all the transfers of energy and materials from one pool to another. Hence **ecosystem ecology** addresses the interactions between organisms and their environment as an integrated system. At first sight this approach seems to follow the European approach to landscape and landscape ecology. But in the strict sense the US-approach marks-off a boundary between organisms on the one hand and their environment on the other, especially between organisms and their abiotic environment. In most cases the focus of scientific work in the field of ecosystem ecology is to

- trophic interactions: the feeding relationships among organisms - foodwebs and foodchains (e.g. Pimm 1982, 1984, Power 1992),
- species distribution, populations (Watts 1999),
- habitat fragmentation (Dunning 1999),
- succession: long-term directional changes in community composition (Vitousek and Reiners 1975),
- resilience of ecosystem properties following disturbance (Turner et al. 2001), and
- biodiversity (D'Antonio and Vitousek 1992).

Often (abiotic) environmental conditions are only considered insofar as they are essential for the explanation of the organisms' occurrence or behavior: atmosphere, oceans and climate as well as geology and soils are considered as a background of the ecosystem but not as an inherent part of the system. But also human can't be excluded – they are an inseparable integral part of the environmental system (Haber 1996, 2001). Chapin's term **ecosystem** bears a comparison with our term **physical region** but has nothing to do with **landscape** in the sense discussed above. A holistic view to the whole system is missed. This points up, that the use of the landscape term is often restricted to a specific scale but not to a system that integrates abiotic and biotic environment as well as land use representing the interface between the natural system and the socio-economic system.

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6 Scales and spatio-temporal dimensions in landscape research

Uta Steinhardt, Martin Volk

6.1 Introduction

The human factor 'land use' affects the interactions between water, soil, geomorphology, vegetation, etc. on several spatial and temporal scales in different manners and intensities. The implementation of strategies for sustainable land use assumes specific research concepts from the local to the global scale (micro-, meso- and macroscale). Therefore, landscape ecology science has to provide investigation methods for all these different scales. A number of papers from different scientific disciplines deal with the hierarchical organization of nature (Burns et al. 1991, O'Neill et al. 1986). The hierarchical concept was introduced into German landscape ecology by Neef (1963, 1967) and continued by several other landscape ecologists (Leser 1997). An overview of hierarchical concepts in landscape ecology is given by Klijn (1995). These concepts are mainly focused on the hypothesis, that each of the scale levels (micro-, meso- and macroscale) is characterized by specific temporal and spatial ranges. As a consequence, each scale level needs specific investigation methods as well as data layers with suitable spatio-temporal resolution on the one hand, and which provide specific knowledge on the other (Steinhardt & Volk 2000).

Due to the increased application of GIS over the past few years, this is often reduced to the spatial resolution of the data layers. This paper stresses the necessity of considering scale-specific investigation methods in landscape ecological research. In connection with this, the difficult question of regionalization will be treated. Several examples will be given of proposals for considering scales and spatio-temporal dimensions in landscape research, as well as of scale-specific problems within process-oriented or structurally oriented investigations. One of the main topics is the definition of a linkage between the different scales. The authors will present a hierarchical approach, their main hypothesis beeing that the basic components for most landscape-ecological processes are similar at all scale levels. It is only the importance of the factors (and the factors themselfs) which changes for each scale and have to be defined (Helming & Frielinghaus 1999, Steinhardt & Volk 2000, Volk 1999). This hypothesis will be discussed in detail in Chap. 7.

6.2 Theory of geographic dimensions

The fundamental idea of a the hierarchical organization of nature follows the holistic axiom, that the whole is more than the sum of all of its parts. It was first mentioned by Smuts (1926) and introduced into ecology by Egler (1942).

There are many theoretical approaches to this problem in the literature, and they will be mentioned and discussed first. Afterwards, we will direct the spotlight on a more applied approach - taking into account the hierarchy of both nature and spatial planning.

6.2.1 The terms 'scale' and 'dimension'

There are probably few linguistic obstacles to understanding the German approach to problems of scale and hierarchy. Many new words have been coined (e.g. Nanochore, Microchore, Macroregion), that are some times difficult to understand even for geographers and even more so for scientists from neighboring disciplines, and which are a nightmare to translate into other languages. On the oher hand, landscape ecology claims to be interdisciplinary. Using terms like micro-, meso- and macroscale instead of topological, chorological, regional, and geospherical dimension in (German) landscape ecology (Fig. 6.2) would promote better acceptance and appreciation from other (bio)ecological and geosciencies. Moreover approaches in German landscape ecology would recieve attention abroad, too (Steinhard 1999). Let us tackle this issue with a specific problem of German landscape ecology: the use and definition of the terms 'dimension' and 'scale'. 'Dimension'was introduced into (German) landscape-ecological research by Ernst Neef 1963, who defined dimensions as "... scale levels bearing identical informations in relation to the contents." If a change in scale leads to a new level of geographic reality a change in geographic dimension occurs, thus enabling different information to be gained. By contrast, the term 'scale' is - especially in the English language literature - used in several contexts. Its meaning varies widely between disciplines and communities (Goodchild & Quattrochi 1997). To landscape ecologists, scale might connote 'grain', a measure of patch sizes in a landscape fragmented into discrete habitats. To a cartographer, scale is defined simply as the ratio between a distance on the map and the distance on the ground, this usage often beeing qualified as 'metric scale'. This issue is further complicated by the use of 'scale' as a basic dimension of generalization. Often generalization adds information rather than reducing it, because some kinds of geographic phenomena can only become apparent from large scale observations. But to a scientist, the representation of topography at 1:10,000 is clearly more accurate than one at 1:100,000. One effect of generalization is growing uncertainty in the representation of real phenomenon that could only be mapped perfectly at a much smaller scale[1]. However to most scientists the term 'scale' is

[1] It has to be mentioned, that there is a completely opposed understandig of "small scale" and "large scale" in German and English or American literature: German landscape ecologist and geographers use the term "scale" in terms of cartographers: So 1:100,000 is a smaller scale than 1:10,000. So *small scale* connotes to a *large area* and vice versa. English and American ecologists use the scale terms contrarily: A small scale is coupled to a small area; a large scale

likely to imply some aspect of the small linear dimension discussed above. The term should be used here as an order in the sense of spatial and temporal spheres. Depending on the specific scale level specific investigation methods need to be applied. Table 6.1 compares levels of geographic dimension and scale.

6.2.2 Hierarchy theory

A hierarchy can broadly be defined as 'a partial ordering' of entities (Simon 1973). Complexity frequently takes the form of hierarchy, whereby a complex system consits of interrelated subsystems that are in turn composed of their own subsystems, and so on, until the level of elementary components is reached. The choice of the lowest level in a given system depends not only on the nature of the system, but also on the research question. This corresponds to Herz's (1973) hierarchy of landscape units (Fig. 6.1). The problem of heterogeneity will be discussed in detail in Chap. 6.2.3.

In the literature of hierarchy theory, the subsystems that comprise a level are usually called 'holons' (from the Greek word *holos* = 'whole' and the suffix *on* = 'part' as in proton or neutron; coined by Koestler, 1967). The word holon has been widely adopted mainly because it conveys the idea that subsystems at each level within a hierarchy are 'Janus-faced': they act as 'wholes' when facing downwards and as 'parts' when facing upwards (Wu 1999). With respect to planning practice the scientific term 'holon' should be substituted by the more common term '(landscape) unit'. It is known, that (landscape) units - considered as subsystems at specific scale levels - can be distinguished and mapped by specific criteria.

A hierarchical system has both a vertical structure composed of levels and a horizontal structure consisting of (landscape) units. Hierarchical levels are always separated, by characteristically dominant structures and different process rates. The boundaries between levels and (landscape) units can be considered as surfaces (comparable to layers with barriers). Surfaces filter the flows of matter, energy and information crossing them, and can thus also be perceived as filters. The relationship between subsystems (units) can be distinguished by the degree of interactions among components. Thus, components interact more strongly or more frequently within than between subsystems or surfaces. These characteristics of hierarchical structure can be explained by virtue of 'loose vertical coupling', permitting distinction between levels, and 'loose horizontal coupling', allowing separation between subsystems (units) at each level (Simon, 1973). The existence of vertical and horizontal loose couplings is precisely the basis enabling complex systems to be broken down (e.g. the feasibility of a system to be disassembled into levels and units without a significant loss of information). While the word 'loose' suggests 'can be broken down', the word 'coupling' implies resistance to breakdown. In a- landscape ecological sense

to a large area. For a consistent understanding we will adopt to the English and American scientific community.

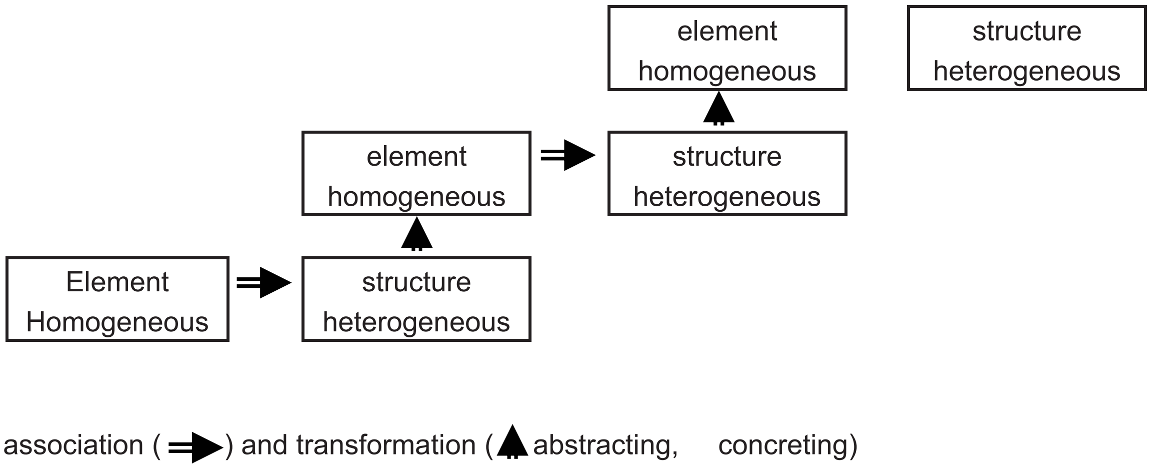


Fig. 6.1: Investigation scheme concerning the hierarchy of landscape units (after Herz 1973)

breakdown means with respect to either considering specific subsystems (soil, water, climate, etc.) or to specific subsets of the earth's surface (landscape units).

According to the principle of breakdown, for a given study focusing on a particular level, constraints from higher levels are expressed as constants or boundary conditions. By contrast, the rapid dynamics at lower levels are filtered (smoothed out) and only manifested as averages or equilibrium values. For a specific problem it is not only possible but also useful to 'scale off' (Simon, 1973) relevant levels from those above and below, thus achieving greater simplification and better understanding. It appears that the tenor reflected in the statement "everything is connected to everything else" often encountered in ecological literature is ultimately unhelpful and perhaps even misleading for understanding complex systems or developing scaling theories. Evidently, for any given phenomenon in this world, some things are more closely connected than others, and most things are only negligibly interrelated with each other (Simon, 1973). Hierarchy theory suggests that when a phenomenon is studied at a particular hierarchical level (the focal level, often denoted as Level 0), the mechanistic understanding comes from the next lower level (Level -1), whereas the significance of that phenomenon can only be revealed at the next higher level (Level +1). It should be pointed out that higher level (Level +1) processes proceed slower and can be considered quasi-constant, while lower level (Level -1) system behaviour operates faster and will be integrated as a mean value. Interestingly, Baldocchi (1993) called the three adjacent scales the reductionist (Level -1), operational (Level 0), and macro (Level +1) scales, respectively. This three-level structure is sometimes referred to as the "triadic structure" of hierarchy (O'Neill, 1989). Thus, three adjacent levels or scales usually are necessary and adequate for understanding most of the behaviour of ecological systems (O'Neill 1988, 1989; Salthe, 1991). The definition and delimitation of a specific hierarchical level is an important step in the problem solution process. The scale level selected determines the main attention to be focused on a specific organizational level of the system being investigated.

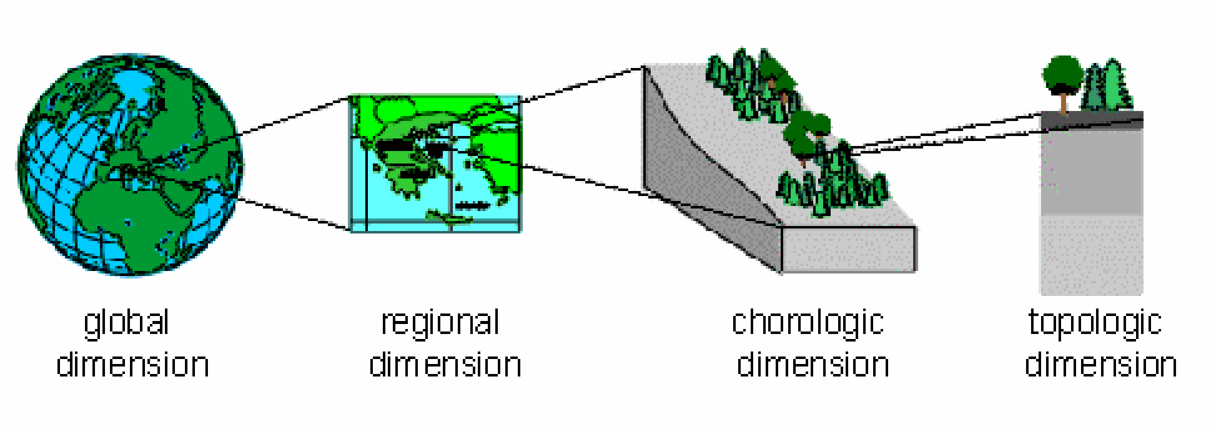


Fig. 6.2: Geographical Dimensions (after http://www.geog.uni-hannover.de/phygeo/trianet/Grafik/Dimensionen.html, 2000)

Starting from this point of view, we can formulate the following premises for further discussions:

1. Spatial and temporal scales are fundamentally interlinked.
2. Complex systems can be broken down in time and space simultaneously.

This is be supported empirically by the fact that many physical and ecological phenomena are arranged along the 45° line in a space-time scale diagram (Fig. 6.3).

Beside hierarchies of space and time there are hierarchies in directions of fluxes of matter and energy, and patterns of nested systems, too. All these hierarchies can be classified in two categories: hierarchies of structures and hierarchies of processes.

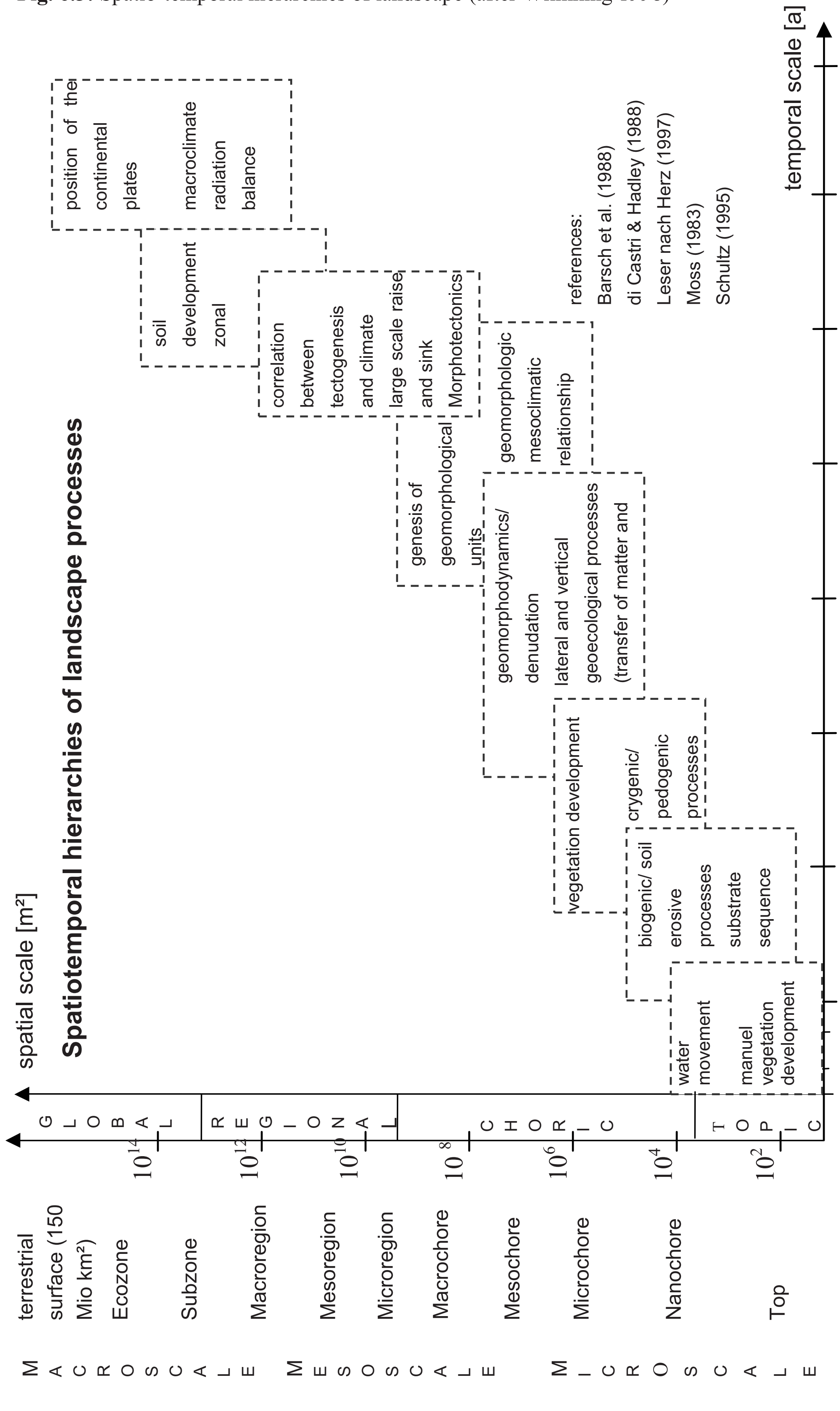
Patterns and processes have components that are reciprocally related, and both patterns and processes, as well as their relationship, change with scale. Different patterns and processes usually differ in the characteristic scales at which they operate. Again, this relates to the near-breakdown ability of ecological systems, and explains why they *can* be studied at a variety of scales, and why they *have* been studied at a variety of scales. To link patterns with processes at the same scale, or to translate them across scales, domains of scale (usually corresponding to hierarchical levels) need to be identified correctly.

The traditional approach was concentrated to a 'vertical' perspective in which a system is viewed as spatially homogeneous and, hence, the internal processes and function is highlighted. In contrast, the landscape approach is directed towards a more 'horizontal' view since it focuses on the spatial distribution of and interactions among ecological entities (Rowe 1961). The vertical perspective promotes a process- or function-based approach (e.g. ecophysiology, population and ecosystem dynamics, etc.), whereas the horizontal perspective tends to encourage a structural, pattern-oriented or geographic approach.

Table 6.1: Scale Levels in Geosciences

Scale Levels of Landscape Ecological Research (after Neef 1967, Barsch 1975)		**Spatial and Temporal Determined Dimensions for Describing Hydrological Processes** (after Dyck 1983)				**Scale Levels in Hydrology** (after Becker 1992)		
Dimension						characteristic Distances	characteristic Areas	*Scale Level*
Geospheric	Geosphere Zone				Climatological level	≥ 100 km 30 - 100 km	≥ 10^4 km² 10^3 - 10^4 km²	*Macroscale*
Regionic	Macroregion	climatic zones			Climatological mean values, ocean currents, large continental areas			
	Microregion	large river catchments regions						
	Macrochore				glacier cover, atmosphere	10 - 30 km	10^2 - 10^3 km²	
	Mesochore			hydrological level		1 - 10 km	1 - 10^2 km²	*Mesoscale*
Chorological	Microchore	catchments		hydrology of catchments		0,1 - 1 km	0,1 - 1 km²	
	Nanochore	small river catchments						
Topological	Geoecotoep	river section	hydrodynamic level			30 - 100 m	0,001- 0,1 km²	*Microscale*
	Physiotope	ground water table hydrotope	hydraulics of rivers and geohydraulics			≥ 30 m	≥ 0,001 km²	
			10^{-6} 10^{-4}	10^{-2} 10^{0} years	10^{2} 10^{4}			

Fig. 6.3: Spatio-temporal hierarchies of landscape (after Wilmking 1998)



6.2.3 The problem of homogeneity and heterogeneity

Neef (1963) reserved the property *homogeneous* for the topological dimension. He described a top as "the basic and indivisible (landscape) unit characterized by a homogeneous combination of all features." By contrast, all other dimensions are determined by a heterogeneity. However, this approach to homogeneity and heterogeneity needs to be reconsidered. Kolasa and Picket (1992) gave a more conceptual definition of heterogeneity: "A system is heterogeneous in time and/or space if a specific temporal interval and/or different locations is characterized by different values." This definition implies that heterogeneity can be achieved at each temporal and spatial level consideration. Herz (1973) also contributed to overcoming the rigid separation between homogeneity and heterogeneity. According to his theoretical studies, homogeneity and heterogeneity can be used to define the 'border' between individual levels of hierarchy: Taking into account the synthetic aspects of transformation and association homogeneity can be attained at each dimension level.

In addition to the main features of geosystems (connected to the earth's surface, high degree of complexity, and spatiotemporal differentiation (Neef 1967), Herz (1984) also developed the principles of landscape ecology (areal-structural principles) taken up by other authors (Bailey 1996):

1. The *principle of correlation* describes the existence of correlative coherences between the partial complexes of the landscape and results in the vertical structure of the landscape (Fig. 6.4).
2. The *principle of areality* assumes that all of these feature combinations are spatially bounded. Taking into account that landscape boundaries are boundaries in a continuum with the character of hemlines (ecotones) and do not isolate parts of the earth's surface, the horizontal or lateral structure of the landscape can be characterized (Fig. 6.5).
3. Following the *principle of polarity* we can observe a constitutional neighborhood between units of the same hierarchical level as well as their dynamic coupling (similarity or contradistinction) - hence the resulting source-sink relations (Fig. 6.6a,b).

Last but not least, the principle of hierarchy allows the delimited units to be classified or subdivided.

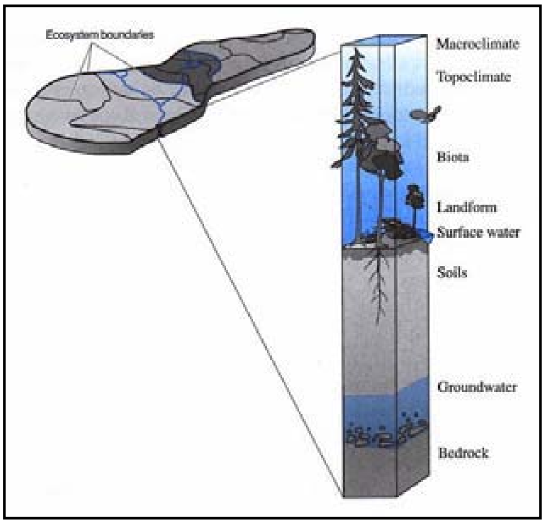


Fig. 6.4: Vertical landscape structure

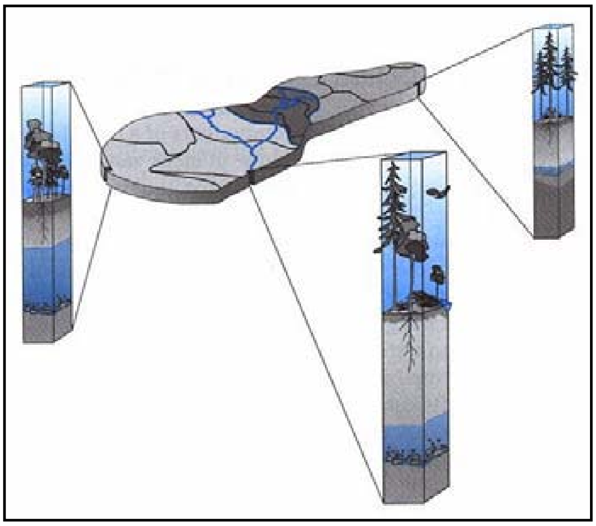


Fig. 6.5: Horizontal landscape structure (boundaries / ecotones)

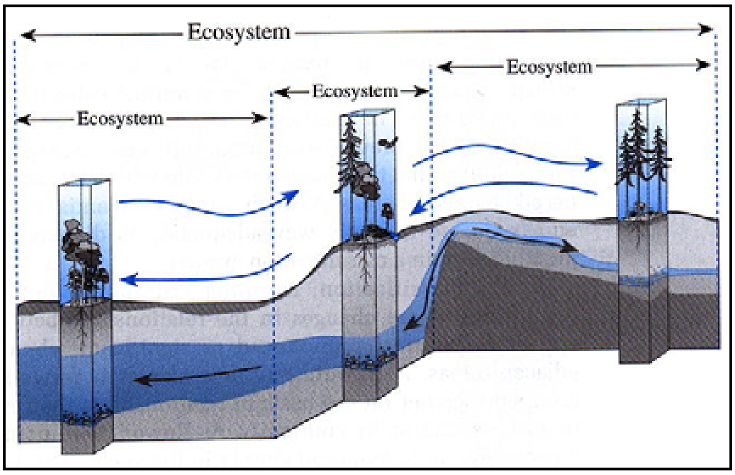


Fig. 6.6a: Permeable landscape

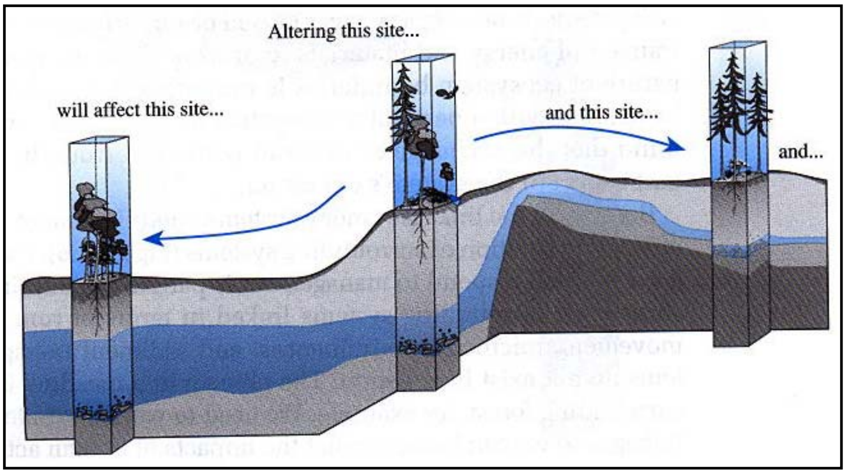


Fig. 6.6b: Sorce-sink effects boundaries (all figures from Bayley 1996)

6.2.4 Landscape heterogeneity and change

Spatial heterogeneity within ecosystems and among ecosystems arrayed on a landscape is critical to the functioning of individual ecosystems and of entire landscapes. For example, the patterns and distribution of plants within arid and semiarid ecosystems control patterns of nutrient cycling processes, with the highest accumulations of organic matter and the highest rates of nutrient cycling occurring under plants rather than in open spaces. Similarly, the configuration of ecosystems within watersheds and at even coarser scales determine the transfers and processes occurring at the landscape scale. For example, the adjacency of riparian systems to upland agricultural systems may prevent nitrate movement from the terrestrial watershed to downstream systems (Rau 1998, Steinhardt & Volk 2000). Furthermore, the degree to which a landscape is fragmented into smaller units determines variation in abundance and diversity of animals. All of the processes and mechanisms that operate in ecosystems have specific spatial dimensions. Knowledge of the physical, biological and ecological sources of this variation and of the resulting spatial patterns are essential in order to understand how both ecosystems and whole landscapes function. Additionally, it forms an important basis for predicting how ecosystems change temporally with natural and anthropogenic alterations.

A great variety of landscape metrics is available for describing and quantifying the degree of heterogeneity. Chap. 5 discussed how to apply these metrics and how to relate the metrics based on landscape structures to landscape processes - to use them as indicators for processes. The size, distribution and connectivity of the patches are quantifiable attributes of landscapes that provide a basis for evaluating change over time and space.

The concept of landscape, as used in ecology, considers a landscape as an ecological system comprising recognizable components such as managed forest patches, agricultural fields, human settlements and natural ecosystems. All landscapes can be thought of as mosaics, composed of discrete, bounded patches that have a distinct biotic structure or composition, and which in some cases are embedded in a predominant and more continuous cover-type matrix. Ecologists working at landscape scales are confronted with spatial heterogeneity both within and among patches of the landscape, and their research topics focus on both the interactions that take place among the units or patches on the landscape, and the behaviour and functioning of the landscape as a whole. Spatial heterogeneity on the landscape stems from both natural and human-caused disturbances, the successional status of vegetation communities, human land uses and land management, variations in state factors and environmental resources, and other anthropogenic influences such as the invasion of non-native species.

For instance, one concept for understanding the variation among soils and landscape is the “state factor” model by Jenny (1941), in which parent material, topography, climate, time and biota are all viewed as independently varying factors that exert control over soil development and ecosystem properties and processes. It is the interaction of these state factors with each other that underlies the formation of the very different types of landscapes existing today and have existed in the past. However, these state factors alone do not control the spatial array of the world's landscapes. Instead they determine the natural matrix upon which natural and human caused disturbances and land uses are overlain.

While natural disturbances have always been a force for spatial and temporal change in ecosystems, human activities have added another and increasingly overwhelming layer of change to the matrix of natural spatial variation. Half of the ice-free terrestrial surface has been transformed in some way by human activity (Turner & Gardner 1990). Human-dominated landscapes supply tremendous amounts of food, fibre and other landscape services to human populations. They have also changed the conditions of the landscape processes related to energy, nutrients and water, and have thereby left their mark on landscape-scale interactions and the earth system as a whole.

Concerning the example of intensive agriculture, it should be mentioned here that all of these land use changes result in significant alterations in the way ecosystems function, the way patches on the landscape affect each other, and the way landscapes as a whole function:

The majority of agricultural land use involves tillage as part of the management system, i.e. the ploughing of soil on a regular basis. During the last few decades, potential agricultural yield per unit of farmland has increased substantially, primarily through the development and use of high-yield crop varieties combined with industrially produced fertilizers, pesticides, herbicides and irrigation. This transition in cultivation methods represents a major shift in the way humans have traditionally practised agriculture. The combination of tillage, new cultivars and intensive inputs into agricultural systems has been largely responsible for keeping food production in step with the rapid human population growth of the last several decades. However, the practice of industrial agriculture carries significant consequences for local, regional and global environments (Matson & Boone 1984). Globally, organic matter stored in the soil profile represents a larger carbon pool than both the biota and the atmosphere together. Tillage disrupts the physical structure of the soil and exposes organic matter that is normally physically protected by its aggregation with soil minerals, thereby making it available to microbial decomposition. This physical change, along with the alteration of soil microclimate, results in faster decomposition and greater potential for erosion. Thus, carbon stored as soil organic matter decreases when subjected to tillage. Modern intensive agriculture also plays a significant role in the biogeochemical cycle of nitrogen. Because grain is harvested and removed from the fields yearly, agriculture depends on the regular input of nitrogen to maintain yields. Since 1945, industrial nitrogen fertilizers have largely supplanted organic nitrogen applied as animal manure or supplied by nitrogen fixation by organisms (such as leguminous plants) as a nitrogen source to crops. The increasing use of nitrogen fertilizers in all its forms has consequences for water and air quality as well as downwind and downstream ecosystems. Numerous local and regional studies have measured elevated nitrate concentrations in systems adjacent to intensive agricultural areas with consequences for both human health and for the functioning of ecosystems. The use of nitrogen fertilizers also influences atmospheric processes through the emission of a number of trace gases. Human-induced alteration in the landscape has become an overwhelming force of change. Land use changes are critical in terms of local scale consequences for biological diversity, air and water quality and other landscape services. At the regional and global scale, land use changes influence atmospheric composition and chemistry - and ultimately climate. At the same time, many land use changes are carried out to meet pressing human needs. These interconnecting causes and consequences call for integrative approaches to conservation, habitat protection, and land use planning that recognize the multiple and interacting roles of the landscape.

Spatial mosaics not only result in the variation and heterogeneity of the land surface cover and in landscape processes within the patch, but also determine the ways in which parts of the landscape mutually influence each other. There are three general pathways by which landscapes interact: via topographically controlled interactions (e.g. the topographically controlled redistribution of materials in water and via erosion); via transfers through the atmosphere (nitrogen and sulphur transfers, biomass combustion, dust transport); and via biotic transfers (movement of plants and animals). Some of these aspects will be discussed in detail in Chap. 7.

6.3 Scales and dimensions in landscape ecology

The whole universe can be considered as an organization consisting of a system hierarchy. Each higher level is constructed from systems of lower hierarchy levels. Lower components are relatively dependent on those above and vice versa any reverse influence by the lower components on the upper cannot be neglected either. All natural processes have their own scale domains, our observations are scale-specific, our interpretations depend on scales, and decisions relate to a certain time frame and spatial context. If we neglect scales, we draw wrong conclusions and take shortsighted decisions (Klijn 1995).

When ascending a scaling ladder, the loss of detailed information is compensated for by a gain in overview information about structures, relationships and interactions. Each scale level offers its own cognition facilities. Climbing up and down the scaling ladder reveals completely new insights into structural and functional phenomena which otherwise remain hidden. Even Troll (1939) said about archaeology, “On the ground we are standing in front of a clutter of lines that can only be arranged into a system from a height.” (p. 250f) This view goes for modern landscape ecology, too: different landscape patterns and processes can be made out depending on whether the earth is observed from the ground, an aeroplane or a satellite.

Studies in landscape ecology, hydrology, meteorology and other related earth sciences have shown that different processes tend to dominate in distinctive, characteristic domains of scale in time and space. Thus, observations made on a single scale can, at best, only capture those patterns and processes pertinent to that scale of observation. The situation inevitably becomes complex when a description or explanation simultaneously invokes multiple levels of organization or domains of scale. While the issue of scaling has been widely recognized as essential in both basic and applied research, a general theory of scaling is still elusive due to the complex matter of scaling.

The discussion about the terms ‘scale’ and ‘dimension’ mentioned in Chap. 6.2.1. was accompanied - especially in Germany - by several proposals to indicate specific scale levels resp. dimensions (see overview in Leser 1991, 202pp). Unfortunately, some of the contributions only focus on a formal completion of the system of landscape units and there is no contribution on the development of methods. Table 6.2 gives an overview of some nomenclatura proposals and the related mapping scales and area of the adequate units. It should be mentioned here that this suggestion defining each scale level with a certain area size and mapping scale is ventured because the extent of the units also depends on the general structure (variety, diversity) of the landscape, which is why the extent of landscape units also differs between for example arid desert and temperate zone. Fig. 6.6 gives another example of landscape mapping at different scales. Bailey (1976, 1983, 1995) developed a technique for mapping ecoregions first for the United States that was subsequently expanded to include the rest of North America (Bailey and Cushwa 1981) and the world (Bailey 1989).

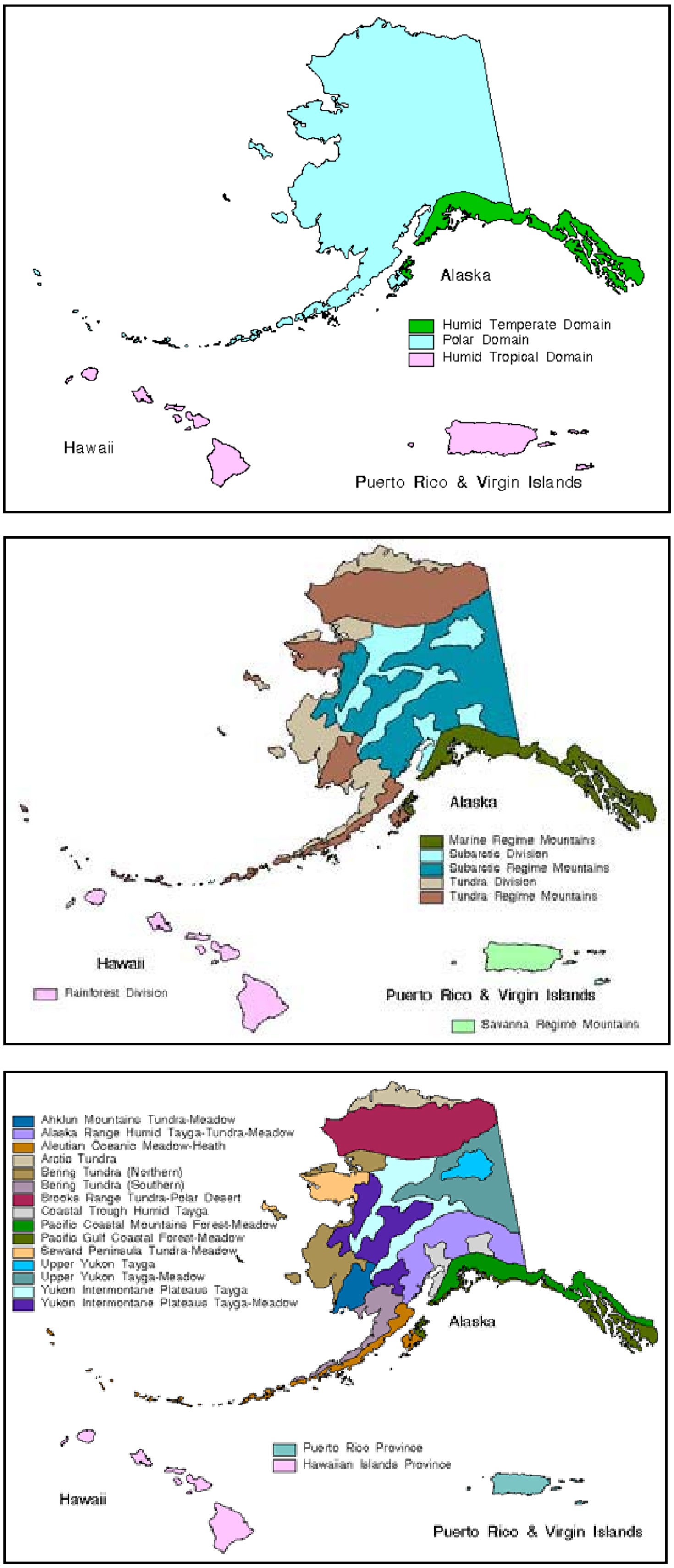


Fig. 6.7: Hierarchical landscape mapping on the example of Alaska (from Bailey 1995)

a) Ecosystem Domains

b) Ecosystem Divisions

c) Ecosystem Provinces

Table 6.2: Different nomenclature proposals for hierarchical ecosystem classification

Barsch 1975	F. Klijn 1997	indicative mapping scale	basic mapping unit
Zone	Ecozone	1:50 Mio and smaller	>62,500 km²
Makroregion	Ecoprovince	1:10 ... 50 Mio	2,500 - 62,500 km²
Subregion	Ecoregion	1:2 ... 10 Mio	100 - 2,500 km²
Mikroregion	Ecodistrict	1:500,000 ... 2 Mio	625 - 10,000 ha
Mesochore	Ecosection	1:100,000 ... 500,000	25 - 625 ha
Mikrochore	Ecoseries	1:25,000 ... 100,000	1.5 - 25 ha
Nanochore	Ecotope	1:5,000 ... 25,000	0.25 - 1.5 ha
Top	Eco-Element	1:5,000 and larger	< 0.25 ha

The regions delineated on the map were adopted for use in ecosystem management and are also used in the proposed National Interagency Ecoregion-based Ecological Assessments.

Each of these scale levels is characterized by specific temporal and spatial ranges and has to be investigated with specific methods (measuring, observation, mapping, modelling). Consequently the temporal and spatial resolution of all data collected or used has to match the scale to which it is to be applied. In reality, data are rarely available in the resolution required (Volk & Steinhardt 1998), which is why we have to tackle data homogenization and data or parameter transfer. Besides the problem of scale-appropriate data, another problem is scale-appropriate investigation methods. What methods of observation are appropriate to what scale level for data collection and what methods should be used to analyse and process them (Table 6.3) must be defined.

With respect to the above discussed problems of patterns and processes as well as scale-specific degrees of complexity, it must be emphasized that landscape ecology cannot be based exclusively on one type of hierarchy. According to Klijn (1995) we have to focus on:

Table 6.3: Complexity of ecological systems and hierarchy theory and appropriated methods

Scales	Dimensions	Ranges of complexity	Methods of data gathering (selection)
Macroscale	regionic - global	disorganized complexity (to be dealt with statistical methods)	remote sensing techniques
Mesoscale	chorological - regionic	organized complexity (quantitative methods are lacking)	combination of fine- and coarse scale methods
Microscale	topological (local) - chorological	organized simplicity (to be dealt with analytical mathematics)	point measurements, field mapping

- Process-functional hierarchies based upon flow directions (relative position of systems in flows of energy, matter and information; ranking according to dependence on other systems)
- Hierarchies in complexity or organizational properties
- Temporal and spatial hierarchies

6.4 Regionalization in landscape ecology

The discussions concerning scale and dimension in landscape ecology resulted in division into specific hierarchical levels, known as micro-, meso- and macroscale. However, this delimitation must not be allowed to lead to the splitting of landscapes into stand-alone hierarchical elements. Despite this hierarchical structuring, a landscape has to be considered as a coherent unit, and so there must be connectivity between all the specific scales. The concept of regionalization can help solve this problem. It can bridge the gap between the ideas of scale-specific and cross-scale approaches. All hierarchical components can be assembled in this way (rather like the pieces of a jigsaw puzzle) to form a consistent landscape. To scale up from a leaf to a continent and beyond, we must understand how information is transferred from a fine to a broad scale and vice versa. We must learn how to aggregate and simplify, retaining essential information without getting bogged down in unnecessary details. Steinhardt & Volk (1999) edited a book about regionalization in landscape ecology which includes papers presented at an interdisciplinary conference in Germany in 1998. It shows the general necessity of regionalization methods and gives an overview of the variety of landscape-oriented research. Yet it brings home the fact that we still seem to be far away from the development of standards (assuming they are possible in the first place).

Initial ideas for the realization of transitions from one scale to another have been developed in hydrology. These transitions have been termed “regionalization” (Kleeberg 1992, 1998). It should be pointed out that this term is not related to the above-mentioned regional dimension. This additional level of dimensions was not introduced into German landscape ecology until 1973 by Haase. ‘Region’ in general refers to widespread areas. Although the term ‘regionalization’ has since come to be used in several disciplines, each discipline has its own narrow understanding of the word, with some interpretations actually being contradictory.

Bach & Frede (1999) launched a new methodological discussion concerning a general definition of regionalization and the development of regionalization strategies. At first sight this discussion seems to be very theoretical, but the more or less formal approach meets the requirements of an interdisciplinary research approach. Thus from a theoretical point of view, all data are characterized by three attributes: object (e.g. soil, climate, vegetation), feature (e.g. grain size, field capacity, mean annual air temperature) and scale (micro-, meso-, macroscale). Usually data have to be transferred to other objects, features and scales. This procedure of data transmission is defined as ‘regionalization’. Depending on which of the three attributes of an existing primary data set is to be regionalized, three fundamental operations of transmission can be distinguished:

Translocation:	The same feature is transferred from one object to <u>other objects</u> of the same class of objects on the same scale.
Transformation:	From one or more features for one object, <u>other features</u> for the same object on the same scale are derived.
Up- or downscaling:	The same feature is transferred from the objects on one scale to the same object on <u>another scale</u>.

Fig. 6.8 shows the three transmission operations graphically. Accordingly, regionalization means the change of data $D_{i,j,k}$ in one or more of its attributes. All the three operations can be performed separately or combined. For this purpose, different methods are available or have to be developed. All transmission rules have to describe the way in which the destination data are generated from the primary data.

The implementation of these operations is currently limited by the rules available in the specific fields of landscape ecology. This must be one of the main tasks to be solved by several geosciences over the next years. Some of the existing transmission rules are mentioned below.

To answer the question: "What about the spatial validity of a data set measured at one point (e.g. different climatic parameters)?", some geostatistical approaches such as Thiessen polygons, kriging or the construction of isobars, isotherms, etc. are already available (Burrough 1986, Oliver 1990, Fohrer et al. 1999). Hence *translocation* is a resolvable problem.

The problem of *transformation* seems to be more difficult. Indicators and transfer functions have to enable new properties to be derived from measured/mapped data. Examples in this field include sediment ratio delivery (SDR) in the field of geomorphology (Hairston 1995), the unit hydrograph in hydrology (Sherman 1932) and pedo-transfer functions in soil sciences (Tietje & Tapkenhinrichs 1993).

The change of scale (*up-/downscaling*) is related to problems of aggregation or disaggregation of data. In this regard, the transmission of runoff data measured at the outlet of a watershed up to the whole watershed is an example of upscaling procedures (Fohrer et al. 1999). Sometimes it is necessary to downscale statistical data ascertained for an administrative unit (e.g. federal state) to the lower district level.

Regionalization is the key concept for reaching a compromise solution between scale-specific and cross-scale investigations. Scale-specific investigations have to be applied in the core areas of the different scale levels (mentioned in Table 6.1), and for the transition zones between the specific levels a cross-scale approach is necessary. Thus, a connection between the separate hierarchical levels and hence an uninterrupted systematical reflection can be implemented.

The problem of scale transfer was not realized a few years ago. The reason is that it occurs mainly in the face of heterogeneity (Bierkens et al. 2000). Despite (or maybe even because of) its recent emergence, research into scale transfer in environmental science has led to an enormous amount of different methods and approaches to upscaling and downscaling information. This makes it difficult for a practitioner to see where transfer occurs in the various steps of a research project scale, and what methods of scale transfer are available and should preferably be used for these cases. Bierkens et al. (2000) describe a number of available methods (Table 6.4) integrated into a decision support system for practitioners.

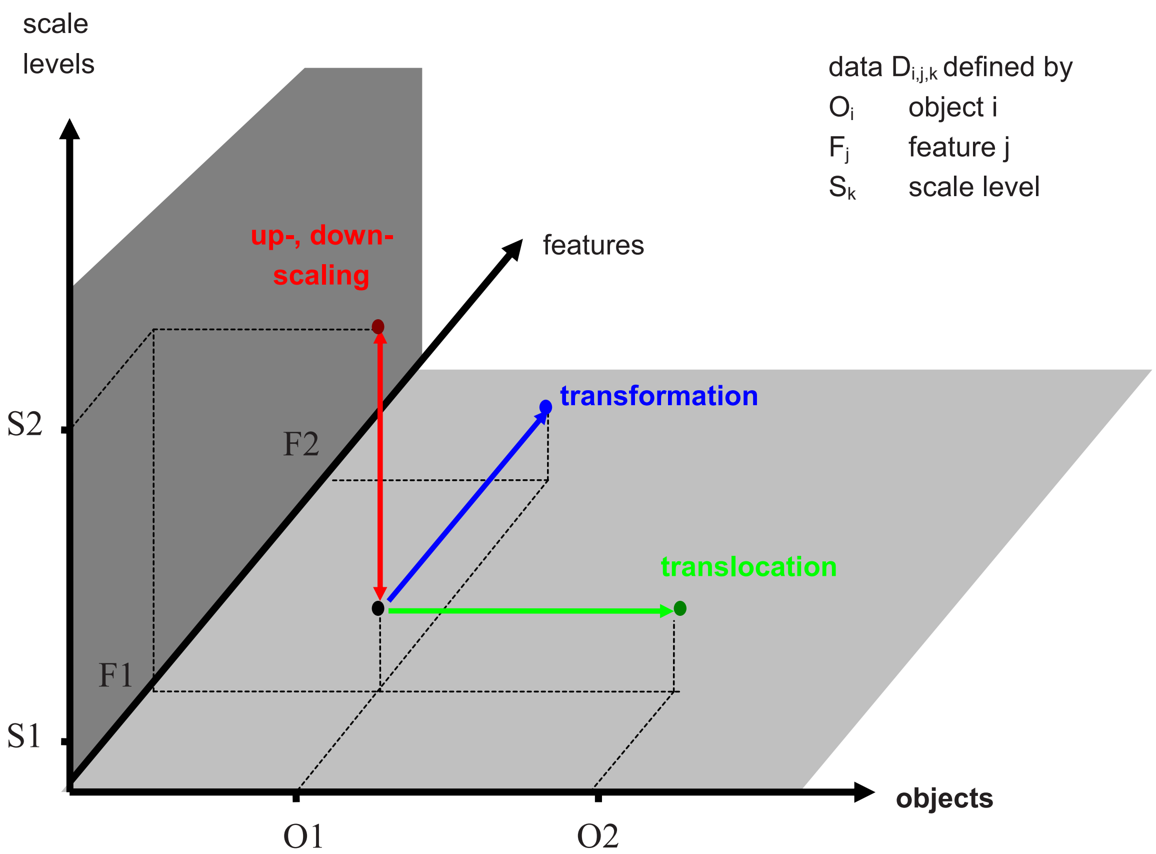


Fig. 6.8: Fundamental operations of regionalization

Table 6.4: Classification of upscaling and downscaling methods (after Bierkens et al. 2000)

UPSCALING	averaging of observations or output variables	- Exhaustive information (e.g. upscaling of measured daily precipitation to average precipitation over a decade) - Design-based methods (e.g. averaging model output parameters to larger map units) - Geostatistical prediction (e.g. block kriging) - Deterministic functions (e.g. delineation of influence zones around sample locations by Thiessen polygons) - Combinations and auxiliary information (e.g. stratified block kriging)
	finding representative parameters or input variables	- Exhaustive information (e.g. finding representative hydraulic conductivity for numerical block models) - Deterministic functions (e.g. spline interpolation, inverse squared distance weighting) - Indirect stochastic methods (e.g. estimating the statistics of the stochastic function from a limited number of observations) - Direct stochastic methods (e.g. estimating statistical properties (e.g. mean, covariance function) from the observation) - Inverse modelling (e.g. finding representative parameters)
	averaging of model equations	- Deterministic: temporal or volume averaging (e.g. estimating the uptake of the whole root system by averaging the uptake simulated for one root) - Stochastic: ensemble averaging (e.g. one-dimensional steady-state groundwater flow in a heterogeneous porous medium)
	model simplification	- Lumped conceptual modelling (no standard solutions exist) - Meta-modelling (e.g. calibrating parameters for a black-box model through regression)
DOWNSCALING	empirical functions	- Deterministic functions (e.g. splines, linear functions, general additive models) - Conditional stochastic functions (e.g. using stochastic wavelets) - Unconditional stochastic functions
	mechanistic models	- Deterministic functions (e.g. adjusting parameter values or boundary conditions of mechanistic models) - Conditional stochastic functions (e.g. constructing equally probable realizations of a stochastic function by adding a noise component) - Unconditional stochastic functions
	fine scale auxilary information	- Deterministic functions (e.g. determining the fine-scale variability of water storage in a sloping landscape using fine-scale topographic data and a value for the over-all water storage) - Conditional stochastic functions (incorporating the ensemble of equiprobable functions instead of only one deterministic function) - Unconditional stochastic functions (e.g. determining the probability density function at the detailed scale directly from the coarser scale)

6.5 Examples for cross-scale and scale-specific investigations

Watersheds are affected by uncertain and complex interactive environmental and socio-poltical trends, some of local and many of regional or even global origin. Our ability to sustainably manage our natural resources is presently still constrained by our lack of knowledge about the hydrologic cycle and its relationship to the geosphere and the biosphere. However, water serves - at least in the temperate climates - as the most important carrying medium of all transport processes (cf. Chap. 7). Hence the watershed approach to the long-term research and monitoring of areas characterized by a high intensity of land use provides important data on ecosystem processes and interactions for detecting both spatial and temporal change in management practices as well as in environmental conditions. Watershed ecosystem studies are based on the collection of long-term data sets of the ecosystem conditions. At this spatial level, research and monitoring contribute to the accumulation of important baseline information on deposition, meteorology, hydrology, ecosystem functioning and land use. The data collections allow the partitioning of cause and effect relationships of ecological and management changes within watersheds. Currently, the investigations are focused on developing, testing and implementing state-of-the-art methods and procedures for application to improve water and land resource management at both the local and regional levels.

The quantification of the hydrologic cycle and chemical fluxes are the major objectives of the watershed programme. Such measurements, when combined with other geographic resources data (e.g. geology, land use, topography, historic and prehistoric records), permit a better understanding of ecosystem-level processes and how watershed ecosystems respond to various natural and human-induced stimuli. During the initial years a core set of variables to be monitored was defined, and sampling and database methods were established. Variables included precipitation, climate, vegetation, soils, hydrology and management practices.

Our integrated approach combines research, inventory, and monitoring within a focused programme for the collection of these data needed to test hypotheses regarding the contribution of human-induced stress to long-term ecological change within agricultural landscapes. To document the relationships between ecosystem effects and anthropogenic influences, long-term monitoring and research are essential. The existence of sites with a commitment to gathering long-term ecosystem level data permits research activities aimed at testing hypotheses relevant to ecosystem processes and structure.

Combination of top-down and bottom-up approaches

Combining ‘top-down’ and ‘bottom-up’ approaches in addition to coupled GIS-model applications and traditional classification and assessment methods appears to provide a way of investigating landscape-ecological structures and processes. Top-down approaches include the use of the inquiry function of the GIS to detect areas that can be defined as potential risk zones with vertical/horizontal material (and nutrient) leaching from agricultural areas. Assessment on this scale level is a rough filter that provides background information and identifies the properties for subsequent analysis. GIS can be used as a powerful tool to provide a process-based landscape typification. The outcome is units with similar conditions, characterized not only by their specific combination of structural components (grain size, slope angle, biotope type, etc.) but by the dominant processes (overland flow, macropore fluxes, percolation, interception, etc.). These process-based units can further indicate the dominant process direction (lateral - vertical) and determine the neighbourhood effects (either the adjacent or the upper/lower layer).

Going in the other direction, in a bottom-up approach the risk areas identified have to be investigated in detail - with other models and a database with a higher spatiotemporal resolution. For this purpose the scales have to be changed. Vertical and lateral water, material and energy fluxes from the designated risk areas can be qualified and quantified. By using a nested approach in small test areas, indicators for sustainable land use systems can subsequently be identified that can be applied to larger areas afterwards.

Examples of specific applications of ‘top-down’ (balancing - modeling - typifying) and ‘bottom-up’ (measuring - mapping - modelling) are discussed in Chap. 7.

6.6 Discussion and conclusion

Based on the fundamental theory of scales and dimensions in landscape ecology, our research components will integrate a series of analyses and assessments designed to create a rigorous context for decision making. We will apply quantitative tools and information systems as well as traditional methods to enable critical interpretation of the uncertainty associated with decisions about future alternatives. We will try to employ and combine three major research approaches and combined steps:

(1) Characterizing landscape status and changes (Chap. 3, 4 and 7);
(2) Identifying and understanding critical processes (Chap. 7); and
(3) Evaluating outcomes (Chap. 8, 9 and 10).

(1) Characterizing status and change

Research will assess trajectories of change from now through alternative future scenarios. If possible, the investigations should be extended to historical situations (retrospectively). The approach should be guided by initial assessments which will influence future research and environmental management. Assessment approaches should:

- Describe historical change;
- Describe current condition and function;
- Identify biophysical and socioeconomic processes and functions that constrain possible future ecosystem trajectories;
- Characterize the level of rigour and the uncertainty and unknowns in the assessment.

(2) Identifying and understanding critical processes

We try to select and apply a set of conceptual, quantitative and evaluative models to identify and analyse critical anthropogenic and non-anthropogenic processes related to ecosystems. This phase of research will:

- Identify critical ecological (biotic), environmental (physical and chemical) and socioeconomic (individual, domestic and institutional) influences on ecosystem structure and function;
- Select indicators in each process category that quantify the magnitude of response to these influences on ecosystems;
- Integrate quantitative tools, information systems and qualitative understanding to describe system responses both within and across process categories.

(3) Evaluating outcomes

We will evaluate and illustrate the social and ecological consequences of potential management practices for future landscape conditions. Furthermore, we will describe sources and levels of uncertainty and define likely boundaries of ecosystem and socioeconomic trajectories. This phase of work will entail:

- Creating alternative futures that illustrate the major strategic choices and explicitly identify the likelihood and advantage of relevant choices;
- Evaluating the consequences of alternative futures on critical anthropogenic and non-anthropogenic processes including the characterization of risks, technical limitations, scientific uncertainty and public response to alternative futures;
- Using different forms to present the results of these evaluations.

6.7 Outlook

Despite all the advantages of hierarchical theory, as well as of its application in landscape-related sciences and spatial planning, there are still many unresolved issues. Therefore, future research must focus on two directions. Firstly, a general theory of hierarchy that is acknowledged through all disciplines of geosciences or landscape-related research must be established. This includes a common use and understanding of terms related to scale and hierarchy, as well as the consideration and application of this theoretical background to the problem to be solved. Secondly, the great (technical) progress made in the development of tools for landscape analysis (GIS, remote sensing, modelling) must be critically examined. All these tools are often applied without respect to scale-related questions.

Bearing these problems, the following questions have to be solved:

- What properties of physical and human systems are invariant with respect to scale?
- What kinds of transformation of scale are available to aggregate or disaggregate data in ways that are logical, rigorous, and well-defined in theory?
- Is it possible to implement methods which assess the impact of scale through measures of information loss or gain, for example?

- How is the observation of processes affected by changes of scale, and how can we measure the degree to which processes are manifested at different scales?
- How is scale represented in the parametrization of process models, and how are models affected by the use of data from inappropriate scales?
- What is the potential for integrated tools to support multiscale databases, and associated modelling and analysis?
- What are the problems that must be resolved when integrating data from different scales?

Especially with respect to the inseparable coherence between nature and society, the following questions need to be answered:

- How can we *measure changes* in ecosystems across scales from individual sites to large basins or regions?
- What is the *current state* of ecological resources within a given region?
- How do *natural* patterns and processes of landscapes or ecosystems *interact with anthropogenic* patterns and processes?
- What *types of interactions* are consistent and what types are contradictory?
- What critical yardsticks are there for comparing and contrasting various alternative *future scenarios* (e.g. biological/ecological, economic, climate/hydrologic, demographic)?
- What *indicators of climatic/hydrologic/geomorphic processes* or components are most useful, meaningful and tractable for describing the historical condition, current status, and alternative futures across multiple spatial scales in a certain area?
- What *indicators of demographic and economic processes* or components are most useful, meaningful and tractable for describing the historical condition, current status and alternative futures across multiple spatial scales in a certain area?
- What *environmental management options* are available to alter future ecosystem conditions across a range of spatial scales?
- How can natural processes and human programmes be used to *maintain or restore ecosystem* processes and patterns?
- What *fundamental limits* govern the achievement of ecosystem management objectives?
- How can human efforts be designed to *enhance natural processes that restore ecosystems* and recognize the ecological benefits of future disturbances?

All the above tasks have to be solved with respect to spatial planning which is also organized hierarchically (Table 6.5., Chap. 9).

Table 6.5. Hierarchies in landscape and spatial planning in Germany (after Kiemstedt et al. 1997)

Scale level	Planning level	Spatial planning	Landscape planning
Macroscale	Country	Spatial development policy	-
Mesoscale	Federal state	*Raumordnungsprogramm*	*Landschaftsprogramm*
	Region	*Regionalplan*	*Landschaftsrahmenplan*
Microscale	Municipality (town, village)	*Flächennutzungsplan*	*Landschaftsplan*
	Parts of municipalities	*Bebauungsplan*	*Grünordnungsplan*

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7 Landscape balance

Martin Volk, Uta Steinhardt

7.1 Introduction

The characteristic distribution of the landscape's components land use, land cover, soil, morphology, hydrology, climate, geology, etc., forms the landscape structure. These components are interrelated by fluxes of water, material, energy and information (landscape-ecological processes), which result in the 'landscape balance'. This term is based on the German concept of the *Landschaftshaushalt*, which describes the associations between the geoecofactors in a geoecosystem due to the laws of nature (Leser 1997, Marks et al. 1992, Troll 1939, Zepp & Müller 1999)[1]. The geoecosystem is regarded as an 'open system' characterized by an equilibrium of flows, with input and output interactions with the landscape balances of the adjacent geoecosystems (the environment). Despite the dimension of the landscape ecosystem, a model of the landscape balance can be created for any order of magnitude. In doing so, methodological extensions or limitations arise for the different dimension steps. Several problems have emerged with the development of scale-specific methods, the improvement of knowledge about the interactions between landscape structure and landscape-ecological processes and the processual interactions and changes within the landscape ecosystem itself at different dimensions and scales. These questions become even more important when considering the impact of land use and its changes on the landscape balance and its assessment as a basis for a sustainable development.

Human impacts - such as land use - affect the interactions within a landscape ecosystem by changing the landscape structure and thus altering conditions for landscape-ecological processes. The human factor 'land use' within the complex ecosystem has a strong impact on the adaptability, regeneration and regulation capability of the landscape balance. It should be mentioned in this context that it is still a problem to assess the adaptability and dynamics of the landscape balance as a reaction to human impacts (feedbacks) within landscape analysis owing to the lack of knowledge about these interactions (Fig. 7.1 and Fig. 7.2). As most of the relevant processes in the landscape depend mainly on the mobile agent water, they have influences ranging from small to large scales. However, understanding of

[1] Schmithüsen (1973) transferred the theories and considerations of thermodynamics and synergism into geography with the term 'geosynergetic landscape research', which describes the totality of all interactions within a landscape (cf. also Müller 1999). Neef (e.g. 1973) also largely developed the system theories for 'his' landscape research on the basis of such knowledge.

these processes - especially on large scales - is still insufficient, as most of the processes take place on small scales. Concepts for sustainable development have to consider the implementation of information about the landscape balance on all scale levels. Special attention should paid to larger scales because most of the environmental conflicts and changes become apparent on the landscape scale. However, most of the useful methods for the analysis and assessment of landscape-ecological processes and parameters are limited to scales up to 1:25,000, and the importance of the parameters - and the parameters themselves - are limited to changes in a hierarchical spatio-temporal way.

To solve these problems, the following questions should be asked:

- *How does the importance of parameters (as well as the parameters themselves) of their landscape balance components (morphology, soil, hydrology, soil, hydrology, land use and cover and climate) change on different scales?*
- *How does the impact of changes to the landscape structure (especially land use) affect the water, material, energy and information fluxes (horizontal and vertical) on different scales?*
- *How does the land use influence the quality and quantity of soil and water?*

This also requires characterizing the processes concerning extension, duration, intensity and continuity - and improvingknowledge about possible feedback. The complex interactions of the landscape balance within the landscape system and the problems of its investigation and assessment are shown in Fig. 7.1. Fig. 7.2. contains an example of positive feedback.

In this paper, several national and international approaches and models for these investigations are presented. Finally, our hierarchical approach is described for mesoscale application. In addition, suggestions are made for the verification of large-scale calculations. For integrated landscape analysis, we aim to combine both ‘top-down’ and ‘bottom-up’ approaches with GIS-coupled model applications and traditional methods (e.g. mapping, measuring, etc). Using traditional methods is an essential part of verifying modelling results, as well as for improving knowledge of how landscape ecosystems function.

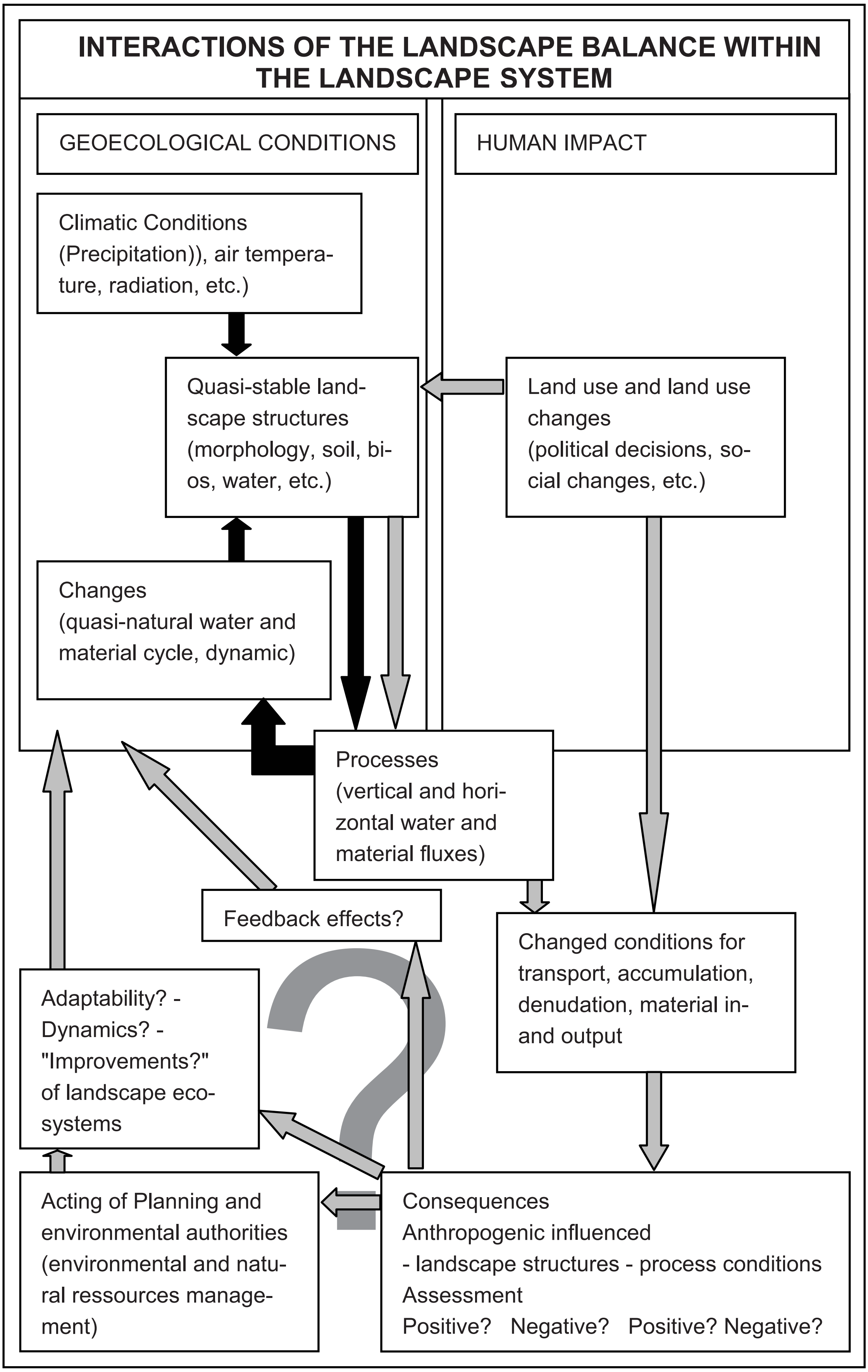

Fig. 7.1: Interactions of landscape balance within the landscape system and the problems of its investigation and the assessment of human impacts. This is even more difficult considering the fact that the intensity of the interactions and influences depend on time and the scales concerned.

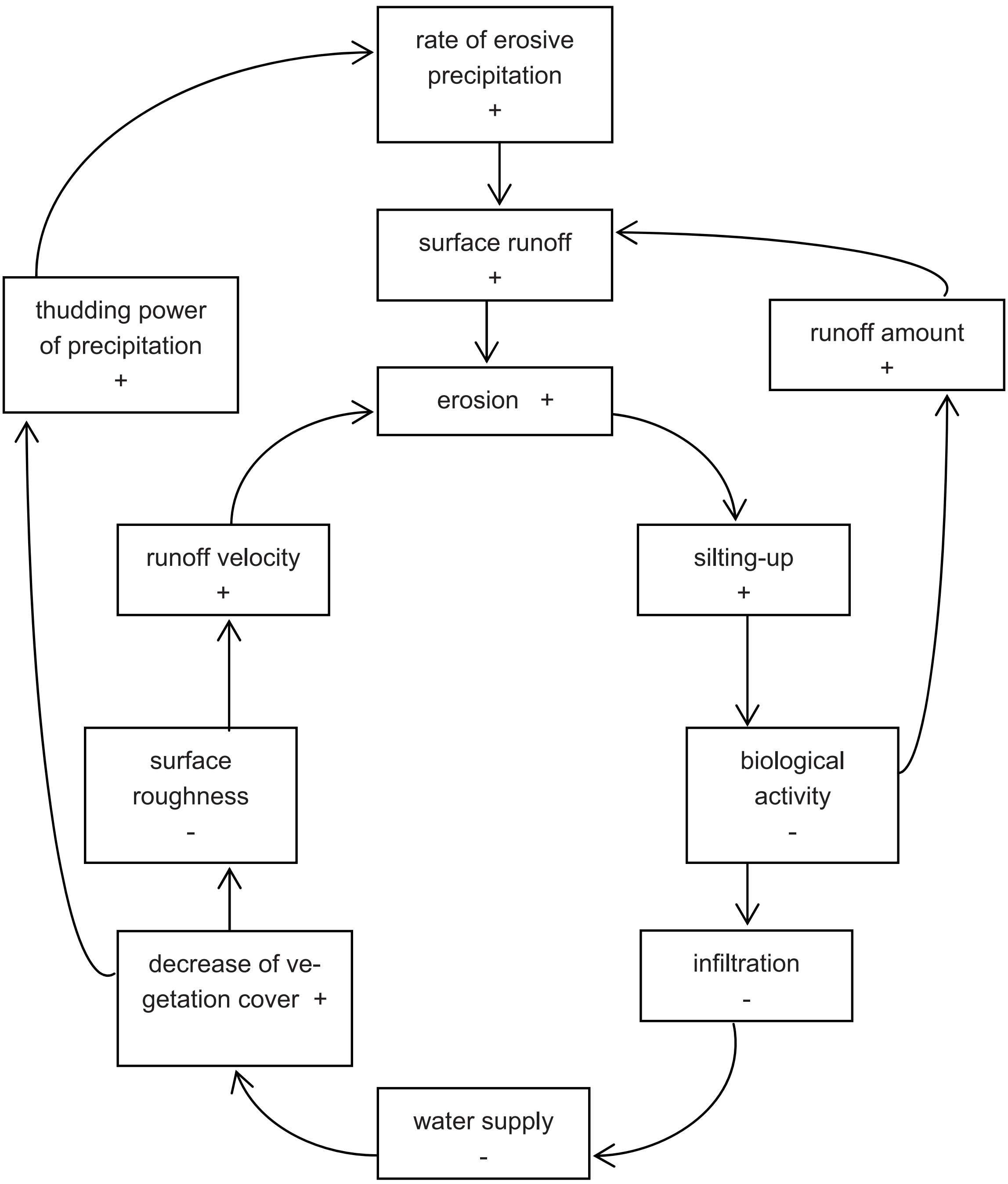

Fig. 7.2: Example for positive feedback: increased erosion and surface runoff caused by a decrease of the vegetation cover (+: increase; -: decrease) (after Rohdenburg 1989).

7.2 Interactions between landscape structures and processes

Due to its importance, some of the earliest works in ecology focused on understanding the relationships between patterns and processes at landscape scales (Cowles 1899, Cooper 1923, Gardner & O'Neill 1991). Klug & Lang (1983) described the investigation of the structure, functioning and dynamics of natural systems and their anthropogenic-technogenic transformation as a main task of geosystems research (see also Turba-Jurcyk 1990).

As mentioned in the introduction, the landscape structure causes different process conditions. "Despite the wealth of empirical and conceptual investigations that have been carried out since these early studies, the problem of predicting ecological processes at broad scales remains largely unsolved. This lack of resolution is due, in part, to the complexity of the problem and an intellectual tradition that has assumed that detailed measurements of fine-scale processes are necessary to predict broad-scale patterns." (Gardner & O'Neill 1991). This statement highlights the connection between landscape structure processes and the problem of scales in landscape ecology. In relation to this topic, special attention is given to the narrow, linear transition zones from one ecosystem to another, which are known as 'ecotones' (i.e. Jedicke 1994). These zones are characterized by a rapid change of the environmental conditions and site factors within a small area. Besides their importance for biodiversity (i.e. the "edge effect", Jedicke 1994), the main functions of ecotones consist in the protection of adjacent ecosystems, i.e. protection against unwanted material, nutrient and water fluxes (barrier effect, buffer function; Bastian & Schreiber 1999). Negative tendencies of land use development can reduce these transition zones and lead to drastic impacts on the living conditions for plants and animals, as well to completely changed fluxes of material, nutrient and water - which influences our natural resources such as water and soil, and in turn the mentioned living conditions for plants and animals (Plachter 1991). Due to the importance of ecotones for both biodiversity and as transition and buffer zone for processes (e.g. the influence of ecotones on flows of energy, material, organisms and water) within the landscape (structure), several publications deal with these topics (Hansen & di Castri 1992, Hansen et al. 1992). Delcourt & Delcourt (1992) tackles ecotone dynamics in time and space, while Weinstein (1992) suggests methods and models for monitoring ecotones to detect global change at different scales (see also Naveh & Liebermann 1994).

One of the most important factors in relation to the landscape structure is georelief. In the landscape balance, georelief is a regulation factor and a structural area in or on which landscape ecological processes act (Leser 1999). Problems arise in connection with the fact that most of the process factors on the landscape scale - e.g. soil erosion - are mainly or partly derived from structural information such as soil maps etc. However, these methods do not enable the deduction of information about the processual and structural transformations at the structural boundaries between different landscape types. In the authors' opinion, due to the morphological conditions of a landscape (and thus with its changing conditions of soil, vegetation, micro- and mesoclimate, etc.), process-structural transformation zones (vertical to horizontal or reverse, interflow, etc.) of different widths have to be defined. This means that these transformation zones are narrow in landscapes with - for example - alpine conditions with high morphological energy, and wide in gently sloping or flat areas with low morphological energy. On the other hand, ‘sharp’ boundaries can often be observed in for example relatively small fluvial plains with a mainly ice-age glacial genesis. In these areas, both the heterogeneous distribution of the substrate and small differences in the morphological conditions of the surface can cause these boundaries.

A useful step to improve knowledge about the complex interactions between landscape structures, biodiversity and relevant processes appears to be to assess landscape structures with landscape metrics derived from satellite images (Turner & Gardner 1991, Antonova 1998, Boyce 1998, Richards 1998, Wallin 1998, Wallin & Boyce 1998, Lausch 1999, Lausch 2000)[2], which also allows the monitoring and documentation of land use changes and their impact on the landscape structure during time steps. Coupled with ‘context-related’ methods of digital image processing (improving land use classification with phenological and DEM-based morphological indicators, ‘hydrological remote sensing’, etc.), these studies seem to allow a connection with the more process-oriented studies. Finally, this method should allow the transferability and applicability of integrated models to the specific natural conditions of an investigated landscape to be examined. This is important in view of the fact that most of the models and algorithms applied are developed for specific research fields and special study areas with very different conditions in comparison to their own study areas. This approach is pursued by the authors and described in Chap.7.5.

[2] Even Troll (1939) mentioned the importance of interpreting aerial photographs for integrated landscape ecological analysis.

7.3 Fluxes of matter, energy and information in landscapes

Due to the complex processes of matter and energy transformation in landscapes, special attention is paid to the water as an essential element and a mobile agent which is the main transport medium in temperate climates. At first our investigations will concentrate on abiotic components of the landscape balance. Keeping its varied interactions with the bios in mind, these processes cannot be understand as purely abiotic (Finke 1994, Wohlrab et al. 1999). This is being tackled by other groups from our department (Lausch 1999, Lausch 2000; see also Chaps. 4 and 5).

7.3.1 Vertical and horizontal fluxes and processes

One of the most important and 'hottest' topics in landscape ecology is differentiation between vertical and horizontal fluxes and processes in landscapes. Most of the process-oriented investigations are concentrated on very small locations, which have all resulted in an improvement in the understanding of the horizontal and especially the vertical processes on the microscale. On this scale, the process system can be characterized as mostly vertical, whereas on the mesoscale horizontal processes are at the focus of consideration (Leser 1997). Schmidt (1978) pointed out that the biggest problem results in the transformation of the rare information about the horizontal processes into natural areas recorded with static methods. Bearing in mind that Schmidt's statement is related to the microscale, this remains valid nowadays, despite several works dealing with theoretical aspects, the improvement in field analysis and 'scale-transferring' techniques concerning this problem.

In an attempt to solve these problems, Menz & Kempel-Eggenberger (1999) combined two landscape-ecological methods on two different scale levels. On the microscale, they followed the concept of the landscape ecological complex analysis, with time-dynamic measurements of the landscape's water and material fluxes under climate-, bio- and hydroecological aspects (Leser 1991). The main step of these investigations is complex local analysis with the conceptual model 'local site regulation cycle' (*Standortregelkreis*, Chorley & Kennedy 1971, Mosimann 1978). This theoretical model includes different spatio-temporal dimensions for the measurements in the study area and should be the basis for upscaling the material fluxes and transformations. This means that material fluxes passing from one spatial dimension to the next higher level do not remain in one dimension. This results not only in a change of the transport direction from vertical to lateral, but also in an overstep to another spatio-temporal transformation network (e.g. the local fluxes are concentrated on direct surface runoff, after a short time they reach the drainage channel of the catchment, and hence a larger transformation network - 'positive feedback'[3]). On the other hand, the lapse of the local fluxes to the subsurface is defined by them as 'negative feedback' Because of the problems of

[3] The definition of 'positive feedback' has a different meaning here compared to Rohdenburg (1989).

transferring local process, information on larger areas - whose complexity is caused by heterogenization (Herz 1994) - they suggest the development of hypothetical key factors or connecting links between the different scale levels. Therefore, the coincident method used in this approach is the development of a digital geoecological risk analysis for a micro- to mesoscale application ("method of processual geoecotop and geoecochoric[4] segmentation"). This method is based on a typological classification of homogeneous assessment units considering processual characteristics as described by Leser & Klink (1988). The data basis of this integrated analysis comprises field data and maps (i.e. substrate, soil type, pH, soil depth, vegetation, land use and land cover, climate, etc.), and morphological parameters derived from a DEM. From the processes, it is possible to derive processual areas (material and nutrient balance, water balance, aerial balance, radiation balance). These process-based models are re-classified and form the basis for deriving ecological indicators (i.e. soil denudation, low soil depth, soil depth, pH, frost risk, etc.). The combination of structural and processual parameters and the application of classification and assessment methods (e.g. Marks et al. 1992) allows the designation of ecological risk zones (sensitivity of landscape to natural and anthropogenic impacts). Depending on the appropriate scales, the derivation of different information is possible. Transfer to larger areas (regions) is possible by modifying the given classification and assessment methods. Besides the problem that there is less information about the process dynamics and process behaviour in these structure-oriented studies, most of the given assessment methods are only valid for scales up to 1:25,000. Nevertheless, Menz & Kempel-Eggenberger (1999) suggest combining these two methods as a basis for the definition of connecting links between the dimensions that allow a scale-specific characterization of the process transformations (although the links are not defined in the publication).

Considering the above-mentioned studies, the following facts should be pointed out. The importance of ecological - and also socioeconomic - parameters changes depending on the spatio-temporal scale level concerned. For instance, within the material transformation process, a change of the scale level can happen - not only as 'downscaling' (top-down), but also as 'upscaling' (bottom-up), if material fluxes overstep under self-intensification into the next higher or lower dimension ('interflow-network'). The present focus on 'upscaling' methods is caused by small-scale concentrated research over decades. By combining both 'top-down' and 'bottom-up' approaches, it seems possible to link both methods. By doing so, suggest the authors, a contribution to the solution of problems related to the questions of continuous scale level transitions and the knowledge of horizontal processes on different scales can be expected (Mosimann 1999, Steinhardt & Volk 2000). It should be pointed out here that such approaches are very important for the progress of scale-related landscape-ecological research, considering questions about system behaviour, adaptability, feedback mechanisms, hierarchies, synergy, etc. The remaining problem is still the definition of the links between the different scale levels. Another question is the degree to which these 'philosophical', very difficult and complex system approaches have to be simplified for application to, for instance, environmental planning. Here, a combination with more practical ap-

[4] The concepts of topes and chores is explained in Chap. 6 of this book.

proaches - as is suggested in Bierkens et al. (2000) - is conceivable and should be aimed at.

7.3.2 Scale specific and cross scale investigations

The consideration and transferability of processes, local conditions and assessment methods to different scale levels plays a central role in landscape-ecological science as well as in planning practice. The main problem here is - as also in many other fields of landscape ecology (Müller & Volk 1998) - the lack of general theories allowing the derivation of rules for regionalization (Richter et al. 1997, Steinhardt & Volk 1999). Thus, two fundamentally different opinions can be pointed out:

- The need for methods that allow a transfer of locally valid (small-scale) information and results to larger areas through suitable indicators or transfer functions (Scheinost 1995, Tietje & Tapkenhinrichs 1993);

or

- The need for special methods on all scale levels.

The two cross-scale approaches mentioned are special studies in the field of soil sciences, with limited possibilities for transfer to other larger areas and fields of landscape ecology. Due to the current lack of cross-scale methods, scale-specific methods should be preferred. Currently, most of the studies in landscape ecology are concentrated on ‘bottom up’ approaches (King 1991, Meyer 1997, Gerold 1999, Diekkrüger 1999), which are defined by the translation or extrapolation of information from small scales to larger landscape or regional scales. King (1991) describes several methods for these ‘scaling up’ approaches. The inverse approach of ‘top down’ or ‘scaling down’ approaches, which can be defined as the translation of information from larger to smaller scales, is less commonly practised in research (it is mainly confined to climatological topics, such as predicting landscape response to climatic change by means of climate models, i.e. Gates 1985). Bierkens et al. (2000) suggest a theoretical framework for the large number of upscaling and downscaling methods used in environmental science. These methods are designed to help the practitioner assess whether scale transfer occurs in the research project, and if so, exactly where, and to descide what upscaling or downscaling methods are suitable for performing these instances of scale transfer. Their book includes a CD-ROM with a simple Decision Support System (DSS), which helps choose the most appropriate upscaling or downscaling method depending on the ‘research chain’ of a project. The book is designed for applied research with a practical approach, and does not deal with more ‘philosophical’ approaches to scale, such as hierarchy, organization and synergy, etc. Steinhardt & Volk (Chap. 6) passed general comment on the above-mentioned regionalization problems. In the authors’ opinion, both approaches - ‘top down’ and ‘bottom up’ - (along with their specific features) - are necessary and have to be combined in order to achieve an integrated landscape analysis for all spatial scales (cf. the following Chaps., see also Wrbka et al. 1999). Hence scale-specific approaches have to be applied to the “core zones” of each scale, and cross-scale investigations are used in the “transition zones” between the scales due to the loose coupling (Wu 1999) between the scales to guarantee a holistic consideration of landscape.

7.4 Water-carried fluxes of nutrients and pollutants

The outwash and transport of material, nutrients and pesticides is mostly linked to an amount of water flowing out of a region. This results in an input of this material into the groundwater and surface water. The investigation of these processes is often concentrated on phosphate (particle-bound transport through erosion: lateral processes) and nitrate (soluble transport through seepage: vertical processes). Because of the huge amount of different pesticides and the resulting complex chemical analysis, work needs to concentrate on estimating the spatio-temporal input behaviour of selected relevant pesticides (i.e. Grunewald et al. 1999).

7.4.1 Investigation methods

A comprehensive description of methods for landscape-ecological analysis applied in Germany is given by Bastian & Schreiber (1999) and Zepp & Müller (2000). The most highly developed investigation methods are for small-scale studies, including several proposals recommended for methodological standards in mapping, measuring and assessing up to a scale of 1:25,000 . There is still a lack of homogenous approaches for the various investigation methods for mesoscale and macroscale integrated landscape analysis (Lenz 1999).

7.4.2 GIS-coupled modelling on different scales

Most of the nutrient load of surface waters originate from non-point sources. To analyse these processes, the application of distributed parameter models in combination with geographical information systems (GISs) seems to be a useful method. Special attention has to be paid to the spatial variability of the landscape characteristics and their influence on the transport of water and nutrients within a given area. At present, many of the physically based approaches with a high spatio-temporal resolution cannot be effectively applied to medium-sized watersheds (Grayson et al. 1992), for example, because of the huge amount of input parameters required. Despite the much greater effort needed to parameterize, validate and run physically based models, simulated results often provide only slightly better or sometimes even worse correspondence with measured values than lumped-parameter models (Seyfried & Wilcox 1995). In this context, it should be mentioned that most of the common empirical models employed by environmental and planning offices and authorities rarely use more than three parameters (Hauhs et al. 2000).

Bearing these problems in mind, several models have been tested for their scale-specific applicability with respect to the time schedule and topics of research projects (Krysanova et al. 1996).[5] As most of the models were developed within research projects carried out in specific study areas, the possibility of transferring these methods to other regions needs to be tested. Table 7.1 gives an overview of several models, their capacities and operations, and their scale-specific applicability. These models have been selected by the authors to test their scale-specific applicability. More models are listed and described by Bork & Schröder (1996) and Grunwald (1997).

All input data used for model applications have to be prepared and modified depending on the specific calculation characteristics of the models (cp. Petry et al. 2000, Volk & Steinhardt 1998). This is also important for deriving indicators for environmental conflicts, land use, water balance and morphology interactions in catchment areas. One main problem of large-scale investigations is verifying the results. As measured data are mostly unavailable, the investigations have to be hierarchically linked to studies on smaller scales (sampling and analysis at representative locations, mapping, measuring, application of small-scale models, see above). Nevertheless, the application of these traditional methods is essential not only for verifying the modelling results, but also for improving basic knowledge about how the landscape ecosystem functions (Hauhs et al. 2000).

[5] Before applying a model, the algorithms used have to be checked. For example, most of the models that have an erosion component are based on different versions of the USLE (Bork & Schröder 1996). It seems important to be able to adapt the model algorithms to the specific conditions of a study area.

Table. 7.1: Selected models and their scale-specific applicability.

Model system	Scales	Objectives, operations and capacities
SWAT (cp. Arnold et al. 1993, Srinivasan & Arnold 1993)	Large river basins, subbasins (up to several thousend square miles)	• Predict the effect of management decisions on water, sediment, nutrient and pesticide yields with reasonable accuracy on large, ungaged river basins. • Daily time step to long term simulations • Groundwater flow model • Basins subdivided to account for differences in soils, land use, crops, topography, weather, etc. • SWAT accepts output from EPIC (see below) • SWAT accepts measured data & point sources • Soil profile can be divided into ten layers • Water can be transferred from channels and reservoirs • Basin subdivided into subbasins or grid cells • Nutrients and pesticide input/output • Reach routing command language to route and add flows • Windows/ArcView Interface • Hundreds of cells/subbasins can be simulated in spatially displayed outputs
ABIMO (cp. Glugla & Fürtig 1997, Rachimov 1996)	Meso- to macro-scale	• Description of the basic elements of the water balance on the landscape scale (long-term values of runoff and evapotranspiration). • "Mean runoff" is defined here as the difference between long-term mean annual precipitation and real evapotranspiration. This difference is equivalent is to the total runoff. In the case of a solely vertical seeping of the water this value corresponds with the groundwater recharge. • Thus, the value must be understand as the sum only indifferent of both surface and subsurface runoff. Therefore, the results have be modified with a runoff quotient (based on slope inclination and groundwater level) after Röder (1998), which allows an estimation of the surface runoff and interflow.
AGNPS-WaSim-ETH Young et al. 1987, Schulla 1997).	Mesoscale watersheds	• System of computer models developed to predict non point source pollutant loadings within agricultural watersheds. It contains a continuous simulation, surface runoff model designed for risk and cost/benefit analysis.

		• The set of computer programs consist of: • input generation & editing as well as associated data bases; • the "annualized" science & technology pollutant loading model (AnnAGNPS); • output reformatting analysis; • the integration of more comprehensive routines (CONCEPTS) for the instream processes; • an instream water temperature model (SNTEMP); • several related salmonid models (SIDO, Fry Emergence, Salmonid Total Life Stage, & Salmonid Economics). • The application of AGNPS (Agricultural Nonpoint Source) can be used for the caculation of soil erosion, sediment transport and nutrient yield (N and P). • The computed runoff and peak flowof the hydrological model WaSim-ETH will be used as input for the linked model AGNPS.
CANDY (cp. Franko et al. 1997)	Lower mesoscale, Farm level, fields	• Analysis of vertical carbon-, nitrogen and water fluxes between crops, soil, and groundwater; daily time step, consideration of land management practices
EPIC	Farm level, fields	• Capable of simulating the relevant biophysical processes simultaneously, as well as realistically, using readily available inputs and, where possible, accepted methodologies; • Capable of simulating cropping systems for hundreds of years because erosion can be a relatively slow process; • Applicable to a wide range of soils, climates and crops; • Efficient, convenient to use, and capable of simulating the particular effects of management on soil erosion and productivity in specific environments. • The model uses a daily time step to simulate weather, hydrology, soil temperature, erosion-sedimentation, nutrient cycling, tillage, crop management and growth, pesticide and nutrient movement with water and sediment, and field-scale costs and returns.
E3D (cp. Schmidt 1991, Von Werner 1995)	Subbasins, farms, fields	• Simulation of erosion processes during single strong rain events as well as for the calculation of annual or several year values. • Based on a physical approach, characterized by a high spatio-temporal resolution and short calculation times

7.5 Theory: A scale-specific hierarchical approach

The terms ‘sustainable development’ and ‘sustainability’ as well as ‘ecological’ are buzzwords which are being increasingly used and misused throughout society. These terms are frequently not understood in the sense of their original meaning as conceptual ideas, as was presented in the Brundlandt Report (Goodland 1992, Gore 1992). According to these ideas, the concept of sustainability is related to the whole earth. However, the realization of this concept requires a hierarchical approach with solutions for all spatial scale levels - from local to global. This must be seen in connection with the fact that human impacts - like land use - affect the landscape balance over a broad range of spatial scales (i.e. use of natural resources, input of agrochemicals, etc.). Several papers from different scientific disciplines deal with the hierarchical organization of nature (Burns et al. 1991, O’Neill et al. 1986). The hierarchical concept was introduced into German landscape ecology by Neef (1967) and continued by several other landscape ecologists (Leser 1997). An overview of hierarchical concepts in landscape ecology is given by Klijn (1995). These concepts are mainly focused on the hypothesis that each of these scale levels (micro-, meso- and macroscale) is characterized by specific temporal and spatial ranges. Due to the size and internal differentiation of the spatial reference level, different proceeds of cognition are related (spatio-temporal hierarchies of landscape-ecological processes). Consequently, each scale level needs layers (and indicators) with suitable spatio-temporal resolution and specific investigation methods, and provides specific information (Steinhardt 1999a, Steinhardt & Volk 2000). Regarding the increasing application of geographical information systems (GISs), this is often limited to consideration of the spatial resolution of the data layer. Technical problems caused by the huge amount of data used and the computer memory required are often the limiting factors for the application of high-resolution data. Despite all the advantages of these systems, GIS users often run the risk of applying a number of formal procedures which often do not help improve knowledge about the process behaviour in landscapes. The derivation of slope curvatures from digital elevation models (DEMs) with different spatial resolutions may be an example of those doubtful applications (Steinhardt & Volk 1998). Therefore consideration of scale-specific methods needs to be underlined. It is at this point that we touch upon the difficult field of regionalization. On the one hand, although small-scale investigations are useful and important for improving our knowledge of processes, they depend on complex, laborious methods such as detailed measurement or mapping. Many studies deal with the translation and extrapolation (‘scaling-up’) of these findings to larger scales (‘bottom up’). By contrast, mesoscale and macroscale approaches enable landscape ecological interactions in large areas such as watersheds and regions to be detected while focusing on relevant areas (‘scaling-down’, ‘top down’). This “step in the scale or consideration level” can be illustrated using the example of data layers based on remote sensing data gained from different recording platforms. As the survey altitude above the earth’s surface increases (aerial photographs, satellite images, etc.), the visible of the whole landscape area grows but is accompanied by a loss of detail. Thus, more abstract information has to be used due to the increasing size of a given study area (e.g. the Normalized Difference Vegetation Index).

Regarding the investigation of vertical and horizontal material, nutrient and energy fluxes, we pursue the following hypothesis:

- *The basic components for vertical and horizontal material and energy fluxes - morphology, soil, hydrology, land use/management, and climate - are similar at all scale levels. It is only the importance of the parameters (and the parameters themselves) of these components which change for each scale (Helming & Frielinghaus 1999, Klijn 1995, Steinhardt & Volk 2000, Volk 1999).*

Main topic is here the definition of a linkage between the different landscape scales, as mentioned in the Chaps. before. To give an example for morphology and erosion: On the local scale, surface roughness is one of the main factors that will affect erosion deposition, whereas for larger scales (up to river catchments) the factors slope inclination, slope length, slope exposition up the shapes of streamlet, -net, -order and direction of flow are responsible for erosion processes. With the GIS-coupled combination of both “Top-down” and “Bottom-up”-approaches and more traditional methods, the investigation of the complex interactions between landscape structures and processes, as well as an assessment of the impact of land use changes on the landscape balance, seems to be enabled. This method is prefered by the authors. In the following Chaps., our approach will be presented with some examples. Fig. 7.3 gives an overview about our method.

Preprocessing, derivation of transfer functions and first assessment of the basic parameters

Soil physics	Retention capability	Land use monitoring (rem.sens.-methods)	Landscape structure (remote sensing methods)
• Filter capability of soils • Adjustment of soil mappings • Derivation of soil related paramers for model calculations • Derivation of soil moisture regimes • Differentiation of horizontal and vertical fluxes of water and matter	• Assessment of the retention- and buffer capability of different landscape types considering soil acidification and biodiversity • Source - sink relations	• land use classification considering phenology and socio-economic and geofactors • integration of context related classification methods (i.e. impact of morphological conditions) • changes of land use probability • Derivation of soil moisture regimes	• landscape metrics in relation to the landscape balance • scale dependence • recommendations of landscape structures for a decrease of nutrient and matter outwash

S M A L L   T O   M E S O S C A L E

Mesoscale modeling and assessment

- Description of processes with models - optimization and recommendations of applications
- characterization of processes concerning extension, duration, intensity and continuity, contributions to the analysis of the specific conditions of the water balance and fluxes of material and energy
- derivation of scale-specific parameters for the assessment of the landscape balance
- Assessment of the sensivity and capability of different landscapes - derivation of land use variants for a decrease of nutrient and matter outwash.

Main objectives

- Assessment of the landscape balance on different scales: combination of „top-down“ and „bottom-up“ approaches with GIS-coupled models and more ”classic“ methods
- Quantitative and qualitative informations about material outwash (watersheds) and –input in streams and rivers.
- Integration of the results for the derivation and assessment of landscape functions of soil and water and land use conflicts
- Check out the integration possibilities of the methods and results in spatial planning processes.

Fig. 7.3: Methodological steps for a hierachical, scale-specific landscape analysis of the landscape balance.

7.6 The study areas

The hierarchically nested approach presented here is tested on different scales in various areas of the states of Saxony and Saxony-Anhalt in eastern Germany with different natural and socioeconomic conditions. Therefore, the applicability and transferability of the method has to be checked. Below, the application of the approach is shown using the examples of the Dessau district (administrative unit) in the east of Saxony-Anhalt and on the River Parthe watershed in Saxony (cf. Fig. 7.4).

These two different types of investigation units are used due to the different objectives of the projects. The natural boundaries of *watersheds* (which can be described as 'quasi-closed systems') and their hierarchical organization form an appropriate structure for process-orientated environmental impact analysis. Administrative units should be used when the project's objective is to provide planning authorities with recommendations for land use and land management.

Generally speaking, it makes sense to consider both types of investigation units to ensure that information about the landscape balance is contributed to the planning processes in order to achieve sustainable development. The studies presented show a good way of combining these two approaches.

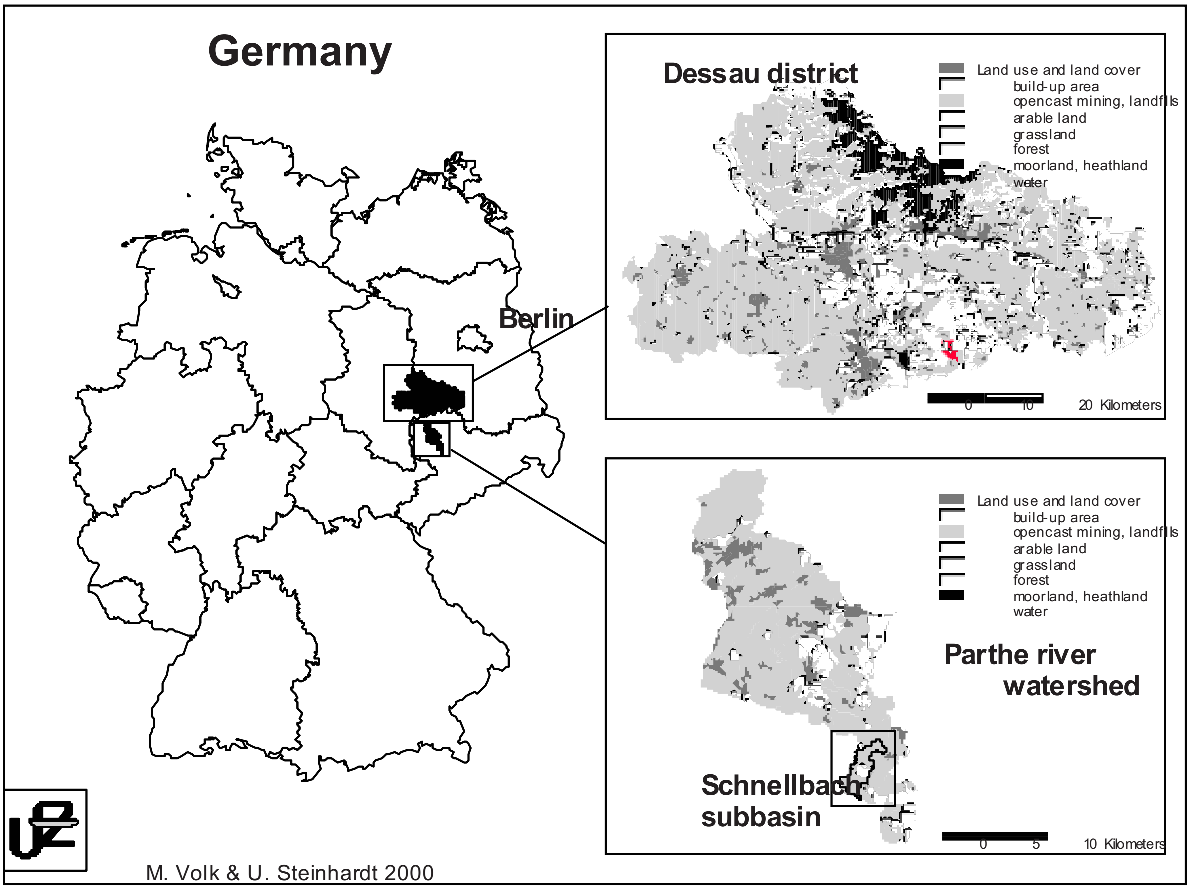

Fig.7.4: Location of the study areas in Germany.

7.6.1 The Dessau district

The district covers an area of about 4,300 km² in Saxony-Anhalt divided by the River Elbe. The study area is composed of various landscapes with very different conditions, ranging from Holocene floodplains and old moraine landscapes to very fertile loess plains. The mean annual precipitation varies between <500 mm in the western parts up to around 650 mm in the northern and southern district. The investigation area is one of the driest regions in Germany. Owing to the widespread fertile soils (chernozems) and lignite resources, agriculture, industry and other human activities have determined the main features of the region. On the other hand, there are almost undisturbed areas such as the riparian zone and the Elbe floodplain, which have been designated a biosphere reserve by UNESCO. The investigations are being carried out within the project "Landscape development, landscape balance and multifunctional land use in the Dessau district". This project is designed to derive strategies for sustainable development on the basis of investigations into the landscape balance (Petry & Krönert 1998, Volk 1999). Special attention is paid to restoring the landscape's multifunctionality by avoiding land use conflicts.

7.6.2 The Parthe river watershed

The River Parthe watershed (400 km²) is a subbasin of the Elbe watershed and located in southeast Leipzig. It can be characterized as a representative part of the northwestern Pleistocene landscape with very different properties ranging from Permian porphyry hills and old moraine landscapes to fertile sandy loess plains. The mean annual precipitation is about 570 mm. The watershed is characterized by strong impacts of land use on the landscape balance resulting from the extraction of groundwater, sand, gravel and porphyry, lignite-mining, and the expansion of built-up areas on the outskirts, especially since 1990. The studies are part of an interdisciplinary project investigating the landscape water balance and the fluxes of material, nutrients and energy within the loess areas of the Elbe watershed. The interactions of the various material and energy fluxes are to be described here using a hierarchical network of various GIS model couplings. The results are to be used to calculate land use scenarios (impact of land use changes on the landscape balance) as a basis for the conclusion of sustainable land use variants.

Both study areas are dominated by agricultural land use. At present, the contrary development of agricultural land use is becoming increasingly dynamic and leading to landscape changes; while the loess-covered parts face further intensification, with marginalization becoming a widespread new phenomenon in the sandy Pleistocene areas.

7.7 ‘Top down’: balancing - modelling - typifying

During the past few years, demands have increased for information about the landscape’s water, material and energy balance to be integrated into planning processes. Consequently, the assessment of the impact of land use changes (i.e. groundwater abstraction, afforestation, etc.) on the landscape balance can be expected. In contrast to the increased application of GIS for environmental and planning surveys, a problem still arises with the availability of the relevant data layer for large areas. Moreover, most of the environmental parameters are only gathered and measured for short periods and in small areas. To solve this problem, investigating the landscape water balance on a regional scale (1:50,000 and higher) is proposed. The results of this investigation provide a basis for the designation of potential risk zones for vertical and horizontal material and nutrient outwash. The majority of the huge amount of data required was obtained by sharing data as part of our collaboration with geological and meteorological surveys, as well as planning and environmental authorities. One advantage of this exchange of data is that it strengthens communication and cooperation between research institutions and the relevant authorities responsible for landscape planning. On the other hand, this diverse information about soil, land use and land cover, groundwater and surface water, etc., is gathered using very different methods and for different projects and aims. As a result, the data are often inadequate for interdisciplinary landscape-ecological applications owing to their spatio-temporal resolution and their quality. Solving this problem requires standards and guidelines for the objective generalization and aggregation of the data layer (Volk & Steinhardt 1998, Petry et al. 2000).

Nevertheless, the calculations allow regional assessment and comparisons between areas of higher and lower *groundwater recharge and runoff* in relation to the prevailing natural conditions and the land use types. The water balances were calculated for both areas using the runoff simulation model ABIMO. Fig. 7.7 shows the groundwater recharge values using the example of the Dessau district. In comparison with other regions in Germany, both study areas exhibit low precipitation and groundwater recharge rates, the highest values in both cases being recorded in the morainic parts of the study areas. The dry western parts of the Dessau district with prevailing chernozems and cohesive substrate (highly important for the function of groundwater protection) are characterized by very low values. Within the existing priority areas for groundwater extraction in these western parts, only the extraction wells are protected. However, these sites are not necessarily the places where groundwater recharge and potential contamination occur. It is self-evident that the calculations can be used for the better designation of priority areas for groundwater extraction.

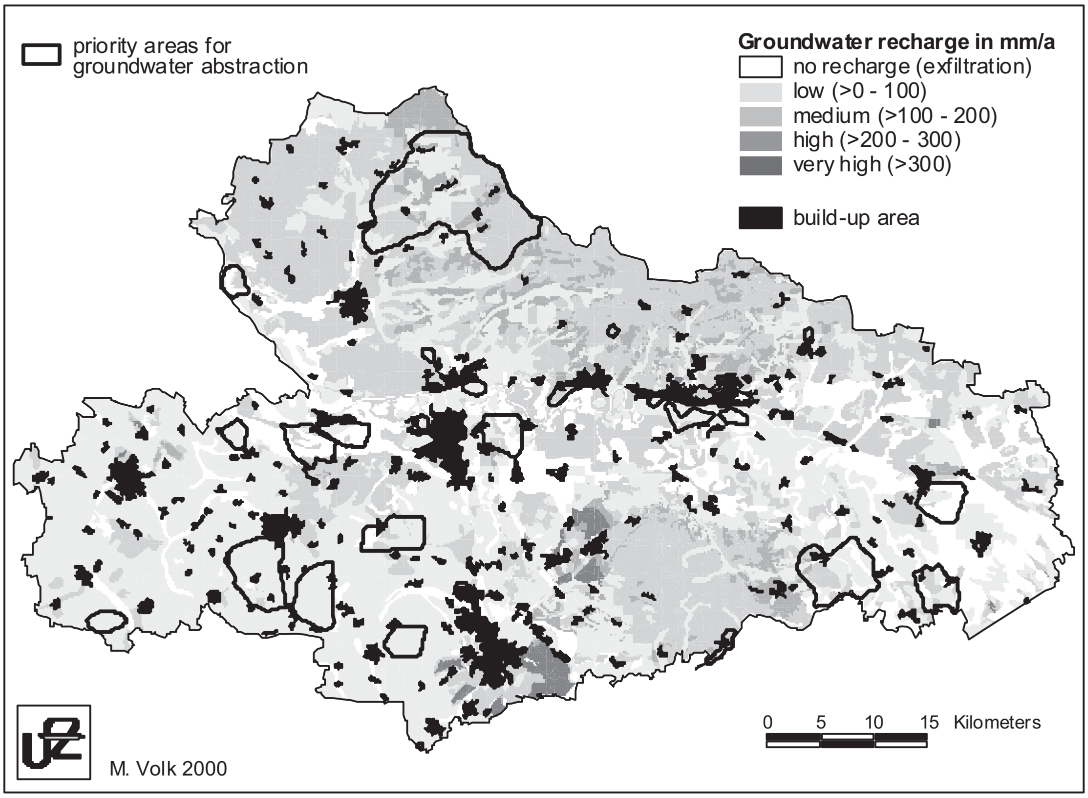

Fig. 7.5*:* Groundwater recharge in the Dessau district

Besides regional analysis, *scenarios* were calculated regarding the impact of land use changes on the water balance for smaller test areas within the Dessau district (Volk & Bannholzer 1999). As the database for the whole district has a relatively coarse spatial resolution, useful calculation of the scenarios is limited to areas >100 km². Fig. 7.6 shows an example of a test area. The conflicts in this part of the region stem from groundwater contamination by agrochemicals (e.g. nutrients and pesticides) and the overlap with priority areas for groundwater extraction and forestry. The calculations with different land use variants show for instance that afforestation in this area only slightly affects the groundwater recharge rate, but could improve the groundwater protection by a decrease in the agricultural areas. Although the potential lowering of the groundwater table caused by increased water extraction would only entail minor changes, its main ecological impact would be experienced by forestland (owing to dryness effects). These scenarios can only give rough indications of the impact of land use changes on the water balance for relatively large areas (average values). More detailed studies concerning the conditions within these areas require the application of other models and a database with a higher spatio-temporal resolution, as is shown in the following sections. Nevertheless, the results presented have been made available to the local water management and regional planning authorities to assist environmental planning decisions.

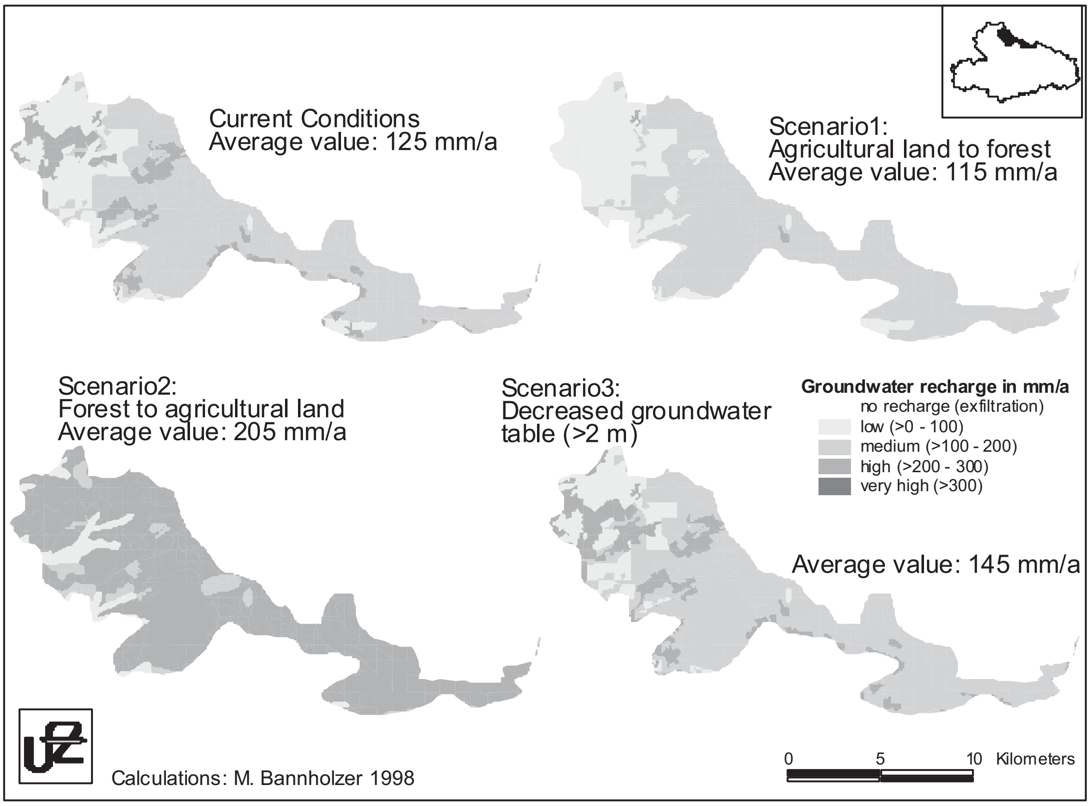

Fig. 7.6: Land use scenarios in a test area: Imapct of land use changes on the groundwater recharge.

In addition to the above studies, estimating *vertical and horizontal material and energy fluxes* on the regional scale is important in this part of our approach. These water-borne fluxes essentially depend on morphological conditions and surface cover. The percolation rates and an estimation of surface runoff and interflow were obtained by modifying the modelling results calculated with ABIMO after Röder (1998). Relevant geomorphologic parameters (e.g. slope angle and slope exposition) are coupled with the data layers of land use, soil conditions, the modelling results and climate in a GIS (ArcInfo). The soil data were classified by permeability, erosion disposition, etc. using the given assessment methods (AG Boden 1994).

In the first step, areas were identified using the query function of the GIS (ArcView), characterized by “arable land use”, “percolation rate >180mm/a” and “slope inclination 0-2°”. These areas are defined as potential risk areas (‘hot spots’) with vertical material leaching (e.g. nutrients, pesticides) from agricultural areas. According to the calculation results, the main risk areas are situated in the northern, eastern and partly the southern part of the Parthe watershed as well as in the northern and partly the southern part of the Dessau district, with permeable substrates.

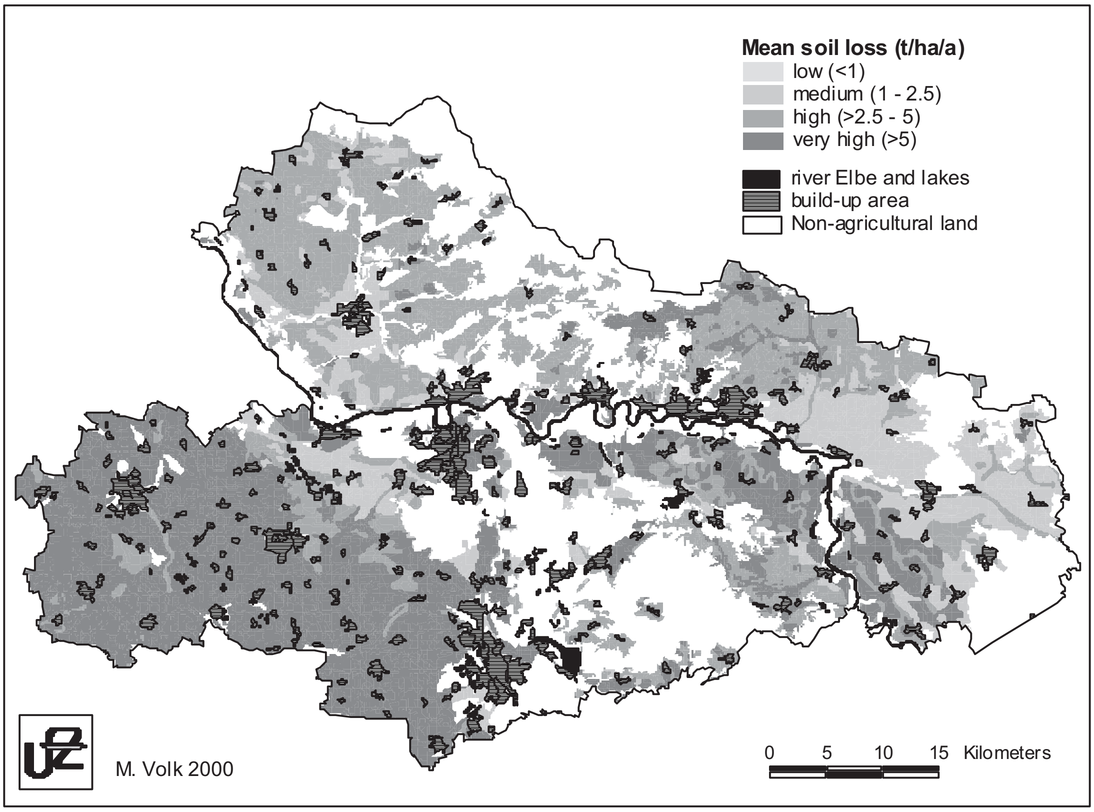

Fig. 7.7: Large scale calculation of mean soil loss of the Dessau distict.

For an initial estimate of the mean soil loss of the whole region, the modified Universal Soil Loss Equation ($R * L * S * K_B$) proposed by BGR (1994) was used. The equation factors were modified and reduced (due to the size of the study area) as follows: the R-factors (precipitation and surface runoff factor) were adapted to the conditions of the region (after Sauerborn 1994), the slope length factor L was equalized to 2.0 (slope factor S remains), and the factor K_B determines the substrate-dependent rate of the erodibility factor K (after Schwertmann et al. 1990). The results of these calculations are shown in Fig. 7.7**.**

In combination with these results, an initial indication of potential risk zones with horizontal material (and nutrient) leaching from agricultural areas is obtained by selecting areas characterized by "arable land use", "cohesive substrate" and "slope inclination >1°" (relatively high soil loss) and "medium to high surface runoff". As the Dessau region is mostly flat, only a few risk zones for horizontal material (and nutrient) flow exist. The map in Fig. 7.8 shows both potential risk zones with vertical and horizontal material (and nutrient) leaching.

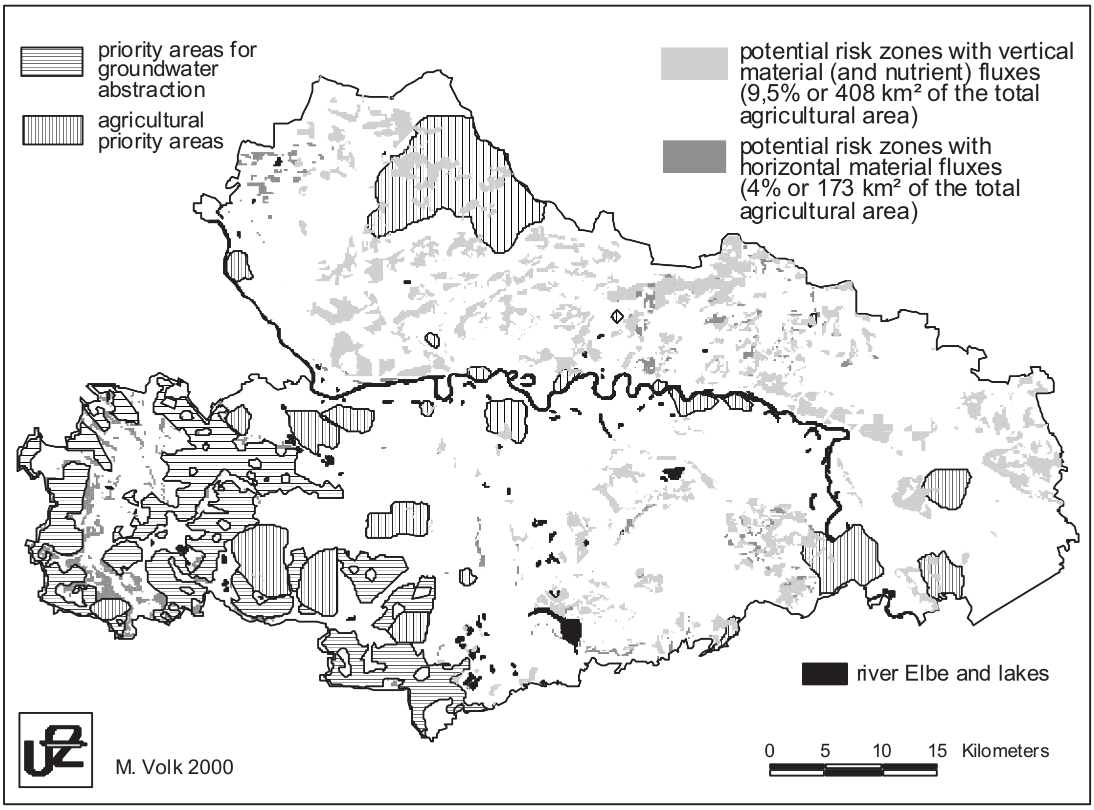

Fig. 7.8: Dessau district: Potential risk areas with vertical and lateral material (and nutrient) leaching

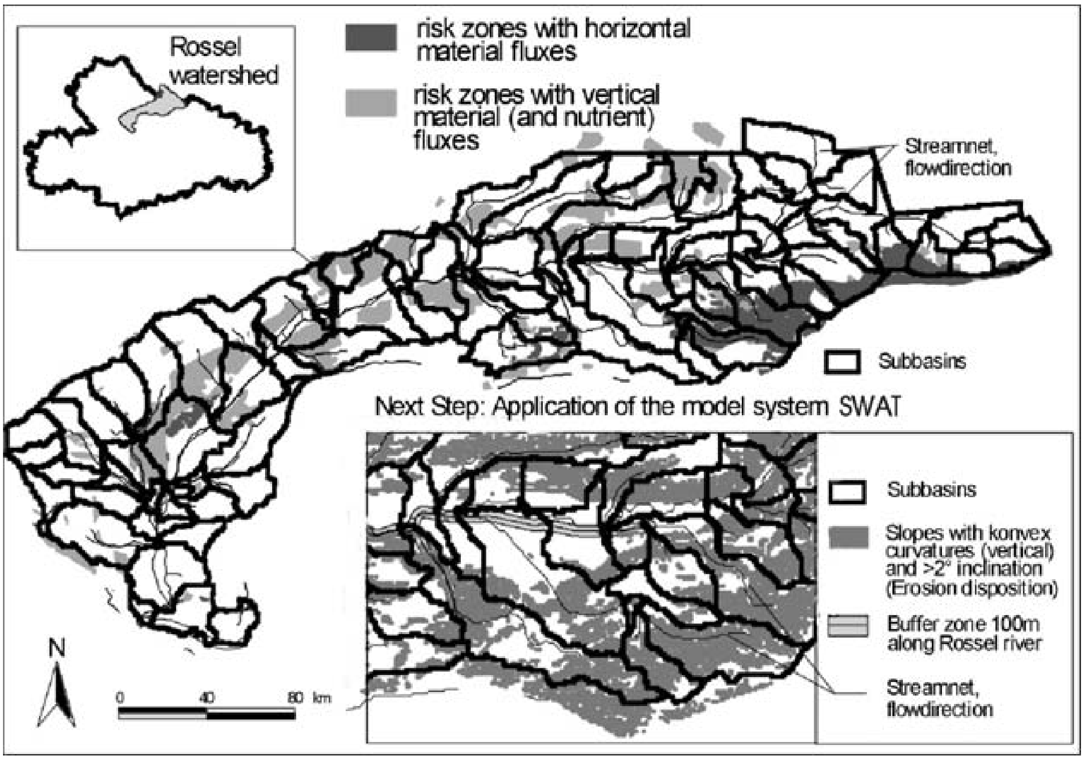

Fig. 7.9: Detailed investigations in the Rossel watershed.

The next step is to qualify and quantify the vertical and horizontal water, material and energy fluxes from the designated risk zones. These more detailed process-oriented studies require using watersheds as investigation units and a change of scale level (Steinhardt 1999b). The application of a database with a higher spatio-temporal resolution enables the derivation of hydro-morphological parameters such as flow direction, stream-order, watersheds, etc., which gives an impression of the transport conditions in the watershed. This analysis is carried out with hydrological functions in the Grid Module in ArcInfo or with the watershed module for ArcView, which is coupled with the HEC-HMS Hydrological Modelling System[6] (see also Olivera et al. 1998).

In addition, using different model systems allows the derivation of quantitative and qualitative information on the water, material, nutrient and energy fluxes in catchments and subbasins. As well as renewed calculation of the groundwater recharge and the surface runoff, this enables the improved differentiation of the risk zones. Moreover, useful information can be derived about potential risk zones in streams and rivers. Concluding the topographic factor LS in ArcInfo (Grid Module, Hickey et al. 1994) and determining the factors K_B (substrate-dependent rate of the erodibility factor K) and R (precipitation and surface runoff factor, modified after Sauerborn 1994) and combining the two enables the modified usage of the Universal Soil Loss Equation after BGR (1994). The results of these more detailed calculations are shown in Fig. 7.9 using the examples of the Rossel watershed in the north of the Dessau district.

[6] A hydromorphological module is also integrated into the ArcView SWAT model.

7.8 ‘Bottom-up’: measuring - mapping - modeling

To verify the mesoscale results and to improve knowledge about the process behaviour, we now attempt to combine the ‘top-down’ and the ‘bottom-up’ approaches. This will be done using the example of the Parthe watershed. For a detailed investigation of the landscape balance, four representative test sites were initially selected (Figs. 7.10, 7.11):

- Glasten - source area, slightly anthropogenically influenced
- Naunhof - middle course, impact through groundwater extraction
- Thekla - final gauge, partly urban influenced
- Schnellbach - subbasin, intensive agricultural impact

In November 1998, the installation of a network of measuring and survey stations to investigate the surface water was commenced (Figs. 7.10, 7.11).

Fig. 7.10: Gauge Glasten, V-weir (right);
Gauge Thekla, water sampler, rain-gauge (left)

Table 7.2: Selected precipitation values (daily totals) of the Parthe area, June 1999

Station	*Precipitation values (mm)*			
	2 July	*18 July*	*19 July*	*20 July*
Glasten	8.4	0.4	27.5	5.9
Naunhof	9.5	0.3	18.6	6.5
Thekla	5.0	0.3	2.6	1.7
Schnellbach	7.1	13.8	18.1	0.0

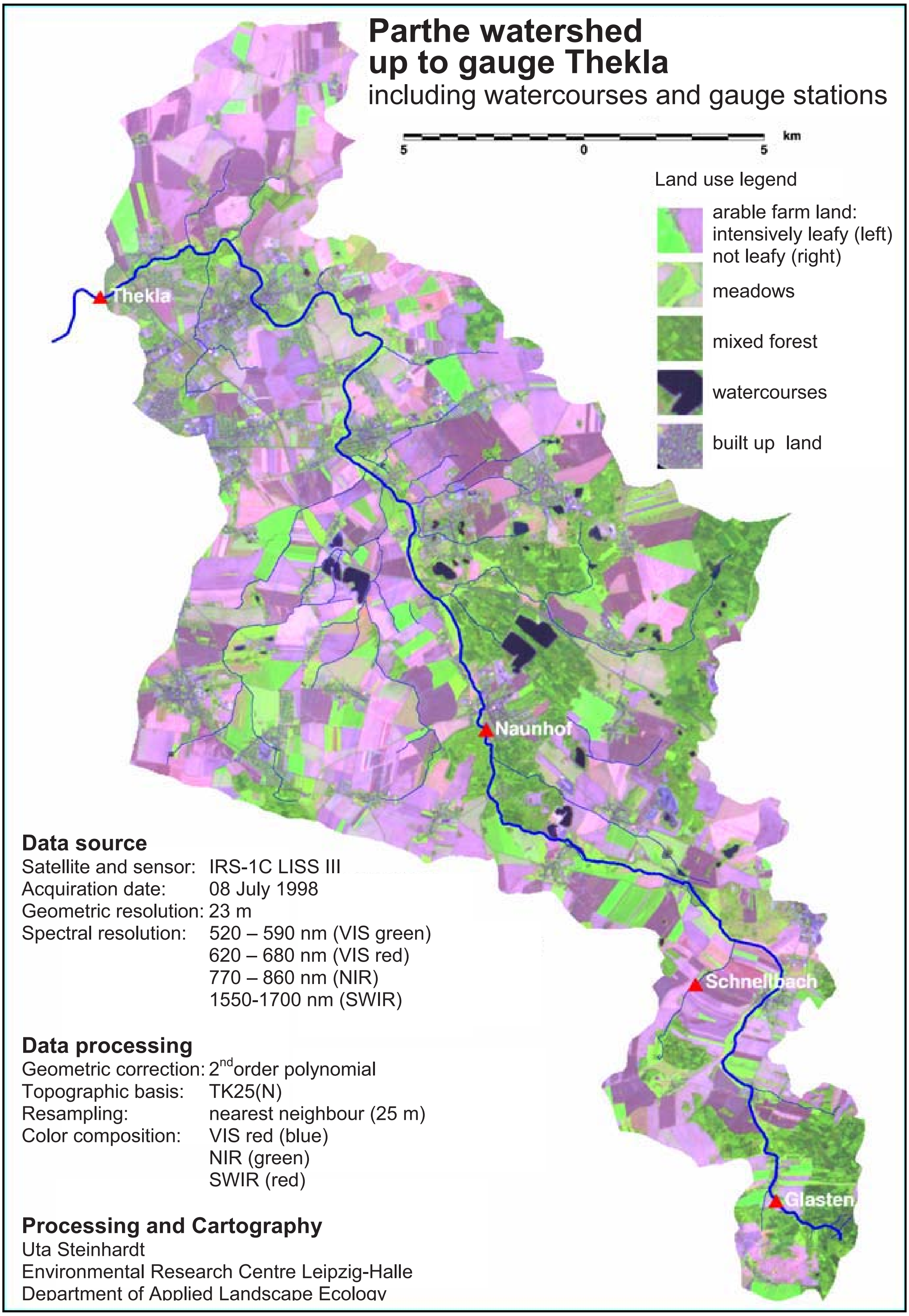

Fig. 7.11: Parthe watershed with its drainage network and location of the gauging stations.

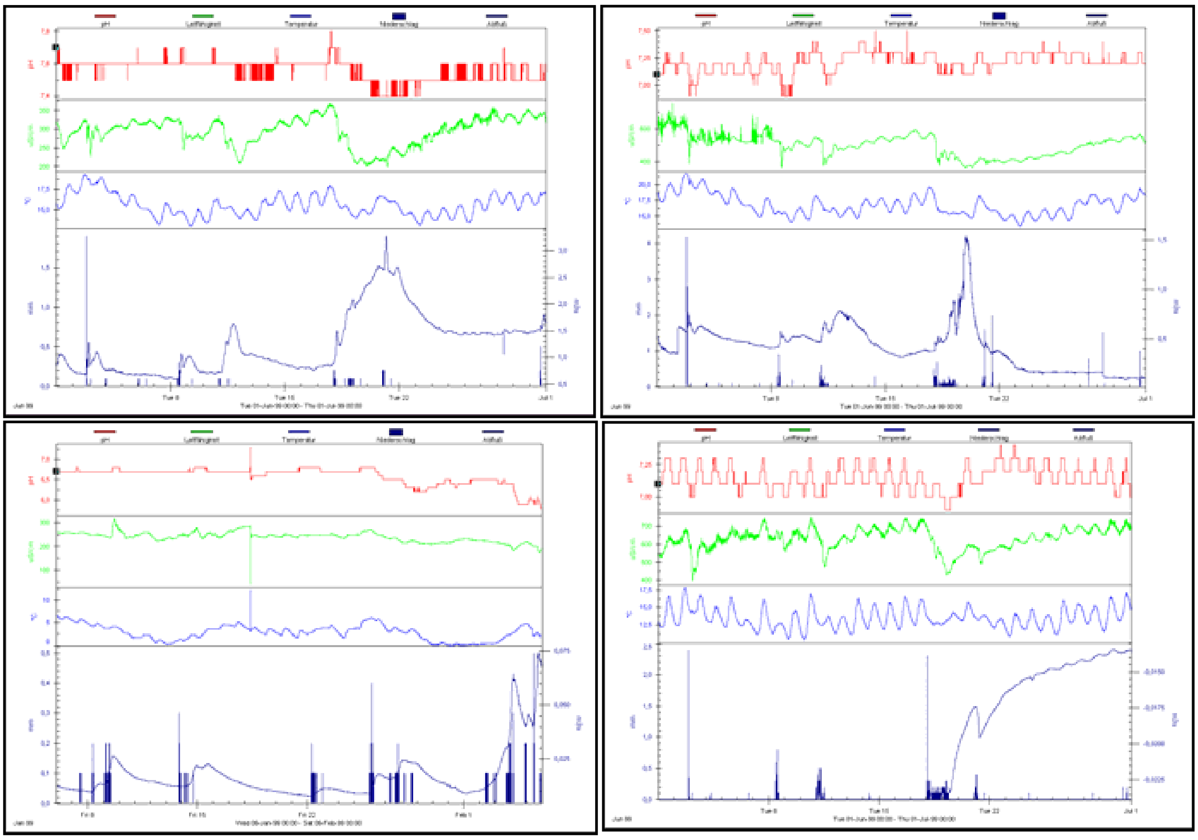

Fig. 7.12: Surface water parameters (pH, conductivity, temperature, precipitation and discharge) of the four gauges along the River Parthe for June 1999: Thekla (upper left), Naunhof (upper right), Glasten (lower left), Schnellbach (lower right).

The parameters discharge, pH, water temperature and conductivity and precipitation are measured at each station with a five-minute resolution. Additionally, the water is automatically sampled. Daily sampling (mixed samples) allows the derivation of information about the base load. The samples are analysed for their content of nitrogen and phosphorus components in the laboratory. At the same time, event-based sampling takes place during automatic sampling if a set flow rate value is exceeded. This enables the acquisition of the material components during the drain peaks. Fig. 7.12 shows the initial results using parameters recorded in June 1999.

The results show the occurrence of extreme precipitation events (short-term heavy rain, long-term light precipitation) throughout the whole watershed simultaneously, as well as those only locally surveyed or with a delay following events (Table 7.2).

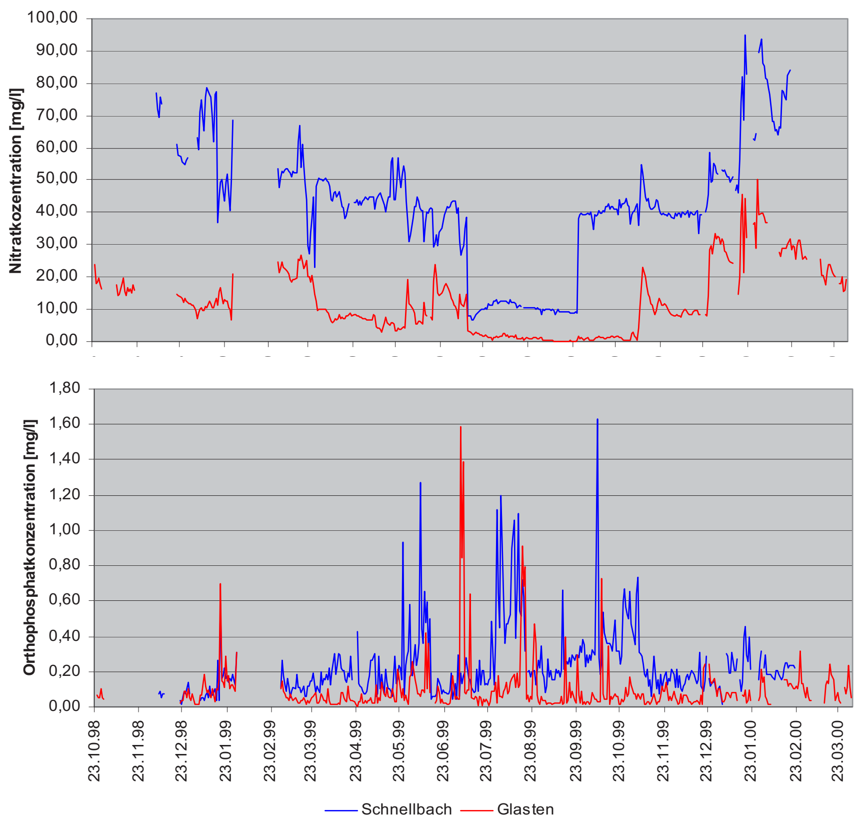

Fig. 7.13: Concentration of orthophosphate (bottom) and nitrate (top) in surface water.

The response of the receiving water to such precipitation events obviously depends more on the size of the watershed than on its natural conditions and land use and land cover situation. Although the size of the subbasin at the Glasten gauge is comparable to the size of the Schnellbach subbasin, both areas differ in their land use structure: nearly the whole subbasin at the Glasten gauge is used by forestry, whereas the Schnellbach subbasin is dominated by intensive agricultural use (Fig. 7.11). No retention influence by forest on drainage can be observed. Immediately after the precipitation event, the receiving water responds with a sharp rise in discharge, but also descends just as fast. At the gauges located downstream, such as Naunhof and to a much higher extent Thekla, the drainage curve rises with a delay after the precipitation event respectively with a stay away of the local precipitation. A decay can also be observed at the descent of the drainage curve.

The increased flow rate and its related dilution effect is naturally accompanied by a decrease in conductivity. The initial results of the investigations into the quality of the surface water indicate seasonal and temporal differences. All the samples were analysed for their orthophoshate and nitrate levels (Fig. 7.13).

These field measurements are planned to be the beginning of a long-term environmental monitoring. The results of the measurements help our understanding of the spatio-temporal distribution and organization of water-borne fluxes of material to be detected and improved, and also serve as input data for model applications. As the mesoscale processing of cultivation-related nutrient input into rivers by surface runoff entails the mathematical description of the fluxes of water, material and energy, the usage of simulation systems is essential.

The qualification and quantification of the vertical and horizontal fluxes of water, material and energy fluxes from the designated risk zones requires a change in the scale level, as mentioned in the previous section. A database with a higher spatio-temporal resolution allows detailed hydromorphological analysis (calculation of subbasins, flow direction, stream net and stream order as potential material transport courses, etc.) and is used as input data for various models. Due to the scale-specific applicability of the models tested, data with different spatio-temporal resolutions are needed. With the common consideration of both ‘top-down’ and ‘bottom-up’ approaches, the two methods converge at the ‘subbasin scale’, which is dealt with below.

As shown above, the application of GIS-coupled model applications also enables the identification of potential risk zones with horizontal material (and nutrients) outwash. However, these ‘rough’ identifications do not provide any indication of the potential input of the material concerned into the receiving water, which is related to two problems.

GIS-coupled model applications for describing the precipitation/runoff/drainage processes assume a ‘depressionless’ (‘filled’) DEM. The relevant GIS routines are only applicable if all drainless depressions of the DEM are filled (this is also true for the natural depressions!) and thus linked to the receiving water. This situation does not correspond to reality. The investigations by Fritsch (1998), for instance, show that in a study area in northeastern Germany, only 10% of the watershed area is linked to the receiving water. The situation differs due to the different natural conditions of landscapes, of course, but even in our own study areas the 100% drainage of the watershed is not guaranteed. Even with a hypothetical assumption of such a situation, the entire surface runoff does not reach the receiving water. This must also be considered in relation to the structure of the near-stream land. Near-stream land can exert a retention influence on the receiving water which is related to the dissolved substances in the water (nitrogen components), as well as the material fixed to the particles transported (i.e. phosphorus). This is the subject of several detailed investigations (e.g. Haycock & Burt 1993).

In our investigations, our goal was to produce a simple method to assess the effectiveness of the near-stream land along the 50 km Parthe riverbanks as a potential nutrient retention zone. This assessment necessitates measuring the relevant qualitative and quantitative parameters of the near-stream land. Therefore, the theoretical fundamentals of the nitrogen and phosphorus cycles had to be studied intensively. This enables the structures and processes to be identified which are relevant for the outwash of these materials out of the landscape and input into the water. This in turn formed the basis for the following development of a special mapping method. In doing so, various suggestions for the recording and assessment of the capability of the near-stream land (LUNRW 1993, Raderschall 1995, SMU 1995, LAWA 1998, DVWK 1997) were taken into account and combined with the above-mentioned aspects and adapted to the conditions of the study area (Fig. 7.14, Rau 1999).

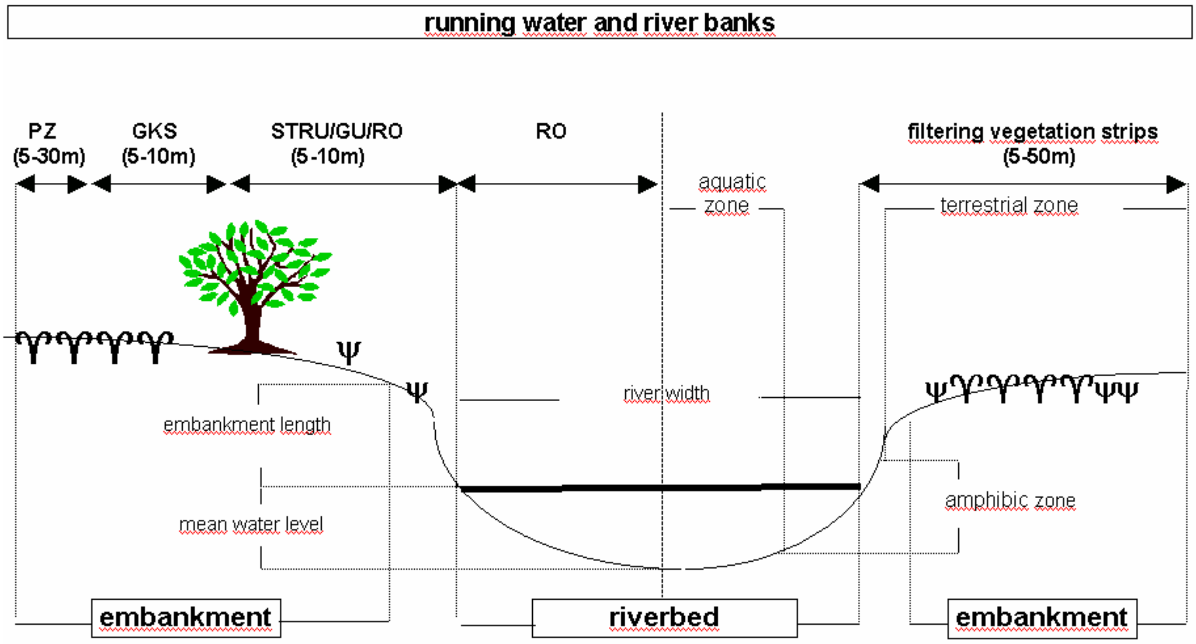

Fig. 7.14: Profile of a stream and structured retention zones (after Rau 1999)

As basic assessment units for recording and assessing, 250m-sections of the river are used. The size of this assessment unit is adapted to the length of the receiving water and suitable for the derivation of information for a watershed of this size. The method was developed using digital mapping and the application of a pen computer and GISPAD software (CON TERRA 1998). The assessment of the retention capability based on the mapping results was carried out using a three-step chart. In view of the amount of work involved, the time required and the related cost, the investigations showed that using the described mapping for a river with a length of 50 km is hardly useful - especially for application by environmental authorities. Nevertheless, the resulting assessments were coupled with other data layers in the GIS. The combination of defined buffer zones with low potential nutrient retention with areas showing high erosion risk, for instance, allows the identification of areas with a high risk of non-point source pollution.

All these investigations can be considered as an important preliminary stage and an addition to the following scale-specific application of models for the mathematical description of the transport processes and the measurement of nutrient input and output. In the Schnellbach subbasin of the River Parthe watershed, for instance, the small-scale model system E3D and the runoff simulation model WaSim-ETH were tested for their scale-specific applicability (Fig. 7.15), as well as to verify the large-scale calculations with ABIMO and later on SWAT. The simulations are based on data with a higher spatial, temporal and thematic resolution: For example, short-term individual rain events are considered as well as field-specific management (e.g. tillage, irrigation, drainage and fertilization). The combination of the investigation methods from different scales allows their scale-specific applicability to be checked. Thus, it should be possible to derive rules for transmission from one scale level to another.

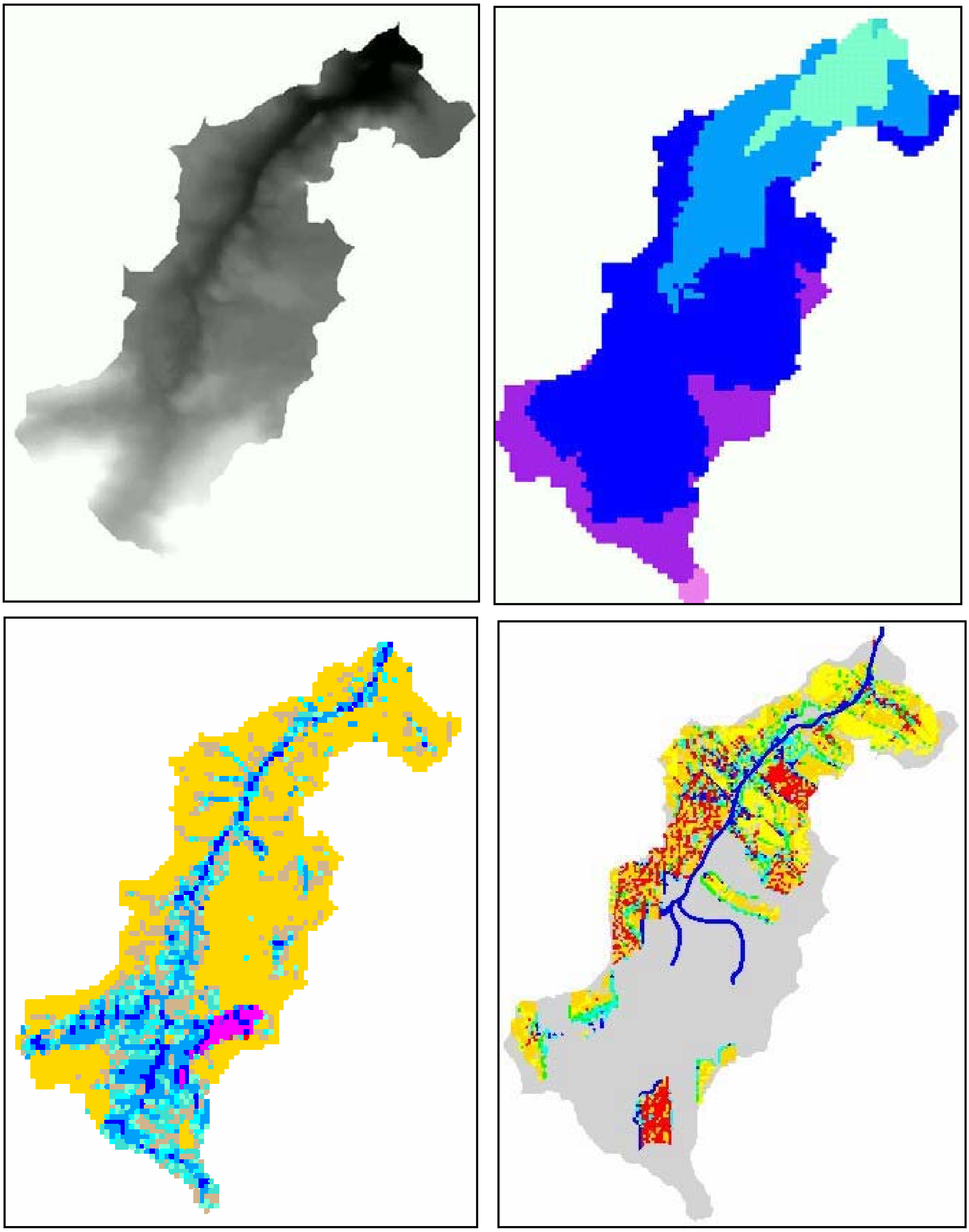

Fig. 7.15: Modeling event based soil loss with E3D and water balance (calculated with the runoff simulation model WaSim-ETH) within the Schnellbach subbasin (DTM (upper left), drainage time (upper right), discharge (lower left), soil erosion/accumulation (upper right))

7.9 Discussion and conclusions

The hierarchical approaches for the investigation and description of landscapes presented here ought to enable the optimization and regulation of process systems at all scale levels. Besides the reduction of material and nutrient outwash and the regulation of water flows, this also includes consideration of the internal interactions, as well as consideration of interactions with adjacent landscape ecosystems. The investigation of the landscape balance and the calculation of different land use scenarios allows an estimation of the impact of land use changes on the landscape's water, material and nutrient fluxes at the different scale levels (Table 7.3).

Table 7.3: A scale-specific approach for the landscape balance investigation and evaluation. The table includes also suggestions for the integration of the results in landscape planning processes.

Scale level	Spatial planning level	Assessment Units	Data base (different spatial resolutions!)	Model applications	Derived informations and application possibilities
1:50.000 and larger (areas >10³ km²)	Regional level (First identifications, coarse classifications)	• Land scape • Units • large river catchments • areas with similar conditions	• Soil • Morphology • (DEM100-250) • Climate • Water • Watersheds • Land Use • Landscape Units • Spatial Planning Targets	• ABIMO Decription of the fundamen-Tal elements of the water cycle (e.g. runoff)	• Water balance and land use scenarios for areas >100 km² • (Coarse) Identifications of potential risk zones with (water) and material fluxes (combination of modeling results with assessments guidelines) • Analysis of land use conflicts on the regional scale and recommendations for land use (environmental and resources management and conservation)

Scale level	Spatial planning level	Assessment Units	Data base (different spatial resolutions!)	Model applications	Derived informations and application possibilities
1:25.000 to 1:50.000 (areas 10 to 10³ km²)	**Regional level** District level (Quantitative and qualitative informations and assessments)	• Watersheds • subbasins • conservation areas • indicated danger zones (cp. above)	• Soil • Morphology • (DEM40-100) • Climate • Water • Watersheds • Subbsins • Land Use • Spatial Planning Targets	• SWAT • ABIMO • WaSIM-ETH • AGNPS • (CANDY) Investigation of vertical and horizontal matter and energy fluxes (Nitrogen, phosphorus and pesticides transport, Erosion)	• Identification of subbasins, streamnet and -order (flow of matter and nutrients) • Indication of rivers and streams affected by matter and nutrient input • Modeling of water balance and material and energy fluxes (qualitative and quantitative informations) within the risk zones • Combination of modeling results with assessment methods: Recommendations for land use variants for decreased material/ nutrient output related to agricultural areas.
1:10.000 to 1:25.000 (areas 100m² to 10 km²)	District level Community level (Detailed quantitative and qualitative assessments)	• Fields • biotopes • river sections	(incl. mapping/ measuring) • Soil • Morphology • (DEM<40) • Climate • Water • Subbsins • Land Use • Spatial Planning Targets	• **Physical and empirical Models** (WEPP, AGNPS, CANDY)	• Material and nutrient output (outwash) related to fields • Polyfunctional landscape assessment and land use optimization

On the basis of the results, recommendations can be concluded for land use variants with positive effects for environmental and natural resource protection.

Considering the reduction of material and nutrient outwash out of agricultural areas (e.g. in conservation areas), this could be an important argument within the planning processes of the relevant agencies. Besides the scale-specific optimization of the model applications and landscape assessment methods, we are working on a hierarchical parameter indicator system, which allows both the assessment of other landscape functions and the integration of the model calculations. The first results of these investigations for the upper mesoscale are presented in Chap. 9 of this book. The augmentation and combination of 'classical' methods such as measuring, mapping and assessment with innovative GIS-model applications ought to solve the problem of verifying mesoscale and macroscale model calculations of the landscape balance. Various methods are being developed and tested by the authors' working group. The next important step is to integrate socioeconomic components into the approach. Examples of such integrated ecological and socio-economical assessments are contained in Chap. 10 as well as in Horsch, Ring & Herzog (2001) and Dabbert et al. (1999).

7.10 Outlook

Considering the 'state of the art' of the field of landscape ecology dealt with here (including our own approach), future research should be focused on the following topics:

▪ *Topic 1:*
- Improving our understanding of landscape-ecological processes: interactions between landscape structures and processes.
- *Possible solution:*
  ñ Further development of models and scale-specific assessment methods (optimization and verification)
  ñ Development and application of 'context-related' remote sensing methods

▪ *Topic 2:*
- Less availability of the required database for large areas
- *Possible solution:*
  ñ Development of transfer functions
  ñ Further development of 'hydrological remote sensing'.

▪ *Topic 3:*
- A lack of knowledge concerning the 'natural' dynamics and adaptability of ecosystems (current 'ecologiocal' assessments are mostly structure-oriented, especially at larger scales). As a result, insufficient attention is paid tothese factors.
- *Possible solution:*
  ñ Enforce of basic science in the field of the behaviour and adaptability of ecosystems to human impacts (with special attention to land use)

In addition, future landscape-ecological research should be directed towards the greater consideration of the 'driving forces' of land use development (supraregional, national and international tendencies). Subsequently, the calculation of scenarios of these probable land use changes will allow better forecasts of their impact on the landscape balance. As socioeconomic parameters are integrated, multicriteria assessment can be carried out to derive land use and water management strategies geared towards sustainable development.

Future models and methods developed in landscape ecological research must be clear and understandable in their structure, procedure and applicability. This entails further investigations that contribute to a better understanding of the interactions between landscape-ecological processes and structures, and the hierarchical concepts of landscape ecosystems (i.e. Klijn 1995). This will lead to these models and methods being increasingly accepted by environmental and planning authorities as important instruments for the simulation of human impacts on the landscape balance and as decision instruments for relevant planning problems. This is all the more important considering the rapid development of computer-based models and methods in landscape ecology.

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An investigation of water and matter balance on the meso-landscape scale: A hierarchical approach for landscape research

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Abstract

The realization of strategies for sustainable land use assumes specific research concepts from the local to the global scale (micro-, meso- and macroscale). Therefore, landscape ecological science has to provide investigation methods for all these different scales. By combining "top-down" and "bottom-up" approaches in addition to coupled GIS-model applications and traditional methods, the investigation of landscape ecological structures and processes seems to be possible. The presented studies show this approach on examples of two study areas in Eastern Germany: A watershed of 400 km^2 and an administrative district of about 4000 km^2. The scale-specific applicability of several models and methods were tested for these investigations, and the validation of the calculated results are presented. An important outcome of the project should be the prevention of conflicts between agriculture, water management and soil, and water and nature conservation; based on recommendations for land use variants with decreased pollutant loading within agricultural areas. The scale specific investigations can be considered as a base for establishing sustainable land use.

Introduction

The human factor "land use" affects the interactions between water, soil, geomorphology, vegetation, etc. on several spatial scales in different manners and intensities. Due to the long tradition in scale-specific research in landscape ecology, several methods are available for the investigation of landscape ecological structures and processes on the microscale (measuring and/or mapping of parameters), as well as for the macroscale (General Circulation Models, GCMs). Only a few studies deal with the optimization of the scale-specific applicability of these methods and models, especially for larger scales (Krysanova et al. 1998).

But most of the environmental changes and conflicts between land use, conservation, and resource management become obvious at the landscape scale (Riitters and Wickham 1997). Special attention should be paid to the investigation of the landscape water balance – at least in temperate and humid zones, because water-carried fluxes of matter and energy play an important role for the landscape balance. All components of the landscape structure are interrelated by these fluxes. Therefore, special indicators and model applications have to be developed for landscape-scale investigations, that allow assessments and forecasts about the impacts of land use changes on the landscape balance. This aim should be reachable by the combining of both "top-down" and "bottom-up" approaches in addition to coupled GIS-model applications and traditional classification and assessment methods. This would be also useful for administrative and environmental agencies, because the integration of landscape ecological knowledge into landscape planning processes is absolutely essential.

The presented approach is a scale-specific, hierarchical method for the investigation of parameters of the landscape balance on test areas in Eastern Germany – at first focused at a "top-down" method. The

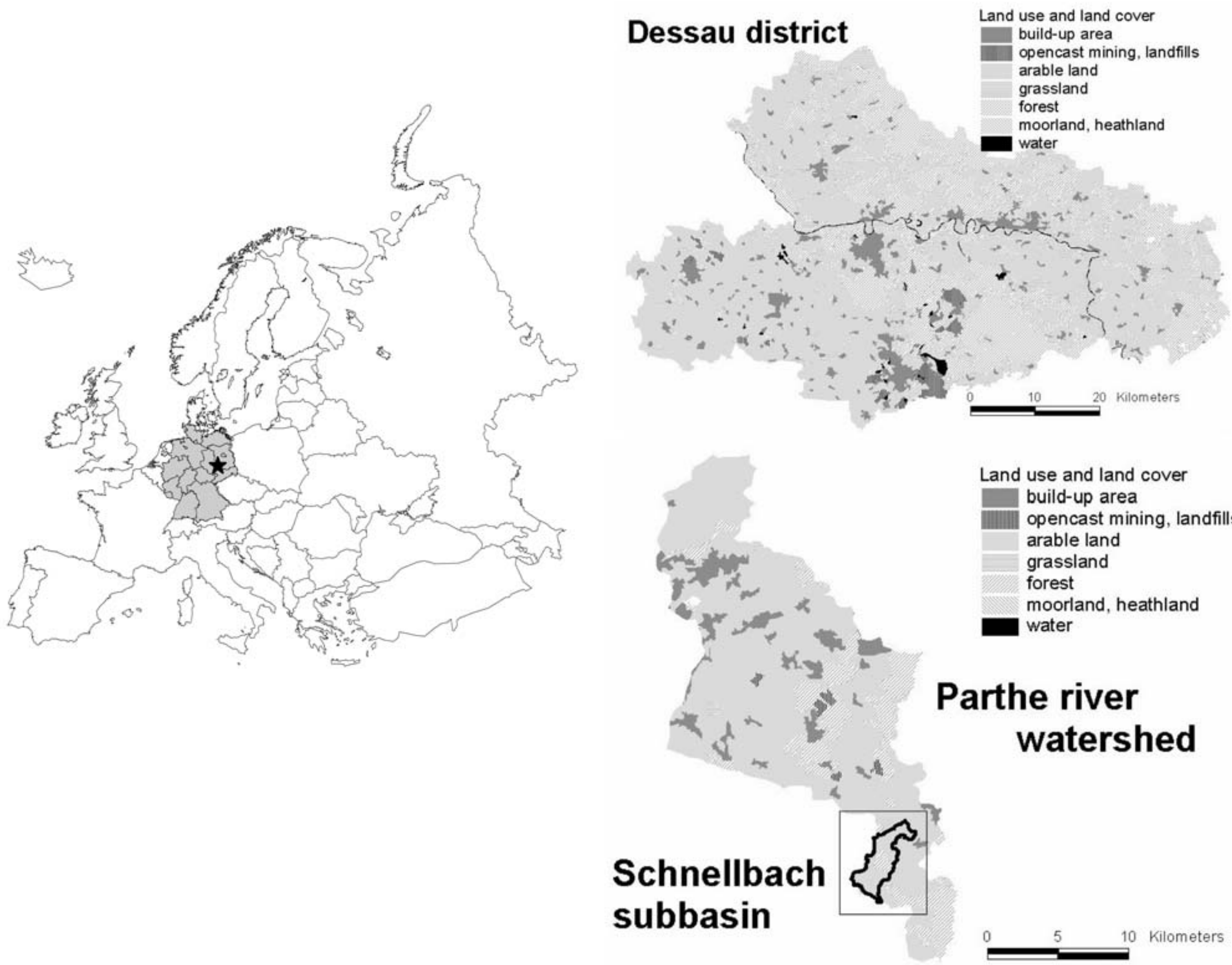

Figure 1. Location of the study areas: The left map shows the position of Germany in Europe. Both study areas (Dessau district and Parthe watershed with Schnellbach subbasin) are located in the central part of Eastern Germany – labeled by the star.

study is focused at the upper (scale 1:50 000 and larger) and middle mesoscale (scale 1:25 000 to 1:50 000). The main hypothesis of the project is that, depending on the specific scale level, application of specifically adapted methods is necessary.

Therefore, the main objectives of the project are: derivation of basic factors and parameters of landscape ecological processes and structures, development of investigation methods (processing, assessment, GIS-coupled modeling), and investigation of their scale-specific applicability.

Study areas

The presented approach is tested in two study areas in the states of Saxony-Anhalt and Saxony (Germany), with different ecological and economic conditions. Thus, the applicability and the transferability of the approach has to be examined. Within the paper, the approach will be shown on examples of the Dessau district (administrative unit) in the east of Saxony-Anhalt and on the Parthe river watershed in Saxony (Figure 1).

These two different types of investigation units are used due to the different objectives of the projects. The natural boundaries of watersheds (quasi-closed systems) and their hierarchical organization form an appropriate structure for process-oriented environmental impact analysis at the landscape scale. Administrative units should be used when the project's objective is to give recommendations for land use and land management to planning authorities. The presented studies show a possibility to combine both approaches in a useful way. Due to the involvement of different projects it was not possible to apply this approach to the same territory: The Parthe watershed is not located in the Dessau district.

The Dessau district

The Dessau district covers an area of about 4300 km^2 in Saxony-Anhalt, divided by the river Elbe. The district is composed of various landscapes with very different conditions, reaching from Holocene floodplains and old moraine landscapes to very fertile loess plains. The mean annual precipitation value, vary from < 500 mm in the western parts up to around 650 mm in the northern and southern district. It belongs to the driest region in Germany. Because of the widespread fertile soils (chernozems), and lignite resources, agriculture, industries and other human activities have determined the main features of the region. On the other hand, there are nearly undisturbed areas like the riparian zone and the floodplain of the river Elbe, which are designated as a biosphere reserve by the UNESO.

The Parthe *river watershed*

The Parthe river watershed (400 km^2) southeast of Leipzig is a representative part of the northwestern Saxonian Pleistocene landscape with rather different properties, reaching from Permian porphyry hills and old moraine landscapes to fertile sandy loess plains. A mean annual air temperature of 8.5 °C and a mean annual precipitation of 570 mm are typical for this site. The area is characterized by strong impacts of land use on the landscape balance, resulting from the extraction of groundwater, mining of lignite, gravel, sand and porphyry, and the expansion of built-up areas at the outskirts of the settlements, especially since 1990.

Both test sites are dominated by agricultural land use. Currently, the contrary development of agricultural land use is becoming increasingly dynamic and leads to landscape changes; while the loess covered parts are confronted with further intensification, marginalization is a widespread new phenomenon in the Pleistocene areas (Petry and Krönert 1998). Beside the impacts of these land use changes, e.g., on soil and water, this is leading to conflicts between agriculture, nature conservation and ground water extracting for drinking water supply.

Landscape hierarchies

A scale-specific approach for the investigation of the landscape balance

Several papers from different scientific disciplines deal with the hierarchical organisation of nature (O'Neill et al. 1986). The hierarchical concept was introduced into German landscape ecology by Neef (1967) and continued by several other landscape ecologists (Leser 1997). An overview about hierarchical concepts in landscape ecology is given by Klijn (1995). These concepts are mainly focused to the hypothesis that each of the scale levels (micro-, meso and macroscale) is characterized by specific temporal and spatial ranges. As a consequence, each scale level needs data layers with suitable spatio-temporal resolution and specific investigation methods and provides specific knowledge (Steinhardt and Volk 2000). Main topic is here the definition of a linkage of the different landscape scales.

The main hypothesis of our hierarchical approach is that the basic components for vertical and horizontal material and energy fluxes – morphology, soil, hydrology, land use and land management, and climate – are similar over all scale levels. It is only the importance of the factors of these components which changes for each scale (Helming and Frielinghaus 1999). To give an example for morphology and erosion: On the local scale, surface roughness is one of the main factors that will affect erosion disposition, whereas for larger scales (up to river catchments) the factors slope inclination, slope length, slope exposition up to the shapes of streamlet, -net, -order and direction of flow are responsible for erosion processes. In the following sections, our approach will be presented with some examples.

It has to be mentioned here, that there is a completely opposed understanding of "small scale" and "large scale" in German and English or American literature: German landscape ecologists and geographers use the term "scale" in terms of cartographers: So 1:100 000 is a smaller scale than 1:10 000. Hence *small scale* connotes to a *large area* and vice versa. English and American ecologists use the scale terms contrarily: A small scale is coupled to a small area; a large scale to a large area. For a consistent understanding we will adopt the English and American scientific community's standard.

Table 1. Hierarchies in landscape and spatial planning in Germany (after Kiemstedt et al. (1997))

Scale level	Planning level	Spatial planning	Landscape planning
Macroscale	Country	Spatial development policy	–
Mesoscale	Federal state	*Raumordnungsprogramm*	*Landschaftsprogramm*
	Region	*Regionalplan*	*Landschaftsrahmenplan*
Microscale	Municipality (town, village)	*Flächennutzungsplan*	*Landschaftsplan*
	Parts of municipalities	*Bebauungsplan*	*Grünordnungsplan*

Hierarchies in landscape planning

According to the hierarchical organization of nature spatial planning is also organized hierarchically (Table 1). In Germany, regional management plans are instruments within the spatial planning system used to set guidelines for landscape development on the regional scale. Designated priority areas for "landscape functions" designated at the community level. But there are only few application of landscape ecological information – especially about the regulation capacity – included, and the realization of sustainable development requires the recognition of all landscape functions. Therefore, we are deriving useful indicators and parameters for analyzing and optimizing the regulation and production functions of the existing landscape types for the concerned scales (Petry and Krönert 1998; Volk 1999).

Data processing

Database and Geographical Information Systems (GIS)

Integrated landscape ecological analysis requires much different information about soil, morphology, both surface and ground water, land use and land cover, climate, as well as about socioeconomic conditions and governmental spatial planning targets. For the management of this information and the analysis of landscape ecological processes, the Geographical Information Systems Arc/Info and ArcView are used. The largest part of the database was gained by data exchange within our cooperation with geological and meteorological surveys, and planning and environmental management authorities. For the upper mesoscale, the land use data were derived from the CORINE project Land Cover (Statistisches Bundesamt 1996). Additional information about the land use and land cover structure is derived from satellite images with remote sensing tools (Erdas Imagine). Morphological data are derived from Digital Elevation Models (DEM) with spatial resolutions of 250 m (upper mesoscale) and 40 m/25 m (scale 1:25 000 to 1:50 000). For the scale 1:25 000 to 1:50 000, the climate data are received from weather stations. The soil data for this scale had to be derived from soil maps. Additional information is derived from our own hydrochemical analysis.

Modeling concept

Most of the nutrient load of surface waters originates from nonpoint sources. For the analysis of these processes (spatial variability of landscape characteristics and their influence on the transport of water and nutrients within a given area), the application of distributed parameter models seems to be an useful method. At present, a lot of the physically based approaches with a high spatio-temporal resolution cannot be effectively applied to medium-sized watersheds (Grayson et al. 1992). Thus, several models are tested for their scale-specific applicability due to the time-schedule and topics of our projects.

The short- and long-term impact of different types of land use on the quality and quantity of surface and ground water has to be quantified to formulate environmental objectives. In order to get a first description of the basic elements of the water balance at the landscape scale (long-term values of runoff and evapotranspiration), we use the runoff simulation model ABIMO (Glugla and Fürtig 1997). "Mean runoff" is defined here as the difference between long-term mean annual precipitation and real evaporation. This difference is equivalent to the total runoff. In the case of a solely vertical seeping of water this value corresponds with ground water recharge. Due to the fact that this situation is very rare in reality, this value must be understood as the sum only indifferent of both surface and subsurface runoff. Therefore, the results were modified with a runoff quotient (based on

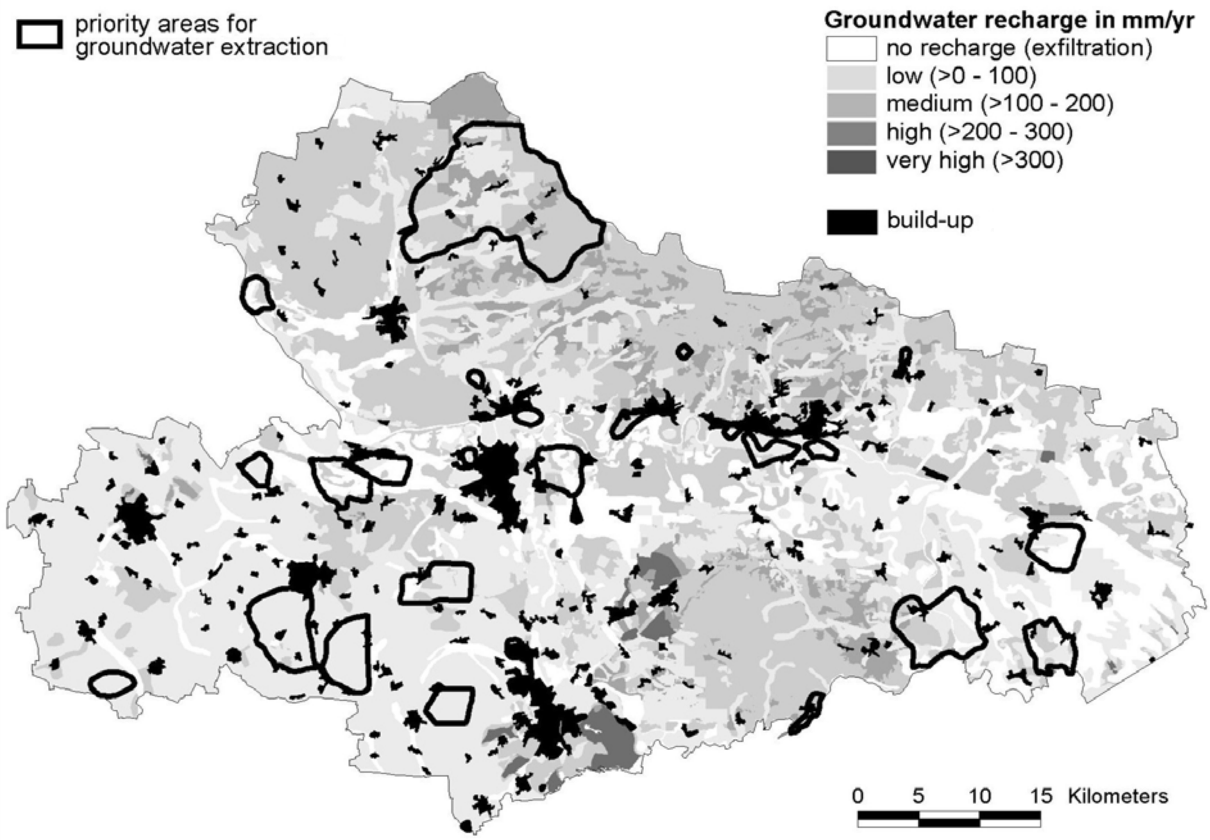

Figure 2. Dessau district: Groundwater recharge calculated with the runoff-simulation model ABIMO. The framed areas indicate priority areas for groundwater extraction designated by the regional planning authority. These calculations contribute to the improvement of the governmental designation of priority areas and the protection of water and soil.

slope angle and soil moisture) determined by Röder (1998), which allows an estimation of the surface runoff and interflow. For rough identification of the mean soil loss of large areas, different modified versions of the Universal Soil Loss Equation (Wishmeier and Smith 1978) of the Federal Agency of Geology and Natural Ressources (Bundesanstalt für Geowissenschaften und Rohstoffe 1994) were tested.

To provide a detailed mathematical description of the transport process and the coupled nutrient transport a physically based model has to be used. Thus, several models (i.e., AGNPS, ASGI etc.) have been tested for their suitability according to our needs, and as a result the model SWAT (Soil and Water Assessment Tool) (Arnold et al. 1993; Srinivasan and Arnold 1993) seems to meet these requirements. SWAT is applied to predict the impact of land management practices on water, sediment and agricultural chemical yields in complex watersheds with varying soils, land use and management conditions over long periods of time. Rather than incorporating regression equations to describe the relationship between input and output variables, specific information about weather, soil properties, topography, vegetation, and land management practices occurring in the watershed are required. The physical processes associated with water movement, sediment movement, crop growth, nutrient cycling, etc. are directly modeled, and users are able to study long-term impacts.

For the simulation of erosion processes in subbasins, the physically based model Erosion 3D (Sächsische Landesanstalt für Landwirtschaft 1996) was tested. It can be used for the simulation of erosion processes during single strong rain events as well as for the calculation of annual or several year values.

All input data used for model applications had to be prepared and modified according to the special calculation characteristics of both models (Volk and Steinhardt 1998). This is also important for the derivation of indicators for environmental conflicts, land use, water balance and morphology interactions in catchment areas.

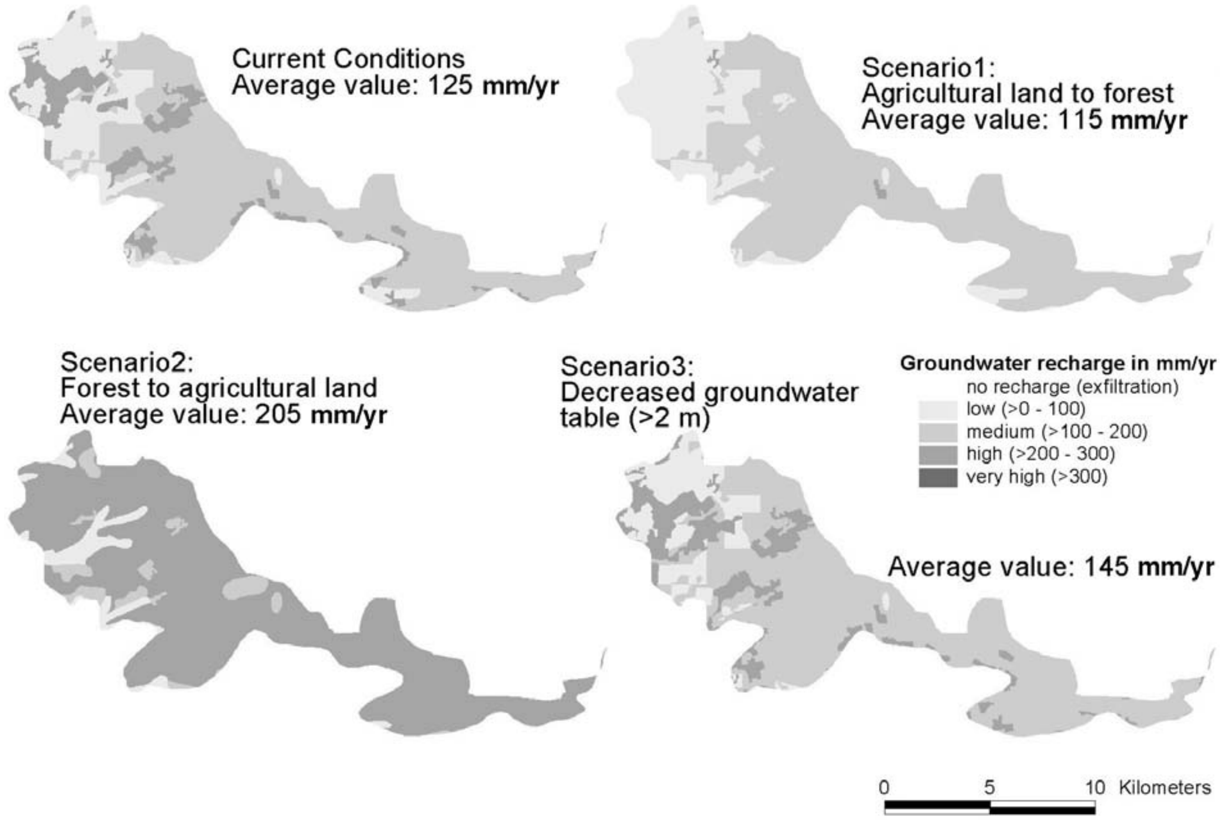

Figure 3. Land use scenarios: Several land use changes have been simulated for a landscape unit in the northern part of the Dessau district ("Fläming"). The calculations show the impact of possible land use changes on the water balance of this region.

Results

Water balance, scale larger than 1:50 000

Calculation of the basic elements of the water cycle
The calculations allow regional assessment and comparisons between areas of higher and lower groundwater recharge and runoff in relation to the prevailing natural conditions and the land use types. The water balances were calculated for both test areas (Figure 2). In comparison with other areas in Germany, both test regions show low precipitation and groundwater recharge rates. The highest values are registered in both cases in the morainic parts of the study areas. Very low rates characterize the dry western parts of the Dessau region with prevailing black soils and cohesive substrate, with a high importance for the function of groundwater protection. Within the priority areas for groundwater extraction in these western parts, only the extraction wells are protected. But these sites are not necessarily the places where groundwater recharges and potential contamination take place. It becomes obvious that our calculations can be used for a better designation of priority areas for groundwater extraction.

Land use scenarios
Besides this regionwide analysis, scenarios were calculated about the impact of land use changes on the water balance for smaller test areas within the Dessau region (Volk and Bannholzer 1999). The database given for the whole district has a relatively coarse spatial resolution, so the scenarios are useful only for areas > 100 km^2 (Figure 3). The conflicts in this part of the district originate from groundwater contamination by nutrients and agrochemicals and the overlap with priority areas for groundwater extraction and forestry. Our calculations with different land use variants show for instance, that afforestation in this area affect the groundwater recharge only slightly, but could improve the groundwater protection by a decrease of the agricultural areas. A potential decrease of the groundwater table caused by increased water extraction would only lead to slight changes but to ecological impacts on the forest (dryness effects).

These scenarios can only give coarse identification of the impacts of land use changes on the water bal-

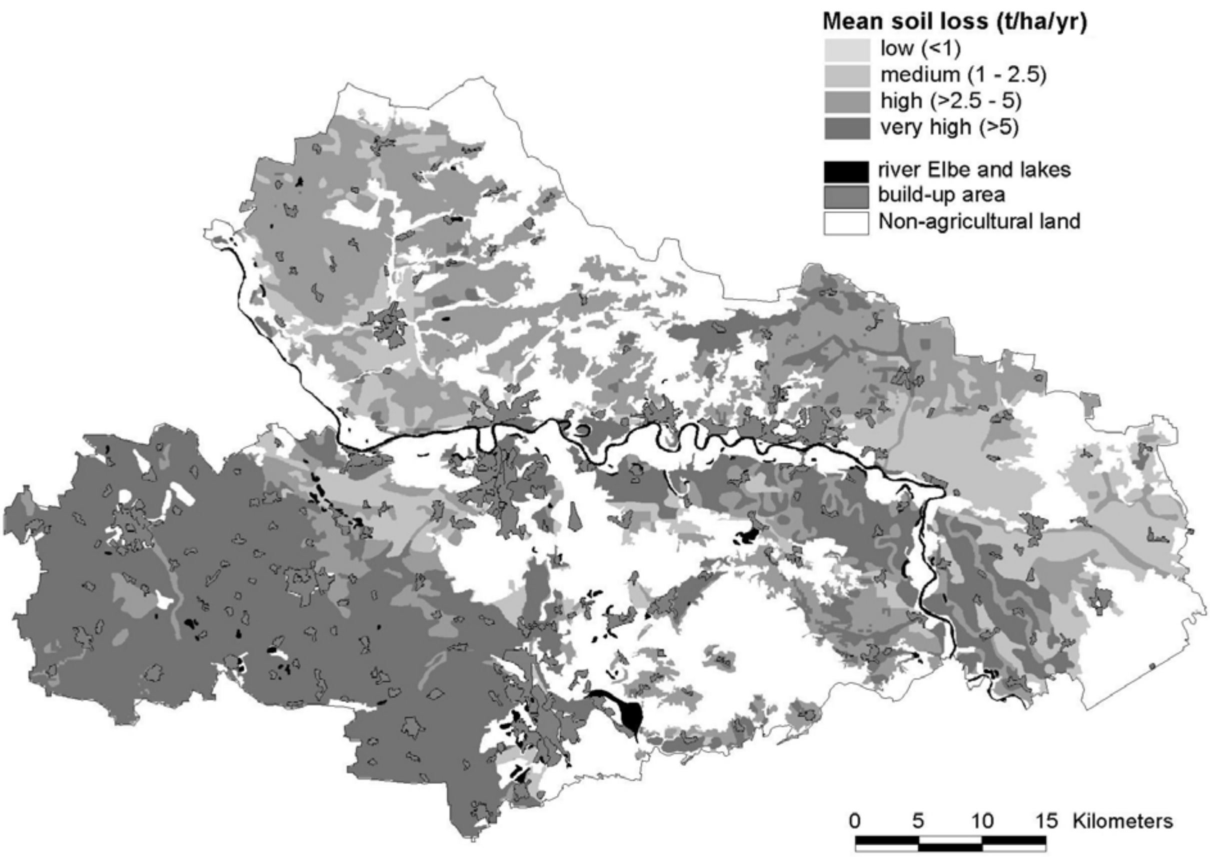

Figure 4. Estimation of the mean soil loss of agricultural areas in the Dessau distict: The potential mean soil loss is calculated with a modification of the USLE – adapted to regional conditions.

ance for relative large areas (average values). For more detailed studies concerning the conditions within these areas, the application of other models and a database with a finer spatial and temporal resolution is required, as shown below. Nevertheless, the presented results of the calculations are placed at the disposal of the local water management authority as well as the regional planning authority as a decision instrument.

Vertical and horizontal material and energy fluxes, Scale larger than 1:50 000

These water-carried fluxes depend essentially on morphological conditions and surface cover. The percolation rates and an estimation of surface runoff and interflow were received by the modification of the modeling results from ABIMO after Röder (1998). Relevant geomorphologic parameters (e.g. slope angle, slope exposition) are coupled with the data layers of land use, soil conditions, the modeling results and climate in GIS. The soil data were classified according to permeability, erosion disposition, etc. with given assessment methods (AG Boden 1994).

In a first step, areas were identified with the query function of the GIS, characterized by "arable land use", "percolation rate > 180 mm/yr" and "slope inclination 0–2°". These areas are defined as potential risk areas ("hot spots") with vertical material leaching (e.g., nutrients, pesticides) from agricultural areas. According to the calculation results, the main locations of the risk areas are situated in the northern, eastern and partly in the southern part of the Parthe area as well as in the northern and partly in the southern part of the Dessau region, with permeable substrates.

For a first estimation of the mean soil loss of the whole region, the modified Universal Soil Loss Equation ($R \times L \times S \times K_B$) as suggested by Bundesanstalt für Geowissenschaften und Rohstoffe (1994) was applied. The equation factors were modified and reduced (due to the size of the area) as follows: The R-factor (precipitation and surface runoff factor) was adapted to the conditions of the region (after Sauer-

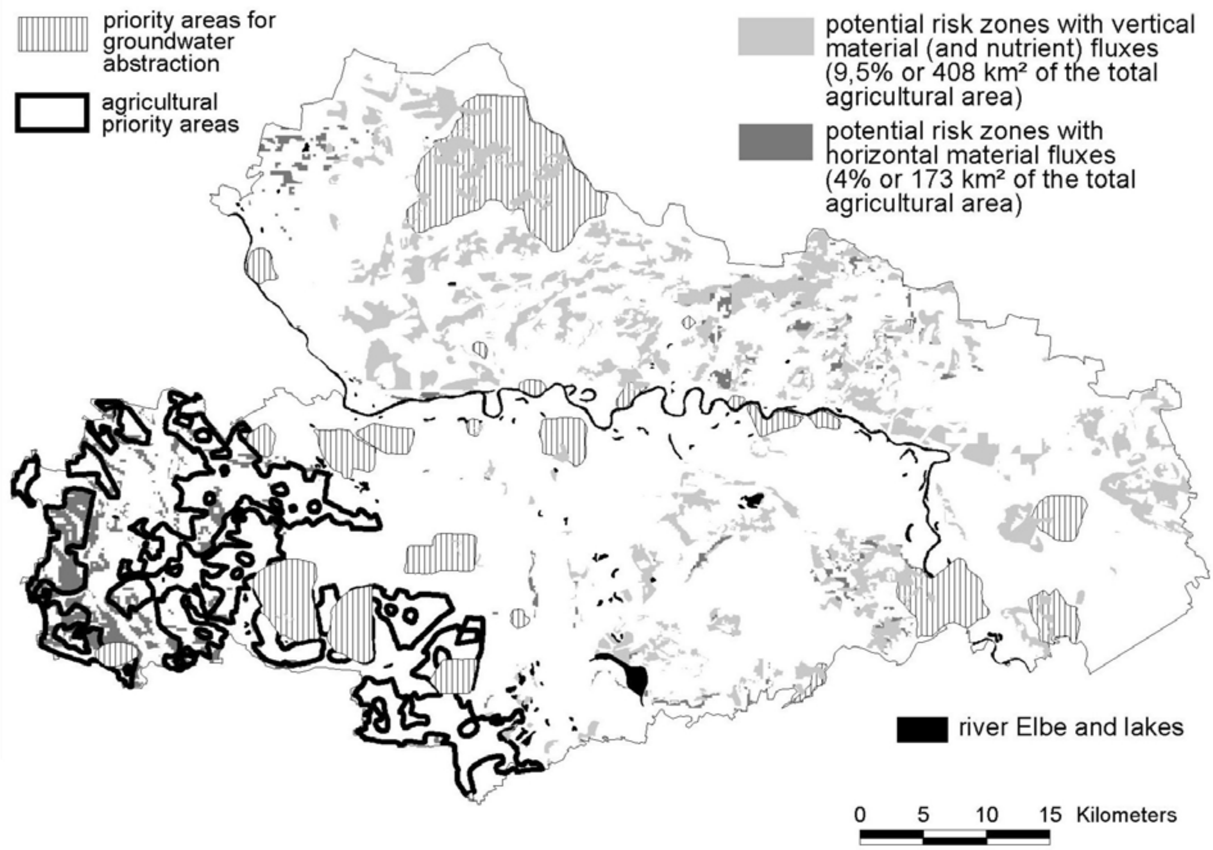

Figure 5. Dessau district: Potential risk zones with vertical and lateral material (and nutrient) leaching. Large parts of risk zones with lateral material leaching (erosion) determined mainly in the western parts are located in agricultural priority areas. In contrast, some risk zones with vertical material fluxes can be found in the northern district with permeable soils within the priority areas for groundwater extraction. Thus, environmental impairments resulted by multiple land use requirements (land use conflicts) can be indicated.

born (1994)), the slope length factor L was equalized to 2.0 (slope factor S remains), and the factor K_B determines the substrate dependent rate of the erodibility factor K (after Schwertmann et al. (1990)). In combination with these results (Figure 4), a first focus potential risk areas with lateral material (and nutrient) leaching from agricultural areas is enabled on selecting areas characterized by "arable land use", "cohesive substrate" and "slope inclination > 1°" (relatively high soil loss) and "middle to high surface runoff".

The study area is for the largest part flat, and as a consequence there are only few risk areas for horizontal material (and nutrient) flow. Nevertheless, the slopes of both the moraine and porphyry hills in the Parthe area and the chernozems in the western part of the Dessau region are predisposed for surface runoff (Figure 5).

Vertical and horizontal material and energy fluxes, Scale 1:25 000 to 1:50 000

The next step is the qualification and quantification of the vertical and horizontal water, material and energy fluxes from the designated risk areas.

These more detailled process-oriented studies require the application of watersheds as investigation units and a change of scale (Steinhardt 1999). Beside hydrological and morphological analyses, the application of a database with finer spatio-temporal resolution and different model systems allows the derivation of quantitative and qualitative information on the water, material, nutrient and energy fluxes in catchments and subbasins. In addition to a renewed calculation of the groundwater recharge and the surface runoff, this enables an improved differentiation of the risk areas. Additionally, useful information can be derived concerning potential risk areas at streams and rivers. The derivation of the topographic factor LS in Arc/Info (Grid Module, Hickey et al. (1994)) and the

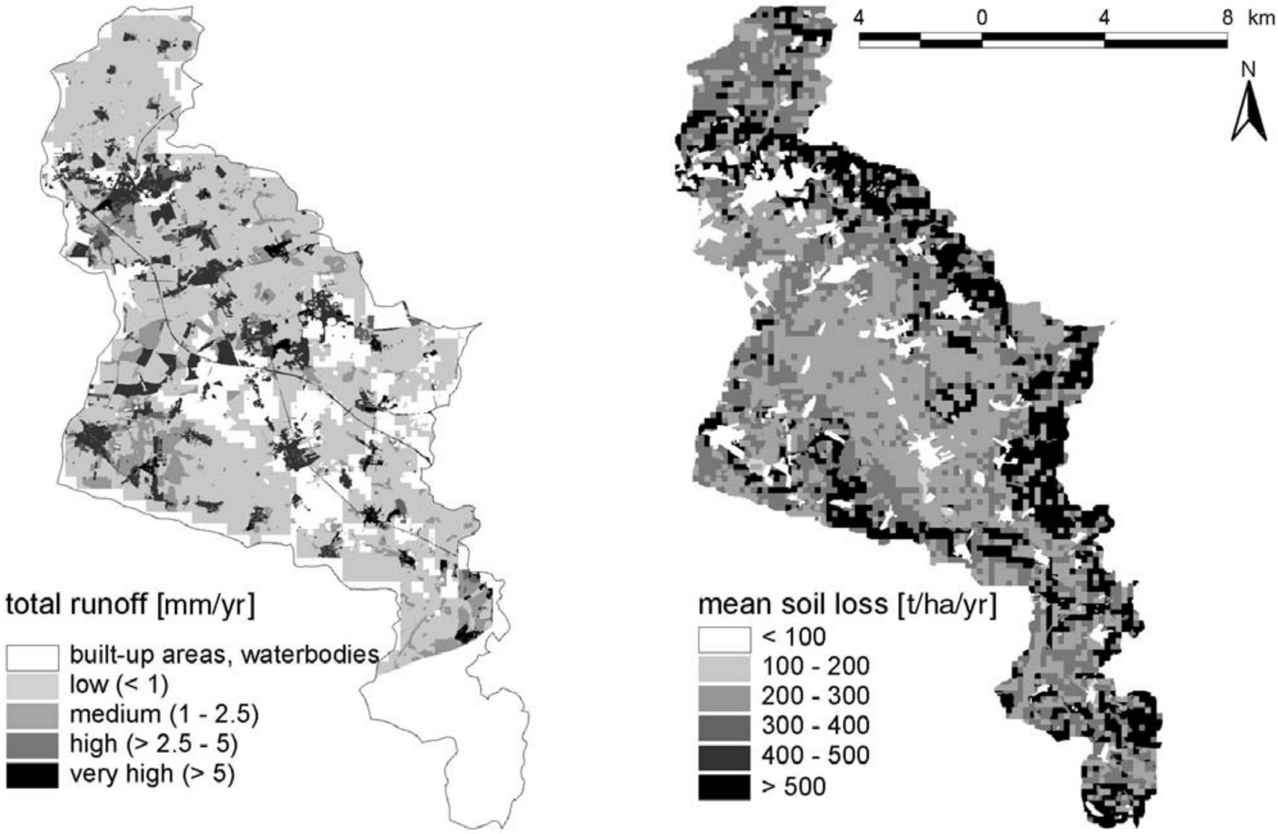

Figure 6. Estimation of surface runoff and mean soil loss within the Parthe watershed: Surface runoff values have been derived from ABIMO total runoff simulations using a method of Röder (1998). The estimation of the mean soil loss was estimated by using a modified version of the USLE (Figure 4). ABIMO is not suitable for the porphyry hill areas (solid rock) in the southern part of the area.

determination of the factors K_B (substrate dependent rate of the erodibility factor K) and R (precipitation and surface runoff factor, modified after Sauerborn (1994)) allows with their combination the application of a modified usage of the Universal Soil Loss Equation after Bundesanstalt für Geowissenschaften und Rohstoffe (1994) (Figure 6).

For the verification of the investigations (including studies about the interactions between surface runoff and material transport), several mappings and hydrochemical analyses of water samples are carried out. In a subbasin of the Parthe river watershed, the physically-based small-scale model systems E3D and SWAT were also tested for their scale specific applicability, as well as for the verification of the large-scale calculations (Figure 7).

The combination of the investigation methods from different scales allows us to check their scale specific applicability. Thus, the derivation of rules for the transmission from one scale level to another should be possible.

For instance, mapping as a small-scale method (digital mapping using a pen computer and GISPAD software) has been applied for the assessment of the effectiveness of the near-stream land along the 50 km Parthe riverbanks as a potential nutrient retention zone (Haycock and Burt 1993). The combination of defined buffer zones with a low potential nutrient retention with areas showing a high erosion risk, for instance, allow the idendification of areas with a high risk of nonpoint source pollution. All these investigations can be considered as an important forestep and a completion to the following application of the model SWAT for the mathematical description of the transport processes and the registration of nutrient in- and output.

Discussion and conclusions

The presented approach shows a hierarchical method for the investigation of the landscape balance on scales larger than 1:50 000 and scales between

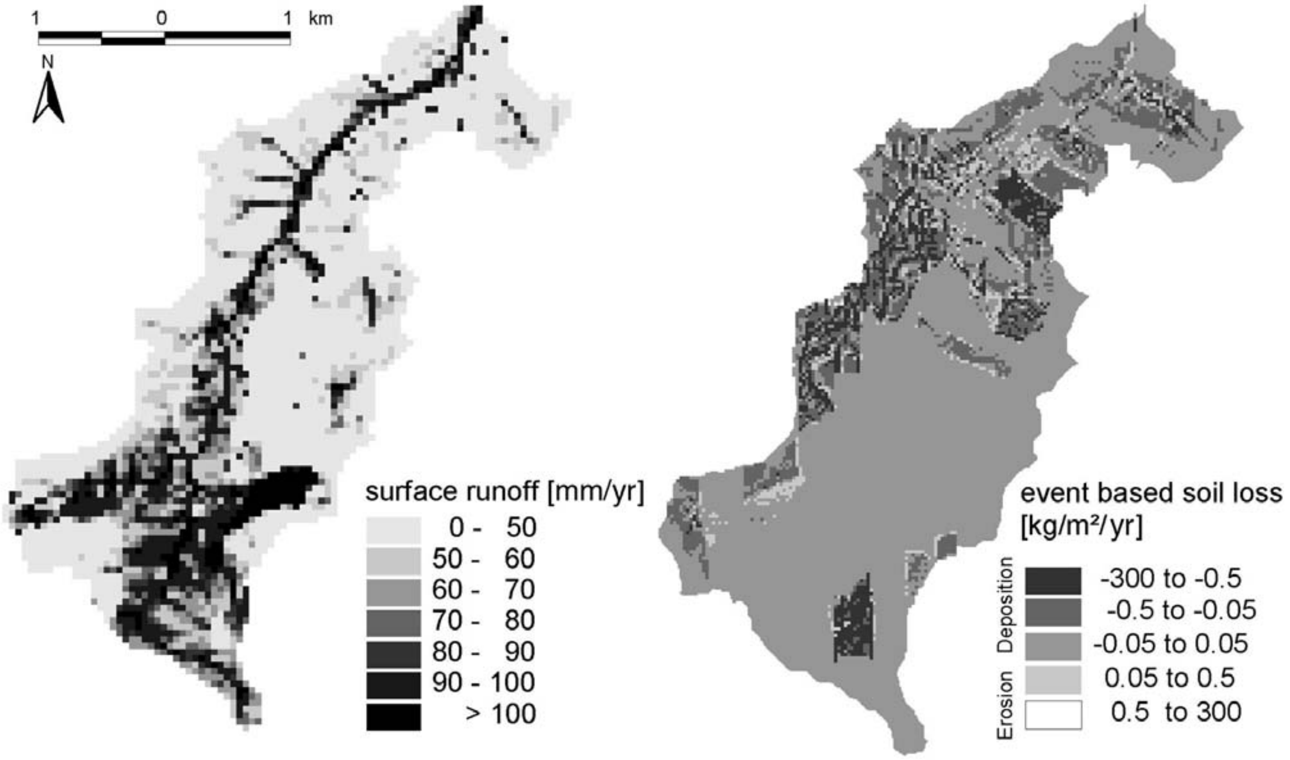

Figure 7. Modeling event-based soil loss and water balance within the Schnellbach subbasin: Physically-based models have been applicated for detailed investigations in this subbasin, using data with a higher spatio-temporal resolution (e.g., DEM, land use, soil). The simulations enable differentiations between both denudation (erosion) and sedimentation.

1:25 000 and 1:50 000 (Table 2). In a first step, the method allows the analysis of the basic elements of the water balance (runoff, evaporation) on the landscape scale. The calculation of scenarios (impact of land use changes on the water balance) for areas larger than 100 km^2 is possible, and the combination with other data layers allows a first rough identification of the mean soil loss, and potential risk areas with material and nutrient output and a regionwide classification. Assessment on this scale level is a coarse filter that gives the background information and identifies the properties for subsequent analysis.

In the second step, the scale level is changed to 1:25 000 to 1:50 000. At this scale, the identified risk areas are investigated in detail - with other models and a database with higher resolution in space and time. These calculations allow the recognition of water, material and nutrient transport mechanisms and the identification of nutrient or material input and output zones of areas or along streams and rivers. Additionally, qualitative and quantitative information about water and material transport can be derived. On this scale, the results provide detailled information about the impact of land use and land use changes on the landscape balance. One result is the derivation of recommendations for land use variants to achieve less material loss and nutrient leaching.

This method requires the availability of different data layers in a more or less homogeneous spatial resolution, because the integration of several data layers with different spatial resolution can lead to incorrect results. An additional integration of data layers including "time-space" components like climate parameters can even aggravate this problem. However, at present the data layers for most parts are available only on different scales. For the solution of this typical problem, we are developing intelligible methodical modifications for aggregation and generalization of input data in order to establish indicators for environmental conflicts, land use, water balance, and morphology interactions in catchment areas. Future increased application of GIS should improve the availability of the required databases.

A main problem of large-scale investigations arises with verification of the results. As measured data are mostly not available, the investigations have to be hierarchically linked to studies on smaller scales. Therefore, we are working together with groups spe-

Table 2. Scale specific approach for landscape balance investigation and evaluation. Suggestions for integrating the assessment results into landscape planning

Scale level	*Spatial planning level*	*Assessment Units*	*Data base (different spatial resolutions)*	*Model applications*	*Derived information and application possibilities*
1:50 000 and larger (areas $>10^3$ km^2)	**Regional level** (First identifications, coarse classifications)	– Landscape units	– soil	**ABIMO**	– water balance and land use scenarios for areas >100 km^2
		– Large river catchments	– morphology (DEM 100-250)	Description of the fundamental elements of the water cycle (e.g. runoff)	– (coarse) identifications of potential risk zones with (water) and material fluxes (combination of modeling results with assessment guidelines)
		– Areas with similar conditions	– climate		– analysis of land use conflicts on the regional scale and recommendations for land use (environmental and ressources management and conservation)
			– water		
			– watersheds		
			– land use		
			– landscape units		
			– spatial planning targets		
1:25 000 to 1:50 000 (areas 10 to 10^3 km^2)	**Regional level**	– Watersheds	– Soil	**SWAT ABIMO WaSIM-ETH AGNPS (CANDY)**	– identification of subbasins, streamnet and –order (flow of matter and nutrients)
	District level (Quantitative and qualitative information and assessments)	– Subbasins	– Morphology (DEM 40-100)	Investigation of vertical and horizontal matter and energy fluxes (Nitrogen, phosphorus and pesticides transport, Erosion)	– indication of rivers and streams affected by matter and nutrient input
		– Conservation areas	– Climate		– modeling of water balance and material and energy fluxes (qualitative and quantitative information) within the risk zones
		– Indicated danger zones (as above)	– Water		– combination of modeling results with assessment methods: recommendations for land use variants for decreased material/ nutrient output related to agricultural areas.
			– Watersheds		
			– Subbasins		
			– Land use		
			– Spatial planning targets		
1:10 000 to 1:25 000 (areas 100m^2 to 10 km^2)	**District level**	– Fields	(incl. mapping/measuring)	**Physical and empirical Models**	– Material and nutrient output (outwash) related to fields
	Community level (Detailed quantitative and qualitative assessments)	– biotopes	– soil	WEPP, AGNPS, CANDY	– Polyfunctional landscape assessment and land use optimization
		– river sections	– morphology		
			– (DEM < 40)		
			– climate		
			– water		
			– subbsins		
			– land use		
			– spatial planning targets		

cialized in microscale modeling and check the results in smaller areas in cooperation with water management authorities.

Future work will include the derivation and examination of further parameters of landscape ecological processes and structures (e.g., potential nitrate leaching, Franko et al. (1997)). We will strengthen the communication with administrative and environmental authorities to promote contribution of landscape ecological knowledge to landscape planning processes. The investigations also serve as a basis for the implementation of sustainable land use.

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6.4 Models in landscape ecology

6.4.1 Introduction

Society needs a way to handle a landscape as a whole, so that the human manipulative capabilities do not have too much headstart over our knowledge about the impacts of these manipulations (Odum 1969). However, extent and rate of effectuate changes in landscapes still exceeds, to a high degree, the scientific capability to reliably predict long-term impacts of technological developments on natural cycles and processes. Human impact on landscape pattern, material fluxes, habitats for plants and animals, but also on socio-economic situations has in fact reached a degree that may lead to irreversible changes and put at risk the natural systems essential for life support. Thus, landscape ecology and other environmental sciences have to develop suitable and improved methods to assess the impacts of anthropogenic changes in landscapes and to develop a conceptual base for sustainable land use.

During the last few decades it has turned out that models are suitable instruments to improve understanding of natural or economic systems. Additionally, they seem to enable comparison and assessment of results from factors that are assumed to influence these systems. By formalization and generalization of the complex reality, landscape models – like any other kind of model – provide the opportunity to connect detailed knowledge of different disciplines (Leser 1991a). Thus, it becomes possible to assess the related ecological and economic consequences of alternative management strategies or potential impacts of human induced landscape changes. In spite of the recent progress, the evaluation of integrated dynamic landscape models is only at the beginning of a far-reaching development. This shortcoming stands to reason considering the lack of quantified data on some topics, the high complexity of the task, as well as the methodological problems to get data in landscape ecosystems. Wenkel (1999) describes the five steps of development from single models to complete model-GIS-integration, which is characterized by coupling and interactive information exchange between sectoral dynamic process models among each other and with a GIS (see Chapter 6.2), as well as interactive handling. This chapter deals with the development and application of models for the investigation of several parts of the landscape ecosystem including the state of the art on integrated dynamic landscape models. This includes both technical and theoretical aspects.

6.4.2 Landscape ecology: Models for the investigation of complex topics

A **model** is a simplifying simulation of complex shapes from the reality and not the reality itself. Complexity is, thereby, a feature that results from the modeler's perception of the system in question (Schultz 1997, Wenkel 1999). The view of the modeler and thus the spatio-temporal resolution of the treated system are, of course, influenced by the modeling objective.

In the past, several attempts have been made in order to approach methodologically this complex topic within its theoretical framework (Finke 1994). However, this goal has yet to be reached. This insufficiency is caused on the one hand by a specialization of bio- and geo-sciences. On the other hand, a lack of suitable methods to combine the perceptions of different ecological disciplines, socio-economy and computer sciences was and mostly still is the reason for insufficient entire landscape synthesis or modeling. Thus, a **huge amount of models** have been developed within the single subject areas of landscape research. These have mostly synthesized the existing sectoral process knowledge (Wenkel 1999). Using the example of models to investigate water balance and waterbound material fluxes, we will highlight some general tendencies, problems and potentials of their application.

6.4.3 Modeling the water balance

The first models for the calculation of the landscape water balance stem from the late 1940s. Since that time, and due to the manifold requirements of the investigation of the water balance, the development of these models has undergone rapid progress in various manners (Dyck 1983, Xu et al. 1996). In general, one can differentiate between three **methodological approaches** to the modeling of the landscape water balance today:

– **physical-deterministic models** that are based on the fundamental laws of physics (mainly hydro- and thermodynamics), chemistry, biology, etc.,
– **conceptual models** that consider these laws in a simplified way and work simultaneously with empiric approaches, and
– **empiric-statistical models** that are only based on empiric measured cause-effect-relations of system in- and outputs, without the demand to comprehend the basic legalities.

The transitions between these approaches are fluid. Furthermore, hydrological processes always show deterministic as well as stochastic features. Both are based on the inevitable simplification of the complex reality and the appearing defects and uncertainties that occur with the gathering of the input data (Nemec 1993).

According to the model type and purpose of modeling, it is possible to handle different **spatio-temporal resolutions**. In doing so, compromises have to be made mostly between targeted accuracy and the available data. In the case of investigating non-linear processes (e.g. precipitation - runoff), it has to be worked in hourly or daily steps, whereas for seasonal or year-specific qualities monthly or annual steps are sufficient. The potential degree of spatial resolution reaches from greatly aggregated approaches, in which the investigated watershed is subdivided in only few sub-basins with similar geophysical characteristics (lumped models), up to models that consider the variability of spatial structure (distributed models).

The possibility to work with data with a high spatial resolution is improved due to increased computer capacities, development of geographic information systems (GIS, see Chapter 6.2), and the increasing availability of digital data. In general, all input data used for model applications have to be prepared and modified depending on the specific calculation characteristics of the models (Petry et al. 2000, Volk and Steinhardt 1998). This is also important for deriving indicators for environmental conflicts, land use, water balance and morphological interactions in catchment areas. One main problem of large-scale investigations is verifying the results. As measured data are mostly unavailable, the investigation has to be hierarchically linked to studies on smaller scales (sampling and analysis at representative locations, mapping, measuring, and application of small-scale models) (Steinhardt and Volk 2000). Nevertheless, the application of these traditional methods is essential not only for verifying the modeling results, but also for improving basic knowledge about how the landscape ecosystem functions (Hauhs et al. 2000).

Society affects the fluxes of water, matter and energy within a landscape by the parameter land use. Models are used to describe the impact of land use changes on the **potential groundwater recharge** (Volk and Bannholzer 1999). For the most part, variants or **scenarios** (see Chapter 4.3) are investigated which base on assumptions on climate change or impacts of political decisions (Table 6.4-1, Figure 6.4-1, Volk et al. 2001).

Quite obvious land use changes result in appreciable shifts of the simulated total run-off, if related to the whole study area. The listed results (Table 6.4-1) do not allow derivations about local changes or conditions; which can be much higher than the averaged values. In this connection, an algorithm has to be considered which takes into account the predicted land use changes upon the area. The assumptions about the spatial distribution of land use changes can be made on the basis of considerations of plausibility, or additional models might be used (Fohrer et al. 1999).

Table 6.4-1: Examples of scenarios of land use changes and landscape water balance

region	orientation of the scenarios	land use changes	run-off change	authors
Northeast Germany	EU-agricultural reform	afforestation (4% of farmland)	-1%	Werner et al. (1997)
Northeast Germany	EU-agricultural reform	afforestation (32% of farmland)	-10%	Werner et al. (1997)
Hesse	agricultural policy: pasture premium	decrease of forest (42% to 13%) increase of farmland (44% to 73%)	+8%	Fohrer et al. (1999)
Hesse	agricultural policy: loss of animal kee-ping	increase of forest (42% to 49%) decrease of farmland (44% to 37%)	+2%	Fohrer et al. (1999)
Saxony-Anhalt	analysis of land use conflicts in priority areas (agriculture vs. groundwater protection)	afforestration of farmland	-9% to -2%	Volk and Bannholzer (1999)
Saxony	regional political decisions for the conservation of natural resources	consequences of different devel-opment scenarios (changes of protected areas, mining acti-vities, sealed areas, cultivation practice, afforestation)	-2,3% (in aver-age)	Volk et al. (2001)

Figure 6.4-1: Due to land abandonment and afforestation of mostly poor sandy soils in North-Eastern Germany, water balance and alterations are expected: Terminal morain land-scape at the Parstein lake near Eberswalde (Brandenburg, Germany) (Photo: O. Bastian 1990)

6.4.4 Modeling waterbound material fluxes and water quality

The **outwash and transport of material, nutrients and pesticides** is mostly linked to an amount of water flowing out of a region. This results in an input of this material into the groundwater and surface water with an impact on the water quality. The investigation of these processes is often concentrated on phosphate (particle-bound transport through erosion: horizontal processes) and nitrate (soluble transport through seepage: vertical processes).

Examples of such **nutrient transport models** are shown in Figures 6.4-2 and 6.4-3. Most of the nutrient load of surface waters originates from **non-point sources**. To analyze these processes, the application of distributed parameter models in combination with GIS seems to be a useful method. According to the relation of the material fluxes in landscapes to hydrological processes (see above), most of the models investigating waterbound lateral and vertical material fluxes consist of a hydrological model combined with a material transport component. Several of these models are listed and described by Bork and Schröder (1996) and Grunwald (1997). One of the latest innovative models based on physically approaches is EROSION 2D/3D – developed in the 1990s in Germany (Figure 6.4-4). Several studies are dealing with the application of models to investigate the impact of political decisions and related land use changes on waterbound-material fluxes and water quality (Franko et al. 2001).

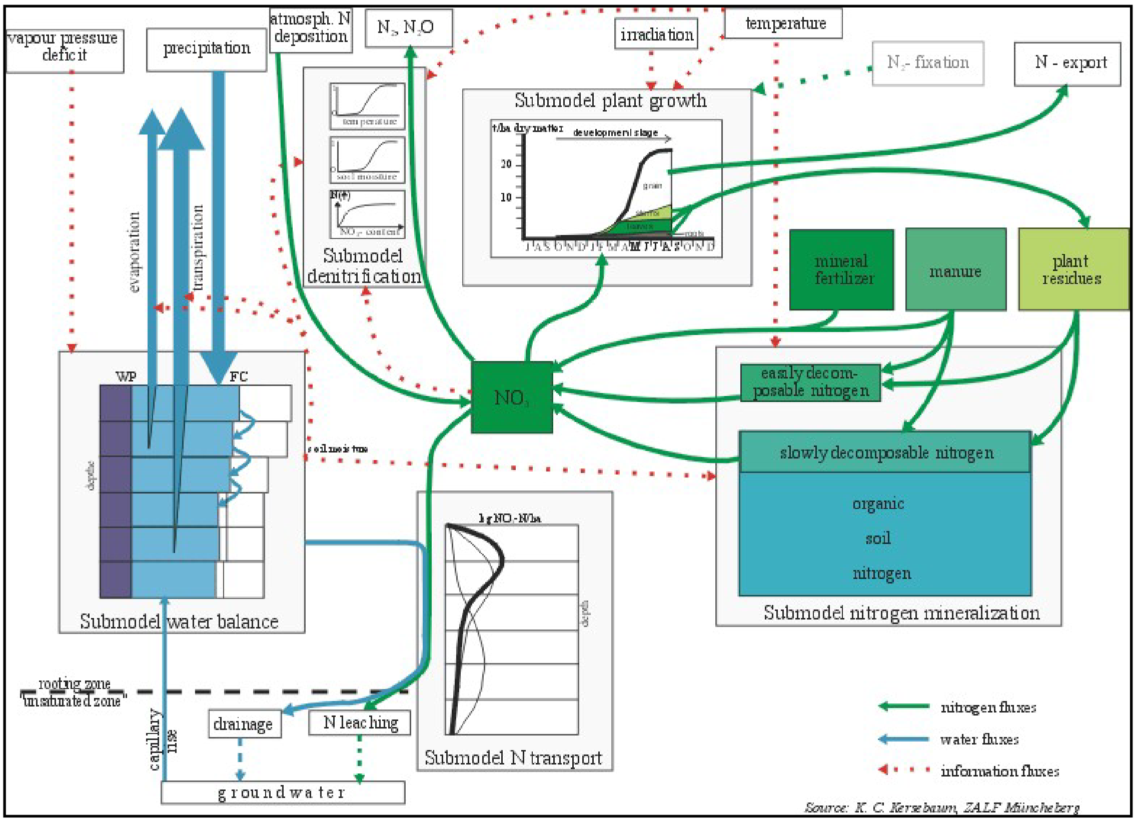

Figure 6.4-2: Modeling the nutrient balance: HERMES (Kersebaum 1995)

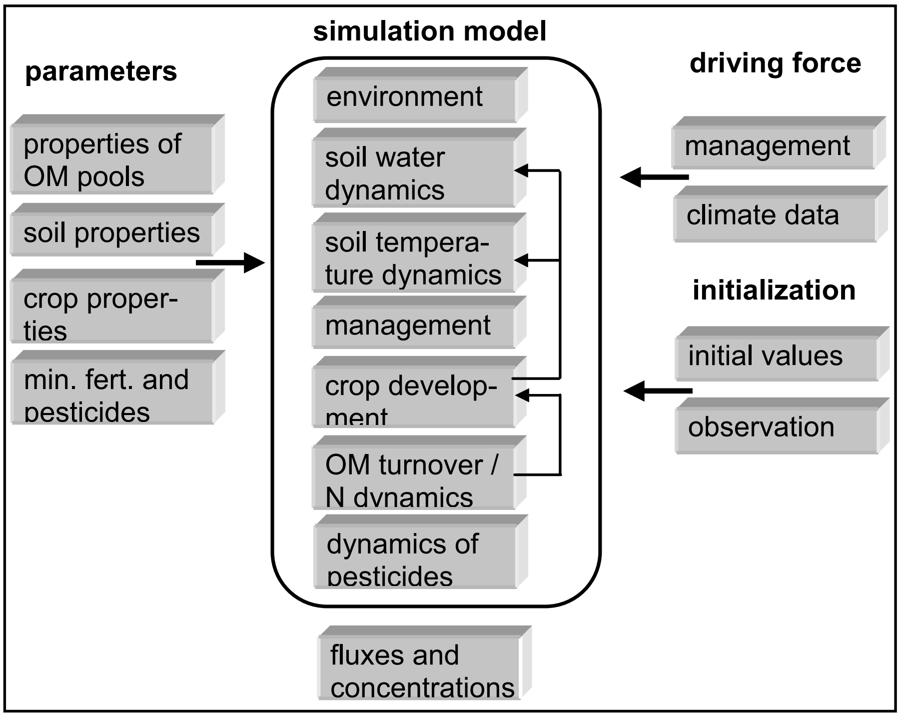

Figure 6.4-3: Modeling the carbon and nitrogen dynamics: CANDY (Franko 1997, Franko et al. 2001). The simulation system CANDY (CArbon and Nitrogen DYnamics) has been developed in order to describe the dynamics of the carbon and nitrogen turnover in the soil, as well as the dynamics of soil temperature and soil water content. All processes in the unsaturated zone are described for a one-dimensional soil profile. The system consists of both a simulation model which is imbedded into a user interface, and an environmental data base providing information about driving forces, initial values and series of measurements

6.4.5 Research sectors, models and scales

At present, many of the physically based approaches with a high spatio-temporal resolution cannot be effectively applied to medium-sized watersheds (Fohrer and Döll 1999, Grayson et al. 1992) because of the huge amount of input parameters required. Despite the much greater effort needed to parametrize, validate and run physically based models, simulated results often provide only slightly better or sometimes even worse correspondence with measured values than lumped-parameter models (Seyfried and Wilcox 1995). In this context, it should be mentioned that most of the common empirical models employed by environmental and planning offices and authorities rarely use more than three parameters (Hauhs et al. 2000).

Bearing these problems in mind, several models have been tested for their **scale-specific applicability** with respect to the time schedule and topics of research projects (Krysanova et al. 1996). Before applying a model, the algorithms used have to be checked. For example, most of the models that have an erosion component are based on different versions of the **USLE**

(Bork and Schröder 1996, see Chapter 5.2.2). It seems important to be able to adapt the model algorithms to the specific conditions of a study area. As most of the models were developed within research projects carried out in specific study areas, the possibility of transferring these methods to other regions needs to be tested.

input parameters		
relief parameters	precipitation parameters	soil parameters
digital elevation model	duration of rainfall rainfall intensity	roughness (Manning) erosion resistivity canopy cover texture bulk density initial moisture organic contents

preprocessor module		
control parameters (user defined)		
parameter for flow distribution threshold for flow accumulation		correction factor deposition coefficient
computation of		
slope aspect real (filled) area flow paths drainage network flow and path length upslope area catchment boundary		infiltration rate

calculation module	
momentum of droplets	momentum of flow

output parameters	
output parameters related to a point	output parameters related to element
runoff [m³/m] transported sediment [kg/m] sediment concentration [kg/m³] grain size distribution [%]	overland flow erosion / deposition for element [kg/m²] or element's upslope area resp. [t/ha] channel flow runoff [m³/m] transported sediment [kg/m] grain size distribution [%] erosion [t/ha]

Figure 6.4-4: EROSION 2D/3D (Schmidt 1994, von Werner et al. 1999). E2D is a physically based soil erosion model for single slopes. The model calculates the amount of eroded material, the runoff volume, as well as the material deposition along a slope profile at single precipitation events. The model consists of three parts: the digital slope model, the erosion model, and the infiltration model. E3D is mainly based on the same algorithms like E2D. Additionally, the description of the spatial distribution of erosion processes is enabled by including a digital terrain model into the calculations

During the last years it has become clear that the solution of complex problems requiring knowledge from different scientific disciplines cannot always follow a single model application. Integrated modeling requires the usage of a common database, spatial and temporal scales have to be compatible "and a smooth exchange of data and results between the sub-models must be guaranteed" (Weber et al. 2001). For these purposes, oftentimes two, three or more stand-alone models in the fields, e.g. of (agricultural) economics, ecology and hydrology are developed or adapted joining in an **integrated model system** (Horsch et al. 2001, Weber et al. 2001). This requires the close cooperation of the research groups of the different scientific disciplines. The models are mostly integrated using GIS, and as "integrated model system" they are thought as instruments or tools with which political decision-makers will be able to evaluate land use variants or alternatives. However, as mentioned above, Wenkel (1999) differs between the following five development steps from single models to integrated dynamic landscape models:

Step 1: development of sectoral ecosystem models and application of GIS for landscape analysis,
Step 2: coupling of a GIS with statistical assessment models (model-based assessment of the landscape potential),
Step 3: coupling of a GIS with sectoral dynamic process models (spatio-temporal assessment of selected landscape functions, see Chapter 5.2),
Step 4: partial model-GIS-integration (data bank-based automatic coupling and mutual information exchange of GIS with sectoral dynamic process models), and
Step 5: complete model-GIS-integration (coupling and mutual information exchange between sectoral dynamic process models among each other and with a GIS, as well as with interactive operating).

Analyzing recent development, it has to be pointed out that most of the models can be assigned to the steps 1 to 4. In spite of various approaches to this direction one will find only few examples following the idea of dynamic landscape models (step 5). The main reasons for this lack may be the manifoldness and complexity of the methodological and research organizational problems to master. However, most of the modeling is still sector-oriented but uses increasingly the potential for coupling dynamic process models with GIS in the sense of landscape models (step 5).

6.4.6 Landscape models

The view taken in landscape modeling is that a landscape is understood as a spatio-temporal structure. The research object determines which components of the entire complex "landscape" have to be included in the scientific consideration and description. It is, thereby, not possible to include the description of the overall complexity of a real existing landscape. Hence, a landscape is described mostly on a meso-scale level, which enables a higher area-acridity in comparison to the global level, but not reaching the high spatio-temporal resolution of the local level (micro-scale).

First prototypes of dynamic and transferable landscape models were developed at the beginning of the early 1990s at the University of Maryland (USA). They integrate biological and physical processes and consider essential processes and their interactions with landscape structures (Maxwell and Costanza 1994). Beside this they enabled a distributed respectively spatial explicit simulation of process behavior in landscapes.

Nowadays work with **mesoscale level** models has become more and more important. They have to fulfil primarily a strategic task and serve as an assessment of the efficiency of alternative measures (Horsch et al. 2001). As these models should enable political decision making, they are considering cost-benefit aspects and thus include both **ecological and economic components**.

Because of their intermediary reference level, the conception of landscape models often requires a tightrope walk. On the one hand, the complexity of the man-environment-system has to be considered in the sense of the holistic approaches of global models. On the other hand the model structure is determined by the necessity of reduction to a few relevant factors in order to illustrate cause and effect correlation with the aid of technical-functional partial models. This results in a **simplification of the reality**, but also in a **systematization of complex correlation and interactions**.

With the simultaneous consideration of ecological and economic factors a resolution of the problem of coupling the different spatio-temporal scales of the different scientific disciplines can be found. This is especially true in consideration of the fact that the factor "space" is rarely of interest for economic models. Additionally, ecological and economic models consider the factor "time" to a very different degree. Therefore often a comparative-static approach is used that compares two static mapped conditions with each other. An **interdisciplinary modeling** requires the coordination between the time horizons of each research disciplines.

We now present **two examples of landscape models**. Formation and structure of the landscape model, as well as the couplings between the modules of the model are depending on the objective of the research project. Table 6.4-2 shows an example for modules of the landscape model "Kraichgau" by Dabbert et al. (1999) that has been developed for the analysis and assessment of environmental impacts in agrarian landscapes. These modules are represented by several single assessment algorithms or models and form the landscape model by various input- and output-connections among each other.

The prime objective of the most landscape models is to create an integrated approach to economic and ecological processes in a watershed. Figure 6.4-5 shows an example of an integrated ecological-economic modeling and evaluation framework. The objective of the study determines very much the spatial, temporal and structural resolution of the model. The following parts show the structure of a landscape and related topics on the example of the Patuxent watershed model (Voinov et al. 1999, http://iee.umces.edu/PLM).

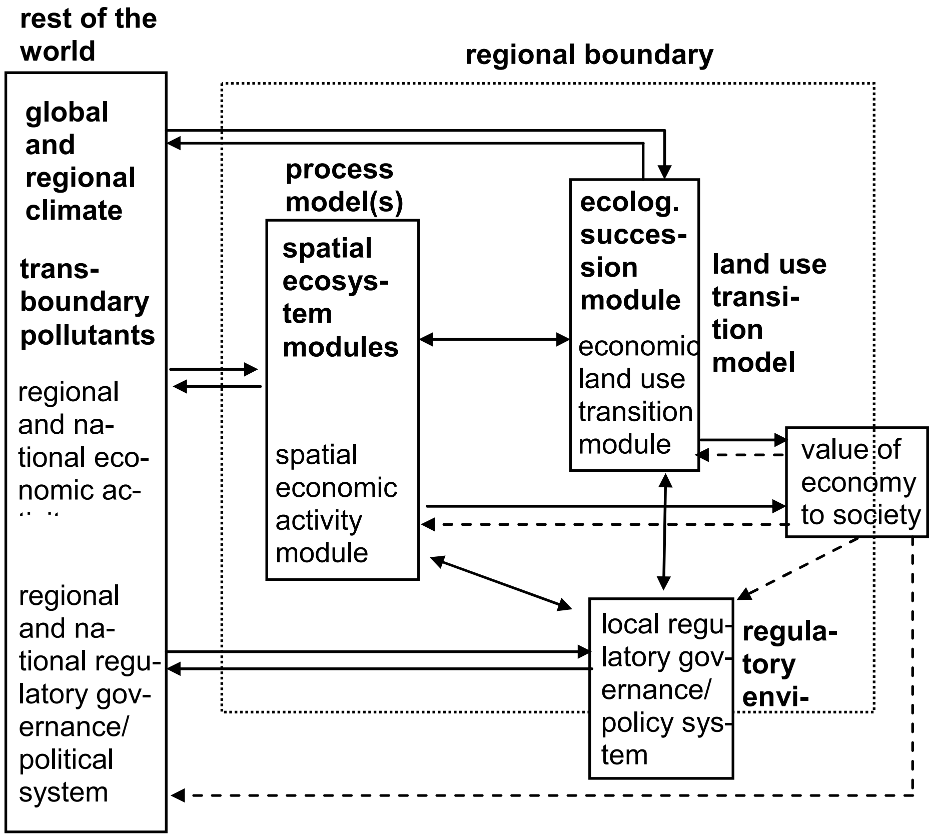

Figure 6.4-5: Integrated ecological-economic modeling and valuation framework: driving forces, initial values and series of measurements (http://iee.umces.edu/PLM)

In the spatial domain it has to be assured that the ecological, hydrological heterogeneity in the area can be represented as well as the socio-economic heterogeneity. Two types of **spatial design** have been mostly used in watershed modeling:

Lumped network based units: the whole area is subdivided into regions based on certain hydro-ecological criteria. These may be subwatersheds of certain size, hillslopes, areas with similar soil and habitat properties, etc.

Grid-based units: the landscape is partitioned into a spatial grid of unit cells. The cells may have different size but their geometry is the same. This approach allows cell attributes to change during the model run.

Table 6.4-2: The thematic modules of the landscape model "Kraichgau" (Dabbert et al. 1999)

thematic module	description
module nitrogen	soil-related description of the potential risk of groundwater contamination by nitrate input
module erosion	soil-related description of the potential risk for soil denudation by erosion
module economy	illustration of the impact of agrarian policy on agricultural companies (farms) (close sectors)
test modul farm modeling	estimation of changed boundary conditions on companies (farms) (local levels)
module nitrate	description of the nitrate loading in dependence to the cultivation practice
module nutrient input	modeling of the nutrient input in biotopes
module area-relation	generation of area-concrete data from aggregated data

One possibility for the **temporal design** of landscape modeling is the definition of fixed time steps according to the goals of a study, e.g. they have to be long enough to illustrate the impacts of political decisions by models or limited by the temporal borders for assumptions on economic structure and development. Other approaches assume that in time it is possible to represent the system as a sequence of independent discrete events.

With respect to **structural design** we have to state that landscape models are more and more process-based. The processes considered are mostly related to climatic conditions, hydrology, nutrient movement and cycling, terrestrial and estuarine primary productivity, and decomposition, etc. As mentioned in Chapter 6.4.3, the hydrologic processes are fundamental for the models, simulating water flow vertically within the cell and horizontally between cells. Nutrients cycled through plant uptake and organic matter decomposition, etc. The model should incorporate a modular structure. This allows individual modules to be designed and tested independently, prior to running the full model with all modules. Figure 6.4-6 shows an example of the structure of a landscape model. A landscape model is not a "universal model" but a "meta model" which holds a multitude of very different modules in a model bank.

The success of **model calibration** is very much dependent upon the available data. Calibrating and running a model of this level of complexity and resolution requires a multi-stage approach (see Chapters 6.4.3 and 6.4.4). However, from a scientific point of view the **validation** of dynamic landscape models is awaiting a satisfactory solution to the problem caused by lack of available high-resolution data, as well as by a lack of suitable strategies for the validation of complex models. Wenkel (1999) points out that an ensemble of methods could lead to a solution of the problems.

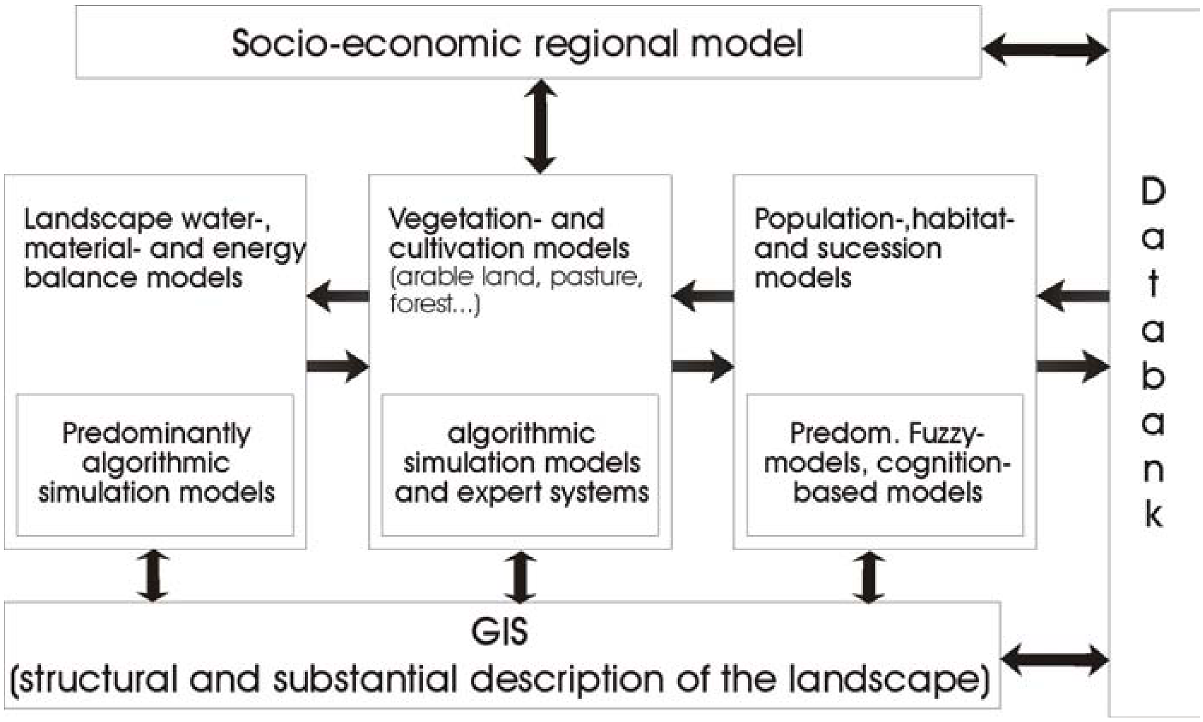

Figure 6.4-6: Main modules of a the dynamic landscape model „MLM“ (after Wenkel 1999)

6.4.7 Conclusions and outlook

There is an obvious trend from the development and application of single models to the development of integrated dynamic landscape models holding a multitude of very different modules in a model bank. Landscape models aim at the analysis and assessment of medium- to long-term ecological and socio-economic consequences of human caused landscape changes. Landscape ecology is understood as an inter- and transdisciplinary scientific branch (see Chapter 1.3). That means that an instrument trying to consider the landscape ecosystem from a holistic perspective and bridge the methodological and technical difference between scientific disciplines can only be developed in a multidisciplinary cooperation of many scientific fields. Due to Wenkel (1999) the future progress in landscape modeling will depend particularly on the success of unite theoretical and experimental ecologists with system analysts, computer scientists, and socioeconomists. Beside many scientific and technical open questions, some complex problems have to be solved in the future (Wenkel 1999, Volk & Steinhardt 2001).

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Meso-scale landscape analysis based on landscape balance investigations: problems and hierarchical approaches for their resolution

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Abstract

Varied utilization demands of society to the landscape are leading to an overlay of interests and thus to land use conflicts. Thereby, essential landscape functions like the regulation function (i.e. run-off regulation, groundwater recharge, groundwater protection, buffer functions of the soil, etc.) may be affected, and result in stresses to our natural resources like soil and water. The land use conflicts become especially obvious in a regional context. The diminution of such land use conflicts in terms of a regional management of environment and natural resources requires the knowledge of the response of the landscape balance to land use changes. The results of integrated landscape analysis enable the calculation of scenarios that allow the derivation of site-suitable land use variants with positive effects (decrease) to material out-wash from landscape parts and material inputs into surface water and groundwater. Numerous and complex methodological problems arise with such analysis, as well as with the investigation and assessment of the landscape water balance and water-bound material fluxes on the mesoscale.

As a contribution for the resolution of these problems, the authors present a hierarchical nested approach that interlinks scale-specific methods. Due to the complexity and difficult implementation from purely system-oriented approaches in both applied landscape research and planning, the connection to more pragmatic approaches is herewith striked. Thus, information about the impact of land use changes on the landscape balance, as well as the assessment of landscape functions for both watersheds and administrative units should be enabled. Beside the check of the scale-specific applicability of models (i.e. E2D/3D, ABIMO, ASGi, SWAT, modifications of the USLE), the transferability of parameter- and indicator systems for the assessment of the landscape balance on the concerned scale levels is also investigated. An important objective is thereby the optimization of the validity of landscape information for the spatio-temporal levels of the mesoscale.

Keywords: Scales and dimension; Hierarchical nested approach; Scaling; GIS-coupled modelling; Landscape functions

1. Introduction

Solutions to ecological and environmental problems require the understanding and prediction of natural and anthropogenic patterns and processes on all spatial and temporal scales. However, most ecological studies have been carried out on small-scales, and thus our knowledge is mostly limited to local scale environmental systems and interactions. For bridging this gap, recent and future landscape research dealing with regional scale analysis and assessment has to focus on the question: *How does spatial heterogeneity at meso-scale levels affect ecological processes?* On the

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base of such investigations, the development of scaling strategies and rules for extrapolating information from the local ecosystem to landscapes and regions is one of the most important challenge of landscape related research. Thus, we propose a hierarchical nested approach combining traditional investigation methods like field measuring and mapping with GIS-coupled modelling, scenario and assessment techniques. This approach is used for the investigation and solution of one of the most pressing environmental and ecological problems of today: *How does land use and land management changes affect the landscape structure and ecosystem processes respectively their interactions at the regional scale?* The treatment of such a complex topic has to include the consideration of the related impairment of essential landscape functions like the regulation function (i.e. runoff regulation, ground water recharge, ground water protection, buffer functions of the soil, etc.)—resulting in stresses of our natural resources like soil and water. The results of such integrated landscape analysis enable the calculation of scenarios allowing the derivation of land use variants adapted to the landscapes natural conditions showing positive effects (decrease) on material out-wash from landscape parts and material inputs into surface water and ground water. Numerous and complex methodological problems arise with such analysis, as well as with the investigation and assessment of the landscape water balance and water-bound material fluxes on the meso-scale. The results will help to improve the understanding of the interactions between ecosystem processes and land use/land cover change, and demonstrate a useful spatial hierarchical modelling and scaling approach for regional scale analysis and assessment.

2. Theoretical background: scales and dimensions

Because of their capability to manage and combine huge amounts of spatio-temporal data, GIS and GIS-based model systems and remote sensing methods are indispensable instruments in landscape-related sciences, especially at meso-scale levels today. On the other hand, more "classic" methods like mapping and measuring are used at local scales. In our system analytical approach, we combine theoretical and empirical methods, spatial modelling, GIS, and remote sensing in order to contribute to the discussion concerning scale and landscape theory. One important goal of this complex topic consists in the identification of spatio-temporal hierarchies of landscape processes in order to classify them according to their temporal (duration: short-term to long-term) and spatial scale/dimension (range: small to large), as well as to their intensity. We assume that as an important indicator for sustainability.

Scale is an essential concept in both natural and social sciences, and has been defined in several different ways (Neef, 1963; Goodchild and Quattrochi, 1997). In landscape ecology, scale refers primarily to grain (or resolution) and extent in space and/or time. Scale may be absolute (measured in spatial or temporal units) or relative (denoted as a ratio). Scaling, on the other hand, is usually defined as the process of extrapolating or translating (transferring) information from one scale to another, including scaling up and scaling down. Scale and scaling have become buzzwords in ecology in recent years, since environmental research emphases are changing increasingly from local to broader scales. The relationships between spatial pattern and ecological processes over a range of spatial scales is one of the most important investigation topics with the most unsolved problems in landscape ecology. Because of the scale multiplicity in spatial pattern and ecological processes, scale holds the key to the understanding of pattern–process interactions and becomes one of the corner stones concepts in landscape ecology (Urban et al., 1987; O'Neill et al., 1989; Dollinger, 1998).

Processes like macro-pores fluxes, soil erosion, air mass exchange, humus formation and decomposition, relocation of heavy metals, ground water table oscillation, climatic change (e.g. global warming), etc. will be assigned in such a matrix. Empirical studies indicate that many physical and ecological phenomena tend to line up approximately along the diagonal direction in a space–time scale diagram (cp, i.e. Wu, 1999, Fig. 1) although variations may be large sometimes (Innes, 1998). For instance, small-scale processes last short time periods and large-scale processes last accordingly longer.

Most of the terms and definitions regarding the spatial scales mentioned in Fig. 1 are based on varied investigations of German physical geographers. But no matter if the dimensions are referred to *topological*, *chorological* and *zonal* or if they termed *micro-*,

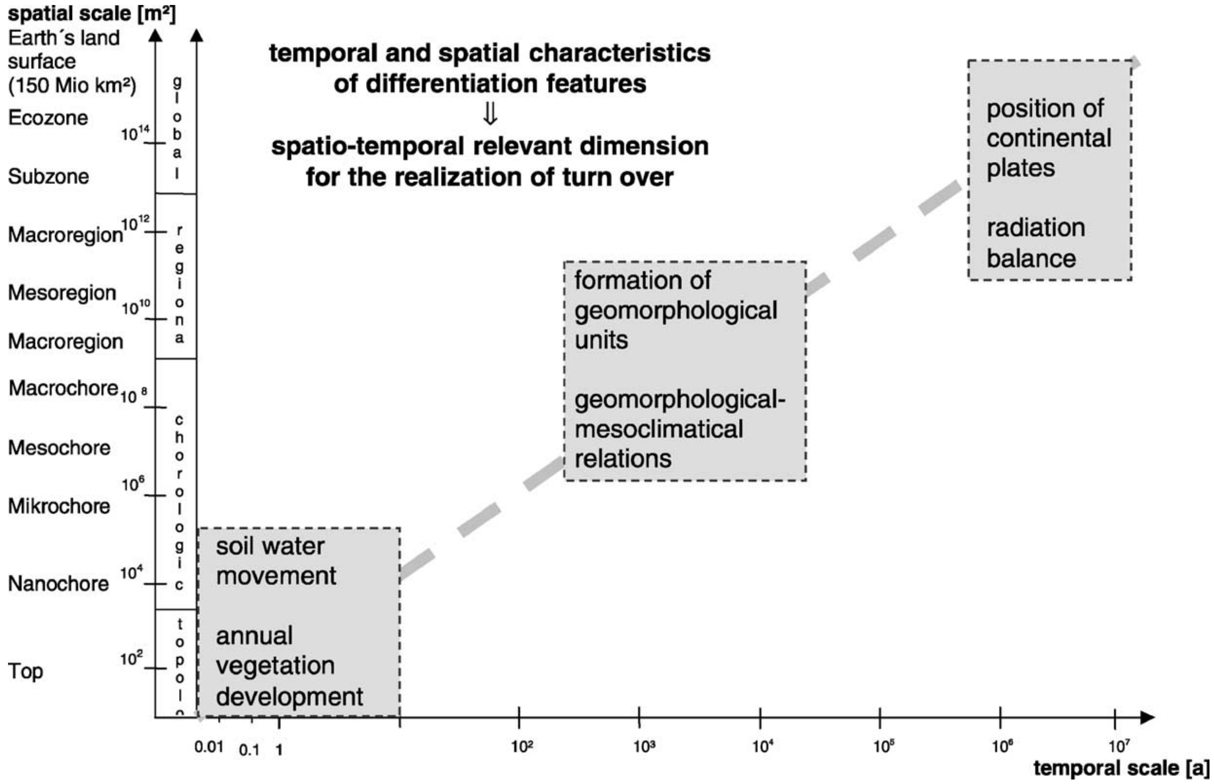


Fig. 1. Spatio-temporal hierarchies of landscape ecological processes (according to Wilmking, 1998; Barsch et al., 1988; di Castri and Hadley, 1988; Leser (1997); Moss, 1983; Schultz, 1995).

meso- and *macro-scale* there is a clear consensus concerning the principle verbalised by Neef (1963): Scale-specific approaches require scale-specific investigation methods and result in scale-specific information and insights. In addition to this axiom, we have formulated the following hypotheses:

1. A "sharp" scale delimitation is not possible. Scale levels are connected through loose couplings.
2. The basic components of the landscape balance are scale invariant. Solely their single factors vary from scale to scale.

A short explanation concerning the first hypothesis: The definition and delimitation of a specific hierarchical level is an important step within the problem solution process of our investigations. The selected scale level determines the main attention to be focused on a specific organisational level of the investigated system. Higher level processes proceed slower and can be considered quasi-constant. Constraints from these higher levels are expressed as boundary conditions. By contrast, lower level systems operate faster in their behavior. The rapid dynamics at lower levels are filtered (smoothed out) and only manifested as averages or equilibrium values. A system's 'description can only be effective when relating the selected focal level both to the adjacent higher and lower hierarchical levels. If one is interested in the effect of nutrients carried by a five minutes' precipitation event to a plant stand, the relevant subjects of observation are leaves, litter surface, fungi and fine roots. But if one is interested in effects of climatic changes over centuries, attention has to be focused on organic matter accumulations, while their hourly, daily and seasonal variations can be ignored. Hierarchy theory suggests that when a phenomenon is studied at a particular hierarchical level, the mechanistic understanding comes from the next lower level, whereas the significance of that phenomenon can only be revealed at the next higher level. Baldocchi (1993) called the three adjacent scales the reductionist, operational, and macro-scales, respectively.

The second hypothesis can be explained exemplary at morphology and erosion: On the local scale, surface roughness is one of the main factors that will affect erosion disposition, whereas for larger scales (up to river catchments) the factor's slope inclination, slope length, slope exposition up to the shapes of streamlet, -net, -order and direction of flow are responsible for erosion processes.

3. A hierarchical nested approach for watershed modelling

According to the complex processes of matter and energy transformation in landscapes, special attention is paid to the water as an essential element and mobile agent, and thus as the main transport medium in at least temporate climates. Watersheds in particular are suitable for the investigation of horizontal processes. They are considered as more or less closed systems suitable for the modelling of cycles of water and matter on the landscape scale. Their natural boundaries and hierarchical organisation form an appropriate structure for environmental impact analysis. The importance of such integrated watershed modelling becomes particularly obvious regarding the implementation of the European water framework directive that requires the comprehensive investigation and assessment of whole watersheds.

Thus, our investigations are focused on four nested watersheds of different size in central Germany: the Saale watershed (23.000 km^2), the Weiße Elster watershed (5.000 km^2), the Parthe watershed (350 km^2) and the Schnellbach watershed (8 km^2) (Fig. 2).

In order to achieve an implementation of the research results into spatial planning, science has look for the units that are relevant for planners. Spatial planning is organised hierarchically too, and until now the corresponding levels are administrative units like communities, counties, districts or federal states. Environmental research has to aim to a consideration and application of its knowledge, concepts and results in society and thus in spatial planning. Thus, the investigations have been focused not only on watersheds but also on administrative units. Administrative units have been used as assessment units when the project's

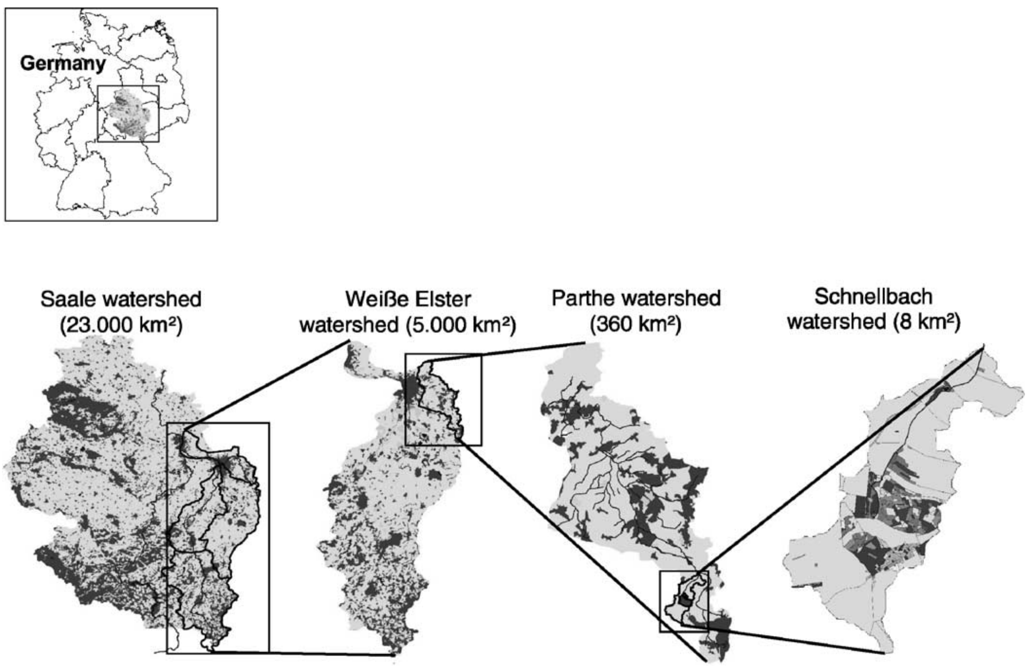


Fig. 2. Hierarchical nested watershed approach.

objective was to give recommendations for land use and land management to planning authorities. But now the EU water framework directive demands the development of management plans for meso-scale watersheds within the next nine years to obtain a sound quality and availability of both surface and ground water (qualitative as well as quantitative). That means that for the first time planning has to be cross-border and process-related, because (meso-scale) watersheds are now objects of both water agencies and spatial planning—which is also associated with an enormous reorganisation and reorientation of these authorities (cp, i.e. Leymann, 2001).

Independent from the type of the investigation area, the connection from one investigation level to the adjacent level has to be realised. This requires an extent of the above-mentioned scale-specific approaches by cross-scale investigations. Thus, we suggest a combination of top–down and bottom–up approaches. Thereby, we approach to the meso-scale from different directions: On the one hand, detailed investigations are carried out by using measurements and mappings on smaller scales. Because of the high time and work exposure most of the environmental parameters can only be gathered and measured for short periods and in small areas. Thus, the suitable investigations carried out on larger scales are balancing, modelling, typifying and classification (also based on remote sensing data) on the other hand. Hence, the loss of detailed information is compensated by a gain in overview information about structures, relationships and interactions. The results of the top–down investigations provide a basis for the designation of potential risk zones, for instance, for vertical and horizontal material and nutrient out-wash. By contrast, bottom–up investigations are essential on the one hand for the validation of the large-scale studies, and on the other hand they are needed for improving the knowledge about the process behavior and systems. GIS-coupled modelling is a method applied across all scales.

4. Modelling on different scales

During the last decades simulation systems have been developed to powerful tools supporting the description of landscape processes at different scales. But the description of natural processes is inevitable aligned to an abstraction and simplification of the complex processes and interactions within the landscape ecosystem in reality. The degree of abstraction and simplification is determined by the treated object and the investigator and thus has been resulted in the development of several model types in respect of basic model principles, conceptual background, time frame and spatial approach. For micro- to meso-scale applications a large number of process-oriented (hydrological) models have been developed in the last decades (Herrmann, 1999; Volk and Steinhardt, 2001; Wenkel, 1999). The major problem limiting the possibility of modelling large watersheds is the increasing heterogeneity of environmental parameters, which is associated with a decreasing data accuracy and availability at these larger scales. Additionally it can be stated, that most of the small-scale models have been developed for the investigation of specific key questions, reflecting the research focus of the development team. As a result most models address only some aspects of landscape processes very detailed and sophisticated but often neglect others by simulating them with simplified algorithms. Compared to the large number of models for micro- and lower meso-scale catchments only few models are available which are developed specifically for applications in large river basins. Compared to small-scale models, the concept of such models is mostly much simpler. This is correlated to the simulation of the single processes and the implemented algorithms and methods, as well as their distribution concept and their temporal resolution. As a contribution to bridge the gap between small-scale and large-scale models, we have tested a series of existing models for their applicability especially on meso-scale levels in order to check out the upper boundary of small-scale models and the lower boundary of large-scale models assuming an overlap. We have been focused on the check of the scale-specific applicability of the following models describing (parts of) landscape processes:

- erosion 2D/3D
- modifications of the USLE
- the runoff simulation model ABIMO
- the integrated models ASGi and
- Soil and Water Assessment Tool (SWAT)

Table 1 gives an overview of several models, their capacities and operations, and their scale-specific applicability.

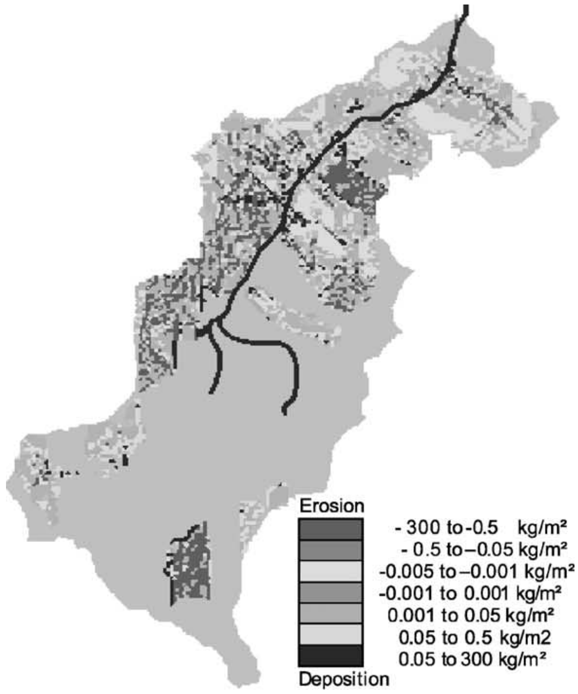
Table 1
Selected models and their scale-specific applicability

Model system	Scales	Objectives, operations and capacities
SWAT (Arnold et al., 1993; Srinivasan and Arnold, 1993)	Large river basins, sub-basins (up to several thousand square miles)	• Predict the effect of management decisions on water, sediment, nutrient and pesticide yields with reasonable accuracy on large, ungaged river basins. • Daily time step to long-term simulations. • Ground water flow model. • Basins subdivided to account for differences in soils, land use, crops, topography, weather, etc. • SWAT accepts output from EPIC (see below). • SWAT accepts measured data and point sources. • Soil profile can be divided into 10 layers. • Water can be transferred from channels and reservoirs. • Basin subdivided into sub-basins or grid cells. • Nutrients and pesticide input/output. • Reach routing command language to route and add flows. • Windows/ArcView Interface. • Hundreds of cells/sub-basins can be simulated in spatially displayed outputs.
ABIMO (Glugla and Fürtig, 1997; Rachimov, 1996)	Meso- to macro-scale	• Description of the basic elements of the water balance on the landscape scale (long-term values of runoff and evapotranspiration). • “Mean runoff” is defined here as the difference between long-term mean annual precipitation and real evapotranspiration. This difference is equivalent to the total runoff. In the case of a solely vertical seeping of the water this value corresponds with the ground water recharge. • Thus, the value must be understand as the sum only indifferent of both surface and subsurface runoff. Therefore, the results have been modified with a runoff quotient (based on slope inclination and ground water level) after Röder (1998), which allows an estimation of the surface runoff and interflow.
ASGi (AGNPS + WaSiM-ETH) (Young et al., 1987; Schulla, 1997)	Meso-scale watersheds	• System of computer models developed to predict non-point source pollutant loadings within agricultural watersheds. It contains a continuous simulation, surface runoff model designed for risk and cost/benefit analysis. • The set of computer programs consist of: ○ input generation and editing as well as associated data bases; ○ the “annualised” science and technology pollutant loading model (AnnAGNPS); ○ output reformatting analysis; ○ the integration of more comprehensive routines (CONCEPTS) for the in-stream processes; ○ an in-stream water temperature model (SNTEMP); ○ several related salmonid models (SIDO, Fry Emergence, Salmonid Total Life Stage, and Salmonid Economics). • The application of AGNPS can be used for the calculation of soil erosion, sediment transport and nutrient yield (N and P). • The computed runoff and peak flow of the hydrological model WaSiM-ETH will be used as input for the linked model AGNPS.
USLE (Wishmeier and Smith, 1978)	Local (field) scale Modified versions allow limited application at meso-scale levels (cp, BGR, 1994)	• Calculation of the mean annual soil loss in t/ha under consideration of the pluviometric regime, the soil characteristics, the terrain morphology, the land cover, and the conservative practices eventually adopted.
Erosion 2D/3D (Schmidt, 1991; von Werner, 1995)	Sub-basins, farms, fields	• Simulation of erosion processes during single strong rain events as well as for the calculation of annual or several year values. • Based on a physical approach, characterised by a high spatio-temporal resolution and short calculation times.

Erosion 2D/3D (Schmidt, 1991; Schmidt et al., 1997; von Werner, 1995) is a computer-based model for the simulation of soil erosion by water. It has been developed mainly for the application on the soil erosion problems of the agricultural landscapes of Central Europe with respect to municipal and regional planning (respectively inventories and assessments), as well as for farmer advisory. The model allows the prediction of the amounts of soil erosion and deposition from both a single extreme rainstorm and a series of numerous rainfall events which occur within a longer period such as one year or a decade. E2D is a physically based soil erosion model for single slopes. The model calculates the amount of eroded material, the runoff volume, as well as the material deposition along a slope profile at single precipitation events. The model consists of three parts: the digital slope model, the erosion model, and the infiltration model. E3D is mainly based on the same algorithms like E2D. Additionally, the description of the spatial distribution of erosion processes is enabled by including a digital terrain model into the calculations. A small number of input parameters and a minimum expense required for their determination make a good case for the application of this model. According to the simplifications inherent to any kind of physically based mathematical simulation, certain limitations have to be considered when applying the model (e.g. no simulation of infiltration into macro-pores, neglect of impact of suspended matter on runoff or turbulence, assumption of uniform spatial distribution of rainfall intensity across the slope, erosion caused by intercepted rain (throughfall) and stemflow is ignored). The model approach is transferable to any other region. Another advantage is the temporal variability of the main input parameters, such as tillage, plant cover, initial soil moisture, etc.

The model has been applied at the small Schnellbach watershed. Fig. 3 shows the spatial distribution and the amount of eroded/deposited material in the case of a rainstorm event with a repetition probability of 100 years. For more advantages and disadvantages see Table 2.

The *Universal Soil Loss Equation* (Wishmeier and Smith, 1978), developed at the United States Agricultural Department (USDA) during the 1950s, is the most used equation to determine the mean annual soil loss, depending on the pluviometric regime, the soil



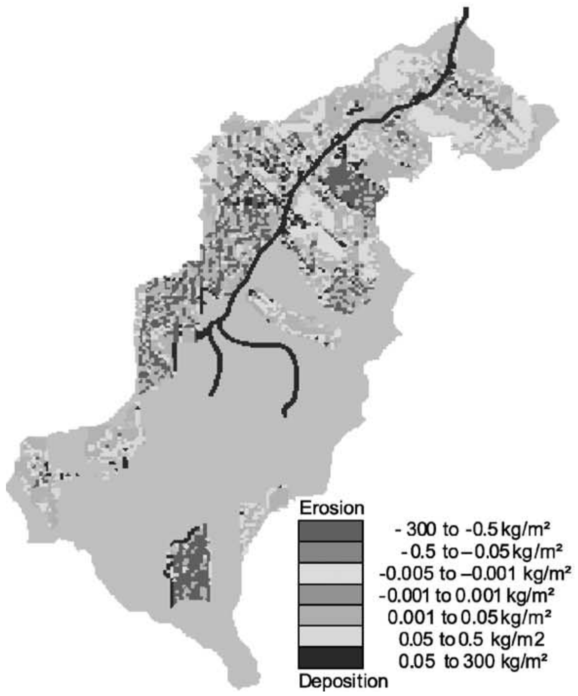


Fig. 3. Spatial distribution and amount of eroded/deposited material in the case of a rainstorm event with a repetition probability of 100 years within the Schnellbach watershed—simulated with E3D.

Table 2
Advantages and disadvantages of the tested models with respect to their applicability

Model	Advantage	Disadvantage
USLE	Simple single factors	Parameters are not site adopted
	Conditional comparability	Simple versions not universal applicable
Erosion 2D/3D	Included parameter catalogue	Application in larger areas problematic
	GIS-coupled (ArcView)	Transfer to areas out of central Europe difficult
ABIMO	Sufficiently tested for the central-German pleistocene sediment area	Unsuitable for higher spatio-temporal resolution
	Small number of parameters allow wide application	No differentiation between several runoff components
ASGi	Modular construction, GIS-coupled (ArcInfo)	No update due to project-related development
	Selection of algorithms depending on database	Errors within the matter and nutrient components
(AV)SWAT	Very user-friendly due to modular construction and the ArcView Interface	Risk to over-parametrise, difficult in larger areas
	Meso-scale application	Transfer and adaptation of the huge amount of parameters to regions out of the US is very time-consuming and problematic
	Selection of algorithms depending on database	

characteristics, the terrain morphology, the land cover, as well as and on the conservative practices eventually adopted. It does not predict the deposition, and considers only the rill and inter-rill erosion, not the gully erosion. The equation has the following structure: $A = R\ K\ L\ S\ C\ P$. The meaning of the single factors is assumed to be known. Schwertmann et al. (1990) adapted the equation—respectively their parameters—to the conditions in Bavaria (southern Germany). The structure of the USLE allows the calculation within a Geographic Information System: Each factor can be derived from existing data and represents a layer in a GIS environment. The multiplicative overlay between the factors produces the final erosion result. A lot of integrated models (e.g. SWAT, AGNPS) use the USLE or its modified versions as a core algorithm for describing soil erosion processes and related lateral material transport. This is the critical point, because the Wishmeier and Smith developed the algorithm originally for the application on the field scale. However, considering the simple structure of the equation and the availability of the basic data are an advantage and up to now there is no real alternative for erosion calculations. We have tested modified versions of the USLE at the Schnellbach and Parthe watersheds, as well as in the Dessau district. The Dessau district covers an area of about 4300 km^2 in Saxony-Anhalt (central Germany). We have applied different existing methods to derive the single parameters (Volk et al., 2001a). The results of these investigations contribute to an optimisation of the scale-related application of the USLE, respectively its single parameters. By showing the uncertainties, the results can also improve the interpretation of simulations calculated with model systems containing the USLE algorithm. Our special focus was on the regression equations calculating the R-factor on the one hand and on GIS-programmed algorithms calculating the LS-factor (Hickey et al., 1994; Hickey, 1999). Fig. 4 shows the results of some variants for the above-mentioned test sites.

Comparing the results of the different variants, it can be pointed out that the USLE algorithm provides satisfyingly results regarding a differentiation within a meso-scale study area which allows comparative estimations (areas with higher or lower erosion rates). Because of the huge amount of uncertainties of the meso-scale application of the modified versions, it should not be used to provide absolute values (amount of erosion in t/ha/a). For further advantages and disadvantages see Table 2.

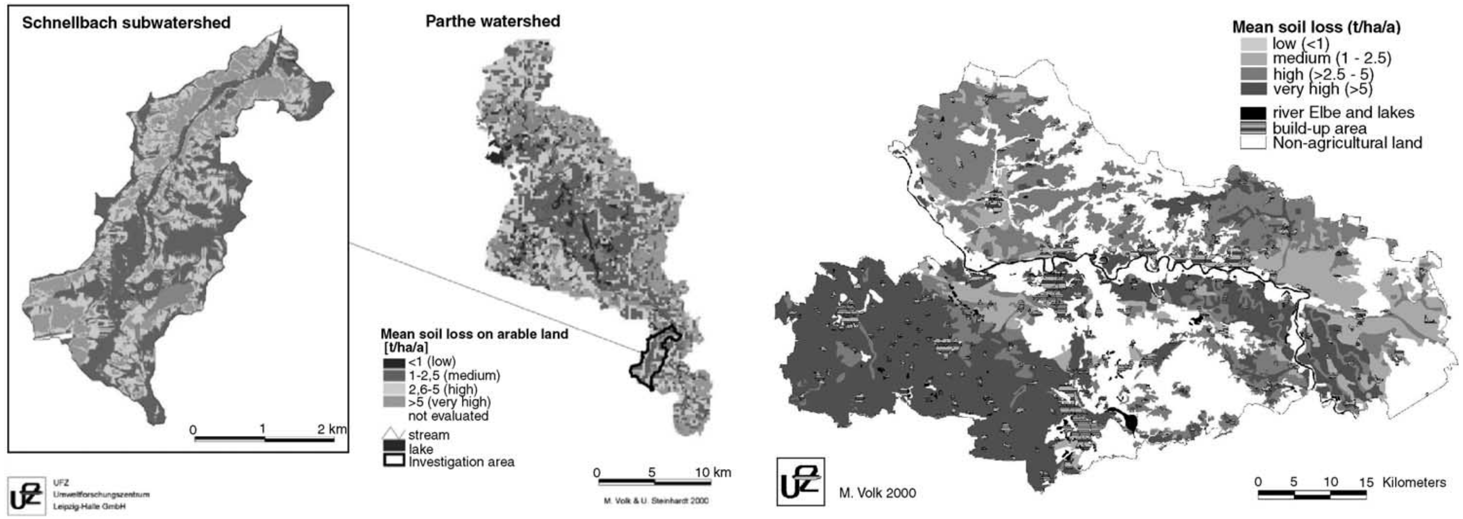


Fig. 4. Application of the USLE in the Schnellbach and Parthe watersheds as well as in the Dessau district. The single factors have been derived by using different methods. At the Dessau district, the highest values are calculated for the western parts and shows thus a high risk within the agricultural priority areas.

In order to get a first description of the basic elements of the water balance on the landscape scale (long-term values of runoff and evapotranspiration), we use the runoff simulation model *ABIMO* (Glugla and Fürtig, 1997; Rachimov, 1996). "Mean runoff" is defined here as the difference between long-term mean annual precipitation and real evaporation. This difference is equivalent to the total runoff. In the case of a solely vertical seeping of the water this value corresponds with the ground water recharge rate. Due to the fact that this situation is very rare in reality, this value must be understand as the sum only indifferent of both surface and subsurface runoff. Therefore, the results were modified with a runoff quotient (based on slope angle and soil moisture) determined by Röder (1998), which allows an estimation of the surface runoff and interflow (Fig. 5).

The calculations allow regional assessments and comparisons between areas of higher and lower ground water recharge and runoff in relation to the prevailing natural conditions and the land use types. The water balances were calculated for both test areas, Fig. 5 shows the ground water recharge values on the example of the Dessau district and the Parthe watershed.

Such calculations can be used, for instance, for a better designation of priority areas for ground water abstraction in order to improve ground water protection (cp, Volk and Steinhardt, 2001). Besides these region wide analysis, scenarios were calculated about the impact of land use changes on the water balance for smaller test areas within the Dessau region (Volk and Bannholzer, 1999) or within an integrated ecological–socioeconomic project dealing with natural resources protection and economic development (cp, Horsch et al., 2001; Franko et al., 2001; Volk et al., 2001b). For further advantages and disadvantages see Table 2.

ASGi has been developed between 1993 and 1997 by the University of the German Federal Armed Forces Munich in cooperation with the Bavarian Water Management Authority. ASGi is an GIS-coupled integrated modelling system continuously simulating runoff and matter transport. It is a grid-based modular model system, calculating the fluxes of water, sediment and nutrients in meso-scale watersheds. ASGi consists of two more or less independent subsystems: the widely physically based water balance model WaSiM-ETH (Schulla, 1997) and the deterministic-analytical AG-NPS model for the simulation of the water-bound

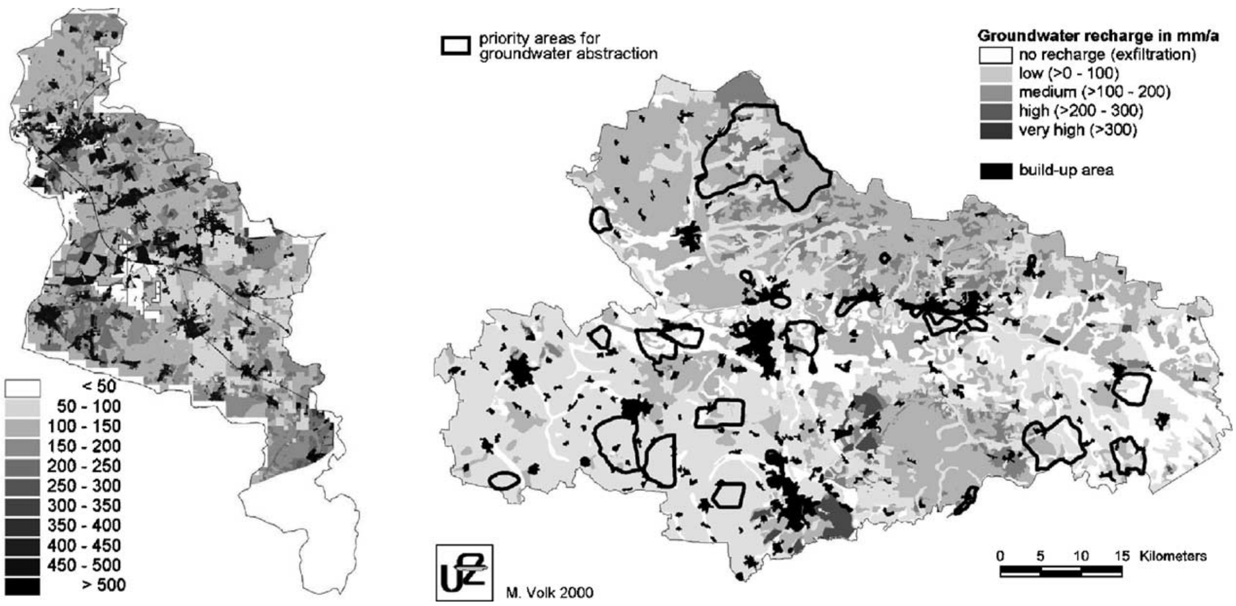


Fig. 5. Total runoff (mm/a) calculated with ABIMO (left: Parthe watershed, right: Dessau district).

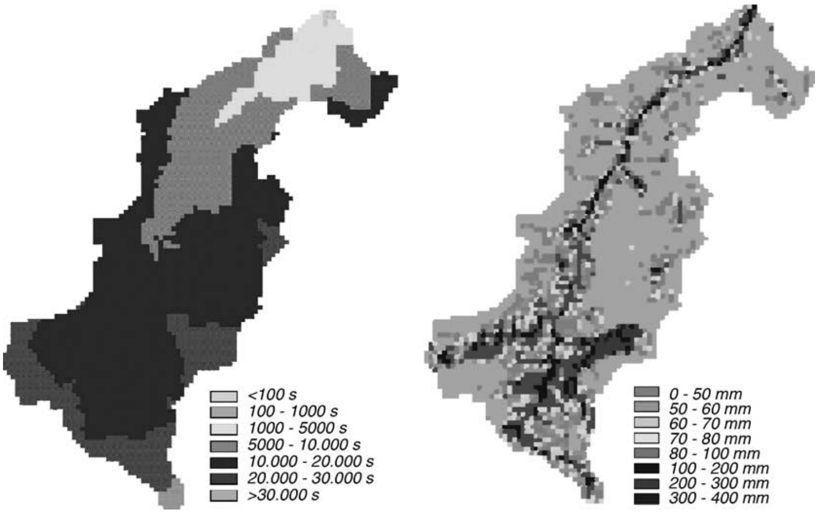


Fig. 6. Flow time (left) and amount of runoff (right) calculated with ASGi for the Schnellbach sub-basin. Land use scenarios can show the effect of different land use systems on these hydrological factors and enable evaluations about their potential impairment.

matter fluxes. The Agricultural Non-point Source (*AGNPS*) pollution model is a single-storm event-based simulation model. It requires the division of a watershed into grids, with each grid having 22 input parameter values. The input variables can be grouped into six categories: (1) watershed, (2) topography, (3) stream, (4) soils, (5) land use/cover, and (6) point sources. Output from AGNPS includes both a watershed outlet summary and detailed cell information. The outputs are related to hydrology (runoff and peak flow), sediment/erosion, and nutrient (nitrogen, phosphorus, and Chemical Oxygen Demand).

ASGi has also been tested in the Schnellbach watershed. The hydrological part of the model (WaSiM-ETH) can be evaluated as very good. Experiences of other research projects circumstantiate this (Bronstert et al., 2001; Niehoff and Bronstert, 2001). Fig. 6 shows flow time and surface runoff for the investigation area. In contrast, the water-bound matter transport model did not provide any satisfying results. This was due to the fact, that the AGNPS model has been reduced to the matter transport and put under a new environment. A lot of bugs have been recorded in this subsystem. The further development of ASGi has ended with the finish of the project which is in general a problem at project-related development of models. Rode and Lindenschamidt (2001) have been combined WaSiM-ETH and AGNPS also and received better results with their approach. For further advantages and disadvantages see Table 2.

SWAT is a river basin, or watershed scale model developed by Arnold et al. (1993, 1998) to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time. The model is physically based and uses readily available inputs. It is computationally efficient and enables the user to study long-term impacts. SWAT is a continuous time model, i.e. long-term yield model. Specific models have contributed significantly to the development of SWAT: CREAMS (Knisel, 1980), GLEAMS (Leonard et al., 1987) and EPIC (Williams et al., 1984). We are using the version SWAT2000 with a ArcView Interface (cp, DiLuzio et al., 2001). SWAT2000 has extended capabilities and a number of improvements (cp, Neitsch et al., 2001). The model allows the simulation of numerous different physically processes including the land phase of the hydrological cycle as well as its routing phase.

Fig. 7 shows the simulation results for a 14-year period in the Weiße Elster watershed (ca. 5.000 km^2).

The model is actively supported by the USDA, which can be seen as an advantageous point. The development team monitors three forums: one for the stand-alone ArcView Interface, one for the interface

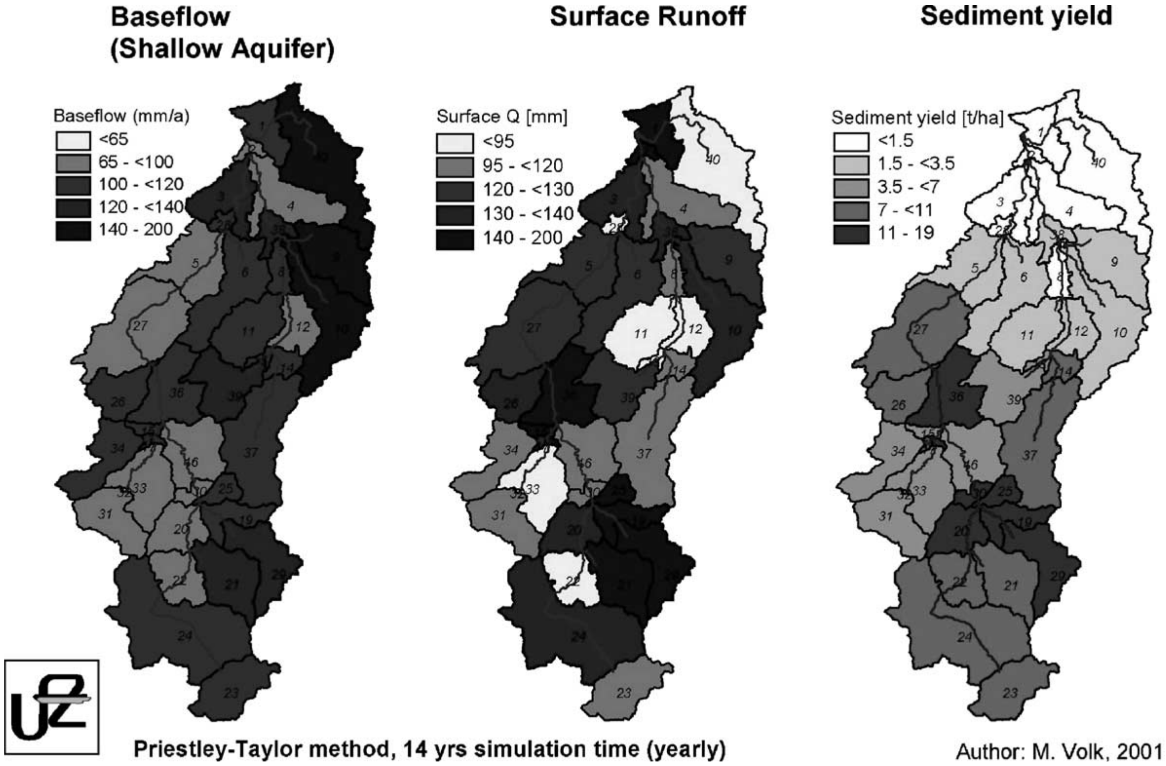


Fig. 7. Simulation of baseflow, surface runoff and sediment yield in the Weiße Elster watershed. The figure shows one of the results of our experiments with different methods for PET calculation. Such calculations are a base for integrated watershed management by enabling estimations of regional potentials and environmental risks which allow the derivation of related protection measures and land use systems.

in Better Assessment Science Integrating Point and Non-point Sources 3.0 (BASINS 3.0), and one for SWAT2000 allowing users to post questions. Together with the fact of free available software and source codes, this enables, for instance, modifications (cp, i.e. Fohrer et al., 2001) of the model, corrections of source code errors or the solution of interface problems. The modular and open construction of the model in addition to its coupling to the Desktop GIS ArcView forms an user-friendly instrument. Depending on the database a selection of algorithms for the calculations of different parameters is available. For instance, the potential evapotranspiration (PET) can be calculated with Priestley-Taylor-, Penman-Monteith- or Hargreaves-method. Due to the above-mentioned open construction, the user is also allowed to implement additional methods. Thus, we have been added the FAO Penman-Monteith method (cp, Wendling, 1995) which seems to represent the central European conditions quite better. The model is not designed to simulate detailed, single-event flood routing. According to the temporal resolution of one day, the model should not be applied to very small watersheds, where the (hydrological) processes possibly proceed very fast within this time step. But it is qualified to simulate hydrological processes and related material fluxes in meso-scale watersheds.

5. Conclusions

The applicability of models is determined by their use restriction to certain (temporal and spatial) scales due to their model “philosophy”. There is no scale-independent model similarly suited for different questions and controllable by a set of accurately defined parameters and input variables. The user has to examine carefully for what purpose the model should be applied. He has to answer the following questions:

- Does the model meet the requirements of the problems to be solved?
- Which processes are described by the model?
- Which algorithms are implemented?
- What about the temporal and spatial resolution of the model?
- Which input data are needed?
- Which output data will be produced?

After testing all models mentioned above, we have concluded SWAT to be the model best suited for the application on the meso-scale. It links the advantages of being an integrated model (e.g. describing the water balance as well as water coupled fluxes of matter) and being applicable in a wide spatial range (i.e. from small to very large watersheds). It should be used neither to simulate water and matter fluxes within a single soil column nor for single rainstorm events. If someone is interested in a more detailed description of the erosion process, we can recommend to use the E3D model, and for merely hydrological processes WaSiM-ETH seems to be the best.

6. Outlook

Based on the modelling results we try to carry out, for instance, a process-related landscape classification.

With respect to the above-mentioned theory, the following questions have to be answered:

- Are there only scale-specific processes?
- Do scale-independent respectively cross-scale processes also exist?
- If so, are they restricted to disastrous events like volcanic eruptions, insect outbreaks, floods, etc.?

We try to classify the landscape processes according to features as continuity, periodicity and intensity (Fig. 8). This classification will be the basis for a process-related landscape typification. In contrast to previous landscape classifications based on structural characteristics (e.g. grain size distribution, slope angle, aspect, mean annual air temperature, natural vegetation), this is a new approach. Thus, a large-scale survey about regions characterised by similar process events will be provided. According to Burak and Zepp (2000) the prime features deriving a large-scale landscape process texture are soil water dynamic, relief, climatological water balance and land use. Via land use manner and intensity human-induced impacts of matter dynamic will be determined.

Developing model systems that describe the great variety of (land use related) landscape balance processes is only one part of scientific research. On the other hand, we have to look for their practical application, which is related to policy and thus to socioeconomy. Multipurpose environmental analysis systems for use by regional, state and local agencies in performing watershed- and water-quality based studies are needed. BASINS is a system which includes several different existing models, developed to meet the needs of such agencies (EPA, 1998). On the other hand, modelling and simulation today becomes more and more interesting for environmental science because of highly available computing power on desktops and more efficient software support in terms of methodology. Toolboxes which provide the ability for non-programmers and scientists to build an optimised executable model

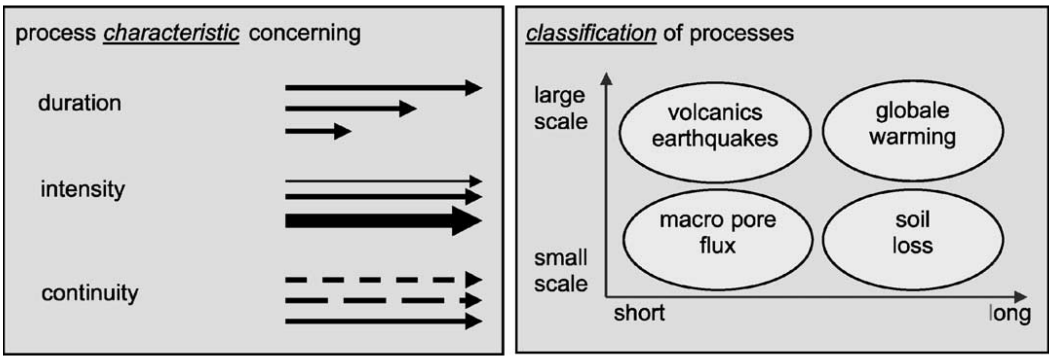


Fig. 8. Approaches aimed at a process-related landscape typification.

without any detailed knowledge in programming languages based on a set of modules implementing different kinds of modelling approaches. Thus, current research focuses also on the design and the implementation of such Object-Oriented Modelling Systems (OMS) to match these requirements (cp, David, 1997, 1999). Several types of environmental programs, e.g. the above-mentioned EC Water framework directive (EC, 2000), can benefit from the use and application of such integrated systems in various stages of environmental management planning and decision making.

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Abstract: A considerable reduction in the nutrient and pesticide inputs into the rivers and lakes of Germany is required in order to meet the "good ecological status" as demanded by the European Water Framework Directive (WFD). Sub-surface tile drainage systems are one of the main pathways for such diffuse nutrient and pesticide inputs. However, the simulation of water and matter fluxes under tile-drained land on the landscape scale is still problematic in many countries, mainly due to a lack of data about the existing drainage systems. The present study examines for the first time whether an existing method to calculate the usually unknown proportions of tile-drained areas could be transferred to a large river basin, for which minimal data about drained areas is available. The study area was the Saale river basin (24,000 km²) in central Germany, with a broad variety of soils and site characteristics. The share of tile-drained areas in the Saale river basin was calculated to be 11% of the agricultural area. Apart from that, the calculated proportion of tile-drained areas corresponded satisfactory with the statistical data of the meliorated areas of the former German Democratic Republic. The successful application of the promising method is considered as an important step towards the calculation of the proportion of tile-drained areas for the whole Germany and Europe.

Quantifying the proportion of tile-drained land in large river basins as a contribution to modelling water and nutrient fluxes

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Abstract

A considerable reduction in the nutrient and pesticide inputs into the rivers and lakes of Germany is required in order to meet the "good ecological status" as demanded by the European Water Framework Directive (WFD). Sub-surface tile drainage systems are one of the main pathways for such diffuse nutrient and pesticide inputs. However, the simulation of water and matter fluxes under tile-drained land on the landscape scale is still problematic in many countries, mainly due to a lack of data about the existing drainage systems. The present study examines for the first time whether an existing method to calculate the usually unknown proportions of tile-drained areas could be transferred to a large river basin, for which minimal data about drained areas is available. The study area was the Saale river basin (24,000 km²) in central Germany, with a broad variety of soils and site characteristics. The share of tile-drained areas in the Saale river basin was calculated to be 11% of the agricultural area. Apart from that, the calculated proportion of tile-drained areas corresponded satisfactory with the statistical data of the meliorated areas of the former German Democratic Republic. The successful application of the promising method is considered as an

important step towards the calculation of the proportion of tile-drained areas for the whole Germany and Europe.

1 Introduction

Approximately 14% of the agricultural regions in Germany are characterized by tile drainage systems (Werner et al. 1991; Werner and Wodsak 1994) that are used to reduce poor drainage problems in crop fields. Interferences with the landscape water balance and agricultural contamination of the environment through these subsurface or tile drainage systems have been increasingly investigated during the recent decades (Baker et al., 1975; Logan et al., 1994; Du et al., 2006).

Nitrate-nitrogen (NO_3-N) is one of the main pollutants produced from tile drainage that passes into the rivers of these areas (Jaynes et al., 1999; Cambardella et al., 1999). The shortened nitrogen outflow via tile drainage systems, and the reduced denitrification capacity in drained soils, contribute to the high concentration of nitrogen drainage waters.. A reduction of nutrient inputs, especially nitrate, from tile-drained land is strongly recommended in order to achieve the environmental objectives of the European Water Framework Directive (WFD).

Considerable methodical deficits exist in the quantification of tile-drained land as a basis for quantification of drainage discharge on a range of scales, especially on the river basin scale (Haag and Kaupenjohann, 2001; Arabi et al., 2006). Hence, the discharge via tile drainage systems is one of the main missing links in large scale modelling, especially because databases for tile-drained land do not exist for western European countries. For Germany, some documentation about the location of tile drainage systems exists, but these are for smaller areas such as farms. For the Eastern part of the present day Germany that was the former German Democratic Republic (GDR), limited tile drainage data were held by the bodies responsible for the amelioration of soils and collective farms in the GDR. However, many documents got lost during the time of the German reunification, and the remaining data are stored at different places and at different authorities; consequently there is still no systematic documentation of the locations of tile drainage systems in Germany.

Data on tile drainages are urgently needed as basis for the simulation of the related fluxes of water, nitrogen, phosphorus, and pesticides that come along with the drainage water. Furthermore, information about the proportion of tile-drained land is essential for the development of measures to reduce diffuse nutrient inputs into the river systems.

A method to estimate the proportions of tile drained land for large river basins by means of soil characteristics was firstly developed by Behrendt et al. (2000). They choose representative areas in different landscapes in the former German Democratic Republic (GDR) with different soil conditions where they obtain the information whether an area is drained or not from maps of drained areas. Information about drained areas was digitized in these representative areas in order to obtain a complete data set of the location of drained soils. The proportion of the drained land was then determined for every aggregated form of soil type ("site type", German: "Standorttyp") of the "meso-scale soil map in agricultural areas" (MMK), which is the soil map of GDR that reflects the soil and site conditions. This proportion of drained areas for every "site type" was extrapolated to the entire area of the former GDR. Thus, differences in the proportion of tile-drained land could be detected for soil associations in the macro-scale.

Behrendt et al. (2000) used the most aggregated "site type" for the extrapolation of the proportion of tile-drained areas to larger regions, e.g. for the whole of Eastern Germany. However, this method does not include the comprehensive hydromorphical characteristics described in the "regional site type" (that contains 15 instead of three classes of hydromorphology). Their method is therefore mainly applicable in aggregated analyses. The inclusion of the detailed "regional site type" enables differentiated analyses of the geographical distribution of the tile-drained areas, and focuses on the hydromorphical soil conditions, which is the most relevant information concerning the necessity to drain a soil. An additional limitation of Behrendt et al. (2000) is that most data sets were for drained areas for the soils of the northern part of the former GDR.

The method developed by Behrendt et al. (2000) was modified by Hirt et al. (2005a,b) by using: a) a higher disaggregation level of the soil and site conditions (regional site type of the MMK), and b) existing data sets of drained areas of the southern part of the GDR. This extrapolation of the method was demonstrated for the Mulde river basin (2,700 km^2) in Central Germany (Hirt et al. 2005 a,b). Because information about tile-drained land was already available for approximately 80% of the river basin area, the extrapolation was only performed to obtain a complete data set of the proportion of drained areas in the Middle Mulde river basin.

Following from this, we wanted to investigate how far a transfer of the method to a greater river basin, where no data is available, would be possible, and which inaccuracies would be involved. The transfer of the promising method to a larger river basin is now proven for the first time, and helps to improve the knowledge and database about tile-drained areas, for example for modelling purposes. Therefore, the objective of the present study was to investigate the transferability of the Hirt et al. (2005 a,b) method to the Saale river basin, which covers an area of 23,719 km^2. For the Saale river basin, which is adjacent to the Mulde river basin, only minimal data about drained areas were available. The plausibility of the results are demonstrated using the digital soil map of Germany BÜK-1000 (in German ***B**oden**ü**bersichts**k**arte*, scale 1:1,000,000; Hartwich et al., 1995) and a comparison with existing statistical data of the former GDR on the district level is performed.

2. Study area: Saale River Basin

The Saale river basin (23,719 km²) lies in central Germany, predominantly in the state of Thuringia, and extending to Saxony and Saxony-Anhalt (Figure 1); the river runs for some 320 km.

The basin can be subdivided into three sub-regions: the Pleistocene lowlands, the loess sub-region, and the mountainous sub-region. The geology is characterised by shale bedrock in the mid-mountain range, sandstone in the forelands, and Karstic limestone at the Thuringian border. Precipitation varies from 500 mm/yr in the dry loess areas to 1200 mm/yr in the forested mountain regions.

The Saale river basin is adjacent to the smaller Mulde river basin (Figure 1), which was investigated by Hirt (2005a,b). An area with pseudogleys and gleys extends from the Mulde to the Saale river basin. Although several soil types are the same or comparable in the two basins, the larger Saale basin has more soil types covering much larger areas than does the Mulde basin, including large areas on black earth in the former.

Please insert Figure 1

Our study area does not include the parts of the Saale river basin that lie in the state of Bavaria and Lower Saxony of the Saale River Basin, because they are very small, and because they were part of the former Western Germany and thus excluded from the East German soil map “meso-scale soil map in agricultural areas (MMK)” (see below).

3. Database and data processing

The data used to determine the proportion of tile-drained areas were derived from three sources: a) maps showing the location of tile drained land, (1:10,000 - 1:50,000), b) the meso-scale agricultural soil map (the MMK) (1:100,000), and c) the unified biotope and land use type maps of the German States of Saxony, Saxony-Anhalt and Thuringia (1:10,000). Each of these is now described in detail.

3.1 Maps showing the location of tile drained land (1:10,000 - 1:50,000)

These maps were available for the Middle Mulde river basin (Hirt 2005a), the sub-basin of the Weida (sub-basin in the southern Saale river basin), and the drained areas (Behrendt et al. 2000) (Figure 2). New information about tile drained areas in regions with black earth soils were obtained from local authorities, and then digitalized to complete information about tile drainages for the different soil regions in the Saale river basin. Information about tile-drained areas is now available for an total area of 14,100 km², in which 2,300 km² tile-drained land extists.

The maps of tile drained land were mainly prepared between 1960 and 1989 as a basis for soil amelioration and collective farm management in the former German Democratic Republic (GDR) (1949 – 1990). Since many maps were lost during the course of restructuring following German reunification, this data set is incomplete.

In order to digitise the topographical positions of tile drained areas in the larger river basins, for representative areas maps about drained areas were first collected and digitalized. For these areas the tile-drained areas and their position are fully known. These tile drained areas are relevant because they are representative for specific soil and landscape characteristics that exist in the larger river basins. Most of these maps with tile-drained information are based on a scale of 1:10,000 or 1:25,000. A topographical map at the scale of 1:50,000 was used as a basis for digitisation for the larger river basin regions. Some small tile-drained areas that were classified as 'non-arable' (due to intersection errors (GIS-operation)) were eliminated by an overlay with land-use data.

Please insert Figure 2

3.2 Meso-scale agricultural soil map of the former GDR (1:100,000), (German abbreviation: MMK, Lieberoth, 1982).

This map is a major source of data for determining the proportions of tile drained land. It contains the „regional site type" (RST), which is determined by a characteristic mosaic of substrate, soil moisture and inclination conditions (Lieberoth*,* 1982). The RST includes information about the hydromorphic characteristics (named hydromorphic area type, Table 1), and soils are differentiated according to whether they are influenced by leaching, groundwater, or stagnant water. The part of the MMK covering the part of the Saale river basin included 561 Regional Site Types that we had to consider.

Please insert Table 1:

3.3 Unified biotope and land use type maps of the German States of Saxony, Saxony-Anhalt and Thuringia (1:10,000).

As the MMK does not record the differentiation between arable land and grassland, this information was obtained by the analysis of the biotope and land use type maps of the German States of Saxony, Saxony-Anhalt and Thuringia. It is derived from the interpretation of colour infrared aerial images on the basis of the "list of biotope and land use mapping" from 1992 (scale 1:10,000).

4. Method

To determine the proportion of tile-drained areas in a river basin, it is assumed that these areas are substantially influenced by the local soil and site characteristics. The local soil and site conditions are described in the "regional site types" (RST) of the MMK.

In order to determine the proportions of tile-drained land throughout the study area, the data set of tile-drained areas in Eastern Germany (described in section 3 above), the soil map MMK, and the data from biotope and land use type map were overlaid using the geographic information system (GIS) ArcInfo® (Figure 3). The MMK provided the necessary information on the soil and site characteristics. The overlay with the biotope and land use type map enabled us to differentiate between grassland and arable land. The GIS is used to calculate the percentage of tile-drained land for each regional site type for all areas with digitalised drainage data. The results of the calculations were subsequently extrapolated to the regional soil types of the Saale river basin (Behrendt et al., 2000; Hirt et al., 2003). For SRT, where no information is given in the data set of tile-drained areas in Eastern Germany, a mean value for the ST with the same substrate (designated by the capital letters contained in the regional site type, e.g. D for diluvial soil) and the same hydromorpholocial conditions (designated by the small letters, e.g. "a" for determined by leachate) are calculated (e.g. a mean value of: Lö4a, Lö3a and Lö5a). With this procedure, the share of drained areas for every regional soil types of the basin is generated with reasonable effort.

Please insert Figure 3

5. Results and Discussion

The calculated share of tile-drained areas in the Saale River Basin is 11% of the agricultural area (Figure 4). The southern and western parts of the river basin show a higher share of tile drained areas with partly over 20%, while the northern part shows lower values (mainly > 5%).

The digital soil map of Germany BÜK-1000 can be used to check the plausibility of the calculated drained areas, according to their degree of stagnant water and groundwater (Figure 5). With an increasing portion of stagnant water, and/or groundwater, there is an increasing need for drainage. Our study shows that areas with gleys and pseudogleys (which are typical soils in areas with stagnant water and high groundwater level) as the dominant soil types (southern and western parts of the basin) are highly tile drained, with 10-40% of the land area requiring tile drains. In contrast, in areas with black earths and brown earths that are characterised by seeping water (northern and central basin regions), only 0 – 5% of the area has tile drains.

Please insert Figure 4

Please insert Figure 5

The distribution of tile-drained areas in relation to the hydromophic conditions (hydromorphic area type of the MMK, Table 1) is shown in Figure 6. The basic trend is that soils affected by groundwater and stagnant water have a higher share of drainage areas (12 - 30%; exception: GS1), whereas the leachate soils have lower proportions (9 - 10%). The highest share of drained areas is found in the soils dominated by groundwater and stagnant water (GS2 and GS3). The type GS1, which could be determined by leachate up to 60%, has a relative low share of drained areas, and appears to have a low need for drainage. The soils determined only by groundwater (G1 to G3) also have relatively high values from 15 to 22%. The soils influenced by groundwater and leachate have a higher share of tile drained areas as two of those determined only by groundwater (G2, G3). A possible reason for this is that the soils determined by groundwater (G1 – G3) mainly occur in valleys, which are mainly drained by ditches. The soils influenced by stagnant water and leachate show values from 10% (SN1) and 15% (SN2), which seems to be feasible due to the higher share of leachate of SN1.

Stagnant soils show a proportion of tile drained areas of 12 – 22%. The lower proportion of drainage in soils which are strongly influenced by stagnant water (S3:12%) is remarkable in comparison to those which are weakly influenced by stagnant water (SN1: 15%, SN2:22%). These results show that analysing the calculated results in detail does not lead in every case to feasible results. The reasons for this could be: i) the method needs more information about drained land in order to produce more plausible results, ii) inaccuracies in the MMK, and iii) insufficient consideration of site factors in drainage-project planning. It could be expected, that the observed discrepancies are caused by a combination of these three potential reasons.

Please insert Figure 6

Regarding the representativeness of the calculated share of tile-drained areas, it is assumed that a minimum size of 1 km² for each RST (n = 561) of the representative areas is necessary to enable the extrapolation to areas for which the proportion of tile drainage area is not known. For the Saale river basin, the extrapolation can be claimed for 99.8 % of the area. The remaining 0.2 % of the study area is not significant for the overall results.

The calculated proportion of tile-drained land was compared with the statistical data of drained areas in the districts of the former GDR (Statistisches Amt der DDR, 1990; Table 2). Three districts, at least 50% of which lay within the river basin, were analysed; for district areas outside the basin, we assumed the same distribution of the tile-drained areas as for the areas within the river basin. A satisfactory correspondence between the calculated tile drained areas and the statistical data was found for two of the largest districts of the Saale river basin: 99% of the drained areas (statistical data) were calculated for Erfurt, and 68% for Gera. The correspondence for the district of Halle was poor (154%).

Please insert table 2

6. Conclusions

Data of the proportion of tile-drained areas are urgently needed for various aspects of landscape modelling and assessment. The method we present can be used to calculate proportions of tile-drained land in large river basins by examining the share of drained areas for every soil type in representative areas, and subsequently extrapolating the data.

One of our next objectives will be the generation of a comprehensive database on the proportions of tile drained land for every soil type (here called: ”regional site types”) of the meso-scale soil map in agricultural areas (MMK) for the former East Germany (GDR). The information was gained from a broad variety of representative areas. On basis of this dataset, the share of drained areas can be extrapolated to the whole area of the former East Germany.

To extrapolate the share of drained areas to western Germany, other soil maps are necessary, because the MMK is only available for the east of the now unified country. Therefore the German soil maps (BÜK 1:200,000 and 1:1,000,000) or the European soil maps have to be used. The share of drained areas in the representative areas then have to be intersected with these soil maps, and extrapolated to areas where no tile drainage data are available. The extrapolation has to be proven with data of tiled drained areas in test areas, because of the different soil assessment methods and data sources used for these soil maps (Behrens and Scholten, 2006). Then the share of drained areas can be calculated for catchments in western Germany, for the whole of Germany, and other countries using the European soil map. This process will provide an important basis allowing the effects of tile drained areas to be taken into account in modelling of water and matter fluxes and landscape assessment, which will in turn be used as a base for implementing measures to reduce nutrient and pesticide input into rivers.

Acknowlegdements

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Figures and tables

Figure 1: Location of the Saale and Mulde River Basins in Germany.

Figure 2: Procedure for determining the proportion of tile drainage area in regional site types (Hirt, 2003)

Figure 3: Position of the digitalized drained areas in the area of the former East Germany (GDR)

Figure 4: Soil types in the Saale catchment (data source: soil map of the FRG 1:1,000,000; BGR)

Figure 5: Calculated share of tile drainage areas for the Saale catchment

Figure 6: Proportions of drainage area (%) in the River Saale catchment classified by hydromorphic area type

Table 1: Criteria used to derive hydromorphic area types (Schmidt and Diemann, 1991: 32)

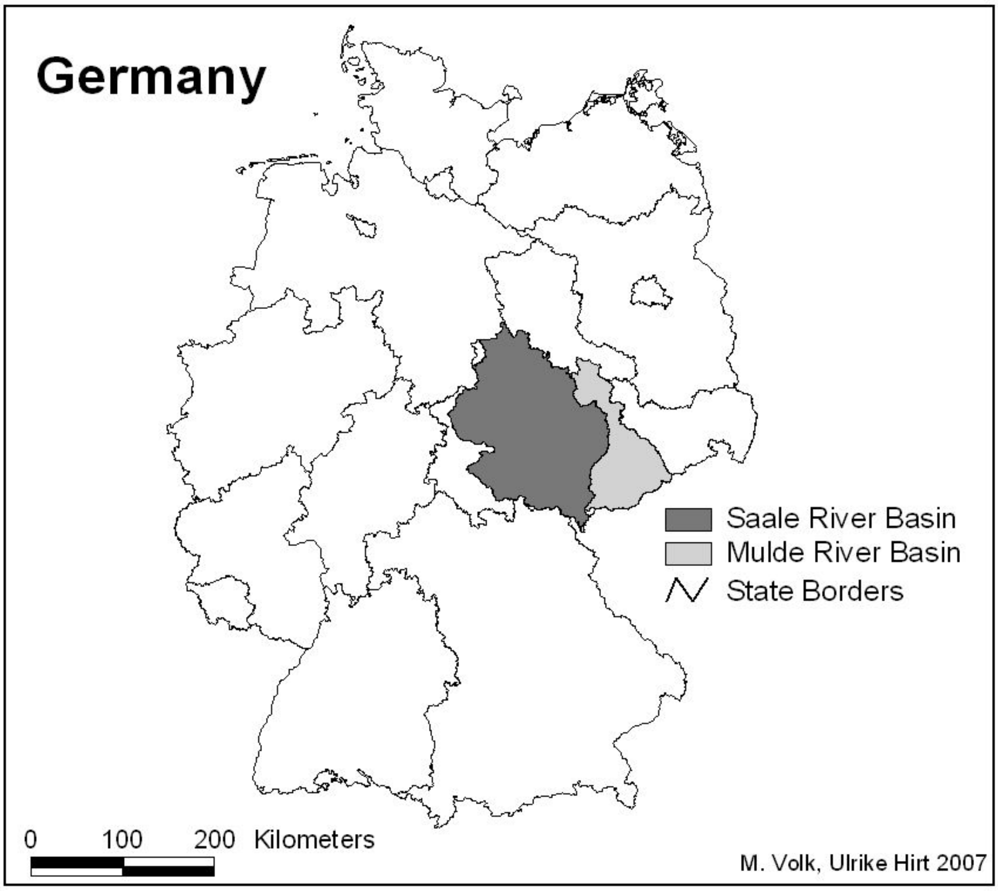
Table 2: Comparison of drained areas derived from calculations and statistical data

Tab. 1

Hydromorphic area type	Symbol	Proportion of areas [%]		
		Leachate	Stagnant water	Ground-water
Totally determined by leachate	N1	>80	-	-
Not completely determined by leachate	N2	61-80	≤20	≤20
Weakly influenced by stagnant water	SN1	21-60	21-40	≤20
Moderately influenced by stagnant water	SN2	≤40	41-60	≤20
Moderately determined by stagnant water	S1	≤20	>40 semi-hydro-morphic	≤20
Highly determined by stagnant water	S2	≤20	>40 completely hydro-morphic	≤20
Extremely determined by stagnant water	S3	-	>80 completely hydro-morphic	-
Determined by stagnant water, groundwater and leachate	GS1	21-60	21-40	21-40
Determined by stagnant water with groundwater	GS2	≤20	41-60	21-40
Determined by groundwater with stagnant water	GS3	≤20	21-40	41-60
Weakly influenced by groundwater	GN1	21-60	≤20	21-40
Moderately influenced by groundwater	GN2	≤40	≤20	41-60
Moderately determined by groundwater	G1	≤20	≤20	>60 (water zone G1)
Highly determined by groundwater	G2	≤20	≤20	>60 (water zone G2)
Extremely determined by groundwater	G3	≤20	≤20	>60 (water zone G3)

Tab. 3

Region	Drained land, calculated (ha)	Drained land, statistical data (ha)	Correspondence (%)
Erfurt	45646.3	45692.5	99.9
Halle	34119.3	22180.0	153.8
Gera	34683.8	51036.4	67.9



Germany
Saale River Basin
Mulde River Basin
State Borders
0
100
200
Kilometers
M. Volk, Ulrike Hirt 2007

Fig. 1

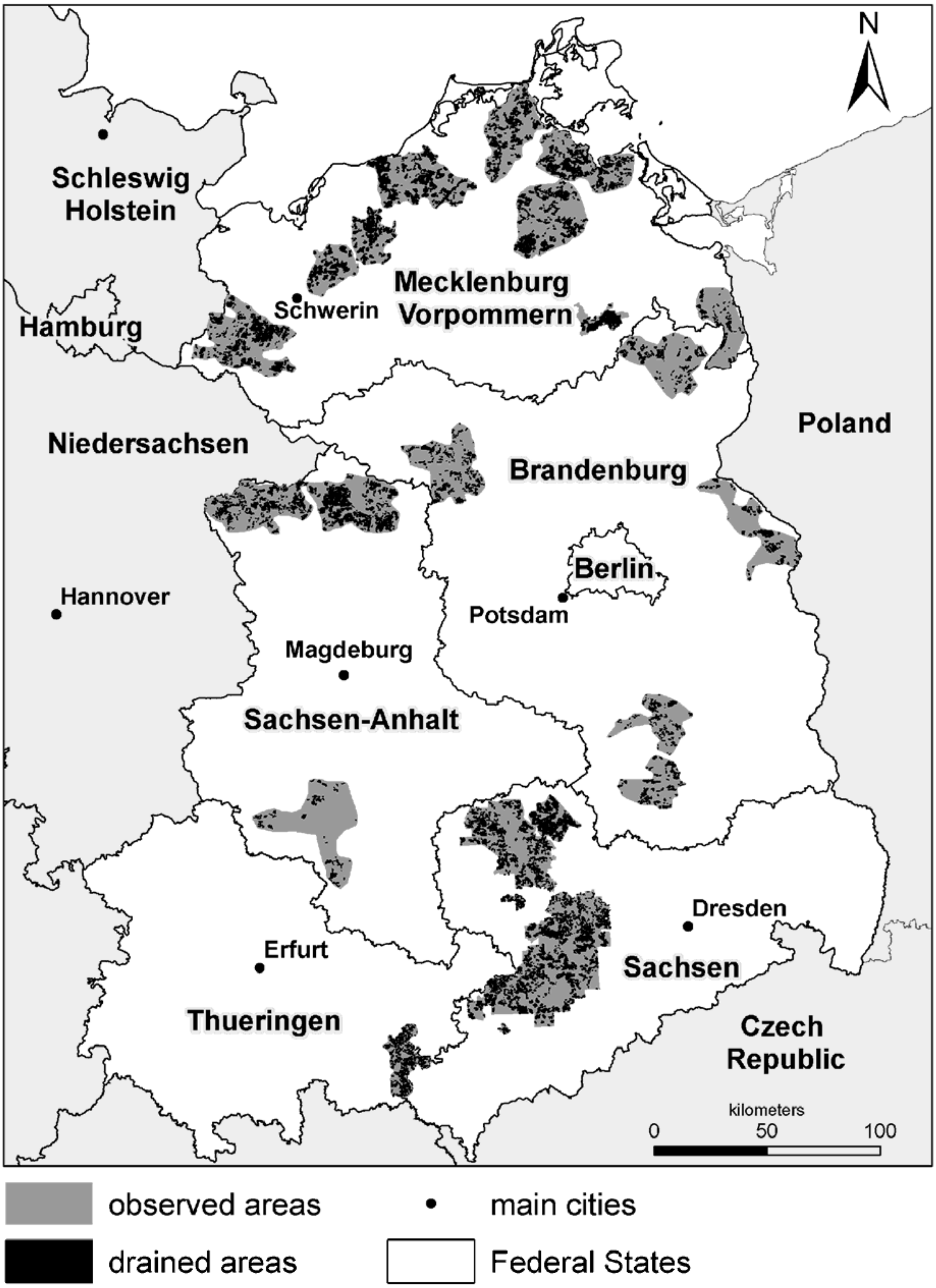
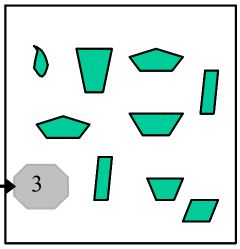

N
Schleswig Holstein
Hamburg
Schwerin
Mecklenburg Vorpommern
Poland
Niedersachsen
Brandenburg
Berlin
Potsdam
Hannover
Magdeburg
Sachsen-Anhalt
Dresden
Erfurt
Sachsen
Thueringen
Czech Republic
kilometers
0
50
100
observed areas
main cities
drained areas
Federal States

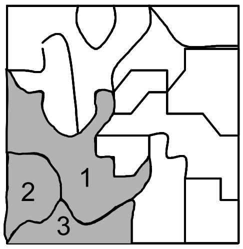
Fig. 2

Digital data set of the proportion of subsurface drainage areas

ID	drained area
1	1
2	2
3	3
4	4
5	5



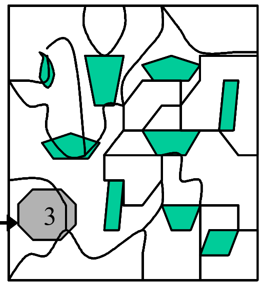
Meso-scale soil mapping
(Mittelmaßstäbige Landwirtschaftliche Standortkartierung, MMK)



ID	RST
1	Lö3a3-09
2	Lö4b3-05
3	Lö4c2-05
4	Lö5b2-05
5	Lö5b3-09

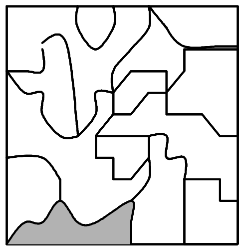
Intersection of both datasets

ID	RST	drained area
1	Lö3a3-09	0
2	Lö3a3-09	3
3	Lö4b3-05	3
4	Lö4b3-05	0
5	Lö4c2-05	3
6	Lö5b2-05	0
7	Lö5b3-09	2
8	Lö5b3-09	0



Calculation of the proportion of drained area on basis of the regional site type (“Standortregionaltyp”)

RST	%
Lö3a3-09	5,9
Lö4b3-05	35
Lö4c2-05	16,2
Lö5b2-05	31,3
Lö5b3-09	21,8



Extrapolation of the proportion of drained area to the entire area

ID	RST	%
1	Lö3a3-09	5,9
2	Lö3a3-09	5,9
3	Lö4b3-05	35
4	Lö4b3-05	35
5	Lö4c2-05	16,2
6	Lö5b2-05	16,2
7	Lö5b3-09	21,8
8	Lö5b3-09	21,8

Presentation

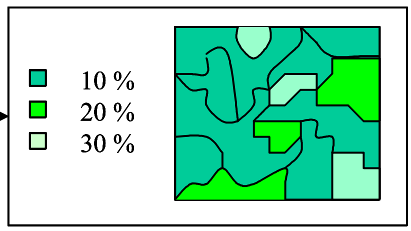


Fig. 3

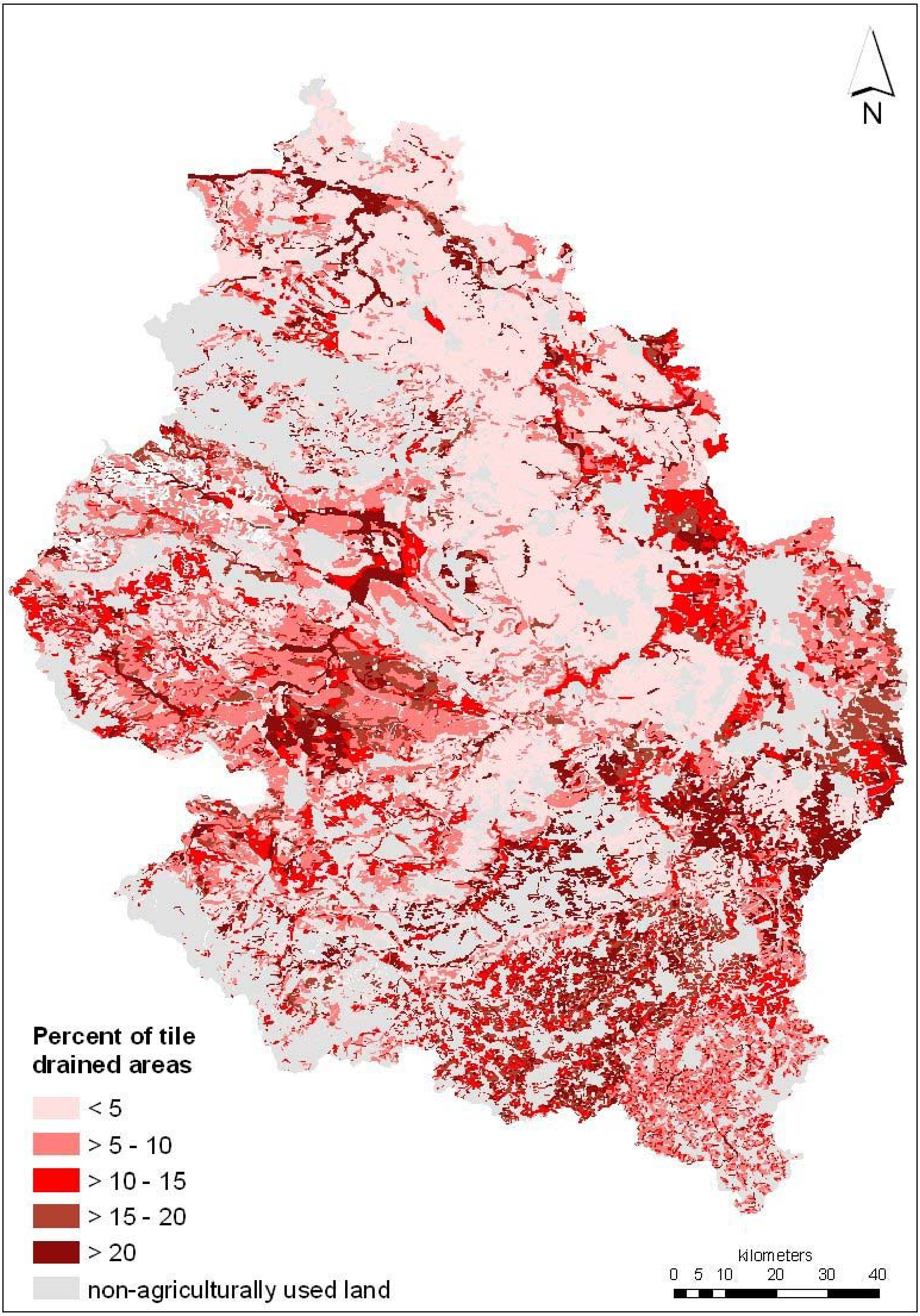

N
Percent of tile drained areas
< 5
> 5 - 10
> 10 - 15
> 15 - 20
> 20
non-agriculturally used land
kilometers
0 5 10 20 30 40

Fig. 4

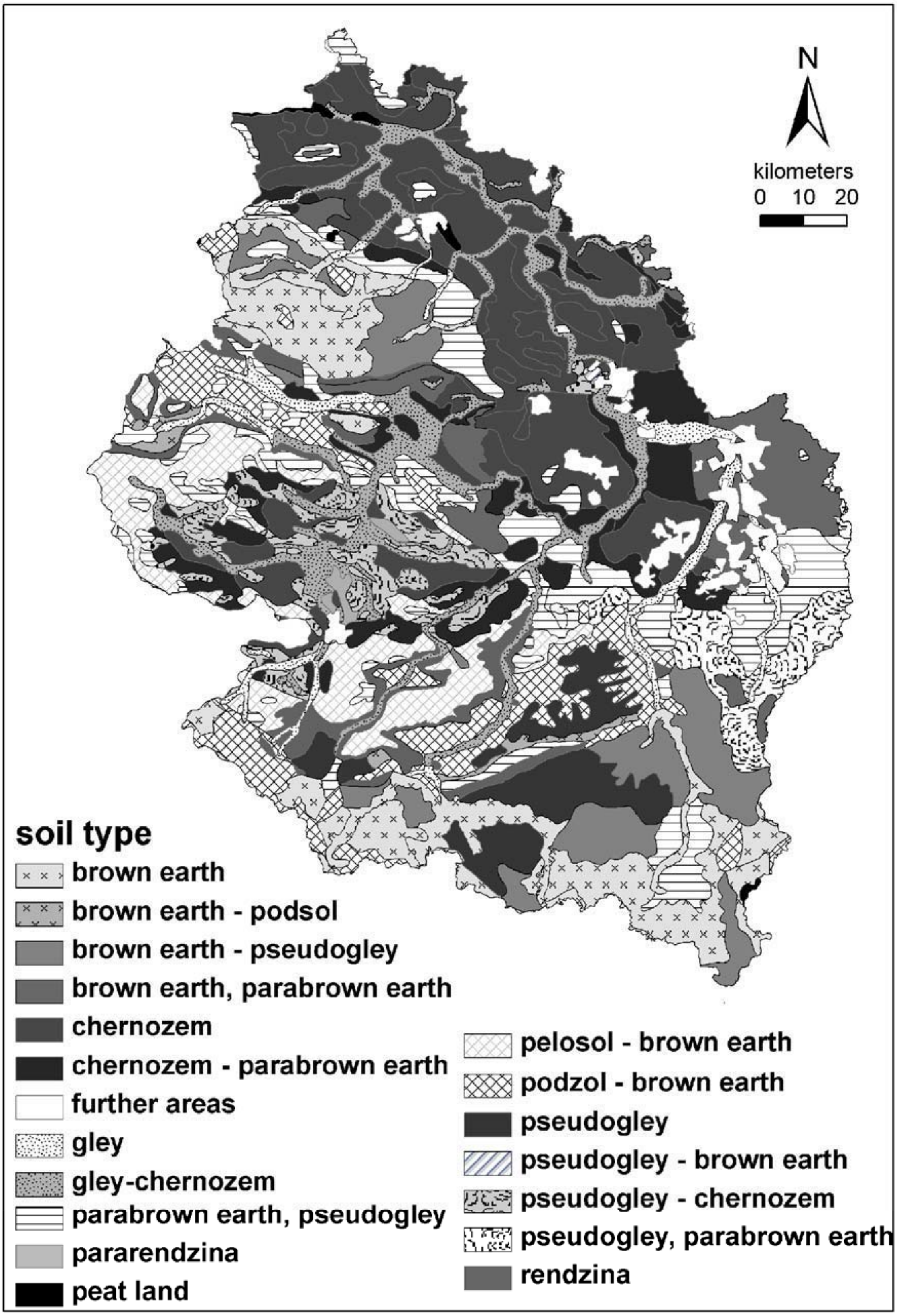
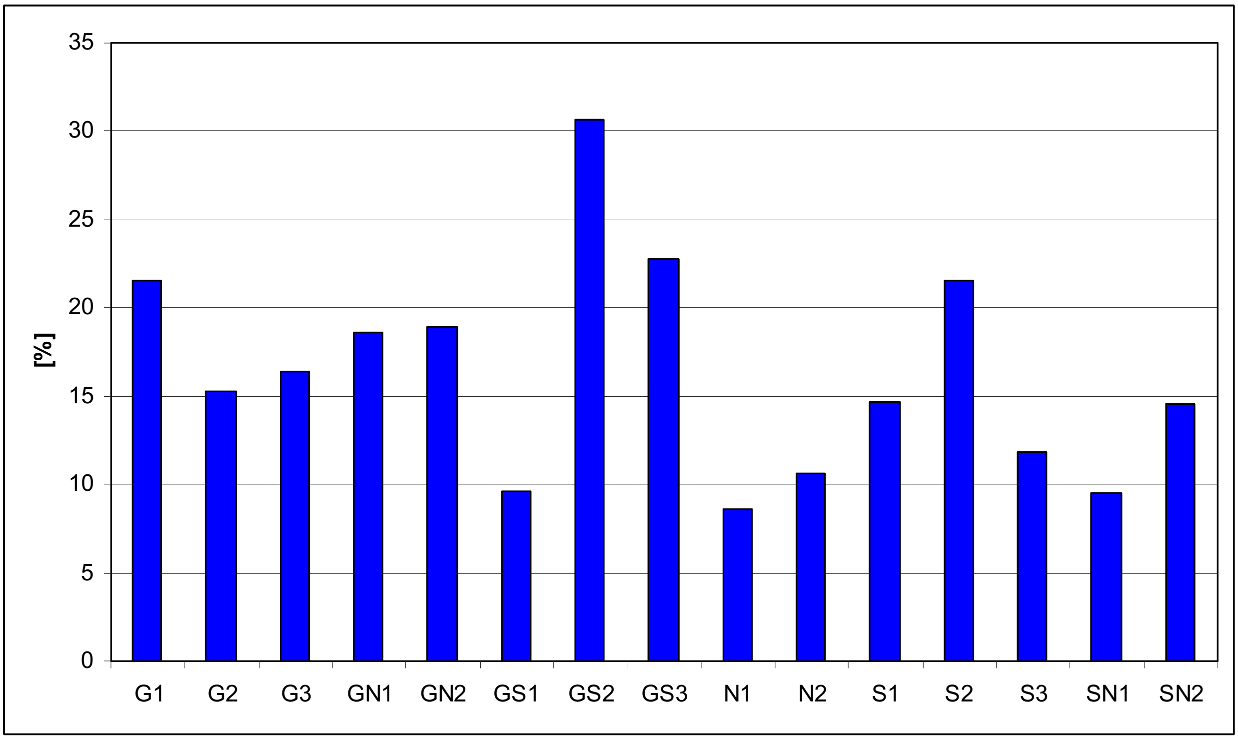

N
kilometers
0 10 20
soil type
brown earth
brown earth - podsol
brown earth - pseudogley
brown earth, parabrown earth
chernozem
chernozem - parabrown earth
further areas
gley
gley-chernozem
parabrown earth, pseudogley
pararendzina
peat land
pelosol - brown earth
podzol - brown earth
pseudogley
pseudogley - brown earth
pseudogley - chernozem
pseudogley, parabrown earth
rendzina

Fig. 5



35
30
25
20
15
10
5
0
[%]
G1
G2
G3
GN1
GN2
GS1
GS2
GS3
N1
N2
S1
S2
S3
SN1
SN2

Fig. 6

A2.8

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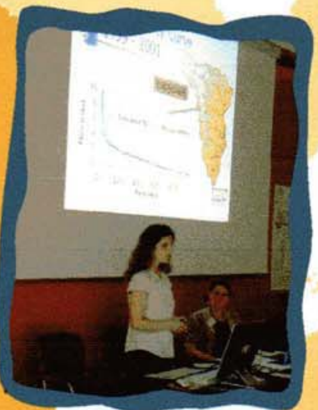

TWRI Technical Report 266

V. Model Integration

The Model Concept in the Project FLUMAGIS: Scales, Simulation and Integration

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Abstract

In order to reach the environmental targets of the European Community (EC) water framework directive on different scales, a concept for the scale-specific simulation of water-bound fluxes in the FLUMAGIS project is presented. According to the interdisciplinary relevance, scale levels have been defined which comprises the micro-, meso- and macroscale. Thus, for the description of the water balance and matter fluxes within the landscape the models NASIM (microscale), ArcEGMO (micro- to macroscale), ABIMO and SWAT (meso- to macroscale) have been selected. The usage of all these models aims to examine the transferability and applicability of the simulation results to the next higher or lower scale, as well as holding the system open for other models. The scale transition and thus the information exchange between the models are scale-specific depending on the application and compilation of existing parameters and indicators. During the first working phase, the behaviour and sensitivity of the models on different frame conditions and factors is checked out and possibly adapted by using artificial areas. Despite the different model concepts and the temporal differentiation of the input parameters, the results show an acceptable accordance. The simulation results of SWAT and ArcEGMO mostly show only small differences, whereas the results of ABIMO and NASIM show greater differences. In general, these differences are caused by different temporal resolutions and parametrization options of the models. As a first step towards the consideration of the whole area, the ABIMO conceptual model was found suitable for estimating the mean runoff for the Upper Ems River Basin.
KEYWORDS. EC water framework directive, scale-specific simulation, parameter and indicator system, artificial areas.

Introduction

The Water Framework Directive (WFD, established in 2000 (EC 2000) forms a framework for the measures of the European Community (EC) in the field of water policy. The directive defines the environmental targets for surface and groundwater for the European Union. Management plans for whole river basins shall serve as the main instrument for the implementation of the directive. Additionally, the participation of all concerned authorities and surveys is required (EC 2000). The FLUMAGIS project[2] (www.flumagis.de) contributes to an improved implementation of participation approaches on the management processes. The project is focused on the development of an interactive tool for the assessment and three-dimensional visualization of the hydrological and ecological conditions in river basins. The simulation of the impact of land use on the water balance fluxes in the landscape is an essential basis for the establishment of a knowledge base for this system. A lot of problems exist regarding the implementation of the directive. Examples are deficiencies in the spatial differentiation of environmental quality targets, planning and measure levels, and the lack of a common and homogeneous database for several scale levels. However, planning for water protection measures requires the designation of further scale levels due to the different regional conditions. Thus, the open scale problems and the regional relationship of management measures made it necessary for us to develop a concept for the scale-specific simulation of the water balance and the matter fluxes.

Study Areas

The studies are carried out in the mainly flat Upper Ems river basin in Northwestern Germany, which covers an area of 3,740 km^2. The hydrological processes in the Upper Ems basin are characterized by increasing precipitation amounts from the Northwest and Central basin (700 mm/a) to the Southeast (1.200 mm/a) but also by the widespread permeable sandy soils. The runoff dynamics are closely related to precipitation patterns. The Ems River has its sources at the foothills of the Teutoburger Wald mountains (altitudes reach only about 360 m above sea level or asl) on the Eastern border of the Upper Ems river basin. The Ems River flows through the North German Lowlands to the North Sea. Detailed investigations will be carried out in three subbasins. The selected subbasins cover areas between 160 km^2 and 350 km^2. The Ems floodplain, between Telgte and Greven (13.5 km^2), represents another study area for investigations with a high spatial-temporal resolution. Figure 1 shows the location of the study areas and its land use pattern (green – forests, red – settlements, light yellow – agricultural land use).

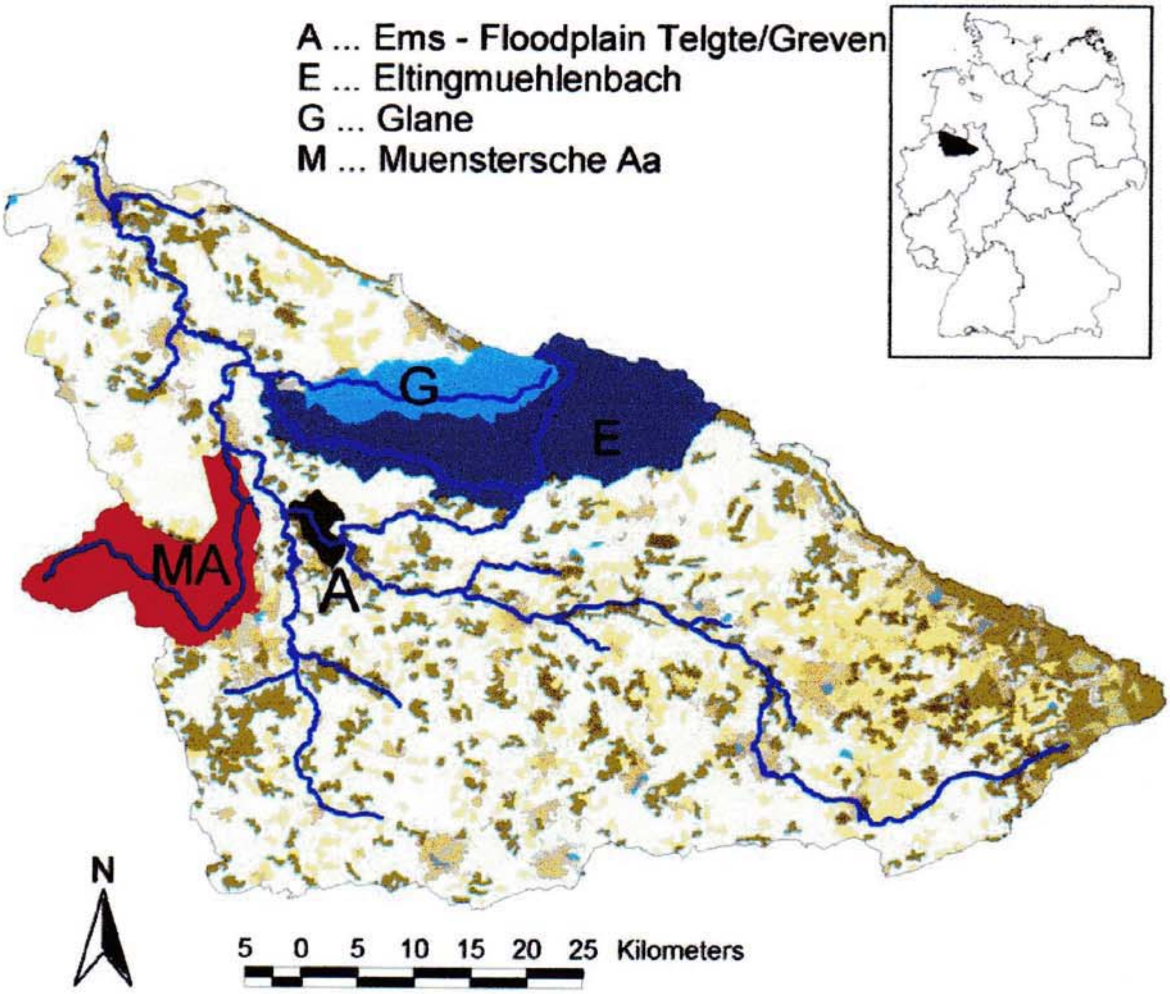

Figure 1. Location of the study areas in Germany and land use pattern of the region. Agriculture (bright colour) is the dominating land use type.

The River Basin is dominated by agriculture and is one of the most intensively used agrarian regions in Europe. Agricultural land (arable land, pasture and heterogeneous agricultural use) covers 81% of the total area, followed by forests (11%) and settlements, industry and infrastructure (9%). Figure 2 gives more detailed information about the land use of the area.

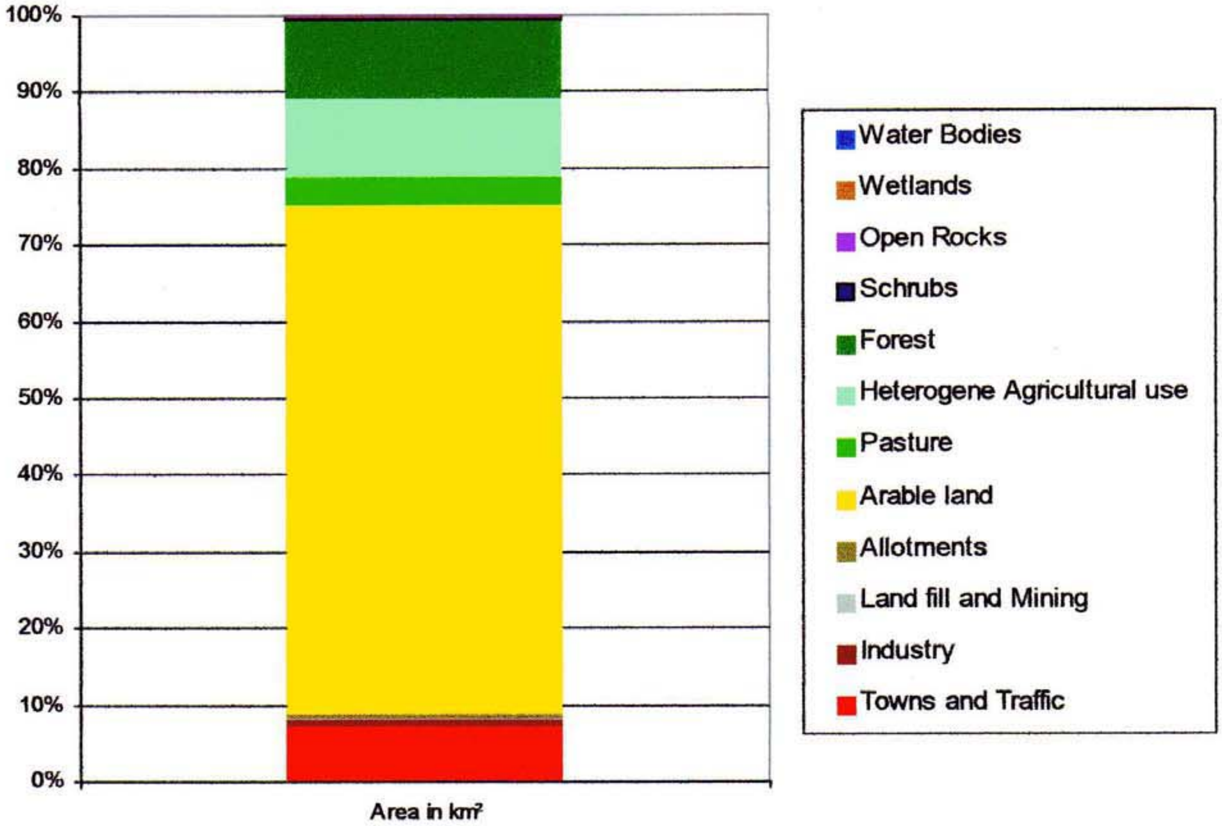

Figure 2. Proportion of the land use types on the area of the Upper Ems River Basin in the year 1997 (CORINE Land Cover)

A detailed analysis shows that arable land dominates the land use structure with approximately 65%. Such a high percentage of arable land in an area with mostly poor soils is caused by the cultivation of forage crops for stock farming. Several environmental problems and land use conflicts are the consequence.

A concept for Scale-Specific modelling

The main objective of FLUMAGIS is the development of an assessment and visualization tool for integrated river basin management. Specific scale definitions of the WFD (report scale, measure scale, etc.) require the use of scale-specific tools for the investigation and visualization of the ecological situation in river basins and the effects of water protection measures. The use of scale-specific methods and tools must be based on a data transfer between these tools. The transfer could be realized by specific indicators resulting from a simulation process for a certain scale level or as input data for another model simulation on another scale.

The scale transition and information exchange between the models should be realized by a selection of existing indicators at the outlets – under consideration of variability and uncertainty (comparison of aggregation levels). An essential basis is the development of a regional- and scale-specific indicator system for the investigation of the water balance and matter fluxes (Schmidt et al. 2003). Thus, existing indicator systems are examined on their suitability for ecological assessments due to the requirements of the WFD. The so-called Indicators of

Hydrological Alteration (IHA) have been proven suitable for the characterization of the runoff dynamics in surface waters (Richter et al. 1996, Ehlert & Van den Boom 1999, Zepp 2002). Further hydrological indicators result from the BWK-instructions (BWK 2001). They play an important role for rivers in urban areas with special impairments by sewage and stormwater discharge. Information which is extracted from existing concepts has to fulfill several requirements to meet the aims of the WFD. This is caused by the required participation, the considered scale levels, and the model selection and application. After passing through this filter, the resulting indicator pool will be examined according to how well it represents a certain scale level (Table 1).

Table 1. Scale matrix for the identification of the scale-specific representiveness of indicators.

Indicator Scale level	**Indicator A**	**Indicator B**	**Indicator C**
MICROSCALE 1:5,000-1,000	+	-	+
MESOSCALE 1:25,000-10,000	+	+	-
MACROSCALE 1:500,000	+	-	-

The indicators form the basis for scale transition between the models and help to prove the quality of the simulation results. The procedure is a combination of "top down" and 'bottom up" approaches (Volk & Steinhardt 2000) and the investigation of processes and conditions on different levels. As a result, knowledge about the efficiency of detailed management measures (microscale) for the whole area is obtained. The macroscale investigations enable a spatial differentiation of the landscape and a designation of regions with environmental risks and land use conflicts. More detailed model systems can be focused on these areas to get exact information about water quality and quantity and the water-bound material fluxes. A "calculation knot" (outlet) forms the central interface of the scale transition. This technique presumes that the applied model systems generate mutually comparable results about the concerned parameters. This is especially necessary regarding the use of quantitative indicators for the investigation of hydrologic and environmental conditions. The method aims to examine the transferability and applicability of the simulation results to the next higher or lower scale level, as well as holding the systems open for other model systems.

Simulation Models and Artificial Areas

According to the textual and scale-specific requirements of the project, several conceptual and physically based simulation models are selected: NASIM 3.10 (Hydrotec 2001), ArcEGMO 2.3 (Pfuetzner et al. 2001), SWAT 2000 (Arnold et al. 1993, 1998, Neitsch et al. 2001) and ABIMO 2.1 (Glugla & Pfuertig 1997, Rachimov 1996). The selection was determined by the scale-specific applicability of the models, the data availability, and the experiences of the working team. The specific approach, and different structures of the simulation models and the spatial-temporal resolution and parametrization requires comparing calculations to detect potential variations (Table 2). At first, a relative comparison of the models with different scale-specifics is carried out on the basis of artificial areas with variable differentiation levels. This method provides information about the model specifics and variations of the results. During the first phase of the relative model comparison, the artificial areas are distinguished by a few homogeneous parameters. This is advantageous because the resulting datasets are concise and manageable and the calculation times are reduced. The spectrum of the properties for each parameter is determined by the given landscape characteristics of the study area (the Ems River Basin). Homogeneous relief and soil parameters (one layer) have been selected, and land use is represented by different scenarios (Table 2). The simple artificial catchment covers an area of 10 km². The catchment is divided by a river into two similar slopes (2.5 km x 2 km).

Table 2. Structure of the artificial area.

Soil	Land use	Relief	Climate
• B1 – Gleyic posdol from sandy river sediments • B2 – Loess soil	• Forest (deciduous) • Pasture • Arable land • Surface sealing	• 5° Slope angle • 1% river channel incline	• Precipitation and PET data: 1970 – 1993

During the phase of creating a simple artificial catchment, 12 scenarios were created. Table 3 shows the main parameters, describing the soil and land use conditions of the scenarios. The parameters of the selected typical soil represent their magnitude in the Upper Ems River basin.

Table3. Main Soil and Land use Parameters of the artificial areas.

Nr.	Soil	Saturated water conductivity [mm/h]	Available water capacity [mm/m]	Land Use	Root depth [cm]	Interception [mm]
1	GLEYIC PODSOL	23.75	139	Forest	150	8
2	GLEYIC PODSOL	23.75	139	Pasture	100	4
3	GLEYIC PODSOL	23.75	139	Arable land	100	3
4	Loess soil	5.42	325,5	Forest	150	8
5	Loess soil	5.42	325,5	Pasture	100	4
6	Loess soil	5.42	325,5	Arable land	100	3
7	GLEYIC PODSOL	23.75	139	F30/P10/A50/S10[1)]	150/100/100/0	8/4/3/2[2)]
8	GLEYIC PODSOL	23.75	139	F10/P5/A75/S10	150/100/100/0	8/4/3/2
9	GLEYIC PODSOL	23.75	139	F10/P20/A50/S20	150/100/100/0	8/4/3/2
10	Loess soil	5.42	325,5	F30/P10/A50/S10	150/100/100/0	8/4/3/2
11	Loess soil	5.42	325,5	F10/P5/A75/S10	150/100/100/0	8/4/3/2
12	Loess soil	5.42	325,5	F10/P20/A50/S20	150/100/100/0	8/4/3/2

1) F – Forest; P – Pasture; A – Arable land; S – Surface sealing; the number shows the relative part of each Land use of the whole artificial area

2) Interception of sealed surfaces

Differentiated climate data series are used according to the model specifications (Table 4).

Table 4. Model types and temporal resolution of the used data.

Model	NASIM	ABIMO	SWAT	ArcEGMO
Type	Physic. based with conceptual parts	conceptional	Physic. based with conceptual parts	Physic. based with conceptual parts
Precipitation	6 min	Long-term mean annual values	Daily values	Daily values
Temperature	Daily values	-	Daily values	Daily values
Evapotranspiration	Daily values	Long-term mean annual values	Daily values	Daily values

This method enables the evaluation of sensitive operating input parameters. Therewith, the requirements on the spatio-temporal discretization of the input data sets can be described more precise. Phase 2 and 3 of the sensitivity analysis are focused on the design of more complex artificial areas (more soil layers, differentiated land use). On the base of the increasing complexity of the artificial watersheds, the simulations will be advanced to the usage of "real" data. The comparison of the simulation results for the artificial areas and the derived parameter is the basis for the identification of the scale-specific relevance of parameters.

First results

First results of the relative model comparison

Selected hydrological parameters (mean, minimum and maximum runoff, mean monthly runoff, etc.) have been used for the first relative model simulation comparison. The method allows users to identify the indicators which could be used later for data transfer between different scales in FLUMAGIS. Figure 3 shows the first simulation results, calculated with datasets of simple artificial areas.

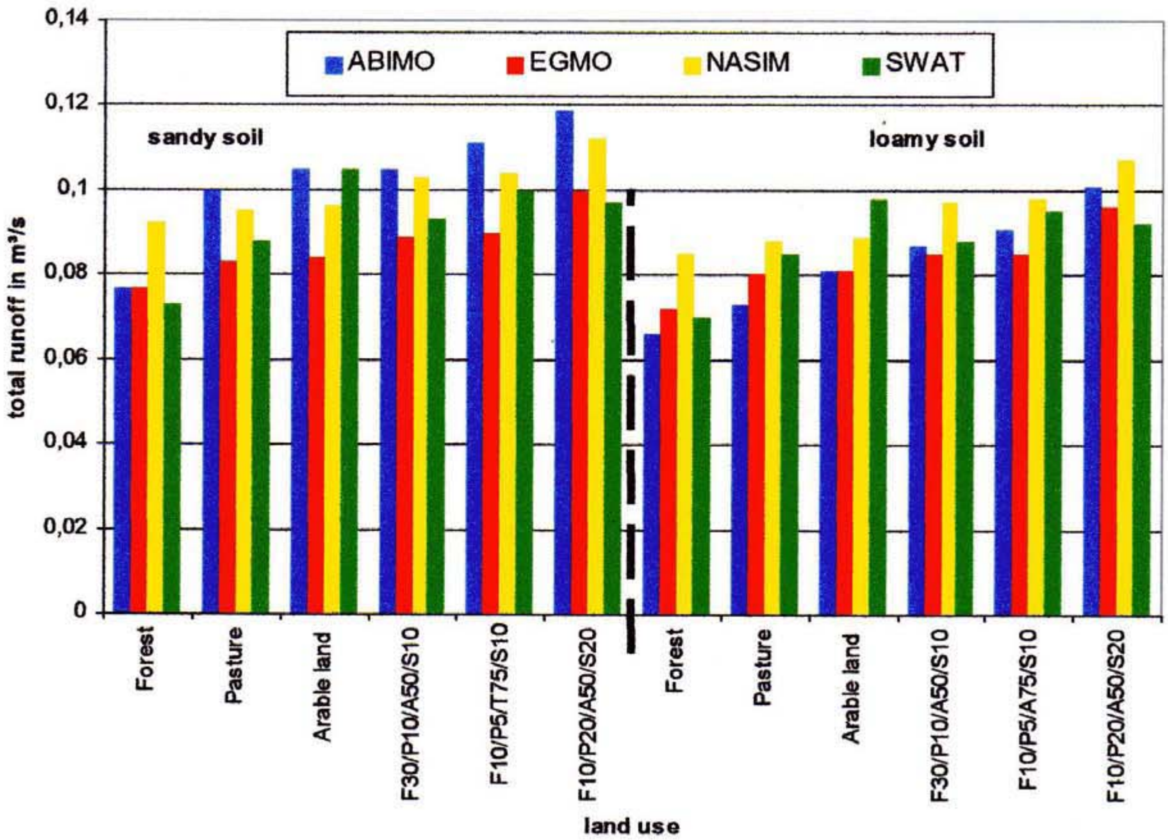

Figure 3. Comparison of simulated mean runoff in a artificial area, calculated with ABIMO, ArcEGMO, NASIM and SWAT (F – Forest, P – Pasture, A – Arable land, S – Surface sealing)

These simulations show similar results for all models: As expected, the highest values of runoff are observed under sandy soil. Under a loamy soil, the values are obviously smaller. The total runoff increases with an increased proportion of surface sealing. The simulation results of SWAT and ArcEGMO show only small differences in most cases. The differences of the simulated mean runoff are, in general, less than 10%. Only for the scenarios "arable land" do the differences rise up to 20%. This is caused by the different parametrization possibilities of the models for the agricultural characteristics. SWAT has the most options to describe agricultural cultivation practices by parameters. In order to enable a comparison to the other model systems, we had to simplify that land use type in which fertilization is controlling plant growth and seasonal evapotranspiration. More detailed parametrization would lead here to more precise results.

The simulation results of ABIMO and NASIM show greater differences (up to 30%), which are mainly caused by the different temporal resolution of the precipitation and evapotranspiration data (Table 2). The simulation results differ more widely at the other indicators like mean runoff (i.e., runoff statistics, mean monthly discharges). Those results are the first step for determining the expected variations and uncertainties of the simulation.

The next steps of the work with artificial catchments include simulations with vertically differentiated soils and a larger semi–artificial catchment. Semi-artificial means that we use a digital elevation model (DEM), a river system, and the soils of a real catchment, but the land use scenarios are created in a synthetic way. This method should enable us to get better information about the effects of input parameters on the simulation results in terms of a sensitivity analysis. Additionally, the method is used as a stepwise approach of the simulations to the more complex scenarios with the study areas.

First simulation of the whole study area using the ABIMO model

The total runoff for the whole study area of about 3,500 km^2 has been calculated with ABIMO as a first step towards a first area-wide differentiation. Due to the characteristics of the ABIMO conceptual model (Table 5), the results are long-term mean annual values of the total runoff. The model best fits flat loose rock areas (Herzog et al. 2001, Volk et al. 2001) which are found in most of the study area. Thus, it can be assumed that most of the precipitation seeps away more or less vertically and the simulated total runoff corresponds approximately with the groundwater recharge of the study area.

Table 5. Input data for the ABIMO calculation

Climate	Soil	Land use	Groundwater level classes	Degree of canalization
Long-term mean annual values of precipitation and Potential Evapo-transpiration (1960-1990, Raster data (Data source: German Weather Service DWD)	Digital soil map 1:1,000,000 (BGR. 1995) - Available water capacity - Soil types sub-divided in Sand, Silt, Clay, Loam and organic	CORINE Land Cover 1:100,000 (Statistisches Bundesamt 1994). - Land use types subdivided in arable, forest (deciduous and evergreen), gardens, sealed surfaces, water, yield classes	Derived from the Digital soil map 1:1,000,000 (BGR 1995)	Derived from Glugla & Fuertig (1996), Kuntze (1998) and CORINE

Figure 4 shows the results for the study area. According to the prevailing sandy soils with a high permeability, the values are relatively homogeneous. The floodplains with a surface-near groundwater levels have a higher evaporation and thus show the lowest runoff. The highest total runoff values are calculated for the hilly regions of the Teutoburger Wald with high precipitation amounts. The results give us a first estimation of the conditions for the whole area. The mean annual runoff of the area amounts to 368 mm/m^2 (area-weighted) which are relatively high in comparison to other lowland regions in Germany. The specific runoff reaches 11.7 l/s/km^2.

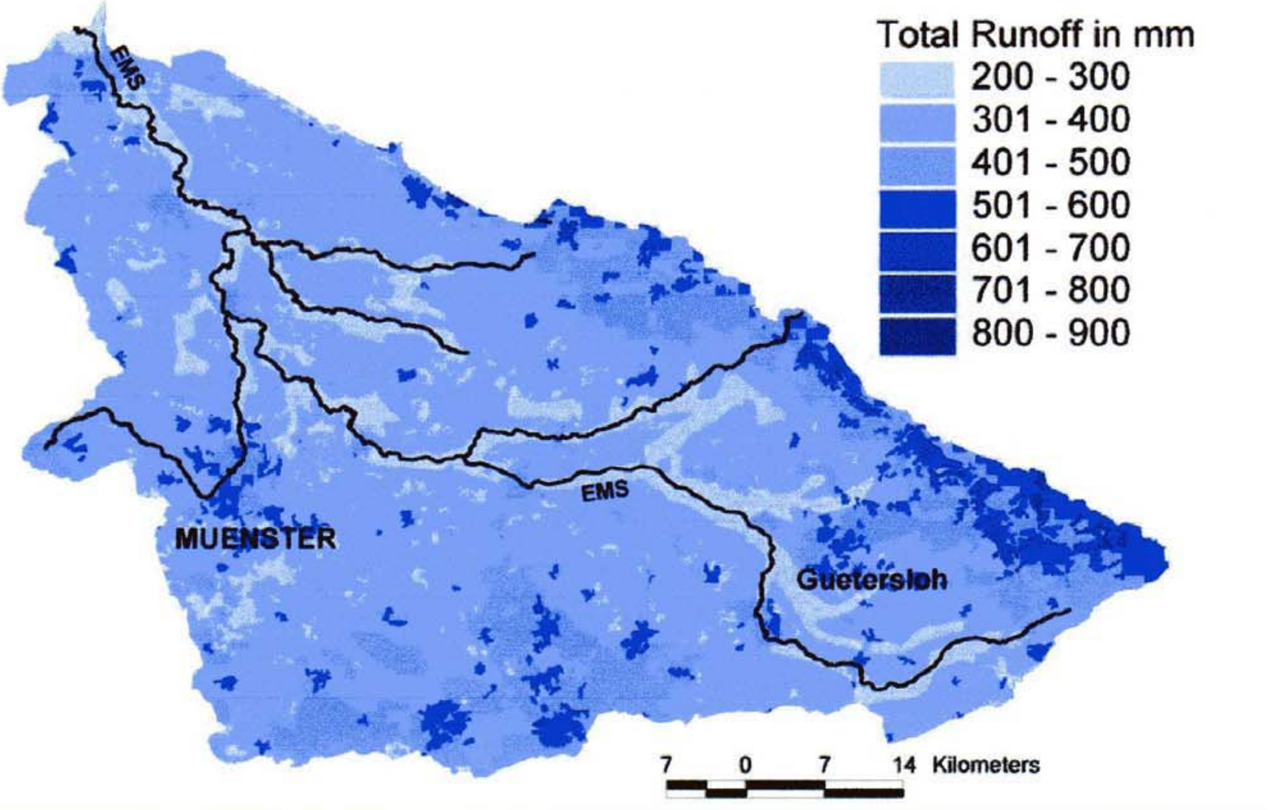

Figure 4. Total runoff for the study area simulated with the conceptual model ABIMO.

For a first validation of the received results, the streamflow data from five gauges at the Ems River were averaged and area-weighted. The daily streamflow data were passed through a digital base flow filter. The method is based on automated base flow separation and recession analysis techniques and is described in Arnold and Allen (1999) and Arnold et al. 1995). One of the results of the filter is the calculation of baseflow fractions (the percentage of baseflow contribution to the streamflow). By relating the baseflow fraction to these converted values, this results in an estimation of the baseflow contribution and a groundwater-recharge rate. This basic hydrologic information is important for river basin management, which is one goal of our project. Values about groundwater recharge and baseflow contributions help track the paths of nutrient inputs in the river system. The results are an important contribution to develop detailed and effective measures for reducing eutrophication processes in the area. Table 6 shows the comparison of the simulated runoff using ABIMO, area-weighted streamflow data and calculated baseflow fractions.

Table 6. Comparison of the ABIMO simulations, area-weighted streamflow data and baseflow calculations.

Gauge	River	Subbasin area [km2]	Mean streamflow [m^3/s]	Mean total runoff (area weighted) [mm/a]	Result of ABIMO simulation [mm/a]	Diff. [%]	Rate of baseflow contribution * [%]	Baseflow contribution (area-weighted) [mm/a]
Einen	Ems	1486	15.93	338.12	379.1	10,8	68.0	229.92
H. Langen	Ems	1616	17.49	341.31	375.6	9,1	69.5	237.21
Haskenau	Ems	1845	18.97	324.25	373.0	13,1	68.0	220.49
Greven	Ems	2842	27.77	308.15	375.1	17,8	63.0	194.13
Rheine-U.	Ems	3740	36.16	304.90	368.6	17,3	65.5	199.71

* This is an average of the first two baseflow filter passages (Fraction 1 and Fraction 2)

The area-related runoff values (converted streamflow data) show relatively small differences regarding the large areas of the gauge-related subbasins of the Ems River. As mentioned before, this is mainly caused by the relatively homogeneous soil and precipitation conditions as well as the dominant agricultural use of the study area. This is confirmed by the slight storage behavior of the river basin (a small amplitude of total runoff and baseflow contributions). The small differences between the measured and the calculated total runoff data are caused by a number of difficulties of the input information used in this study. The main problem is that the area of the given gauged basins differs from the GIS-based approximation. Additionally, the measured and calculated periods are not identically at every gauge. The input data for climate used for the water balance simulation with ABIMO covers 1961 to 1990. Some of the average streamflow data from the gauging stations are generated in longer (1950 to 2000) or shorter (1977 to 2000) time periods. A lot of environmental investigations for large areas over long time periods that use several data sets face this problem. Serious interpretation of results has to point out these uncertainties.

The results received by the ABIMO simulation could be used for the assessment of landscape functions (i.e. groundwater recharge and availability) for water protection projects on larger scales. In connection with land use information, we try to localize hot spots and potential risk zones of eutrophication of ground- and surface- water. This will be done also in more detail by the use of the model systems like SWAT, ArcEGMO and NASIM. This "top down" method enables the further focus on areas with the highest need for the implementation of management measures.

Conclusion

Despite the basically different model concepts and the temporal differentiation of the input parameters, the results show an acceptable accordance for the simple artificial areas. The simulation results of SWAT and ArcEGMO show mostly only small differences, whereas the results of ABIMO and NASIM show greater differences. In general, differences are caused by different temporal resolutions and parametrization options of the models. As a first step towards the consideration of the whole area, the conceptual model ABIMO was found suitable for estimating the indicator mean runoff for the Upper Ems River Basin.

The next phases of the relative model comparison are focused on the further development and differentiation of the artificial areas. The enhanced model comparison (all indicators, higher differentiated artificial areas) shall be the basis for the determination of the scale-specific application of indicators. Additionally, it is seen as the basis of plausible forecasts of the effects of management measures on the water balance and matter fluxes in landscapes. Another important point is the development of links from the model to the developing working platform of FLUMAGIS. It has to be examined if the derivation of parameters and indicators for the assessment of the environmental conditions is transferable to other regions. Thus, cooperation with other projects of the "River Basin Management" research program is aspired.

Acknowledgements

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Abstract: Abstract
Alternative land management practices such as conservation or no-tillage, contour farming, terraces, and buffer strips are increasingly used to reduce nonpointsource and water pollution resulting from agricultural activities. Models are useful tools to investigate effects of such management practice alternatives on the watershed level. However there is a lack of knowledge about the sensitivity of such models to parameters used to represent these conservation practices. Knowledge about the sensitivity to these parameters would help models better simulate the effects of land management. Thus, this paper presents a sensitivity analysis for conservation management parameters (specifically tillage depth, mechanical soil mixing efficiency, biological soil mixing efficiency, curve number, Manning´s roughness coefficient for overland flow, USLE support practice factor, and filter strip width) in the Soil and Water Assessment Tool (SWAT). With this analysis we aimed to improve model parameterization and calibration efficiency. Based on the results we parameterised sensitive parameters like curve number values in detail in contrast to less sensitive parameters like tillage depth and mixing efficiency.
Then the analysis consisted varying selected management practices for different crops and varying operation dates. Results showed that the model is very sensitive to applied crop rotations and in some cases even to small variations of management practices. But the different settings do not have the same

sensitivity. Duration of vegetation period and soil cover over the time with was most sensitive followed by soil cover characteristics of applied crops.

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Yours faithfully

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The use of the SWAT model to predict the impact of tillage on water quality

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Abstract

Alternative land management practices such as conservation or no-tillage, contour farming, terraces, and buffer strips are increasingly used to reduce nonpointsource and water pollution resulting from agricultural activities. Models are useful tools to investigate effects of such management practice alternatives on the watershed level. However there is a lack of knowledge about the sensitivity of such models to parameters used to represent these conservation practices. Knowledge about the sensitivity to these parameters would help models better simulate the effects of land management. Thus, this paper presents a sensitivity analysis for conservation management parameters (specifically tillage depth, mechanical soil mixing efficiency, biological soil mixing efficiency, curve number, Manning´s roughness coefficient for overland flow, USLE support practice factor, and filter strip width) in the Soil and Water Assessment Tool (SWAT). With this analysis we aimed to improve model parameterization and calibration efficiency. Based on the results we parameterised sensitive parameters like curve number values in detail in contrast to less sensitive parameters like tillage depth and mixing efficiency.

Then the analysis consisted varying selected management practices for different crops and varying operation dates. Results showed that the model is very sensitive to applied crop rotations and in some cases even to small variations of management practices. But the different settings do not have the same sensitivity. Duration of vegetation period and soil cover over the time with was most sensitive followed by soil cover characteristics of applied crops.

Keywords: SWAT, management practice, tillage, water balance, nutrient, modelling

1. INTRODUCTION

Alternative land management practices are increasingly used to reduce nonpointsource pollution resulting from agricultural activities. These practices may include reduced tillage such as conservation tillage (e.g. without deep ploughing, field preparation just before planting) or no-tillage (direct drilling). Reduction of soil tillage intensity can positively affect numerous soil properties, such as aggregate stability, macroporosity and saturated hydraulic conductivity which increases infiltration rates and reduces surface runoff, nutrient loss and soil erosion (Jones et al., 1969, Pitkänen and Nuutinen, 1998, Schmidt et al., 2001, Kirsch et al., 2002, Pandey et al., 2005, Tripathi et al., 2005). In Germany the implementation of alternative tillage systems is increasingly supported by agri-environmental programs. In the German State of Saxony, for instance, conservation tillage and mulch seeding on arable land has increased from < 1 % to about 27 % during 1994 to 2004 with support from the Saxonian Program for Environmental Agriculture (LfL, 2006). A number of field studies have illustrated the positive effects of conservation tillage and no-tillage practices on water and material fluxes at the field local level (e. g. Sloot et al., 1994, King et al., 1996, Schmidt et al., 2001), but this effect needs to be assessed on the watershed level to guide river basin management programs (Kirsch et al., 2002, Chaplot et al., 2004, Pandey et al., 2005, Behera and Panda, 2006, Bracmort et al., 2006).

Watershed models have been used for decades to evaluate nonpointsource pollution and the short- and long-term impacts of alternative management practices. However, modelling evaluations of conservation management effects at the watershed-scale are limited by the lack of management operation data. Thus, knowledge is needed about the sensitivity of such models to conservation management parameters and practices to improve the efficiency of model parameterization and the quality of model calibration.

We have chosen the semi-distributed river basin model, SWAT 2005 (**S**oil and **W**ater **A**ssessment **T**ool; Neitsch et al., 2002, Arnold and Fohrer, 2005), to examine the sensitivity of the model to selected management parameters and practices. Gassman et al. (2007) point out that “a key strength of SWAT is a flexible framework that allows the simulation of a wide variety of conservation practices and other Best Management Practices (BMPs), such as fertilizer and manure application rate and timing, cover crops (perennial grasses), filter strips, conservation tillage, irrigation management, flood prevention structures, grassed waterways, and wetlands. The majority of conservation practices can be simulated in SWAT with straightforward parameter changes.”. The SWAT model was developed for application to large complex watersheds over long periods of time (Neitsch et al., 2002). Working on the watershed scale means that required input data are often aggregated in terms of temporal scale (e.g. daily climate data). In contrast, land management parameters (tillage, fertilization, crop rotation, etc.) can be included in high resolution and detail, due to its modular structure and its historical development based on the EPIC (Erosion Productivity Impact Calculator) model (Benson et al., 1988, Neitsch et al., 2002, Arnold and Fohrer, 2005, Gassman et al., 2007).

Furthermore, potential simulation uncertainties based on ranges of realistic parameter values and on influences of scale need to be understood because simulated effects often drive financial and political decisions (Onatski and Williams, 2003). Many studies have used SWAT (Saleh et al., 2000, Shanti et al., 2001, Vache et al., 2002, Shanti et al., 2003, Chaplot et al., 2004, Pandey et al., 2005, Tripathi et al., 2005, Arabi et al., 2006, Behera and Panda, 2006) and EPIC (Sloot et al., 1994, King et al., 1996) to evaluate the effects of land use scenarios and management practices. Several studies have analyzed the long-term effects of structural Best Management Practices on water quality (e.g. Kirsch et al., 2002, Chaplot et al., 2004, Tripathi et al., 2005, Pandey et al.. 2005 or Behera and Panda, 2006, Bracmort et al., 2006). Arabi et al. (2007) investigated the impact of modelling uncertainty on evaluation of management practices using a Monte Carlo-based probabilistic approach. But, to the best of our knowledge, a sensitivity analysis of the model to conservation management parameters and practices has not been conducted. Therefore, the main objective of this study is to analyse the sensitivity of the SWAT model to selected conservation management practices to improve model parameterization and calibration. We used a semi-virtual watershed with homogeneous land use and soil, because the resulting data sets are concise and manageable and calculation time is reduced. Recommendations are given for the parameterization of tillage operations under certain conditions.

2. MATERIAL AND METHODS

2.1 Model Description

SWAT is considered as one of the most suitable models for predicting long-term impacts of land management measures on water, sediment and agricultural chemical yield (nutrient loss) in large complex watersheds with varying soils, land use and management conditions (Arnold and Fohrer, 2005, Behera and Panada, 2006, Gassmann et al., 2007). The model has been gained international acceptance as a robust interdisciplinary watershed modelling tool (Gassman et al., 2007). SWAT is a physically based, conceptual, continuous-time river basin model with spatial distributed parameters operating on a daily time step. It is not designed to simulate detailed, single-event flood routing (Neitsch et al., 2002). The relationship between input and output variables is described by regression equations. The SWAT model integrates all relevant eco-hydrological processes including water flow, nutrient transport and turn-over, vegetation growth and land use and water management at the subbasin scale. Subbasins are further disaggregated into classes of Hydrological Response Units (HRU), whereby each unique combination of the underlying geographical maps (soils, land use, etc.) forms one class. HRU are the spatial unit where the vertical flows of water and nutrients are calculated, which are then aggregated and summed for each subbasin. Water and material from HRU in sub-watersheds are routed to the sub-watershed outlet. The HRU in SWAT are spatially implicit, their exact position in the landscape is unknown, and it might be that the same HRU covers different locations in a subbasin (Neitsch et al., 2002, Di Luzio et al., 2005). The water balance for each HRU is represented by the four storages snow, soil profile, shallow aquifer and deep aquifer. The soil profile can be sub-divided up to ten soil layers. Soil water processes include evaporation, surface runoff, infiltration, plant uptake, lateral flow and percolation to lower layers (Arnold and Allen, 1996, Neitsch et al., 2002). The surface runoff from daily rainfall is estimated with a modification of SCS curve number method from USDA Soil Conservation Service (Arnold and Allen, 1996, Neitsch et al., 2002).

The nitrogen movement and transformation are simulated as a function of nitrogen cycle (Neitsch et al., 2002, Jha et al., 2004). SWAT monitors five different pools of nitrogen in the soils; two inorganic and three organic. Nitrogen is added to the soil by fertilizer, manure or residue application, fixation by bacteria, and rain (Neitsch et al., 2002). Nitrogen losses occur by plant uptake surface runoff in the solution and the eroded sediment (Neitsch et al., 2002, Jha et al., 2004).

Background for the crop growth and the management practices is the EPIC crop growth model, which is a comprehensive field scale model. EPIC was originally developed to simulate the impact of erosion on crop productivity and has now evolved into a comprehensive agricultural management, field scale, non-point source loading model (Benson et al., 1988, King et al., 1996, Neitsch et al., 2002). The management practices are defined by specific operations and parameters such as the beginning and end of growing season, timing of tillage operations as well as timing and amount of fertilizer, pesticide and irrigation application. These operations are taking place in every HRU (Neitsch et al., 2002).

2.2 Input Data

The Parthe watershed was chosen as study area. It is located in the State of Saxony in Central Germany and drains an area of about 315 km² (Figure 1). It is a subbasin of the Weisse Elster catchment in the Elbe River system. The topography of the area is flat with altitudes between 106 and 230 m above sea level. The mean annual precipitation is about 570 mm. The model input data are shown in Table 1. For the sensitivity analysis, we assumed “arable land” to be homogeneous land use without any further differentiation. A typical soil profile was used from a soil map (1:25,000) of the Parthe watershed. The use of homogenous land use and soil (semi-virtual catchment) is advantageous because the resulting data sets are concise and manageable and calculation time is reduced. Daily precipitation data and other climate data are from a weather station in the watershed.

2.3 Sensitivity Analysis

The analysis of the sensitivity of the model to selected management practices consisted of first varying management parameters (tillage depth, mechanical mixing efficiency, biological mixing efficiency, curve number, and Manning´s roughness coefficient for overland flow, USLE support practice factor, and filter strip width). Then, management practices were parameterised and varied for different crops and dates of operation. The influence of varying these practices on water balance components and nutrients was then evaluated

Management parameters

The parameters tillage depth (DEPTIL) and mechanical soil mixing efficiency (EFFMIX) define the applied tillage operation (plough, stubble cultivation, harrow etc.). These parameters define the fraction of crop residue, nutrients, pesticides and bacteria for each soil horizon, which are redistributed within the mixed soil depth (Neitsch et al., 2002). The biological soil mixing efficiency (BIOMIX) defines the activity of soil organisms, such as earthworms as representatives of macrofauna, which influence soil porosity and water fluxes by their grubbing activity. The macrofauna is very sensitive to soil tillage (Kladivoka, 2001). The SCS curve number (CN) defines soil permeability based on soil characteristics and land cover (land use). This parameter routes the process of infiltration and generation of surface runoff. The Manning´s roughness coefficient for overland flow (OV_N) is a parameter to estimate overland flow velocity, which depends on characteristics of the land surface (Neitsch et al., 2002). The management parameter USLE support practice factor (USLE_P) defines the ratio of soil loss with a specific support practice (such as contour tillage, strip cropping, and terraces) to the corresponding loss with up-and-down the cultivation (Neitsch et al., 2002). The width of edge of field filter strip (FILTERW), which affects sediment and nutrient loads in surface runoff, can be defined for each HRU. (Neitsch et al., 2002).

For each simulation only one parameter was varied within its realistic range (see Table 2). The advantage of this method is that the effect on model output is related to the variability of only the selected parameter, but it does not consider the dependency on settings chosen for the other parameters (Arabi et al., 2007).

The used management scenario was a generalised Agricultural Land Close Grown (AGRC) scenario with one fertiliser application (70 kg N/ha) and one tillage operation. The output parameters investigated are surface runoff, baseflow, total water yield, total sediment loading, organic nitrogen, organic phosphorus, nitrate in surface runoff, nitrate and phosphorus leached.

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(2002) as well as by other studies, such as by Sloot et al. (1994), Heuvelmans et al. (2004), Bracmort et al. (2006) or Arabi et al. (2007). Moreover biological soil mixing efficiency is a sensitive parameter mainly to nutrients and sediment loading. The Manning´s roughness coefficient for overland flow is only moderately sensitive to water balance components but sensitive to organic nitrogen followed by sediment loading and organic phosphorus. Both the USLE support practice factor and the width of edge of field filter strip do not influence water balance components. But USLE support practice factor is very sensitive to sediment loading, organic nitrogen and organic phosphorus while the width of edge of field filter strip is sensitive to organic nitrogen and moderately sensitive to sediment loading, organic phosphorus and nitrate in surface runoff (see Figure 3).

In this study the variation of tillage depth and mechanical soil mixing efficiency did not influence neither on water cycle output parameters nor on nutrient and sediment cycle output parameters.

Management Practices

Based on the results of the sensitivity analysis to management parameters, the tillage operations subjected to management practices (conservational (CVT), conservation (CST) and no-tillage (NOT)) were parameterised exemplary (see Table 3). Thereby, conventional tillage primarily is distinguished by tillage practices applied after harvesting with deep ploughing, previous stubble cultivation and following harrow operation before seeding/planting. For conservation management a multiplicity of measures can be taken. For tillage practice we chose altogether three variations: a) deep ploughing operation is replaced by a less intensive operation (CST_A), b) deep ploughing operation is left out and not replaced (CST_B) and c) harrow operation is applied only (CST_C).

Parameters were set as follows. Differentiated by applied tillage operation, we parameterised curve number values in detail. The Curve number adjustment depends on soil dependent basic curve number identified within calibration process, planted crop (grain and root crop), applied tillage operation and residue coverage (defined by applied management practice). The allocation of the SCS curve number is based on the parameterisations suggested by Neitsch et al. (2002) and continuative on the comments of Rawls and Richardson (1983). Rawls and Richardson (1983) recommend lowering the SCS curve number by 2% for soils with poor hydrological conditions when applying conservation tillage (compared to conventional tillage). For fields with good hydrological conditions, the SCS curve number should be lowered by 4% compared to conventional tillage. King et al. (1996)

used with EPIC a curve number value of 87 for conventional tillage and 82 for no-tillage practices responsible for soil hydrological group D (clay soil). Sloot et al. (1994) used the initial curve numbers: A value of 84 for conventional tillage, 83 for minimum tillage (conservation tillage) and 82 for no-tillage. Biological mixing efficiency is a sensitive parameter whichsoever we parameterised in detail depending on the intensity of the applied management practice. Thereby, with increasing tillage intensity the biological mixing activity decreases.

Manning´s roughness coefficient for overland flow was defined subjected to management practise and changing residue cover (Sloot et al., 1994, Neitsch et al., 2002). The parameter increases with increasing soil coverage accordingly with decreasing tillage intensity.

We defined only one tillage depths and mixing efficiencies for one main tillage operation (as applied crop-dependent on the field in reality (Abraham et al., 2004)).

Furthermore the timing of tillage operations affects soil coverage (residue decomposition). For example, fall tillage operation reduces residue over winter and spring period. The timing of tillage and the choice of tillage operation depends on the crop being planted and the chosen management practice (Kirsch et al., 2002). Therefore differences between spring and winter crops as well as between grains and root crops are expected. Hence we applied commonly planted crops: spring barley (Hordeum vulgare) – representative for grains planted in early spring; winter barley – representative for grains planted in early fall and sugar beet (Beta vulgaris) – representative for root crops planted in early spring. For each crop depending on management practice basic scenarios were defined (see Table 4). Thereby, conventional and conservation tillage primarily are distinguished by tillage practices applied straight away after harvesting with deep ploughing, previous stubble cultivation. The harrow operation is applied for all scenarios just before seeding/planting. Following sub-scenarios for conventional and conservation tillage (CVT_1, CVT_2, etc.) were defined where e.g. tillage operations were applied at different dates (for spring planted crops spring tillage instead of autumn tillage) to identify the sensitivity of SWAT model to the timing of the tillage operations. Also varying operation combinations were applied to find out if less intensive operations (e.g. harrowing after ploughing) reasonable needs to be implemented.

Furthermore the conservation management practice contouring and implementation of filter stripes were applied with base tillage scenarios of conservation tillage CST_A and CST_B (e.g. CST_Aa) and no-tillage practise (see Table 5). Within concluding scenario a catch crop (red clover) was implemented (CST_CC) for green manuring (only applied for conservation tillage basic scenario).

3. RESULTS AND DISCUSSION

Generally, our results confirm the outcome of studies undertaken by Kirsch et al. (2002), Chaplot et al. (2004), Pandey et al. (2005), Tripathi et al. (2005), and Behera and Panda (2006) that conventional tillage practices need to be replaced by less intensive tillage practices in order to minimize the sediment yield and nutrient losses. Regarding the influence on hydrology and nutrient and sediment output, with varying tillage operations for each crop we observed the largest differences between the conventional tillage and no-tillage scenarios (see Figure 4). Decreasing tillage intensity resulted in an increase of baseflow while surface runoff and total water yield decreased; organic nitrogen and phosphorus, nitrate in surface runoff, phosphorus leached and total sediment load decreased while nitrate leached increased regarding to the increase of groundwater recharge. Sloot et al. (1994) came to similar results concerning runoff and soil loss under spring wheat.

However in this study we paid major attention to even small variations of tillage operation combinations, dates of application and varying crops and how they affect water balance components, nutrient losses and sediment load. We assume both current and previous applied management practices and their specific parameterisations (crop rotations and tillage operation configuration) to be important for calibration and validation as well as to generate comparable results.

As expected we found the results strongly differing with respect to applied crops (see Figure 4). Thereby, the differences between spring planted crops and winter barley were larger than between sugar beet and grains. Generally we found the highest total water yield with spring barley followed by sugar beet and winter barley. For winter barley total water yield is only half the amount of spring barley. Therefore, based on the longer period of soil cover for winter barley, we found proportional highest baseflow (85% with CVT and 88% with NOT) compared to spring barley (54% with CVT and 66% with NOT) and sugar beet (45% with CVT and 59% with NOT). In contrast for winter barley, we found proportional lowest surface flow (14% with CVT and 11% with NOT) compared to spring barley (46% with CVT and 34% with NOT) and sugar beet (55% with CVT and 41% with NOT). Sediment loading is highest for sugar beet followed by spring barley and winter barley. Nutrient output is not comparable between the different crops because we used crop specific fertilizer amounts.

Small variations (tillage sub-scenarios) of tillage intensity affected the water balance components, nutrient losses and sediment yield also with regard to the crops applied (see Figures 5, 6 and 7). While water cycle output parameters, nutrients and sediment loading for spring barley and sugar beet showed differences already with small changes of tillage intensity, for winter barley significant changes only occurred if strong changes in tillage were applied. Thereby the effect of the single variations was very different to the investigated output parameters. A closer examination of (1) tillage intensity and (2) dates of tillage operation application showed that:

(1a) Leaving out an intensive tillage operation (CVT: CST_A and CST_B) affected the water balance components, the nutrient losses and sediment yield more than replacing it by a less intensive operation.

(1b) Leaving out single less intensive tillage operations followed more intensive operation (e.g. CVT: CVT_1) did not affect water balance components, nutrient losses and sediment yield very much.

Thereby, the influence is larger for nutrients and sediment than for the water balance components; in addition, the influence is larger for spring planted crops than for winter barley.

(2) Significant effects occurred if intensive operation after harvesting (e.g. CVT_6/7) was first applied just before sowing. The effects are negligible if the intensive operation is still first applied in fall. Effects are larger for spring planted crops because with winter barley tillage operations are applied just before sowing anyway.

Results of the implementation of conservation practices contouring and setting filter stripes did not affect water balance components but has lead to a significant decrease of organic nitrogen, organic phosphorus and sediment loading while nitrate in surface runoff is only affected by filter stripes. These effects are visible for all crops but only marginal developed for the winter grain. As expected the application of catch crops showed a decrease of nutrients and sediment loading (see Figure 8).

4. CONCLUSIONS

Based on the initial conditions (semi-virtual watershed with homogenous arable land and soil characteristics) the analysis has shown that the SWAT model is very sensitive to applied crop rotations and in some cases even to small variations of management practices. But the different settings do not have the same sensitivity. Based on the results of our analysis the following sensitivity ranking can be concluded:

1) Duration of vegetation period and soil cover over the time with
    1a) implementation of catch crop;
    1b) dates of planting (winter/spring crop);
    1c) date of first tillage operation applied after harvesting (fall tillage/spring tillage)
2) Soil cover characteristics of applied crops (e.g. grains/row crops);
3) Conservation support practices (contouring) and filter stripes
4) Tillage intensity (means applied tillage practice; basic scenarios);

We consider this ranking as a first recommendation for the parameterisation of tillage operations and management practices for SWAT users and for our further studies - always with the view to the initial conditions of input data.

Also we reason that it is not necessary to implement tillage operation successions in detail into the model especially for winter crops. Less intensive operations in connection with more intensive operations can be left out. Important is to apply the date of first intensive operation (fall/spring) and to know most important crops grown in the investigated area. Furthermore it is important to know and to implement conservation practices like catch crops, contouring and filter stripes.

With these results and based on catchment size and rate and distribution of arable land within watershed area we reason the parameterisation of crop rotations and tillage operation configurations of management practices to be important for the calibration and validation procedure as well as for the generation of comparable results.

OUTLOOK

Our overall goal is to give recommendations for land management parameterisation on different catchment sizes following a nested approach; Parthe (about 300 km²), Weiße Elster (5,300km²), Saale River Basin (23,000 km²). Therefore, our next step is the application of tillage scenarios used in this study to these different watersheds of different sizes and with differentiated land use and soil data input in order to evaluate the results for the virtual area. Afterwards, the application of differentiated crop rotations and tillage operations for different management practices as well as regionalisation of management input data and different management systems are planned.

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Tables

Table 1 Input data

Topography	**Land use**	**Soil**	**Weather**
DEM - area: 315 km² - grid cell size: 30 m	homogenous - arable land	homogenous - Cambisol	daily values - precipitation - wind speed - max. and min. air temperature - solar radiation - relative humidity

Table 2 Basic parameter settings and variation ranges

Parameter	Basic Setting	Parameter Range
CN	75	35 – 95
BIOMIX	0.2	0 – 1.0
OV_N	0.14	0.01 – 0.5
DEPTIL (cm)	30	0 – 95
EFFMIX	0.5	0 – 1.0
USLE_P	1.0	0.1 – 1.0
FILTERW (m)	0.0	0 – 5.0

Table 3 Parameterisation of tillage operations within management practices (*CN is exemplarily used representatively for soil hydrological group A – soil with good hydrological conditions)

<table>
<tr><th rowspan="2">Scenario</th><th rowspan="2">Tillage operation</th><th rowspan="2">DEPTIL (cm)</th><th rowspan="2">EFFMIX</th><th rowspan="2">BIOMIX</th><th rowspan="2">OV_N</th><th colspan="2">CN*</th></tr>
<tr><th>Grains</th><th>Row crops</th></tr>
<tr><td colspan="8">Conventional tillage (CVT)</td></tr>
<tr><td>CVT</td><td>cultivation stubble</td><td>12</td><td>0.45</td><td>0.1</td><td>0.09</td><td colspan="2">76</td></tr>
<tr><td></td><td>plough (bare soil)</td><td>25</td><td>0.85</td><td></td><td></td><td colspan="2">77</td></tr>
<tr><td></td><td>harrow</td><td>7</td><td>0.3</td><td></td><td></td><td colspan="2"></td></tr>
<tr><td></td><td>plant</td><td></td><td></td><td></td><td></td><td>63</td><td>67</td></tr>
<tr><td></td><td>harvest</td><td></td><td></td><td></td><td></td><td colspan="2">74</td></tr>
<tr><td colspan="8">Conservation tillage (CST)</td></tr>
<tr><td>CST_A</td><td>cultivation stubble</td><td>12</td><td>0.45</td><td>0.2</td><td>0.13</td><td colspan="2">76</td></tr>
<tr><td></td><td>harrow</td><td>7</td><td>0.3</td><td></td><td></td><td colspan="2"></td></tr>
<tr><td></td><td>plant</td><td></td><td></td><td></td><td></td><td>62</td><td>66</td></tr>
<tr><td></td><td>harvest</td><td></td><td></td><td></td><td></td><td colspan="2">74</td></tr>
<tr><td>CST_B</td><td></td><td></td><td></td><td>0.3</td><td>0.19</td><td colspan="2"></td></tr>
<tr><td>CST_C</td><td>harrow</td><td>7</td><td>0.3</td><td>0.4</td><td>0.3</td><td colspan="2">74</td></tr>
<tr><td></td><td>plant</td><td></td><td></td><td></td><td></td><td>61</td><td>65</td></tr>
<tr><td></td><td>harvest</td><td></td><td></td><td></td><td></td><td colspan="2">73</td></tr>
<tr><td colspan="8">No tillage (NOT)</td></tr>
<tr><td></td><td>plant</td><td></td><td></td><td>0.4</td><td>0.3</td><td>60</td><td>64</td></tr>
<tr><td></td><td>harvest</td><td></td><td></td><td></td><td></td><td colspan="2">73</td></tr>
</table>

Table 4 Tillage scenarios based on tillage practices; (P)-Plough, (St)-stubble cultivation, (H)-Harrow, (S)-Seed (dates by Abraham et al., 2004)

	Dates of operation					
	Fall tillage		**Spring tillage**			
spring barley	03. August	10. October	01. March	05. March	10. March	15. March
winter barley	12. August	20. August			01. September	05. September
sugar beet	08. October	11. October	26. March	01. April	05. April	10. April
Conventional tillage						
CVT	St	P	-	-	H	S
CVT_1	St	P	-	-	-	S
CVT_2	-	P	-	-	H	S
CVT_3	-	P	-	-	-	S
CVT_4	St	-	-	P	H	S
CVT_5	St	-	-	P	-	S
CVT_6	-	-	St	P	H	S
CVT_7	-	-	St	P	-	S
Conservation tillage						
CST_A	St	St	-	-	H	S
CST_A1	St	St	-	-	-	S
CST_A2	St	-	-	St	H	S
CST_A3	St	-	-	St	-	S
CST_A4	-	-	St	St	H	S
CST_A5	-	-	St	St	-	S
CST_B	St	-	-	-	H	S
CST_B1	St	-	-	-	-	S
CST_B2	-	St	-	-	H	S
CST_B3	-	St	-	-	-	S
CST_B4	-	-	-	St	H	S
CST_B5	-	-	-	St	-	S
CST_B6	-	-	St	-	H	S
CST_B7	-	-	St	-	-	S
CST_C	-	-	-	-	H	S
No-tillage						
NOT	-	-	-	-	-	S

Table 5 Parameter settings for contouring and filter stripes

Scenario	Parameter	
	USLE_P	FILTERW (m)
_a	0.6	0
_b	1.0	2
_c	0.6	2

Figure captions

Figure 1 Location of the study area in Germany

Figure 2 Sensitivity of SWAT model to tillage parameters: CN, BIOMIX and OV_N

Figure 3 Sensitivity of SWAT model to management practice parameters USLE_P and FILTERW

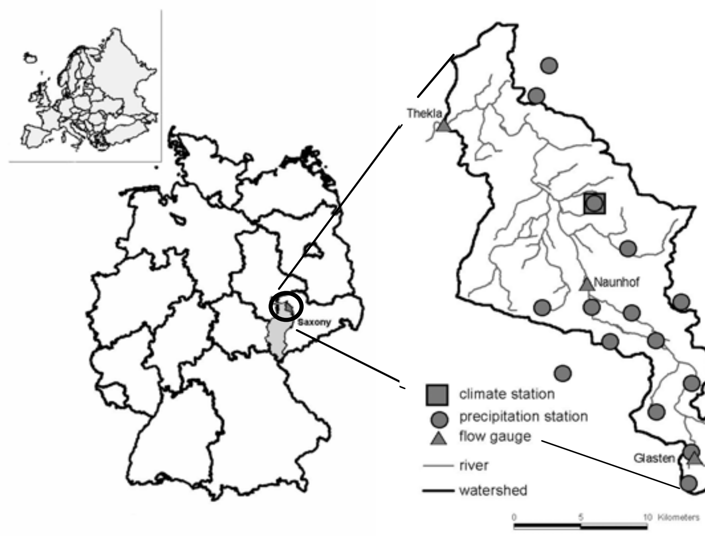
Figure 4 Average values of the sensitivity analysis: Effect of tillage intensity (basic scenarios) with spring barley, sugar beet and winter barley on water balance components, nutrients and sediment loading

Figure 5 Average values of the sensitivity analysis: Effect of different tillage operations with spring barley on water balance components, nutrients and sediment

Figure 6 Average values of the sensitivity analysis: Effect of different tillage operations with sugar beet on water balance components, nutrients and sediment

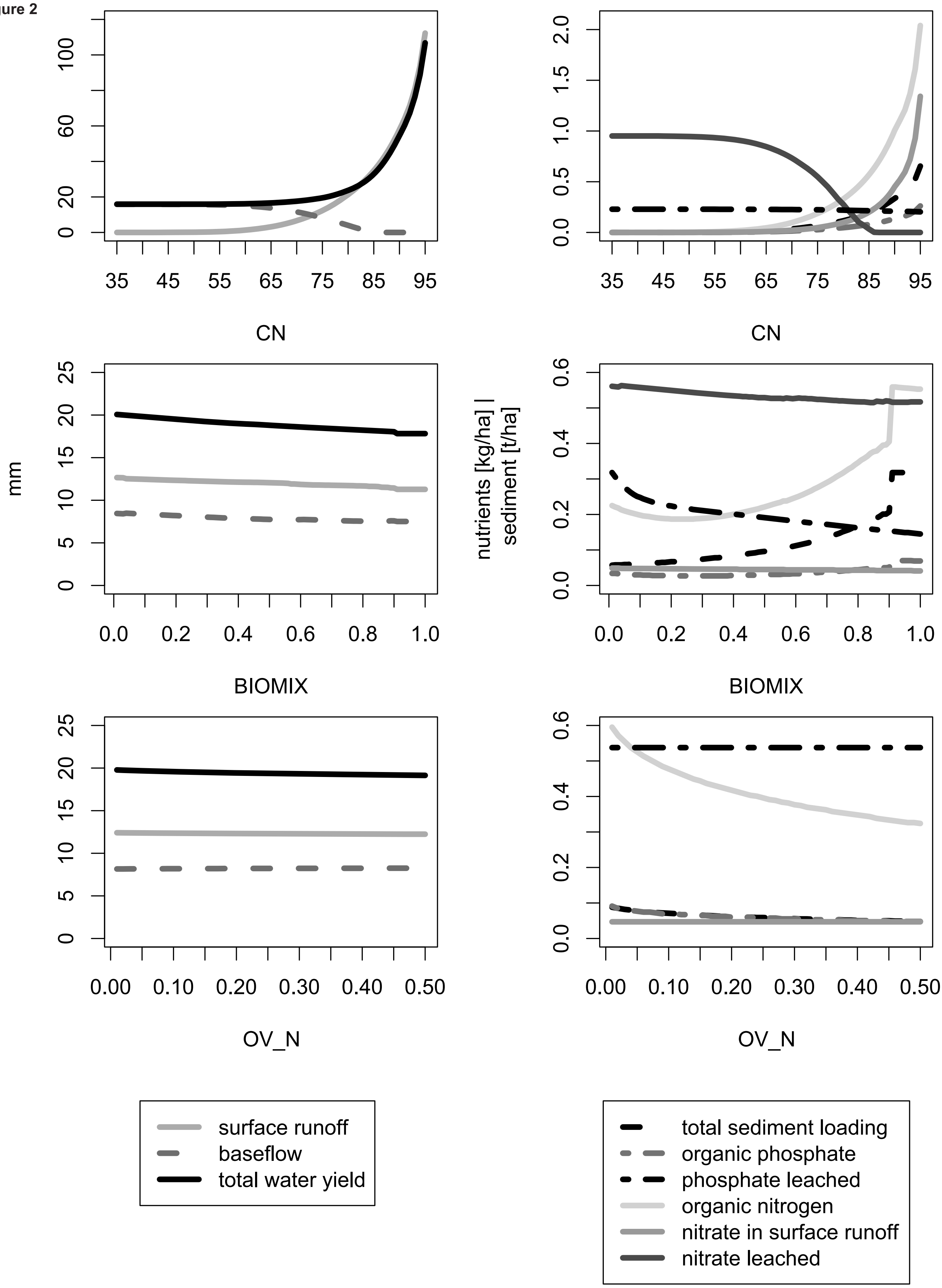
Figure 7 Average values of the sensitivity analysis: Effect of different tillage operations with winter barley on water balance components, nutrients and sediment

Figure 8 Average values of the sensitivity analysis: Effect of different conservation practices on the example of sugar beet on nutrients and sediment



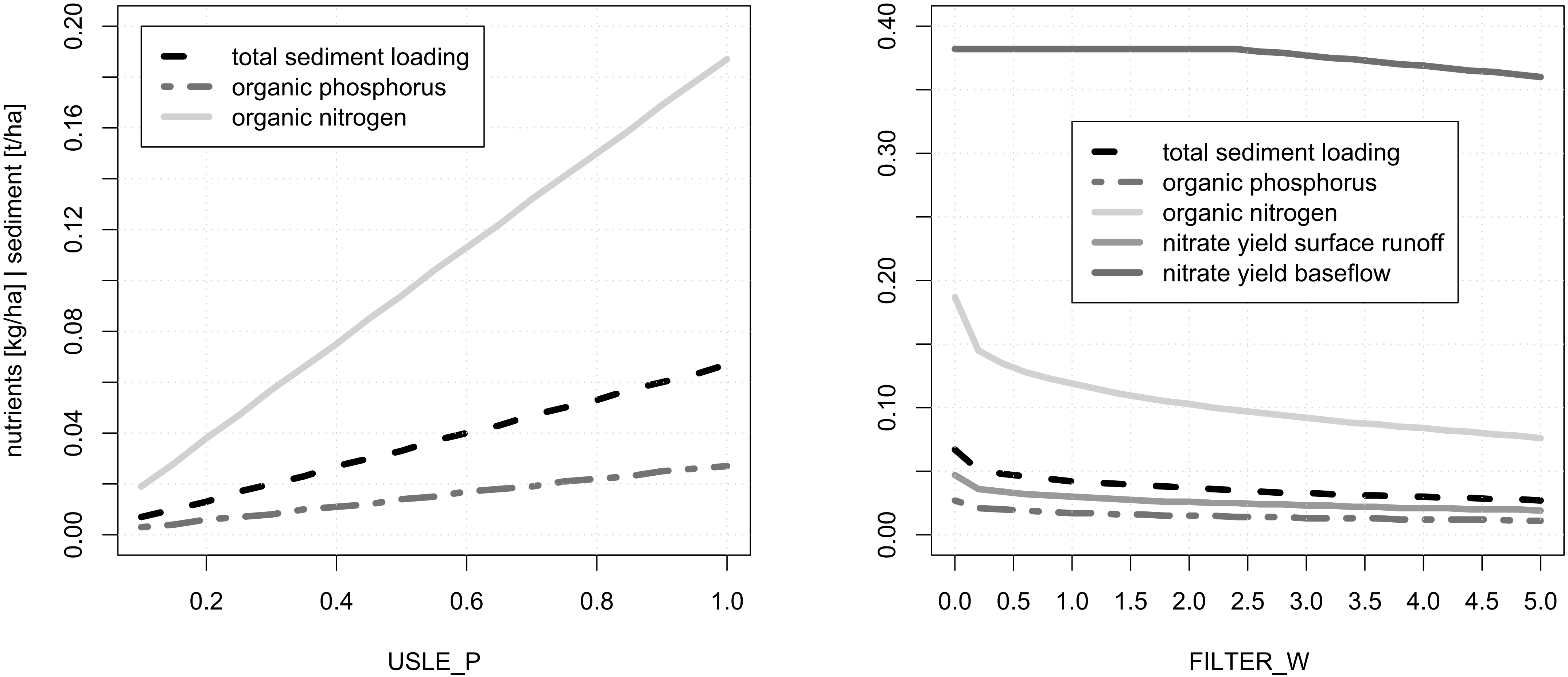
Thekla
Naunhof
Glasten
Saxony
climate station
precipitation station
flow gauge
river
watershed
0 5 10 Kilometers

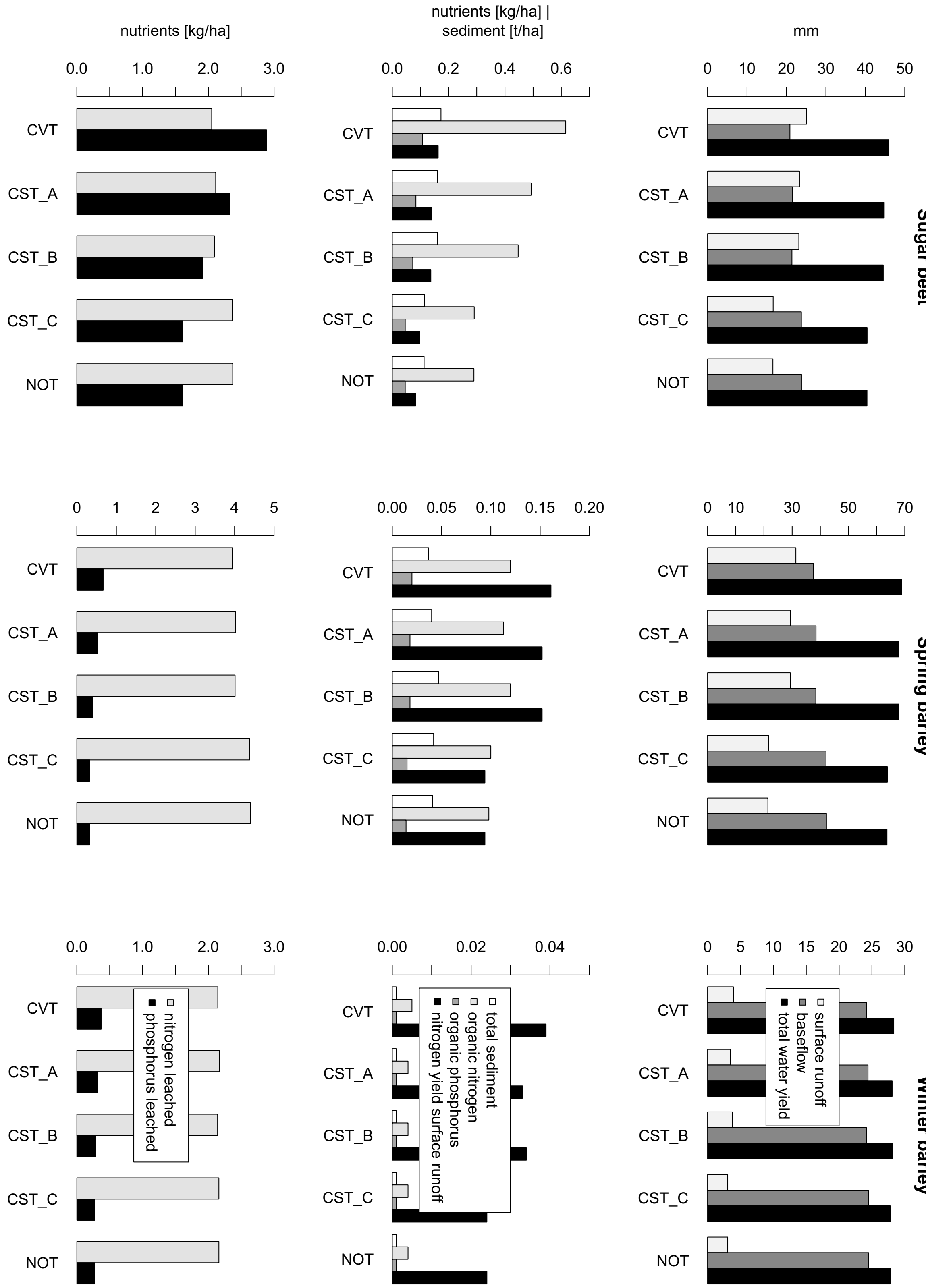
Figure 2



CN
BIOMIX
OV_N
mm
nutrients [kg/ha] |
sediment [t/ha]
surface runoff
baseflow
total water yield
total sediment loading
organic phosphate
phosphate leached
organic nitrogen
nitrate in surface runoff
nitrate leached

Figure 3

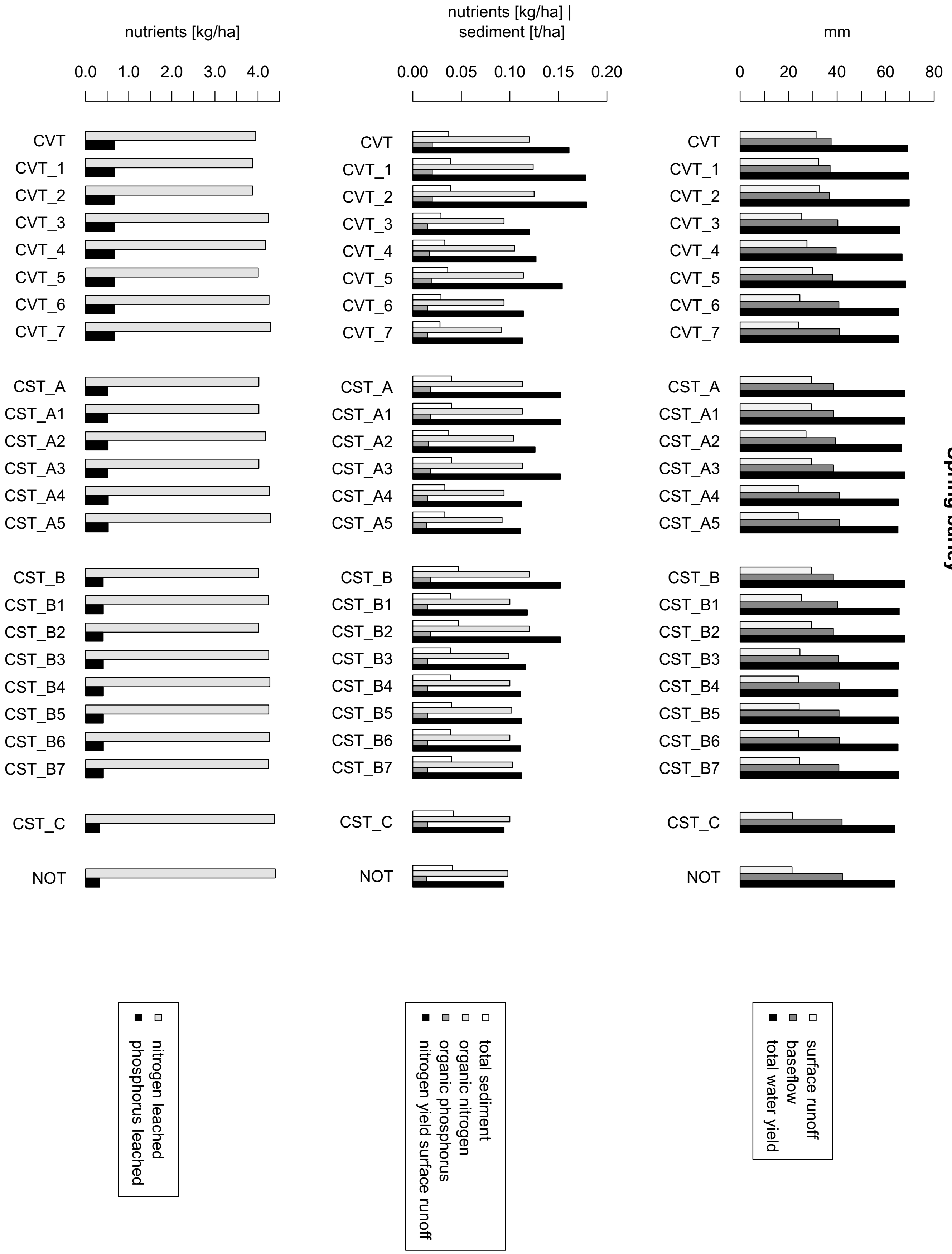

nutrients [kg/ha] | sediment [t/ha]
0.00
0.04
0.08
0.12
0.16
0.20
total sediment loading
organic phosphorus
organic nitrogen
0.2
0.4
0.6
0.8
1.0
USLE_P
0.00
0.10
0.20
0.30
0.40
total sediment loading
organic phosphorus
organic nitrogen
nitrate yield surface runoff
nitrate yield baseflow
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0
FILTER_W



nutrients [kg/ha]
nutrients [kg/ha] | sediment [t/ha]
mm
CVT
CST_A
CST_B
CST_C
NOT
Sugar beet
Spring barley
Winter barley
nitrogen leached
phosphorus leached
total sediment
organic nitrogen
organic phosphorus
nitrogen yield surface runoff
surface runoff
baseflow
total water yield

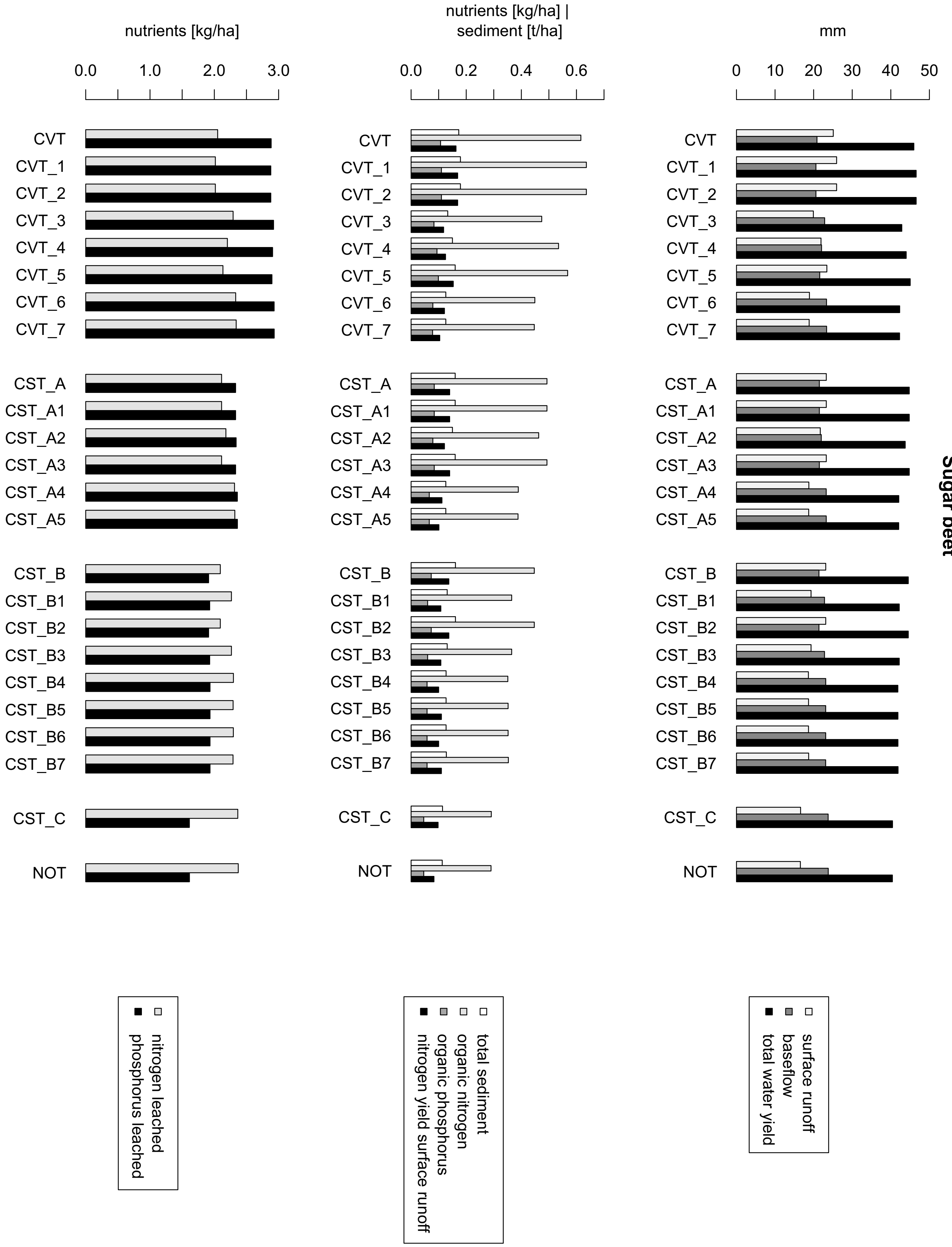
Figure 4

Figure 5



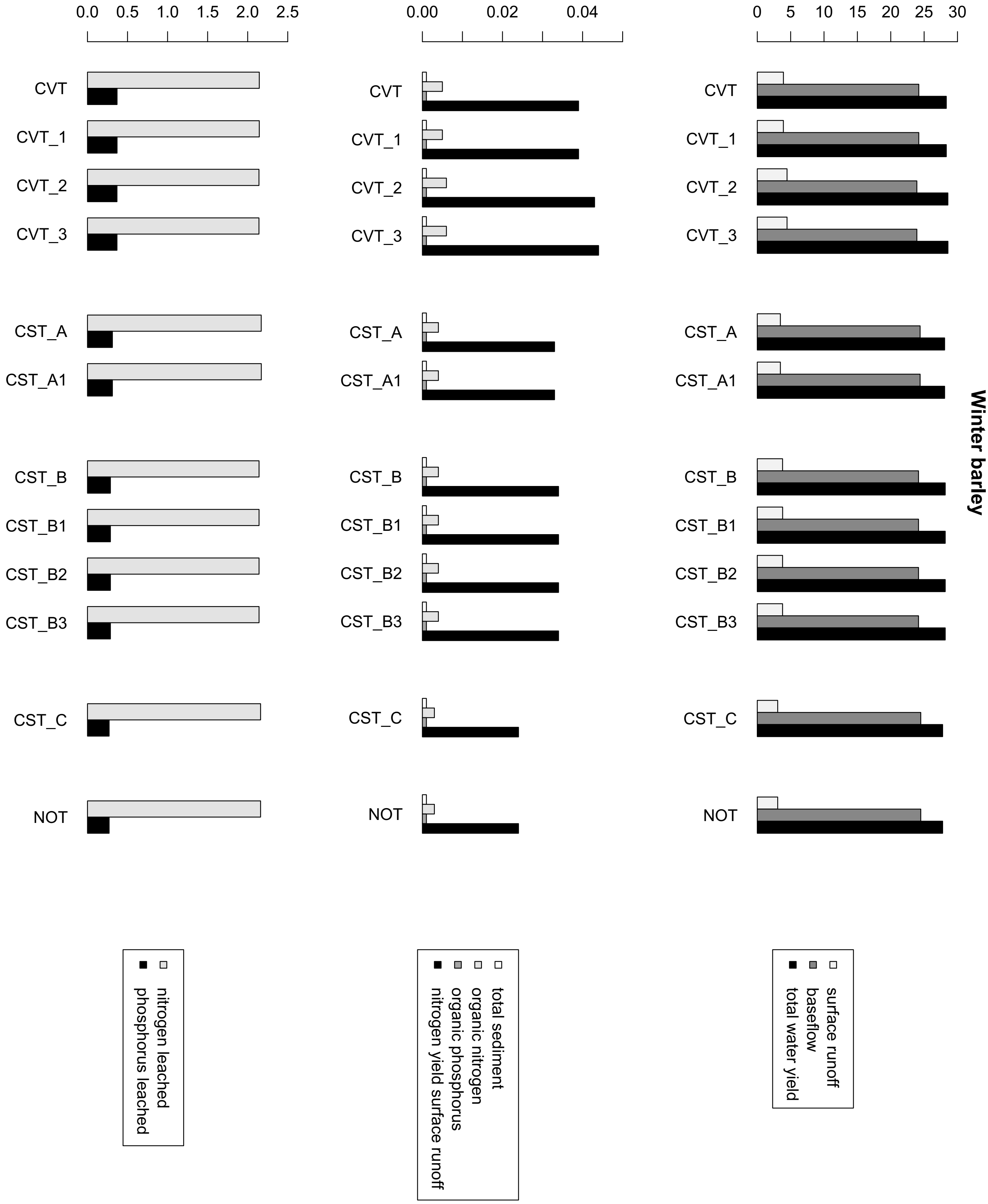
Spring barley
nutrients [kg/ha]
0.0 1.0 2.0 3.0 4.0
nutrients [kg/ha] | sediment [t/ha]
0.00 0.05 0.10 0.15 0.20
mm
0 20 40 60 80
CVT
CVT_1
CVT_2
CVT_3
CVT_4
CVT_5
CVT_6
CVT_7
CST_A
CST_A1
CST_A2
CST_A3
CST_A4
CST_A5
CST_B
CST_B1
CST_B2
CST_B3
CST_B4
CST_B5
CST_B6
CST_B7
CST_C
NOT
nitrogen leached
phosphorus leached
total sediment
organic nitrogen
organic phosphorus
nitrogen yield surface runoff
surface runoff
baseflow
total water yield

Figure 6



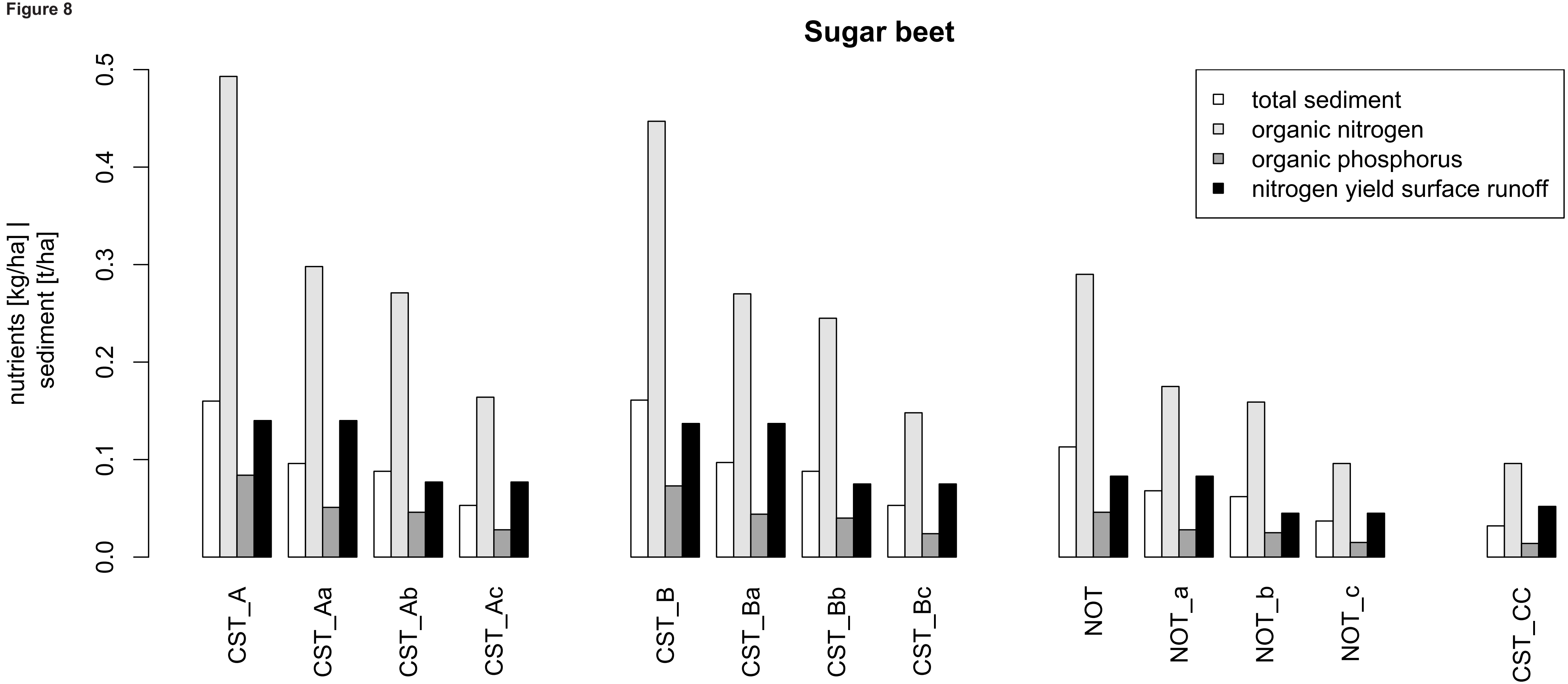
nutrients [kg/ha]
0.0
1.0
2.0
3.0
nutrients [kg/ha] |
sediment [t/ha]
0.0
0.2
0.4
0.6
mm
0
10
20
30
40
50
CVT
CVT_1
CVT_2
CVT_3
CVT_4
CVT_5
CVT_6
CVT_7
CST_A
CST_A1
CST_A2
CST_A3
CST_A4
CST_A5
CST_B
CST_B1
CST_B2
CST_B3
CST_B4
CST_B5
CST_B6
CST_B7
CST_C
NOT
Sugar beet
nitrogen leached
phosphorus leached
total sediment
organic nitrogen
organic phosphorus
nitrogen yield surface runoff
surface runoff
baseflow
total water yield

Figure 7



Winter barley
nutrients [kg/ha]
nutrients [kg/ha] | sediment [t/ha]
mm
CVT
CVT_1
CVT_2
CVT_3
CST_A
CST_A1
CST_B
CST_B1
CST_B2
CST_B3
CST_C
NOT
nitrogen leached
phosphorus leached
total sediment
organic nitrogen
organic phosphorus
nitrogen yield surface runoff
surface runoff
baseflow
total water yield

Figure 8


Sugar beet
nutrients [kg/ha] | sediment [t/ha]
0.0
0.1
0.2
0.3
0.4
0.5
total sediment
organic nitrogen
organic phosphorus
nitrogen yield surface runoff
CST_A
CST_Aa
CST_Ab
CST_Ac
CST_B
CST_Ba
CST_Bb
CST_Bc
NOT
NOT_a
NOT_b
NOT_c
CST_CC

A2.10

Ullrich, A., **Volk, M.**, and Schmidt, G., 2008. Influence of the uncertainties of monitoring data on model calibration and evaluation. In: M. Sànchez-Marrè, J. Béjar, J. Comas, A. Rizzoli and G. Guariso (Eds.): Proceedings of the iEMSs Fourth Biennial Meeting: International Congress on Environmental Modelling and Software (iEMSs 2008). International Environmental Modelling and Software Society, Barcelona, Catalonia, July 2008. Vol. 1: 544-552. ISBN: 978-84-7653-074-0.
http://www.iemss.org/iemss2008/uploads/Main/Vol1-iEMSs2008-Proceedings.pdf (peer reviewed).

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Integrating Sciences and Information Technology for Environmental Assessment and Decision Making
4th Biennial Meeting of iEMSs, http://www.iemss.org/iemss2008/index.php?n=Main.Proceedings
M. Sànchez-Marrè, J. Béjar, J. Comas, A. Rizzoli and G. Guariso (Eds.)
International Environmental Modelling and Software Society (iEMSs), 2008

Influence of the uncertainties of monitoring data on model calibration and evaluation

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Abstract: The model-based prediction of the impact of different land management on nutrient loading requires measured nutrient flux data. Thereby the accurate calibration and evaluation of the models need an adequate data base in form of monitoring data. Uncertainties in the monitoring data influence the calibration and thus the parameter settings which affect the modelling results. Hence, we compared three different time-based sampling strategies and four different load estimation methods for model calibration and compared the results. For our study we used the river basin model SWAT (**S**oil and **W**ater **A**ssessment **T**ool). Study area is the intensively used loess-dominated Parthe watershed (315 km²) in Central Germany.
Nitrate-N load estimation results differ considerable depending on sampling strategy, used load estimation method and period of interest. For study period the load estimation results for the daily composite data set have the lowest ranges (14% and 2% maximum deviation related to the mean value of all applied methods). In contrast estimation results for the sub-monthly and the monthly data set vary in greater ranges (between 25% and 52%). To show differences between sampling strategies we calculated the percentage deviation of mean load estimations of sub-monthly and monthly data sets related to the mean estimation value of composite data set. The maximum deviation of 82% occurs for the sub-monthly data set in 2000. This affects the model and leads to different parameter settings in model calibration and evaluation. Therefore we recommend both the implementation of optimised monitoring programs and the use of more than one load estimation method to describe the water quality situation in a better way and to establish a good calibration base for simulation models.

***Keywords**:* SWAT; modelling; water quality sampling; load estimation; model calibration.

1. INTRODUCTION

With this paper, we postulate the implementation of optimised and effective monitoring programs to support a sustainable water quality protection and to establish a good calibration base for simulation models. Simulation models are powerful tools to evaluate the impact of land management scenarios on water quantity and quality at the watershed scale [Chaplot et al., 2004; Behera and Panda, 2006; Bracmort et al., 2006; Pandey et al., 2005]. The results of such scenarios can be used for the development of efficient water quality management plans in river basin management. However several problems still exist when using models for spatially explicit simulation of the environmental impact of land management options and environmental measures. Accurate calibration and evaluation of the models need an adequate data base in form of monitoring data. Uncertainties in the monitoring data influence the calibration [Harmel et al., 2006a] and thus the parameter settings which affect the modelling results.

There are two main aspects of the influence of monitoring data uncertainty on model calibration and evaluation: i) sampling frequency of water quality data and ii) load estimation method. For water quality data sampling the point of time and frequency at which discrete water samples are collected are important to reflect temporal changes [Harmel et al., 2006b; Tate et al., 1999]. Therefore, sampling strategies should include frequent samples taken for the entire range of observed flow to characterise water quality. However, due to financial and personnel constraints, the number of samples that can be collected is often limited [Harmel et al., 2002, 2003; Harmel and King, 2005; King and Harmel, 2003; Robertson and Roerish, 1999; Tate et al., 1999]. In Germany the basic water quality monitoring is organised by the Federal States. The Regional Authorities for Environment are responsible for water sampling and water quality monitoring. Usually, discrete samples are collected 13 to 24 times per year on a regular time basis [SMUL, 2005]. This is a commonly used sampling strategy for large rivers also in other countries [e.g. Robertson and Roerish, 1999; Robertson, 2003]. However, during storm water events, water levels and pollutant concentrations can change very rapidly especially in small streams and, however, periodic sampling does not adequately describe the rapid changes in water quality [Robertson and Roerish, 1999]. Therefore, small streams (such as the study area) need more intensive sampling strategies to achieve precise and accurate load estimations [Harmel et al., 2003; Harmel and King, 2005]. The accuracy of load estimates depends on the sampling method, sampling frequency, load estimation methodology, and the duration and period of the estimation [Littlewood, 1995; Littlewood and Marsh, 2005]. Inaccuracy or imprecision of load estimates limits its use in environmental assessment and management, trend detection, and watershed simulation [Littlewood and Marsh, 2005]. Therefore, the accurate load estimation and water quality characterization are important to accomplish the objectives of alternative management plans. Diverse studies have dealt with comparative analysis concerning either the use of different sampling strategies or/and the use of different methods for load estimation for different constitutes [e.g. Ferguson, 1987; Izuno et al., 1998, Keller et al., 1997; Littlewood, 1995; Robertson and Roerish; 1999, Stone et al., 2000; Swistock et al., 1997; Walling and Webb, 1981; Webb et al., 1997].

The overall goal of our study was to investigate the influence of uncertainty of monitoring data on model calibration, model parameter settings and model evaluation. Hence, we compared different load estimation methods based on three different time-based sampling strategies to estimate load data for model calibration. For our study we used the continuous-time river basin model SWAT (**S**oil and **W**ater **A**ssessment **T**ool). SWAT has been developed to predict the long-term impacts of land management measures on water, sediment and agricultural chemical yield in large complex watersheds with varying soils, land use and management conditions [Arnold and Fohrer, 2005; Behera and Panda, 2006]. Study area is the intensively used loess-dominated Parthe watershed (315 km²) in Central Germany. The investigated water quality parameter is nitrate-N.

2. MATERIAL AND METHODS

2.1 Study area

The Parthe watershed (study area) is located in the State of Saxony in Central Germany and drains an area of about 315 km² (Figure 1). It is a subbasin of the Weisse Elster catchment in the Elbe River system. The topography of the area is flat with altitudes between 106 m and 230 m above sea level. The mean annual precipitation ranges about 590 mm to 640 mm (1981-2000). The Parthe is a typical lowland river. The runoff dynamics are characterized by high flows in spring due to snow melt and rainfall and long periods of low flows in summer with occasional storm flow events. Mean long-term flow rate is 0.9 m³/s.

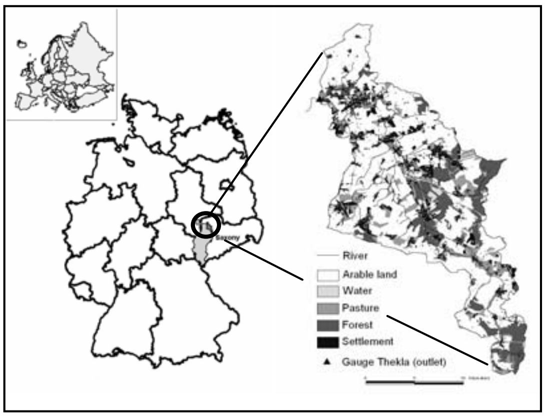

Figure 1. Location of the study area in Germany

2.2 Model description

SWAT is considered as one of the most suitable models to predict the long-term impacts of land management measures on water, sediment and agricultural chemical yield (nutrient loss) in large complex watersheds with varying soils, land use and management conditions [Arnold and Fohrer, 2005; Behera and Panda, 2006; Gassmann et al., 2007]. SWAT is a physically based, conceptual, continuous-time river basin model with spatial distributed parameters operating on a daily time step. It is not designed to simulate detailed, single-event flood routing [Neitsch et al., 2002]. The SWAT model integrates all relevant eco-hydrological processes including water flow, nutrient transport and turn-over, vegetation growth, land use and water management at the subbasin scale. Subbasins are further disaggregated into classes of Hydrological Response Units (HRU), whereby each unique combination of the underlying geographical maps (soils, land use, etc.) forms one class. The water balance for each HRU is represented by the four storages snow, soil profile, shallow aquifer and deep aquifer. The soil profile can be sub-divided up to ten soil layers. Soil water processes include evaporation, surface runoff, infiltration, plant uptake, lateral flow and percolation to lower layers [Arnold and Allen, 1996; Neitsch et al., 2002]. The surface runoff from daily rainfall is estimated with a modification of SCS curve number method [Arnold and Allen, 1996; Neitsch et al., 2002].
In Swat, nitrogen movement and transformation are simulated as a function of nitrogen cycle. SWAT simulates five different pools of nitrogen in the soils; two inorganic and three organic. Nitrogen is added to the soil by fertilizer, manure or residue application, fixation by bacteria, and rain. Nitrogen losses occur by plant uptake surface runoff in the solution and the eroded sediment [Neitsch et al., 2002].

2.3 Input data and model calibration (water cycle)

The applied input data sets are listed following: digital elevation model (30x30 m); several precipitation stations (daily sums, Environmental Operating Company (UBG)); one climate station (daily values, UBG), land use (habitat cartography), 1:10,000, Statistical Office of the Free State of Saxony, aggregated to five classes [arable land, pasture, forest, water, settlement]); waste water treatment plants, State Agency for Environment (StUFA); soil mapping (1968) (1:25,000, M. Thomas-Lauckner, (unpublished)); several crop rotations (conventional managed), including applied tillage operations and fertiliser applications.

A period of three years was used either for model calibration (1994 to 1996) and evaluation (1998 to 2000). First the rates of surface flow, lateral flow and baseflow were adjusted. Basically the following model parameters were adjusted. The curve number (CN2 - lowered), soil evaporation compensation factor (ESCO – 0.85 to 0.9; spatially adjusted), plant uptake compensation factor (EPCO – 0.8), effective hydraulic conductivity in channel alluvium (CH_K1/2 – spatially adjusted), Manning´s roughness coefficient for overland flow (OV_N – increased for agricultural land and pasture), Manning´s roughness coefficient for main and tributary channel (CH_N1/2 – 0.35), surface runoff lag coefficient (SURLAG – 1.0), maximum canopy storage (CANMX – 4.0 for arable land, 5.0 for pasture and 9.5 for forest), groundwater delay times (GW_DELAY – 150 to 350; spatially adjusted), baseflow alpha factor (ALPHA_BF – 0.065), threshold depth of water in the shallow aquifer required for return flow to occur (GWQMN – 0.0), groundwater "revap" coefficient (GWREVAP – 0.03), threshold depth of water in the shallow aquifer for "revap" or percolation to the deep aquifer to occur (REVAPMN – 0.05).

To evaluate the model predictions the following goodness-of-fit parameters were used: mean discharge, standard deviation (STD), coefficient of determination (R^2; indicates the quality of relationship between observed and predicted results); Nash-Sutcliff efficiency (NSE; indicates the model efficiency [Nash and Suttcliffe, 1970]) and prediction efficiency (PE; indicate the model's ability to describe the probability distribution of the observed results) (see Table 1). The measured and predicted monthly discharge values matches quite well at gauge Thekla. The model efficiency dropped only in very dry and in very wet years of prediction.

Table 1. Goodness-of-fit parameters for calibration of monthly predicted discharge values (watershed outlet-gauge Thekla)

	1994-1996		**1998-2000**	
	observed	**predicted**	**observed**	**predicted**
Mean discharge m³/s	1.19	1.16	1.00	1.35
STD	0.80		0.80	
R^2	0.75		0.80	
NSE	0.72		0.56	
PE	0.88		0.72	

The Parthe system is impaired by several small scale human activities which are difficult to simulate. Furthermore, most sections of the river channel are heavily regulated. Therefore, we consider the results as satisfactory.

2.4 Water quality monitoring

The samples for water quality investigation were collected using three different time-based sampling strategies: periodic grab samples taken on regular time intervals corresponding to biweekly or monthly intervals and composite samples. The time span of daily composite sampling strategy was limited from 2000 to 2001. The periodic grab samples were taken at random times within the day and not adjusted to represent selected flow rates. The composite samples are isochronous (40 ml/hour), not flow-weighted, and stored in a single collection bottle that represents the daily mean concentration. None of the three sampling strategies include flow-stratified sampling.

2.5 Load estimation

Most of the commonly applied load estimation methods differ with respect to the sampling strategy and the hydrological characteristics of the investigated streams and constituents. For the presented study, we chose four equations, which were applied to the data sets of the different sampling strategies:

$$L_{(1)} = F\frac{\overline{Q}_{sd}}{Q_{sd}}\left(\frac{1}{N}\sum_{N=1}^{N} cQ_{sd}\right) \tag{1}$$

$$L_{(2)} = F\frac{1}{N}\sum_{i=1}^{N} cQ \tag{2}$$

$$L_{(3)} = L_{(2)} + F\frac{1}{N}\sum_{i=1}^{N} r_{Qc}\sigma_Q\sigma_c \tag{3}$$

$$L_{(4)} = F\frac{1}{N}\sum_{i=1}^{N} cQ\,;\, T(Q) = aQ^b \tag{4}$$

with: L- load (t), F- factor to take account period of record, c- sample concentration (mg/l), Q- discharge at sample time (m³/s), N- number of samples, Q_{sd}- mean discharge for sampling day (m³/s), $\overline{Q}_{sd}$- mean flow (m³/s), r_{Qc}- correlation coefficient of Q and c, σ_Q- standard deviation of Q, σ_c- standard deviation of c, T- mean daily transport (t)

The first three equations are interpolation methods while the fourth equation is an extrapolation method. Equations (1) and (2) are the first and second choice of the OSPAR (Oslo-Paris) Convention [Littlewood, 1995, OSPAR, 2004]. Webb et al., [1997] found the methods (1) and (2) are quite accurate, but suffer from a high degree of imprecision. For solute fluxes, they state that these methods produce generally more reliable estimates. For the equation (2), substantial systematic errors only occur with large variability of discharge and concentration within sampling period. Especially if there is no correlative relation between concentration and discharge, the method should lead to good load estimation results [Keller et al., 1997]. Equation (3) uses statistical values to quantify the variability of concentration and discharge. Equation (4) assumes a relationship between transport and discharge at the time of sampling [Webb et al., 1997]. It uses a correlative relation (rating curve) between these variables.

3. RESULTS AND DISCUSSION

3.1 Load estimation

The annual nitrate-N load estimation results have a wide range with respect to i) the water quality sampling strategy, ii) the applied load estimation methods and iii) the period of interest and its hydrological characteristics (see Table 2). For the study period, the load estimation results for the daily composite data set have the lowest ranges (14% and 2% maximum deviation related to the mean value of all applied methods). In contrast the estimation results for the sub-monthly and the monthly data set vary in greater ranges (between 25% and 52%). Therefore, the daily composite data set is assumed to give best results even if this sampling strategy has also its uncertainty. To show the differences between the different sampling strategies we also calculated the percentage deviation of mean load estimations of sub-monthly and monthly data sets related to the mean estimation value of composite data set. The deviation ranges from -15% for the monthly data set in 2001 up to a maximum deviation of 82% for the sub- monthly data set in 2000. That points out again the importance of the choice of the used sampling strategy as well as the randomness of measurements, especially within time-based discrete sampling strategies.

Table 2. Results of nitrate-N load estimation in tons (t) using different equations (**Max; **Min, ***percentage deviation of mean load estimation values of sampling strategies related to mean load estimation value of daily composite data set)*

	(1) [t]	(2) [t]	(3) [t]	(4) [t]	Mean [t]	Max range [t] (percentage deviation related to the mean value)	Percentage deviation***
Year	**2000**						
Composite		141.5	148.8*	128.8**	**139.7**	20.0 (14%)	
Sub-monthly	246.5	243.2	307.9*	219.4**	**254.3**	88.5 (35%)	82%
Monthly	196.2	152.1**	175.5	242.5*	**191.6**	90.4 (47%)	37%
Year	**2001**						
Composite		91.9*	92.2	90.8**	**91.6**	1.4 (2%)	
Sub-monthly	124.1	142.7	167.8*	98.0**	**133.2**	69.8 (52%)	45%
Monthly	89.7*	71.0	70.0**	80.2	**77.7**	19.7 (25%)	-15%

3.2 Nutrient calibration

On the example of annual nitrate-N loads, we executed three separate calibration procedures based on mean estimated load values of each sampling strategy (Table 2). First we calibrated the model based on the result from daily composite data set because it is assumed to give best results. In Table 3 basic adjustments for nutrient (nitrate) output calibration are listed.

Table 3. Basic adjustments for nutrient (nitrate) output calibration

Parameter		Setting; description
FRT_KG	amount of fertilizer applied to HRU	based on farming scale modelling results
BIOMIX	biological mixing efficiency	0.1-0.4; depending on landuse and applied management practices
SOL_NO31	initial nitrate concentration in the upper soil layer	5.0 mg/kg soil; based on average Nmin of 20 kg/ha
SOL_ORGN1	initial organic nitrogen concentration in the upper soil layer	cropland: 100mg/kg; grazing land: 450 mg/kg; urban area: 180 mg/kg; pasture: 80 mg/kg; mixed forest: 150 mg/kg
GWNO3	nitrate in groundwater	25 mg/l
RCN	nitrogen in rainfall	10.0 mg/l; based on nitrogen deposition of 60 kg/ha/a and average, annual precipitation
RSDCO	residue decomposition coefficient	0.05

The different calibration bases resulted in different parameter settings (Table 4) to describe the same processes within the watershed.

Table 4. Selected nitrate output calibration parameter settings based on: mean estimated loads of: composite (S_com), sub-monthly (S_sm) and monthly (S_mo) data set

		Value range	S_com	S_sm	S_mo
NPERCO	nitrogen percolation coefficient	0.01-1.0	0.9	0.9	0.5
CMN	rate factor for humus mineralization of active organic nutrients	0.0001-0.003	0.0001	0.0017	0.0005
N_UPDIS	nitrogen uptake distribution parameter	0.0-100	100.0	1.0	50.0
FRT_SUF	fraction of fertilizer applied to top 10 mm of soil	0.0-1.0	0.4	0.2	0.2

Simulation results for annual nitrate-N load predictions as well as maximum, minimum and mean calculated load estimation results for all sampling strategies are illustrated in Figure 3. Based on different calibration data sets results met the ranges of maximum and minimum estimation except for 2001 with composite data set. In 2001 for the composite data set the nitrate-N load is under predicted. The minimal range of load estimation results for this dataset and year makes it quite difficult to achieve the absolute value. Furthermore, we assume the unsatisfactory water balance simulation for this year led to this result. But we should keep in mind that it is difficult to appropriately evaluate model performance if the uncertainty in model validation data is high. In that case, we do not really know how well the model is reproducing actual conditions.

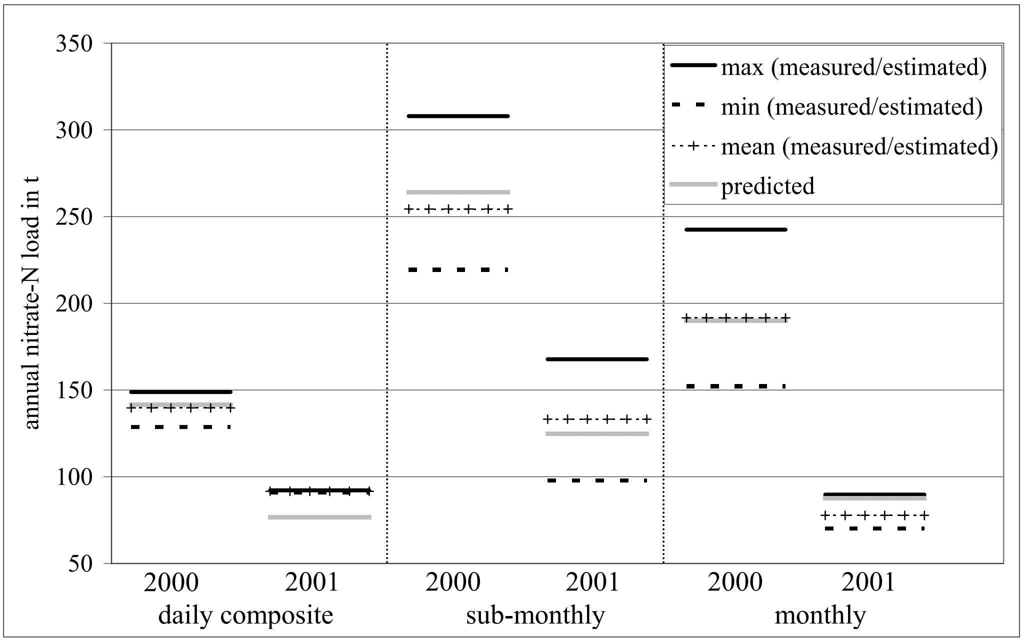

Figure 3. Results of annual nitrate-N load predictions

So load estimation qualities differs every sampling year depending on discharge conditions and randomness of sampling event related to discharge and concentration. This affects simulations of nutrient balance and makes the calibration process difficult.

CONCLUSIONS

For a satisfactory simulation of nutrient processes, a good calibration data base is required. In our case mean load estimation results using different sampling strategies and load estimation methods differ up to 82%, which can affect model evaluation conclusions. Hence, we postulate for the Parthe watershed the implementation of an intensive (daily) monitoring program to reduce measurement uncertainty and allow a more realistic judgement of model performance. Furthermore, we postulate the use of more than one load estimation method because of the fact that load estimations that uses only grab sampling methods are highly-uncertain. Therefore, if only grab sampled data bases are available we recommend the use of value ranges for model simulations. These value ranges should be used for discussion of the results and for suggestion of implementing best management practices.

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Considering spatial distribution and functionality of forests in a modeling framework for river basin management

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Abstract

This paper emphasizes the need of an improved consideration of the spatial distribution and functionality of forests in river basin management. The review of relevant papers has shown that forests, despite their frequent occurrence in temperate zones, play presently only a minor role in river basin management. In general, most of the studies highlight the positive effect of forests on water and nutrient fluxes in river basins. But hydrologists have also reported consistently flood events in or originating from forested areas. In context of the discussion on forest ecology and water quality it became obvious, that forest ecosystems can be sources depending on system properties, time and atmogenic pollution.

The simulation of land use changes on water yield in forested river basins has been carried out in a great number of research projects, but mostly without considering the spatial distributed function of forests. The objective of our work is thus to improve the consideration of spatial distribution of forests in river basins and its effect on water yield and water quality. The most promising approaches in the future are either spatial explicit models or integrated models with both improved forest modules and landscape positioning. The efficiency of these models could be proved by using virtual catchments. As a first conceptual approach towards the base concept of a virtual catchment, we propose a five-units-model (FUM), representing cross-sections with typical land use sequences. The basic idea of our model is to identify major process units and to implement them in the river basin modeling and management.

Keywords: Land use pattern; Forests; River basin; Management; Modeling

1. Introduction

The need of a meso- to macro-scale (for definition see Volk and Schmidt, 2004; Jessel and Jacobs, 2005) based approach in river basin management has emerged during the last decade for example in the context of the coming into effect of the European Water Framework Directive (WFD). Large parts of river basins in the humid and temperate zones of Europe are covered by mostly managed forests (average for Germany approximately 30% [2004]). Surprisingly forests and forest management are *not* explicitly mentioned in the regulations of the WFD, neither as alternative land use nor as potential sources of contaminants—as opposed to industrial activities or agriculture. However, from the author's point of view good reasons exist to integrate forest management into river basin management:

- Forested areas can cover large parts of river basins (especially headwaters).
- Depending on their location and extent, forested areas can contribute considerably to water quality protection especially in landscapes with a low proportion of forests and with the status ''unclear/improbable to meet the goals of WFD''.
- Forest ecosystems have large receptor surfaces causing increased atmogenic inputs, in combination with low retention capacities forest ecosystems are likely to turn from sink into source areas under changing environmental conditions.

Following the general idea of the WFD forests have to be managed in such a way they improve or at least sustain water quality, as stated by the *avoidance rule* of the WFD (Section 1, paragraph a). Therefore, and for reasons of cost efficiency (Krecmer and Perina, 1987) the development of common strategies in river basin and forest management is necessary (LAWA, 2002).

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A crucial point in a combined forest-water-management will be the functionality of forests. This can be defined (1) as the general effect due to properties of forests itself and (2) by their geographic location (spatial distribution) in river basins.

The central point of our contribution is to stress the need of research on the interrelation between location and function (connectivity of processes) on the river basin scale. We will focus on the question *how can the spatial distribution of forests be considered in a modeling framework of river basin management to improve and sustain water quality and quantity in the long run?* We follow a stepwise approach in which we highlight the role of the location of forest ecosystems in river basin management. However, the complexity of forest ecosystems caused by management and natural succession is of high importance, but has to be neglected in that first step for reasons of simplification.

2. The general role of forests in controlling water and nutrient fluxes in river basins

Recent research on the role of forests in river basins has focused mostly on their function as protective forest or as an option for land use change (afforestation of agricultural land, etc., e.g. Fohrer et al., 2003; Wegehenkel, 2003) to improve the water quality of river sections or to manipulate (e.g. reduction of surface runoff) the discharge in river basins. Other studies are dealing with the management and function of forests in smaller water protection areas or the interaction between forest management and pollutant input into stream waters or reservoirs (e.g. Lorz et al., 2003).

Even in the excellent overview of spatial forest planning by Baskent and Keles (2005) the term *watershed* is mentioned only in one sentence. Thus, the inclusion of forest functions is considered only insufficiently for entire river basins—neither the possibility to improve water quality by the variation of the spatial configuration of forests nor as a potential source of nutrients and sediments. Even in the outstanding book *Integrated Watershed Management* by Heathcote (1998), for instance, *forestry* appears only at p. 62, where she points out that *probably forestry, including logging and replanting of trees, agricultural practices such as tillage, planting, harvesting, and drainage works, and construction activities* affecting water resources in a watershed. Nonetheless, watershed hydrologists (e.g. Gravelius, 1914; Black, 1996) and forest hydrologists (e.g. Chang, 2003; Lee, 1980) have recognized the importance of forests for the water and matter cycle for a long time. Chang (2003) states that the function of a specific forest is *more significant than others for a particular forest because of its location and environment*. Although, to our knowledge there is no general approach to account for the importance of the geographical location of forested areas for water protection in river basins; most research is carried out without reference to the geographical location of forests. Planned afforestations for the purpose of increasing water quality are rare. Existing afforestation programs take mostly economic causes into account, where the geographic position is determined, for instance, by the low quality of soils (marginal land). The banks of rivers or water reservoirs were occasionally planted with trees to prevent direct inputs (e.g. Lowrance et al., 1997).

In general the influence of forested areas on water quality and quantity is considered positive (*water protection function*, Lee, 1980). The reasons are (1) more uptake of nutrients and ions by plants, (2) fewer runoff and sediment loss, (3) lower rates of organic-matter decomposition and microbial activity, (4) cooler temperatures, and (5) less management (Chang, 2003). Thus, a large number of rivers showing the reference conditions according to the WFD (Appendix II, 1.3) with a good ecological status are located in forested areas.

Several studies in forest ecosystem research show values for nitrogen leaching up to 62 kg N ha^{-1} a^{-1} (DVWK, 1990; UBA, 1995; Akselsson and Westling, 2005; Van der Salm et al., 2006). Despite the often reported trend of declining nitrogen deposition, one cannot expect a general reduction of the nitrogen leaching from forested areas (Aber et al., 1989, 1998; Gundersen, 1995; Schmidt et al., 2001). Simulated scenarios in the Torgau district in Saxony (Germany) have shown a considerable increase of nitrogen leaching into groundwater under high atmospheric nitrogen deposition (Franko et al., 2001). Future increases in nitrogen leaching under forest (Aber et al., 1989, 1998) could lead to an attenuation of the positive effects of adapted agricultural cultivation practices such as integrated and organic farming, especially in river basins with large forested areas. Consequently, Bastrup-Birk and Gundersen (2004) do not recommend afforestation where N-input exceeds N-demand, because water quality will not improve. Reports on surface water acidification (e.g. Likens, 2004; Ulrich et al., 2006) and on the implementation of the WFD (e.g. Lorz et al., 2005; Meesenburg et al., 2005) are questioning the high quality of water from forested catchments, because of increasing concentrations of DOC, nitrogen, and acidification related pollutants.

The effect of forested versus non-forested areas on the water cycle such as a smaller volume of runoff, lower peak flows and broader time base is widely accepted (Chang, 2003; Lee, 1980; Lewis et al., 2001). It is based largely on the comparison of clearcut catchments versus non-clearcut areas. This observation has lead to the widespread assumption *that forest can reduce flooding downstream and that they are thus a "natural" solution to flood problems* (e.g. Robinson et al., 2003; Lee, 1980 for an extended discussion). However, recent studies have shown that the reduction effect on storm flows for most of European forests is smaller than thought (Robinson et al., 2003; LWF, 2004). Obviously geographical location in combination with site conditions are the controlling factors, since forest ecosystems can be found most times in areas with steep slopes and shallow, stony soils having low retention capacities compared to sites in the lower reaches of river basins. An additional factor in amplifying storm flows from forested catchments is the direct runoff-increasing effect of forest roads (Bowling and Lettenmaier, 2001; Wigmosta and Perkins, 2001).

The fact that forests are substantial water consumers due to greater evapotranspiration carries mostly no weight in the discussion since the uplands in the temperate zones used for water collection receive high amounts of precipitation. The consequences for groundwater recharge and thus baseflow (especially dry weather flow) will only occur in the lowlands with semi-arid summer periods, but can be of considerable ecological and economic relevance (Chang, 2003; Lee, 1980; Robinson et al., 2003; Wegehenkel, 2003) and might be a future problem under changing climate with considerable lower precipitation.

2.1. Simulated land use change and water yield in forested river basins

Usually variants or scenarios are investigated that are for instance based on assumptions concerning climatic change or the impact of political decisions (Table 1). With afforestation scenarios, the results show mostly a reducing effect on discharge (Table 1). The results of a model-based simulation of an afforestation scenario (increase of the forest cover from 34% up to 80%) in a catchment in North-East Germany showed a decrease in discharge of the order of 5–48% with the average of 24% and an increase of evapotranspiration on the order of 3–31% with an average of 14% with the actual land use (Wegehenkel, 2003). The decrease in extreme peak discharges due to the afforestation scenario was of the order of 4–5%. The results show also that if balancing is performed over the entire study area, only very pronounced land-use changes will result in any considerable shifts in the simulated total run-off. Bosch and Hewlett (1983) pointed out that changes in the proportion of forest cover of less than 20% are unlikely to be detected by flow gauging. Wegehenkel (2003) used a drastic land use change scenario for the first test of the conceptual hydrological catchment model THESEUS. What Table 1 fails to show are any local changes, which in certain areas may considerably exceed calculated mean values.

2.2. Spatial distribution and function of forests in river basins

A general assumption in watershed hydrology (Black, 1996), hillslope hydrology (Anderson and Burt, 1990) and landscape geochemistry (Litaor, 1992; Perelman, 1972; Sommer, 2002) is that the magnitude and direction of fluxes of both water and matter is a result of the passage of water from the watershed boundaries to the water body, i.e. through the landscape.

If forested areas are seen as reaction units changing incoming water and matter flow, then, apart from site characteristics (tree species composition, age, management practices, etc.), the geographic location in relation to the water body plays the major role in the function of forests in river basins. A key parameter is the connectivity of a forested area to the water body. Changes of both water and matter fluxes will occur at interfaces, when the properties of ecosystems are substantially different from the upslope system. Such changes are often caused by

- alternating surface roughness and impacts on the overland flow (e.g. created by contrasting land use);
- geochemical changes occurring at the interface between aerobic and anaerobic conditions (e.g. slope drainage versus riparian zone) or neutral and acidic conditions (e.g. calcareous versus non-calcareous soils).

Possible effects of forests with a special geographic location to be used in river basin management in context of spatial distribution of forest are:

- *Water quantity.* Some hints on the effect of different locations of forest in river basins on the water yield can be found in Chang (2003). He states that small openings in upper slopes can cause a smaller impact on water yield than openings in lower slopes (Chang, 2003). The effect of afforestation will be most effective in locations where connectivity and water retention capacity is optimal. But, these sites will not normally be the flood generation areas.
- *Water quality.* The best example for the effect of forest location in river basins is the relevance of near stream zones (NSZ). Effects used in river basin management can be a reduction of sediment yield and sediment production along the stream in specific streamside management zones or BMZ (best management zones) (Chang, 2003). A positive effect of these zones regarding water quality is observed: water of higher quality enters the surface water compared to watersheds without such zones. For non-particulate dynamics biogeochemical processes in the riparian zone (RZ) are to be emphasized. Examples are the microbial reduction of sulfate in the RZ causing sulfate immobilization and consequently

Table 1
Case studies on land use change and water yield

Region	Scenarios in the case studies	Land-use change	Simulated change in total water yield (%)	Reference
NE-Germany	EU agricultural reform	Change of 4% AL into forest, change of 32% AL into forest	−1, −10	Werner et al. (1997)
Hessen (Germany)	Agricultural policy: grassland premium	Forest: 42 → 13%, AL: 44 → 73%	+8	Fohrer et al. (1999)
Hessen (Germany)	Agricultural policy: cessation of animal husbandry	Forest: 42 → 49%, AL: 44 → 37%	+2	Fohrer et al. (1999)
Sachsen-Anhalt (Germany)	Analysis of usage conflicts in priority areas (agriculture vs. groundwater protection)	Change of 10% AL into forest, *Köthener Ackerland*, *Nördlicher Mittelfläming*	−9, −2	Volk and Bannholzer (1999)

AL, arable land.

lower H^+ and Al^{n+} concentrations in stream water (Fitzhugh et al., 1999) or denitrification (Burns, 1998; Hill et al., 2000; Meißner et al., 2006). Additionally, the dependence of denitrification rates on water table elevation (Hefting et al., 2004; Machefert and Dise, 2004) implies the importance of local and global environmental changes especially in systems oversaturated with N or close to N oversaturation.

We identified four typical patterns of forest location in Central Europe depicted in cross-sections, each representing a characteristic land use pattern (Fig. 1). A standard situation is that headwater areas are forested due to the low agroeconomic value of their shallow soils (cross-section a), whereas the lower elevations of the watershed are occupied by soils with higher quality, mainly due to the occurrence of loess or loess like deposits. Therefore a common situation for the lower reaches is the predominant agricultural use (cross-section b).

Forested patches are quite common in areas with mosaic geologic underground and patchy soil geography, e.g. outcrops of hard rock, glacial sediments with high variability (cross-section c), often connected to small hills or ridges surmounted by plains with highly productive soils. Another common feature are forested riparian zones in small headwater courses of the loess regions, where the valley floors are to small and the bank slopes are to steep for agricultural use (cross-section d).

3. The role of forests in river basin modelling—state of the art and research needs

We see an imperative in river basin management to quantify the protective function of forests for water by simulating land use scenarios with different land use configurations. Approaches are needed to investigate what the potential of forests are in the water and matter budget of landscapes that can be used to achieve the objectives of the WFD. The valuation of these potentials is necessary, since the economisation of environmental goods and services of forests will play a major role in the future discussion of river basin management. For example, the monetary benefits of forests for flood protection were recently estimated by LWF (2004).

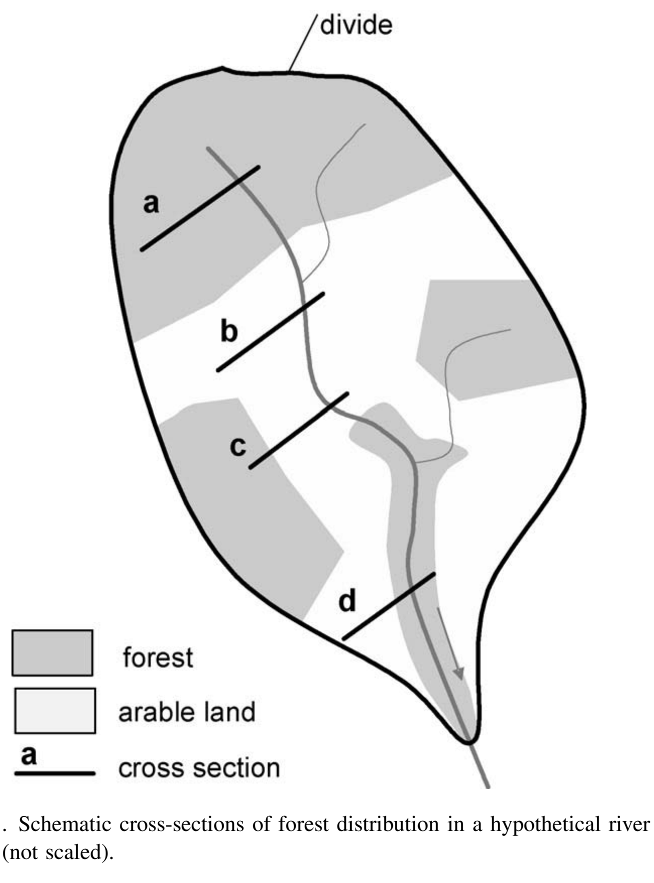

Fig. 1. Schematic cross-sections of forest distribution in a hypothetical river basin (not scaled).

It is important to investigate the potential of forests to modify the water and matter cycle relative to their landscape position in order to achieve the objectives of the WFD. A successful implementation of the WFD requires appropriate mathematical models and other tools to manage the different phases of the planning procedure and to support decision making in various steps of the implementation processes (EUROHARP, 2004; Horn et al., 2004). During the last three decades, computer-based model systems have been increasingly used for the investigation of impact of land use changes on water quantity and quality in river basins of different sizes. Examples for such models are HSPF (Johanson et al., 1980; Bicknell et al., 2001), AGNPS (Young et al., 1987), HBV (Bergström, 1976, 1992), SWAT (Arnold et al., 1993; Neitsch et al., 2002), MIKE-SHE (Refsgard, 1997), or SWIM (Krysanova et al., 1998). Overviews of different models are given in Volk and Steinhardt (2001), Krysanova and Haberlandt (2002), Horn et al. (2004), and Arnold and Fohrer (2005).

There is still a need of a better spatial process description with these (mostly ‘‘integrated’’) models on the catchment scale. Such an improvement would enable to better estimate the impact of the spatial distribution of land use and land cover – including forests – on water quantity and quality. In addition, in most of the cases these models have only very simple approaches (or adapted crop modules) to describe the physical and biological properties of forests. Arnold and Fohrer (2005) stated that current research is, beside landscape position and others, focused on the improvement of the forest growth module in SWAT with (i) the leaf litter layer, (ii) growing trees from seedlings to a mature stand and (iii) simulating the tree canopy and ground cover simultaneously. A potential way to improve this situation would be to test selected integrated models (with and without such improvements) on their response on both different spatial configurations of forests and different forest types. Such tests could be carried out on the example of so-called artificial or virtual catchments (Jetten et al., 1996; Eckhardt et al., 2003; Volk and Schmidt, 2004; Raat et al., 2004), as it is shown in the next section.

A crucial point is how detailed integrated models on the landscape scale have to describe spatially the forest-related processes that influence water quantity and quality. With the application of integrated models in river basin management, it may be in most cases not necessary to have the same complexity level as forest management models such as PnET-CN (Aber and Driscoll, 1997), BALANCE (Grote and Pretzsch, 2002) or COUP Model (Jansson and Karlberg, 2004) with highly sophisticated and elaborated growth and nutrient cycle algorithms. On the other hand, the relevant processes should not be too simplified and thus probably neglect some important influences and interactions. This is also true for the promising landscape modeling frameworks as tools to support ongoing watershed planning and adaptive watershed management efforts. These are mostly modular, multiscale approaches to build, calibrate and test models. The model approach of Costanza and Voinov (2004) for instance incorporates an ecosystem-level unit model that is replicated in each of the unit cells representing the landscape. The model builds on the format of a raster-based geographic information system (GIS) that is used to store all the spatially referenced data. Horizontal fluxes link the cells together across the landscape to form the full landscape model. These spatial fluxes are driven by cell-to-cell head differences of surface and ground water in saturated storage. Water fluxes between cells carry dissolved and suspended materials and determine water quality in the landscape. The modular structure is important for flexible model adjustments and scaling experiments; that could be an option for testing different forest growth, water balance, and nutrient cycle algorithms in a landscape modeling framework. Mladenoff (2004), for instance, used such a raster-based model (LANDIS) to simulate forest landscapes, including succession and wind and fire disturbance that operate spatially. He stated that *"future goals include integration within a larger land use change model, and applications to landscape and regional global change protection based on newly incorporated biomass and carbon dynamics"* (Mladenoff, 2004). Another promising approach under the actual technical restrictions is the disaggregation of spatial model units as shown by Fohrer et al. (2005) and Haverkamp et al. (2005). This could be combined with the stepwise improvement of the forest modules in these integrated models.

4. Sensitivity of simulation models on different land use configurations by using virtual catchments

Volk and Schmidt (2004) carried out a relative comparison of selected models (NASIM 3.10, Hydrotec, 2001; ArcEGMO 2.3, Pfuetzner et al., 2001; SWAT 2000, Arnold et al., 1993, 1998; Neitsch et al., 2002; ABIMO 2.1, Glugla and Fuertig, 1997) with different scale-specifics on the base of virtual catchments with variable differentiation levels. Fig. 2 shows the principle of increasing complexity with the development of virtual catchments. This method provides information about the model sensitivity and specifics and the variations of the results with different land use configurations. In this context results of simulations can be related to "real" data in a stepwise approach. Volk and Schmidt (2004) compared the simulation results for the virtual catchments as base for the identification of the scale-specific relevance of parameters and applicability of their above-mentioned selected models. Volk and Schmidt (2004) used selected hydrological parameters (e.g. mean, minimum and maximum runoff, mean monthly runoff) for the relative model simulation comparison. The method allows it to

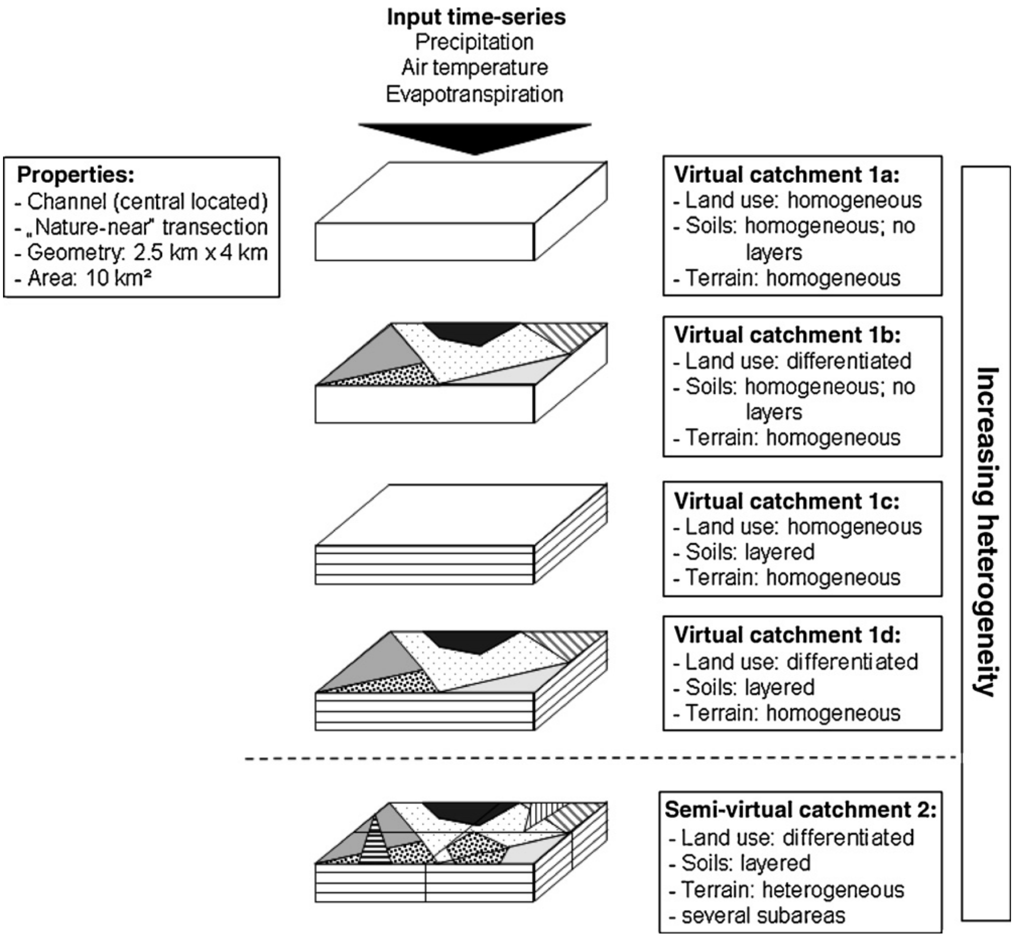

Fig. 2. Principle of the development of the virtual catchments for investigating both model sensitivity and land use impact.

turn out the indicators which could be used later for data transfer between different scales. Fig. 3 shows the simulation results on the example of total runoff, calculated with datasets of simple virtual catchments.

The comparing simulations show similar results for all models: in general, forest scenarios or scenarios with a high proportion of forest show lower runoff values than the others. The simulation results of SWAT and ArcEGMO show in the most cases only small differences. The differences of the simulated mean runoff are in general less than 10%. Only for the scenarios ''arable land'' the differences rise up to 20%. This is caused by the different parameterisation possibilities of the models for the agricultural characteristics. SWAT has the most options to describe agricultural cultivation practices by parameters. In order to enable a comparison to the other model systems, Volk and Schmidt (2004) had to simplify the parameterisation of agricultural land use that has the additional concern of fertilization controlling the plant growth and seasonal evapotranspiration. More detailed parameterisation could lead to results with higher accuracy.

The simulation results of the conceptual model ABIMO and the physically based model NASIM show greater differences (up to 30%), which is mainly caused by the different temporal resolution of the precipitation and evapotranspiration data. Those results are the first step for the determination of the expected variations and uncertainties of the simulation by using different spatial configurations of soil and land use.

Our future steps in working with virtual catchments will include simulations using different model types (integrated models with improved forest modules, such as the soil and water assessment tool [SWAT]; or spatially explicit models such as the spatial modelling environment [SME]), different forest distributions, vertically differentiated soils, and a larger ''semi-virtual catchment'' (see Fig. 2). In that context ''semi-virtual'' means that a digital elevation model (DEM), a river system and a soil classification of a real catchment is used, but the land use scenarios are created in a synthetic way. This method enables us to get better information about the effects of input parameters on the simulation results in terms of a sensitivity analysis. Additionally, the method could be used for a stepwise approach of the simulations of more complex scenarios within real study areas.

5. A conceptual approach—the five-units-model

An important aspect in using virtual catchments is the analysis of the landscape structures and their effect on the connectivity of processes in river basins. As a part of the virtual catchment concept, we developed the five-units-model (FUM).

We used existing approaches (e.g. cascade storage model from hydrology, Dyck and Peschke, 1995 or nine-units-surface-land-surface model from geomorphology, Conacher and Dalrymple, 1977) to create a five-units-model simplifying the complex passage of water through the catchment by using cross-sections. The cross-sections are basically a composition of compartments (ecosystem-level units) representing three major units (1) the slope hydrological system [slope shoulder, upslope, footslope], (2) the riparian zone, and (3) the channel itself.

The FUM is designed as a conceptual model of fluxes in meso-scale river basins. The basic assumption of the model is that fluxes of water and matter are directed by surface and subsurface relief. Detailed land use and regolith are the factors determining fluxes and retention. Flow paths connecting the compartments are either subsurface lateral passage through soil and regolith or overland flow. Compartments are reaction units

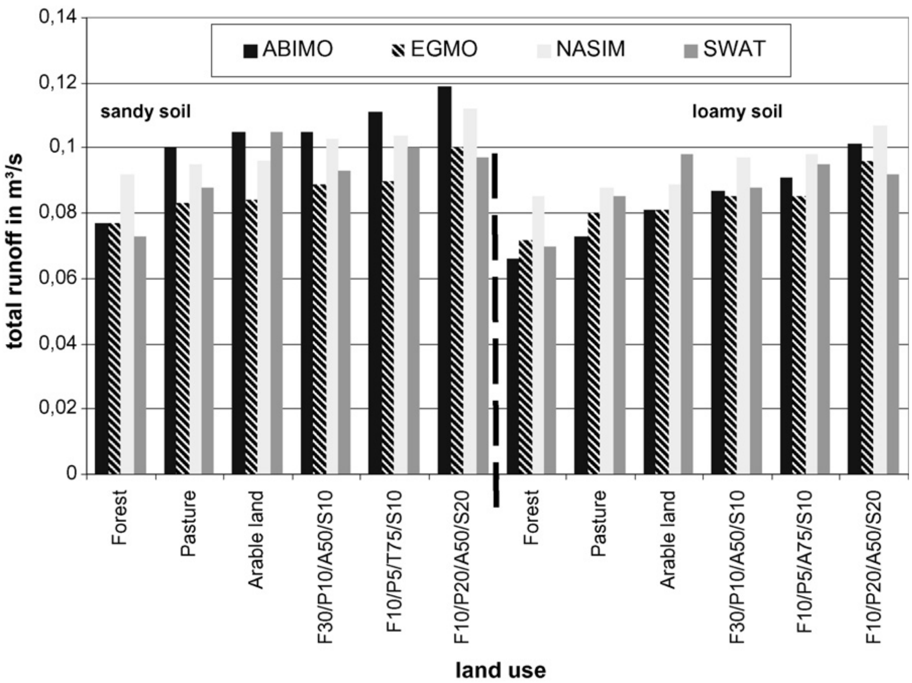

Fig. 3. Comparison of simulated mean runoff in an artificial area, calculated with ABIMO, ArcEGMO, NASIM and SWAT. F, forest; P, pasture; A, arable land; S, surface sealing. The numbers behind the letters represent the proportion of catchment area covered by the land use type in %.

of a more or less uniform soil, regolith and land use composition of topic dimension. The cross-sections are basically sequences of units representing a two-dimensional sector of the river basin. Fig. 4 shows the standard situation for Central Europe in the FUM (see Fig. 1 for location of cross-sections). The connectivity of the slope hydrological systems varies widely. In contrast the riparian zone is an all-time buffer (near-stream processes) towards the channel, which itself acts as reaction unit (in-stream processes).

In a uniform arable land section overland flows are dominating (Fig. 4[1]). All water bound transport systems of particle and dissolved matter are depending on this. The connectivity of all compartments is comparably high because of the high partition of overland flow. Therefore, the retention capacities for water, sediment, and elements bound to particulate matter during storm events are rather low. Although, lateral flows can have an importance in agricultural areas under certain conditions (steep slopes, coarse soils). In a landscape with an uniform forest cover (Fig. 4[2]) subsurface water flow paths are prevailing.

In the cross-sections c and d mixed land uses are shown. While in Fig. 4[3] a typical setting for the lowlands with forested outcrops of hard rock is given, Fig. 4[4] presents a forested buffer in the riparian zone in otherwise agricultural used landscape (typical for loess landscapes). The buffer prevents pollutants and sediment from entering the stream by using mechanical effects (sediment traps) and geochemical effects due to changes in pH (immobilization by adsorption or precipitation) and/or redox potential (immobilization or volatilization by reduction).

A second aspect of an approach on the dynamics in river basins is the in-stream-processes along a longitudinal section. Immobilization and buffering processes can occur, if stream waters flow through contrasting partitions of the catchment, e.g. outflow from a forested part and inflow in agricultural part.

The FUM approach focuses on the optimization of forest location with regard to an integrated water resource management. A crucial point in developing the approach will be the inclusion of forest properties and forest management. A first step is the integration of tree simulation in large river basin models (e.g. McDonald et al., 2005). The consideration of forests of differing age, density or species composition will be a further improvement of our model framework in the future. The combination of adapted tools to simulate forest growth and spatial explicit models will be the biggest challenge to realize our approach finally.

6. Conclusions and outlook

The function of forests on the landscape scale has to be better integrated in river basin management strategies. In the context of the WFD, we have to attach more importance on the protection of long-term renewal and supply of usable surface water than on the economic capacity of forestry. However, the

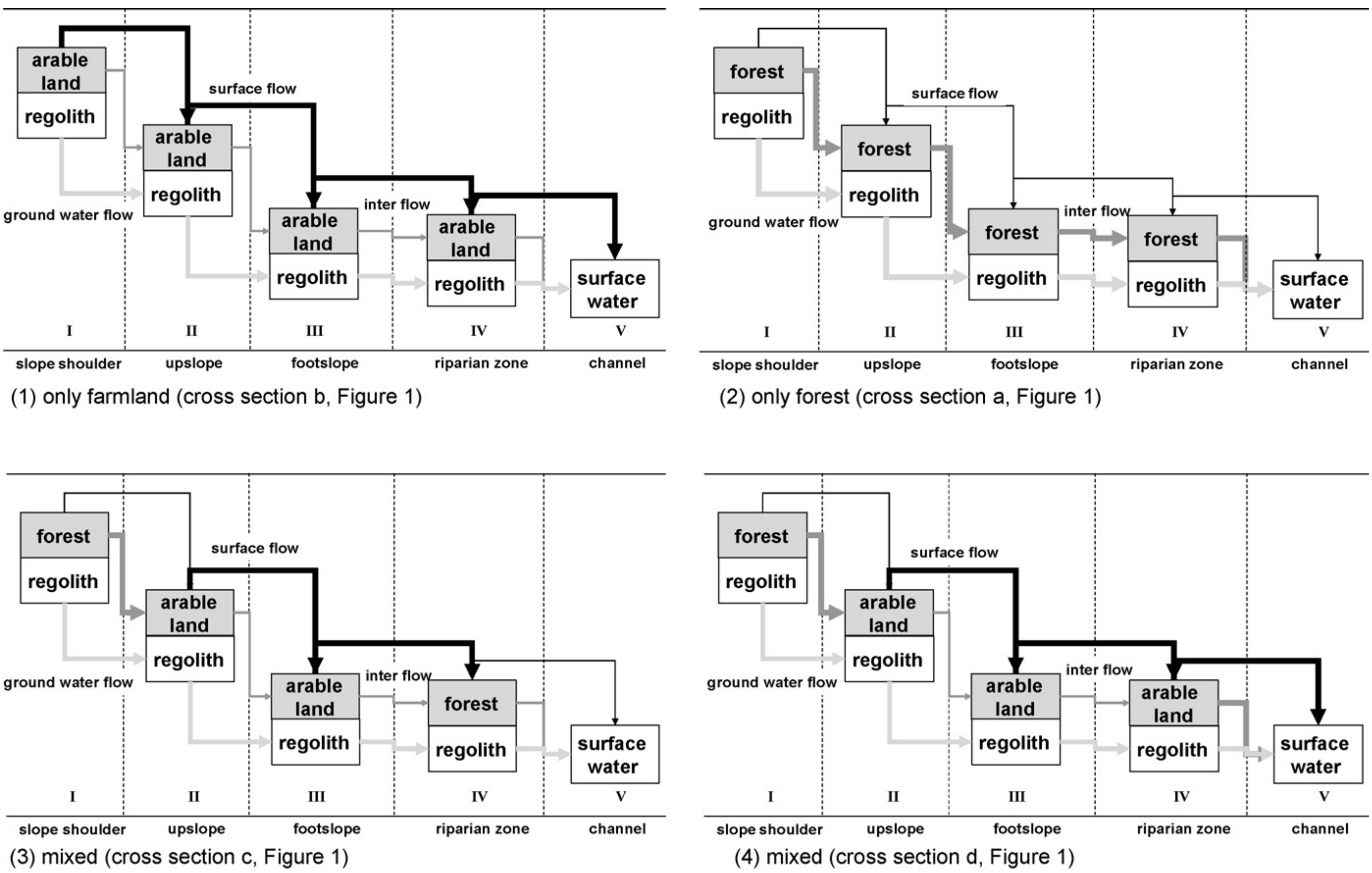

Fig. 4. The five-units-model.

potential to influence the water quality by adapted forest management are not unlimited.

Despite a good understanding of single processes in forests, knowledge gaps exist about the role of forests in large river basins, especially with respect to the spatial distribution of forests (connectivity of processes). We identify four objectives for our future work on this issue:

- Defining important properties of forests that we can use to manipulate water quality and quantity in river basins.
- Defining the best location or spatial distribution regarding the connectivity and site conditions for forests in river basins with the maximum benefit for water quality and quantity.
- Defining the potential risks of oversaturation of pools (sink to source) and by changing environmental conditions.
- Improvement of conceptual and numerical models to simulate the effect of different forest locations and their effect on water quality.

The prognosis of river basin scale dynamics and effects of land-use scenarios is an actual challenge for river basin management. The impact of forests and their location could play a major role in regards of financial as well as ecological aspects.

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SWAT Revisions for Simulating Landscape Components and Buffer Systems

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Abstract. *Methods for simulating different landscape positions within the SWAT model are being examined. A three component system, consisting of the watershed divide, the hillslope, and the floodplain landscape positions, has been developed to address flow and transport across hydrologic response units prior to concentration in streams. The modified SWAT model is capable of simulating flow and transport from higher landscape positions to lower positions within a single river basin. The revision was developed to address variable source areas within watersheds and stream-side buffer systems which exist alongside many streams. The enhanced model will allow for more accurate simulation of natural transport processes within a hillslope. The revision was tested using data collected from a low-gradient watershed near Tifton, Georgia, USA which contains heavily vegetated riparian buffers. The modified model provided reasonable simulations of surface and subsurface flow across the landscape positions without calibration. The application demonstrates the applicability of the model to simulate filtering of surface runoff, enhanced infiltration, and water quality buffering typically associated with riparian buffer systems.*

Keywords. Watershed modeling, natural resource modeling, surface hydrology

Introduction

Watershed models are valuable tools for examining the impact of land-use on hydrology and water quality. While extensive research has been done to describe the impact of management practices on field and farm runoff, less is known about how these changes are reflected at the watershed scale. The success of the Total Maximum Daily Load (TMDL) program will be based on water quality improvements that result at the watershed scale. Additionally, a national assessment of the effects of conservation practices on watershed scale water quality is underway which relies heavily upon the reliability of watershed flow and transport models (Mausbach and Dedrick, 2004).

The SWAT model has been applied to watersheds throughout the world (Arnold and Fohrer, 2005). The model has received extensive testing in Texas (Saleh et al., 2000; Santhi et al., 2001; Srinivasan et al., 1997), Kentucky (Spruill et al., 2000), Wisconsin (Kirsch et al., 2002), Mississippi (Bingner, 1996), Indiana (Smithers and Engel, 1996), Pennsylvania (Peterson and Hamlett, 1998), and Georgia (Bosch et al, 2004; Van Liew et al., 2006). In most cases, the prediction accuracy was satisfactory to obtain working knowledge of the hydrologic system and the processes occurring in the watersheds. One of the shortcomings of SWAT has been an inability to model flow and transport from one position in the landscape to a lower position prior to entry into the stream. The model utilizes a Hydrologic Unit Area (HUA) concept which combines a unique combination of land-use and soil type within a defined subbasin. Transported water, sediment, and chemicals from the HUA are routed directly into the stream channel. As currently configured, SWAT does not allow transport from upslope HUAs to flow through lower landscape position HUAs prior to entry into the stream.

Arnold et al. (2007) have developed a modification to the model which facilitates such a process (SWAT-L). The modification divides the catchment into three units, the upland divide, the hillslope, and the floodplain (Fig. 1). The modified model routes surface runoff, lateral subsurface flow, and shallow ground water flow from the divide, through the hillslope, through the floodplain, and eventually to the stream. Additional details are provided by Arnold et al. (2007).

The objectives of this manuscript were to test SWAT-L for a small watershed in South-central Georgia. Simulations were conducted which incorporated the landscape routing and a comparison made to observed data.

Methods

Site Description

A site located near Tifton, GA was selected for this research (Fig. 2). The study site consists of two paired watersheds, 57 and 47 ha (Table 1), which join to form a larger watershed (123 ha). Soils in the watershed consist of loamy sands, with Tifton loamy sand (Plinthic Kandiudults; fine loamy, siliceous, thermic) being the dominant soil type in the upland (Calhoun, 1983). The Tifton soil contains subsurface horizons with reduced infiltration rates which perch water and initiate lateral flow during wet conditions (Hubbard, 1983). The Tifton soil contains 7-14 % plinthite from 0.8 to 1.4 m. Over the year, the shallow aquifer water-table varies from 0 to 7 m below the ground surface, depending upon landscape position. The watershed contains dense riparian buffers in the flood plain. The dominant soil type in the flood plain is an Alapaha loamy sand. The uplands consist of tilled fields and some forest (Fig. 2).

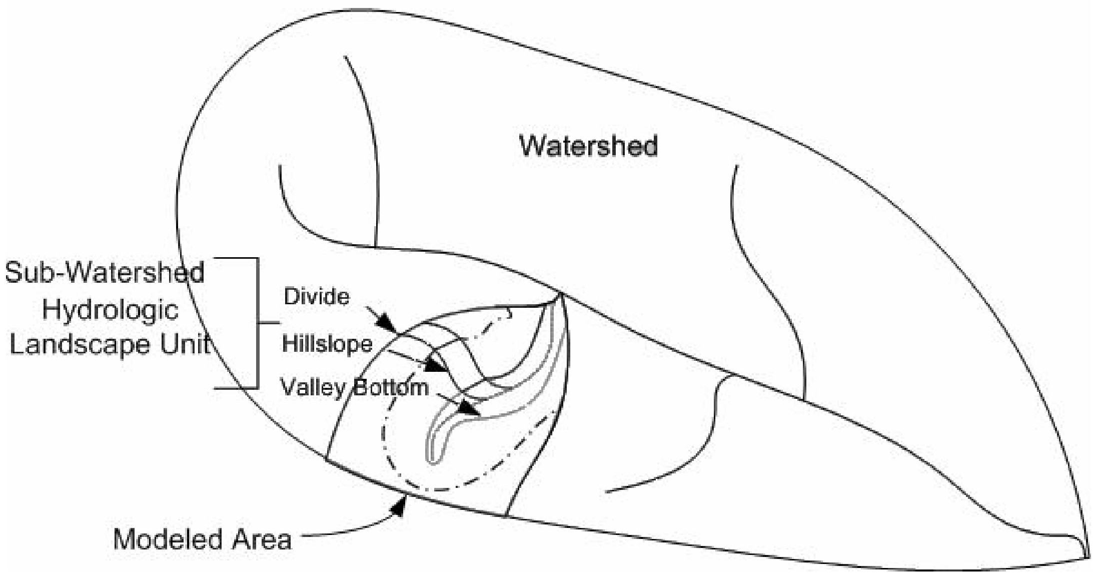

Figure 1. SWAT-L subwatershed landscape delineation within a watershed.

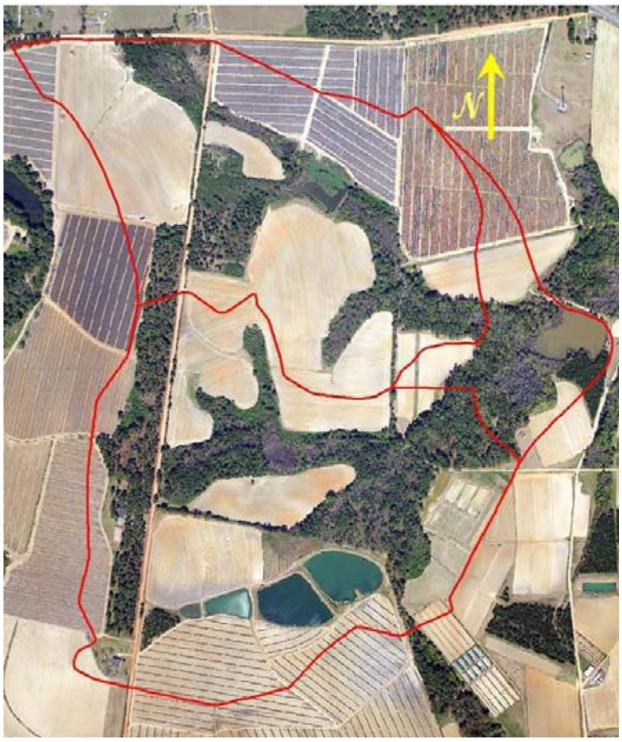

Figure 2. Gibbs Farm Watershed, Tifton, Georgia, illustrating north and south branch watersheds.

Table 1. Land-use for the studied watersheds.

Land-Use	Gibbs South ha (%)	Gibbs North ha (%)
Ponds	3.2 (5.5%)	0.8 (1.7%)
Fields	28.9 (50.6%)	32.0 (68.4%)
Forest	24.2 (42.4%)	13.5 (28.9%)
Roads	0.8 (1.5%)	0.5 (1.0%)
Total	57.1 (100%)	46.8 (100%)

The North and the South basins join and eventually flow into the outlet pond (Fig. 2). Both watersheds are instrumented with weirs within the streams for streamflow measurement. Hydrologic and water quality data were collected in the watershed from October 1996 through November 2004 (Lowrance et al., 2007).

Both basins include stream reaches bordered by mature riparian forests (Fig. 2). Upland areas within the watersheds are tilled. Row crops grown include corn, peanuts, cotton, and vegetables. Most of the field borders between the upland fields and the riparian forests are grassed and used as turn-around areas for farm implements. The grassed areas vary from 5 to 20 m in width. The riparian buffers vary in width from 20 to 100 m.

SWAT-L simulations

The Gibbs Farm Catchment was manually configured for SWAT-L as shown in Figure 3. The simulation was established for one subbasin and three landscape units (divide, hillslope, and flood plain). One HRU was simulated for each landscape unit. A transect through the Fox Den Field at the University of Georgia Gibbs Farm was simulated (Fig. 4). This site has been extensively studied, particularly the riparian buffer (Bosch et al., 1996; Lowrance et al., 1997). The field drains into the lower part of the Southern basin (Fig. 2). Corn, peanuts, and cotton have been grown in the field. There is a grass edge downslope from the field and a woody riparian buffer between the grass buffer and the stream.

SWAT-L input datasets were developed for the watershed and landscape units using the landscape unit configuration shown in Figure 3. Three landscape positions and vegetation types were simulated. A peanut / cotton rotation was assumed in the upland divide, a bermuda grass in the hillslope, and a pine forest in the floodplain. A catchment area of 10 ha was simulated. The divide corresponded to 70% of the area or 7 ha with a slope of 3.0%. The hillslope made up 10% of the area or 1 ha with a slope of 2.4%. The floodplain made up 20% of the area with a slope of 2.0%. The upland and the grass buffer were simulated with the soil type of a Tifton loamy sand while the riparian buffer was simulated with a soil type of a Alapaha loamy sand. Rainfall data were obtained from an on-site recording rain gage while climate data were obtained from a nearby weather station (University of Georgia, 2007). A 5 year simulation was conducted using observed rainfall.

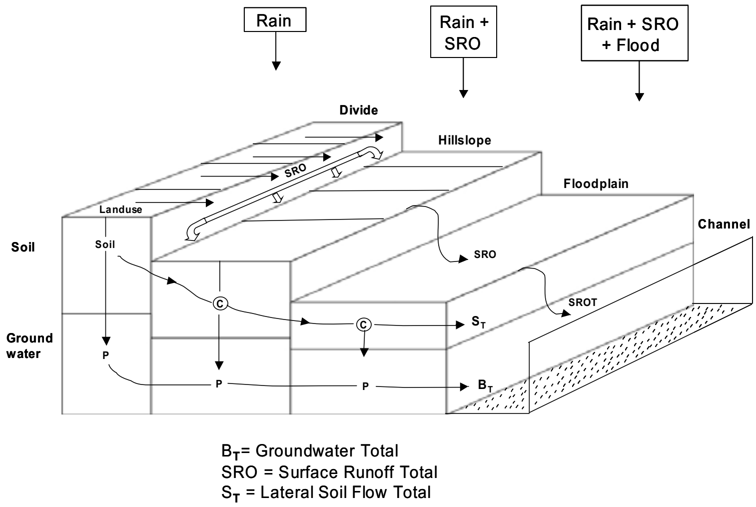

Figure 3. Processes considered in landscape routing units.

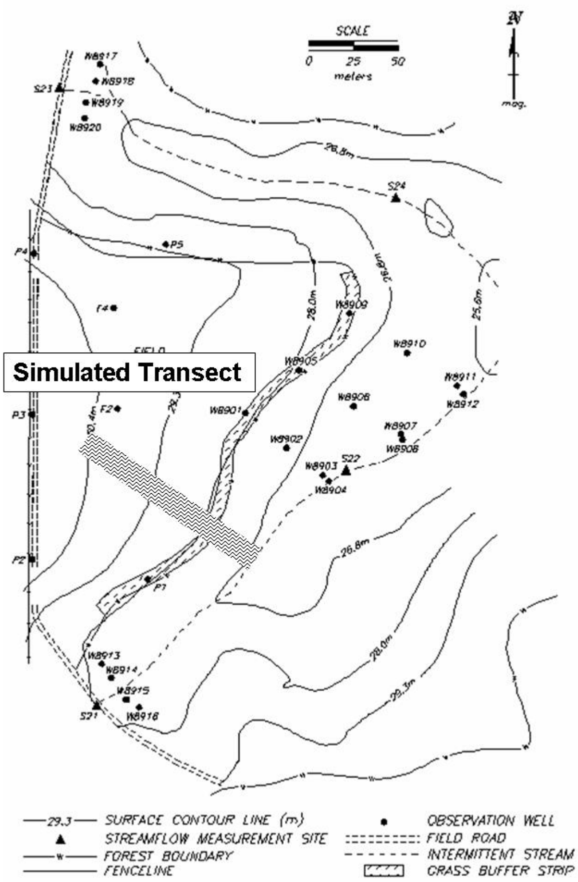

Figure 4. Simulated SWAT-L transect through the Fox Den field.

Results

The summary results for the five year simulation are shown in Table 2. The average annual simulated precipitation for the five year period was 1298 mm. The total water yield on a per area basis for the simulated catchment was 355 mm or 27% of the annual precipitation. Evapotranspiration for the catchment was 702 mm, or 54% of the annual precipitation, while transmission losses from the plants accounted for 16 mm.

The water balance for each of the individual landscape units was also calculated (Table 2). Each component is on a per area basis using the area of that individual landscape unit. The largest contributor to flow from the upland divide was the surface runoff, while the largest contributor from the floodplain was the groundwater (Table 2). On a per area basis, surface runoff within the upland and the hillslope were roughly equivalent although the volume was significantly greater from the upland due to the larger upland area. Surface runoff within the floodplain was only 61% of that in the upland despite the contributions of overland flow from the hillslope to the floodplain. Evapotranspiration within the three units was fairly constant (Table 2). There was a large increase in the groundwater component of the flow within the hillslope landscape component, increasing from 33 mm from the divide to 745 mm in the hillslope. Groundwater within the hillslope includes contributions from both the upland and the hillslope units and is also impacted by the smaller area of this unit.

Table 2. Average annual results for the five year simulation of the Gibbs Farm landscape using the SWAT-L model.

Landscape Unit	Precipitation (mm)	Surface Runoff (mm)	Lateral Flow Contribution (mm)	Groundwater Contribution (mm)	Evapotranspiration (mm)
Divide / Peanut-Cotton	1298	260	99	33	716
Hillslope / Bermuda	1298	304	312	745	642
Floodplain / Pine	1298	159	133	335	683
Watershed Outlet	1298	244	127	164	702

Simulated evapotranspiration (ET) within each landscape unit makes up the largest component of the water balance. In the divide where the row crops are grown ET is simulated to take up 55% of the precipitation. In the hillslope, Bermuda, unit it is 49% and it is 53% in the floodplain pine unit. Simulated surface runoff remains fairly constant across the first two landscape units, 20% in the divide and 23% in the hillslope, but it decreases to 12% in the floodplain landscape unit.

The water balance within each landscape was investigated by calculating the fraction of each water component as a percentage of catchment yield of that water component (Fig. 5). Precipitation is evenly distributed with area, with 70% in the upland, 10% in the hillslope, and 20% in the floodplain. Evapotranspiration is similarly distributed. Surface runoff is disproportionally distributed with respect to the area, with greater surface runoff generated in the upland and the hillslope and less generated in the floodplain. Lateral runoff and groundwater flow are also disproportionally distributed with larger contributions than the fraction of the area in the hillslope and the floodplain. The hillslope generates four times the groundwater flow than its fraction of the area.

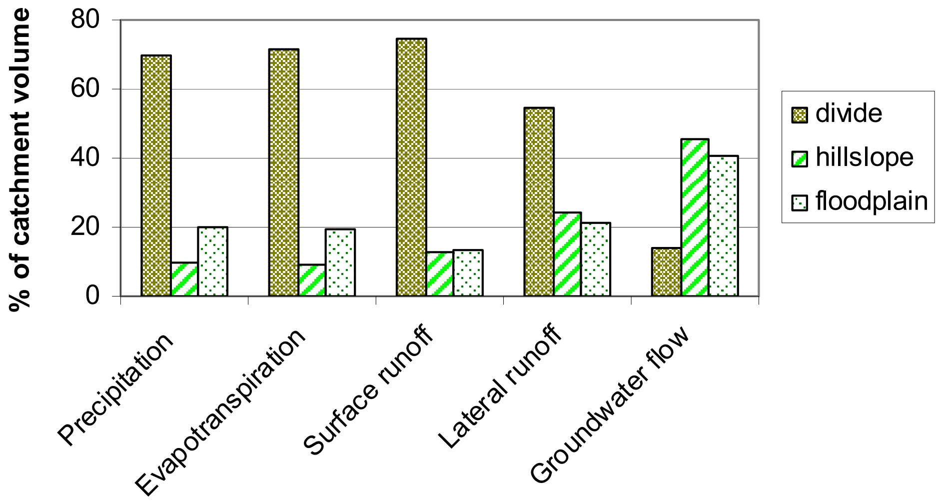

Figure 5. Average annual water balance for each landscape unit expressed as a percentage of the catchment yield of each hydrologic component.

Summary and Conclusions

Lowrance et al. (2007) reported the streamflow for the entire South watershed (Fig. 2) was approximately 15% of the precipitation that fell from 1996 through 2004. The yield from the North watershed was 27% of the precipitation during the same period. While the observed yield from the South watershed was less than the 27% simulated by the SWAT-L model, the measurements of Lowrance et al. (2007) included the area of the watershed which contained several irrigation ponds (Fig. 2) which could increase losses considerably.

Bosch et al (2005) observed that for similar soils 29% of precipitation in conventionally tilled upland fields is lost as surface runoff while other measurements of surface runoff in regional soils have been as low as 7% (Shirmohammadi et al., 1984). Simulated average annual runoff values for the divide (20%) and the hillslope (23%) for this study fall within this range. Prior research within this watershed indicates a 56 to 72% decrease in surface runoff as the flow moves from the upland fields into the grassed buffers (Sheridan et al., 1999). Surface runoff was fairly consistent moving from the grassed buffers into the riparian forest (Sheridan et al., 1999). The average annual volume of surface runoff simulated from the upland was 18172 m^3 while the average annual volume of surface runoff simulated from the hillslope was 3037 m^3. The large decrease indicates a large infiltration component in the hillslope, supporting prior research findings. Simulated surface runoff volume for the hillslope and the floodplain was fairly consistent (Fig. 5), also in agreement with prior field observations (Sheridan et al., 1999).

Estimates for ET in watersheds dominated by pine forests range from 60 to 80% of precipitation per year (Riekerk, 1985). Knisel et al. (1991) reported ET from an upland field in this region as 69% of precipitation for a corn/soybean rotation with an oats winter cover. Bosch et al. (1996) reported ET for the riparian forests in this watershed at 67%. Estimates for ET losses for the corn and fallow upland fields obtained using the GLEAMS model ranged from 700 to 1000 mm per year for the observation period (Bosch et al., 1996). Simulated estimates of ET from this study ranged from 55% for the divide to 53% for the hillslope, slightly below the reported ranges.

While additional calibration and testing of the SWAT-L model is necessary, the results are encouraging. The modifications will allow the model to more realistically represent actual landscape flow and transport processes. The relocation of water flow between surface, lateral, and groundwater flow appears to be represented with the model. As testing of the model is expanded to examine water quality effects the full utility of the model will be utilized.

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Watershed Configuration and Simulation of Landscape Processes with the SWAT Model

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EXTENDED ABSTRACT

Recent and future river basin management requires a more spatially distributed description of basin hydrology and nutrient transport processes to enable land use management as a process controlling factor to realize sound river basin management. The spatial description of these processes in the Soil and Water Assessment Tool (SWAT) watershed model is presently realized by aggregating the flows from overlaid soil and land use patches in subbasins with averaged slope angles. Many concepts with different degrees of complexity have been developed in river basin modelling to aggregate units with similar hydrologic behavior (Hydrological Response Units). Watershed configuration for SWAT currently consists of: 1) subbasins defined by surface topography and 2) hydrologic response units in each subbasin to account for heterogeneity in soils and land use. The hydrologic response units do not account for landscape position within the subbasin. Until recently, many existing watershed models did not implicitly account for landscape processes within a subbasin. Other smaller scale models do account for hillslope transfer (e.g. WEPP, REMM, APEX, HYDRUS-2D).

In an attempt to account for landscape position and processes, SWAT was modified to simulate landscape units within subbasins. Surface, lateral vadose zone, and groundwater flows are routed between landscape units (while allowing for hydrologic response units within each landscape unit). Surface runoff can be overland or channelized when routed from one landscape unit to the next. The model is being tested on the USDA-ARS experimental Y-watershed at Riesel, Texas, USA, using soil moisture and groundwater data. Using GIS techniques, the watershed was divided into three landscape units - valley bottom, hillslope, and upland. Further development will include landscape unit routing of sediment and nutrients and stream interaction with the valley bottom (i.e.; riparian/flood plain landscape unit). Simulated daily stream flow at the watershed outlet after routing across the landscape units, compared well to measured flow ($R^2 = 0.7$). Mean annual lateral flows across landscape units were also realistically simulated. Soil moisture (upper 1 m) was compared to measured soil moisture at one monitoring site in each landscape unit with the model predicting drying early in the summer but following general wetting/drying cycles. The revised version of the model is also tested using data collected from a low-gradient watershed near Tifton, Georgia, USA which contains heavily vegetated riparian buffers. The modified model provided reasonable simulations of surface and subsurface flow across the landscape positions without calibration. The application demonstrates the applicability of the model to simulate filtering of surface runoff, enhanced infiltration, and water quality buffering typically associated with riparian buffer systems. Future validation will include comparison with: 1) the Riparian Ecosystem Management Model (REMM) and riparian data sets; 2) with data from larger basins with defined floodplains; and 3) watersheds having well defined variable source contributing areas. The concept assumes the controlling factors for hydrological processes and functions must be adequately described at different spatio-temporal scales to accurately delineate such response units. This requires a sound description of the characteristics by using physically based parameters and indicators, but also simplified solutions at larger scales. Presentation of the new model concept and first results of testing simulations of different aspects of catchment-related control of landscape processes, pattern hydrology, and spatially distributed modelling are discussed.

1. INTRODUCTION

Watershed models are valuable tools for examining the impact of land use on hydrology and water quality. While extensive research has been done to describe the impact of agricultural management practices on small scales (field and farm level, hillslopes or headwaters), less is known about how these changes are reflected at the watershed scale. While linkages are being developed between the micro- and meso-scale (Shaman *et al.* 2004), the lack of reliable field data limits testing to a few specific linkages such as stream chemistry or groundwater flow, but not the many other features which actually occur. However, the success of programs such as the Total Maximum Daily Load (TDML) in the United States and the European Water Framework Directive (WFD) will be based on water quality improvements that result at the watershed scale.

Recently, Wolock *et al.* (2004) have proposed a linkage between basin scales that is based on a fundamental hydrologic landscape unit. According to the authors, this unit is defined as an upland and lowland separated by a valley side slope. They assert that hydrological landscapes can be conceived as variations and multiples of this fundamental unit. Bogaart and Troch (2006) investigations into the flow processes follow a similar approach in that they indicate that an ideal catchment would be characterized into a fixed drainage network and a fixed hillslope that folds around the channel network.

The Soil and Water Assessment Tool (SWAT) has been applied to watersheds throughout the world (Arnold and Fohrer, 2005). In most cases, the prediction accuracy was satisfactory to obtain working knowledge of the hydrologic system and the processes occurring in the watersheds. One of the shortcomings of SWAT has been an inability to model flow and transport from one position in the landscape to a lower position prior to entry into the stream. The model utilizes a Hydrologic Response Unit (HRU) concept which combines a unique combination of land use and soil type within a defined subbasin. Transported water, sediment, and chemicals from the HRUs are routed directly into the stream channel. Due to the importance of the different hydrological processes and transport mechanisms related to specific landscape positions, the purpose of this study is to document a new modelling approach which links these watershed processes from the hillslope to the watershed scale using the concept of hydrological landscape units. The modification divides the catchment into three units, the upland divide, the hillslope, and the floodplain. The modified model routes surface runoff, lateral subsurface flow, and shallow ground water flow from the divide, through the hillslope, through the floodplain, and eventually to the stream. By linking these units within watersheds, processes at the micro scale can be more appropriately summed for assessment of impacts and flow regimes at the watershed scale within a reasonable programming architecture for rapid assessment of land use and management scenarios. The specific objectives of this study are: 1) to develop a simple yet realistic model for landscape processes that can be generally applied at the river basin scale, 2) incorporate the landscape model into SWAT, and 3) test it at the USDA-ARS experimental watersheds at Riesel, Texas. The revised model is also tested using data collected from a low-gradient watershed near Tifton, Georgia, USA, which contains heavily vegetated riparian buffers (Bosch *et al.* 2007).

2. CURRENT LANDSCAPE APPROACHES IN MODELS

There have been numerous attempts to simulate landscape processes at various scales with varying complexity. Merrit *et al.* (2003) and Drewry *et al.* (2006) provide excellent reviews of and references for the following and numerous other models with details on how they spatially represent the processes in a watershed. The WEPP model simulates flow and sediment transport across a hill slope using multiple overland flow elements. HYDRUS-2D uses a numerical model to route surface and subsurface flow across a hill slope. Riparian zones near a stream are simulated in REMM, which needs inputs from upland models such as GLEAMS or EPIC or observed data.

There are also several different approaches to simulating landscape processes when scaling up to watersheds. One common approach, used in TOPMODEL, AGNPS, ANSWERS, and several numerical models like MIKE SHE, is to divide the watershed into cells. This accommodates significant spatial detail but for larger watersheds does not preserve channel reaches. Another approach is to divide a watershed into subwatersheds defined by topography (typically using a DEM), ensuring all surface water within the subwatershed flows to the outlet and each subwatershed contains a channel reach for routing. Models differ on accounting for heterogeneity within each subwatershed. The WEPP watershed model assumes a representative hill slope within each subwatershed, while models like DWSM, PRMS and KINEROS use overland flow planes or segments. HSPF allows pervious and impervious areas within a subwatershed. The HRU approach of SWAT is described in sections 1 and 3.1.

3. METHODS

3.1. SWAT Model Background

SWAT (Arnold *et al.* 1998) is continuous time and operates on a daily time step. The objective in model development was to predict the impact of management on water, sediment and agricultural chemical yields of long periods in large ungaged basins. A command structure is used for routing runoff and chemicals through a watershed. Using the routing command language, the model can simulate a basin sub-divided into grid cells or subwatersheds. Additional commands have been developed to allow measured and point source data to be input and routed with simulated flows.

In SWAT, a watershed is divided into subwatersheds with unique soil/landuse characteristics called hydrologic response units (HRUs). The water balance of each HRU in the watershed is represented by four storage volumes: snow, soil profile (0-2m), shallow aquifer (typically 2-20m), and deep aquifer (>20m). Flow, sediment, nutrient, and pesticide loadings from each HRU in a subwatershed are summed, and the resulting loads are routed through channels, ponds, and/or reservoirs to the watershed outlet.

3.2. Methods for Landscape Delineation

Existing methods to delineate landscape units range from simple soil considerations to complex methods using multivariate statistics and iterative segmentation algorithms to interpolate the continuous character of the landscape (MacMillan 2004). Gallant and Dowling (2003) point out, that "there are no published methods for mapping valley bottoms by automated algorithms although a number of methods exist that are designed to map floodplains". We searched for an effective but simplified solution for large-scale application and for potential integration into the SWAT. After an intensive method evaluation we selected the slope position method (USDA Forest Service 1999) as a useful method to delineate the landscape units.

The slope position of a cell is its relative position between the valley floor and the ridge top. Filling sinks and leveling peaks is the first step of the method and important to make the valleys and ridges fairly continuous. Downhill and "uphill" flow accumulation values greater than user specified limits are used to identify valleys and ridges, respectively. When large limits are used only large valleys and ridges will be identified as such, and small valleys and ridges will be considered somewhere mid-slope. Slope position is calculated for the cells in the output grid as the elevation of each cell relative to the elevation of the valley the cell flows down to and the ridge it flows up to. This is presented as a ratio, ranging from 0 (valley floor) to 100 (ridge top). Hill slope areas are represented by the values between these two ranges. Delineations in several watersheds have been validated by calculated relief amplitudes (moving window method) and soil maps of different scales.

3.3. Data

The test study site is located within the USDA-ARS Grassland, Soil and Water Research Laboratory watershed network near Riesel, TX, USA. The selected study watershed, Y2, drains 53.4 ha and includes three smaller upland subwatersheds of varying sizes between 6.6 and 8.4 ha. Convective thunderstorms during the warmer months contribute intense, short duration rainfall events. Long-term records collected at the site indicate an annual mean rainfall of 890 mm with relatively wet spring and fall seasons and drier summer and winter seasons.

Clay soils (Vertisol) dominate the site. The soil series consists of deep, moderately well-drained soils formed from weakly consolidated calcareous clays and marls and generally occurs on 1-3% slopes in upland areas. This soil is very slowly permeable when wet (saturated hydraulic conductivity = 1.52 mm/hr). However, when dry, preferential flow associated with soil cracks contributes to rapid infiltration rates. A shallow groundwater system follows local topography at an average depth of 3 meters. Recharge occurs through aerial infiltration at the outcrop.

4. DEVELOPMENT OF THE LANDSCAPE MODEL

To simulate water flow across the landscape, we propose a conceptual model similar to WEPP using a representative hill slope with landscape units within each subwatershed (Figure 1). We used the slope position method to delineate landscape units from a DEM. An example landscape unit delineation at the USDA-ARS experimental watershed in Riesel, Texas, USA, is shown in Figure 2. In this example, three landscape units were delineated: the divide, hill slope and floodplain (Figure 3). In a small watershed like Riesel, the floodplain unit would behave similar to a small stream riparian zone. The model still allows multiple hydrologic response units based on soil and land use within each landscape unit.

Surface Runoff/Run-on:
Surface runoff for each landscape unit is computed with the curve number method or the Green and Ampt infiltration equation. Run-on to an adjacent down slope landscape unit is estimated using a coefficient to partition the amount of flow that is channelized before leaving the landscape unit and the amount that is direct surface run-on. The amount of surface run-on that infiltrates is determined by multiplying the travel time by the saturated conductivity of the soil. To determine velocity and ultimately travel time, Manning's equation is used assuming a one-meter overland flow strip:

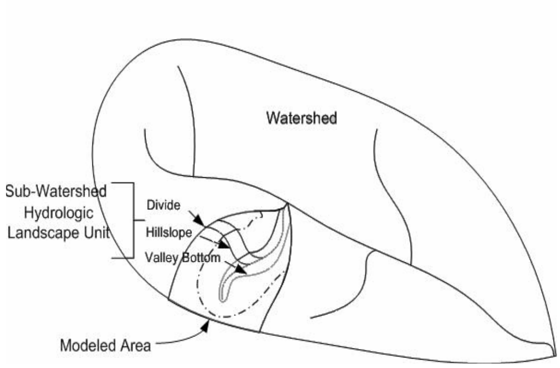

Figure 1. Subwatershed landscape delineation within a watershed.

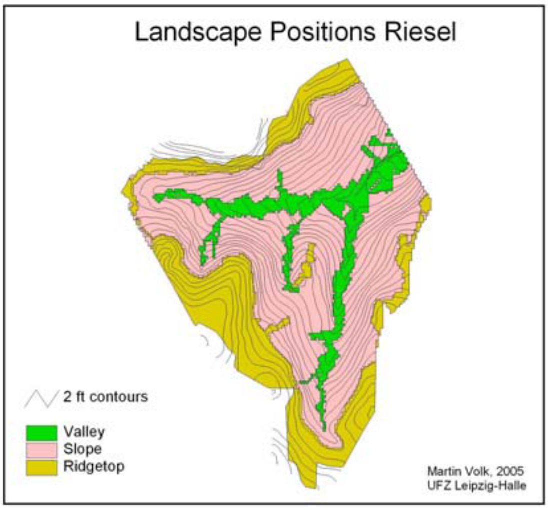

Figure 2. Landscape positions delineated at the Riesel experimental watershed.

$$V_s = (q_s)^{0.4} s^{0.3} / n^{0.6} \tag{1}$$

where q_s is the flow rate, s is slope and n is Manning's n. Then travel time (hours) is:

$$trt = sl/(3{,}600) * V_s \tag{2}$$

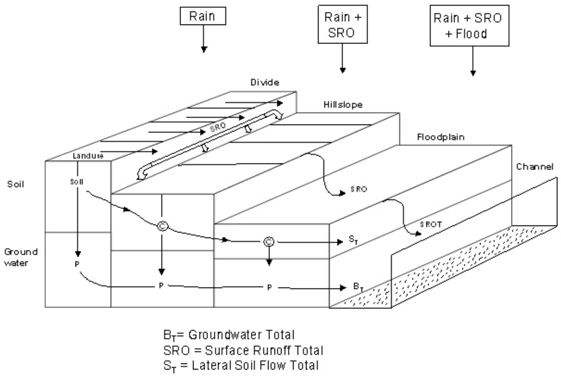

Figure 3. Processes considered in landscape routing units.

where sl is the slope length. Infiltration is calculated by multiplying the travel time by the saturated hydraulic conductivity:

$$I = trt * sc \tag{3}$$

where I is infiltration and sc is saturated hydraulic conductivity.

Lateral Soil Flow:
The model accommodates multiple soil layers as required to account for vertical heterogeneity and soil horizons typically defined in U.S. soil surveys. Lateral flow volumes are calculated using a kinematic storage model (Arnold *et al.* 1998) as a function of saturated conductivity, slope, slope length, and porosity.

$$Q_{lat} = 0.024 \cdot \left(\frac{2 \cdot SW_{ly,excess} \cdot K_{sat} \cdot slp}{\phi_d \cdot L_{hill}} \right) \tag{4}$$

The kinematic model also estimates surface seeps during saturated conditions, which is considered as surface run-on to the next landscape unit. Total lateral flow (summed from each soil layer) flows to the adjacent down slope landscape unit and is distributed to each soil layer weighted by depth of the soil layers. Lateral flow from the flood plain unit enters the channel.

Shallow Groundwater Flow:
Conceptually groundwater flow is simulated as routing through a series of linear storage elements as shown in Figure 4. This is the classic linear tank storage model as summarized by Brutsaert (2005) (5):

$$u(t) = \left(e^{(-t/k)}/k\right) * \left(\alpha_1/2(t/k)^2 + \alpha_2(t/k) + \alpha_3\right)$$

where u(t) is groundwater flow at time t, k is the recession constant, and α area of each landscape unit. The recession constant, k, can be determined from analysis from daily base flow recession curves.

Interaction with Stream Channel:
Groundwater flow from the flood plain landscape unit contributes flow directly to the stream. During low flow, channel seepage or transmission losses, recharges the shallow aquifer of the floodplain unit. When over bank flow occurs, depression

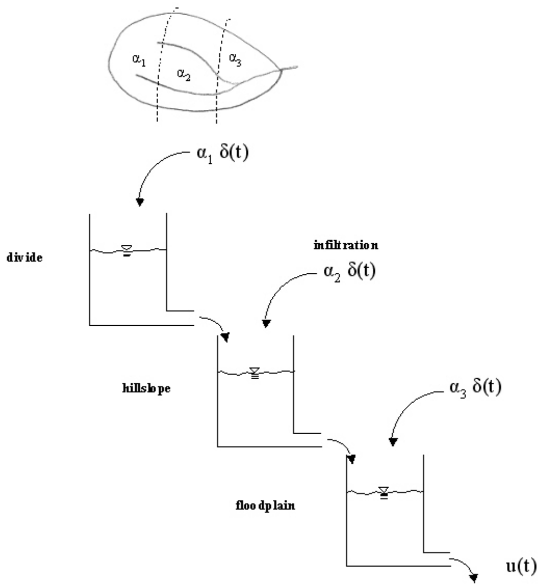

Figure 4. Linear storage elements for routing groundwater flow.

storage in the landscape unit is filled and subsequently allowed to infiltrate into the soil or evaporate.

Landscape Unit Configuration:
The example given here shows a simple three landscape unit hill slope representation on a small experimental watershed. To keep the model flexible to accommodate more complex watersheds and landscape unit configurations, we developed a command routing structure similar to the subwatershed routing used in SWAT. Four commands are used to route water through landscape units: landscape, route, add, and finish commands. The landscape command initializes the units and hydrologic response units in each, and sets the overland routing fraction. The route and add commands set the interaction between landscape units. In this example, the divide unit is routed through the hill slope unit, which is then routed through the floodplain unit. A more detailed structure can easily be accommodated using this command structure.

5. LANDSCAPE MODEL EVALUATION AT RIESEL WATERSHED

The SWAT landscape model was configured as shown in Figure 1 with one subbasin and three landscape units (divide, hillslope, and flood plain) and one HRU in each landscape unit. Soils are relatively uniform across the landscape and a Houston Black soil series was used. Slopes are relatively flat on the divide (about 1%), fairly steep on the hillslope (4%) and then flatten out in the flood plain (1%). The dominate land use was cropland on the divide, and pasture on the hillslope and flood plain. SWAT datasets were developed for the watershed and landscape units using the landscape unit configuration shown in Figure 3 and one HRU in each landscape unit.

Calibration Procedure:
Watershed Y-2 at Riesel was one of the original validation watersheds for the SWAT model and its predecessors and thus inputs had been developed and calibrated from previous studies. Also, Arnold et al. (2005) evaluated the model for watershed Y-2 and found good agreement with measured flow (R^2=0.87 and regression slope near 1.0). There were 66 measurable runoff events during the 1998-1999 period with measured runoff of 228 mm and simulated runoff of 245 mm. In this study with landscape units included, we started with inputs as they were calibrated in the 1998-1999 study and made adjustments to two inputs. Curve number was lowered by 2 and saturated conductivity of the lower soil layers was set at 30 mm/h to account for the impact of cracking on lateral soil flow.

6. RESULTS

Simulations were performed over a three year period during which soil moisture, lateral flow, and surface runoff data were jointly measured. All measurements were daily with the exception of lateral flow, which was recorded on a 2-3 day cycle. Results are described by routing structure within the model beginning with soil moisture.

Streamflow:
Regression of measured and simulated daily stream flow at gage Y-2 is shown in Figure 4. Stream flow is the sum of surface runoff and lateral soil flow as it leaves the valley bottom (landscape unit 3) and enters the channel – groundwater flow is negligible. We assume that all runoff that is channelized as it leaves the landscape unit does not infiltrate and reaches the subbasin outlet.

Mean measured and simulated daily stream flow is within 10% with a regression slope of 0.85 and R^2 = 0.70 and Nash-Sutcliffe coefficient = 0.67 (Figure 5). This is reasonable compared to other model studies comparing daily flows. In a previous

SWAT study at Riesel, for the period of 1998-1999, total flow at Y-2 was also within 10% with an $R^2 = 0.87$.

Lateral Soil Flow:
Lateral soil flow from landscape unit 2 to landscape unit 3 is compared to the hillslope seepage rates collected in the drainage trench. By adjusting the saturated conductivity of the lower

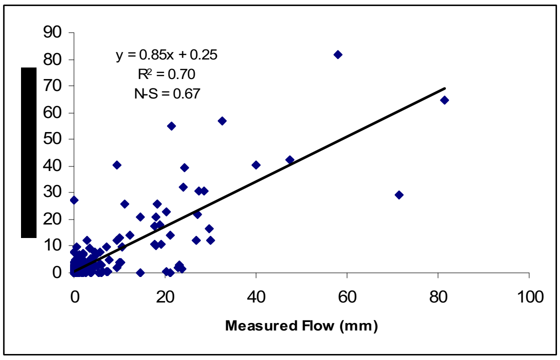

Figure 5. Measured and simulated daily streamflow at gauge Y-2.

layers to 30 mm/hr to account for crack flow, average annual measured and simulated lateral soil flow were 96 and 102 mm, respectively. However, regression of daily lateral soil flow resulted in a relatively low $R^2 = 0.20$. Figure 6 shows that ranges of peaks and recessions are realistic but the model typically overpredicts peaks. It is also important to recognized that flows are low (0-2 mm) which magnified any uncertainty or minor errors. While this needs to be addressed in future simulations, the model is routing water reasonably between the landscape units and does contribute water from divide, through the hillslope to the valley bottom. Another possible cause for the discrepancy between measured and simulated lateral flow during the three large storms was discovered after the data collection efforts. It was found the top over the collection pit had been left open and that rainwater was entering the weir pit. This may have influenced the storm volume on the days with major discrepancies as noted.

Soil Moisture:
Simulated soil moisture (total moisture for the upper 1 m) is compared against measured soil moisture at locations in each landscape unit. SWAT simulated soil moisture is printed on a daily time step while measurements were taken every 2 weeks. Results are reasonable for the divide landscape unit except in the summer of 2004 when the model predicts a significant drying while measurements suggest continued wetness into mid summer.

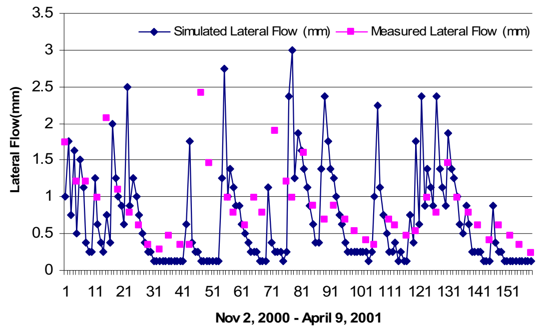

Figure 6. Measured and simulated daily lateral soil flow

The comparison for the hillslope landscape unit shows an over prediction in the winter of 2003 and under prediction of soil moisture in the early summer of 2004. The soil profile stays much dryer on the hillslope due to increased runoff and lateral soil flow caused by the steep slope; the soil is draining appropriately downslope. Similar to the divide, for the flood plain the model predicts low soil moisture in the early summer of 2004 and then overpredicts in the fall/winter. This could be caused by an overestimation of evapotranspiration in the summer growing season or by underestimating the surface runoff and lateral flow from the hillslope landscape unit. It should be noted that the soil moisture is averaged for the 1.0 meter profile and some of the discrepancies may be associated with this effect. Recent work on moisture regimes in vertisol soils at Riesel suggest that moisture levels are staying wetter at about 50 cm depth for the entire year and that the majority of flux is within the top 60 cm. Averaging values then over the whole profile will not accurately reflect these conditions.

7. CONCLUSIONS AND OUTLOOK

A landscape model was developed for simulating surface runoff, lateral soil flow, and shallow aquifer flow between landscape units. The landscape model was integrated into the SWAT watershed model and tested on the USDA-ARS Y2 experimental watershed (53.4 ha) near Riesel, Texas. In addition, a GIS based technique was developed and applied to delineate landscape units. Simulated daily stream flow at the watershed outlet after routing across the landscape units, compared well to measured flow ($R^2 = 0.7$) while mean annual lateral flows across landscape units were also realistically simulated. Soil moisture (upper 1 m) was compared to measured soil moisture at one monitoring site in each landscape unit with the model predicting drying early in the summer but following wetting/drying cycles.

The revision was developed to address variable source areas within watersheds and stream-side buffer systems which exist alongside many streams. The enhanced model will allow for more accurate simulation of natural transport processes within a hillslope. The revision was also tested using data collected from a low-gradient watershed near Tifton, Georgia, USA which contains heavily vegetated riparian buffers (Bosch *et al.* 2007). The modified model provided here reasonable simulations of surface and subsurface flow across the landscape positions without calibration. The application demonstrates the applicability of the model to simulate filtering of surface runoff, enhanced infiltration, and water quality buffering typically associated with riparian buffer systems.

Future planned development includes: 1) additional testing groundwater heights, and lateral soil flow at the Riesel Y2 watershed, 2) additional calibration and testing of the model for the USDA-ARS Gibb's Farm experimental watershed at Tifton, Georgia with "classic' riparian zones, 3) using the kinematic wave equation for overland and channel routing between landscape units, 4) incorporation of sediment and nutrient routing across the landscape. Presently, the plans are here to route firstly sediment with an overland flow and channel component across each landscape unit. Organic N and P will be routed with the sediment, and nitrates and phosphates will be assumed soluble and allowed to infiltrate as the water is routed across the units. Nitrates and phosphates will also be routed through the subsurface (soil and shallow aquifer) and denitrification will be simulated in riparian zones. Finally, 5) includes model testing on larger watersheds with defined flood plains.

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Photogrammetrie • Fernerkundung • Geoinformation 6/1998, S. 349–362, 3 Abb., 5 Tab.

Integration unterschiedlich erhobener Datenebenen in GIS für landschaftsökologische Bewertungen im mitteldeutschen Raum

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Zusammenfassung: Integrative landschaftsökologische Bewertungen erfordern eine Vielzahl unterschiedlicher Informationen, um den Regel- und Modellvorstellungen des Landschaftsökosystems gerecht zu werden. Zur Erfassung, Verwaltung, Kombination und Auswertung dieser Datenebenen stellen Geoinformationssysteme wichtige Instrumente dar, die neben Forschungseinrichtungen auch zunehmend Umweltbehörden und Planungsbüros zur Verfügung stehen. Die Verfügbarkeit und Paßfähigkeit der Daten und Systeme stellt jedoch noch immer ein großes Problem dar. Anhand von zwei Projekten im mitteldeutschen Raum wird gezeigt, wie durch die Kooperation solcher Einrichtungen fast alle relevanten Datenebenen gewonnen werden können, es werden aber auch die Probleme dieser Datenvielfalt dargelegt. Am Beispiel der Aufbereitung von zwei Eingangsparametern für eine Modellrechnung werden Lösungsansätze zur nachvollziehbaren Generalisierung/Aggregierung dieser „öffentlich verfügbaren" Daten für landschaftsökologische Untersuchungen vorgestellt. Auf diese Erfahrungen aufbauend werden Vorschläge zur Verbesserung von Datenlage, Laufendhaltung und Aufbereitung von Datensätzen für ihre Anwendung in Modellen und Bewertungsverfahren unterbreitet.

Summary: *Integration of different data layer in GIS for landscape ecological assessments in multifunctional landscapes.* Current research and discussions in landscape ecology and related disciplines make the need for integrated assessments on landscape scales obvious. The importance of GIS as an integrative tool and as an issue-based information system is increasingly recognised, especially by those who are concerned with environmental planning and management in science, legal authorities and business. However, there are still problems like incompatible systems and a lack of approbiate data. The two examples in eastern Germany show how to get all important data layers in cooperation with environmental authorities and agencies in consideration of the problems linked with that bulk of different data. Additionally the modification of input data for a ground water recharge model is presented as an example for an understandable aggregation and generalization of different scale information that is required for integrated landscape assessments.

1 Einleitung und Problemstellung

Im Sinne einer Erfassung und Bewertung der Gesamtraumfunktion der Landschaft wurde bereits seit den frühen siebziger Jahren immer stärker gefordert, in bezug auf die Prioritäten der räumlichen Entwicklungen neben den sozioökonomischen auch die ökologischen Gegebenheiten in Bewertungen und Planungen einzubeziehen (BUCHWALD & ENGELHARDT 1980). Mit der fortschreitenden Entwicklung und Verwendung von Geoinformationssystemen und deren Koppelung mit modernen Methoden der Landschaftsbewertung wächst die Hoffnung, immer komplexere landschaftsökologische Sachverhalte bearbeiten zu können. Dadurch scheint es in Zukunft möglich zu sein, den Handlungsbedarf zur Verbesserung der ökologischen Situation – unter Einbeziehung der ökonomischen Interessen und Bedürfnisse – besser herauszuarbeiten. Dennoch sind bei der Betrachtung der Entwicklung und des heutigen Standes der landschaftsökologischen Forschung noch zahlreiche offene Fragen beim Ablauf von Landschaftsbewertungen erkennbar (HOBBS 1997, MÜLLER & VOLK 1998).

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Die Anwendung von Geographischen Informationssystemen und Methoden der digitalen Bildverarbeitung für integrierte, landschaftsökologische Bewertungen setzt eine solide Datenbasis voraus. Es hat den Anschein, daß die benötigten Grundlagendaten im erforderlichen Umfang häufig nicht verfügbar sind. Tatsächlich jedoch liegen Daten auch in digitaler Form sowohl bei den unterschiedlichsten Ämtern und Behörden als auch bei verschiedensten Umwelt- und Planungsbüros und Forschungseinrichtungen vor (vgl. KRATZ & SUHLING 1997). Deshalb ist es unumgänglich zu recherchieren,

- wo die benötigten Basisdaten verfügbar sind,
- in welcher Form diese vorliegen,
- in welcher Form die Daten benötigt werden und
- ob es Regeln gibt, nach denen sich die Daten von der einen Form in die andere überführen lassen.

Allerdings kommt es oft vor, daß der vorliegende Datenbestand zu heterogen ist, weil er im Laufe der Zeit aus Ansprüchen heraus entstanden ist, die voneinander unabhängig sind oder waren. Viele dieser Datenbestände werden jedoch nicht fortgeführt, bleiben also als einmalige Momentaufnahmen stehen. Unabhängig davon ist eine Grundvoraussetzung die Verbesserung der Zusammenarbeit zwischen Ämtern, Behörden und Forschungseinrichtungen und in direktem Zusammenhang damit eine rechtliche Vereinfachung des erforderlichen Datenaustausches (DFG 1996).

Im vorliegenden Aufsatz werden Lösungsansätze für die oben angesprochenen Problemkreise am Beispiel der ausgewählten Untersuchungsgebiete des Regierungsbezirkes Dessau und des Einzugsgebietes der Parthe im mitteldeutschen Raum vorgestellt.

Im Projekt zum Regierungsbezirk Dessau werden basierend auf Untersuchungen zum Landschaftshaushalt Strategien zur zukünftigen Landschaftsentwicklung erarbeitet. Dabei wird dem Aspekt der Erhaltung und Wiederherstellung der Mehrfachnutzung der Landschaft bei Vermeidung von Nutzungskonflikten besondere Aufmerksamkeit gewidmet.

Im ca. 400 km^2 großen Einzugsgebiet der Parthe, einem repräsentativen Ausschnitt aus der nordsächsischen Altpleistozänlandschaft, laufen Untersuchungen zum Gebietswasser- und Stoffhaushalt in der Lößregion des Elbegebietes. Das Zusammenwirken der einzelnen Stoffströme wird hierbei durch eine hierarchische Vernetzung verschiedener GIS-gekoppelter Modelle beschrieben. Im Ergebnis dessen werden verschiedene Szenarien zum Gebietswasser- und Stoffhaushalt für Formen nachhaltiger Landnutzung berechnet und bewertet.

Die Landschaftsstrukturen und der Landschaftshaushalt der beiden ausgewählten Gebiete sind bereichsweise stark gestört von den Folgen jahrelanger monofunktionaler Nutzung durch Tagebau, Kiesabbau, Industrie und Intensivlandwirtschaft, enthalten aber andererseits auch einen hohen Anteil an schützenswerten, naturnahen Landschaftselementen. Durch die verschiedenen Nutzungsansprüche entstehen Konflikte – auch zwischen Ökologie und Ökonomie –, die sich nur mit dem Wissen um die ökologischen Zusammenhänge und Wechselwirkungen innerhalb der Landschaftstypen umweltverträglich lösen lassen.

2 Anforderungen an ein integratives GIS

Der *integrative Ansatz* orientiert sich an Regelkreis- und Modellvorstellungen des Geo(öko)systems (MOSIMANN 1985, KLUG & LANG 1983). Ziel ist es dabei, in regionalen Ansätzen die Bereitstellung von Leistungen für Umweltschutz und Erholung sowie die gleichzeitige Versorgung mit Nahrungsmitteln und Umweltgütern besser zu verknüpfen (SRU 1996). BARSCH & SAUPE (1994) sehen in der modularen Integration landschaftsökologischer und sozioökologischer Daten in die gebietliche Planung ein Mittel zur Verhinderung und Lösung von Nutzungskonflikten zwischen ökologischen und ökonomischen Interessen. Die in Abb. 1 dargestellte Systematik zeigt die Vorgehensweise zur Ableitung landschaftsökologischer Bewertungen in diesem integrativen Sinne.

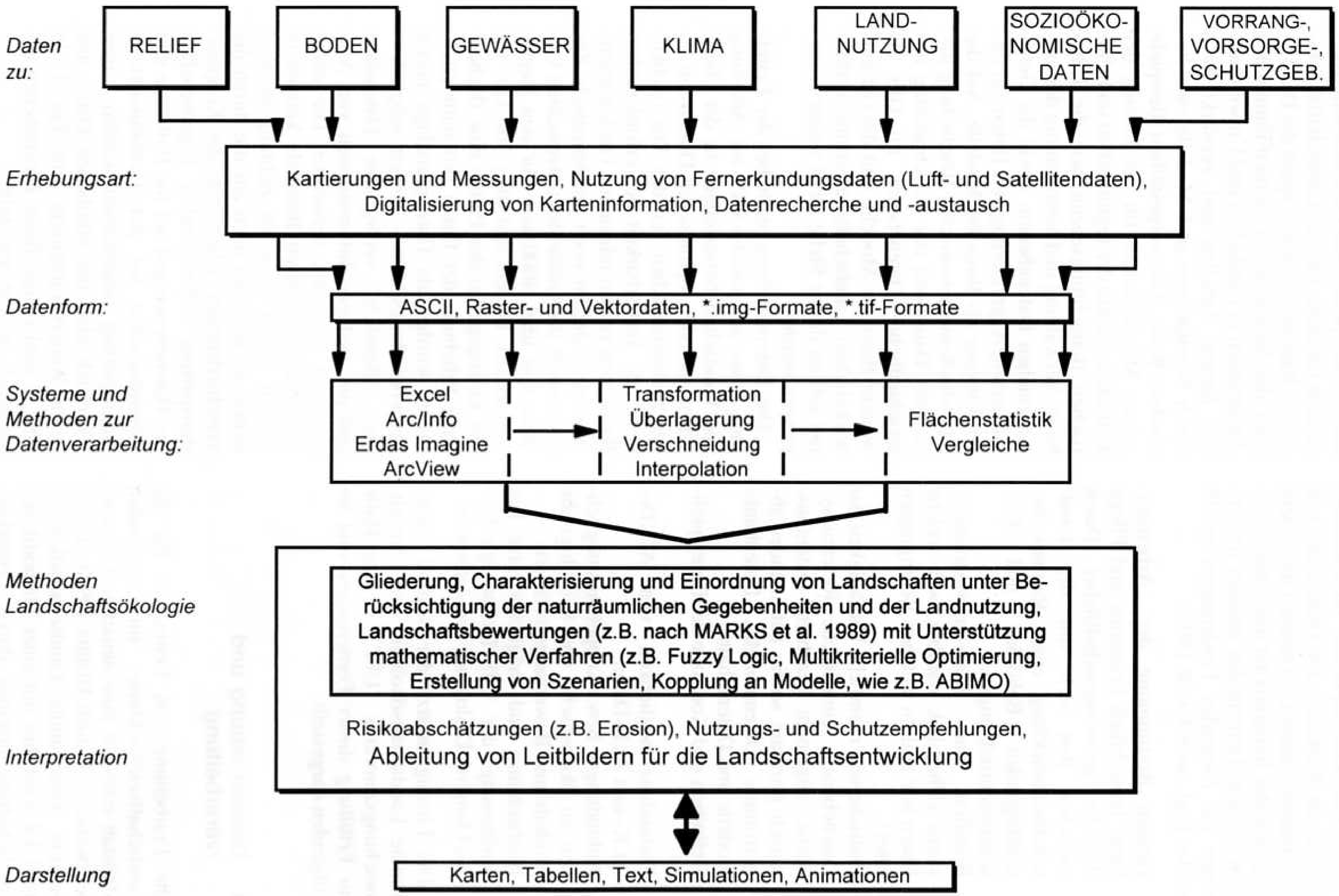

Abb. 1: Integration und Kombination unterschiedlich erhobener Datenebenen in einem GIS zur Ableitung landschaftsökologischer Bewertungen.

Nutzt man Geographische Informationssysteme als Werkzeug zur Umsetzung von Bewertungsverfahren, so müssen diese dem Anspruch der Integrativität also ebenso gerecht werden. Demzufolge müssen im einzelnen die folgenden Forderungen erfüllt werden (vgl. auch JÄGER 1997):

- Genaue Bestimmung der „Arbeitsteilung" hinsichtlich Erfassung und Pflege digitaler geowissenschaftlicher Daten zwischen den sich mit der Landschaftsentwicklung eines Raumes beschäftigenden Behörden und Forschungseinrichtungen.
- Erstellen eines Meta-Informationssystems (Überblick darüber, wer welche Daten wie und zu welchen Bedingungen führt).
- Zeitliche und räumliche Paßfähigkeit der Datenebenen (einheitliche Referenzsysteme, allgemein beachtete Daten(austausch-)formate sowie aufeinander abgestimmte Datenmodelle, Datenbankmodelle und Dateninhalte.
- Definition und Kontrolle der Datenqualität.
- Einbindungsmöglichkeit von ATKIS-, ALK- und ALB-Daten.
- Anbindungs- bzw. Koppelungsmöglichkeit an ökologische und hydrologische Modelle und Bewertungsverfahren.
- Beobachtung und möglicherweise Beeinflussung der GIS-Technologie-Entwicklung im Dialog mit den Anbietern.

Die Lösungsansätze der Sektion Angewandte Landschaftsökologie des Umweltforschungszentrums (UFZ) Leipzig-Halle zur Erfüllung dieser Forderungen sind im Folgenden dargestellt.

3 Datenerhebung und -verarbeitung

Die Erarbeitung von Leitbildern für die Landschaftsentwicklung im regionalen Maßstab erfordert eine neuartige Herangehensweise (vgl. auch HOBBS 1997). An der Sektion Angewandte Landschaftsökologie des UFZ wird dies mit einer Methodik des Landschaftsmonitorings durch Fernerkundung in Koppelung mit Geoinformationssystemen versucht. Für die Landschaftsökologie als Raumwissenschaft spielt die Darstellung der beobachteten zeitlich-räumlichen Änderungen in Landschaft und Umwelt mittels Karten, Tabellen und verstärkt auch durch Simulationen und Modelle eine besondere Rolle. Die vorgestellten Beispiele zeigen Möglichkeiten für die Lösung der kritischen Punkte der regionischen und chorischen Betrachtungsebene, wie die Erhebung, Verwaltung und Verarbeitung der umfangreichen Datenebenen, sowie die Generalisierung/Aggregierung von Daten vor deren Eingang als Basis für Modelle. Auf die (statistisch-mathematische) Bearbeitung unsicherer Daten und der Verschneidung unterschiedlicher Datenebenen im GIS als weitere Basis für Modelle und die (zu entwickelnden) Landschaftsbewertungsverfahren soll an dieser Stelle nicht weiter eingegangen werden.

Die Datenerhebung stellt bei der Erstellung eines integrativen GIS zur Ableitung von Landschaftsbewertungen in der chorischen und/oder regionischen Dimension einen problematischen Teil dar. Die flächendeckende Verfügbarkeit der erforderlichen Basisdaten zu den relevanten Geokompartimenten ist derzeit noch problematisch. Zudem macht die Größe der untersuchten Gebiete von über 4000 km^2 für den Regierungsbezirk Dessau und über 360 km^2 für das Einzugsgebiet der Parthe eine flächenhafte Erhebung der Daten fast unmöglich. Daher wurden als Datengrundlage neben Fernerkundungsdaten – soweit möglich – die „öffentlich" verfügbaren Datensätze und die analogen Informationen von Ämtern und Behörden verwendet. Die unten angeführten positiven Beispiele können in diesem Bezug zwar als richtungsweisend gelten, es soll aber auch auf die durch die verschiedenenen Arbeitsziele der Kooperationspartner differierenden Eigenschaften der Datenebenen und auf die Probleme und Lösungsansätze bei der Generalisierung/Aggregierung hingewiesen werden. Einen Überblick über die erhobenen Daten und deren Weiterverarbeitung gibt Tab. 1. In Abb. 2 sind einige dieser Datenebenen anschaulich zusammengestellt.

Tab. 1: Datenerhebung und -verarbeitung am Beispiel der Untersuchungsgebiete.

Datenebene	**Quelle**		**Ursprüngl. Format**		**Maßstab**		**Umgesetztes Format**	
	Dessau	***Parthe***	***Dessau***	***Parthe***	***Dessau***	***Parthe***	***Dessau***	***Parthe***
Boden nFK, Bodenformen, Bodenart	MMK, GLA (BÜK)	Boden-karten, Literatur	analog, E00-Format /Arc/info	analog	1:100000 1:200000	1:25000	Arc/Info-Coverage	Arc/Info-Coverage
Relief, historisch aktuell	MTB TK	- TK	analog analog	- analog	1:25000 1:10000 1:25000 1:100000	1:10000 1:25000	ASCII, Grid, Arc/Info-Coverage	- ASCII, Grid Arc/Info-Coverage
Gewässer, historisch aktuell als Verkehrsweg	MTB TK WWK RP Dessau	- LysBrand -	analog Raster-daten analog E00-Format /Arc/info	- analog -	1:25000 1:25000 1:200000	1:25000	Arc/Info-Coverage	- Arc/Info-Coverage -
chem.-physik. **Gewässer**parameter biolog.-ökolog. **Gewässer**parameter	STAU	LUG	*.txt	*.dbf	-	-	PC-Map, Arc/Info-Coverage	-
Klima	DWD	DWD Meßnetz (17 Statio-nen)	Datei	Datei analoge Erfassungs-bögen	1:100000	1:100000 1:25.000	Grid (Arc/Info) Rasterbild (Imgine, IDRISI)	Grid (Arc/Info) Rasterbild (Imgine)-
Landnutzung (teilw. historisch)	Sat-Szenen CORINE	SatSzenen	*.img (digitalis.) Arc/info	*.img (-)	geometr. Auflösung 10- 30 m 1:100000	geometr. Auflösung 5 - 30 m	*.img (Imagine) Arc/Info-Coverage	*.img (Imagine) Arc/Info-Coverage)
Flächennutzung (Gemeindeebene)	StatLand	-	*.txt	-	Gemeinde-ebene		PC-Map, Arc/Info-Coverage	-
Biotoptypen/Nutzungstypen	LAU	SLUG	E00-Format /Arc/info	E00-Format /Arc/info	1:10000	1:10000	Arc/Info-Coverage, Rasterbild (Imgine, IDRISI)	Arc/Info-Coverage, Rasterbild (Imgine, IDRISI)
Schutzgebiete (NSG, LSG, etc)	LAU	-	E00-Format /Arc/info	-	1:200000		Arc/Info-Coverage, Rasterbild (Imgine, IDRISI)	-
Vorrang- und Vorsorgegebiete	RP Dessau, LAU	-	E00-Format /Arc/info	-	1:200000		Arc/Info-Coverage, Rasterbild (Imgine, IDRISI)	-
Schienennetz und Straßennetz	RP Dessau	-	E00-Format /Arc/info	-	1:200000		Arc/Info-Coverage	-
Bevölkerung	StatLand	-	*.txt	-	Gemeinde-ebene, Kreisebene, Regierungs-bezirks-ebene		PC-Map, Arc/Info-Coverage	-
Zentrale Orte und regional bedeutsame Standorte	RP Dessau	-	E00-Format /Arc/info	-	1:200000		Arc/Info-Coverage	-
Erholung	StatLand, Umfragen	-	*.txt analog	-	Gemeinde-ebene, Kreisebene		PC-Map, Arc/Info-Coverage	-

Abkürzungen: **BÜK**=Bodenübersichtskarte; **GLA**=Geologisches Landesamt Sachsen-Anhalt; **LAU**=Landesamt für Umweltschutz Sachsen-Anhalt; **LysBrand**=Lysimeterstation Brandis; **MMK**=Mittelmaßstäbige Standortkartierung; **MTB**=historische Meßtischblätter; **RP**=Regierungspräsidium; **SatSzenen**=Satellitenbildszenen; **SLUG**=Sächsisches Landesamt für Umwelt und Geologie; **StatLand**=Statisches Landesamt Sachsen-Anhalt, **STAU**=Staatliches Amt für Umweltschutz Dessau-Wittenberg; **TK**=Topographische Karten, **WWK** = Wasserwirtschaftliche Karten vom LAU

Beispiel Parthegebiet

Die *Bodendaten* wurden aus analogen Karten im Maßstab 1:25.000 entnommen und mit hohem Arbeitsaufwand in digitale Form gebracht. Vom Deutschen Wetterdienst (DWD) liegen im 1 km-Raster (interpolierte) *Niederschlagsdaten* vor. Außerdem wurden die Meßreihen von 17 Hellmann-Niederschlagssammlern im Untersuchungsgebiet verwendet, die vom Sächsischen Landesamt für Umwelt und Geologie betreut werden. Die Situation der *Landnutzung* wurde aus Satellitenbildszenen Landsat-TM (30 m Auflösung) von 1989, 1992, 1994 und 1996 abgeleitet. Zudem werden Daten des indischen Satelliten IRS-1C des Jahres 1996 mit einer Auflösung von 25 m im

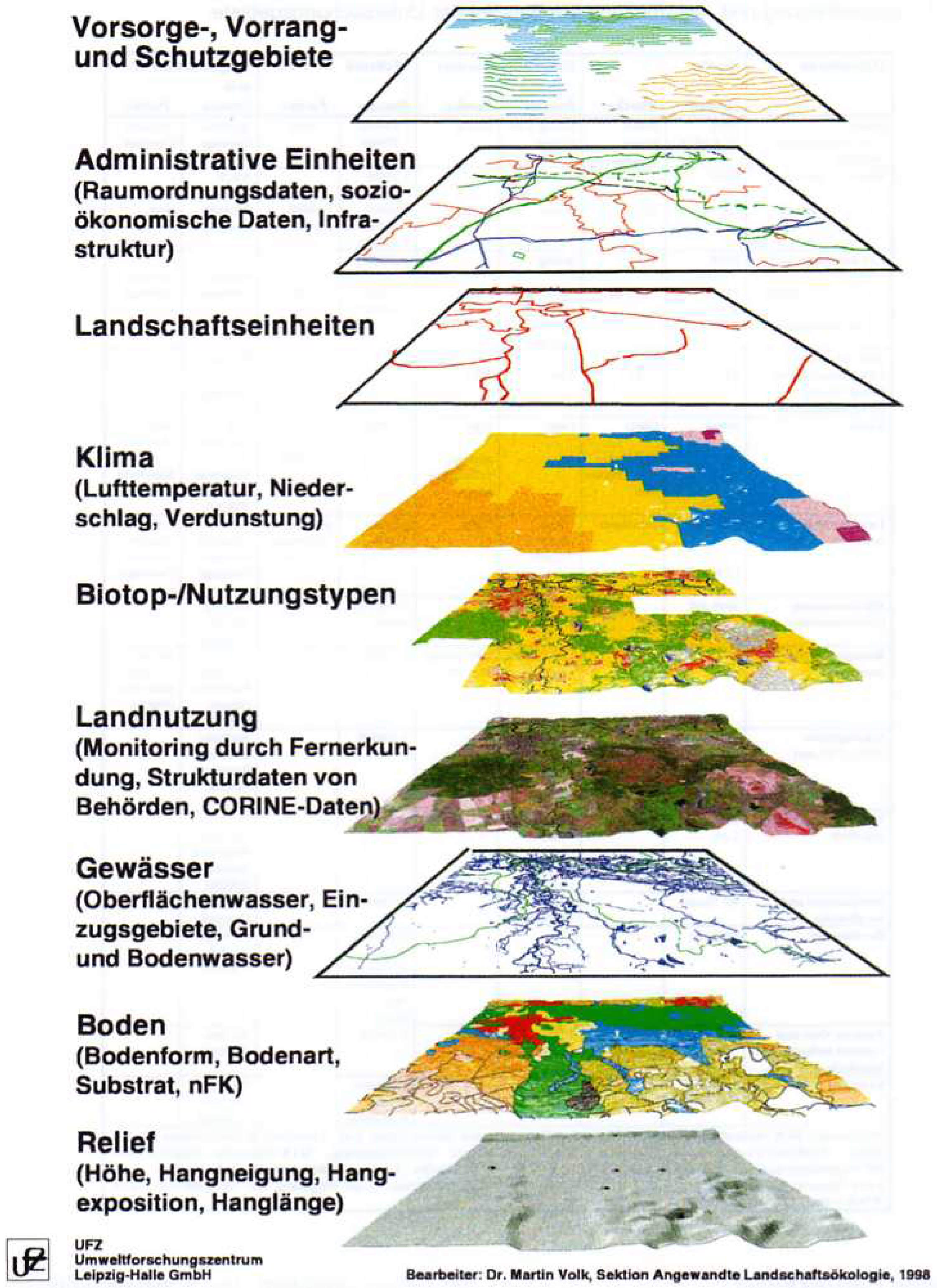

Abb. 2: Basisdaten zur Landschaftsbewertung.

Multispektralbereich und von 5 m im panchromatischen Modus verwendet. Der Vergleich der Szenen unterschiedlicher Jahre erlaubt die Feststellung von Landnutzungsänderungen.

Als weitere *Informationsquelle zur Landnutzung* stehen die *Biotop- und Nutzungstypen* des Sächsischen Landesamtes für Umwelt und Geologie in digitaler Form zur Verfügung.

Die steuernden und regelnden Eigenschaften des *Reliefs* werden aus *Digitalen Geländemodellen* (DGM) abgeleitet. Die Erstellung erfolgte aus der Ableitung der Höhenwerte aus Topographischen Karten im Maßstab 1:25.000 für das gesamte Gebiet und für einen Ausschnitt im Maßstab 1:10.000.

Die Informationen zum *Gewässernetz* wurden aus den Topographischen Karten TK25 entnommen und durch spezielle Karten vom Sächsischen Landesamt für Umwelt und Geologie ergänzt. Für die ober- und unterirdischen Gewässer existiert ein ausgedehntes Meßnetz zu Mengen- und Güteparametern.

Beispiel Regierungsbezirk Dessau

Die *Bodendaten* wurden im September 1997 in digitaler Form vom Geologischen Landesamt Sachsen-Anhalt zur Verfügung gestellt. Die *Niederschlagsdaten* wurden, wie oben schon genannt, vom DWD als interpolierte Werte im 1 km-Raster bezogen. Auch die Informationen zur Situation und zeitlichen Änderung der *Landnutzung* sollen in diesem Untersuchungsraum vorwiegend aus Satellitenbildern (siehe oben) abgeleitet werden. Zum einordnenden Überblick wird auf die Daten des CORINE-Projektes 'Land Cover' (Erfassungsmaßstab 1:100.000) zurückgegriffen. Vom Landesamt für Umweltschutz Sachsen-Anhalt wurden zudem die Ergebnisse der *Biotoptypen- und Nutzungstypenkartierung* in digitaler Form bereitgestellt. Außerdem wurden vom Landesamt sowie dem Regierungspräsidium Dessau digitale Daten zu *Schutzgebieten, Vorrang- und Vorsorgegebieten* sowie zur *Infrastruktur* (*Verkehrswege und schiffbare Flüsse*) zur Verwendung freigegeben. Die Informationen zum *Relief* wurden, wie oben beschrieben, aus Topographischen Karten abgeleitet, wobei ein DGM im 1 km-Raster bereits für den gesamten Regierungsbezirk vorliegt und im 250 m-Raster und 100 m-Raster bereits große Teile des Raumes abgedeckt sind. Das *Gewässernetz* liegt einerseits innerhalb von Rasterdaten der Topographischen Karten vor, die vom Landesamt für Landesvermessung und Datenverarbeitung erworben werden mussten, anderseits wurde es aus Wasserwirtschaftlichen Karten (inkl. Wassereinzugsgebiete) des Landesamtes für Umweltschutz sowie aus historischen Karten digitalisiert, um Veränderungen feststellen zu können. Sozioökonomische Daten wurden größtenteils vom Statistischen Landesamt Sachsen-Anhalt erworben.

Die *Verwaltung und Laufendhaltung* dieser immensen Datenmenge und Informationsfülle, die allein bei der Bearbeitung der beiden vorgestellten Projekte entsteht, ist eine schwierige Aufgabe. Daher wird am UFZ seit 1995 ein *Metainformationssystem* zur Beschreibung der Projekte und Daten aufgebaut, für das auch die Anbindung an das Internet vorgesehen ist. Ein erstes Beispiel hierfür liefert die Sektion Gewässerforschung des UFZ unter http://ifgsun1.gm.ufz.de:8000/.

Im Projekt Dessau verläuft der *Datenaustausch* mit den Kooperationspartnern in bezug auf die Austauschformate optimal, da sowohl beim Landesamt für Umweltschutz Sachsen-Anhalt, dem Geologischen Landesamt Sachsen-Anhalt als auch beim Regierungspräsidium Dessau die Geoinformationssysteme Arc/Info und ArcView verwendet werden. „Sofortige" inhaltliche Bearbeitung dieser Informationen stellen somit also auch kein Problem dar. Lediglich einzelne spezielle Informationen zum Boden, zum Gewässer sowie einige sozioökonomischer Informationen müssen/mußten digitalisiert werden oder lagen in Form von statistischen Tabellen vor. Beim Projekt Parthe konnten die Biotoptypen-/Nutzungstypen vom Sächsischen Landesamt für Umwelt und Geologie digital übernommen werden, fast alle anderen Informationen lagen in Form von analogen Karten oder in Tabellenform (Datei) vor. Dabei muß aber darauf hingewiesen werden, daß die Zielsetzung dieses Projektes einerseits zwar nicht unbedingt die Fülle an unterschiedlichen Datenebenen erfordert wie beim Projekt Dessau, aber anderseits eine höhere Informationsdichte (Genauigkeit) der Informationen in Sinne einer Systembetrachtung in einem Einzugsgebiet dieser Größe von Vorteil ist.

Die qualitative und quantitative Verfügbarkeit von *Reliefdaten* – z.B. bei Landesvermessungsämtern – stellt noch immer ein großes Problem dar, so daß diese, wie oben genannt, aus Topographischen Karten mit der Rastermethode entnommen werden mußten. Dies stellt bei einem Untersuchungsgebiet von der Größe des Regierungsbezirkes Dessau einen enormen zeitlichen Aufwand dar, zumal dabei zugleich die Anwendbarkeit dieser Informationen bzw. von digitalen Geländemodellen für landschaftsökologische Untersuchungen unter verschiedenen Rasterauflösungen/Maßstäben getestet werden mußte (vgl. z.B. VOLK & STEINHARDT 1996). Dieser Ansatz, *sinnvolle* Anwendungen von DGM unterschiedlicher Genauigkeit zur Simulation von Reliefeigenschaften für landschaftsökologische Analysen kritisch zu untersuchen, wird leider noch immer zu wenig verfolgt.

Ein großes Problem stellen die verschiedenen und stark *differierenden Maßstäbe der Datenebenen* dar. Von LESER (1991) z.B. wird gefordert, daß „die großen Maßstäbe von 1:25.000 oder 1:50.000 zwar nicht die Traditionsmaßstäbe der Regionalplanung sind, daß sie es aber werden sollten“, um „halbwegs fundierte landschaftsökologische Aussagen zu liefern“ und eine „Ökologisierung“ in der Regionalplanung und in der Fachplanung auf regionaler Ebene zu erreichen. Dies ist sicher zutreffend, dennoch ist die Verfügbarkeit der vielen Datenebenen in diesem Maßstab problematisch und die Erhebung für alle Teildisziplinen stellt für mesoskalige Untersuchungs- bzw. „Bewertungsräume“ noch eine fast unlösbare Aufgabe dar. Daher muß der Weg dorthin schrittweise erfolgen, z.B. die verfügbaren Informationen „standardisiert“ zu generalisieren bzw. zu aggregieren, um zu einem relativ einheitlichen Betrachtungsniveau zu gelangen. Bisher geschieht dies weitgehend subjektiv. Damit gestalten sich dann auch Vergleiche von Untersuchungsergebnissen als schwierig, in denen z.B. die gleichen *Modelle* in unterschiedlichen Untersuchungsgebieten verwendet wurden, da die Eingangsdaten oft nach unterschiedlichen Kriterien generalisiert und/oder aggregiert wurden (vgl. Kap. 4). Dies ist u.a. auch auf den interdisziplinären Ansatz der Landschaftsökologie zurückzuführen bzw. den unterschiedlichen *Arbeitszielen und Genauigkeitsansprüche* der einzelnen Fachdisziplinen. Insgesamt lassen sich angesprochenen Probleme nur im Dialog von Forschung und Praxis lösen. Daher streben die Arbeitsgruppen in den vorgestellten Projekten regelmäßige Beratungen an, bei denen einerseits die oben genannten Probleme angesprochen werden, anderseits aber auch *gegenseitige Kontrollen der Datenqualität* stattfinden. Für die mögliche, zukünftige *ATKIS- und Fachdatenintegration* am UFZ wird derzeit eine Diplomarbeit angefertigt (LINK 1998).

4 Modellgerechte Aufbereitung von Daten am Beispiel von Wasserhaushaltsberechnungen mit ABIMO

Bei der Modellierung relevanter Prozesse und der Landschaftsbewertung soll versucht werden, einen Kompromiß zu finden zwischen der landschaftsökologisch-holistischen, komplexanalytischen Herangehensweise und einer nachvollziehbaren, praxis- bzw. planungsorientierten Methodik (FINKE 1994, MÜLLER & VOLK 1998). Der erste Schritt hierzu besteht darin, größtenteils auf „öffentlich zugängliche“ Datenquellen von Behörden und Ämtern sowie auf kommerzielle Soft- und Hardware zurückzugreifen. Diese Anforderung gestaltet sich allerdings bei den Methoden und Instrumenten zur Modellierung und Bewertung etwas schwieriger. Im Vergleich mit der verfügbaren Datengrundlage stellt sich ein Problem, das bei vielen Modellanwendungen offensichtlich wird: Die Fachinformationen, die als Eingangsdaten für die Modellrechnungen dienen sollen, müssen klassifiziert und/oder standardisiert werden, damit sie überhaupt verwendet werden können. Oft existiert ein Widerspruch zwischen der sachlichen Gliederungstiefe der verfügbaren Basisdaten und der erforderlichen Modellparameter, der vor der Modellanwendung gelöst werden muß. Am Beispiel des Modells ABIMO (**A**bfluß**BI**ldungs**MO**dell), das langjährige Mittel des Wasserhaushaltes für den Lockergesteinsbereich berechnet, soll die Herangehensweise an dieses Problem im folgenden dargestellt werden.

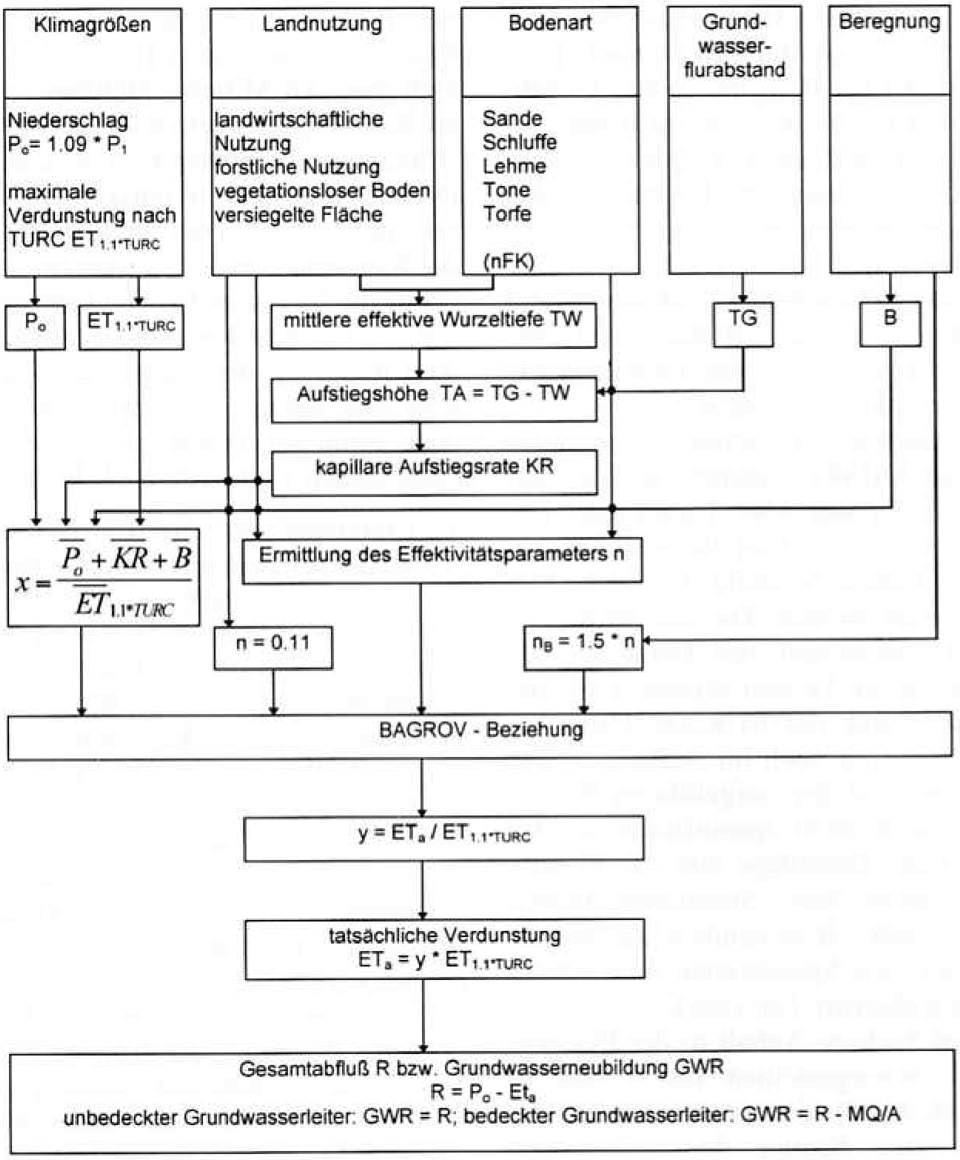

Abb. 3: Berechnungsschema für die Abflußbildung in ABIMO (nach GLUGLA & FÜRTIG 1997).

Dem Landschaftskompartiment Wasser kommt als wesentlicher Träger von Transportprozessen eine herausragende Bedeutung zu (KLEEBERG 1992). Um zunächst eine Grundvorstellung zu den Grundgrößen des Wasserhaushaltes im jeweiligen Untersuchungsraum zu erlangen, soll das Programm ABIMO zur Anwendung kommen (GLUGLA & FÜRTIG 1997). Dabei wird unter „mittlerer Abflußbildung" die Differenz aus vieljährigem Mittel von Niederschlag und realer Verdunstung verstanden, die dem mittleren Gesamtabfluß entspricht. Unter der Prämisse einer ausschließlich vertikalen Sickerung bis zur Grundwasseroberfläche stimmt dieser Wert mit der Grundwasserneubildung überein. Da dies jedoch nur in den seltensten Fällen gegeben ist, kann die Größe des mittleren Gesamtabflusses nur als Summe indifferent für ober- und unterirdischen Abfluß aufgefaßt werden. Abb. 3 zeigt das dem Programm zugrundeliegende Berechnungsschema für die Abflußbildung. Eingangsgrößen zur Berechnung der Abflußbildung sind Parameter des Klimas (Niederschlag, pot. Verdunstung), der Landnutzung (Nutzungstypen, Versiegelungsgrad) und des Bodens (Bodenart, nutzbare Feldkapazität) sowie Grundwasserflurabstand und Beregnung. Mit Hilfe dieser Parameter werden programmintern wichtige Kenngrößen wie die kapillare Aufstiegsrate berechnet. Zentraler Bestandteil des Modells ist die BAGROV-Beziehung (vgl. DVWK 1996) zur Berechnung des vieljährigen Mittels der Verdunstung in Abhängigkeit von den genannten Klimaparametern und den die Verdunstung beeinflussenden Größen Landnutzung und Boden (Effektivitätsparameter 'n').

Für die Datenebenen „Boden" und „Landnutzung" soll die erforderliche Generalisierung/Aggregierung im folgenden exemplarisch diskutiert werden.

Zur Festlegung der kapillaren Aufstiegsrate sind in ABIMO Angaben zur Bodenart erforderlich. Gemäß Abb. 3 muß jeder Bezugseinheit eine der fünf Bodenartengruppen Sande, Lehme, Schluffe, Tone oder Torfe zugewiesen werden. Die zur Verfügung stehenden bodenkundlichen Daten für den Regierungsbezirk Dessau stammen aus der Flächendatenbank der BÜK200. Diese befindet sich derzeit noch im Aufbau, so daß die Heterogenität der aufgeführten Bodeneinheiten noch nicht quantifiziert werden kann. Auf der Grundlage von ca. 20 Substratschichtungs- bzw. Substratmischungstypen, die jeder Bodeneinheit zugeordnet sind, lassen sich Spannweiten auftretender Bodenarten ableiten. Die vom Geologischen Landesamt Sachsen-Anhalt in der Flächendatenbank bereitgestellten Daten sind so strukturiert, daß für den entsprechenden Bezugsraum eine Spanne der auftretenden Korngrößenklassen definiert ist (Bodenart-min und Bodenart-max). Die in Tab. 2 aufgeführten Beispiele repräsentieren die verschiedenen auftretenden Klassifikationen der Eingangsdaten.

Daraus ergibt sich im Zusammenhang mit der in ABIMO erforderlichen Zuweisung *einer* Bodenart zu jeder Bezugsfläche ein Problem in zweifacher Hinsicht. Zum einen muß eine Art Mittelwertbildung in vertikaler Richtung erfolgen, um dem Problem der Schichtung Rechnung zu tragen, zum anderen aber muß auch in horizontaler Richtung gemittelt werden, um auch dieser Varianz der Substrate gerecht zu werden.

Für die hier erforderliche Generalisierung soll ein Lösungsweg aufgezeigt werden, der dem Anspruch der Nachvollziehbarkeit genügt und damit exemplarisch bei gleichen oder ähnlichen Problemstellungen Anwendung finden kann (vgl. auch PETRY 1998):

1. Ermittlung der charakteristischen Korngrößenzusammensetzung im Oberboden
   Für die unter BODENART-MIN und BODENART-MAX aufgeführten Bodenartenuntergruppen des Oberbodens werden aus den entsprechenden Klassenbreiten der Kornfraktionen Klassenmittel gebildet. Die beiden Klassenmittel jeder Kornfraktion werden dann als Minimum bzw. Maximum aufgefaßt.
2. Ermittlung der charakteristischen Korngrößenzusammensetzung im Unterboden (analog zur Vorgehensweise im Oberboden).
3. Mittelung zwischen Ober- und Unterboden unter Beachtung der Schichtmächtigkeit.
   Aus den ermittelten Klassengrenzen für Ober- und Unterboden werden die gewichteten Mittel der Klassengrenzen berechnet. Es wird dabei von einer Gesamtmächtigkeit des Profils von 2m ausgegangen.
4. Zuordnung der ermittelten Bodenartendefinition zu der am nächsten liegen Bodenartenuntergruppe.
   Über die Berechnung der Standardabweichung zu allen Bodenartenunter-

Tab. 2: Beispiel für Bodenartenspektren für Bodeneinheiten der BÜK 200 (nach KAINZ & HARTMANN 1997).

Bodenart-min		Bodenart-max	
Ss	reiner Sand	Su2	schwach schluffiger Sand
Uu/Sl3	reiner Schluff über mittel-lehmigem Sand (Schichtwechsel in ca. 6 dm Tiefe)	Ut4/Ls2	stark toniger Schluff über schwach sandigem Lehm
Sl3/Lt2	mittel-lehmiger Sand über schwach tonigem Lehm (Schichtwechsel in etwa 10 dm Tiefe)	Tu4/Ls4	stark schluffiger Ton über stark sandigem Lehm

Tab. 3: Übertragung der Bodenartenspektren in Kenngrößen der Korngrößenverteilung und Ermittlung der repräsentativen Bodenartenuntergruppe (nach AG Bodenkunde).

	Klassenbreite (nach AG Boden 1994)			Klassenmitte		
	Ton (min–max)	Schluff (min–max)	Sand (min–max)	Ton	Schluff	Sand
Oberboden						
Uu	0– 8	80–100	0–20	4	90	10
Ut4	17–25	65– 83	0–18	21	74	9
Oberboden, gesamt	4–21	74– 90	9–10			
Unterboden						
Sl3	8–12	10– 40	48–82	10	25	65
Ls2	17–25	40– 50	25–43	21	45	34
Unterboden, gesamt	10–21	25– 45	34–65			
Ober- und Unterboden	8,2–21	39,7–58,5	26,5–48,5	kleinste Standardabweichung zu Slu		

gruppen wird die Bodenartenuntergruppe bestimmt, zu der die geringste Standardabweichung auftritt.

Grundlage dieser Überlegungen sind die in der Bodenkundlichen Kartieranleitung (AG Bodenkunde 1996) aufgeführten Grenzwerte der Bodenarten (Anteile der Fraktionen) zur Definition der Bodenartenuntergruppen. Für das in der zweiten Zeile der Tab. 2 aufgeführte Bodenartenspektrum erhält man demzufolge die in Tab. 3 zusammengestellten Werte.

Die hier erarbeitete Vorgehensweise läßt sich mit einer einmal erarbeiteten Prozedur in EXCEL routinemäßig auf jede mögliche Bodenartenkombination anwenden und ist somit operationell einsetzbar.

Dem Berechnungsalgorithmus in ABIMO folgend werden Daten zur Landnutzung benötigt, um den in BAGROV-Beziehung eingehenden Effektivitätsparameter zur Ableitung der reellen Verdunstung zu ermitteln.

Vor diesem Hintergrund sind die in ABIMO erforderlichen Daten zur Landnutzung zu gliedern in:

- landwirtschaftliche Nutzflächen (L)
- forstliche Nutzung (W)
- gärtnerische Nutzung (K)
- vegetationslose Flächen (D)
- Gewässerflächen (G)
- bebaute Flächen (mit Berücksichtigung des Versiegelungs- (VER) und Kanalisationsgrades (KANN))

Als Eingangsdaten stehen die Kartiereinheiten der CIR-Biotoptypen und Landnutzungskartierungen der Länder Sachsen und Sachsen-Anhalt in digitaler Form zur Verfügung. Diese sind in einem bis zu 9-stelligen Zahlen- bzw. Buchstabenschlüssel erfaßt und damit in ihrer sachlichen Gliederungstiefe den Modellanforderungen nicht adäquat. Die Strukturierung der Kartiereinheiten in Sachsen wird an dem in Tab. 4 dargestellten Beispiel verdeutlicht. Dabei ist jeder Strukturebene der Nutzung eine Zahl zugeordnet, Zusatzinformationen bzw. Besonderheiten werden mit Buchstaben (Abkürzungen) charakterisiert und der jeweiligen Zahl zugewiesen.

Tab. 4: Struktur der Kartiereinheiten der CIR-Biotoptypen- und Landnutzungskartierung Sachsen (nach Luftbild Brandenburg 1994).

Hierarchische Strukturebenen	Schlüssel	Beispiel
Hauptgruppe	4	Grünland, Ruderalflur
Untergruppe	1	Wirtschaftsgrünland
Bestand	4 0 0	Feucht-, Naßgrünland
Ausprägung	4/ga	mit Gehölzaufwuchs
Nutzung		
Sekundärnutzung	1/gr	mit Gräben durch-
Sondernutzung		zogen

Bei der Zuordnung der Einheiten der Biotoptypenkartierung zu den für die Berechnung mit ABIMO erforderlichen Landnutzungsformen wurden maximal die drei ersten Gruppeneinteilungen (Haupt- und Untergruppen sowie u.U. Bestand) berücksichtigt, wie Tab. 5 exemplarisch für die ersten fünf Hauptgruppen der sächsischen Kartieranleitung zeigt.

Die hier erforderliche Generalisierung ist aufgrund der Strukturierung der Basisdaten weitaus einfacher zu realisieren als bei den Bodendaten. Bedingt durch den hierarchischen Aufbau des Schlüssels der Biotoptypenkartierung sind diese Daten für Aussagen auf verschiedenen Maßstabs- und Betrachtungsebenen nutzbar. Die Frage, wie tief man in diesen Strukturschlüssel eindringt, ist dabei immer von den zu bewältigenden Anforderungen abhängig und nicht in jedem Fall eindeutig zu beantworten. Für die Nutzung dieser Daten in ABIMO ist es wie oben bereits angedeutet, nicht generell ausreichend, sich auf Haupt- und Untergruppen zu beschränken. Insbesondere im Zusammenhang mit den erforderlichen Angaben zum Versiegelungsgrad bebauter Flächen, müssen bei den Siedlungs-, Infrastruktur- und Grünflächen auch Angaben zum Bestand berücksichtigt werden.

Nutzt man diese Daten der Biotoptypenkartierung beispielsweise als Referenzdaten für Landnutzungsklassifikationen von Satellitendaten, so setzt dies voraus, den Einheiten der Biotoptypenkartierung Spektralklassen der Landnutzung gegenüberzustellen. Bei dieser erforderlichen Parallelisierung wird der Klassifikationsansatz mitunter durchbrochen, da sich in der Hauptgruppe Siedlungs-, Infrastruktur- und Grünflächen Objekte mit unterschiedlichstem Spektralverhalten verbergen (Wohngebiete, Grün- und Freiflächen, Abgrabungen, Bauflächen).

Die hier diskutierten Beispiele für mögliche Lösungsansätze beziehen sich lediglich auf die Aufbereitung von *nur zwei* Eingangsparametern für *eine* Modellanwendung als Bestandteil landschaftsökologischer Untersuchungen und erheben auch nicht den Anspruch eines standardisierten Verfahrens. Jedoch wird angesichts der in Tab. 1 zusammengestellten Datenebenen der enorme Bedarf an weiteren derartigen nachvollziehbaren Ansätzen zur Generalisierung/Aggregierung für integrative Betrachtungen zu größeren Räumen deutlich.

Tab. 5: Zuordnungsbeispiele der Kartiereinheiten der Biotoptypenkartierung Sachsen zu den Landnutzungsformen in ABIMO.

Hauptgruppe	Untergruppe	ABIMO
Gewässer	Fließgewässer	G
	Stillgewässer	G
	gewässerbegleitende Vegetation	W
	Bauwerke an Gewässern	G
Moore, Sümpfe	Hochmoor, Übergangsmoor	L
	Niedermoor, Sumpf	L
Gründland, Ruderalflur	Wirtschaftsgrünland	L
	Ruderalflur, Staudenflur	L
Magerrasen, Felsfluren, Zwergstrauchheiden	anstehender Fels	D
	Blockschutthalden	D
	größere Lesesteinhaufen, offene Steinrücken	D
	offene Flächen	D
	Zwergstrauchheiden und Borstgrasrasen	L
	Magerrasen trockener Standorte	L
Baumgruppen, Hecken, Gebüsch	Feldgehölz, Baumgruppe	W
	Baumreihe	W
	Allee	W
	Einzelbaum, Solitär	W
	Hecke	W
	Gebüsch	L

5 Schlußfolgerungen und Diskussion

Die hier vorgestellten Arbeiten zu landschaftsökologischen Fragestellungen im mitteldeutschen Raum zeigen, daß bei eingehender Recherche die Verfügbarkeit von Datenebenen für diesen integrativen Ansatz kein unüberwindbares Problem mehr darstellen muß. Dies setzt jedoch eine Kooperationswilligkeit aller Beteiligten (Ämter, Behörden, Planungsbüros, Forschungseinrichtungen) voraus. Wie die Beispiele zeigen, werden geographische Informationssysteme immer öfter auch bei den o.g. öffentlichen und privaten Einrichtungen verwendet, so daß eine wesentliche Voraussetzung zur Verbesserung dieser Kooperation gegeben ist.

Bedingung dafür ist jedoch eine weitgehende Übereinstimmung der verwendeten Systeme und Datenformate, wie es bei den hier diskutierten Beispielen der Fall ist, jedoch derzeit noch immer nicht die Regel ist. Aus diesem Grund sind auch weiterhin Bemühungen um die Standardisierung von Datenformaten bzw. deren Kompatibilität erforderlich. Die gute Paßfähigkeit der Systeme ArcInfo, ArcView und Erdas-Imagine kann in diesem Zusammenhang als beispielgebend gelten.

Die erfolgreiche und effektive Realisierung von Projekten, die den Einsatz von integrativen GIS erlauben, erfordert die generelle Orientierung an gemeinsamen Richtlinien hinsichtlich Laufendhaltung, Beschreibung, Verwaltung, Einbindungsmöglichkeit an Modelle. Die Aufstellung eines diesbezüglichen Kriterienkataloges wäre eine wesentliche Erleichterung dafür.

Ein Problem ist noch immer die Fülle vorhandener Daten, die in der vorliegenden Form in den seltensten Fällen für Modellanwendungen und Bewertungsansätze nutzbar sind. Die beiden dargestellten Beispiele zu Boden und Landnutzung zeigen einen Lösungsweg zur nachvollziehbaren Aggregation und Generalisierung von Eingangsparametern für Wasserhaushaltsberechnungen als Grundlage landschaftsökologischer Bewertungen. Der Bedarf an standardisierten Verfahren, die eine Generalisierung und Aggregation von Informationen objektivieren, wird in diesem Zusammenhang offensichtlich.

Arbeitsteilung zwischen Ämtern und Behörden einerseits und Planungsbüros und Forschungsinstitutionen andererseits sollte angestrebt werden, um u.a. eine Laufendhaltung der Datenbestände zu gewährleisten. Damit werden temporäre Vergleiche zur Landschaftsentwicklung möglich, die mit GIS optimal zu realisieren sind.

Neben den oben beschriebenen Verfahren zur standardisierten Datenaggregation arbeiten die Autoren an Lösungsansätzen zur Kopplung von GIS mit Daten und Methoden der Fernerkundung, da auf diesem Wege ein großräumiges Monitoring von Landnutzungsänderungen optimal ermöglicht wird. Diese Überlegungen fließen dann in landschaftsökologische Bewertungen im mesoskaligen Bereich ein.

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Auswirkungen unterschiedlicher Datengrundlagen auf mesoskalige Wasserhaushaltsmodellierungen: Beispiele aus dem mitteldeutschen Raum

Potentials and Limits of Mesoscale Water Balance Modelling with Varying Input Data

D. Petry, F. Herzog, M. Volk, U. Steinhardt und S. Erfurth

Zusammenfassung: Die Landnutzung und ihre Veränderung prägen und verändern den Landschaftswasserhaushalt. Mesoskalige Wasserhaushaltsmodellierungen quantifizieren diese Auswirkungen für größere Räume und sind eine wichtige Grundlage für einen differenzierten Grundwasserschutz. Am Beispiel zweier Untersuchungsgebiete in Sachsen und Sachsen-Anhalt wird die Eignung unterschiedlicher Datengrundlagen sowie deren Einfluß auf die Berechnung wasserhaushaltlicher Kenngrößen unter Verwendung des Modells ABIMO diskutiert. Im Mittelpunkt stehen Probleme der inhaltlichen und räumlichen Datenauflösung, der Regionalisierung von Daten und der Validierung von Modellierungsergebnissen.

Schlüsselwörter: Grundwasserneubildung, Landschaft, mittlere Maßstabsebene, GIS

Summary: Land use and land use dynamics influence the landscape water balance. Mesoscale models quantify these impacts at the regional scale and form the basis for groundwater protection. The quality of input data determines the model output to a high degree. By using the runoff model ABIMO in two adjacent test areas with differing input data the suitability of varying data sets is discussed. Main emphasis is laid on spatial resolution and heterogeneity, data regionalisation, and validation of modelling results.

Key words: Groundwater recharge, landscape, mesoscale, GIS

1 Einleitung

Seit einigen Jahren wird nicht nur von der landschaftsökologischen Forschung sondern verstärkt auch von der Planung die Einbeziehung landschaftshaushaltlicher Faktoren in landschaftsbezogene Planungen gefordert (z. B. *Finke* et al., 1993). Eine zentrale Bedeutung kommt dem Wasserhaushalt aufgrund seiner Steuerungsfunktion, z. B. für Stofftransportprozesse, zu. Landnutzung und deren Änderungen prägen und verändern den Landschaftswasserhaushalt in quantitativer (Grundwasserneubildung und Abflußregime) und qualitativer Weise (stoffliche Belastung von Grund- und Oberflächengewässern). Dem muß bei der Entwicklung dauerhaft umweltgerechter Landnutzungskonzepte vermehrt Beachtung geschenkt werden. Die Bereitstellung wasserhaushaltlicher Kenngrößen trägt dazu bei:

- Grundwasserdargebotsprognosen für größere Räume zu erstellen;
- die Ausweisung von Vorrang- und Vorbehaltsgebieten für den Grundwasserschutz besser an den regionalen Infiltrationsbedingungen zu orientieren;
- durch Kopplung an ökonomische Modelle, die Kosten für den Grundwasserschutz regional zu differenzieren.

Wasser- und Stoffhaushaltsmodelle verfolgen in der Regel einen *bottom-up*-Ansatz, mit dem durch die Regionalisierung standörtlicher Bedingungen auf größere Räume geschlossen werden soll (*Kleeberg*, 1992). Der Ausweitung des Raumbezugs sind auf diese Weise jedoch enge Grenzen gesetzt (wenige km²), da die Komplexität solcher Modelle einer Datenfülle bedarf, die auf meso- bis makroskaliger Ebene oft nicht verfügbar ist. In der vorliegenden Studie werden daher die Möglichkeiten und Grenzen mesoskaliger Wasserhaushaltsmodellierungen diskutiert, die einem *top-down*-Ansatz folgen und mit flächendeckend verfügbaren Daten arbeiten. Dazu wurde mit dem Modell ABIMO (*Glugla* und *Fürtig*, 1997) in zwei benachbarten Untersuchungsgebieten mit teilweise unterschiedlichen Datengrundlagen die langjährige mittlere Grundwasserneubildung berechnet. Die Analyse der Modellierungsergebnisse konzentrierte sich auf die Fragen:

- Wie wirken sich unterschiedliche Datengrundlagen auf das Modellierungsergebnis aus?
- Welche Möglichkeiten der Validierung bestehen für mesoskalige Modellierungen?

2 Modell ABIMO

Das Abfluß-Bildungs-Modell ABIMO der Bundesanstalt für Gewässerkunde ist ein deterministisches konzeptionelles Modell, mit dem langjährige Mittelwerte von Gesamtabfluss und Grundwasserneubildung größerer Räume berechnet werden können. Die Kalibrierung erfolgte anhand umfangreicher Daten von Lysimeterstationen im Jung- und Altmoränengebiet der ehemaligen DDR. Das Modell ist in ebenen Lockergesteinsbereichen anwendbar, in denen laterale Wasserbewegungen zu vernachlässigen sind (*Glugla* und *Fürtig*, 1997). ABIMO ist damit auch in Untersuchungsgebieten mit relativ trockenen Klimabedingungen einsetzbar, in denen vergleichbare mesoskalig einsetzbare Verfahren (*Renger* et al., 1990) die Sickerwassermenge häufig überschätzen (*Frede* und *Dabbert*, 1998). Die Entwicklung des Modells geht auf Modifikationen der BAGROV-Beziehung zur Berech-

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nung der realen Evapotranspiration (ET_a, s. u.) durch *Glugla* und *Tiemer* (1971) zurück (DVWK 1996). In den 1980er Jahren erfolgten erste praktische Anwendungen an der ehemaligen Wasserwirtschaftsdirektion Berlin (Modell RASTER, Arbeitsplatz Grundwasserdargebot; *Glugla* und *König* 1989).

Im Modellansatz ergibt sich der Gesamtabfluß (R) aus der Differenz der langjährigen Mittel von Niederschlag (P_o) und aktueller Evapotranspiration (Et_a):

$$R = P_o - ET_a \tag{1}$$

ET_a wird mittels der BAGROV-Beziehung berechnet, die klimatische Gegebenheiten (Niederschlag, potentielle Verdunstung (ET_p) und weitere Standorteinflüsse (Landnutzung, Boden) in physikalisch und empirisch begründeter Weise verknüpft (vgl. DVWK 1996):

$$\frac{dET_a}{dP_o} = 1 - \left(\frac{ET_a}{ET_p}\right)^n \tag{2}$$

Die Faktoren Landnutzung und Boden gehen über den Effektivitätsparameter n in die BAGROV-Beziehung ein. Auf grafischem Wege läßt sich n aus Abbildung 1 ableiten. Je größer n, desto mehr nähert sich ETa an ETp an. Versiegelte Flächen erhalten den Wert $n = 0{,}1$. Die mittlere jährliche kapillare Aufstiegrate errechnet sich aus Grundwasserflurabstand und Bodenart in Abhängigkeit von der Landnutzung.

Allgemein gilt für den Einfluß auf den Gesamtabfluß die Rangfolge (*Renger*, 1992): *Klima (Niederschlag+Verdunstung) > Boden (nFK+Grundw.flurabstand) > Nutzung.*

In Tabelle 1 sind die Eingangsdaten für ABIMO zusammengestellt. Ein Teil der obligatorisch notwendigen Daten kann durch zusätzliche Angaben stärker differenziert werden. Mittels einer Sensitivitätsanalyse (*Herzog* & *Kunze*, im Druck), deren Ergebnisse hier nur auszugsweise wiedergegeben werden können, wurde der Einfluß der obligatorischen und der optionalen Daten auf das Gesamtergebnis untersucht. Dabei wurde von mittleren Werten ausgegangen, wie sie für die Untersuchungsgebiete typisch sind.

Eine Änderung der nFK um 1 Vol-% führt zu Änderungen des Abflusses R von 6 bis 16 mm/a, wobei sich ein Unterschied der nFK um 1 Vol-% bei höheren nFK-Werten stärker auswirkt als bei geringeren (Abb. 2a). Die Einflußstärke der

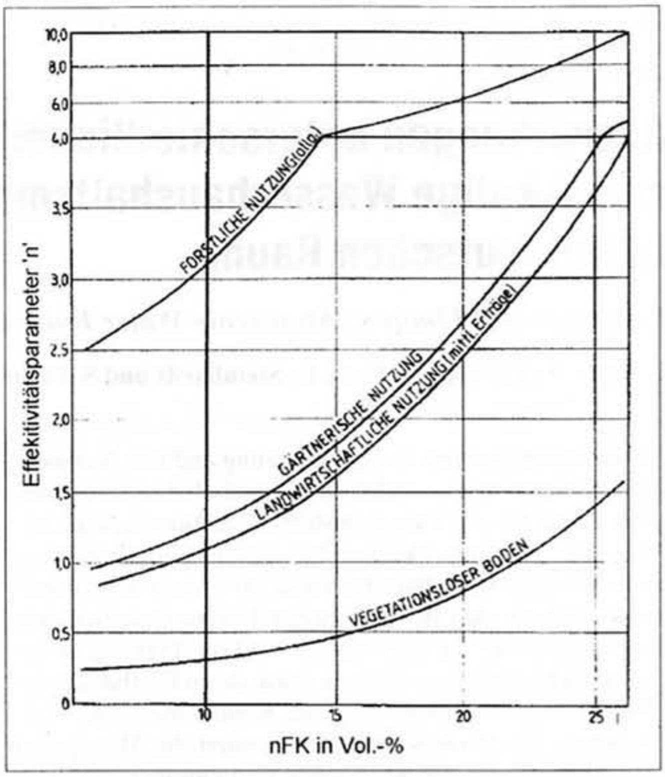

Abb. 1: *Ausprägung des Effektivitätsparameters n in Abhängigkeit von Landnutzung und nFK (DVWK 1996, verändert).*

nFK hängt auch von der Landnutzung ab. Unter Wald wirkt sich die nFK deutlich geringer auf den Abfluß aus (2–3 mm Abfluß/1Vol-% nFK-Änderung) als unter landwirtschaftlicher Nutzung (8–14 mm Abfluß/1Vol-% nFK-Änderung). Am höchsten ist der Gesamtabfluß auf vegetationslosen Flächen, am geringsten unter Wald (Abbildung 2b). Eine weitere Regelgröße von entscheidender Bedeutung ist die Grundwasserflurabstandsklasse (FLK). Bei einer FLK '< 1 m' setzt ein modellinterner Algorithmus generell $ET_a = ET_p$, unter Wald gilt dies auch für die FLK '1–2 m'.

Durch die Schätzung des lateralen Abflusses, wie er z. B. auf versiegelten Flächen auftritt, kann aus dem Gesamtabfluss auf die Grundwasserneubildung geschlossen werden, so dass diese beiden Begriffe im folgenden synonym verwendet werden.

Tab. 1. *Eingabedaten für das Abflußbildungsmodell ABIMO zur Berechnung der Grundwasserneubildung (in Klammern die Basisdate der Sensitivitätsanalyse).*

Table 1. *Input data of the runoff generation model ABIMO for calculating groundwater recharge (data used in sensitivity analysis in brackets).*

obligatorische Daten		**optionale Daten**
Jahresniederschlag (600 mm/a)		Sommerniederschlag (320 mm/a)
Potentielle Jahresverdunstung (600 mm/a)		Potentielle Verdunstung im Sommer (470 mm/a)
Grundwasserflurabstand (1-2 m)		Bodenart (Lehm)
Landnutzung:	- Landwirtschaft (L)	Ertragsklasse (ERT=35), Beregnungsmenge (50 mm/a)
	- Wald	Baumart, Begründungsjahr (1942)
	- Gewässer	
	- Gärtnerische Nutzung	
	- Vegetationslose Flächen	
Nutzbare Feldkapazität (17 Vol-%)		
Versiegelungsgrad (0 %)		
Kanalisationsgrad (0%)		

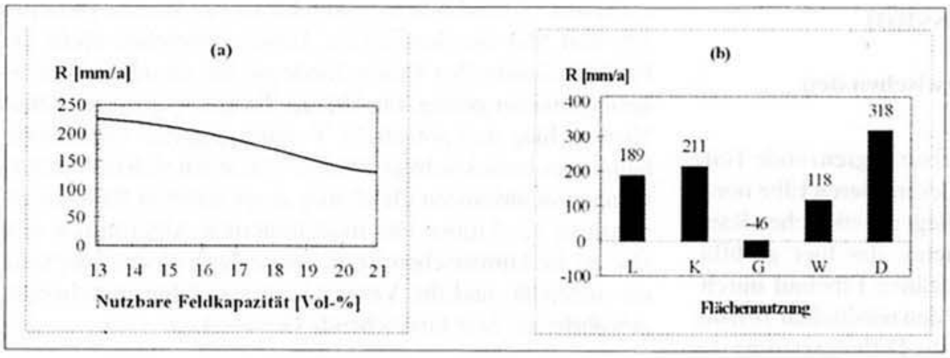

Abb. 2. Sensitivitätsanalyse: Gesamtabfluß in Abhängigkeit von der Nutzbaren Feldkapazität (a) und von der Flächennutzung (b) (L: Landwirtschaftliche Nutzfläche, K: Gärtnerische Nutzung, G: Gewässer, W: Wald, D: Vegetationslose Flächen). Quelle: Herzog u. Kunze, 1999.

***Fig. 2.** Sensitivity analysis: Total runoff (R) depending on available water capacity (a), and on land use (b) (L: agriculture, K: allotments, orchards, G: water, W: forest, D: unvegetated areas). Source: Herzog and Kunze, 1999.*

3 Bezugsräume, Datengrundlagen und -aufbereitung

Für die vorliegende Studie wurden zwei aneinander grenzende Gebiete ausgewählt:

- Altkreis Torgau (Sachsen; südl. Teil von Abbildung 3) (766 km²)
- Regierungsbezirk Dessau (Sachsen-Anhalt; nördl. Teil von Abbildung 3) (4295 km²)

Entsprechend des jeweiligen Arbeitsmaßstabes sowie der Datenverfügbarkeit kommen unterschiedliche Basisdaten zum Einsatz (Tabelle 2).

Für den Parameter **'Landnutzung'** wurden im Regierungsbezirk Dessau die Daten des CORINE-land-cover Projektes und im Altkreis Torgau die Biotoptypenkartierung des Freistaates Sachsen verwendet. Die CORINE-Daten wurden u. a. aus Satellitenbildern mit einem europaweit einheitlichen Differenzierungsschlüssel abgeleitet. Die Mindestgröße dargestellter Strukturen beträgt 25 ha. Hingegen erfolgte die Biotoptypenkartierung mit CIR-Luftbildern im Maßstab 1:10.000. Aus diesen Datengrundlagen lassen sich, teilweise unter Hinzuziehung weiterer Informationsquellen (*Herzog* und *Kunze*, 1999) auch die optionalen Parameter Versiegelungs- und Kanalisationsgrad, Baumart sowie Baumalter (nur mit Biotoptypenkartierung) festlegen.

Im Regierungsbezirk Dessau dienten die Substrattypen der Bodenübersichtskarte 1:200.000 (BÜK200) zur Ableitung der **nutzbaren Feldkapazität (nFK)** auf Grundlage der Bodenkundlichen Kartieranleitung (KA4) (*AG Boden*, 1994). Die Ermittlung der **Klasse des Grundwasserflurabstandes (FLK)** erfolgte ebenfalls auf deduktivem Wege über die Bodentypen der BÜK200, unterstützt durch Angaben der Mittelmaßstäbigen Landwirtschaftlichen Standortkartierung (MMK) zur Hydromorphie von Bodenformengesellschaften. Für den Altkreis Torgau wurden Bodenschätzungsdaten zur Ableitung der nFK verwendet. Die Dokumentation des Modells ABIMO enthält für diesen Zweck eine Zuordnungsvorschrift für Bodenarten der Bodenschätzung. Die FLK wurde aus der Grundwasserdargebotsprognose für den Regierungsbezirk Leipzig übernommen, der eine Interpolation von Meßstellenwerten auf 250 m-Raster zugrunde liegt.

***Tab. 2.** Datengrundlagen zur Wasserhaushaltsmodellierung mit ABIMO in den Untersuchungsgebieten 'Dessau' und 'Torgau': Maßstab bzw. räumliche Auflösung, Datenquelle, zugrunde gelegte Meßreihe bzw. Jahr der Datenerhebung.*

***Table 2.** Data sources for water balance modelling with ABIMO in the test areas 'Dessau' and 'Torgau': scale and resolution, data source, year of publication or sampling.*

Daten	Regierungsbezirk Dessau	Altkreis Torgau
Niederschlag[1) (30-jähriges Mittel)	1 km-Raster 1961-1990	
Potentielle Verdunstung[1) (30-jähriges Mittel)	1 km-Raster 1961-1990	
Landnutzung	CORINE land cover 1989-92[2)	Biotoptypenkartierung 1:10 000, 1992[3)
Grundwasserflurabstand	Bodenübersichtskarte 1:200.000 (BÜK200)[4) und Mittelmaßstäbige landwirtschaftliche Standortkartierung der DDR (MMK)	Flurabstandsmodell der Grundwasserdargebotsprognose Leipzig 250 m-Raster[5)
Nutzbare Feldkapazität	BÜK200	Bodenschätzung, 500 m-Raster
Hauptbodenart	BÜK200	Bodenschätzung 500 m-Raster
Sommerniederschlag[1) (30-jähriges Mittel)	–	1 km-Raster 1961-1990
Potentielle Verdunstung im Sommerhalbjahr[1) (30-jähriges Mittel)	–	1 km-Raster 1961-1990
Baumart	CORINE land cover	Biotoptypenkartierung 1:10 000, 1992[3)
Begründungsjahr der Forstflächen	–	Ableitung aus der Biotoptypenkartierung 1:10 000, 1992[3)

[1) Deutscher Wetterdienst (DWD);
[2) Statistisches Bundesamt (1996), Daten zur Bodenbedeckung für die neuen Länder und Berlin, CD-ROM, Wiesbaden;
[3) Sächsisches Landesamt für Umwelt und Geologie (1992);
[4) Schröder, H., Knauf, C. & W. Kainz (1997): Bodenübersichtskarte im Maßstab 1:200.000, Region Dessau, Geologisches Landesamt Sachsensen-Anhalt, Halle.
[5) Grundwasserdargebotsprognose für den Regierungsbezirk Leipzig 1996, Abschlussbericht. HGN Hydrogeologie, NL Torgau; G.E.O.S. Ingenieurgesellschaft mbH Freiberg; IBGW Ingenieurbüro für Grundwasser GmbH Leipzig.

4 Ergebnisse und Diskussion

4.1 Datenbedingte Unterschiede zwischen den Modellierungsergebnissen

Die Karten der Abbildungen 3a–d geben angrenzende Teile der beiden Untersuchungsräume an der mittleren Elbe nördlich von Torgau wieder. Das Gebiet liegt am südlichen Rand des norddeutschen Altmoränengebietes, das hier großflächig vom spätpleistozänen und holozänen Elbelauf durchschnitten wird. Abbildung 3b läßt für den nördlichen Teil der Elbaue im Regierungsbezirk Dessau die Differenzierung der Aue (verlandete Altarme (FLK < 1 m), höher gelegene Dünenzüge (FLK > 2 m), erkennen. Nach W und SW erfolgt über eine ca. 10–15 m hohe Geländekante der Übergang zu den glazialen, überwiegend sandigen Schmelzwasserablagerungen der Dübener Heide. Den östlichen Kartenrand bildet der Übergang des Elbtales in die weichselzeitliche stark sandige und daher größtenteils bewaldete Niederterrasse.

Die unterschiedlichen Datengrundlagen und ihr Einfluß auf das Modellierungsergebnis lassen sich anhand des Grenzgebietes zwischen Regierungsbezirk Dessau und Altkreis Torgau analysieren, da datenbedingte Brüche der natur- und kulturräumlichen Kontinuität an der gemeinsamen Grenze deutlich werden.

Niederschlag und Verdunstung

Im betrachteten Gebiet heben sich der W und SW (höher gelegene Ausläufer der Dübener Heide) mit Niederschlagswerten zwischen 580 und 600 mm vom Elbtal mit Werten zwischen 530 und 550 mm deutlich ab. Davon abgesehen bleibt der Einfluß klimatischer Unterschiede auf das räumliche Versickerungsmuster gering. Im Altkreis Torgau wurden zusätzlich Niederschlag und potentielle Verdunstung des Vegetationshalbjahres berücksichtigt. Dadurch änderten sich die Abflußmengen in ansonsten gleichartig ausgestatteten Räumen geringfügig (2–3 mm). Die modellinternen Algorithmen sind also an die klimatischen Verhältnisse der Untersuchungsräume angepaßt, und die Verwendung von Jahresmittelwerten gewährleistet eine hinreichende Genauigkeit.

Landnutzung (Abbildung 3a)

Kleinräumig ergeben sich durch die Verwendung von CORINE-Daten bzw. Biotoptypen erhebliche Ergebnisunterschiede. Die geschlossenen Forste auf Sanderflächen im Nordosten des Kartenausschnittes sind im Altkreis Torgau durch ein differenziertes Versickerungsmuster gekennzeichnet (Verwendung des Parameters Baumalter). Mittelt man diese Werte, erhält man eine Versickerungsrate von 110 mm/a, was etwas unter der Versickerungsrate des angrenzenden forstlichen Landschaftsausschnittes im RB Dessau (126 mm/a) liegt. Der höhere Wert für die Forste im RB Dessau ist mit der Verwendung der internen Standardeinstellung (> 85 Jahre) für das Baumalter zu erklären, wohingegen knapp die Hälfte der Waldflächen im Altkreis Torgau ein geringeres Alter (< 60 Jahre) haben, was eine Verminderung des Gesamtabflusses um 10–15 mm bewirkt. Die Kleinmaß-

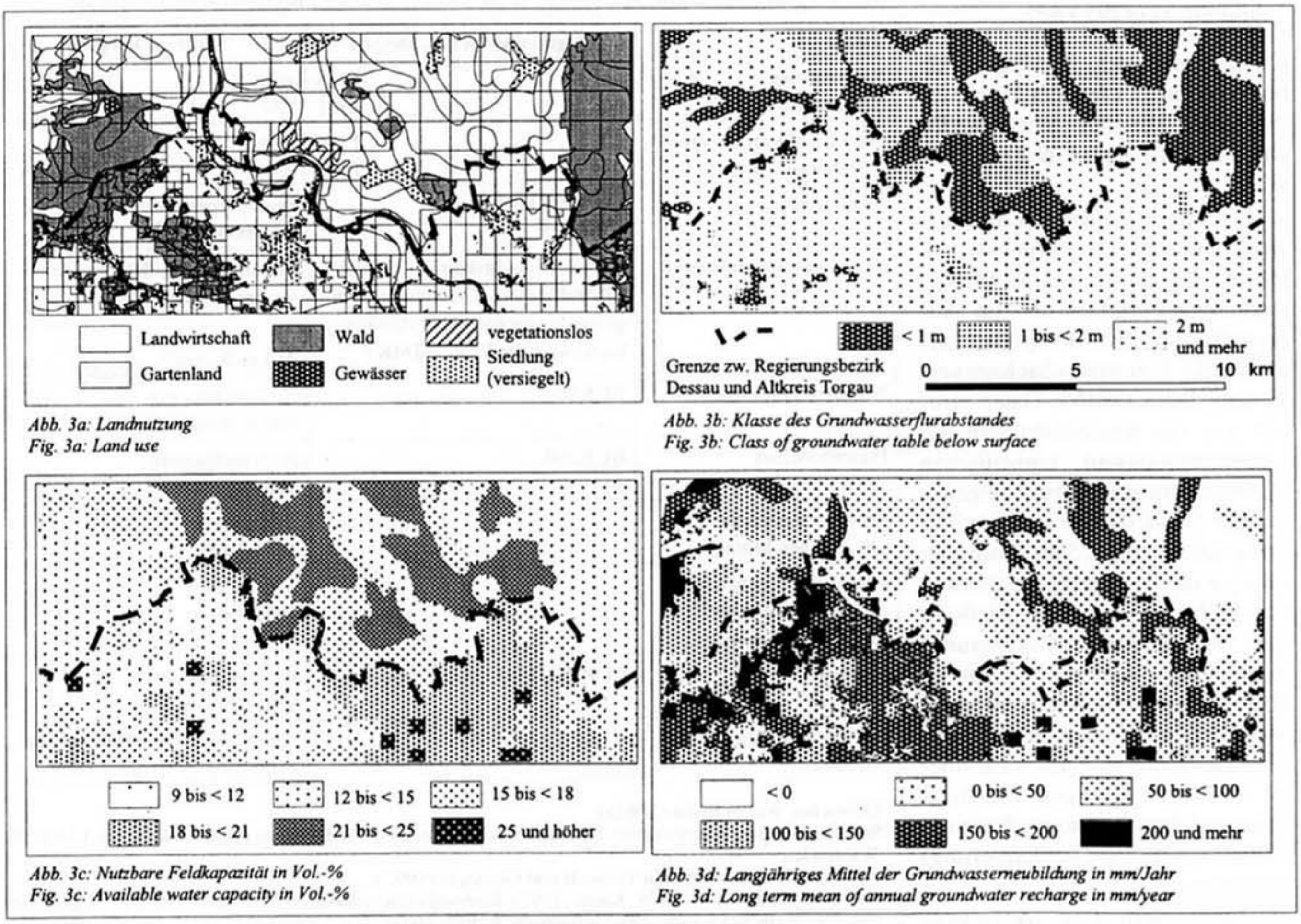

Abb. 3a: Landnutzung
Fig. 3a: Land use

Abb. 3b: Klasse des Grundwasserflurabstandes
Fig. 3b: Class of groundwater table below surface

Abb. 3c: Nutzbare Feldkapazität in Vol.-%
Fig. 3c: Available water capacity in Vol.-%

Abb. 3d: Langjähriges Mittel der Grundwasserneubildung in mm/Jahr
Fig. 3d: Long term mean of annual groundwater recharge in mm/year

Abb. 3a–d. *Räumliche Differenzierung ausgewählter Eingangsdaten und des Modellierungsergebnisses*
Fig. 3a–d. *Spatial differentiation of selected input data and the modelling results*

Tab. 3. Landnutzungsstruktur des Altkreises Torgau nach Biotoptypenkartierung und CORINE land cover
Table 3. Land use structure of the Torgau test area derived from biotope mapping and CORINE land cover.

Landnutzung	Biotoptypen-kartierung		CORINE land cover	
	km²	%	km²	%
Acker- und Grünland	480,1	62,7	512,8	66,9
Wald	221,4	28,9	213,2	27,8
Gewässer	12,4	1,6	8,4	1,1
Gartenland	12,2	1,6	1,0	0,1
Devastierte Fläche	8,3	1,1	0,6	0,1
Versiegelte Fläche	31,7	4,1	30,0	3,9

stäbigkeit der CORINE-Daten führt zu Fehlklassifikationen kleinerer Teilflächen. Für den Altkreis Torgau wurde die Flächennutzungsstruktur auf Grundlage der Biotoptypenkartierung derjenigen nach den CORINE-Daten gegenübergestellt (Tabelle 3). Durch die begrenzte Auflösung unterrepräsentieren die CORINE-Daten Nutzungen bzw. Landschaftselemente geringer Flächenausdehnung (Gewässer, devastierte Flächen, Hecken und Gehölzinseln). Dies bedingt auch den höheren Anteil landwirtschaftlich genutzter Flächen (Tabelle 3).

Grundwasserflurabstand (FLK) (Abbildung 3b)

Besonders auffällig sind die Unterschiede der Parameterausprägungen beim Grundwasserflurabstand. Ursache ist die unterschiedliche Definition von Grundwasser in Bodenkunde und Hydrogeologie. In der Bodenkunde wird der Grundwasserflurabstand über diagnostische Horizonte (z. B. Gr-Horizont) ermittelt. In der Hydrogeologie bezieht sich der Grundwasserflurabstand auf den obersten Grundwasserleiter.

In den Auelehmen haben sich Vega-Gleye mit einem diagnostischen Gr-Horizont im Bereich < 1 m unter Flur entwickelt, woraus die FLK < 1 m abzuleiten ist. Der oberste Grundwasserleiter aus pleistozänen Kiesen und Schottern steht erst unter den ca. 2 m mächtigen Auelehmen an, so daß aus hydrogeologischer Sicht eine FLK > 2 m gegeben ist. Die naturräumliche Differenzierung des Elbtales wird daher im nördlichen Teil gut wiedergegeben, im südlichen Teil überhaupt nicht.

Für die Wasserhaushaltsmodellierung sind bodenkundlich abgeleitete Daten jedoch eher geeignet, da entscheidend ist, ob oberhalb 2 bzw. 1 m unter Flur pflanzenverfügbares Wasser für die Verdunstung verfügbar ist – bzw. über kapillaren Aufstieg verfügbar wird – oder nicht. Aber auch dies gilt nicht ohne Einschränkung, da die bodengenetische Ableitung wesentliche Faktoren nicht berücksichtigt:

a) *Anthropogene Einflüsse*: Meliorierte Auenböden haben tatsächlich einen Grundwasserflurabstand > 1 m. Wasser, das vor der Melioration verdunstet ist, wird heute den Vorflutern zugeführt.
b) *Saisonale Schwankungen:* Während des Vegetationshalbjahres dürfte der Grundwasserflurabstand von Vega-Gleyen teilweise tiefer als 1 m liegen. Dadurch wird die tatsächliche Versickerung in der Modellierung unterschätzt.

Nutzbare Feldkapazität (nFK) (Abbildung 3c)

Die Verwendung von BÜK200 bzw. Bodenschätzung führt zu unterschiedlichen Ausprägungen der nFK (siehe Tabelle 4):

a) In der Bodenschätzung kommt im Unterschied zur heutigen Terminologie die Hauptbodenart 'Schluff' nicht vor. Dies verändert im Vergleich zur BÜK200 die räumliche Bodenartenstruktur, da 'schluffige' Substrate anderen Hauptbodenarten zugeordnet werden. Lehmige Böden der Bodenschätzung besitzen gegenüber der BÜK200 um 10–15 % (tonige Böden um ca. 15–20 %) höhere nFK-Werte, sandige Böden hingegen ca. 5–10 % niedrigere Werte.
b) Geringeres Bodenartenspektrum nach den generalisierten Bodenschätzungsdaten als nach der BÜK200, damit zwangsläufig eine geringere Differenzierung der nFK (Tabelle 3b).
c) Eingeschränkte Übersetzbarkeit der Substrat-/Bodenartbezeichnungen der Bodenschätzung auf heutige Bezeichnungen und damit auch bei der Ableitung der nFK

Tab. 4. (a) Bodenarten und nFK, (b) Mittelwerte der nFK in Vol.-% nach Hauptbodenarten[1]
Table 4. (a) Soil texture classes and available water capacity, (b) Means of available water capacities in Vol.-% for soil texture classes[1]

(a)	Dessau[1] (BÜK200)[2]	Torgau[3] (Bodenschätzung)	nFK in Vol.-%[4]
Sand		S	9
	Ss/Su2		13,3
		Sl	14
	Su2	lS	16
	Sl4		17
	Sl2		17,5
	Sl3		18
	Su3		20,5
Lehm	Lts		13,5
	Lt3		14
	Ls3		15
	Ls4		16
		SL	18
		sL	19
	Slu	L	21
Schluff	Uls		22
	Ut3		23,5
Ton	Lt3		14
	Tl		14,5
		T	17
		LT	16
b)	**Dessau**	**Torgau**	
Sand	14,9	13,8	
Lehm	17,3	19,6	
Schluff	22,6	–	
Ton	14,0	16,9	

[1] Bodenartenkürzel der KA4
[2] repräsentative Bodenarten der Substratypen der Bodeneinheiten
[3] Bodenartenkürzel der Bodenschätzung
[4] Zuordnung der nFK zu Bodenarten in Dessau über KA4, in Torgau über Zuordnungsvorschrift Dokumentation ABIMO

(*Fleischmann, Hacker* und *Oelkers* 1979); diesem Umstand wird durch die starke Generalisierung von Bodenarten und nFK in der Zuordnungsvorschrift der ABIMO-Dokumentation Rechnung getragen.

Die Unterschiede zwischen den Grundlagendaten für die bodenbezogenen Parameter haben eine zunächst paradox erscheinende Wirkung: Aus der kleinmaßstäbigen BÜK200 abgeleitete Daten zeichnen die naturräumliche Struktur des Raumes besser nach als diejenigen der großmaßstäbigen Bodenschätzung. Neben der geringeren Bodenartendifferenzierung der Bodenschätzung, liegt dies auch an der Überbetonung naturräumlicher Strukturen in der BÜK200. Die BÜK200 enthält heterogene Bodeneinheiten (Bodenformengesellschaften). In die Modellierung geht aber nur ein Wert für jede Bezugseinheit ein. Die Bodenschätzung kann solche heterogenen Strukturen prinzipiell genauer wiedergeben. Durch die schwierige Übertragbarkeit in die heutige Terminologie und die Verwendung aggregierter Bodenschätzungsdaten kommt es jedoch zu den o.g. Verzerrungen.

Hennings (1998) kommt in einer Untersuchung am Beispiel des Blattes Vechta (Niedersachsen) der Bodenkarte 1:5.000 zu dem Ergebnis, dass bei der Berechnung der Sickerwasserrate mit der BÜK200 keine größere Genauigkeit zu erzielen ist als mit der BÜK1000. Diese Aussage lässt sich nicht ohne Weiteres auf andere Gebiete übertragen, da die Datengrundlagen, mit denen die einzelnen Blätter dieser Kartenwerke erstellt werden, sehr unterschiedlich sind. Das Blatt Dessau der BÜK200 weist eine deutlich höhere räumliche und inhaltliche Differenzierung auf als die entsprechenden Bodeneinheiten der BÜK1000, da sie im wesentlichen auf den Bodeneinheiten der Mittelmaßstäbigen Standortkartierung im Maßstab 1:100.000 beruht.

4.2 Vergleich der Modellierungsergebnisse

Im Altkreis Torgau spiegelt sich in der Grundwasserneubildung durch die geringe Differenzierung von nFK und FLK die Nutzungsstruktur stark wieder. Gerade die Parameter nFK und FLK haben im Regierungsbezirk Dessau entscheidenden Einfluß auf das Versickerungsmuster, hinter denen die Nutzung deutlich zurücksteht. Dennoch läßt sich für den gesamten Grenzraum eine Dreiteilung anhand der Versickerung vornehmen:

a) Die vorwiegend bewaldete Niederterrasse östlich des Elbtales mit negativer bis geringer Grundwasserneubildung (überwiegend –25 bis 75 mm). Höhere Werte ergeben sich lediglich für Agrarinseln und höher gelegene Bereiche.
b) Die zentrale Elbaue mit negativer Grundwasserneubildung in tief gelegenen Bereichen und mittlerer Versikkerung in den übrigen Teilen (überwiegend 80 bis 180 mm). Hier ist davon auszugehen, daß ein Teil des aus dem Wurzelraum versickernden Wassers in den undurchlässigen Auelehmen lateral den Vorflutern zuströmt.
c) Die sowohl land- als auch forstwirtschaftlich geprägten Bereiche auf periglazialen Sanden westlich des Elbtales weisen unter Landwirtschaft mittlere bis hohe (überwiegend 120 bis 220 mm) und unter Wald geringe bis mittlere Grundwasserneubildungsraten auf (überwiegend 80 bis 120 mm).

Trotz der vorhandenen datenbedingten Unterschiede differieren die Modellierungsergebnisse für ähnlich strukturierte Räume nur in relativ engen Grenzen: Ein Vergleich anhand der höher gelegenen Agrargebiete im Elbtal sowie den Wäldern auf den Sandern ergibt datenbedingte Abweichungen der Versickerungsraten von ca. 20 mm. Kleinräumig und in Bereichen, in denen unterschiedliche Datengrundlagen zu stark abweichenden Ausprägungen einzelner Parameter führen, sind die Unterschiede jedoch groß: Im zentralen Elbtal verursacht der niedrige Grundwasserflurabstand auf Dessauer Seite eine negative Versickerungsrate von < –50 mm, während auf der unmittelbar angrenzenden Torgauer Seite Versickerungsraten von ca. 100 mm erreicht werden.

4.3 Möglichkeiten der Validierung

Betrachtet man die Versickerungshöhe des Gesamtraumes, ergibt sich für den nördlichen Teil eine Versickerungsrate von 69 mm/a (= 13 % des mittleren Niederschlags) im langjährigen Mittel und für den südlichen Teil von 132 mm/a (= 23 % des mittleren Niederschlags). Diese Differenz relativiert sich, wenn man im Regierungsbezirk Dessau die Flächen mit einer FLK < 1 m aus der Betrachtung herausnimmt, die aus o.g. Gründen nicht mit naturräumlich ähnlich strukturierten Flächen des Altkreises Torgau vergleichbar sind. Dann erhöht sich die Versickerung auf 118 mm/a (= 22 % des mittleren Niederschlags). Diese Werte stimmen auffallend mit dem von *Hölting* (1996, nach *Baumgartner* und *Liebscher*, 1990) für Ostdeutschland genannten mittleren Abfluss von 145 mm/a (= 22 % des mittleren Niederschlags) überein. Mesoskalige Modellierungen langjähriger Mittel wasserhaushaltlicher Größen lassen weder in ihrem Raumbezug noch in der Höhe der Grundwasserneubildung exakte Aussagen zu. Dies hat seine Ursache in der zwangsläufigen Generalisierung von Eingangsparametern, der notwendigen Interpolation von Meßwerten und der maßstabsbedingten Vernachlässigung kleinräumig relevanter Differenzierungen z. B. des Bodenwasserhaushaltes. Die Differenzen der berechneten Versickerungsraten von ca. 20 mm/a liegen daher unserer Meinung nach innerhalb eines vertretbaren Toleranzbereiches für meso- bis makroskalige Modellierungen.

Die Verwendung des hydrogeologisch definierten Grundwasserflurabstandes in der Aue oder des bodengenetisch abgeleiteten Grundwasserflurabstandes in großflächig meliorierten Räumen muss jedoch zwangsläufig zu falschen Ergebnissen führen.

Die genaue Validierung mesoskaliger Modellrechnungen ist prinzipiell anhand von Pegeldaten in anthropogen möglichst unbeeinflußten Einzugsgebieten möglich. Es existieren verschiedene Verfahren zur Berechnung des Anteiles des grundwasserbürtigen Abflusses am Gesamtabfluß, wie Abflußganglinienseparation und MoMNQ-Verfahren, die u. a. *Hölting* (1996) und *Wohlrab* et al. (1992) beschreiben. Die Ergebnisse der Wasserhaushaltsmodellierung mit ABIMO lassen sich durch die Datenhaltung im GIS auch auf Einzugsgebiete beziehen. Hier besteht allerdings das Problem, daß für kleinere Einzugsgebiete, die vollständig im Regierungsbezirk Dessau liegen, keine Tagesabflußwerte oder differenzierte gewässerkundliche Hauptzahlen vorliegen. In Zusammenarbeit mit den Staatlichen Umweltämtern wird jedoch versucht zu einer qualitativen Einschätzung der Modellierungsergebnisse zu kommen.

Vom Maßstabsbereich her vergleichbare Untersuchungen arbeiten i. d. R. mit zeitlich höher auflösenden Modellen (Tagesabflußwerte). Für das Einzugsgebiet der Oberen Leine (Südostniedersachsen) wurden im Rahmen des DFG-

Schwerpunktprogramms „Regionalisierung in der Hydrologie" in verschiedenen Teilprojekten (z. B. *Diekkrüger*, 1999 (Modell SIMULAT) *Gerold*, 1999 (Modell WASMOD) Wasserhaushaltsberechnungen durchgeführt. Hier bestand die Möglichkeit, die Ergebnisse mit Pegelwerten zu validieren. Als bodenkundliche Datengrundlagen dienten auch hier mittel- und kleinmaßstäbige Bodenkarten (BÜK50 und BÜK200), aus denen die entsprechenden Modellparameter über Pedotransferfunktionen des Niedersächsischen Bodeninformationssystems NIBIS abgeleitet wurden. Diese Vorgehensweise entspricht weitestgehend derjenigen im Regierungsbezirk Dessau, wo die Modellparameter mit den Pedotransferfunktionen der KA4 aus der vorläufigen Flächendatenbank der BÜK200 abgeleitet wurden.

5 Schlußfolgerungen

Aus den durchgeführten Analysen können folgende Schlußfolgerungen abgeleitet werden:

a) Das Modell ABIMO liefert bei einem Fehler-Toleranzbereich von 20 bis 25 mm/a für die beiden Untersuchungsgebiete auf meso- bis makroskaligem Niveau gültige Ergebnisse. Dies gilt unter der Voraussetzung der ausreichenden Validierung und Kalibrierung des Modells an Lysimeterdaten, von der die Modellentwickler ausgehen (*Glugla*, mündl. Mitteilung). Der Toleranzbereich wird bewußt in absoluten Werten angegeben, da prozentuale Angaben hohen Versickerungsraten ungleich größere Abweichungen zugestehen würden als niedrigen Versickerungsraten.
b) Die Datengrundlagen einer mesoskaligen Modellierung müssen daraufhin überprüft werden, ob sie die auf regionaler Ebene relevanten Differenzierungen der Modellparameter wiedergeben (z. B. Unterschiede zwischen bodenkundlich und hydrogeologisch definiertem Grundwasserflurabstand).
c) Regionale Modellierungen zur Bilanzierung des Landschaftswasserhaushaltes sind demzufolge ein wichtiges Instrumentarium beispielsweise zur Abschätzung der Folgen von Landnutzungsänderungen. Zur Ausdehnung der Anwendungsmöglichkeiten ist es nun erforderlich, die quantitativen Größen über geeignete Modelle zu Aussagen über die daran gekoppelten Stoffausträge zu nutzen.

Die datenbedingten Ergebnisunterschiede lassen sich auf bestimmte Ursachen zurückführen:

a) Räumliche Auflösung und inhaltliche Differenzierung der Daten (Dimensions- und Aggregationsproblematik)

CORINE-Daten genügen in räumlicher Auflösung und Nutzungsdifferenzierung den Modellanforderungen, bergen aber ein gewisses Maß an Fehlklassifikationen. Die Biotoptypenkartierung minimiert solche Fehler, muß jedoch den Modellanforderungen entsprechend stark aggregiert werden. Die Aggregation ist durchaus aufwendig, da zahlreiche Übergangstypen zwischen den Hauptnutzungstypen existieren und geklärt werden muß, wie mit Kleinstrukturen (z. B. Feldgehölze) umgegangen werden soll. Die Entscheidung für eine Datengrundlage ist also nicht nur von der Datenverfügbarkeit, sondern auch vom Aufwand der Datenaufbereitung abhängig.

Ein weiteres Problem ist die Arbeit mit kleinsten gemeinsamen Geometrien. Beim Einsatz von GIS ist die Verschneidung ein beliebtes Mittel zur scheinbaren Homogenisierung von Datengrundlagen. Es werden differenzierte Aussagen für einen größeren Maßstab suggeriert, auf dem die verwendeten Daten nicht differenziert sind. Zudem geht der Bezug zur realen Landschaft teilweise verloren, da für die räumliche Differenzierung von Landschaften Daten entscheidend werden, die eben diese Landschaft in sehr unterschiedlicher Weise abstrahieren. Um die Zahl der zu modellierenden Einzelflächen zu verringern, können vor der Verschneidung Flächen zusammengefaßt werden, die in Bezug auf den jeweiligen Modellparameter eine einheitliche Ausprägung besitzen. *Gerold* (1999) hat eine solche Zusammenfassung auf der Grundlage aller Modellparameter vorgenommen und mit Hilfe der Clusteranalyse die durch Verschneidung entstandenen ca. 10.000 Einzelflächen des 992 km² großen Einzugsgebietes auf 401 Flächen (sog. Hydrological Response Units, HRUs) reduziert. Als zusätzlicher Nebeneffekt läßt sich so auch der Rechenaufwand verringern.

b) Methodik der Datengewinnung und Parameterableitung (Regionalisierungsproblematik)

Hierzu zählen die Ableitung des Grundwasserflurabstandes nach bodenkundlichen oder hydrogeologischen Kriterien, die Verwendung induktiver oder deduktiver Bodenkarten zur Ableitung der nFK (vgl. *Hennings* 1998). Auf mesoskaliger Ebene ergibt sich zudem das Problem der Generalisierung heterogener Strukturen. Der von *Hennings* (1998) festgestellte große Fehler kleinmaßstäbiger Bodenkarten (BÜK200 und BÜK1000), läßt sich nach seiner Aussage durch Aggregierung großmaßstäbiger Bodenkarten verbessern. Dem sind jedoch, durch die mit der Maßstabsebene steigende Heterogenität der Bodeneinheiten, enge Grenzen gesetzt.

Eine umfassende Diskussion und Darstellung wasserhaushalts- und landschaftsbezogener Regionalisierungmethoden findet sich bei *Steinhardt* und *Volk* (1999).

Jede Modellanwendung sollte mit vorhandenen Ergebnissen aus anderen Vorhaben im Untersuchungsraum verglichen werden, und die Diskussion mit Praktikern aus der Wasserwirtschaft ermöglicht die Berücksichtigung zusätzlichen Expertenwissens. Auf diese Weise kann zumindest die Plausibilität mesoskaliger Modellierungen erhöht werden.

In der Praxis stellt sich seltener die Frage „Welches sind die richtigen Daten?" sondern viel häufiger die Frage: „Welche Daten sind verfügbar und geben möglichst schnell Auskunft über wasserhaushaltliche Größen?" In diesem Zusammenhang hat der hier vorgenommene Vergleich gezeigt, dass sich auf mesoskaliger Ebene hinreichend verlässliche Aussagen durchaus mit unterschiedlichen Daten erzielen lassen. Er hat aber auch gezeigt, daß wasserhaushaltliche Modelle nicht als 'Black Box' betrachtet werden dürfen. Entscheidend für die fundierte Interpretation und Nutzung von Modellen ist das Wissen über die Art der verwendeten Daten und deren Einfluss auf das Modellierungsergebnis.

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Probleme und Möglichkeiten der mesoskaligen Abschätzung des Bodenabtrages mit einer Variante der ABAG

Meso-scale estimation of soil erosion with a variant of the universal soil loss equation (USLE): problems and possibilities

Zusammenfassung

Modifizierte Varianten der allgemeinen Bodenabtragsgleichung bilden zumeist die Komponente zur Berechnung von Erosionsgefährdung und lateralem Stofftransport in integrierten Wasser- und Stoffhaushaltsmodellen. Aufgrund mangelnder Möglichkeiten der Anpassung von Einzelfaktoren im Modell und der fraglichen Übertragbarkeit kann dies zu Fehlern bei den Berechnungsergebnissen führen. Über den Vergleich von Varianten zur Ableitung der Einzelfaktoren wird deren Einfluss auf das Berechnungsergebnis im mittleren Maßstab dargelegt und Empfehlungen zur Verwendung von Varianten je nach Datenlage und Zielstellung abgeleitet. Die Ergebnisse sollen einen Beitrag zur Optimierung der Anwendung von regional und maßstabsgerecht angepassten Faktorenvarianten der ABAG und von Modellen liefern.

Summary

Modified variants of the universal soil loss equation (USLE) are normally used to calculate the risk of erosion and horizontal material transport within integrated hydrological models. Due to limitations in adapting the relevant factors in the model and the problematic transferability, the calculated results may be faulty. The errors can be compounded if the values are used in other components of the model. The influence of different variants on the calculated results for the derivation of the USLE factors is shown comparatively for meso-scale applications. Recommendations for the application of variants are given dependent on the data base and the project goal. The present study is a contribution towards optimised application of different, regionally adapted and scale-oriented factor variants in the USLE and integrated models. It also aims at optimisation of a scale-specific interpretation of the results (accuracy evaluation).

1 Einleitung und Zielstellung

Die Landnutzung beeinflusst über veränderte Landschaftsstrukturen die Transportbedingungen für wassergebundene Stoffflüsse (Sedimente, Nährstoffe, Pestizide) und beeinträchtigt dadurch Quantität und Qualität von Boden und Wasser in unterschiedlicher räumlicher Ausdehnung und Intensität. Seit einigen Jahren werden deshalb auf Basis von GIS gestützten Modellierungen die Auswirkungen von Landnutzungsänderungen auf den landschaftlichen Wasser- und Stoffhaushalt abgeschätzt. Zur Bewertung der (potenziellen) Erosionsgefährdung und wassergebundenen Stoffausträge (zumeist N und P) ist dazu meistens die eigentlich für Ackerschläge entwickelte Bodenabtragsgleichung (ABAG oder USLE) (Schwertmann et al. 1990, Wishmeier und Smith 1978), oder deren modifizierte Varianten in die Modelle integriert (Bork und Schröder 1996, Grunwald 1997, Lang 1997). Im Modell können die Einzelfaktoren der Formel dann jedoch zumeist nicht regionalspezifisch und maßstabsgerecht angepasst werden. Die Folge sind Fehler in den Berechnungsergebnissen (Deumlich 2000, Wishmeier 1976), die sich durch Verwendung in andere Modellkomponenten fortpflanzen oder potenziert werden können. Demgegenüber stehen zahlreiche Vorschläge für angepasste Varianten zur Ableitung der Einzelfaktoren der ABAG (BGR 1996, Sauerborn 1994, Schwertmann et al. 1990). Ziel des vorliegenden Aufsatzes ist es, den Einfluss solcher Ableitungsvarianten auf das Berechnungsergebnis darzulegen und Empfehlungen entsprechend der Datenlage und den Projektzielen abzuleiten sowie die Aussagemöglichkeiten zu bewerten.

2 Untersuchungsgebiete

Als Untersuchungsraum dienten Gebiete in Mitteldeutschland mit unterschiedlichen naturräumlichen Bedingungen und Nutzungsstrukturen (Bild 1). Die Untersuchung wurde in mittleren Maßstäben unter Verwendung unterschiedlicher Basisdaten vorgenommen. Damit soll die Anwendbarkeit und Übertragbarkeit der Methoden sowie die skalenspezifischen Aussagemöglichkeiten der Berechnungsergebnisse überprüft werden.

2.1 Regierungsbezirk Dessau und Einzugsgebiete der Rossel und Fuhne

Der ca. 4300 km² große Regierungsbezirk Dessau befindet sich im Osten Sachsen-Anhalts. Die größtenteils schwach reliefierte Region umfasst holozäne Auenlandschaften (Elbe, Mulde, etc.) und die größtenteils bewaldeten Moränengebiete des Flämings und der Dübener Heide sowie die fruchtbaren Lößgebiete im Westen. Braunerden, Schwarzerden, Braunerde-Podsole und Rosterden sowie Gleye nehmen die größte Fläche ein. Die mittleren Niederschlagswerte variieren zwischen 450 mm/a im Westen (Einzugsgebiet der Fuhne) bis 650 mm/a (Dübener Heide und Fläming, EZG Rossel). Die intensiv agrarisch genutzten Schwarzerde-

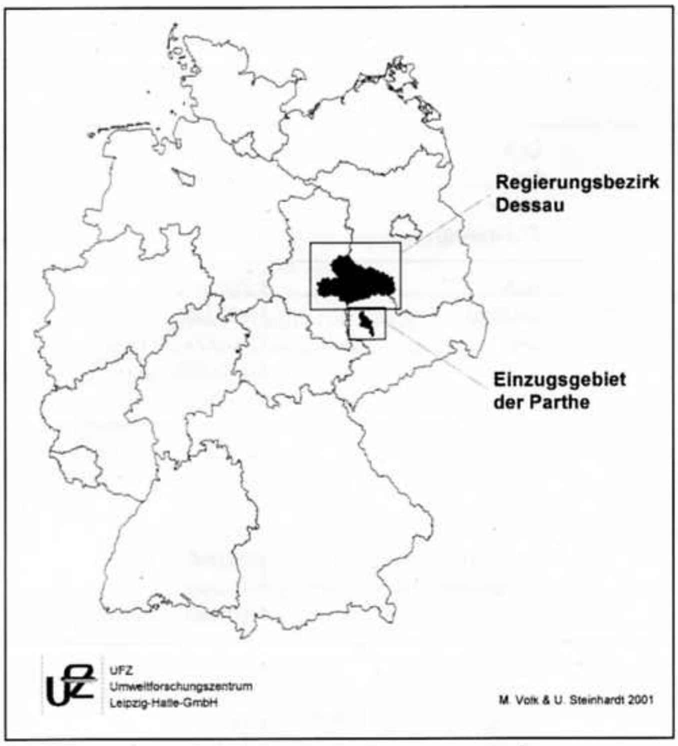

Bild 1 Lage der Untersuchungsgebiete in Deutschland.
Figure 1 *Study localities in Germany.*

gebiete im Westen der Region müssen aufgrund der bindigen Ausgangssubstrate als potenziell gefährdet gegenüber Wassererosion angesehen werden (sommerliche Intensiv-Niederschlagsereignisse). Hier befindet sich das zu etwa 81 % landwirtschaftlich genutzte Einzugsgebiet der Fuhne, dessen bearbeiteter Teil ca. 320 km² beträgt. Demgegenüber weist das im Bereich des Fläming liegende Einzugsgebiet der Rossel (ca. 200 km²) sandige Braunerden, Podsole und Rosterden sowie Fahlerden auf. Die Nutzungsstruktur ist durch ein Mosaik aus forstwirtschaftlich (ca. 47 % des Einzugsgebietes) und landwirtschaftlich genutzten Flächen (ca. 25 %) charakterisiert. Diese Grenzertragsstandorte werden – gestützt durch die europäische Agrarpolitik – zunehmend aus der landwirtschaftlichen Nutzung genommen. Während der ersten Jahre nach der Nutzungsaufgabe sind hier erhöhte Bodenabträge durch Wasser- und Winderosion zu erwarten.

2.2 Einzugsgebiet der Parthe und Teileinzugsgebiet Schnellbach

Das etwa 300 km² umfassende Einzugsgebiet ist durch permische Porphyrhügelländer, Altmoränengebiete und fruchtbare Sandlößebenen charakterisiert. Klimatisch ist es durch eine Jahresmitteltemperatur von 8,5 °C sowie einen mittleren Jahresniederschlag von 570 mm gekennzeichnet. Die bedeutendsten Bodentypen sind Braunerden, Parabraunerden und Staugleye. Mehr als 50 % der Fläche des Gebietes werden landwirtschaftlich genutzt; der Anteil der versiegelten Flächen beträgt demgegenüber etwa 10 %. Aufgrund der flächenhaft verbreiteten bindigen Ausgangssubstrate ist das Gebiet zum größten Teil als potenziell gefährdet durch wasserbedingte Bodenerosion einzustufen.

Das ca. 8 km² große Teileinzugsgebiet des Schnellbaches im Süden des Parthegebietes (Sandlößgebiet) wird intensiv landwirtschaftlich genutzt. Die landwirtschaftliche Bodennutzung im Gebiet erfolgt auf 30 Ackerschlägen mit Größen bis zu 90 ha (mittlere Schlaggröße 16,5 ha), die zumeist in Hangrichtung bearbeitet werden.

3 Arbeitsmethoden und Datengrundlagen

Die Universal Soil Loss Equation (USLE, vgl. Wishmeier und Smith 1978) ist im deutschen Raum als Allgemeine Bodenabtragsgleichung (ABAG) bekannt. In der Gleichung wird die Vielzahl der den Erosionsprozess beeinflussenden Faktoren in den sechs Hauptfaktoren zusammengefasst (Schwertmann et al. 1990). Schwertmann et al. (1990) bearbeiteten die Einzelfaktoren der Formel und passten sie an bayerische Verhältnisse an. Seitdem wird die Gleichung in Deutschland sowohl bei schlagbezogenen Untersuchungen (Löwa 1997, Meyer und Grabaum 1996), als auch in größeren Gebieten (Gündra et al. 1995) angewendet. In BGR (1994) sind zahlreiche Varianten von Gleichungen zur Bestimmung der potenziellen Erosionsgefährdung durch Wasser sowie deren Einzelfaktoren zusammengestellt, die zumeist an die ABAG angelehnt sind und eine Anwendung auch auf größere Räume erlauben sollen.

Für die eigenen Untersuchungen wurde die folgende Variante ausgewählt, die nach (BGR 1994) für alle Maßstäbe anwendbar ist:

$$EfW = K_B \cdot R \cdot L \cdot S$$

K_B = Bodenerodierbarkeitsfaktor,
R = Niederschlags- und Oberflächenabflussfaktor,
L = Hanglängenfaktor,
S = Hangneigungsfaktor [LS = Topographiefaktor]

„Der Kennwert EfW ist konzeptionell identisch mit dem Produkt aus R-, K- und S-Faktor der Allgemeinen Bodenab-

Tabelle 1 **Verwendete Varianten von Einzelfaktoren**
Table 1 ***Variants of the USLE factors used***

Faktor	Regierungsbezirk Dessau und Einzugsgebiet der Rossel
K_B	*Schwertmann et al. (1990),* KS-Faktor = 1 (Skelettgehalt < 1 %)
R	*Schwertmann et al. (1990):* Jahresniederschlag: R = 0,083 * Nj – 1,77 Sommerniederschlag: R = 0,141 * Ns – 1,48 *Saupe (1984/1985) in Sauerborn (1994):* Jahresniederschlag (Mitteldeut.): R = 0,0958 * Nj – 3,46 (für RP Dessau und Rossel) *Sauerborn (1994):* Jahresniederschlag (Sachsen): R = 0,1165 * Nj – 16,39 (für Parthe und Schnellbach) *Sauerborn (1994):* Sommerniederschlag: R = 0,2206 * Ns – 24,83 *Deumlich (1993) in BGR (1994):* Sommerniederschlag: R = 0,152 * Ns – 6,88
L	*BGR (1994):* L = 2.0
S	*Schwertmann et al. (1990)*
LS	GIS-Analyse im Grid-Modul von Arc/Info, nach *Hickey* *(Stand 1999,* vgl. auch *Hickey et al. 1994)*

Tabelle 2 Verwendete Datengrundlagen
***Table 2** Data base employed*

Untersuchungsgebiet	Datenebene	Quelle	Maßstab
Regierungsbezirk Dessau	Bodenübersichtskarte	GLA	1:200 000
($4300\ km^2$)	Nutzung (CORINE)	StBa	1:100 000
	Niederschlag	DWD	1:100 000
	Digitales Geländemodell	TK (analog)	1:25 000
Einzugsgebiet der Rossel	Bodenübersichtskarte	GLA	1:200 000
($160\ km^2$)	MMK	(analog)	1:100 000
	Nutzung (Biotop-/Nutzungstypen)	LAU	1:10 000
Einzugsgebiet der Fuhne	Niederschlag	DWD	1:100 000
($260\ km^2$)	Digitales Geländemodell	LfLD	1:4000
Einzugsgebiet der Parthe	Bodenkarten (analog)	SLUG (analog)	1:25 000
($400\ km^2$)	Nutzung (SatSzenen,	Landsat	Geom. Aufl. 5–30m
	Biotop-/Nutzungstypen)	SLUG	1:10 000
	Niederschlag	DWD	1:100 000
		Meßnetz (anal.)	1:25 000
	Digitales Geländemodell	TK (analog)	1:25 000
			1:10 000
Einzugsgebiet des Schnellbach	Bodenkarten (analog)	SLUG (analog)	1:25 000
($8\ km^2$)	Nutzung (SatSzenen,	Landsat	Geom. Aufl. 5–30m
	Biotop-/Nutzungstypen)	SLUG	1:10 000
	Niederschlag	DWD	1:100 000
		Meßnetz (anal.)	1:25 000
	Digitales Geländemodell	TK (analog)	1:10 000

Abkürzungen: DWD: Deutsches Wetterdienst; GLA: Geologisches Landesamt Sachsen-Anhalt; LAU: Landesamt für Umweltschutz Sachsen-Anhalt; LfLD: Landesamt für Landesvermessung und Datenverarbeitung Sachsen-Anhalt; SLUG: Sächsisches Landesamt für Umwelt und Geologie; SatSzenen: Satellitenszenen; StBa: Statistisches Bundesamt; TK: Topographische Karten;

tragsgleichung (ABAG) und ist ein Maß für die Standortempfindlichkeit bei konstanten Belastungsfaktoren", (BGR 1996). In Tabelle 1 sind die verwendeten Varianten für die Einzelfaktoren dargestellt.

Die Datengrundlagen zur Ermittlung der Einzelfaktoren sind größtenteils öffentlich verfügbar. Teilweise mussten Boden- und Reliefinformationen digitalisiert werden. Informationen zur Landnutzung wurden den CORINE-Daten entnommen (Statistisches Bundesamt 1996), für größere Maßstäbe wurden die CIR-Biotoptypen-/Nutzungstypenkartierungen verwendet. Eine Übersicht über die verwendeten Basisdaten, deren Maßstäbe sowie die Angabe der Bezugsquellen gibt Tabelle 2.

Verfügbarkeit flächendeckender Daten für das Einzugsgebiet der Rossel musste hier auch auf Basisinformationen zurückgegriffen werden, deren räumlich-zeitliche Auflösung nicht den Bearbeitungsmaßstäben entspricht. Dies kann unter Umständen zu einer Beeinträchtigung der Ergebnisqualität führen (Steinhardt und Volk 1999, 2000).

Die Reliefanalysen der digitalen Geländemodelle zur Ableitung der L-, S- bzw. LS-Faktoren sowie die Verschneidung von Datenebenen wurden mit dem Geographischen Informationssystem (GIS) Arc/Info (Version 7.2.1) ausgeführt. Die weitere Bearbeitung der Daten erfolgte mit dem GIS ArcView (Version 3.1a) sowie Erdas Imagine (Version 8.3). Zur Untersuchung des Einflusses von Einzelfaktoren auf das Gesamtergebnis der Bodenabtragsberechnung wurden die in Tabelle 3 aufgelisteten Varianten getestet.

4 Vergleich und Bewertung der Ergebnisse

Die Ergebnisse zeigen teilweise große Differenzen, die auf unterschiedlichen naturräumlichen Bedingungen in den Untersuchungsgebieten, aber auch auf die Herleitung der betreffenden Faktoren zurückzuführen sind (Tabelle 4).

4.1 Vergleiche der Varianten zur Ableitung von Einzelfaktoren

Beim bodenartbedingten Erosionsfaktor K_B werden im Einzugsgebiet der Fuhne aufgrund der relativ geringen Bodendiversität und damit gleichen Bedingungen für die Wassererosion die geringsten Varianzen erreicht. Dagegen spiegelt sich im Einzugsgebiet der Parthe die Substratheterogenität in einer sehr hohen Spannbreite wider.

Die höchsten Varianzen bei den R-Faktoren werden erwartungsgemäß im Untersuchungsraum Regierungsbezirk Dessau erreicht, da hier auf großer Fläche starke klimatische Differenzierungen auftreten (Trockengebiete im Westen und Fläming mit höheren Niederschlägen im Norden).

Geringere Varianzen können im zweitgrößten Untersuchungsraum, dem Einzugsgebiet der Parthe, beobachtet werden. Die höchsten R-Faktoren ergeben sich im südlichen bis südöstlichen, stärker reliefierten Bereich, die geringsten im nördlichen Teil des Einzugsgebietes. Die geringsten Varianzen werden erwartungsgemäß im Einzugsgebiet des Schnellbaches erreicht. Die Differenzierung der R-Faktoren in den Gebieten lässt sich größtenteils auch auf die errechneten LS-Faktoren bzw. S-Faktoren übertragen. Auch hier werden die höchsten Werte in den stärker reliefierten Teilräumen erzielt. Dabei wird deren Abhängigkeit von den jeweils verwendeten Digitalen Geländemodellen und den damit verbundenen Auswirkungen auf die Bodenabtragswerte offensichtlich. Insgesamt lassen alle verwendeten Varianten eine Differenzierung innerhalb der Gebiete erkennen. Die Unterschiede der errechneten Bodenabträge bei den verwendeten Varianten bewegen sich zwischen 0,29 t/ha/a und 0,92 t/ha/a (Tabelle 5).

Tabelle 3 **Faktorenvarianten bei der Berechnung des Bodenabtrages durch Wassererosion**
Table 3 ***Factor variants used in the calculation of soil erosion values***

Variante	
1	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Jahresniederschlag, Schwertmann et al. 1990) LS-Faktor (Hickey, Stand 1999)
1a	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Sommerniederschlag, Schwertmann et al. 1990) LS-Faktor (Hickey, Stand 1999)
2	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Jahresniederschlag, Saupe 1984/85 in Sauerborn 1994 (RP Dessau bzw. Sauerborn 1994 [Parthe]) LS-Faktor (Hickey, Stand 1999)
2a	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Sommerniederschlag, Sauerborn 1994) LS-Faktor (Hickey, Stand 1999)
3	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Sommerniederschlag, Deumlich 1993) LS-Faktor (Hickey, Stand 1999)
4	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Jahresniederschlag, Schwertmann et al. 1990) L-Faktor = 2.0 (konstant) (BGR 1994) S-Faktor (Schwertmann et al. 1990)
4a	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Sommerniederschlag, Schwertmann et al. 1990) L-Faktor = 2.0 (konstant) (BGR 1994) S-Faktor (Schwertmann et al. 1990)
5	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Jahresniederschlag, Saupe 1984/85 in Sauerborn 1994 (RP Dessau bzw. Sauerborn 1994 [Parthe]) L-Faktor = 2.0 (konstant) (BGR 1994) S-Faktor (Schwertmann et al. 1990)
5a	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Sommerniederschlag, Sauerborn 1994) L-Faktor = 2.0 (konstant) (BGR 1994) S-Faktor (Schwertmann et al. 1990)
6	K_B-Faktor (Schwertmann et al. 1990) R-Faktor (Sommerniederschlag, Deumlich 1993) L-Faktor = 2.0 (konstant) (BGR 1994) S-Faktor (Schwertmann et al. 1990)

Eine Bewertung der Berechnungsergebnisse in Bezug zur Realität ist natürlich nur durch Messungen an Testflächen möglich! Dies war jedoch nicht realisierbar. Bei allen Varianten sind die gleichen Tendenzen und Größenordnungen zu erkennen, die eine Einordnung der Erosionsgefährdung ermöglichen. Die höchsten Werte werden bei den Varianten 4 bis 6 erreicht, bei denen der Hanglängenfaktor L = 2.0 gesetzt wurde (Tabelle 5, Bild 2).

Auch hier stimmt allerdings die Differenzierung der Gebiete in Teilräume mit „hohem", „mittlerem" oder „geringem" Abtrag im Vergleich mit den anderen Varianten überein (Bild 3 und 4).

Insgesamt werden bei den Varianten, die bei den R-Faktoren Jahresniederschläge verwenden, im Mittel die geringsten Bodenabträge bei der Variante 1 erreicht. Die mittleren Bodenabträge, die mit Variante 2 errechnet wurden, liegen im Mittel zwischen 11 % und 14 % darüber. Diese Feststellung gilt auch für die Varianten 4 und 5, bei denen L = 2.0 gesetzt wurde.

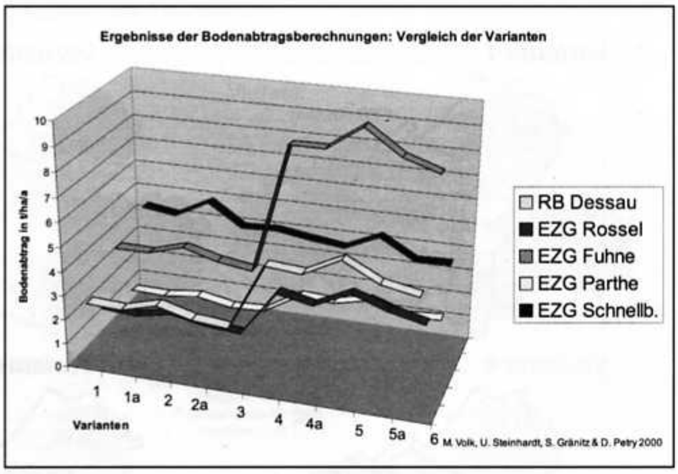

Bild 2 Der Variantenvergleich weist auf den sprunghaften Anstieg der Bodenabtragswerte bei den Varianten 4 bis 6 hin, bei denen der Hanglängenfaktor L = 2.0 gesetzt wurde.
Figure 2 *Comparison of the variants highlights the sudden increase of soil erosion values with the variants 4 to 6 assuming a slope length factor of L = 2.0.*

Bei den auf Sommerniederschläge beruhenden Varianten liegen die mittleren Bodenabträge generell unter den oben genannten Werten. Die geringsten Abträge ergeben sich demnach bei der Variante 3 bzw. bei den Untersuchungsgebieten Parthe und Schnellbach bei Variante 2a. Diese Feststellungen können ebenfalls auf die Varianten 4 bis 6 übertragen werden. Insgesamt sind die Unterschiede der errechneten mittleren Bodenabträge zwischen den Varianten – abgesehen von den Unterschieden zwischen den Variantenkomplexen 1 bis 3 und 4 bis 6 – als gering einzustufen.

4.2 Empfehlungen

Unter Berücksichtigung der mangelnden Validierungsmöglichkeiten bei großräumigen Abschätzungen des Bodenabtrages werden zu den Ergebnissen (Differenzierung innerhalb der Untersuchungsräume und Variantenvergleichen) folgende Feststellungen getroffen:

- Bei der Variante 1a bzw. 4a (R-Faktoren auf der Basis von Sommerniederschlägen) kommen die geringen Niederschlagswerte in Trockengebieten zum Ausdruck. Bei den anderen Varianten liegen die R-Faktoren der Varianten mit Sommerniederschlägen teilweise weit unter denen, die Jahresniederschläge verwenden oder die existierenden Niederschlagsverhältnisse im Gebiet werden nicht abgebildet. Dies führt folglich auch zu einer geringeren Differenzierung der Bodenabträge.
- Die in der Variante 2 bzw. 5 verwendeten R-Faktoren sind den naturräumlichen Bedingungen der Untersuchungsgebiete angepasst (Sauerborn 1994) und sollten daher bei der Verwendung von mittleren Jahresniederschlagswerten verwendet werden.
- Bei der Verwendung von Digitalen Geländemodellen mit horizontalen Auflösungen von < 10 bis ca. 50 m kann der LS-Faktor zur Berechnung der Bodenabträge verwendet werden. Einen Vorteil bietet das zur Ableitung nutzbare Verfahren nach Hickey (Stand 1999) im Grid-Modul von Arc/Info. Bei gröberen Auflösungen muss mit starken Verfälschungen der tatsächlichen Reliefverhältnisse gerechnet werden.

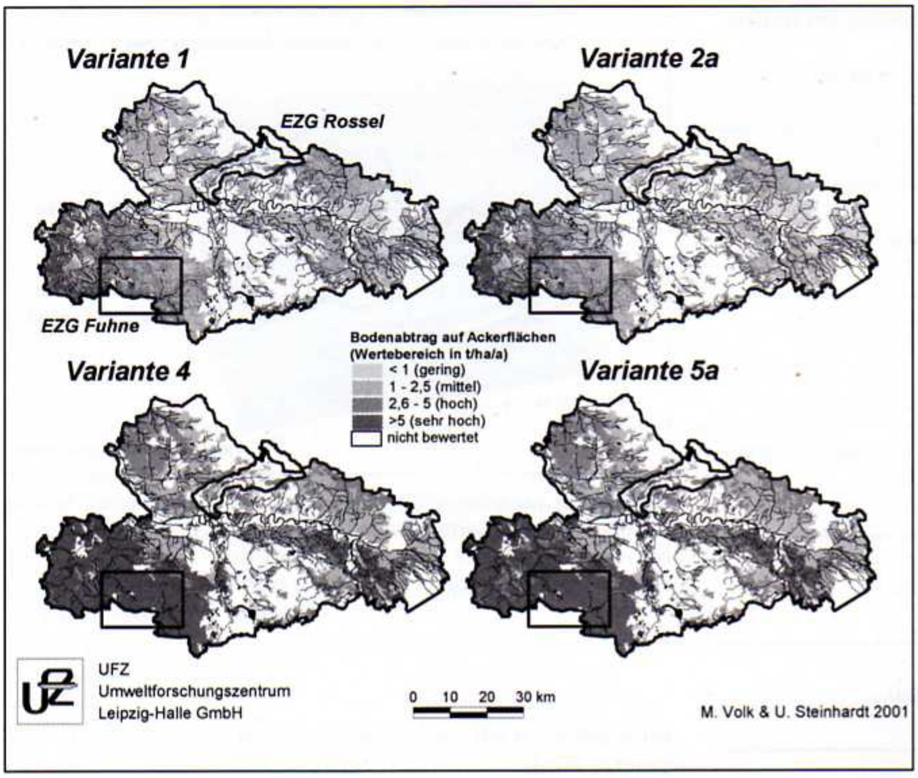

Bild 3 Mittlerer Bodenabtrag im Regierungsbezirk Dessau: Vergleich ausgewählter Varianten mit unterschiedlich hoher Differenzierung des Raumes.
Figure 3 *Average soil erosion values in the Dessau district. Comparison of selected variants with varying spatial differentiation of the research area.*

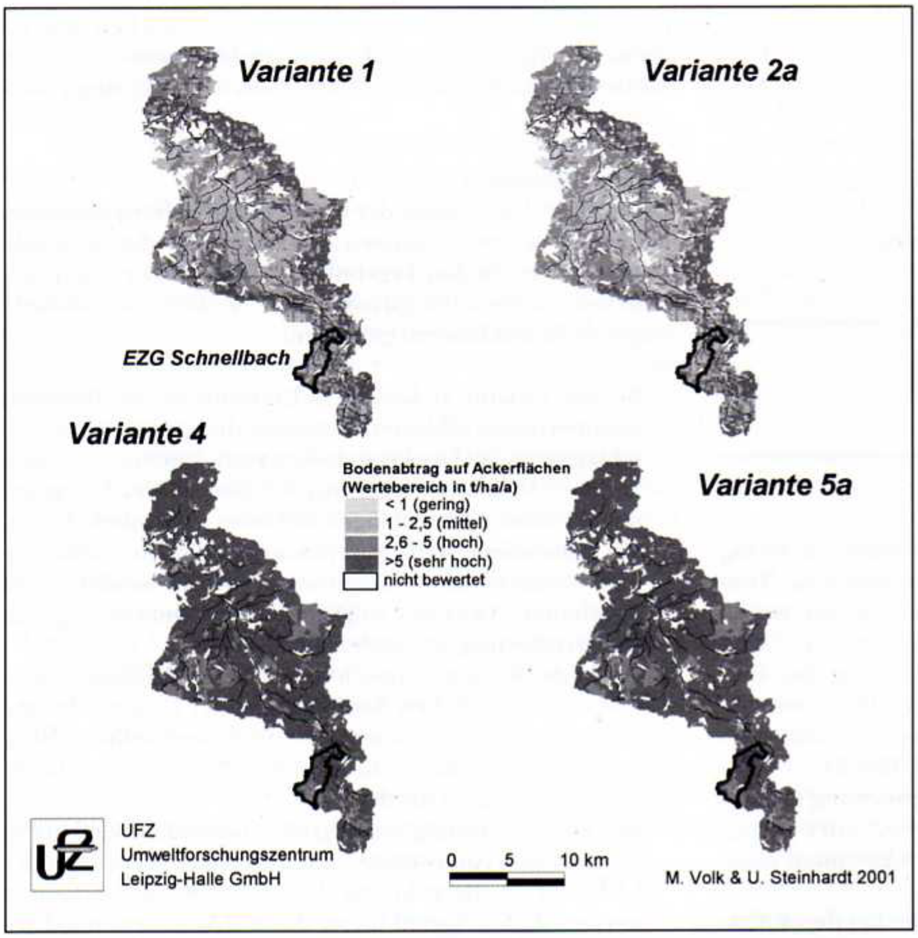

Bild 4 Mittlerer Bodenabtrag im Einzugsgebiet der Parthe: Vergleich ausgewählter Varianten mit unterschiedlich hoher Differenzierung des Raumes.
Figure 4 *Average soil erosion values in the Parthe watershed. Comparison of selected variants with varying spatial differentiation of the research area.*

– Für regionale Abschätzungen bzw. großräumige Einordnungen des Bodenabtrages auf Basis relativ grob aufgelöster Datengrundlagen stellt die Variante 5 (L = 2.0 und regional angepasster R-Faktor) eine gute Lösung dar. Bei dieser Variante werden zwar sehr hohe Bodenabträge ermittelt, sie erlaubt aber bereits eine grobe Differenzierung des Raumes und bildet eine gute Basis für vertiefende Untersuchungen in gefährdeten Gebieten.

Aufgrund der Größe der Gebiete sowie der Qualität der Ausgangsdaten können die errechneten Werte eine Einordnung der Erosionsgefährdung geben bzw. als Wertebereiche dienen. Eine Annäherung an die wirklichen Verhältnisse im Gelände („absolute Beträge") kann nur durch die Anpassung der Allgemeinen Bodenabtragsgleichung mittels weiterer Untersuchungen im Gelände unter verschiedenen naturräumlichen Bedingungen bzw. einer verbesserten Datenbasis erreicht werden.

5 Schlussfolgerungen

Der durchgeführte Vergleich von Varianten zur Ableitung der Einzelfaktoren der Allgemeinen Bodenabtragsgleichung hat gezeigt, dass alle eine tendenziell gleiche Differenzierung von größeren Räumen nach der Bodenerosionsgefährdung und eine gebietsspezifische Einordnung nach „Wertebereichen" wie z. B. „hoch", „mittel" und „gering" ermöglichen. Bei der (häufig angewendeten) Variante „Schwertmann" (unter Verwendung des Sommerniederschlags) ergab sich bei der Betrachtung des R-Faktors auch eine deutliche Differenzierung des behandelten, als Trockengebiet einzustufenden Untersuchungsgebietes. Bei den anderen getesteten Varianten war dies nicht der Fall, sodass hier Beeinträchtigungen bei der Berechnung der Bodenabträge die Folge waren. Stehen als Datengrundlage lediglich mittlere Jahresniederschläge zur Verfügung, sollten R-Faktoren verwendetet werden, die den naturräumliche Bedingungen der jeweiligen Untersuchungsgebiete angepasst sind (Sauerborn 1994).

Bei der Verwendung von Digitalen Geländemodellen mit horizontalen Auflösungen von < 10 bis ca. 50 m kann der LS-Faktor – abgeleitet nach Hickey (1999) – in die Berechnung eingehen.

Tabelle 4 Varianz der Einzelfaktoren (bezogen auf landwirtschaftlich genutzte Flächen)
Table 4 *Variance of USLE factors (in relation to arable land)*

Faktoren	Varianz				
	RB Dessau (4300 km²)	EZG Rossel (43,5 km²)	EZG Fuhne (260,2km²)	EZG Parthe (305 km²)	EZG Schnellbach (8 km²)
K_B-Faktor (Schwertmann et al. 1990)	0,13–0,56	0,15–0,56	0,48–0,56	0,02–0,42	0,13–0,40
R-Faktor (Jahresniederschlag, Schwertmann et al. 1990)	35,16–51,52	43,13–50,69	36,16–42,72	44,88–56,16	48,03–49,94
R-Faktor (Sommerniederschlag, Schwertmann et al. 1990)	36,00–48,00	41,81–46,18	36,87–42,37	43,22–52,81	46,32–47,87
R-Faktor (Jahresniederschlag, Sauerborn 1994)	39,17–58,04	48,37–57,09	40,32–47,89	49,08–64,93	53,51–56,19
R-Faktor (Sommerniederschlag, Sauerborn 1994)	33,00–52,00	42,61–49,67	35,12–43,70	37,30–56,04	43,36–46,40
R-Faktor (Sommerniederschlag, Deumlich 1993)	33,1–46,02	39,78–44,50	34,46–40,39	41,30–51,64	44,65–46,32
S-Faktor (Schwertmann et al. 1990)	0,20–6,80	0,20–1,30 (bei DGM250: 0,20–0,5)	0,31–1,00	0,20–0,70	0–1,70
LS-Faktor (Hickey, Stand 2000)	0,10–3,90	0,10–3,90 (bei DGM250: 0,10–3,9)	0,30–9,70	0,10–1,90	0,10–1,90

Tabelle 5 Ergebnisse der Berechnungen (Bodenabtrag in t/ha/a bezogen auf landwirtschaftlich genutzte Flächen, Durchschnittswerte flächengewichtet)
Table 5 *Results of the calculations (soil erosion values in t/ha/a in relation to arable land, average values area-weighted)*

Var.	RB Dessau (2693,7 km²)		EZG Rossel (43,5 km²)		EZG Fuhne (260,2 km²)		EZG Parthe (164,4 km²)		EZG Schnellbach (6,3 km²)	
	Varianz	Durchschnitt	Varianz	Durchschnitt	Varianz	Durchschnitt	Varianz	Durchschnitt	Varianz	Durchschnitt
1	0,48–86,4	2,61	0,66– 3,52	1,7	0,64–194,7	3,7	0,09–40,91	1,23	0–18,13	4,38
1a	0,48–85,18	2,54	0,63–12,12	1,6	0,64–191,14	3,7	0,09–38,20	1,18	0–17,01	4,18
2	0,53–96,64	2,92	0,74–15,23	1,9	0,71–217,90	4,1	0,10–46,33	1,37	0–26,23	4,91
2a	0,47–85,18	2,54	0,65–12,87	1,6	0,63–189,60	3,7	0,08–37,02	1,08	0–15,98	3,98
3	0,45–80,59	2,41	0,60–11,65	1,5	0,6–180,42	3,5	0,09–36,96	1,13	0–16,41	4,04
4	1,91–254,39	5,14	2,63–19,22	3,4	11,7–46,0	8,8	0,38–30,14	2,00	0–61,66	3,84
4a	1,92–251,6	5,02	2,53–17,62	3,1	11,77–47,04	8,8	0,36–28,15	1,91	0–58,20	3,66
5	2,13–283,97	5,76	2,95–21,63	3,8	13,08–51,6	9,8	0,42–34,14	2,23	0–69,24	4,31
5a	1,87–244,8	5,02	2,60–18,77	3,3	11,7–46,08	8,8	0,32–27,28	1,75	0–55,02	3,49
6	1,81–236,4	4,75	2,41–16,94	2,98	11,10–44,39	8,3	0,34–27,24	1,84	0–56,16	3,53

Für regionale Abschätzungen bzw. großräumige Einordnung des Bodenabtrages auf Basis relativ grob aufgelöster Datengrundlagen, stellt die Variante 5 eine gute Lösung dar. Bei dieser Variante werden zwar sehr hohe Absolutbeträge zur Erosion ermittelt, jedoch wird eine gute gebietsinterne Differenzierung ermöglicht.

Weitere Untersuchungen sollten sich der Anpassung der Einzelfaktoren der Allgemeinen Bodenabtragsgleichung an naturräumlichen Bedingungen von größeren Teilräumen in Deutschland/Europa widmen. Der vorliegende Aufsatz geht mit dem Vergleich von vorhandenen Varianten einen Schritt in diese Richtung, da auf diese Weise deren Stärken und Schwächen besser erkannt werden können. Mit der oben genannten Anpassung der Einzelfaktoren könnten auch die Berechnungsalgorithmen von Erosionskomponenten in Modellen angepasst und verbessert werden. Dies setzt voraus, dass die Einzelfaktoren dieser Komponenten bei zukünftigen Simulationsmodellen gebietsspezifisch variabel gestaltet werden.

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Ground Water Monitoring&Remediation

Surveying Ground Water Level Using Remote Sensing: An Example over the Seco and Hondo Creek Watershed in Texas

by Pei-yu Chen, Jeffrey G. Arnold, Raghavan Srinivasan, Martin Volk, and Peter M. Allen

Abstract

The Normalized Difference Vegetation Index (NDVI) derived from satellite data has been applied to various vegetation studies. The objective of this study was to assess the feasibility of using the NDVI response to plant water content to predict ground water level over a watershed located in the Edwards Aquifer of Texas, USA. Results showed that the precipitation data collected inside the watershed were not highly correlated to ground water depth within 10 d of the event, though a 60-foot sinkhole in the study site was expected to collect rainfall and recharge ground water in a short time. Alternatively, the NDVI derived from SPOT-VEGETATION satellite data and potential evapotranspiration (PET) based on the Hargreaves PET model were significantly correlated to ground water depth. Moreover, the stream flow measurements were correlated to ground water level as well. Two simple models were developed for estimating ground water levels in the artesian and recharge zones. Independent validations were performed to verify both models. All three variables (NDVI, PET, and stream flow) were directly or indirectly related to the precipitation. The PET was mainly controlled by air temperature, and the temperature was negatively related to precipitation. The NDVI values were affected by both temperature and precipitation, and the amount of rainfall was strongly correlated to the stream flow. This study initiated a unique approach to surveying ground water level based on satellite information and meteorological data.

Introduction

Several places on the earth have experienced severe drought in the past decade (e.g., McCabe et al. 2004; Munne-Bosch and Penuelas 2004; Delissio and Primack 2003), while demand for water has increased. Drought conditions have caused losses in agricultural productivity and damaged the environment through vegetation loss and soil erosion. Several studies have introduced satellite data to real-time drought monitoring programs to detect potential droughts across North America, India, and China (e.g., Wan et al. 2004; Singh et al. 2003; Su et al. 2003). Protecting limited water resources has become the first priority for water management.

The Edwards Aquifer, on which the city of San Antonio is located and relies for its water supply, is one of the largest ground water sources in south central Texas. The aquifer has provided the water supply for agricultural, industrial, recreational, and domestic needs. In recent years, the aquifer's capacity to provide fresh water could barely meet the demand (e.g., Chen et al. 2001). The Edwards Aquifer is divided into three main zones: the contributing zone, the recharge zone, and the artesian zone (Figure 1). The contributing zone is rugged and covered with mature live oak-Ashe juniper woodlands. Highly fractured limestones outcrop at the land surface of the recharge zone, which allows large quantities of water to rapidly flow into the aquifer. According to the studies by Eckhardt (2004), a small percentage of recharge occurs when precipitation falls directly on the outcrop, but >75% of recharge occurs when streams and rivers cross the permeable limestone. A ground water recharge project for the aquifer has been conducted over the Seco Creek area, where water is purposely collected and diverted into a sinkhole (e.g., Eckhardt 2004). The artesian zone is covered by relatively impermeable limestone, and the water is trapped inside. This aquifer is one of the most productive artesian aquifers in the world.

Several studies concluded that precipitation infiltration or seepage was the major source for the ground water recharge (e.g., Liu and Zhang 1993; Gau and Liu 2000). The study by Gelt et al. (1999) mentioned that stream flows strongly contributed to ground water recharge. Moreover, Sato et al. (1999) found that the drainage from river basins plays an important role in ground water recharge. In

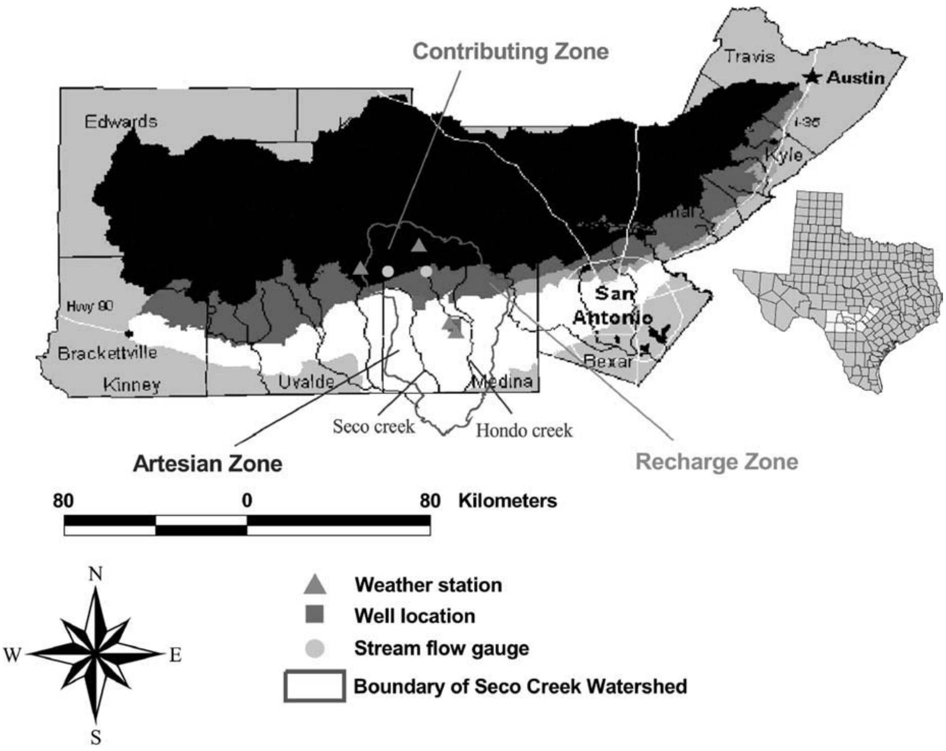

Figure 1. Geopolitical boundary of the Seco and Hondo creek watershed in the Edwards Aquifer, distributions of the contributing zone (shaded in black), the recharge zone (shaded in gray), and the artesian zone (shaded in white), and weather stations, wells, and stream gauges in the Seco and Hondo creek watershed.

general, recharge rate and amount were strongly related to local weather conditions (precipitation, temperature, solar radiation) and rock formation as well as slope, aspect, land use, plant water content, soil moisture, and evapotranspiration (ET) (e.g., Liang et al. 1994; Mitchell and DeWalle 1998; Wooldridge and Kalma 2001). The amount of precipitation affected not only the ground water recharge but also the plant water content (e.g., Lotsch et al. 2003; Martinez-Meza and Whitford 1996). Published studies showed that vegetation density responds to the plant water content, and the Normalized Difference Vegetation Index (NDVI) derived from digital satellite data corresponds to the density of green vegetation (e.g., Boone et al. 2000; Chen and Brutsaert 1998; Gao 1996).

The NDVI derived from the visible and near-infrared reflectance has been widely applied to a diversity of plant-related environmental studies (e.g., Baynes and Dunn 1997; Chen et al. 2003). The 10-d NDVI composites at a spatial resolution of 1000 m are available from the near-real time SPOT-VEGETATION (VGT) data (http://free.-vgt.vito.be). Two types of 10-d NDVI composites were produced from the VGT data. One is based on a revised maximum value compositing (MVC) method with improved removal of clouds and aerosol (e.g., Holben 1986), and the other one is bidirectional compositing (BDC) developed by Duchemin et al. (2002). Published results showed that the patchworks and orbital track patterns resulting from the association of adjacent pixels from orbits with significantly different satellite zenith angles were visible on the MVC images, but they were removed on the BDC images. The BDC approach took the average of the last 12 bidirectional reflectance distribution function (BRDF)-corrected and cloud-free single-date images to represent the 10-d composite, which typically removes visually noisy pixels (e.g., Duchemin et al. 2002).

Most hydrological studies using satellite data have focused on surface water flow modeling and soil moisture monitoring (e.g., Dettinger (2003) and the contributing and artesian zones was the recharge zone (Figure 1). The digital elevation model (DEM) and land use/Cayan 2003; Das et al. 2002; Moran et al. 2002). Little has been published on the use of a vegetation index to predict ground water level. The first objective of this study was to assess the statistical correlation between NDVI and ground water level. If the NDVI data proved to be a significant variable correlated to ground water level, then meteorological data, stream flows, and NDVI data could be combined to survey ground water level.

Study Site

The 3000-km^2 Seco and Hondo creek watershed was located mostly in the western part of Medina County, Texas. Most of the watershed was located in the artesian zone, while the upper 20% of the watershed belonged to the contributing zone. The strip between land cover and soil maps for the study site of the Seco and Hondo creek watershed were available on the Web site of the Environmental Protection Agency (http://www.epa.gov/waterscience/ftp/basins/gis_data/huc/). Elevation data were in raster format, and both land use/land cover and soil maps were in vector format. The DEM elevations decreased from north to south. The soil maps from the State Soil Geographic database were based on the detailed soil survey data, which were aggregated to a mapping scale of

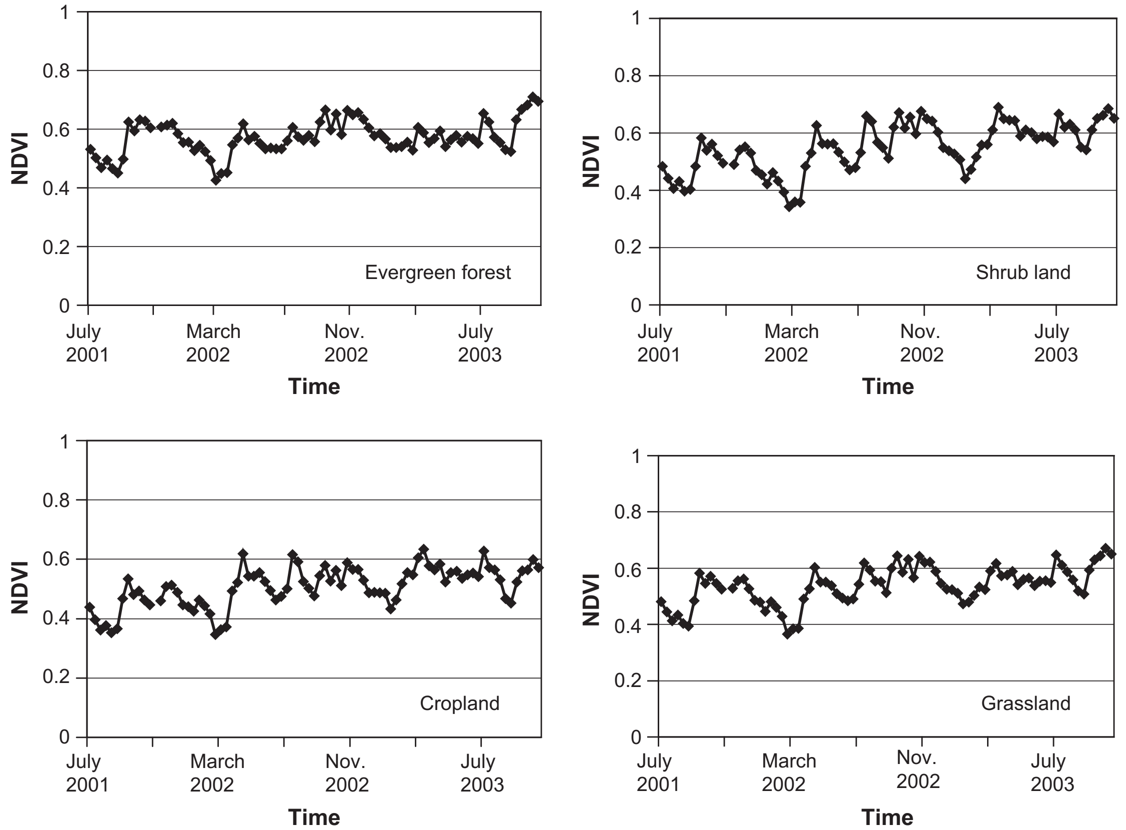

Figure 2. Temporal NDVI profiles of four major vegetation types (evergreen forest, shrub land, cropland, grassland) over the Seco and Hondo creek watershed.

1:250,000. The study area contained 14 soil types, most of which were poorly drained clayey soils with low permeability. A land-use map obtained from the LANDSAT multispectral images at 30-m resolution displayed that >99% of the watershed was dominated by evergreen forest (41%), shrub land (29%), cropland (21%), and grassland (9%).

Two weather stations operated by the National Climatic Data Center were located in the artesian zone, while the other two stations were in the contributing zone (http://www.ncdc.noaa.gov/oa/climate/stationlocator.html) (Figure 1). The stream flows in the watershed area were measured daily by the USGS stream gauges (http://tx.usgs.gov). One gauge was located in the contributing zone, and the other one was located in the recharge zone (Figure 1). Ground water levels in the study area were measured from two wells managed by the USGS. The well in the recharge zone had a depth of 538 feet (164 m) below ground level (BGL), and the other well in the artesian zone had a depth of 1600 feet (488 m) BGL (Figure 1).

Methodology

The VGT-BDC 1000-m resolution NDVI data were selected for this study because they contained less data noise (e.g., Duchemin et al. 2002). Moreover, the average value used in VGT-BDC data was more appropriate for representing vegetation conditions in a period of 10 d compared to the maximum value used in VGT-MVC data, since cloudy pixels were detected and sun as well as satellite zenith were corrected for. Averaged NDVI values acquired from the VGT-BDC data sets were recorded for each vegetation type over the study watershed every 10 d from July 2001 to October 2003 (http://free.vgt.vito.be) (Figure 2). A representative NDVI value for the vegetation greenness over the entire study area was computed according to the

Table 1
Summary of Statistical Correlations between Temperature, PET in the Artesian (A) Zone, and Ground Water Level in the A Zone and Recharge (R) Zone (p value = 0.05)

	Dependent Variables		
Independent Variables	**PET (A zone)**	**Ground Water Level (A zone)**	**Ground Water Level (R zone)**
Temperature I (A zone)	$p = 0.00^1$, $r^2 = 0.42$	$p = 0.00^1$, $r^2 = 0.27$	$p = 0.48$, $r^2 = 0.01$
PET (A zone)	—	$p = 0.00^1$, $r^2 = 0.58$	$p = 0.00^1$, $r^2 = 0.45$

[1]Significant outcomes.

Table 2
Summary of Statistical Correlations between Stream Flow in the Contributing (C) and Recharge (R) Zones and Ground Water Level in the Artesian (A) Zone and R Zone (p value = 0.05)

Independent Variables	Dependent Variables: Ground Water Level (A zone)	Dependent Variables: Ground Water Level (R zone)
Stream flow (C zone)	$p = 0.12$, $r^2 = 0.04$	$p = 0.94$, $r^2 = 0.00$
Stream flow (R zone)	$p = 0.02$[1], $r^2 = 0.09$	$p = 0.80$, $r^2 = 0.00$

[1]Significant outcomes.

proportion of each vegetation type. The NDVI values were scaled between –1 and +1. A large NDVI value indicated a high density of green vegetation, and negative NDVI values denoted the presence of snow, ice, water, or clouds.

Most soils in the study watershed consisted of >34% clays and <35% sands (http://www.epa.gov/waterscience), and a small portion (<10%) of sandy soil was located in the artesian zone close to the watershed outlet. No weather stations, stream gauges, and wells for this study were located in well-drained sandy soils for areas with low elevation. Due to homogeneous soil properties in the study area, the soil data were not considered as a correlation variable for this study.

All four weather stations provided daily precipitation data, but only the two stations in the artesian zone provided daily temperature data. Since the NDVI values were available every 10 d, the sum of precipitation in millimeters and mean of temperatures in degrees Fahrenheit were computed every 10 d for this study. The precipitation and temperature data obtained from each weather station were treated as an independent data set. The averaged amount of potential evapotranspiration (PET) (mm/d) was calculated every 10 d using the Hargreaves PET method and was dependent on the minimal/maximum temperatures and latitudes of the watershed, as well as the day of year (e.g., Hargreaves 1994). The units of stream flow and ground water depth were converted to the metric system for this study. Stream flow (m^3/s) data were averaged every 10 d for each gauge location. Two sets of averaged depths of ground water in meters every 10 d were transformed from BGL to heights above well bottom. Two sets of ground water level data were treated as the dependent variables for the recharge and artesian zones, while one NDVI, four precipitation, two temperature, two PET, and two stream flow data were the independent variables. This study included correlation development and data validation. Two-thirds of the data set (the last 20 d per month) was used for statistical correlation analysis, and the remaining one-third (the first 10 d per month) was used for independent data validation.

The intercorrelations between independent variables were examined first, and then linear regression was applied to identify significant independent variables. The level of significance (p value) was set at 0.05 in this study. The predicted and measured values of ground water levels were compared using the Nash and Sutcliffe (1970) equation:

$$E = 1 - \left(\frac{\sum_{i=1}^{n} (Q_{mi} - Q_{ci})^2}{\sum_{i=1}^{n} (Q_{mi} - Q_{m})^2} \right) \qquad (1)$$

where E is the estimation efficiency, n the number of data samples, Q_{mi} the measured value, Q_{ci} the estimated value, and Q_m the mean measured value. The value of E could range from negative infinity to 1.0, where $E = 1.0$ indicates a perfect model. The E is similar to a correlation coefficient obtained from linear regression; however, the E compares the measured values to the 1:1 line of measured equals predicted (perfect fit) rather than to the best-fit regression line (e.g., Saleh et al. 2000). This statistic has been widely used for evaluating the performance of hydrologic simulation models (e.g., Legates and McCabe 1999).

Results

Intercorrelation of Independent Variables

The temperature data collected from the two stations in the artesian zone were very similar and highly correlated

Table 3
Summary of Statistical Correlations between Precipitation in the Contributing (C) and Artesian (A) Zones, Stream Flow in the Recharge (R) Zone, and Ground Water Level in the A and R Zones (p value = 0.05)

Independent Variables	Dependent Variables: Stream Flow (R zone)	Dependent Variables: Ground Water Level (A zone)	Dependent Variables: Ground Water Level (R zone)
Precipitation (C zone)	$p = 0.00$[1], $r^2 = 0.23$	$p = 0.28$, $r^2 = 0.03$	$p = 0.18$, $r^2 = 0.05$
Precipitation (A zone)	$p = 0.01$[1], $r^2 = 0.18$	$p = 0.19$, $r^2 = 0.05$	$p = 0.45$, $r^2 = 0.02$
Stream flow (R zone)	—	$p = 0.02$[1], $r^2 = 0.09$	$p = 0.8$, $r^2 = 0.00$

[1]Significant outcomes.

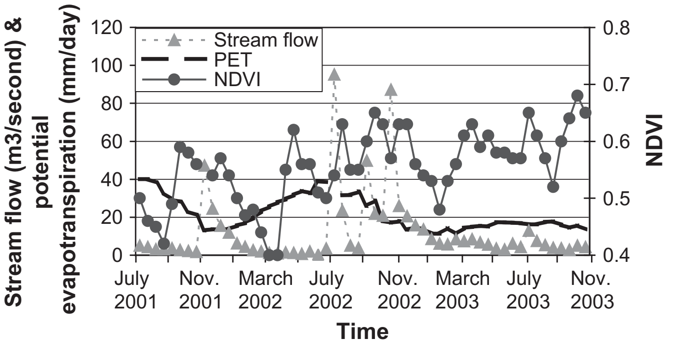

Figure 3. Ten-d averaged stream flow ×10 (m^3/s), PET (mm/d), and NDVI from July 2001 to October 2003.

($p = 0.00$, $r^2 = 0.98$). The PET data for this study were derived based on the temperature data acquired from the station between the two wells, and certainly, it was strongly correlated to the temperature data (Table 1). The results (Table 1) showed that the temperature correlated less to the ground water level in both the recharge and artesian zones than the PET did. Hence, the temperature data were removed from this study.

Similar high correlations occurred with stream flow data in the contributing and recharge zones ($p = 0.00$, $r^2 = 0.79$). Both stream flows barely correlated to the ground water level in the recharge zone (Table 2). However, the stream flow from the recharge zone had a significant correlation to the ground water level in the artesian zone, unlike the stream flow from the contributing zone (Table 2). Thus, the stream flow from the recharge zone was selected as an independent variable for this study.

Two sets of precipitation data obtained in the contributing zone were significantly correlated ($p = 0.00$, $r^2 = 0.74$), and a similar correlation ($p = 0.00$, $r^2 = 0.77$) occurred for the precipitation data acquired in the artesian zone. Moreover, the precipitation data from different zones were significantly correlated with a lower r^2 value ~0.42. The results (Table 3) showed that significant correlations occurred between the precipitation and stream flow, and no significant correlations were observed between the simultaneous precipitation and ground water level in both recharge and artesian zones. Therefore, the precipitation data were eliminated from this study. A total of three independent variables, NDVI, PET, and stream flow, were applied to this study (Figure 3).

Depth of Ground Water in the Artesian Zone

The ground water level was independently correlated to each variable, PET, NDVI, and stream flow. Among them, PET carried the most information ($r^2 = 0.58$) for estimating the water level, and the stream flow provided the least information ($r^2 = 0.09$). The NDVI was the second most important variable ($r^2 = 0.25$). The estimation efficiency (E) was >0.71 when all three independent variables were applied for multiple regression (Figure 4a), and the fitted relationship was:

$$\text{ground water level} = 440.43 + (-0.40) \times \text{PET} + 15.21 \times \text{NDVI} + 0.89 \times \text{stream flow} \qquad (2)$$

The time series between dependent and independent variables was considered in this study. Results showed that the present water level was significantly correlated to the current and future 10-d NDVI and PET. However, the estimation efficiency of close to 0.64 for the future 10-d data was lower than the 0.71 for the current 10-d data. Moreover, the objective of this work was to develop a method using remote sensing data to monitor the real-time ground water level. Thus, the multiple regression of future 10-d NDVI and PET data was not pursued further.

Depth of Ground Water in the Recharge Zone

The ground water level in the recharge zone was less predictable compared to the one in the artesian zone. Only two variables were significantly correlated to the water level in the recharge zone. One was PET and the other one was NDVI. The PET provided the most information and had an r^2 value of 0.45. The NDVI variable had an r^2 value of ~0.30. The PET and NDVI variables produced an

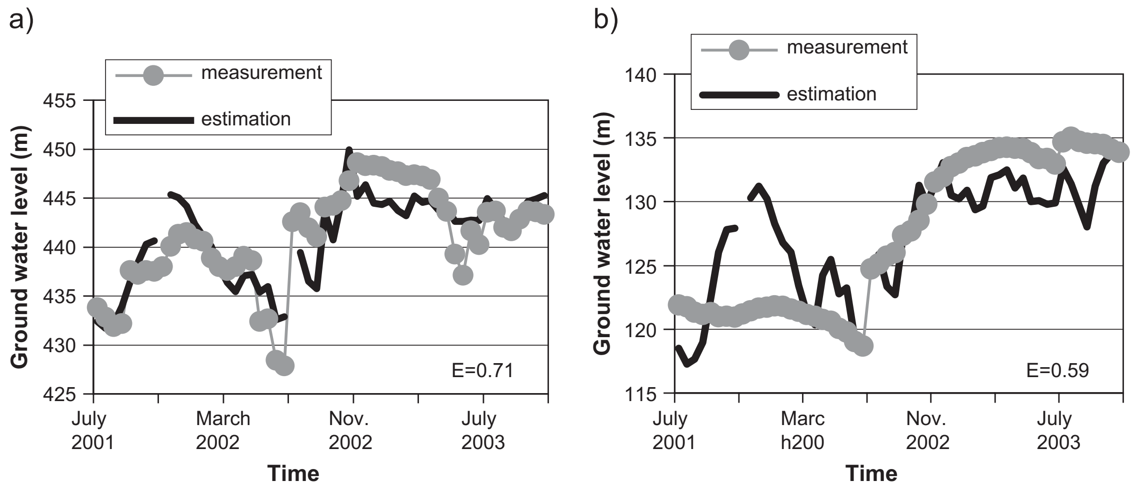

Figure 4. Ten-d profiles of measured and estimated ground water level (above well bottom) from July 2001 to October 2003 in (a) the artesian zone and (b) the recharge zone. The few missing estimations were caused by lack of NDVI or evapotranspiration data.

estimation efficiency of ~0.59 (Figure 4b), and the relationship fitted to the data was:

$$\text{ground water level} = 119.24 + (-0.4) \times \text{PET} + 30.57 \times \text{NDVI} \quad (3)$$

A similar time difference occurred between the variables of PET and water level, but it was not of concern because of a low estimation efficiency ($E < 0.50$).

Precipitation

The results revealed that the precipitation falling in the contributing and artesian zones did not recharge the ground water in the recharge and artesian zones within a short time (10 d). Additional work was pursued to determine the relationship between precipitation and ground water level in the study area. Monthly precipitation and ground water level data were collected for the past 10 years. No significant correlations were found when time lags from a minimum of zero months to a maximum of 12 months were considered. Moreover, the results showed that no significant correlation was found between the differences of 10-d precipitation and of 10-d ground water level. This finding conflicted with most published results where precipitation was significantly correlated to ground water recharge. One of the possible reasons is that the barrier faults control the movement of water in the Edwards Aquifer.

Independent Validation

Accuracy assessment is essential to the validation of research methods using remotely sensed data (e.g., Congalton and Green 1999). A total of 27 data (average of the first 10 d of each month) from August 2001 to October 2003 were used to validate the remote sensing approaches for ground water level estimation. The estimated ground water levels acquired from Equations 2 and 3 were evaluated using corresponding real measurements. The results showed that the estimation efficiency (E) value of ground water level was 0.67 for the artesian zone (Figure 5a) and 0.47 for the recharge zone (Figure 5b). The validated E value of 0.67 was very close to the originally developed E value of 0.72, which indicated that the approach (Equation 2) for the artesian zone is reliable for estimating ground water level. However, the E value 0.47 was lower than the original (0.59) for the recharge zone, and moreover, Figure 5b showed a poor correlation between the measured and estimated ground water levels. Further investigation is required to reliably estimate the ground water level in the recharge zone.

Discussion

In general, it takes a long time, several weeks to years, for surface water to permeate through soil layers and rock formations and to replenish the aquifer. Studies related to ground water recharge normally used monthly to annual precipitation data (e.g., Hadzisehović et al. 1995; Ginting et al. 2000). The results of this study indicate that local precipitation did not notably affect ground water levels within 10 d in the study area. Several measurements in the recharge zone showed that rainfall events had little input on ground water levels, with several rainfall events coinciding with a decrease in ground water levels (Figure 6a). Conversely, the measurements of ground water level in the artesian zone correlated well to precipitation data (Figure 6b), though the increase of water level was not in proportion to the increase of precipitation. Moreover, the additional 10-year monthly data for the study area did not establish the relationship between the difference of precipitation and ground water recharge and discharge. All the results illustrated that the precipitation did not recharge the ground water in the short term in this study. Ground water level could be influenced more by decreased pumping after a rainfall event than by direct recharge. Ground water discharge was normally related to several circumstances, such as natural flow when water levels were very high, water pumped from the aquifer by industrial and residential consumers, and high ET (e.g., Laczniak et al. 1999; DeMeo et al. 2003). Certainly, a great number of barrier faults in the recharge and artesian zone play an important role in ground water recharge and discharge. More investigations are required to identify the source for ground water recharge and the cause of ground water discharge.

Although precipitation data were not correlated to the ground water level, this study showed that stream flows

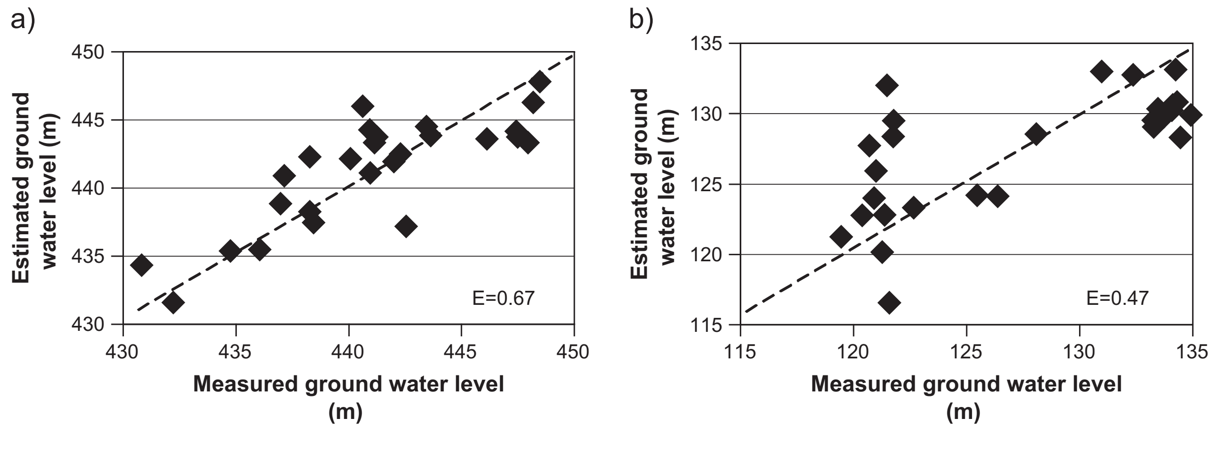

Figure 5. Estimation efficiency of ground water level (above well bottom) from July 2001 to October 2003 in (a) the artesian zone and (b) the recharge zone.

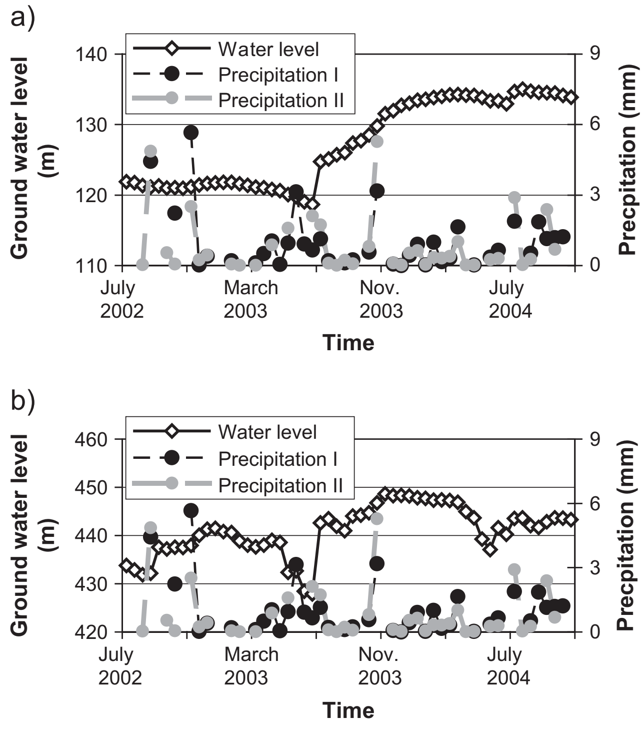

Figure 6. Ten-d profiles of precipitation and ground water level (above well bottom) in (a) the recharge zone and (b) the artesian zone.

significantly contributed to ground water recharge within 10 d in the artesian zone. According to the measured data, the stream flows responded to the rainfall, and the length of response time was dependent on the amount of precipitation. In addition to increasing stream flows, a portion of the precipitation was lost to ET and another portion to runoff. This study showed that stream flow contributed <10% ($r^2 = 0.09$) of information for ground water level estimation in the artesian zone, which could be related to the impermeable limestone in the artesian zone controlling water movement. No correlation was found between stream flows and the ground water level in the recharge zone. One possible reason was that complex faults in the recharge zone acted as barriers or partial barriers to ground water flow. Further studies are required regarding ground water levels in the recharge zone in response to stream flows.

This study showed that the NDVI was correlated to ground water levels in both the recharge and artesian zones. The vegetation index is a response of the green vegetation condition to environmental factors including soil moisture and water availability. Higher NDVI values implied sufficient surface water to stimulate vegetation growth and to support local water needs, and less demand for ground water was consequently expected. Hence, the NDVI value was positively correlated to ground water level. Studies done in Death Valley, California, and Nye County, Nevada, found that the ground water discharge was largely lost to ET (e.g., Laczniak et al. 1999; DeMeo et al. 2003). Higher values of ET indicated less ground water available for use in their studies. As a result, the PET was negatively correlated to ground water level in our study. However, the deep ground water had little potential to lose to ET directly. The negative correlation was in consequence that high PET corresponded to a high demand of ground water (pumping) due to less available surface water. This study showed that both variables of NDVI and PET carried more information for ground water level estimations in the artesian zone than in the recharge zone.

Results showed that the data in the artesian zone had a higher estimation efficiency compared to the data in the recharge zone. Moreover, the differences of estimation efficiency between original data and validated data were 0.04 for data in the artesian zone and 0.12 for data in the recharge zone. The Figure 6a showed that ground water levels in the recharge zone were barely affected by precipitation measured in the watershed. All results in this study showed that the ground water level in the recharge zone is difficult to estimate or model. Continued studies are needed to improve the approach for estimating ground water levels in the recharge zone.

Conclusions

Several research studies have applied digital satellite data to study surface water, such as flood mapping, seashore (or coastline) change, and sea surface temperature monitoring. Few studies have been conducted using remote sensing methods to survey ground water level. This study successfully linked satellite data to ground water levels in the artesian zone of the Seco and Hondo creek watershed, while more studies are required to develop a reliable approach for ground water level estimation in the recharge zone. Our study found that local precipitation did not recharge ground water. Numerous faults in the watershed may obstruct water movement or prevent uniform mixing of water throughout the aquifer. More research is required regarding tracking the fate of runoff and infiltration of surface water downward into the aquifer. The NDVI values from satellite data and PET from meteorological data provided an additional data source for monitoring ground water levels. Both NDVI and PET did not correlate to amount of ground water recharge but of ground water use. Since two-thirds of the world's fresh water is found underground, ground water studies are needed to efficiently protect the decreasing water resource.

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The comparison index: A tool for assessing the accuracy of image segmentation

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Abstract

Segmentation algorithms applied to remote sensing data provide valuable information about the size, distribution and context of landscape objects at a range of scales. However, there is a need for well-defined and robust validation tools to assessing the reliability of segmentation results. Such tools are required to assess whether image segments are based on 'real' objects, such as field boundaries, or on artefacts of the image segmentation algorithm. These tools can be used to improve the reliability of any land-use/land-cover classifications or landscape analyses that is based on the image segments.

The validation algorithm developed in this paper aims to: (a) localize and quantify segmentation inaccuracies; and (b) allow the assessment of segmentation results on the whole. The first aim is achieved using object metrics that enable the quantification of topological and geometric object differences. The second aim is achieved by combining these object metrics into a 'Comparison Index', which allows a relative comparison of different segmentation results. The approach demonstrates how the Comparison Index CI can be used to guide trial-and-error techniques, enabling the identification of a segmentation scale H that is close to optimal. Once this scale has been identified a more detailed examination of the CI–H- diagrams can be used to identify precisely what H value and associated parameter settings will yield the most accurate image segmentation results.

The procedure is applied to segmented Landsat scenes in an agricultural area in Saxony-Anhalt, Germany. The segmentations were generated using the 'Fractal Net Evolution Approach', which is implemented in the eCognition software.

Keywords: Segmentation; Landsat; Field detection; Validation; Accuracy; Object metric

1. Introduction

Image segmentation algorithms such as those contained within eCognition^{TM} are increasingly popular for a wide range of image processing tasks, and the advantages of working with image segments, rather than individual pixels are widely recognized (Fortin et al., 2000; Shi et al., 2005). However, there is a wide range of variables to manipulate, whereas segmenting an image and identifying an 'optimal' result can be difficult. The tools developed in this paper aim to make this process more objective and rigorous.

In this study, image segmentation algorithms were applied to Landsat TM data for an agricultural region in Saxony-Anhalt, Germany. The accuracy of the field boundary delineation was highly important because of its impact on the accuracy of the sediment/pollutant transport model that uses the field boundaries as model input (Van Oost et al., 2000; Takken et al., 2001). In addition, measures of soil protection legislation like

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'European Common Agricultural Policy' or 'Good Farming Practice' (EU, 2002) refer to up-to-date field shapes. However, the existing public vector data sets provided by land surveys are not able to define field boundaries accurately. This is mainly due to the large temporal variations of crop structures (Mysiak et al., 2004). Satellite sensor data have the capacity to fill this 'temporal gap' thereby providing up-to-date field boundary information.

To assess the accuracy of the field boundary delineation we focus on the geometric quality of a single class (cp. Zhan et al., 2005) rather than classification accuracy (sensu Foody, 2002). The image segmentation was carried out using the 'fractal net evolution approach' (FNEA) (Benz et al., 2004). FNEA is constrained using a set of user-defined parameter settings, which affect the segmentation results. In this paper, we investigate the influence of segmentation parameter settings on segmentation results with a view to identifying the parameter settings that provide optimal segmentation results for a specific target. The main objectives of this study are:

(1) the development of a user-friendly evaluation procedure to visualize and quantify segmentation inaccuracies. Inaccuracies refer to over-segmentations or under-segmentations, which stand for generating too many or too few segments (Delves et al., 1992). This level is referred to as local validation, because single objects are considered;
(2) the assessment of segmentation results on the whole, by which we want to achieve an optimal parameter setting of the applied segmentation method. This level is referred to as global validation, because the entire image is considered.

These objectives were achieved by visualizing two object metrics (local validation) and using the Site Comparison Method SICOM (Deumlich et al., 2006) which aggregates the classified object metrics to a map complexity metric Comparison Index CI (global validation). The extrapolation of the complexity metric to the whole study area was realized using random sampling methodology (Congalton and Green, 1990; Stehmann, 1992).

2. Methods

2.1. Image segmentation

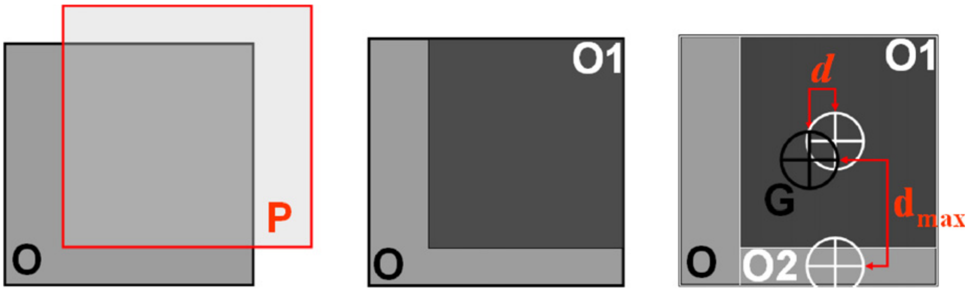
There are various methods for automatic field detection that are based on the application of segmentation algorithms to remote sensing data (e.g., Fuller et al., 2002; Evans et al., 2002; Betenuth, 2004; Mueller et al., 2004; Devereux et al., 2004). Image segmentation is a spatial clustering technique, which leads to a complete image sub-division into non-overlapping regions or segments. The wide variety of segmentation approaches can be distinguished in two broad categories (Fortin et al., 2000; Muñoz et al., 2003; Mueller et al., 2004). The first category uses boundary techniques that apply edge detection methods to locate boundary elements, which are then filtered by using element attributes like angle measures or minimum length. After that, the remaining elements are connected into segments or objects. The second category uses region-growing algorithms that rely on 'seed' pixel groups (local and/or global minima), which grow until an abortion criterion is fulfilled (e.g., homogeneity, meeting of another boundary).

The functionality of the FNEA-algorithm used in this paper is described in detail by Baatz and Schäpe (2000) and Benz et al. (2004). The hierarchical region growing algorithm is widely accepted in the remote sensing community so that a multitude of references emerged in the last few years (see http://www.definiens.com/documents/publicationsearth.php). The crucial parameters are the homogeneity criteria H (scale parameter) and the weight parameters w_{color} and w_{shape} which allows adaptation of the heterogeneity definition to the desired target objects. While H affects the heterogeneity for each segmentation level, w_{color} and w_{shape} balance the spectral and shape heterogeneity ($w_{color} + w_{shape} = 1$). w_{shape} can be influenced by the weight parameters compactness w_{compt} and smoothness w_{smooth} ($w_{compt} + w_{smooth} = w_{shape}$). As the name implies, the higher w_{compt} the more segmentation results tend to compact shapes.

2.2. Segmentation validation

2.2.1. Object metrics

The quality of a segmentation result is connected with data quality (e.g., noise, spatial and spectral resolution) (Fortin et al., 2000) as well as the optimal customization of parameter settings, which enable the adaptation of segmentation results on target objects (Delves et al., 1992). The problem is that the customization is often a result of trial-and-error procedures (Hay et al., 2003; Stein and de Beurs, 2005). Thus, in recent years various object validation techniques were developed for assessing uncertainties in segmentation-based object extraction (Shi et al., 2005). Pixel metrics are appropriate to validate the quality of detected edges or boundaries (Delves et al.,



(a) Overlap (b) Containment (c) Containment and differences between object centers and gravity center G

Fig. 1. Topological and geometric relationships between two object levels.

1992; Prieto and Allen, 2003; Lucieer and Stein, 2002). However, Zhan et al. (2005) suggest object metrics for the validation of segments.

Object-based segmentation validation can be described as 'the problem of matching objects' (Zhan et al., 2005) where at least two hierarchical object-levels have to be considered. Object differences can be specified by, (1) topological and (2) geometric relationships (Molenaar, 1998; Ragia and Winter, 2000; de Bruin et al., 1999; Zhan et al., 2005):

(1) the topological relationships of interest are 'containment' and 'overlap' (Fig. 1). In Fig. 1a object P overlaps object O. Both object levels are not hierarchically connected. If a topological overlay GIS-operation was carried out, the resulting object $O1$ is contained in the primary object O (Fig. 1b). $O1$ and O are hierarchically coupled if a 'part of'-relation was created (de Bruin et al., 1999). As a result, $O1$ is a sub-object of the superior or super-object O. Metrics of containment arise from the comparison of object sizes between related super- and sub-object;
(2) geometric object differences can be determined by the comparison of object positions. Common metrics arise from distances between the gravity centers or skeletons as well as super- and sub-objects (Ragia and Winter, 2000; Zhan et al., 2005). Fig. 1c shows the gravity center G of the super-object O and the centers of sub-object $O1$ and $O2$.

In this study, we calculated the hierarchical object metrics 'relative area in super-object' RA_{SO} and 'relative position to super-object' RP_{SO}. Both metrics are implemented in the eCognition software (Benz et al., 2004). RA_{SO} expresses the topological relationship of two objects connected by a 'part of'-relation and arises from the ratio of the area size A of the object of interest $O1$ and the area size covered by its super-object O (Fig. 1b; Eq. (1)). The metric values are within the values 0 and 1. The value 1 indicates a complete match between sub- and super-object whereas values smaller than 1 represent sub-objects that are smaller than their super-objects.

$$RA_{SO} = \frac{A_{O1}}{A_O} \quad \text{with } RA_{SO} \in [0, 1] \tag{1}$$

The hierarchical metric RP_{SO} relies on the gravity center G of the super-object and is calculated by dividing the distance d from the center of the object of interest C_{O1} to G by the distance d_{max} of the center of the most distant image object, which has the same super-object (Eq. (2); Fig. 1c: object $O2$). The metric value tends toward 0 if d reaches the minimum i.e., if the centers of both objects are in the same location. The metric value comes up to 1 if the distance between the centers of gravity of both objects is large.

$$RP_{SO} = \frac{d}{d_{max}} \quad \text{with } RP_{SO} \in [0, 1] \tag{2}$$

2.2.2. *Integral consideration of object metrics*

Zhan et al. (2005) emphasized that topographical and geometric metrics have to be considered integrally in order to reach a more accurate detection of mismatched objects. A simple option is a threshold-based classification of the above-defined object metrics. Since a classification schema does not exist, we carried out a qualitative grouping of both metrics RA_{SO} and RP_{SO} by means of the K-means clustering algorithm (see Bishop, 1995; McGarigal et al., 2002) within the statistica-environment (http://www.statsoft.com; option 'maximum distance between clusters'). An advantage of this approach is that the two-dimensional metric feature space can be structured in an automatic manner. The resulting clusters show which objects represent the best

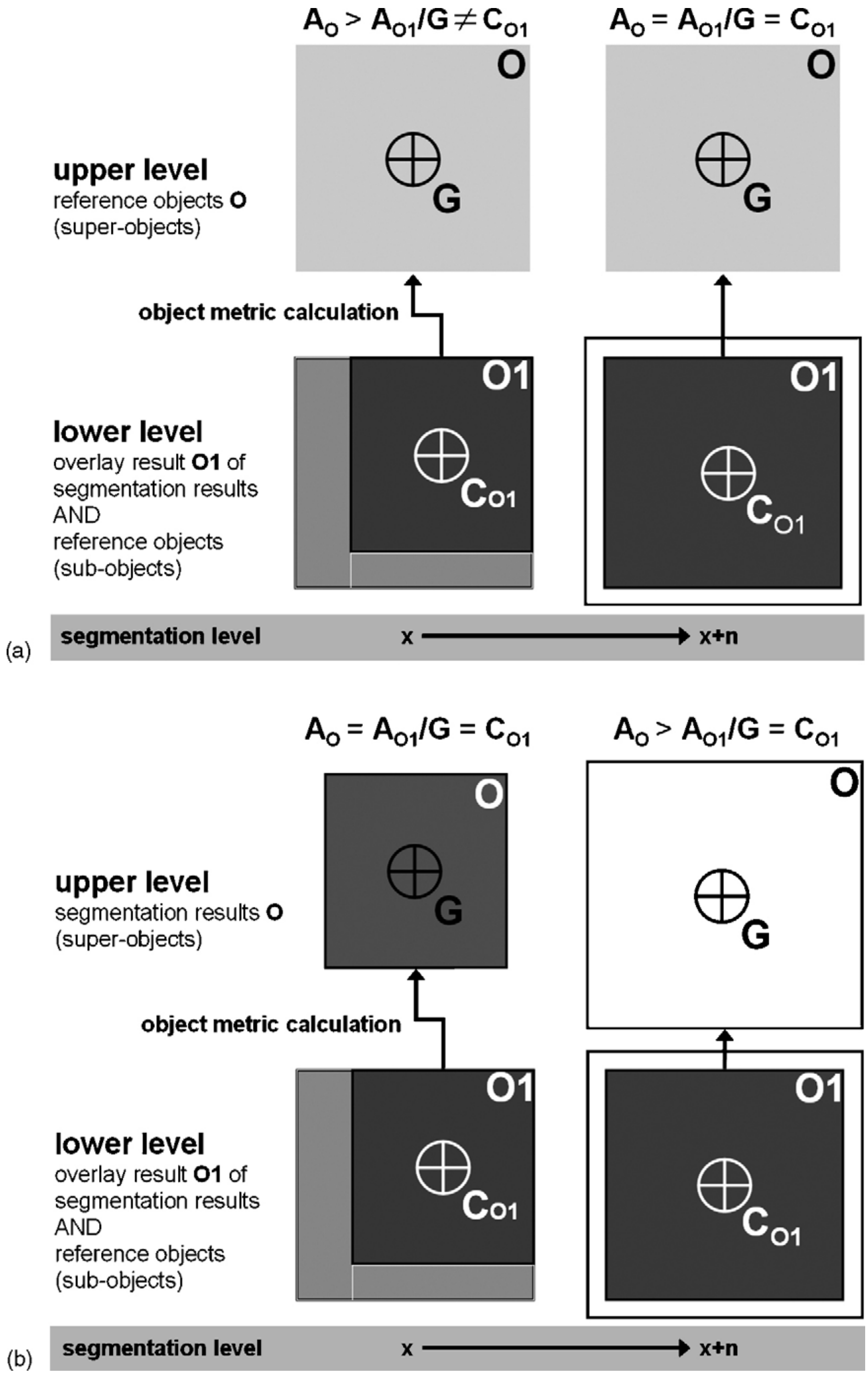


Fig. 2. Principle of local validation (a) test for over-segmentation; (b) test for under-segmentation.

geometrical (high cluster means of RA_{SO}) and topological (low cluster means of RP_{SO}) match of super- and sub-objects.

2.2.3. Local validation

Lucieer and Stein (2002) demonstrated that the application of topological object metrics enables the reference-based and object-specific validation of under- and over-segmentation. Our approach is similar, but uses a topological metric in combination with the geometric metric that was described in the last section. The calculation of both these metrics is carried out from a lower level to an upper level. The lower level represents the overlay result of reference objects and segmentation results (sub-objects). The upper level corresponds to either (1) reference objects or (2) segmentation results (super-objects). Fig. 2 shows the two cases where we distinguish the segmentation levels x and $x + n$:

(1) over-segmentation can be identified by considering the reference objects O as the upper level. Over segmentation has occurred when the size of segmented objects is smaller than the size of reference objects ($A_O > A_{O1}$) (Fig. 2a: level x) or where object centers G and C_{O1} are different. If over-segmentation has occurred the resulting $\mathrm{RA_{SO}}$-metrics are less than 1 and the $\mathrm{RA_{PO}}$-metrics are greater than 0.

If the segmented object size is equal to or exceeds A_O then both A_O is identically to the size of overlay result A_{O1} and the locations of G and C_{O1} are congruent. That means that $\mathrm{RA_{SO}} = 1$ and $\mathrm{RA_{PO}} = 0$. Under-segmented objects cannot be detected in this instance because the reference object is now contained within the segmentation result. This means that A_O and A_{O1} are congruent (Fig. 2a: level $x + n$);

(2) under-segmentation can be identified by considering segmentation results as upper level (Fig. 2b). The topological and geometric similarity between A_O (level x) and A_{O1} lasts until A_O (level $x + n$) exceeds A_{O1}. The under-segmented objects can be identified using $\mathrm{RA_{SO}}$- and $\mathrm{RA_{PO}}$-metrics. For under-segmented objects $\mathrm{RA_{SO}}$ tends to 0 and $\mathrm{RA_{PO}}$ to 1 for larger A_O (level $x + n$).

By the simultaneous visualizing of the clustered object metric values (see Section 2.2.2) it is possible to estimate the balance between under- and over-segmentation related to each reference object.

2.2.4. Global validation

In contrast to the local validation that refers to single objects, the global validation is related to the entire image in order to deduce an optimal parameter setting H_{opt} and a segmentation accuracy $\mathrm{CI_A}$. H_{opt} and $\mathrm{CI_A}$ are derived from (1) the aggregation of the calculated local validation results (clustered object metrics $\mathrm{RA_{SO}}$ and $\mathrm{RP_{SO}}$ in Section 2.2.3) and (2) the creation of CI–H-diagrams:

(1) the aggregation is described by the map complexity metric 'Comparison Index' CI. Complexity metrics are common in landscape analysis (O'Neill et al., 1986; McGarigal and Marks, 1994; Gustavson, 1998). In the connection with segmentation validation, Stein and de Beurs (2005) applied various semantic metrics to measure the complexity of segmentation results in order to quantify the semantic object accuracy. In our case the complexity metric is applied to assess the heterogeneity of the clustered object metrics related to the reference space. The reference space covers the spatial extent of all reference objects.

CI is calculated using Eq. (3). C_i is the comparison class, which represents clustered and ranked object metrics. The ranking is carried out by the highest value of $\mathrm{RA_{SO}}$ and the lowest value of $\mathrm{RP_{SO}}$ (highest ranking). A_{Ci} is equivalent to the proportion of C_i within the reference space. A complete topological and geometric match (i.e., image objects are identical to reference objects) is achieved if CI equals 100.

$$\mathrm{CI} = \frac{\sum_{i=1}^{n}(C_i \times A_{\mathrm{Ci}})}{n} \quad \text{with } \mathrm{CI} \in [0, 100] \tag{3}$$

(2) In the CI–H-diagrams, the resulting CI values and the corresponding scale parameter H are plotted. Because of the consideration of two calculation directions (see Section 2.2.3) the resulting graphs intersect where over- and under-segmentation are balanced. $\mathrm{CI_A}$ and H_{opt} correspond to the intersection point on the CI- and H-axis (as shown in Fig. 6).

3. Study area

The study area (435 km^2) is situated in the south of the German state Saxony-Anhalt near the city of Halle (Saale) (Fig. 3). The accurate delineation of agricultural field boundaries is important because this area is a study site of a major soil erosion study within a project about 'Integrated River Basin Management on the example of the Saale River Basin' (Rode et al., 2002). The field boundaries are used here as model inputs into empirical and physics-based erosion modelling (cp. Merritt et al., 2003). In this paper, we use the 'Comparison Index' CI (Eq. (3)) to disaggregate the thematic class 'intensive agriculture' of the digital biotope and land use types in a scale of 1:10,000 (FANC, 2002; Rosenberg et al., 2003) with multi-temporal Land-sat Thematic Mapper (TM) and Landsat Extended Thematic Mapper (ETM) imagery. The images with a spatial resolution of 30 m were supplied by Eurim-age (http://www.eurimage.com) and cover three significant acquisition dates of the growth season for the year 1999 (April 30, July 3 and September 13). That means that 18 multi-temporal Landsat bands were available. Thermal infrared and panchromtic bands were not considered.

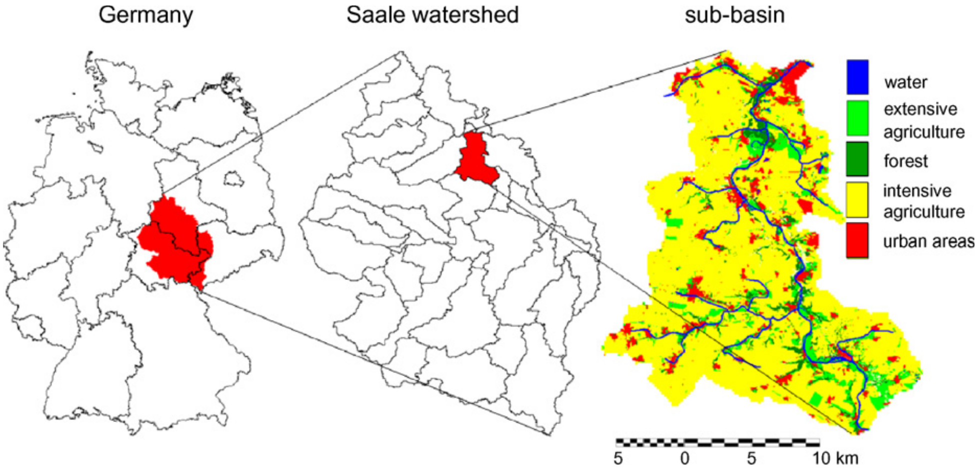


Fig. 3. Position of study area: sub-basin within the German Saale watershed.

Geo-referencing (to the Gauss Kruger projection, with Potsdam datum and the Bessel ellipsoid) was carried out for the images using 200 ground control points (GCPs) taken from topographic maps at a scale of 1:25,000. A linear resampling with the nearest neighbour algorithm was applied using ERDAS Imagine 8.5 (http://gis.leica-geosystems.com). The resampling was done using an output pixel size of 30 m with a RMSE of less than one pixel. The validation objects (agricultural fields) used in this study were selected from the Landsat imagery using stratified random sampling (Stehmann, 1992). The buffering of sample points produced sample areas which provide the basis for a manual on-screen digitizing of field parcels within the ArcInfo -environment (http://www.esri.-com). Like Devereux et al. (2004), independent interpreters collected samples from the test image on the basis of visual interpretation. This approach was used to ensure that the validation objects were derived from the same image resolution as the image segments.

4. Results

4.1. Reference objects and segmentation

The results of manual field detection are visualized in Fig. 4a. Reference objects (400) were digitized. The validation procedure will be exemplified by the reference object shown in red in the south of the study area (Fig. 4b). The round field boundaries are a function of the circular buffer of 1000 m that we used around the randomly selected points.

The segmentation parameter settings are listed in Table 1. The H-, w_{compt}-and w_{shape}-parameters were generated using trial-and-error tests to narrow the

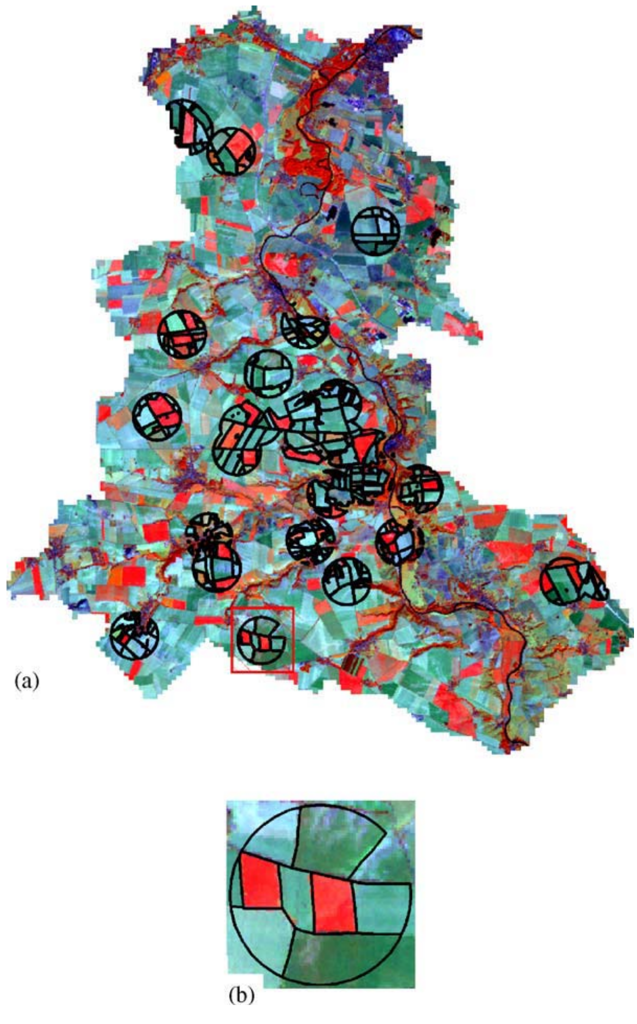


Fig. 4. Positions of reference data within the thematic land-use class 'intensive agriculture' (a) and reference example objects (b); Landsat ETM image from September 13, 1999 (band combination 4-5-3).

Table 1
Parameter setting versions for segmentations of Lands at imagery with FNEA-algorithm

Parameter	Version				
	*S*1	*S*2	*S*3	*S*4	*S*5
w_{shape}	0.2	0.4	0.4	0.4	0.5
w_{compt}	0.5	0.5	0.3	0.7	0.3
H	10, 20, . . ., 100				

parameter range. It is common for all parameter setting versions that ten segmentation levels were produced whereas each level is represented by a specific scale parameter H value (10, 20, . . ., 100). From the multi-temporal data set all 18 Landsat bands were used as input data.

4.2. Local validation

The local validation procedure is illustrated by means of segmentation results of *S*1 version (cp. Table 1). Fig. 5e–h show the influence of scale parameter H. A greater H value affects more heterogeneous and greater segmentation objects.

As explained in Section 2.2.3, the calculation of object metrics and the K-means clustering was carried out in two directions. The cluster means refer to all reference objects.

The clustered and ranked object metrics RA_{SO} and RP_{SO} calculated using the manually digitized reference objects are presented according to the corresponding scale parameter H in Fig. 5a–d. The RA_{SO} and RP_{SO} results calculated using the segmentation results are

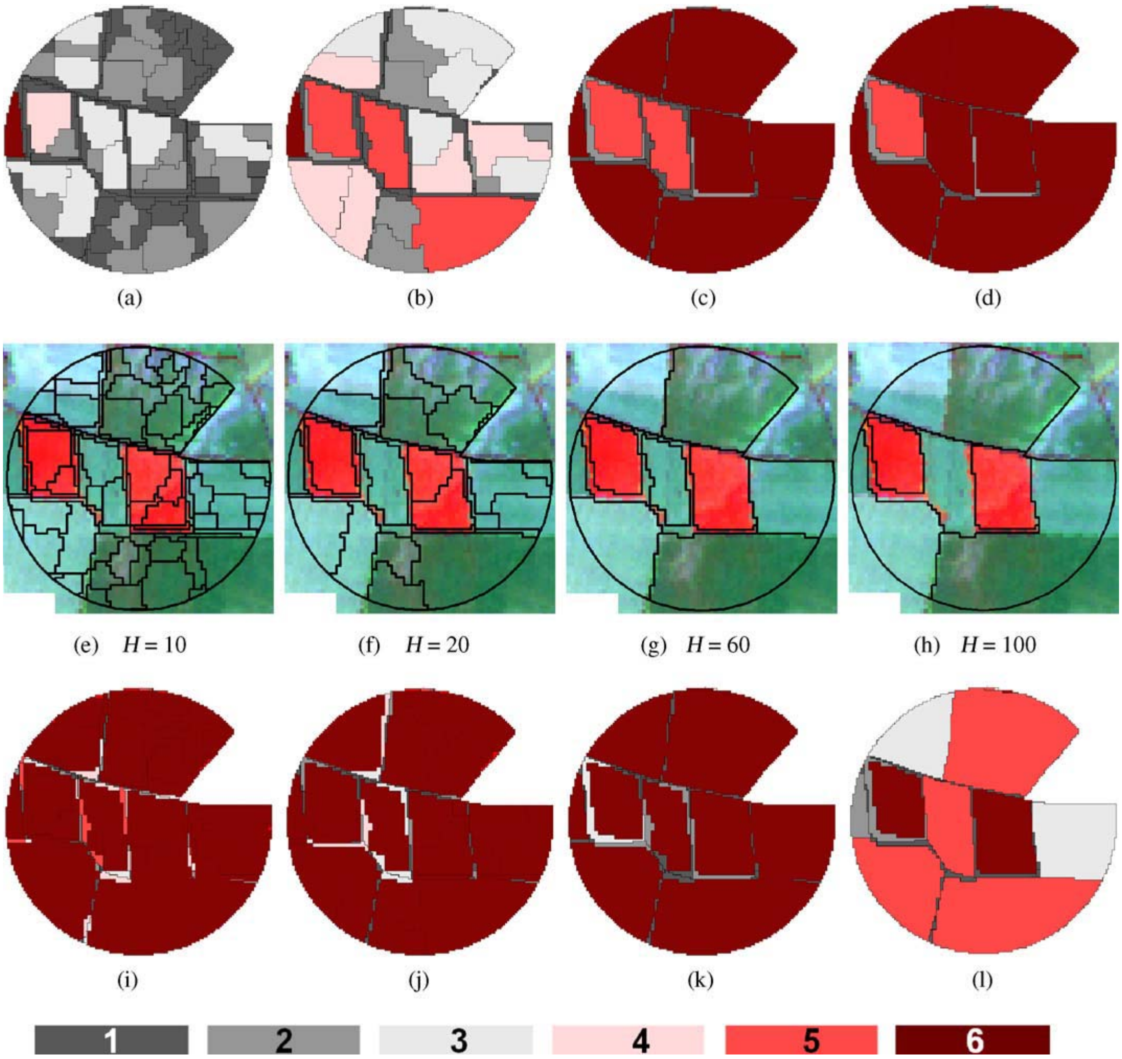


Fig. 5. Visualization of clustered and ranked object metrics RA_{SO} and RP_{SO} by example of segmentation version *S*1 (cp. Table 1). e–h: segmentation results layed over the Landsat ETM image from September 13, 1999 (band combination 4-5-3); a–d: metric calculation relative to reference objects (cp. Fig. 4b) = test for over-segmentation; i–l: metric calculation relative to segmentation results = test for under-segmentation. Attributes of corresponding comparison classes are listed in Table 2.

Table 2
Ranked *K*-means cluster means *C*, class proportions A_C (%), 'Comparison Indices' CI of object metrics and corresponding scale parameters *H* for segmentation version SI (see Table 1)

H	object metric	comparison class 0 A_C	0 C	1 A_C	1 C	2 A_C	2 C	3 A_C	3 C	4 A_C	4 C	5 A_C	5 C	CI
10^A	RA_{SO} RP_{SO}	11.0	0.01 0.98	33.8	0.05 0.68	30.2	0.07 0.35	0.8	0.33 0.95	19.9	0.41 0.32	4.2	0.91 0.05	39
20^A	RA_{SO} RP_{SO}	3.0	0.01 0.99	8.3	0.05 0.71	3.3	0.08 0.35	1.1	0.33 0.96	36.6	0.46 0.35	26.8	0.89 0.05	65
40^A	RA_{SO} RP_{SO}	5.7	0.01 0.99	3.2	0.03 0.75	6.1	0.09 0.41	0.8	0.35 0.96	17.7	0.57 0.30	66.4	0.93 0.03	**84**
50^A	RA_{SO} RP_{SO}	5.2	0.01 0.99	1.9	0.03 0.76	5.9	0.11 0.42	0.9	0.36 0.94	13.2	0.63 0.25	72.8	0.94 0.03	**87**
60^A	RA_{SO} RP_{SO}	5.0	0.01 0.99	2.0	0.03 0.76	2.2	0.11 0.41	1.0	0.36 0.94	10.0	0.61 0.26	$\boxed{79.7}$	$\underline{0.94}$ $\underline{0.03}$	**90**
70^A	RA_{SO} RP_{SO}	4.6	0.01 0.99	1.6	0.03 0.76	2.2	0.11 0.41	0.8	0.36 0.94	7.3	0.61 0.26	83.6	0.94 0.03	**91**
100^A	RA_{SO} RP_{SO}	4.4	0.01 0.99	1.5	0.03 0.76	1.7	0.11 0.41	0.8	0.36 0.94	5.1	0.61 0.27	86.5	0.94 0.03	92
10^B	RA_{SO} RP_{SO}	2.9	0.05 0.99	1.2	0.11 0.66	1.0	0.13 0.28	2.1	0.36 0.96	5.1	0.60 0.28	87.8	0.98 0.02	94
20^B	RA_{SO} RP_{SO}	3.7	0.02 0.99	1.5	0.08 0.68	1.0	0.10 0.29	1.5	0.35 0.97	4.7	0.58 0.28	87.7	0.96 0.02	93
40^B	RA_{SO} RP_{SO}	4.4	0.01 0.99	2.9	0.07 0.67	0.9	0.09 0.28	1.1	0.35 0.96	6.3	0.58 0.30	84.5	0.95 0.02	**91**
50^B	RA_{SO} RP_{SO}	4.4	0.01 0.99	1.1	0.07 0.29	4.2	0.08 0.67	1.4	0.35 0.96	10.2	0.56 0.31	78.7	0.95 0.03	**90**
60^B	RA_{SO} RP_{SO}	4.5	0.01 0.99	1.6	0.07 0.30	4.4	0.08 0.68	1.6	0.35 0.96	11.1	0.55 0.31	$\boxed{76.7}$	$\underline{0.95}$ $\underline{0.03}$	**89**
70^B	RA_{SO} RP_{SO}	4.6	0.01 0.99	1.6	0.09 0.30	4.8	0.09 0.68	1.4	0.33 0.97	16.1	0.56 0.31	71.4	0.95 0.03	**87**
100^B	RA_{SO} RP_{SO}	4.7	0.01 0.99	1.9	0.07 0.31	6.0	0.08 0.68	3.0	0.35 0.97	19.7	0.55 0.31	64.6	0.95 0.03	85

A: Object metric calculation relative to reference objects = test for over-segmentation.
B: Object metric calculation relative to segmentation results = test for under-segmentation.

shown in Fig. 5i–l. The ranked cluster means correspond to colored comparison classes. The comparison classes symbolize high (= dark red) and low (= gray) topological and geometric similarities between reference and segmentation objects. In other words, image segments shown in dark red represent a geometrical and topological match with their reference object.

The comparison classes refer to Table 2 where the attributes of the displayed results are listed. For the scale parameter $H = 60$ for instance (see Fig. 5c and k and gray highlighted cells in Table 2) the topological and geometric similarity between reference objects and segmentation results is high in both calculation directions because of the high proportion of the comparison class 5 (see framed values in Table 2). The corresponding cluster mean values 0.94, 0.95 (=high topological similarity) and 0.03 (high geometric similarity) are underlined. A high topological and geometric similarity also means that under- and over-segmentation are balanced. In contrast, an unbalanced situation indicates under- or over-segmentation. Under-segmented objects are detected by object metrics calculated relative to reference objects (Fig. 5a–c, comparison classes 0–4 in Table 2). However, over-segmented objects are detected by object metrics calculated relative to segmentation results (Fig. 5i–l, comparison classes 0–4).

4.3. Global validation

The aggregation of comparison class proportions leads to the complexity metric 'Comparision Index' CI and CI–*H*-diagrams (see Section 2.2.4). The aggregation was carried out for each segmentation level (defined by scale parameter *H*) and each segmentation version (see Table 1). The CI-values were calculated from the comparison class proportions A_C (see Eq. (3)) for CI and *H* values associate with the point where

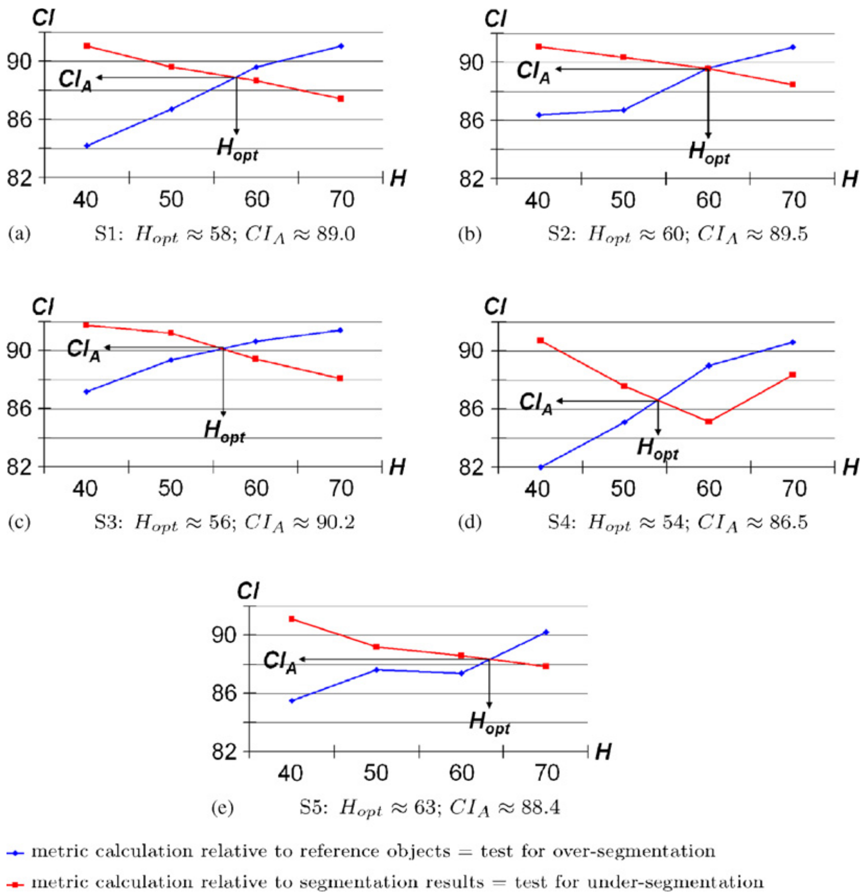


Fig. 6. Estimation of segmentation accuracy CI_A and optimal heterogeneity H_{opt} for segmentation versions of Table 1.

over- and under-segmentation are balanced. Segmentation tests showed that this point is achieved around the scale parameter $H = 60$. In order to meet the H_{opt}-value, the CI values for $H = 40, 50, 60, 70$ are calculated (bold highlighted CI values in Table 2). The resulting CI–H-diagrams show differences regarding the 'segmentation accuracy' CI_A (Fig. 6). CI_A values vary from 88.4 (*S*5) to 90.2 (*S*3). Accordingly, the results of *S*3 version show the best topological and geometric similarity. This means that the optimal scale parameter setting for detecting agricultural field boundaries in the study area is H_{opt} of 56, and this scale parameter should be used in combination with a w_{shape} parameter of 0.4 and a w_{compt} parameter of 0.3 (which were the settings used for the *S*3 segmentation).

5. Conclusions

The main objective of this paper was the determination of an optimal parameter setting of the FNEA-segmentation method in eCogntion™ based on the assumption that an optimal parameter setting is reached when over-and under-segmentation are balanced. The approach developed in this paper demonstrates how a two-stage process can be used to identify a segmentation scale H that is close to optimal using trial-and-error tests in combination with the CI metric. Once this scale has been identified a more detailed examination of the CI–H-diagrams can be used to identify precisely what H value and which w_{shape} and w_{compt} values will yield the most accurate image segmentation results.

The proposed segmentation validation procedure uses object-based metrics and a complexity metric, which were already applied to other studies (e.g., Lucieer and Stein, 2002; Zhan et al., 2005; Stein and de Beurs, 2005). The novelty of our approach is that we combine both enabling (1) local and (2) global validation of segmentation results:

(1) the local validation is based on the consideration of both topological and geometric object metrics. The metrics characterize differences in size and position between segmentation results and reference objects. In the study, we calculated metrics for 400 reference objects. The statistical metric grouping by *K*-means clustering and ranking revealed which objects show

high and low topological and geometric similarities. The metric calculations were carried out with reference to manually digitized field boundaries and super-objects within the segmentation hierarchy. Thus, under- and over-segmented objects could be visualized and classified;

(2) global validation enables the assessment of segmentation results throughout the whole study area. For this purpose, the clustered and ranked metrics were aggregated to 'Comparison Indices' CI by comparing the proportions of cluster areas where the clusters were weighted according to their rank. Depending on the input data and parameter settings used, CI–*H*-diagrams allow the estimation of 'optimal segmentation results'. This way parameters of the used segmentation algorithm can be optimized.

As the name implies, the 'Comparison Index' enables a relative comparison of segmentation results. Here, different field detection results based on FNEA algorithm were compared with reference objects. Other possible fields of application are the investigation of different segmentation procedures (see Delves et al., 1992; Meinel and Neubert, 2004). The local validation procedure is suitable for the assessment of existing thematic data sets or change detection analysis of objects shapes (e.g., field sizes) which both also could help to improve the spatial input data sets for erosion and water quality models for more realistic simulations.

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Placing soil-genesis and transport processes into a landscape context: A multiscale terrain-analysis approach

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Abstract

Landforms and landscape context are of particular importance in understanding the processes of soil genesis and soil formation in the spatial domain. Consequently, many approaches for soil generation are based on classifications of commonly available digital elevation models (DEM). However, their application is often restricted by the lack of transferability to other, more heterogeneous, landscapes. Part of the problem is the lack of broadly accepted definitions of topographic location based on landscape context. These issues arise because of: (1) the scale dependencies of landscape pattern and processes, (2) different DEM qualities, and (3) different expert perceptions. To address these problems, we suggest a hierarchical terrain-classification procedure for defining landscape context. The classification algorithm described in this paper handles object detection and classification separately. Landscape objects are defined at multiple scales using a region-based segmentation algorithm which allows each object to be placed into a hierarchical landscape context. The classification is carried out using the terrain attribute mass-balance index across a range of scales. Soil genesis and transport processes at established field sites were used to guide the classification process. The method was tested in Saxony-Anhalt (Germany), an area that contains heterogeneous land surfaces and soil substrates. The resulting maps represent adaptation degrees between classifications and 191 semantically identified random samples. The map with the best adaptation has an overall accuracy of 89%.

Key words: landforms / terrain analysis / landform semantics / segmentation / mass-balance index

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1 Introduction

Landforms are an important controlling boundary condition for current geomorphic processes (*Dehn* et al., 2001). Soil-related processes such as soil erosion and accumulation occur at multiple spatial and temporal scales, in each case controlled by different factors and with different intensities (*Steinhardt* and *Volk*, 2003). The development of effective soil-protection measures, such as those provided by soil-erosion models, requires the availability of scale-specific soil information (*Kirkby* et al., 1996; *Helming* and *Frielinghaus*, 1999). However, high-resolution soil data are often not available (*Steinhardt* and *Volk*, 2002; *Möller* and *Helbig*, 2005; *Behrens* et al., 2005). In contrast to soil data, digital elevation models (DEM) are usually available on different scales and typically have higher spatial resolution than soil maps. It is well known that strong relationships exist between the spatial distribution of soils and the topography of a given landscape (*Conacher* and *Dalrymple*, 1977; *Speight*, 1988). The use of digital terrain analysis can help to reduce the need for costly conventional survey methodologies by establishing a relationship between terrain attributes, soil genesis/transport processes, and different soil types. This process when combined with field validation can be used to provide high-resolution soil information. This has resulted in the increasing use of topography in many digital–soil mapping (DSM) projects (*McBratney* et al., 2003; *Behrens* and *Scholten*, 2006; *Lagacherie* et al., 2006). There are three key factors to consider when performing a DEM-based landform classification:

(1) Landforms occur on different scales (*Schmidt* and *Dikau*, 1999; *Evans*, 2003). Several approaches tackle the scale problem by using different window sizes—representing scales of interest—for the derivation of multiscale terrain attributes (*Gallant* and *Dowling*, 2003; *Fisher* et al., 2004; *Schmidt* and *Hewitt*, 2004; *Jenness*, 2005). The attributes show scale-specific alterations (*Gallant* and *Hutchinson*, 1997; *Thompson* et al., 2001; *Shary* et al., 2005). Their classification enables consideration of spatial context and uncertainty. The main disadvantage in this approach is that the large moving window sizes reduce the resulting output coverage. Coverage is defined here as the spatial extent of the input and resulting data set (*Bierkens* et al., 2000).

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(2) Common landform semantics do not exist because of their dependence on user's perception which reflects the user's discipline paradigm (*Bishr*, 1998; *Dehn* et al., 2001). For example, an ecologist and an engineer may define "floodplains" using completely different criteria. Semantics denote the relationships between computer representations and the corresponding real-world feature within a certain context (*Bishr*, 1998). The semantic issue is often counteracted by either

(a) using fuzzy rules in the classification process (*Burrough* et al., 2000; *MacMillan* et al., 2000; *Fisher* et al., 2004; *MacMillan* et al., 2004; *Schmidt* and *Hewitt*, 2004; *Drâgut* and *Blaschke*, 2006) or

(b) using an expert knowledge base that considers the geometrical and topological features as well as object and semantic hierarchies (*de Bruin* et al., 1999; *Wielemaker* et al., 2001; *Drâgut* and *Blaschke*, 2006).

The heuristic classification approach is subjective, but enables better inclusion of expert knowledge (*MacMillan* et al., 2000; *Drâgut* and *Blaschke*, 2006) whereas automatic classifications have the advantage of greater objectivity. However, problems may arise from the semantic interpretation of the automatically defined classes (*Burrough* et al., 2001).

(3) Landform-classification approaches are generally difficult to transfer to heterogeneous landscapes because of the aforementioned scale and definition issues (*Schmidt* and *Hewitt*, 2004; *MacMillan* et al., 2004). This is of particular concern for statistically based approaches. Because of their rigid thresholds, heuristic approaches are unable to take into account specific landscape conditions in large study areas. The implementation of fuzzy rules and class definitions with relative values and relative positions to neighboring objects can increase the transferability of heuristic approaches (*Drâgut* and *Blaschke*, 2006).

This paper focuses on the development of an automatic procedure of terrain-object delineation and classification which

(1) takes into consideration landscape heterogeneity and scale without coverage reduction and

(2) allows the adaptation of landform definitions to user's perception.

Our method aims to classify four simple landforms: floodplain, depression, plain, and slope. The classification algorithm treats terrain segmentation and classification separately. The terrain-segmentation process generates discrete landscape units, represented by polygons, at multiple scales. These polygons are related *via* hierarchy, *i.e.*, a larger-scale "parent" polygon may contain a series of smaller "children" polygons, where each child polygon may be unique, but each child polygon also "inherits" attributes from its parent. This hierarchy can also be established across multiple scales ("grandparents" and "great grandparents"), thereby enabling the definition of hierarchical multiscale terrain-object structures (*cf.*, section 2.3). The classification of these polygons is carried out by means of the terrain attribute "mass-balance index" (*cf.*, section 2.2) across a range of spatial scales using a multihierarchical query procedure, a statistically and probability-based operator (*cf.*, section 2.4).

2 Material and methods

2.1 Site description

A study area with heterogeneous soil and relief conditions was selected to demonstrate the applicability of our new methodology. The study area of Könnern, which represents such conditions, is situated in the S of the German State of Saxony-Anhalt near the city of Halle (Fig. 1). The area of 100 km^2 corresponds to the land area equivalent to the offi-

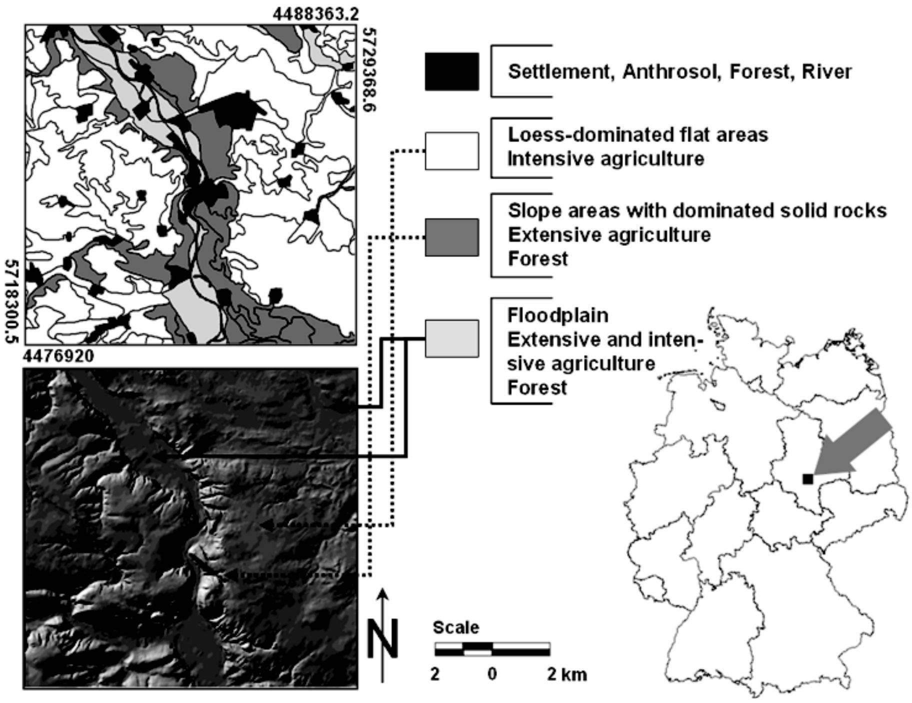


Figure 1: Study area Könnern: Shaded relief and soil-related nature units according to mesoscale agricultural site-mapping program MMK (http://www.lagb.sachsen-anhalt.de [soil data] and http://www.lvermgeo.sachsen-anhalt.de [DEM]).

cial topographic map of Könnern at a scale of 1:25,000. The region is among the driest regions of Germany with a mean annual precipitation <500 mm (*Trefflich*, 1997).

Soil and relief formation of the study area was dominated by processes under glacial and periglacial conditions during the Saalian and Weichselian glacial stages (*Altermann*, 1970). The plateau unit, mainly covered by Chernozem soils developed in the Weichselian Loess and Saalian moraine material, forms the largest and central part of the study area. It is divided by the valley of the Saale river, extending in a S–N direction. In the N and S part, the valley floor was enlarged by solution subsidence processes (*Kunert*, 1970). Neighboring relief units of the Loess plateau are the Fuhne floodplain in the NE and the Schlenze valley in the SW. Fluvisols and Gleysols have developed in the floodplain sediments. The plateau margins are characterized by sharp and shallow valleys (*Möller*, 2005). Depending on landforms and substrate, Leptosols or Regosols have developed in the sandstones or claystones of Palaeozoic ages.

Due to the fertile soils of the study area (Chernozems), the landscape is influenced by an intensive agriculture. The soils are at strong risk of erosion because of heterogeneous landscape morphology, the erodibility of the dominant loess substrate and the intense summer rainstorm events.

2.2 Data base and preparation

The study was carried out using publicly available elevation data with a resolution of 10 m and vertical and horizontal accuracy of approx. 0.5 m (see www.lvermgeo.sachsen-anhalt.de/de/main.htm). The DEM was originally generated *via* the digitization of elevation contours. Structure elements (*e.g.*, dams) or lakes are not included. The ANUDEM algorithm by *Hutchinson* (1989) was applied in order to create a hydrological sound DEM.

The following geomorphometric attributes listed in Tab. 1 were derived from the DEM. The variables *h, n, k,* and *ht* were used

Table 1: Inputs for terrain-classification algorithm.

Terrain attribute		Program
Elevation	*h*	ANUDEM 5.2[a]
Slope	*n*	Landserf 2.2[b]
Mean curvature	*k*	
Vertical distance to channel network	*ht*	SAGA1.1[c]
Mass-balance index	MBI	own application
Vertical distance to neighboring objects	*hd*	eCognition3.0[d]
Vertical distance between neighboring objects within a superobject	*ra*	

[a] http://cres.anu.edu.au/outputs/anudem.php
[b] http://www.landserf.org/
[c] http://www.saga-gis.uni-goettingen.de/html/index.php
[d] http://www.definiens-imaging.com

as input to the terrain structuring (*cf.*, section 3.1). Attributes *ra* and *hd* refer to neighboring, sub- and superobjects (*cf.*, section 2.3) and enter into the floodplain-classification process (*cf.*, section 2.4.1).

Previous landform-classification approaches have used process-based terrain attributes (*e.g., Blaszczynski*, 1997; *Park* et al., 2001). In this study, we used the mass-balance index MBI (*Friedrich*, 1996, 1998) based on the assumption that different soil-related landforms can be identified based on their MBI values. We assume that negative MBI values represent areas of net deposition such as depressions and floodplains; positiveMBI values represent areas of net erosion such as hill slopes, and MBI values close to zero indicate areas where there is a balance between erosion and deposition such as low slopes and plain areas.

The mass-balance index is derived from transformed *f*(*k, ht, n*) values (Eq. 1). As shown in Fig. 2a, high positive MBI values occur at convex terrain forms, like upper slopes and crests,

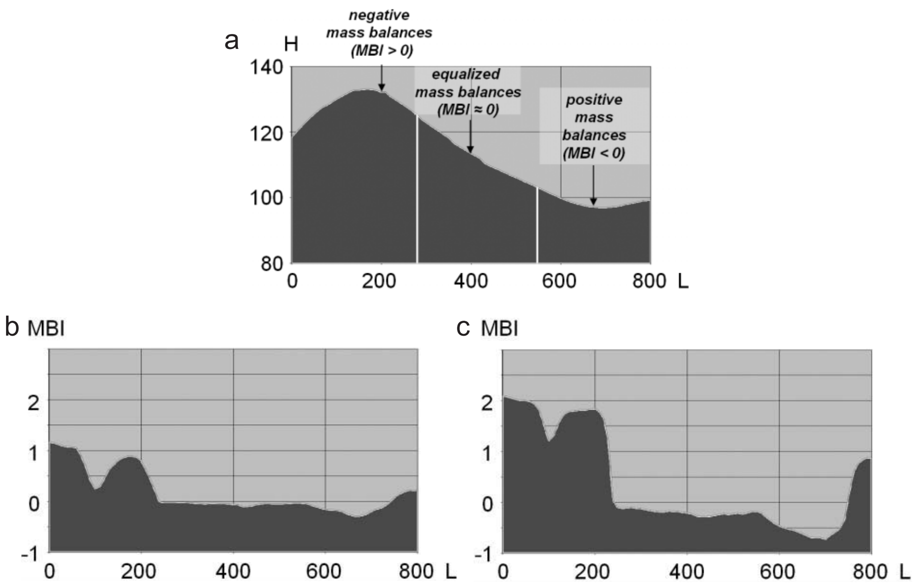


Figure 2: Relation between slope cross section and mass-balance index (MBI) (H, height; L, length of cross section); a) cross section and positions of negative, positive, and equalized mass balances, b) MBI with $T_{ht,n}$ = 15 and T_k = 0.067, c) MBI with $T_{ht,n}$ = 15 and T_k = 0.67.

while lower MBI values are associated with valley areas and concave zones at lower slopes. Balanced MBI values close to zero can be found in midslope zones and mean a location of no net loss or net accumulation of material. Figure 2 also demonstrates how MBI values provide information about relatively balanced states of potential material transport but do not quantify the volume of material in flux.

$$\mathrm{MBI} = \begin{cases} f(k) \times [1 - f(n)] \times [1 - f(ht)] \ \ for \ f(k) < 0 \\ f(k) \times [1 + f(n)] \times [1 + f(ht)] \ \ for \ f(k) > 0 \end{cases} \ with \ \mathrm{MBI} \in [-1, 3]. \quad (1)$$

The attributes were transformed according to Eq. 2 (*Friedrich*, 1996, 1998):

$$f(x) = \frac{x}{(|x| + T_x)}$$

$$with\ x = k,\ n,\ ht,\ h,\ f(k) \in [-1,\ 1],\ f(n,\ ht,\ h) \in [0,\ 1]. \quad (2)$$

This reciprocal operation is extended by the transfer constant T_x which allows different value ranges to be stretched or smoothed: the smaller T_x, the more the value range in the histogram is stretched. This has a large effect on the curvature attribute k which is considered most significant for changing both soil conditions (*Friedrich*, 1996; *Ad-hoc-AG Boden*, 2005) and MBI value range. The comparison of the two MBI versions makes the outcome of the T_k values for MBI characteristic clear: the lower T_k, the greater the relative difference within the value range (Fig. 2b and c).

2.3 Terrain structuring

Landscapes are hierarchically structured. In concepts of hierarchical landscape structuring (*cf., Steinhardt* and *Volk*, 2001, 2003), the delineation of the largest spatial units (hereafter referred to as terrain superobjects) arises from the significant alteration of landscape-related attributes on the one hand and the arrangement of subordinate units or subobjects within hierarchical superobjects on the other hand (Fig. 3).

An automatic implementation of the hierarchical–landscape structuring concept can be achieved by using a region-growing segmentation algorithm applied to continuous digital spatial data like remote-sensing data or DEMs (*Woodcock* and *Harward*, 1992; *Burnett* and *Blaschke*, 2003; *Hay* et al., 2003; *Drâgut* and *Blaschke*, 2006). Here, the fractal–net evolution approach (FNEA) was executed which is described in detail by *Baatz* and *Schäpe* (2000) and *Benz* et al. (2004). Using a hierarchical and bottom-up region-growing algorithm, the FNE algorithm merges single pixel elements (terrain attributes) to terrain objects on different spatial scales building up a hierarchical network of terrain-object levels (Fig. 3). This means that all objects are surrounded by neighboring objects and each object is related to larger and smaller scales *via* parent–children relationships (*cf.*, section 1). As a consequence, each object carries a data set of information including attributes of statistics, neighboring and hierarchical relationships (*e.g.*, attributes *hd* and *ra* in Tab. 1). These data make it possible to implement a multiscale classification algorithm based on hierarchical features.

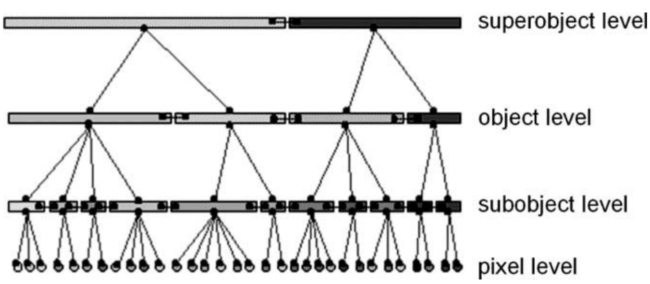


Figure 3: Four-level hierarchical network of terrain objects (*Benz* et al., 2004).

The FNE segmentation algorithm can be considered as an optimization process which minimizes the heterogeneity H of each spatial object for a given resolution over the entire continuous data set with constraints based on local and global conditions. The user-defined heterogeneity H refers to both heterogeneity of pixel values h_{color} and shape heterogeneity h_{shape} according to Eq. 3:

$$H = w_{color}\Delta h_{color} + w_{shape}\Delta h_{shape}$$
$$\text{with } w_{color,shape} \in [0,\ 1],\ w_{color} + w_{shape} = 1. \quad (3)$$

While h_{color} results from the difference between object parameters like object variance, h_{shape} arises from the balance of the object shape features smoothness h_{smooth} and compactness h_{compt} (Eq. 4):

$$\Delta h_{shape} = w_{compt}\Delta h_{compt} + w_{smooth}\Delta h_{smooth}. \quad (4)$$

The parameters w_{color}, w_{shape}, w_{smooth}, and w_{compt} allow finally the weighting of the heterogeneity factors in order to achieve an application-related adaptation of the segmentation results.

2.4 Landform classification

For the purposes of this study, landforms are defined using the following semantics:

- Floodplains are low and flat relative to their surroundings and occur on different scales (*Gallant* and *Dowling*, 2003).
- Depressions and floodplains represent fluvial landforms (*Friedrich*, 1996). They are different in size (floodplains are larger than depressions). Depressions are also low relative to their surroundings but they need not to be flat.
- Slopes, plains, and depressions represent specific scales (*Fisher* et al., 2004; *Jenness*, 2005).
- Slopes, plains, depressions, and floodplains can be differentiated according to their mass balances. Depressions and floodplains show positive mass balances (areas of net deposition), slopes are characterized by negative mass balances (areas of net erosion), and plains are equilibrated. Potential sediment accumulation is therefore more likely to take place in flat than in steeper depression areas. Accumulation reaches a maximum at intense concave cur-

vatures and in a small distance from areas where erosion is occurring. The potential for soil erosion increases with more convex curvature with increasing distance from the channel network (*Friedrich*, 1996; *Möller*, 2005).

This means for our approach that floodplains have to be classified in a multihierarchical manner (*cf.*, section 2.4.1) whereas for the classification of the remaining landforms, specific scales need to be defined (*cf.*, section 2.4.2).

2.4.1 Floodplains

The detection of floodplains is based on a multihierarchical query procedure (Fig. 4a). For each considered scale level (n) resulting from multihierarchical segmentation, a query according to Eq. 5 is performed. The levels that do not fulfil the conditions of the query are transferred to the segmentation level ($n-1$). The procedure is reiterated until no segmentation level is available anymore or the user sets the termination manually, since depression areas appeared from this level onwards.

$$\text{Floodplain} = (hd < 0) \cup \min(\text{MBI}) \cup ra$$
$$\textit{with } ra \in [0,\ x],\ y_n \neq y_{n+1}, y = hd, \text{MBI},\ ra. \tag{5}$$

The term $hd < 0$ means that floodplains on each hierarchy level are located lower than their surroundings. A terrain object with min(MBI) has the smallest positive mass balance within the corresponding superobject. The variable ra means that the objects with a defined mean change in relief are recorded, whereby x represents a maximum of the relief amplitude that has to be determined by the user. In accordance to *Bernhardt* et al. (1991), a value of $ra = 2$ has been used here. The criterion $y_n \neq y_{n+1}$ is applied to avoid a scenario in which objects are classified that have not experienced a spatial differentiation with the transition to the segmentation levels n to n-1))

2.4.2 Slopes, plains, and depressions

The classification procedure combines a statistic structuring method (k means-cluster analysis) with a probability-based approach (maximum-likelihood algorithm) (*cf., McGarigal* et al., 2002). In order to take into account the landscape heterogeneity of the study area, the classification follows a hierarchical approach by which all subordinated (sub-)objects are classified separately according to the spatial extent of the superior (super-)objects (Fig. 4b).

Samples were selected by the following criteria which correspond to context-based landform definitions:

- Minimal MBI values represent depressions, and maximal MBI values indicate slopes.
- Samples for the plain class occur in a cluster where the values lie in the positive and negative value range close to a value of zero (neutral mass balance).

The following two variables influence the classification results, and these parameters can be adjusted so that the outputs are consistent with reference information:

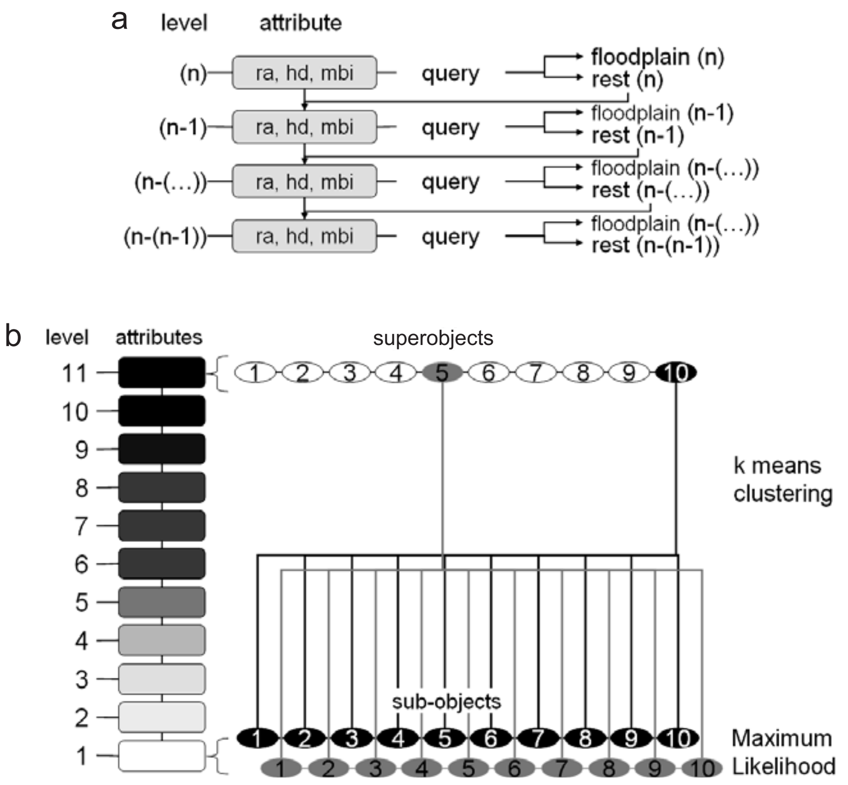


Figure 4: Landform-classification scheme; a) floodplain detection, b) classification of depressions, slopes, and plains.

1) The transfer constant T_k affects the value range of the attribute curvature *k* which is the crucial attribute for MBI calculation (*cf.*, section 2.2).

2) The hierarchy variable determines from which hierarchical level the superobjects are used for the classification procedure. Their modification alters the number of the resulting samples.

2.4.3 Validation

Based on 191 random samples in the study area, elevation cross profiles were set for each point using the Erdas Imagine 8.4 spatial profile tool. From these profiles, we carried out an on-screen determination of the particular landforms considering the local landscape conditions (digital manual mapping, *cf., Möller*, 2005). The sample definition represents the expert knowledge of the user but may also reflect certain class definitions used in a scientific discipline or institution (*e.g.*, soil survey). Figure 5a exemplifies the methodology for a random sample which is situated in a depression landform. The reference information was used to determine the accuracy with which the classification results matched with semantically identified random samples. As adaptation measures the overall accuracy (OA), user's accuracy (UA) and producer's accuracy (PA) were calculated for each landform class deriving from confusion matrix (Fig. 5b; *Stehmann*, 1997; *Foody*, 2002; *Zhan* et al., 2005). The highlighted elements are the main diagonal and contain the cases where the labels depicted in the classification and reference data set agree. The off-diagonal elements represent the cases of label disagreement. *Thomlinson* et al. (1999) stated as a target of a minimum overall accuracy of 85% with no class <70% accuracy (*cf., Foody*, 2002).

Overall accuracy belongs to the most popular measures and is the percentage of all cases correctly allocated to classification. With UA and PA, two class-specific views on confusion matrix can be distinguished depending on whether the calculations are based upon the matrix's row or column marginals (*Foody*, 2002). Producer's accuracy indicates thereby the real hit rate of the classification regarding the reference information (sum of columns). User's accuracy results on the other hand from the "used" classification product. The information content of the classification product is assigned to the reference points (sum of rows).

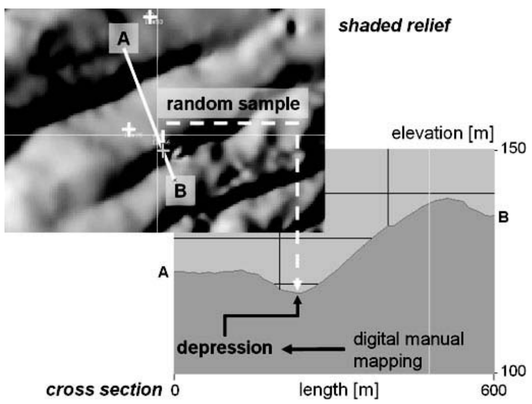
3 Results

3.1 Terrain objects

The transformed attributes *f*(*h*), *f*(*n*), *f*(*k*), and *f*(*ht*) determine the segmentation and the object generation. Their selection was based on two factors:

(1) the published relationships between terrain attributes and their influence on the soil formation and transport processes (*McBratney* et al., 2003; *Ad-hoc-AG Boden*, 2005) and

a



b

Actual Class

Predicted Class	A	B	C	D	Σ
A	n_{AA}	n_{AB}	n_{AC}	n_{AD}	n_{A+}
B	n_{BA}	n_{BB}	n_{BC}	n_{BD}	n_{B+}
C	n_{CA}	n_{CB}	n_{CC}	n_{CD}	n_{C+}
D	n_{DA}	n_{DB}	n_{DC}	n_{DD}	n_{D+}
Σ	n_{+A}	n_{+B}	n_{+C}	n_{+D}	n

$$\text{Percentage correct} = \frac{\sum_{k=1}^{q} n_{kk}}{n} \times 100$$

$$\text{User's accuracy} = \frac{n_{ii}}{n_{i+}}$$

$$\text{Producer's accuracy} = \frac{n_{ii}}{n_{+i}}$$

Figure 5: Validation scheme; a) methodology for sampling of reference data, b) the confusion matrix and some common measures of classification accuracy that may be derived from it (*Foody*, 2002).

(2) the generation of color composites and the related visualization of landforms (Fig. 6a).

The Figs. 6 b–l show the different segmentation levels that represent objects in a multiscale and multidimensional context. The average object sizes OS identify the particular scale area of the segmentation level. The term "multiscale" means that all terrain objects (*e.g.*, of level 7) are both constituted by terrain subobjects (*e.g.*, of level 1) and elements of superobjects (*e.g.*, of level 11; Fig. 6b, h, and l). Multidimensional objects correspond to the classic idea of landform elements or landform facets (*Friedrich*, 1998; *Blaschke* and *Strobl*, 2003; *Drâgut* and *Blaschke*, 2006, *cf.*, section 2.3). However, this only considers objects of the level 1 or 2 because of their low heterogeneity (Fig. 6b and c).

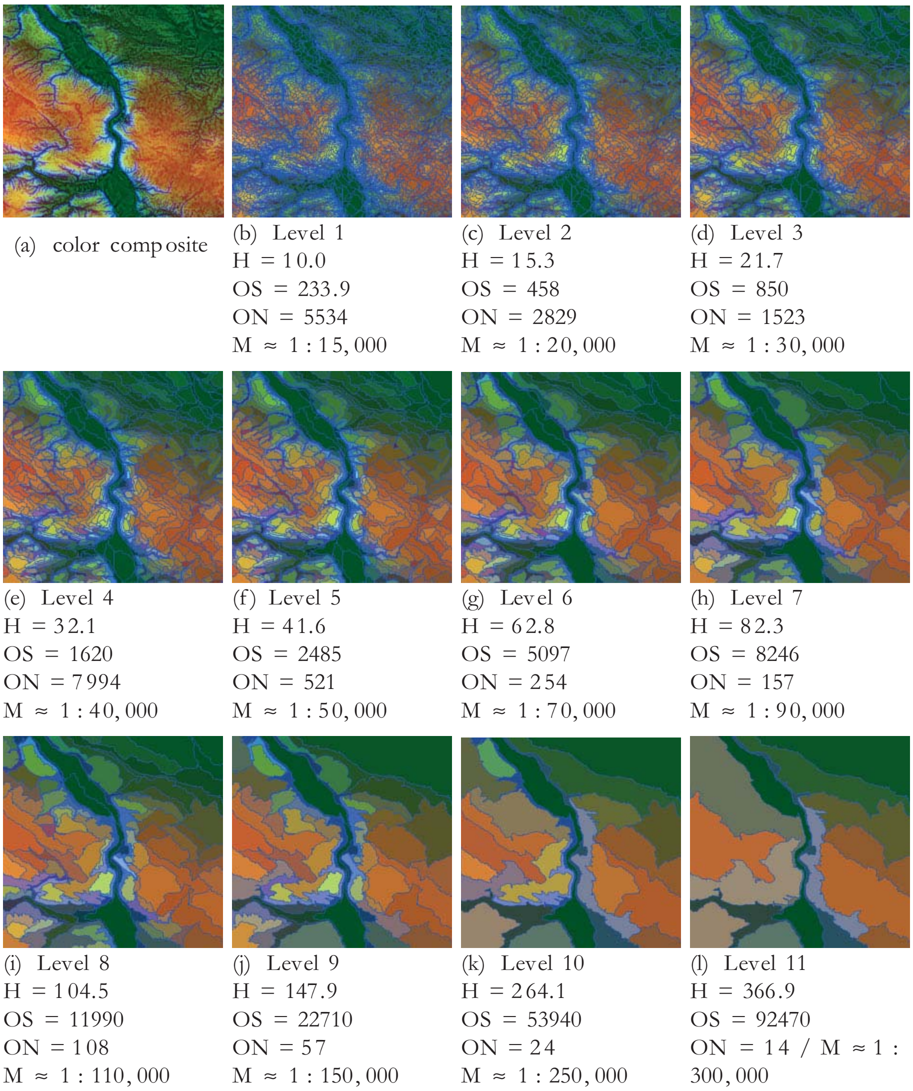
Multidimensionality is exhibited by the fact that on certain aggregation levels, terrain objects emerge or recede. For instance, in the segmentation level 11 (Fig. 6l), the Saale River floodplain appears as a single object, whereas depression areas—which dominate in the segmentation levels shown in the Figs. 6 b to f—are merged in into terrain superobjects.

3.2 Floodplains

The query results and the statistical values of the used attributes are presented in Tab. 2. Six floodplain objects were detected on six different hierarchical levels which are shown in column "Level" (*cf.*, Fig. 6). On level 6 (Fig. 6g), the query was terminated.

3.3 Depressions, slopes, and plains

The segmentation level 1 corresponds to the average object scale of 1:15,000 (Fig. 6b). The modification of (1) the attri-



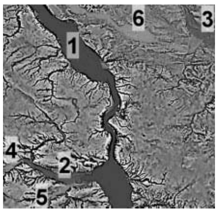
parameter settings: w_{color} = 0.8 / w_{shape} = 0.2 / w_{smooth} = 0.9 / w_{compt} = 0.1

color composite: f (h),f (ht) = red / f (k) = green / f (n) = blue

H = scale parameter / ON = objects number/ OS = object size [m²] / M = mean scale

Figure 6: Scale levels based on the segmentation of transformed terrain attributes *f*(*h*), *f*(*ht*), *f*(*n*), and *f*(*k*).

Table 2: Multihierarchical query results of floodplain detection (*cf.*, Fig. 4).

Floodplains	No	Level	MBI[a]	*hd*[a]	*ra*[a]
	1	11	–0.03	–30.16	0.88
	2	9	–0.15	–8.22	1.93
	3	8	–0.04	–5.25	0.53
	4	7	–0.25	–0.85	1.85
	5	6	–0.06	–0.85	1.42
	6	6	–0.15	–3.21	0.77

[a] *cf.*, Tab. 1

bute variable T_k (*cf.*, Eq. 1) and (2) the hierarchy variable (object number ON) causes different proportions of the resulting classes slope, depression, and plain. The starting point of the classification procedure is the classification variant with the parameter adjustments T_k = 0.0067, $T_{n,ht}$ = 15, and ON = 14 (Fig. 6l and 7e).

3.3.1 Modification of the attribute variable

Figures 7 a–d and 8a clarify the effects of T_k modifications. According to Fig. 2, low T_k values emphasize both terrain forms in flat and sloped areas (Fig. 7a) whereas high T_k values only highlight terrain forms in sloped areas (Fig. 7d). One consequence is that the same landform is described by different MBI values depending on the used T_k values. Thus, an increase of the T_k values is associated with a distinct increase of flat areas and a decrease of the slope areas at the same time. In contrast, the area proportions of the depression class remain stable.

3.3.2 Modification of the hierarchy variable

All classifications of level 1 subobjects (Fig. 6b) refer to a different level of superobjects (*cf.*, Fig. 4b). Apart from level 9 (ON = 57), level 10 (ON = 45), and level 11 (ON = 14; Fig. 6j, k, and l), an additional level was created (ON = 30). The lar-

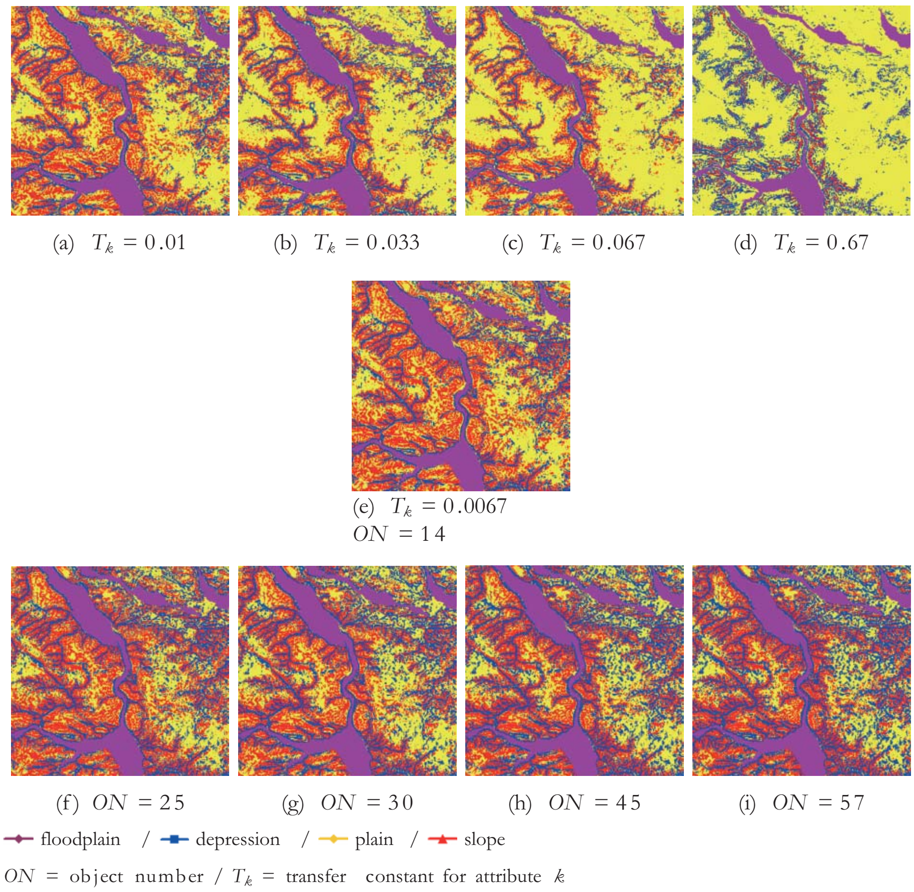


Figure 7: Landform-classification results.

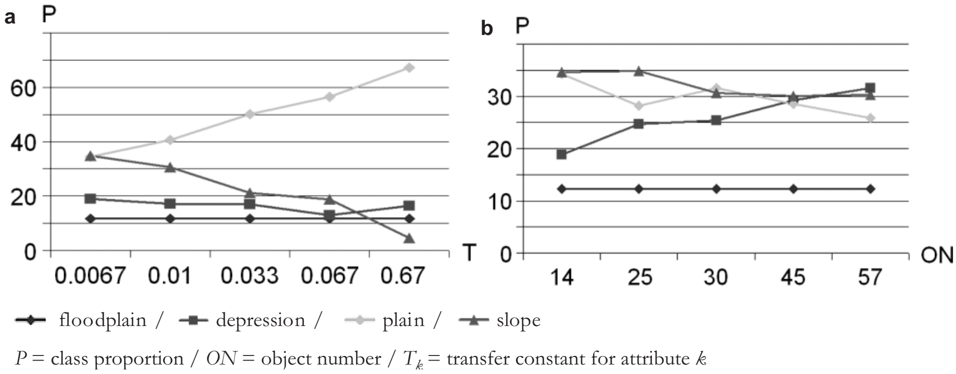


Figure 8: Relations between variable variants of terrain classification and proportions of landforms. (a) Attribute variable with ON = 14, (b) hierarchie variable with T_k = 0.0067.

ger the number of superobjects, the more samples can enter into the classification procedure. Following the sampling strategy (*cf.*, section 2.4.2), the number of samples results from four times the number of superobjects.

As shown in Figs. 7 f–i and 8b, the effect of ON modifications is less sensitive than using the attribute variable. However, as opposed to the attribute variable, an increase of the sample size leads to an increase of the proportion of depression class whereas the proportion of slope areas declines. The resulting proportion of the flat areas varies.

3.4 Validation

Figure 9 summarizes the overall, producer's, and user's accuracies (OA, PA and UA). Accordingly, the attribute variable is the decisive factor for affecting the classification accuracy. This is also shown in Fig. 8. The highest accuracy or the best adaptation between classification results and reference base is achieved with the transfer constant T_k = 0.033 and a number of superobjects ON = 14 (Fig. 9a and b). In terms of classification accuracy, an overall accuracy of 89% was achieved. The classification accuracy of all single classes exceeds the minimum accuracy of 70% (*cf.*, section 2.4.3).

4 Discussion and conclusions

We present a new innovative procedure for the mapping landforms on a soil-genesis and transport basis. The procedure considers multiple spatial scales and can be applied in heterogeneous landscapes. The classified landforms do not inevitably represent soil units, since other factors influence the soil distribution, too. However, the results indicate that this approach will improve existing digital soil-mapping (DSM) methodologies.

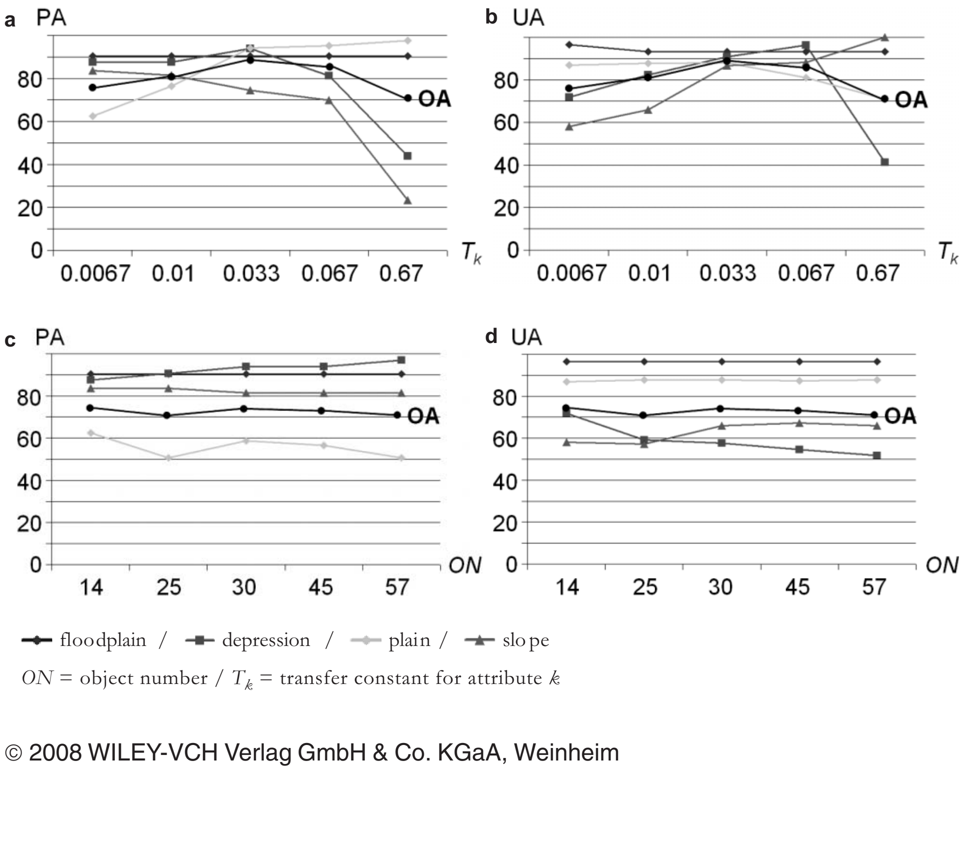


Figure 9: Relations between overall accuracy (OA), user's accuracy (UA), and producer's accuracy (PA) as well as variable variants of landform classification. (a) PA and OA of attribute variable variants with *ON* = 14, (b) UA and OA of attribute variable variants with *ON* = 14, (c) PA and OA of hierarchy variable variants with T_k = 0.0067, (d) UA and OA of hierarchy variable variants with T_k = 0.0067.

4.1 Landscape heterogeneity, scale, and coverage

Our terrain-classification procedure treats object generation and classification separately:

(1) Object generation is based on soil-relevant terrain attributes that are transferred by a region-based segmentation procedure to multihierarchical object structures. As shown by our results, multihierarchical object structures can avoid the limitation of present approaches, like hierarchical moving-window classification procedures (*Gallant* and *Dowling*, 2003; *Fisher* et al., 2004; *Schmidt* and *Hewitt*, 2004; *Jenness*, 2005), which lead to the loss of the resulting coverage by the identification of fluvial landforms (depression, floodplains) on different scales.

(2) The classification procedure distinguished between floodplain detection on the one hand and classification of depressions, slopes, and plains on the other hand:

- Floodplain detection was realized by a multihierarchical query procedure.
- The classification of depressions, slopes, and plains was carried out on a specific target scale (here: approx. 1:15,000) and combined a statistical structuring method (cluster analysis) with a probability-based operator (Maximum Likelihood) and is based on the terrain attribute MBI. The hierarchical relations to superior objects enabled a landscape-specific selection of training areas. This enables landscape heterogeneity to be considered (*cf., MacMillan* et al., 2004; *Schmidt* and *Hewitt*, 2004).

4.2 Landform definition

A key advantage of our classification procedure is the option to modify the area assigned to classes using two different variables, (1) the hierarchical variable and (2) the attribute variable. These two variables allow the adaptation of classification results to reference information and specific class definitions:

(1) The hierarchical variable enables the alteration of sample size and their spatial distribution depending on superobjects which are used for the classification procedure.

(2) The attribute variable affects the value range of MBI by changing of a transfer constant T_x. The MBI has proved to be easily interpretable regarding landform definitions. In this study, all landforms were described by relative values (*e.g.*, maximum MBI value = slope). Thus, the landform definitions are transferable (*cf., Drâgut* and *Blaschke*, 2006).

4.3 Validation

In the majority of soil-related landform-classification approaches, classification quality was deduced from statistical relations between soil and terrain properties (*Pennock* et al., 1987; *Zhu* et al., 1997; *Park* et al., 2001; *Park* and *Vlek*, 2002; *Pennock*, 2003; *Park* and *van de Giesen*, 2004; *Ryan* et al., 2000; *Schmidt* and *Hewitt*, 2004; *MacMillan* et al., 2004). In this study, each classification is labeled by a specific accuracy metric (here: overall, user's, and producer's accuracy). Thus, our approach enables additional applications such as the revision of existing soil maps (*Friedrich*, 1998; *Möller*, 2005). Finally, while reference information is usually mapped during soil survey, our approach realizes the mapping of landforms by an efficient on-screen mapping according to *Möller* (2005).

4.4 Further research

An unsolved problem is the determination of classification-relevant hierarchy levels, for instance, for the delineation of floodplains. One possibility is the identification of landscape-scale thresholds (*Hay* et al., 2001; *Hall* et al., 2004). In this context, the observed relations between hierarchy variable variants and alternating area proportions of landforms require additional research. Furthermore, it remains unclear which parameter adjustments of the segmentation algorithm best represents the underlying terrain units. One option is an object validation based on object-related reference information (*Möller* et al., 2006). This also includes the validation process of landforms itself. Finally, the number of clusters is chosen subjectively. Thus, algorithms have to be included enabling an optimum number of clusters (*e.g., de Bruin* and *Stein*, 1998).

Further possible applications exist in connection with the integration of the MBI attribute in qualitative soil-erosion assessments. Work being undertaken by us aims at the modification of length-slope factor in the universal soil-loss equation (USLE, *e.g., Moore* and *Burch*, 1986; *Hickey*, 2000). This could help to overcome limits of existing USLE-based erosion assessment methods namely the classification of accumulation areas (*Merritt* et al., 2003). Finally, the use of natural system units, such as watershed hierarchies, will be used as a basis for the classification to enable a linkage to hydrological models with the objective to improve their spatial process description (*Volk* et al., 2007).

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AUSWIRKUNGEN VON LANDNUTZUNGSÄNDERUNGEN AUF DEN GEBIETSWASSERHAUSHALT:

Anwendungsmöglichkeiten des Modells "ABIMO" für regionale Szenarien

MARTIN VOLK, LEIPZIG & MICHAEL BANNHOLZER, STUTTGART

SUMMARY

Impacts of land use changes on the landscape water balance - application possibilities of the runoff simulation model ABIMO.

The application possibilities of a runoff-simulation model were examined on the landscape scale by scenarios on land use changes. At first, the different and heterogenous input data were modified (aggregation of data, etc.) to enable the calculation processes of the simulation model. For the definition of the scale-dependent application possiblities of the model, the calculations were carried out at different scales. In cooperation with environmental and governmental authorities, test areas with specific land-use conflicts (between forestry, agriculture, nature conservation and water resources management) were selected. Based on the model output, those land-use variants can be highlighted which show positive effects on the quality and quantity of the water resources. The study shows a useful application of a runoff model servingas an instrument for landscape planning and an integral part of landscape-ecological assessments.

ZUSAMMENFASSUNG

Anhand von Berechnungen der Grundwasserneubildung im Regierungsbezirk Dessau (Sachsen-Anhalt) wurden die Anwendungsmöglichkeiten des Rechenprogrammes "ABIMO" (**Ab**fluß**bi**ldungs**mo**dell) für großräumige Wasserhaushaltsmodellierungen und Landnutzungsszenarien überprüft. Vor der Modellierung mußten die heterogenen Eingangsdaten teilweise verändert und vereinfacht bzw. angepaßt werden, um den Bedürfnissen des Rechenprogramms gerecht werden zu können. Mit der Überprüfung sowohl im regionalen als auch in größeren Maßstäben konnten die dimensionsbedingten Grenzen der Modellanwendung abgesteckt werden. Unter Einbeziehung planungsrelevanter Ämter und Behörden sowie Betroffener aus Land-, Forst- und Wasserwirtschaft wurden für ausgewählte Konfliktregionen (Nördlicher Mittelfläming, Köthener Ackerland) Landnutzungsszenarien erarbeitet und deren Auswirkungen auf die Grundwasserneubildung unter diesen verschiedenen Verhältnissen simuliert. Auf Basis der Berechnungsergebnisse wurden anschließend Empfehlungen für Landnutzungsvarianten unterbreitet, die sich auf den Gewässerhaushalt und die Grundwasserneubildung und damit auch auf Umwelt- und Naturschutz positiv auswirken. Die Studie soll das Verfahren transparent und verständlich gestalten und kann zukünftigen Anwendern von Wasserhaushalts-Modellen als exemplarisches Beispiel dienen.

1.0 EINLEITUNG

Landnutzung und Landnutzungsänderungen beeinflussen durch veränderte Wasser- und Stofffluß-bedingungen innerhalb und zwischen größeren Räumen den Landschaftshaushalt und können durch die ökologischen Folgewirkungen die Regulationsfunktionen der Landschaft stark beeinträchtigen. Im Sinne des Ressourcenschutzes und zur Verringerung von Landnutzungs- und Umweltkonflikten auf regionaler Ebene ist die Entwicklung von Konzepten für eine umweltverträgliche Mehrfachnutzung der Landschaft erforderlich, die diese landschaftshaushaltlichen Faktoren berücksichtigen. Dem Wasser als erneuerbare Ressource, als Klimaelement und Träger wichtiger Stofftransportprozesse kommt hier eine besondere Rolle zu (vgl. KLEEBERG 1992). Die Kenntnisse über regionale Wasser- und Stoffflüsse sind allerdings noch immer unzureichend, landschaftshaushaltliche Faktoren werden somit nur wenig in landschaftsökologischen Bewertungen und planerischen Vorgaben beachtet. Aufgrund der großen Flächen der Untersuchungsräume scheiden die "klassischen" Meßverfahren aus finanziellen, technischen und zeitlichen Gründen und der Notwendigkeit einer flächendifferenzierten Aussage aus. Zahlreiche Wasser- und Stoffhaushaltsmodelle wurden daher in den letzten Jahren entwickelt, deren Anwendungsmöglichkeiten aber zumeist noch abgegrenzt und optimiert werden müssen (vgl. KRYSANOVA et al. 1996, STEINHARDT & VOLK 1998). Auf diese Weise könnten solche Modelle dann auch verstärkt Eingang in die planerische Praxis finden und bei der Analyse von Nutzungskonflikten und bei der Ausweisung von Vorrang-, Vorsorge- und Schutzgebieten (insbesondere Grundwasserschutz) als zusätzliches Instrumentarium verwendet werden.

Neben der Bestimmung der Grundwasserneubildung für den gesamten Regierungsbezirk Dessau sollen am Beispiel von Testgebieten die Anwendungsmöglichkeiten eines Abflußbildungsmodells (ABIMO) für Szenarioberechnungen der Auswirkungen von Landnutzungsänderungen auf den Gebietswasserhaushalt dargelegt werden. Der Untersuchungsraum verfügt neben den bekannten Bergbau- und Industrielandschaften im Raum Bitterfeld auch über die Wälder und Forsten des Flämings und der Dübener Heide, teilweise naturnahe Auen und das Biosphärenreservat "Mittlere Elbe" sowie wertvolle Kulturlandschaften wie den "Wörlitzer Park". In Kombination mit den vorhandenen Agrarlandschaften, an denen alle Aspekte des Agrarlandschaftswandels zu beobachten sind, weist die Region eine bedeutende Landnutzungsdynamik sowie ein hohes Potential an Landnutzungskonflikten auf, so daß die Untersuchungen hier beispielhaft an ausgewählten Testgebieten ausgeführt werden können. Die vorgestellten Arbeiten wurden im Rahmen einer Diplomarbeit durchgeführt (vgl. BANNHOLZER 1998) und sind Teil eines Projektes, bei dem basierend auf Untersuchungen zum Landschaftshaushalt Strategien zur zukünftigen Landschaftsentwicklung im Regierungsbezirk Dessau erarbeitet werden.

2.0 ARBEITSGEBIET

Der Untersuchungsraum "Regierungsbezirk Dessau" befindet sich im Osten des Bundeslandes Sachsen-Anhalt und umfaßt eine Fläche von ca. 4280 km². Die pleistozänen Bedingungen und deren morphogenetische Auswirkungen waren prägend für die Gestaltung der derzeitigen Oberflächenformen und des Fließgewässernetzes im Regierungsbezirk Dessau. Der heutige Aquifer befindet sich in den mächtigen periglaziären, glazialen und glazifluvialen Sedimenten mit sehr heterogenen Eigenschaften. Auf diesem Ausgangssubstrat haben sich bereichsweise sehr fruchtbare Böden gebildet, die wichtigsten seien hier mit den Braunerden (14,9% der Fläche des Regierungsbezirkes), den Schwarzerden (13,6% Flächenanteil), den Braunerde-Podsole und Rosterden (13,3% Flächenanteil) und den Gleyen (12,2% Flächenanteil) genannt. Mit mittleren Niederschlagswerten zwischen 450 mm/a (Raum Köthen) und 650 mm/a (Dübener Heide und Fläming) gehört der Regierungsbezirk Dessau zu den niederschlagsärmsten Regionen Deutschlands, was sich neben den Bodenverhältnissen ebenfalls auf die Grundwasserverhältnisse auswirkt.

In bezug auf den Landschaftswasserhaushalt einer Region stellt die Art der Landnutzung einen wichtigen Faktor dar. Sie beeinflußt die Gesamtverdunstung durch den Boden und die Pflanzen und hat damit auch einen entscheidenden Einfluß auf die Grundwasserneubildung. Etwa 58% der Gesamtfläche des Regierungsbezirkes Dessau werden landwirtschaftlich genutzt. Intensive Nutzung findet hauptsächlich im Schwarzerdegebiet im Westen des Raumes um Köthen und im nordwestlichen Bereich um Zerbst statt. Diese Gebiete können als ausgeräumte Intensivagrarlandschaften mit bis zu 85% landwirtschaftlicher Nutzung eingestuft werden. Forstwirtschaftlich werden ca. 18% der Gesamtfläche genutzt. Dabei handelt es sich hauptsächlich um Kiefernforste.

2.1 Landschaftstruktur, Landnutzung und Landnutzungskonflikte

Wie bereits dargelegt, ist die Region ihrer Landschaftsstruktur und ihren Landschaftstypen nach sehr vielfältig. Neben Stadt- und Industrielandschaften und Bergbaufolgelandschaften existieren die Auenlandschaften der Mulde und der mittleren Elbe, Intensivagrarlandschaften und Agrar-Forstlandschaften mit ihren jeweils sehr unterschiedlichen ökologischen und sozioökonomischen Funktionen. Bisher unzureichend bekannt sind die Wechselwirkungen zwischen den jeweiligen Landschaftsfunktionen und die sich aus Nutzungsüberlagerungen ergebenden Konflikte. Bei der Regional- und Landschaftsplanung dominiert die Festschreibung von Funktionen durch Ausweis von Vorrang- und Vorsorgegebiete für Einzelfunktionen. Die Regulationsfunktionen der Landschaft, ökologische Folgewirkungen der Landnutzung und von Landnutzungsänderungen werden unzureichend berücksichtigt. In bezug auf den Gebietswasserhaushalt und insbesondere auf die Grundwasserneubildung wurde daher das *Regionale Entwicklungsprogramm* für den Regierungsbezirk Dessau (vgl. MRLU 1996) sowie ausgewählte *Agrarstrukturelle Vorplanungen*[1] untersucht, um folgende Fragen klären zu können:

- *Wie ändert sich die Grundwasserneubildung im Regierungsbezirk Dessau aufgrund von Landnutzungsänderungen?*
- *Welche Konflikte können aufgrund bestehender Planungen entstehen?*

Im Ergebnis dieser Untersuchung wurden die beiden Landschaftseinheiten[2] “Köthener Ackerland“ und “Nördlicher Mittelfläming“ für Szenarioberechnungen ausgewählt, da diese Gebiete ein erhöhtes Konfliktpotential aufweisen. Anschließend wurden zahlreiche Gespräche mit den Verantwortlichen in verschiedenen Dezernaten im Regierungspräsidium Dessau, in Forstämtern und Umweltbehörden geführt, um zusätzliche Informationen zu den behandelten Problembereichen zu bekommen.

Köthener Ackerland

Auszüge aus der AVP für "Köthen Süd/West"(Landschaftseinheit Köthener Ackerland): Das Untersuchungsgebiet stellt mit seinem Lößboden durchgängig einen für die landwirtschaftliche Nutzung in hohen Maße geeigneten Raum dar und ist traditionell von intensivem Ackerbau geprägt. Die Ackerzahl beträgt von wenigen Ausnahmen abgesehen 100. Das Oberflächenbild wird fast völlig von weiten ebenen Flächen beherrscht. Die Region ist laut dem Regionalem Entwicklungsprogramm Dessau von 1996 (vollständig) Vorranggebiet für Landwirtschaft und wird zu 85% landwirtschaftlich genutzt. Gleichzeitig bestehen in der Region Vorranggebiete für Wassergewinnung.

[1] Die Agrarstrukturelle Vorplanungen (AVP) werden vom Dezernat für Agrarstrukturen am Regierungspräsidium Dessau erstellt. Die AVP's beziehen sich auf kleinräumigere Flächen innerhalb des Regierungsbezirkes Dessau und sind nahezu flächendeckend vorhanden.

[2] Die Landschaftseinheiten wurden von Krönert (vgl. Krönert 1997) ausgewiesen. Es handelt sich dabei um Bewertungseinheiten eines Forschungsprojektes, die sich von den Landschaftseinheiten des Landschaftsprogrammes Sachsen-Anhalt (vgl. MRLU 1994)

Da 70% des Trinkwassers in der Region aus Grundwasser gewonnen wird, leitet sich für die Zukunft die Erhaltung eines stabilen Grundwasserhaushalts ab. Eine Gefährdung des Grundwassers ist auch in Zukunft aufgrund der bindigen Deckschichten und des hohen GW- Flurabstands von größtenteils mehr als 10 Metern nicht zu erwarten. Schadstoffe dringen nur sehr langsam ein. Der Charakter des Vorranggebiets darf durch andere Flächennutzungen nicht verändert werden.

Der geringe jährliche Niederschlag im Raum (<500mm/a) beeinträchtigt die Ertragsfähigkeit des Bodens erheblich. Eine ausreichende Wasserversorgung durch Beregnung ist in Zukunft Grundvoraussetzung für die volle Ausschöpfung des möglichen Ertragspotentials.

Nördlicher Mittelfläming

Auszüge aus der AVP für "Vorfläming Roßlau-Zerbst" (Landschaftseinheit Nördlicher Mittelfläming): Im Untersuchungsgebiet, ein ländlich geprägter und strukturierter Raum, werden in Zukunft die Hauptnutzungsansprüche Trinkwasserschutz, Natur- und Umweltschutz, Bodenschutz, Fremdenverkehr, Forstwirtschaft und Landwirtschaft mit ihren Wechselwirkungen die Akzente setzen. Planungen und Begründungen aus der AVP: Die Region ist in großen Teilen Vorranggebiet für Forstwirtschaft. Mittel- bis langfristig sollen die dominierenden Kiefernbestände in Mischwaldgesellschaften mit dominierendem Laubholzanteil umgewandelt werden, wobei die natürliche Verjüngung bei Buche und Kiefer durch Lichtungshiebe gefördert werden soll. Die wenigen Reste der Eichenmischwälder, sowie die Erlenbruchwälder sollen erhalten werden. Eine Erhöhung der Umtriebszeit wird angestrebt. Die Region bleibt als Vorranggebiet für Wassergewinnung bestehen. Für die Landwirtschaft ergeben sich Nutzungsbeschränkungen, prinzipiell ist aber in den Schutzzonen II und III der Wasserschutzgebiete eine landwirtschaftliche Nutzung möglich. Es wird eine Hochwassersicherheit des Grabennetzes an landwirtschaftlich genutzten Flächen mit entsprechender Lage des Mittelwassers unter Geländeoberfläche gefordert. Eine Erweiterung der Wasserschutzgebietsflächen ist nicht vorgesehen. Die landwirtschaftliche Nutzung der Flächen ist zu erhalten, um die typische Kulturlandschaft zu erhalten. Das Untersuchungsgebiet ist auch künftig als landwirtschaftlich benachteiligtes Gebiet einzustufen.

Folgende Informationen konnten in zusätzlichen Gesprächen mit den Verantwortlichen aus den Dezernaten im Regierungspräsidium und Forstämtern gewonnen werden: In der Landschaftseinheit "Nördlicher Mittelfläming" liegen im Bereich um die Ortschaft Nedlitz drei Brunnengalerien zur Entnahme von Trinkwasser, wobei hauptsächlich Magdeburg aus diesem Grundwasser versorgt wird. In den letzten 20 Jahren wurde die Anzahl von ehemals flächig verteilten 23 Brunnen auf nunmehr drei Brunnengalerien reduziert - wobei das Volumen der Wasserentnahme gleich blieb. Gleichzeitig verschlechterte sich in diesem Zeitraum der Gesundheitszustand der Wälder. Zu Beginn wurden jährliche Sanitärhiebe in Kiefernbeständen notwendig, um Trockniserscheinungen und allgemeines Baumsterben örtlich einzudämmen bzw. zu begrenzen. Ab 1986 traten dann mit zunehmender Intensität auch in Eichen-Buchenwaldbeständen Absterbeerscheinungen auf, die sich langfristig durch einen immer deutlicher werdenden Vitalitätsverlust bemerkbar machten.

Im April 1994 wurde deshalb ein "Arbeitskreis zur forstwirtschaftlichen Beweissicherung" ins Leben gerufen und in enger Zusammenarbeit mit der Landesanstalt Sachsen-Anhalt, Abt. Forsteinrichtung, insgesamt acht Beobachtungsflächen eingerichtet. In fünfjährigem Turnus soll eine Vollaufnahme aller numerierten Bäume sowie eine Vegetationsaufnahme durchgeführt werden. Zweimal pro Jahr werden die Grundwasserstände im oberen Grundwasserleiter ermittelt.

2.1.1 *Landnutzungskonflikte*

Aus den beschriebenen verschiedenen Planungen und unterschiedlichen Landnutzungsansprüchen ergeben sich Nutzungsüberlagerungen und damit Landnutzungskonflikte. Die wichtigsten, die den

Wasserhaushalt bzw. das Grundwasser betreffen, seien hier genannt. Die beschriebenen Konflikte sind die Grundlage für erstellte Szenarien, die die Beeinflussung der Grundwasserneubildung durch verschiedene Landnutzungsvarianten aufzeigen.

Nutzungskonflikte im Köthener Ackerland:

Im Köthener Ackerland wird seit Generationen Landwirtschaft betrieben. Konflikte ergeben sich mit dem Gewässerschutz und mit dem Natur- und Landschaftsschutz, der extrem eingeschränkte Entfaltungsmöglichkeiten hat. Landschaftsschutzgebiete gibt es mit einem Flächenanteil von ca. 1% ausschließlich auf wenig ertragreichen Standorten. Konflikte ergeben sich auch aus der agrarischen Intensivnutzung und gleichzeitigen Trinkwassergewinnung. Die Vorranggebiete für Wassergewinnung liegen gänzlich unter landwirtschaftlich genutzten Flächen. Übergeordnete Planungen (Landschaftsrahmenplan, Landesentwicklungsplan, Regionales Entwicklungsprogramm) weisen für die Region Gebiete für Erholungsnutzung, Landschafts- und Naturschutz aus. Die Umsetzung stößt jedoch wie auch immer gelagert auf Widerstand aus ökonomischer Sicht.

Nutzungskonflikte im Nördlichen Mittelfläming:

27% der Wälder im Trinkwasserschutzgebiet des Gebietes unterliegen direkt dem Einfluß von Schwankungen des Grundwasserspiegels. Irreversible Schäden sind bei entsprechend starken und dauerhaften Absenkungen des Grundwasserspiegels zu erwarten. Die derzeitigen Flurabstände unterschreiten auf den grundwassernahen Standorten die als optimal anzusehenden Werte. Ein effektiver Rückhalt der gesamten Niederschlagsmengen über mehrere Jahre ist im Trinkwasserschutzgebiet "Westfläming" dringend notwendig. Jeglicher Oberflächenabfluß muß gestoppt werden, um den GW- Speicher aufzufüllen, was für das Waldwachstum und die Grundwasserneubildung notwendig ist. Die Absenkungen des Grundwasserspiegels in den 80er Jahren durch zuvor durchgeführte Meliorationsprojekte in der Landwirtschaft haben maßgeblichen Anteil bei der Destabilisierung des Gesundheitszustandes des Waldes.

Das Wasserwerk Lindau II entnimmt zur Zeit bis zu 50.000 m^3 Wasser pro Tag zur Trinkwasseraufbereitung. Die Reduzierung der Anzahl der Brunnen zur Wasserentnahme von 23 auf drei führt lokal an den drei Stellen zu GW- Senkungstrichtern, es besteht die akute Gefahr des Vertrocknens der Bäume. Flächen in der Schutzzone II und III werden landwirtschaftlich genutzt.

Die Landwirtschaft fordert Erhalt, Pflege und Ausbau des Grabensystems, um einen geordneten Bodenwasserhaushalt und damit erfolgversprechende landwirtschaftliche Nutzung zu gewährleisten. Aus wasserwirtschaftlicher Sicht werden Gewässerschonstreifen, sowie eine Flächenbereitstellung für Windschutzpflanzungen auf erosionsgefährdetem Gelände gefordert. In diesen Gebieten ist für eine möglichst ständige Bodenbegrünung, sowie die Anpflanzung von Flurgehölzen und pfluglose Bodenbearbeitung zu sorgen.

3.0 METHODIK

3.1 Verwendete Systeme

Für die Verarbeitung der Daten wurde eine Sun Solaris Workstation und ein PC in einem Rechnernetz verwendet. Für die Verarbeitung der Daten wurde folgende Software verwendet:

Workstation:	ArcView, Version 3.0 b Arc/Info, Version 7 ERDAS Imagine	**PC:**	ArcView, Version 3.0 a Microsoft Excel, Version 7.0 ABIMO, Programm zur Grundwassermodellierung, Version 2.1

3.2 Datenbasis und Berechnung

Folgende Daten gingen in vorliegender Form oder verändert in die Berechnungen ein:

- Bodenübersichtskarte (BÜK200) vom geologischen Landesamt Sachsen- Anhalt im Maßstab 1 : 200.000 (vgl. SCHRÖDER, KNAUF & KAINZ 1997).
- Mittelmaßstäbige Landwirtschaftliche Standortkartierung der DDR (MMK), Blatt Dessau, im Maßstab 1 : 100.000 von 1979.
- CORINE Land Cover Daten zur Flächennutzung bzw. Bodenbedeckung im Maßstab 1 : 100.000 von 1996.
- Rasterdaten vom Deutschen Wetterdienst (DWD) zum Niederschlag und der potentiellen Verdunstung, je im langjährigen Mittel von 1961–1990 gemessen und auf ein 1- Kilometerraster interpoliert.

Zur Berechnung der Grundwasserneubildung wurde das Programm "ABIMO" benutzt (vgl. GLUGLA & FÜRTIG 1997). Mit dem Programm werden vieljährige Mittel der Abflußbildung und des Wasserhaushalts berechnet. Unter "mittlerer Abflußbildung" wird die Differenz vieljähriger Mittel von Niederschlag und realer Verdunstung verstanden. Diese Differenz entspricht dem mittleren Gesamtabfluß. Bei ausschließlich vertikaler Sickerung stimmt dieser Wert mit der Grundwasserneubildung überein.

In ABIMO gilt folgende Gleichung:

$$R = P0 - ETR$$

R: Gesamtabfluß
P0: Korrigierter Niederschlag
ETR: Reale Verdunstung

Zur Berechnung der für die Abflußbildung benötigten vieljährigen Mittel der realen Verdunstung wird die BAGROV-Beziehung eingesetzt (DWVK 1996). Die BAGROV- Beziehung verknüpft die gegebenen Standorteinflüsse (Niederschlag, potentielle Verdunstung Landnutzung, Boden) und ermittelt die reale Verdunstung.

Die BAGROV- Beziehung lautet:

$$dETR / dP0 = 1 - (ETR / ETP)^{n}$$

ETP: Potentielle Verdunstung
n: Effektivitätsparameter

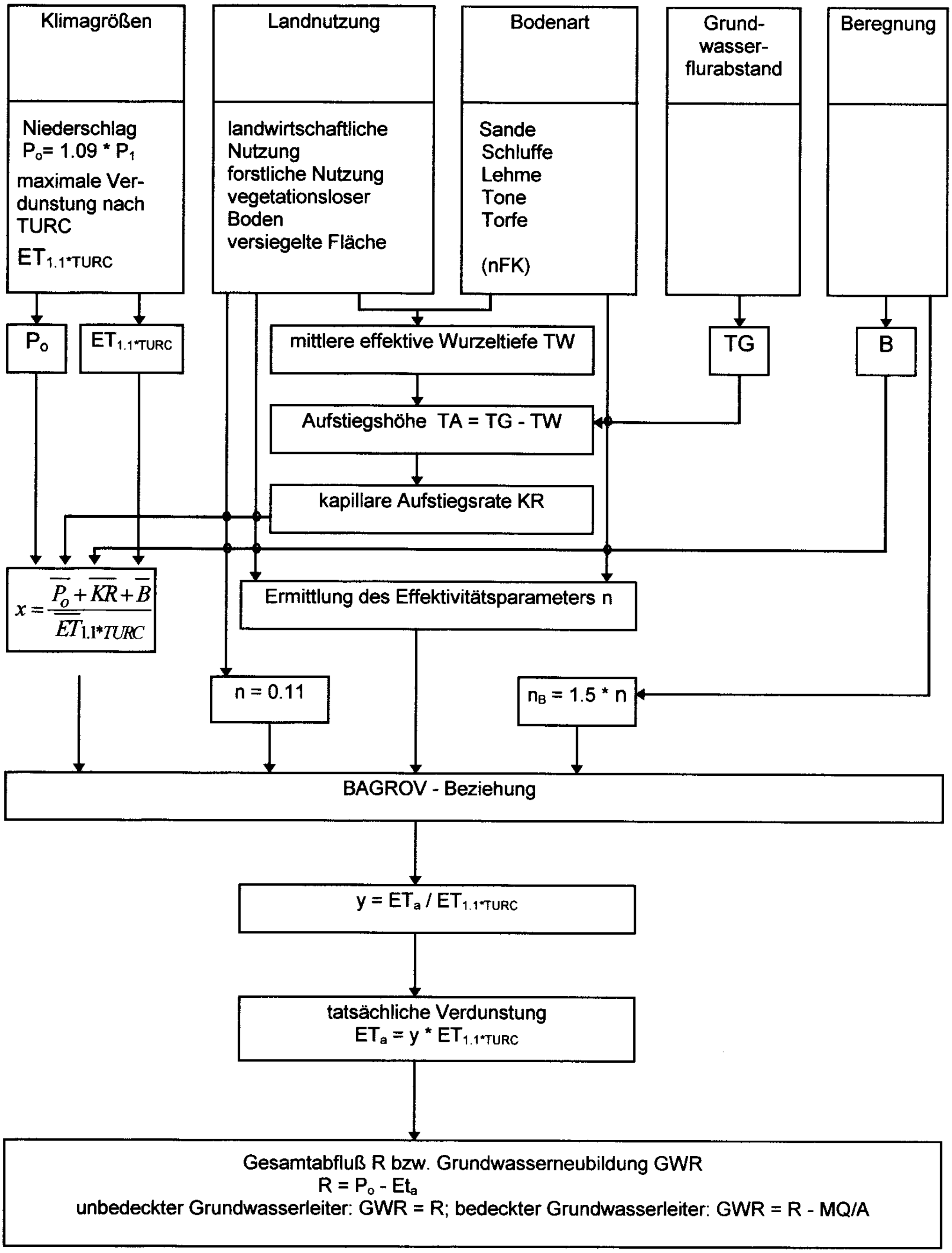
Diese Beziehung ist für nicht oder nur gering geneigtes Gelände geeignet, wo das Wasserdargebot durch den mittleren Jahresniederschlag repräsentiert wird und der Oberflächenabfluß vernachlässigt werden kann. Auf ca. 95% der Fläche des Regierungsbezirkes Dessau (ca. 4.000 $km^{2)}$ ergeben sich maximale Höhenunterschiede von lediglich ca. 70 Metern, so daß diese Bedingung größtenteils erfüllt wird. Abb. 1 zeigt das dem Programm zugrundeliegende Berechnungsschema für die Abflußbildung.

ABIMO benötigt verschiedene Eingangsdaten zur Berechnung, die wichtigsten von ihnen seien kurz genannt: Niederschlag und potentielle Verdunstung, nutzbare Feldkapazität, Flurabstände des Grundwassers und Arten der Landnutzung. Ein Großteil dieser Daten mußte modellgerecht aufbereitet werden, wie im Folgenden dargelegt (vgl. dazu VOLK & STEINHARDT 1998).

Zur Ableitung der "*modellgerechten Böden*" sowie zur Ermittlung der *nutzbaren Feldkapazität (nFK)* wurden die Bodendaten aus der digital vorliegenden Bodenübersichtskarte (BÜK200) des

Geologischen Landesamtes Sachsen- Anhalt (GLA) entnommen (vgl. SCHRÖDER, KNAUF & KAINZ 1997). Zur Aufbereitung der *Flurabstandsklassen* wurde die Mittelmaßstäbige Landwirtschaftliche Standortskartierung (MMK), die analog vorliegt, herangezogen. Jeder Leitbodenform ist ein Grundwasserflurabstand bzw. bestimmte Bodenwasserverhältnisse zugeordnet.

Abb. 1: Berechnungsschema für die Abflußbildung in ABIMO (nach GLUGLA & FÜRTIG 1997).



Diese Bodenformen wurden mit den Bodenformen der BÜK200 verglichen, so daß jeder Bodenform der BÜK200 eine Flurabstandsklasse zugeordnet werden konnte. Schließlich wurden die Originaldaten aus der MMK, die in sechs Wertebereiche eingeteilt sind, an die von ABIMO geforderten drei Klassen angepaßt. Um die für ABIMO notwendigen *Landnutzungsdaten* aufzubereiten, wurde auf den digitalen Datensatz der CORINE-Daten zur Landnutzung und Bodenbedeckung (vgl. STATISTISCHES BUNDESAMT 1996) zurückgegriffen.

Im nächsten Schritt wurden alle aufbereiteten Eingangsdaten in Arc/Info einander räumlich zugeordnet. Das Resultat der Verschneidung war ein Coverage, das die Gesamtheit aller Attributdaten der einbezogenen Daten enthielt. Insgesamt entstanden 24.323 Polygone. Die von ABIMO gesetzte Grenze von bis zu 30.000 zu verwaltenden Elementen wurde nicht überschritten, so daß eine weitere Aufteilung des Untersuchungsgebietes in verschiedene Bilanzgebiete entfiel. Zur Berechnung verschiedener Szenarien mußten die Daten flexibel gehalten und mittels Abfragen aufbereitet werden.

Für den Modelldurchlauf in ABIMO werden folgende Basisdaten benötigt und innerhalb des Programms behandelt: Klimadaten: Niederschlag P0, potentielle Verdunstung ETP; Landwirtschaftliche Nutzflächen L; Forstliche Nutzung W; Gärtnerische Nutzung K; Vegetationslose Flächen D; Gewässerflächen G; Versiegelungsgrad VER [%] bebauter Flächen; Grad der Regenwasserkanalisation KAN [%] versiegelter Flächen; Nutzbare Feldkapazität nFK [Vol%]; Bodenart BOD; Mittlere effektive Wurzeltiefe TW; Mittlere Aufstiegshöhe TA; Mittlere kapillare Aufstiegsrate KR; GW- Flurabstand; Beregnungsmenge BER; Rechts- (RW) und Hochwert (HW).

3.2.1 Überprüfung des Berechnungsverfahrens

Basis für die Szenarioberechnungen war zunächst die Bestimmung der Grundwasserneubildung für den gesamten Regierungsbezirk Dessau (vgl. Abb. 2). Die höchsten Werte sind dabei in den Moränengebieten im Norden (Fläming) und Südosten (Dübener Heide) der Region zu verzeichnen, während die geringsten Werte im Westen (Raum Köthen) und Osten (Elbtal) ermittelt wurden. Um einen Einblick in das Rechenverhalten des Programms ABIMO und dessen Parametergewichtung zu erhalten, wurde eine Sensitivitätsanalyse durchgeführt (KUNZE 1998), deren Ergebnisse nachfolgend dargelegt werden sollen.

Die Daten zur nutzbaren Feldkapazität (nFK) beeinflussen im Rechenprozeß entscheidend die Versickerungswerte. Eine Änderung der nFK-Werte um 1 Vol.% führt zu Änderungen der Versickerungswerte von 6 mm/a auf 16 mm/a. Daher muß die Ableitung der nutzbaren Feldkapazität sehr genau erfolgen. Die Angabe der Bodenart wird programmintern zur Berechnung der kapillaren Aufstiegsrate bei flurnahem Grundwasser benutzt. Eine Variierung der Bodenart führt nur bei der Flurabstandsklasse FLK < 1 m zu einer Änderung der Versickerung.

Um verschiedene Landnutzungsformen zu vergleichen, wird die FLK > 2 m gesetzt, da sonst für Wald bei kleineren FLK mit ETR = ETP gerechnet wird. Ermittelt wurde, daß die Versickerung von devastierten Flächen über Kleingärten und Landwirtschaft zu Wald sinkt.

Die Ertragsklasse ist ein Maß für die Vegetationsdichte und beeinflußt die Versickerungswerte entscheidend. Je höher die Ertragsklasse, desto niedriger der Versickerungswert RU. Eine Variierung der Ertragsklasse ERT um 5 ergibt eine Änderung von RU um 12 mm/a.

Bei landwirtschaftlicher Nutzung wird bei einem Flurabstand < 1 m mit ETR = ETP gerechnet. Bei 600 mm/a Niederschlag errechnet sich dann für die Versickerung: RU = P1 – ETP (z. B.: RU = 600 * 1,09 – 600 = 54).

Als Fazit ist festzustellen, daß besonderes Augenmerk ist auf die Qualität der Daten zur nutzbaren Feldkapazität zu richten ist, da dieser Faktor den größten Einfluß auf die Ergebnisse hat.

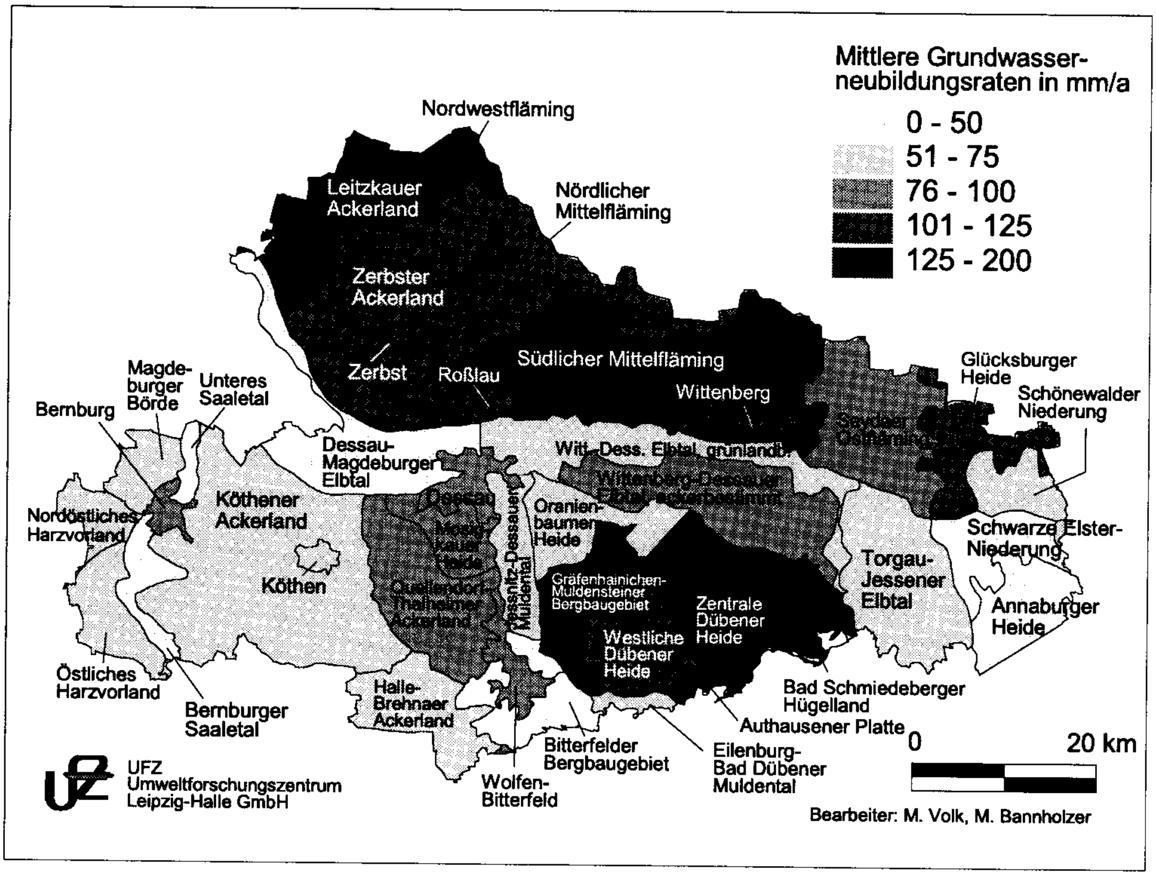


Abb. 2: Mittlere Neubildungsraten im Regierungsbezirk Dessau (bezogen auf Landschaftseinheiten).

Da nahezu keine Informationen über die Grundwasserneubildungsverhältnisse für den gesamten Untersuchungsraum existieren, mußte bei der Überprüfung der von ABIMO errechneten Werte auf Daten und Informationen von verschiedenen Untersuchungen sowohl in noch kleineren als auch in größeren Maßstäben zurückgegriffen werden. Die Werte wurden mit den Ergebnissen einer Wasserhaushaltsberechnung für den Raum Leipzig-Halle-Bitterfeld verglichen, die mit einer Vorläufer-Version des Modells ABIMO erarbeitet wurden (vgl. KRÖNERT & ERFURTH 1998) und eine geringere inhaltliche Dichte und räumliche Auflösung aufweisen. Dabei wurde unter Berücksichtigung dieses Umstandes eine weitgehende Übereinstimmung der Ergebnisse für den Bereich des Regierungsbezirk Dessau festgestellt. Ein weiterer Vergleich wurde mit den Ergebnissen einer bundesweiten Modellierung der Versickerung von Pflanzenschutzmitteln vorgenommen (vgl. HUBER O.J.). Die Eingangsdaten bezüglich Niederschlag und Verdunstung behandeln den gleichen Zeitraum (1961 – 1990) und weisen mit einer Rasterweite von 1 km die gleiche räumliche Auflösung auf wie die Basisdaten für die Berechungen mit ABIMO. Größtenteils wurden Übereinstimmungen mit den eigenen Ergebnissen erzielt, erhebliche Abweichungen konnten jedoch im Fläming verzeichnet werden. Zudem wurde eine von FELDHAUS & WILCZYNSKI (1996) vorgenommene Modellierung der Grundwasserneubildung für die Dübener Heide im Südosten des Regierungsbezirkes zum Vergleich herangezogen. Sie benutzten das Modell "WHAT". Bei der Hauptbodenart "Sand" und einem mittleren Niederschlag von 640mm/a (Vergleichswert) wurden bei dieser Studie etwas geringere Grundwasserneubildungsraten erzielt, was u.a. in der Berücksichtigung von zeitlichen Verläufen (z.B. Berücksichtigung der saisonalen Schwankungen des Niederschlag, etc.).begründet ist. Schließlich wurden die Ergebnisse Personen aus verschiedenen

Tätigkeitsbereichen vorgelegt. Für die Überprüfung der Berechnungsergebnisse für den gesamten Untersuchungsraum ergaben sich aufgrund fehlender großflächiger Messungen sowie unterschiedlicher Modellansätze erhebliche Probleme. Dennoch konnten Vergleiche mit Untersuchungen kleinerer Regionen innerhalb des Regierungsbezirk gemacht und punktuelle Messungen zum Vergleich herangezogen werden. Unter Einbeziehung aller oben genannten Vergleichsmöglichkeiten konnten die Berechnungen als realistisch beurteilt werden.

4.0 BERECHNUNG VON LANDNUTZUNGSSZENARIEN UND EMPFEHLUNG VON NUTZUNGSVARIANTEN

Um Veränderungen der Grundwasserneubildung aufgrund von Landnutzungsänderungen zu aufzuzeigen, wurden für die Landschaftseinheiten "Köthener Ackerland" und "Nördlicher Mittelfläming" Szenarien erstellt. Hierbei wurden die aus der Konfliktanalyse resultierenden, speziellen Probleme in den Landschaftseinheiten berücksichtigt. Es wurden grundlegende Landnutzungsänderungen von theoretischem Charakter erstellt, aber auch zukünftige Planungen simuliert und alternativ zu diesen Planungen Landnutzungsszenarien im Sinne des Grundwasser- und Ressourcenschutzes erstellt. Die kontaktierten Behörden, Ämter und Arbeitsgruppen zeigten Interesse an den Szenarien, die als Argumentationshilfe und Entscheidungsgrundlage dienen sollen.

4.1 Szenarien für das "Köthener Ackerland"

Das Köthener Ackerland weist eine Fläche von ist 454,9 km² auf. Die ausgeräumte Agrarlandschaft verfügt über minimale Reliefunterschiede mit maximalen Höhenunterschieden von etwa 20 m. Das Gebiet weist ein kontinental getöntes Klima mit geringen Jahresniederschlägen auf, wobei jedoch im Sommer bodengefährdende Starkniederschläge zu verzeichnen sind. Die Bodensubstrate der landwirtschaftlich äußerst intensiv genutzten Region kann größtenteils als gering wasserdurchlässig eingestuft werden. Das Berechnungsergebnis weist daher für das Testgebiet mit 55 mm/a an Grundwasserneubildungsrate (Istzustand) einen der geringsten Werte aller Landschaftseinheiten des Regierungsbezirkes auf. Wie aus Kap. 2.1 zu entnehmen, handelt es sich um eine äußerst strukturarme Landschaft. Aus der Sicht des Umwelt- und Naturschutzes wird gefordert, daß 10% der Fläche mit absolutem Vorrang dem Biotop- und Artenschutz zur Verfügung zu stellen sind und weitere 10% der Gesamtfläche mit relativem Vorrang für Biotop- und Artenschutz benötigt werden. Diese Eckwerte sind als das absolute Minimum anzustreben, um eine repräsentative, naturraumspezifische biotische Ausstattung zu schaffen und zu erhalten (vgl. HEYDEMANN 1983; JEDICKE 1994). Bestandteile dieser Ausstattung können z.B. Hecken, Gebüsche, Gewässerrandstreifen und lokale Aufforstungen sein, die sich gerade für diese ausgeräumte Region anbieten würden. Daher wurden Szenarien konzipiert, die eine Einschätzung der möglichen Auswirkungen von lokalen Aufforstungen landwirtschaftlich genutzter Flächen auf die Grundwasserneubildung und somit auch auf die Wasserressourcen zur Trinkwassergewinnung im Untersuchungsraum erlauben. Die Ergebnisse sind in Tab. 1 dargestellt.

Die lediglich als Testuntersuchung zu wertenden Szenarien „Aufforstung aller landwirtschaftlich genutzten Flächen" bzw. „Aufforstung in Bereichen mit Grundwasserflurabstand >1 und >2m" erbrachten aufgrund der resultierenden stark erhöhten Verdunstung erwartungsgemäß eine deutliche Verringerung der Grundwasserneubildungsraten. Die anderen, als realistisch einzustufenden Szenarien zeigen, daß z.B. Aufforstungen von 10 bzw. 20 % der landwirtschaftlich genutzten Fläche eine Absenkung der Grundwasserneubildungsrate von lediglich 5 bzw. 9 mm/a zur Folge haben würde, was als geringfügig und akzeptabel eingeschätzt werden kann. Diese Ergebnisse würden also die Umwidmung von Flächen für den Umwelt- und Ressourcenschutz rechtfertigen, was nachfolgend kurz diskutiert werden soll.

Infolge der Ausräumung der Landschaft und einer größtenteils fehlenden, ganzjährigen Vegetationsbedeckung kommt es zur einer starken, potentiellen Gefährdung durch Wasser- und Winderosion. Daher müssen weitere Flächen zur Anlage von Windschutzhecken, Gebüschen und Baumgruppen zur Verfügung gestellt werden (vgl. z.B. **BENJES 1994**). Weitere Bodenerosionsschutzmaßnahmen sind durch vermehrte Grünlandbewirtschaftung durchzuführen.

Die Wasserentnahme aus Wasserschutzgebieten zur Beregnung sollte in Anbetracht der sehr geringen Grundwasserneubildung unterbleiben oder zumindest verringert werden. Dabei müßte jedoch eine Kompromißlösung gefunden werden, da aus dieser Maßnahme Einbußen des Ernteertrags für die Landwirtschaft ergeben könnten.

Eine Ausdehnung der Wasserschutzgebiete sowie eine Erhöhung der Schutzkategorie mit den damit verbundenen Nutzungseinschränkungen bzw. -verboten würde die Qualität des Grundwassers sicherstellen.

Bereits seit einigen Jahrzehnten nimmt die landwirtschaftlich genutzte Fläche, vor allem für die Nahrungsmittelproduktion, in Deutschland kontinuierlich ab. Mit abnehmender Bedeutung agrarstruktureller Belange stellt sich die Frage nach Maßnahmen zur Flurbereinigung. Erste Ansätze bietet die Erweiterung des Anwendungsbereichs bestimmter Verfahrenstypen der Flurbereinigung. Die Entflechtung der Nutzungskonflikte zwischen Naturschutz und Landwirtschaft wird erleichtert, Landtausch für Naturschutzzwecke einfacher, was positive Auswirkungen auf den Grundwasserschutz hat.

4.2 Szenarien für den "Nördlichen Mittelfläming"

Die Landschaftseinheit "Nördlicher Mittelfläming" weist im Bereich des Regierungsbezirkes Dessau eine Fläche von 169,3 km² auf. Mit der Berechnung des "Ist-Zustandes" konnte zunächst gezeigt werden, daß die Landschaftseinheit Nördlicher Mittelfläming mit 125mm/a die höchste Grundwasserneubildungsrate im Regierungsbezirk aufweist. Dieses Ergebnis ist u.a. mit den im Vergleich zu anderen Landschaftseinheiten höheren Niederschlagswerten und dem durchlässigen Substrat begründet, so daß die Ausweisung des Gebietes als Vorranggebiet für Wassergewinnung zunächst als durchaus gerechtfertigt erscheint. Die höheren Niederschlagswerte resultieren einerseits in der Exposition des Gebietes in Hauptwindrichtung Südwest, anderseits wird durch die größere Entfernung die niederschlagsabschirmende Wirkung des Harzes abgeschwächt. Gemäß dem Klassifizierungsvorschlag von MARKS et al. (1992) ist dieser Wert aber immer noch als gering einzustufen. Das Gebiet wird durch große Kiefernwaldbestände geprägt. Auf einigen kleineren Flächen befinden sich Bestände von Rotbuchen- Traubeneichen- Mischwäldern. Die Waldbestände sind in jüngster Zeit durch die Entnahmen von Grundwasser für die Trinkwassergewinnung (Absenktrichter durch Wasserentnahme etc., vgl. Kap. 2.1) gefährdet. Gleichzeitig wird von einer Gefährdung des Grundwassers durch Stoffaustrag aus den vorhandenen landwirtschaftlich genutzten Flächen ausgegangen. Die Lösung der erwähnten Probleme erfordert die Empfehlung von Landnutzungsvarianten, die sich positiv auf die Erhaltung und den Schutz des Grundwassers und der Waldbestände auswirken. Daher wurden Szenarien erarbeitet, die die Auswirkungen von Landnutzungsänderungen und flächenhafte Grundwasserentnahme auf die Grundwasserneubildung in diesem Sinne aufzeigen:

- *Grundwasserschutz (Verringerung bzw. Vermeidung von Stoffaustrag aus landwirtschaftlich genutzten Flächen durch Aufforstung):* Welche Auswirkungen haben teilweise oder komplette Aufforstungen von landwirtschaftlich genutzten Flächen auf die Grundwasserneubildung bzw. die Gesamtverdunstung?

- *Schutz und Erhalt der Waldbestände (Erhöhung von Wasserentnahmestellen zur flächenhaften Entnahme, Vermeidung ausgedehnter, punktueller Absenktrichter):* Welche Auswirkungen haben flächenhafte Absenkungen auf die Grundwasserneubildung?

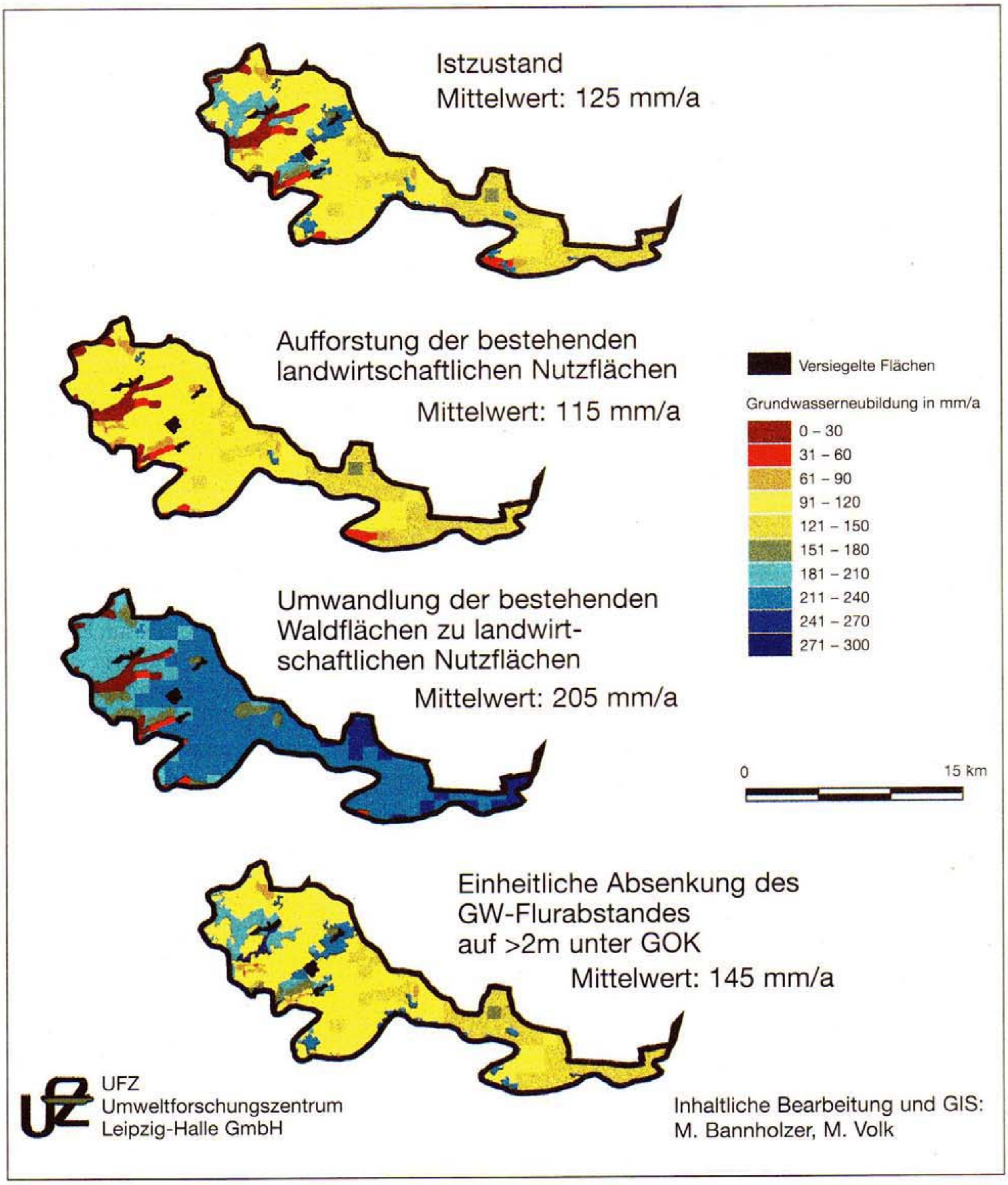


Abb. 3: Grundwasserneubildung in der Landschaftseinheit „Nördlicher Mittelfläming": Ist-Zustand und Szenarien

Tab. 1: Grundwasserneubildungsraten im Köthener Ackerland: Ist-Zustand und Landnutzungsszenarien

Ist-Zustand und Landnutzungsszenarien	GW-Neubildung in mm/a
Ist-Zustand	55
Aufforstung von bestehenden landwirtschaftlichen Nutzflächen	23
Aufforstung von landw. Nutzflächen in Bereichen mit GW-Flurabstand >1m	22
Aufforstung von landw. Nutzflächen in Bereichen mit GW-Flurabstand >2m	24
Aufforstung von 10% der landwirtschaftlichen Nutzflächen	50
Aufforstung von 20% der landwirtschaftlichen Nutzflächen	46

Tab. 2: Grundwasserneubildungsraten im Nördlichen Mittelfläming: Ist-Zustand und Landnutzungsszenarien

Ist-Zustand und Landnutzungsszenarien	GW-Neubildung in mm/a
Ist-Zustand	125
Aufforstung von bestehenden landwirtschaftlichen Nutzflächen	115
Umwandlung von bestehendem Wald zu landwirtsch. Nutzflächen	205
Aufforstung landw. Nutzflächen in Bereichen mit GW-Flurabst. >2m	114
Aufforstung von 10% der landwirtschaftlichen Nutzfläche	123
Einheitliche Absenkung des GW-Flurabstandes auf >2m	145

Den Berechnungsergebnissen (vgl. Tab. 2 und Abb. 3) zufolge sinkt die Grundwasserneubildung aufgrund des relativ geringen Anteils dieser Flächen nur unerheblich bei einer kompletten (um 10mm/a) bzw. teilweisen Umwidmung landwirtschaftlicher Flächen zu Wald (um 2mm/a bei 10% Umwidmung landwirtschaftlicher Fläche zu Wald). Dieses Ergebnis kann sehr gut als Argumentationshilfe hinsichtlich der Verbesserung des in der Region notwendigen Grundwasserschutzes (Verringerung des Anteiles landwirtschaftlicher Flächen durch Aufforstung, Veränderung der Anzahl der Entnahmebrunnen, etc.) verwendet werden.

Bei einer einheitlichen Absenkung des Grundwasserspiegels auf >2m würde sich aufgrund geringerer Oberflächenverdunstung und verringerten kapillaren Aufstiegs eine Zunahme der Grundwasserneubildung um 20mm/a ergeben, was demnach für eine Erhöhung der Entnahmebrunnen bei gleichbleibender Entnahmemenge sprechen würde Eine komplette Umwandlung der Waldbestände in landwirtschaftliche Flächen würde aufgrund der stark verringerten Oberflächenverdunstung sogar zu einer Zunahme von 80mm/a auf 205mm/a zur Folge haben würde. Diese beiden Szenarien besitzen allerdings einen theoretischen Charakter, die als weitere Überprüfung des Verfahrens gewertet sollten. Im Hinblick auf die eher geringen Auswirkungen der Landnutzungsvarianten auf die mittleren Grundwasserneubildungsraten der gesamten Landschaftseinheit sind dennoch deutliche Veränderungen innerhalb der Bezugseinheit (z.B. Szenario „Aufforstung landwirtschaftlicher Nutzflächen") zu verzeichnen (vgl. Abb. 3) . Die Grundwasserneubildungsrate verändert sich je nach gewähltem Szenario in diesen kleinen Flächen bei einer Aufforstung um etwa 100mm/a, wobei hier keine Aussagen über die ökologischen Folgen getroffen werden können. Für detaillierte Untersuchungen dieser Flächen sind andere Modellsysteme (z.B. ASGi, AGNPS, etc.) und höher aufgelöste Grundlagendaten erforderlich. Mit der gewählten Methode können aber Größenordnungen des Wasserhaushaltes großer Flächeneinheiten bestimmt und sensible Flächen - wie oben genannt - ausgewiesen und auf kleinräumige Unterschiede innerhalb von Bezugseinheiten hingewiesen werden.

Es wurde versucht, die Problematik um Grundwasserneubildung und Grundwasserentnahme in der Landschaftseinheit anhand einer Wasserbilanz transparent zu machen:

Wasserentnahme in den Wasserschutzgebieten durch das Wasserwerk Lindau:
50.000 m^3/d entspricht: **18.250.000 m^3/a**

Gesamtfläche der Wasserschutzgebiete:
15.757 ha entspricht: 157.570.000 m^2
Grundwasserneubildung im langjährigen Mittel im Nördlichen Mittelfläming:
125 mm/a entspricht: 0,125 m/a

Grundwasserneubildung multipliziert mit Gesamtfläche:
0,125 m/a * 157.570.000 m^2 = **19.696.250 m^3/a**

Das Ergebnis weist darauf hin, daß bis zu 93% des gebildeten Grundwassers für die Trinkwassergewinnung wieder entnommen werden. Da bei einer großräumigen Modellrechnung dieser Art auf eine eher unsichere Datenlage mit geringer räumlich-zeitlicher Auflösung zurückgegriffen werden muß und somit auch das Ergebnis beeinflußt, stellt dies nur einen theoretischen Wert dar, der lediglich als Größenordnung gewertet werden darf. Berücksichtigt man aber, daß das Modell ABIMO reliefbedingte Oberflächenabflüsse nicht mit in die Berechnung einbezieht, ist davon auszugehen, daß der prozentuale Anteil der Entnahme noch höher liegt. Dies ist in der Tatsache begründet, daß das Untersuchungsgebiet als eine der wenigen Landschaftseinheiten im Regierungsbezirk durchaus Reliefdifferenzierungen aufweist (Δh 93,0m) und der daraus resultierende (aber nicht erfaßte) Oberflächenabfluß die Grundwasserneubildungsrate reduziert.

Neben der Problematik einer nicht zu erwartenden Beschränkung der Grundwasserentnahmemenge, stellt sich zusätzlich das Problem der Kontaminationsgefährdung des Grundwassers durch Stoffaustrag aus landwirtschaftlich genutzten Flächen..

Die Sickergeschwindigkeit ist eines der maßgeblichsten Kriterien für die Kontaminationsgefährdung. In der Landschaftseinheit "Nördlichen Mittelfläming" liegen zum größten Teil sandige Böden vor, die durch eine hohe Sickergeschwindigkeit und damit auch eine hohe Transportrate charakterisiert sind. Der Speichereffekt von Böden unter Wald kann hier nur bedingt berücksichtigt werden. Diese Faktoren kombiniert mit dem Auskämmeffekt von Schadstoffen - vor allem in Nadelwäldern - beeinträchtigt die Qualität des Grundwassers. Schlußfolgernd ist davon auszugehen, daß bei der derzeitigen Situation mit einer Gefährdung von Quantität und Qualität des Grundwassers zu rechnen ist.

Angesichts dieser Überlegungen muß die Einführung einer naturnahen und an die gebietsspezifischen Erfordernisse des Grund- und Trinkwasserschutzes, des Naturschutzes und der Erholung angepaßten Forstwirtschaft durchgesetzt werden. Das Ziel muß darin bestehen, dem Aquifer möglichst viel Niederschlag zuzuführen.

Aufgrund der zu beobachtenden tendenziellen Absenkung des Grundwasserspiegels muß darauf hingearbeitet werden, die Entnahmesituation im Nördlichen Mittelfläming zu entschärfen. Dies kann durch eine Erhöhung der Anzahl von Entnahmebrunnen zwecks flächenhafter Grundwasserentnahme erreicht werden, um Absenktrichter in der bestehenden Größenordnung zu vermeiden bzw. zu verringern.

Eine weitere Möglichkeit der grundwasserschonenden Wasserentnahme sind zeitlich koordinierte Entnahmen: Die Grundwasserneubildung ist saisonalen Schwankungen unterworfen; Perioden starker Grundwasserzehrung wechseln mit Perioden starken Grundwasseranstiegs.

Eine Einbindung der Wasserverbände in das "GW-Überwachungssystem Westfläming" ist anzustreben, um dem Zustand des Grundwassers entsprechende Abstimmungen der Entnahmen zu ermöglichen.

Bezüglich der im Rahmen eines Meliorationsprojekts gezogenen Gräben ist ein Rückbau anzustreben. Die von der Landwirtschaft geforderten Entwässerungstiefen sind der Höhenlage des Grundwasserspiegels abträglich. Eine intensive landwirtschaftliche Nutzung auf Flächen der Schutzkategorie II und III muß unterbleiben. Dort ist eine zumindest teilweise Aufforstung als sinnvoll zu erachten. Gemäß den Berechnungsergebnissen würde diese Maßnahme lediglch einen geringen Einfluß auf die Grundwasserneubildungsraten besitzen.

Die naturnahen Rotbuchen- Traubeneichen- Mischwälder bei Nedlitz sind schützenswert und damit zu erhalten und zu erweitern. Die im Nördlichen Mittelfläming vertretene Hauptbaumart Kiefer soll allmählich durch Eiche und Buche ersetzt werden.

Kiefernmonokulturen können zu anthropogener Versauerung der Böden führen, was abhängig vom pH-Wert zur Mobilisierung von Schwermetallen und deren Anreicherung im Sickerwasser, gerade in den durchlässigen, sandigen Böden des Gebiets, führen kann. Der ganzjährige Effekt der Schadstoffauskämmung bei Nadelwäldern kann durch Umwandlung in Laubwald bzw. Einmischen von Laubbaumbestand reduziert werden. Damit sinkt auch die Gebietsverdunstung.

5.0 GRENZEN DER MODELLANWENDUNG

Die Möglichkeiten und Grenzen einer Wasserhaushaltsmodellierung im regionalen Maßstab hängen im Wesentlichen von dem verwendeten Modell, der Qualität der Datengrundlagen sowie deren räumlich-zeitlicher Auflösung ab. Um die Anwendungsmöglichkeiten kleinmaßstäbiger Wasser- und Stoffhaushaltsmodellierungen verbessern zu können, müssen daher die Stärken und Schwächen des Rechenverfahrens und die Datengrundlage detailliert überprüft werden, um die Aussagefähigkeit der Ergebnisse einschätzen zu können. Vor der Anwendung des Modells mußten die Eingangsparameter Boden und Landnutzung durch eigens entwickelte Verfahren (vgl. VOLK & STEINHARDT 1998) aufgrund unterschiedlicher räumlicher und inhaltlicher Auflösung bearbeitet werden. Diese Aggregierung bzw. Disaggregierung spielt für landschaftsökologische Analysen, bei denen Informationen unterschiedlicher Inhalte und Maßstäbe miteinander gekoppelt werden, eine entscheidende Rolle. Hier ist die Entwicklung standardisierter Verfahren erforderlich, um inhaltliche Verfälschungen zu minimieren. Zudem können bei der Verschneidung von Informationen unterschiedlichen Maßstabs und Inhalts in zunehmendem Maß real nicht existente Einzelflächen entstehen, die zur einer Ergebnisverfälschung führen können. Diesem Umstand muß bei landschaftsökologischen Komplexanalysen und Modellierungen besondere Beachtung geschenkt werden.

Bei dem hier angewendeten Verfahren des Abflußbildungsmodells (ABIMO) wird - wie eingangs dargestellt - unter "mittlerer Abflußbildung" die Differenz aus vieljährigem Mittel von Niederschlag und realer Verdunstung verstanden, die dem mittleren Gesamtabfluß entspricht. Unter der Voraussetzung einer ausschließlich vertikalen Sickerung bis zur Grundwasseroberfläche stimmt dieser Wert mit der Grundwasserneubildung überein. Da dies jedoch nur in den seltensten Fällen gegeben ist, kann die Größe des mittleren Gesamtabflusses nur als Summe indifferent für ober- und unterirdischen Abfluß aufgefaßt werden. Da laterale Flüsse wie reliefbedingter Oberflächenabfluß oder Interflow somit nicht mit in die Berechnung eingeht, ist eine sinnvolle Anwendung des Modells nur in Gebieten möglich, die keine oder nur eine sehr geringe Reliefenergie aufweisen. Dies ist im Untersuchungsgebiet der Fall. Für kleinere Untersuchungsgebiete innerhalb der Region mit höherer Reliefdifferenzierung (wie z.B. dem Testgebiet "Nördlicher Mittelfläming") sind im Rahmen des Projektes zusätzliche Reliefanalysen und Wasser- und Stoffhaushaltsmodellierungen mit höher auflösenden Modellen vorgesehen.

Um die dimensionsspezifischen Grenzen der Anwendung des Modells und der verfügbaren Basisdaten zu überprüfen, wurde als drittes Testgebiet die flächenmäßig kleinere Landschaftseinheit "Oranienbaumer Heide" (mit 93,4 km² etwa 2,3% der Gesamtfläche des Regierungsbezirkes Dessau) ausgewählt. Im Ergebnis der Berechnungen und der visuellen Interpretation der Karte zeigte sich, daß sich die räumliche Auflösung der Klimadaten von 1km als limitierender Faktor abzeichnet. Ein weiterer Fehler ergab sich bei den verwendeten CORINE-Landnutzungsdaten, bei dem die ehemalige Nutzung des Gebietes als Truppenübungsplatz aufgrund der zu geringen räumlichen Auflösung falsch klassifiziert ist. Zudem liegt die Vermutung nahe, daß die ehemalige militärische Nutzung der Oranienbaumer Heide durch schwere Fahrzeuge zu starken Bodenverdichtungen geführt haben, die in den vorliegenden Bodendaten (z.B. nFK) nicht berücksichtigt werden können. Die Ergebnisse der verschiedenen Berechnungen zur Grundwasserneubildung liegen sehr eng beieinander, die Unterschiede lassen sich nur schwer darstellen. Die genannten Faktoren weisen darauf hin, daß zur Berechnung der Grundwasserneubildung und anderer Parameter des Wasserhaushaltes für ein Gebiet dieser Größe Basisdaten von höherer räumlicher und zeitlicher Auflösung erforderlich sind. Dazu werden derzeit für den Gesamtraum vergleichende Berechnungen mit den Biotoptypen/Nutzungstypen als Eingabeparameter "Landnutzung" durchgeführt, die eine weitaus höhere räumliche Auflösung aufweisen.

Weitere Probleme ergaben sich bei der Behandlung von ehemaligen Abbauflächen (bes. Braunkohle) und deren Kippen und Halden, die insgesamt etwa 1,5 % der Gesamtfläche ausmachen. Die Ableitung der erforderlichen pedologischen Parameter für Kippböden ist sehr problematisch. Zudem sind durch die langjährigen Grundwasser-Absenkungen in den Bergbaugebieten keine Informationen über die aktuellen Flurabstände vorhanden. Da Bodenart, sowie die Parameter nutzbare Feldkapazität und Flurabstandsklassen aber wichtige Eingangsdaten für das Modell darstellen, wurden diese Flächen nicht bewertet, da die Berechnungsergebnisse unrealistisch wären. Auch Informationen über das Fließverhalten des Grundwassers in Grundwasserleitern oder Veränderungen von Flurabständen können nicht abgeleitet werden.

Die Anwendung des Modellsystems setzt sowohl Kenntnis unterschiedlicher Software als auch eine entsprechende Hardware-Austattung voraus. Das Verhältnis von zeitlichem Arbeitsaufwand und Ergebnisqualität ist mitunter unausgewogen. In Zukunft ist eine Vereinfachung des Ablaufs der Modellierung, z.B. durch die Reduzierung der anzuwendenden Software erstrebenswert, um die Bedienungsfreundlichkeit und damit die Akzeptanz zu erhöhen. Bearbeitungszeit und Benutzerfreundlichkeit sind Punkte, die aufgrund von wirtschaftlichen Interessen und Fragen der Motivation zur Benutzung nicht vernachlässigt werden dürfen, wenn man über den Einsatz eines solchen Modellsystems im planerischen Bereich nachdenkt.

Ein sehr wichtiger Punkt ist die Validierung der Ergebnisse. Eine Überprüfung von Modellierungsergebnissen im regionalen Maßstab ist bisher nicht oder nur sehr bedingt möglich. Daher sind ergänzend Befragungen von Fachleuten und Vergleiche mit Berechnungen und Messungen in kleineren Testgebieten unabdingbar. Die Aussagekraft der ermittelten Werte hat daher einen stark qualitativen Charakter. Es muß klar hervorgehoben werden, daß für großräumige Betrachtungen mit dem verwendeten Modell sowohl bezüglich des Istzustands als auch für Landnutzungsszenarien "nur" Größenordnungen bzw. Einordnungen erarbeitet werden können. Dies kann jedoch bereits eine wertvolle Zusatzinformation für die Landes- und Regionalplanung darstellen, die eine verbesserte Ausweisung von Vorrang-, Vorsorge- und Schutzgebieten im Hinblick auf den Umwelt- und

Ressourcenschutzes ermöglichen und Landnutzungskonflikte vermeiden helfen. Für detaillierte Wasser- und Stoffhaushaltsberechnungen mit qualitativem und quantitativen Charakter, die zudem laterale Stoffflüsse berücksichtigen, sind andere Modelle und eine bessere Datengrundlage nötig.

6.0 SCHLUSSFOLGERUNGEN UND AUSBLICK

Die durchgeführten Untersuchungen haben gezeigt, daß das verwendete Modell ABIMO ein geeignetes Rechenprogramm zur Modellierung der Grundwasserneubildung in regionalem Maßstab ist. Die Ergebnisse erlauben eine großräumige Abschätzung bzw. Einordnung der Grundgrößen des Landschaftswasserhaushaltes. Regionale Unterschiede sowie qualitative und quantitative Gefährdungspotentiale für den Wasserhaushalt können deutlich gemacht werden und in planerische Fragestellungen einbezogen werden. In Gebieten mit Nutzungsüberlagerungen können die Ergebnisse der Modellierungen als Entscheidungsgrundlage für Landnutzungsvarianten im Sinne des Umwelt- und Ressourcenschutzes dienen. Dadurch wird der Landes- und Regionalplanung ein Instrument für die verbesserte Ausweisung von Vorsorge-, Vorrang- und Schutzgebieten sowie zur Konfliktminderung bereitgestellt.

Mit den Simulationsergebnissen der Landnutzungsszenarien für die beiden Testgebiete "Köthener Ackerland" und "Nördlicher Mittelfläming" konnten die Auswirkungen von Landnutzungsänderungen auf die Grundwasserneubildung verdeutlicht und darauf basierende Landnutzungsempfehlungen abgeleitet werden. Die Ergebnisse sollen von den verantwortlichen Dezernaten im Regierungspräsidium Dessau bzw. von zuständigen Forstämtern als Argumentationsgrundlage für den Grundwasserschutz verwendet werden.

Die Berechnungsergebnisse präsentieren sich ich zwar in Zahlen, geben jedoch aufgrund des Modellcharakters und der geringen räumlich-zeitlichen Auflösung der Eingangsdaten nur Größenordnungen wider. Für Gebiete mit mittlerer bis hoher Reliefenergie, sowie für detailliertere Untersuchungen (laterale und vertikale Stoffflüsse mit qualitativen und quantitativen Charakter) in größeren Maßstabsebenen ist die Verwendung anderer Modelle und genauerer Basisdaten erforderlich. Dabei muß es Ziel künftiger Untersuchungen sein, die Grenzen der Aussagefähigkeit jeder Dimensionsstufe zu definieren. Die rasante Entwicklung von Geoinformationssystemen und Wasser- und Stoffhaushaltsmodellen kann diesen Prozeß beschleunigen, da dies neben der Genauigkeit der Basisdaten auch von der Leistungsgrenze des Rechners bzw. dem vertretbaren Verhältnis von zeitlichem Aufwand und erzieltem Ergebnis abhängt. Die inzwischen weit verbreitete Verwendung von Geoinformationssystemen in öffentlichen Einrichtungen scheint diesen Trend im positiven Sinne zu bestätigen. In Rahmen des Projektes, in dem die vorgestellte Studie angesiedelt ist, verwenden die Kooperationspartner (Landesamt für Umweltschutz Sachsen-Anhalt, Geologische Landesamt Sachsen-Anhalt, Regierungspräsidium Dessau) Arc/Info und Arc/View, was den Datenaustausch erleichert und die Zusammenarbeit zwischen Forschung und Praxis fördert.

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The impact of land use changes on natural groundwater recharge

Martin Volk, Felix Herzog, Thomas Schmidt and Stefan Geyler

1 Introduction and aims

As well as playing a host of ecological functions, natural groundwater recharge is enormously important within the renewal of drinking water resources. Land use (and changes thereto) affect the evapotranspiration of soil and plants, and hence also significantly influence natural groundwater recharge via the landscape water balance. The different interests of farming, forestry, and the water industry as well as nature conservation and landscape protection can generate conflicts which can only be solved by taking an integrated approach to evaluating landscape and socioeconomic components (cf. Horsch/Ring in Chapter 1; cf. also O'Callaghan 1996; Dabbert et al. 1999). This article uses the example of the Torgau district to present a way of modelling how land-use changes influence natural groundwater recharge. Although the study area is mainly used for agriculture, it also contains extensive drinking water protection zones as well as landscape protection areas and nature reserves. Our aim here is to quantitatively assess how land-use changes affect natural groundwater recharge. Moreover, by using the assessment criterion 'natural groundwater recharge minus groundwater extraction', we can also roughly determine the sustainability of the land-use developments considered with respect to quantitative groundwater resources. These findings can then be considered in the multicriteria analysis of the action alternatives, and also provide a basis for investigations into leachate quality. Apart from these objectives, this examination of model algorithms, the modification of the input data and sensitivity analyses is designed to help optimize usage of the run-off formation model ABIMO.

2 Modelling water resources; land-use changes

The first water balance models were produced back in the late 1940s. Since that time they have evolved in a number of different directions depending on the many purposes for which they are required (Dyck 1983, Xu et al. 1996). Generally speaking, nowadays there are three different approaches to modelling the landscape water balance:

1 *Physical-deterministic models*, which are based on the fundamental laws of physics (chiefly hydrodynamics and thermodynamics), chemistry, biology, etc.;

2 *Conceptional models*, which take these laws into account in a simplified manner and also use empirical approaches:

3 *Empirical-statistical models*, which are solely based on the empirically measured cause-effect relations of system inputs and outputs without trying to fathom the laws on which they are based.

The transitions between the three approaches are in a state of flux. Moreover, hydrological processes always include both deterministic and stochastic characteristics owing to the unavoidable simplifications of complex reality and the error occurring when capturing input data (Nemec 1993).

Depending on the model type and its purpose, different scales of temporal and spatial resolution are used. Compromises usually have to be made between the accuracy desired and the availability of data. If non-linear processes are being studied (e.g. precipitation and run off), hourly or daily steps have to be used, whereas for seasonal or annual properties monthly or even yearly steps suffice. The possible degree of spatial resolution ranges from highly aggregating approaches in which the study area is divided into a few sub-units with similar geophysical properties ('lumped models') to models which as far as possible take into account the variability of spatial structures (‘distributed models’). The scope for using a higher spatial resolution has improved with the advent of faster, more powerful computers, the development of geographic information systems, and – especially recently – the growing availability of data stored in a GIS.

Land use is the parameter via which human society affects the landscape water balance (e.g. Calder/Newson 1979). Water balance models are used to depict the impact of land-use changes on potential natural groundwater recharge (e.g. Liebscher et al. 1996, Volk/Bannholzer 1999). Usually variants or scenarios are investigated which are for instance based on assumptions concerning climatic change or the impact of political decisions (Tab. 1). If balancing is performed over the entire study area, only very pronounced land-use changes will result in any considerable shifts in the simulated total run-off. What the list in Tab. 1 fails to show are any local changes, which in certain areas may well considerably exceed the means calculated. The algorithm used to introduce the land-use changes forecast into the area also needs to be considered. Depending on local characteristics, the effect on further modelling exerted by a change in land use varies greatly. Assumptions regarding the spatial distribution of land-use changes can be made by using plausibility considerations. Alternatively, models can also be used (Fohrer et al. 1999).

Table 1: Case studies on land-use changes and the landscape water balance.

Region	Scenario basis	Land-use changes	Forecast change to total run-off	Authors
Northeast Germany	EU agricultural reform	Converting 4% of agricultural land to forest	–1%	Werner et al. (1997)
Northeast Germany	EU agricultural reform	Converting 32% of agricultural land to forest	–10%	Werner et al. (1997)
Hessen	Agricultural policy: grassland premium	Forest: 42% → 13% Agricultural land: 44% → 73%	+8%	Fohrer et al. (1999)
Hessen	Agricultural policy: cessation of animal husbandry	Forest: 42% → 49% Agricultural land: 44% → 37%	+2%	Fohrer et al. (1999)
Saxony-Anhalt	Analysis of usage conflicts in priority areas (agriculture vs. groundwater protection)	Switching 10% of agricultural land to forestland *Köthen farmland* *Nördl. Mittelfläming*	–9% –2%	Volk & Bannholzer (1999)

3 The ABIMO run-off formation model: methodology and data basis

3.1 Methodology

The conceptual model ABIMO (Version 2.1) by the Bundesanstalt für Gewässerkunde/Federal Institute of Hydrology was used (cf. Rachimov 1996; Glugla/Fürtig 1997) to calculate the natural groundwater recharge. Examples of using ABIMO to calculate natural groundwater recharge in central Germany have been published by the Sächsisches Landesamt für Umwelt und Geologie/Saxon Department of the Environment and Geology (LfUG 1995a), Herzog/Kunze (1999), Volk/Bannholzer (1999) and Petry et al. (2000). The program is employed to calculate mean long-term run-off formation and water balances. 'Mean run-off' refers to the difference between the long-term mean precipitation and actual evapotranspiration. This difference corresponds to the average total run-off. Assuming solely vertical leaching, this figures matches the natural groundwater recharge. In order to calculate the above-mentioned actual evapotranspiration, the BAGROV formula is used. (DVWK 1996). The BAGROV formula combines the local inflows (precipitation, potential evaporation, land use, soil) in a physical and empirically based manner to determine the actual evapotranspiration (cf. Fig. 1).

The empirical component (effectiveness parameter *n*) is based on an extensive evaluation of lysimeter measurements. The parameter *n* expresses the extent of vegetation development. Given the same conditions, *n* is generally higher for grassland than farmland, since grassland spends more of the year covered by vegetation. ABIMO is suitable for flat areas or slightly sloping terrain where surface run-off can be neglected. This is largely true of the study area, and so the total run-off there approximately corresponds to natural groundwater recharge.

Fig. 1: Algorithm of the ABIMO run-off model

ABIMO

Abflussbildungsmodell

$$R = P_0 - ETa$$

BAGROV-Beziehung

$$\frac{dETa}{dP_0} = 1 - \left(\frac{ETa}{ETp}\right)^n$$

Langjährige Mittelwerte von:

R	Gesamtabfluss
P_0	Niederschlag
ETa	Evapotranspiration
ETp	potenzielle Evapotranspiration
n	f(Boden, Bewirtschaftung)

3.2 Input data, GIS and digital transfer

ABIMO requires various input data for calculation. The information used is listed in Tab. 2. Much of the data had to be adapted to make it suitable for the model. Substantial methodological dilemmas regarding the data had to be solved, especially concerning the soil data (e.g. should data be used from medium-scale site mapping or soil taxation assessment?), climatic data (measuring series from 1931–60 or 1961–90?), spatial aspects (raster squares or the smallest common geometries?) (Herzog/Kunze 1999, cf. Volk/Steinhardt 1998, Petry et al. 2000, Herzog et al. 2001b).

Tab. 2: Input data used for the ABIMO calculations

Data level	**Source**	**Explanation**	**Data form**
Precipitation	DWD	Mean annual precipitation level (1961–90)	1km x 1km raster data
Potential evapotranspiration	DWD	Mean annual potential evapotranspiration rate (1961-1990)	1km x 1km raster data
Land use	LfUG	Digital biotope type mapping (1993)	Vector data, 1:10,000
Land surfacing	LfUG	Derived from biotope type mapping	Vector data, 1:10,000
Depth of groundwater table	LfUG	Groundwater table model of groundwater forecasting for the Leipzig region (HGN et al. 1996)	250m x 250m raster data
Soil data: - Soil type - Usable field capacity	LfUG	Soil taxation data and forestry site survey from groundwater forecasting (HGN et al. 1996)	500m x 500m raster data
Tree type	LfUG	Derived from biotope type mapping	Vector data, 1:10,000

DWD – Deutscher Wetterdienst/German Meteorological Service

LfUG – Sächsisches Landesamt für Umwelt und Geologie/Saxon Department of the Environment and Geology

Source: Kunze 1998; Herzog/Kunze 1999; Neubert 2000.

As an example of a set of model input data, Map 9 in the appendix shows the usable field capacity of soil in the study area. Clear differences are to be seen with soils in the sandy heathland areas containing relatively low values and soils in the Elbe floodplain with relatively high usable field capacities.
The geographic information systems Arc/Info and ArcView Processing were used to manage data, process the findings of modelling (e.g. combine them with other information levels), and then evaluate them. The approach taken can be summed up as follows:

– *Data preparation:* Aggregation, classification and combination of the individual information levels in the GIS for the suitable arrangement and formation of the smallest common geometries.

– *Processing the database:* Exporting the resulting database into Excel for processing to produce a structure suitable for modelling.

– *Calculation in ABIMO:* Loading the data table into ABIMO; calculation.

– *Further processing of the results:* Assigning the calculated values to the corresponding areas using the ID numbers in Excel.
– *Evaluation of the calculated results:* Statistical evaluation (area-weighted assessment) and cartographic evaluation in the GIS.

The results of the ABIMO calculations can be validated by using the long-term mean levels of rivers draining the assessment area (cf. HGN et al. 1996). Both subterranean and surface inflows and run-offs need to be considered. The accuracy of the calculation is estimated from the ratio between the mean run-off measured and calculated. Unfortunately, however, this is not possible for the assessment areas Elbe and Schwarzer Graben. In the Elbe area the surface inflow is 250 times higher than the calculated run-off, meaning the latter can vary enormously without essentially affecting the quality criterion. And as far as Schwarzer Graben is concerned, level data are only available from a series of measurements dating back to 1912–44, which do not permit any conclusions about current conditions. Nevertheless, previous investigations in which the natural groundwater recharge was estimated for the entire Leipzig region using the Glugla method (HGN et al. 1996) produced a validatable result for 10 of the 15 assessment areas. It can thus be assumed that the model calculations at least represent plausible solutions.
The different conditions for gravel extraction assumed result in three scenarios each for the two development frameworks REALIST (realistic tendencies) and EXPANSION (optimistic tendencies). The scenarios are each based on the period until 2030. The base year chosen is 1993 – the year from which the biotope typing mapping stems.

4. Calculated natural groundwater recharge in the Torgau district

4.1 Natural groundwater recharge in the base year 1993

The calculations produce an average natural groundwater recharge rate of about 133 mm/a for the Torgau district. Compared to other regions in Germany, this is rather low (cf. for instance Hölting 1996). The results of the calculations allow regional differentiation of groundwater recharge taking into account the prevailing natural conditions and land-use types (Map 10 in the appendix). As expected, this indicates that the highest values exist in heathland areas, which contain permeable, sandy soil beneath arable farmland. These areas also feature the highest precipitation. Far lower recharge rates occur in the Elbe floodplain due to the lower permeability of the soils predominating and the lower precipitation in the Elbe floodplain. Generally speaking, assuming the same soil and precipitation conditions, natural groundwater recharge is lower beneath forestland owing to the higher evapotranspiration rates than for example below arable land or grassland. Apart from areas of water, the areas with negative or very low groundwater recharge are mainly located in Annaburger Heide as well as between Torgau and Mockrehna. This is accounted for by the low depth of the groundwater table there and the resulting higher evapotranspiration.

4.2 Groundwater recharge in 2030 for each scenario

Although regional differentiation persists within natural groundwater recharge in the future scenarios, certain changes occur compared to the base year (1993; cf. Map 11 in the appendix). Whereas the additional surfaced areas and afforested areas lead to a reduction of the natural groundwater recharge, the simulated forest conversion measures (coniferous to deciduous forest) result in greater groundwater recharge, particularly in heathland areas.

On average, the assumed land-use changes prompt a reduction of the natural groundwater recharge rate by about 2.3% to around 129 mm/a (Tab. 3). The main reason for this is the increase in land surfacing, resulting in reduced leaching and higher evapotranspiration. The expansion of forestland also diminishes the natural groundwater recharge rate owing to greater evapotranspiration. The simulated additional areas of water resulting from the assumed gravel extraction schemes have a minor effect (higher evapotranspiration being offset by decreasing natural groundwater recharge). By contrast, the forest conversion schemes boost natural groundwater recharge since deciduous forest is characterized by lower evapotranspiration than coniferous forest.

Tab. 3: Mean natural groundwater recharge (weighted by area) in the Torgau district

	Base year: 1993	Scenarios: 2030					
		REALIST			EXPANSION		
		R_1	**R_2**	**R_3 & R_4**	**G_1**	**G_2**	**G_3 & G_4**
Natural groundwater recharge rate [mm/a]	132.9	129.9	130.3	129.9	129.4	129.5	129.7
Volume [million m^3/a]	91.2	89.1	89.4	89.1	88.9	88.9	89.0

The low differences between the individual scenarios become apparent when comparing the area-weighted means (cf. Tab. 3). On the whole, effects which reduce natural groundwater recharge slightly predominate. However, the positive effects of forest conversion almost compensate for the impact on the entire area. The reduction of natural groundwater recharge in the EXPANSION framework compared to REALIST is due to the higher proportion of land surfacing contained in the optimistic assumptions. The changes to the natural groundwater recharge are minor with respect to the means for the entire area. Even when considering the individual land-use types separately, clear changes are only apparent with regard to forest and surfaced land (cf. Map 11 in the appendix).

The most obvious differences result from local consideration of the changed individual areas. Compared to the base year 1993, the converted areas (afforestation and built-up areas) have reduced groundwater recharge rates far exceeding 80 mm/a in some areas (cf. Map 11 in the appendix and Tab. 4) – although this does not say anything about the ecological consequences.

Table 4: Natural groundwater recharge rates in areas put to different use[1]

[1] Area-weighted means of annual groundwater recharge for those areas whose use is altered in the scenarios.

Area	State: 1993 Use stays the same [mm/a]	State: 2030 Use changes [mm/a]	[%]
Afforestation: About 1,900 ha	163.6	81.4	–50.2
Forest conversion: About 20,000 ha	91.7	93.7	+2.2
Surfaced land: REALIST: about 1,533 ha	168.5	131.4	–22.0
Surfaced land: EXPANSION: about 2,530 ha	164.0	132.0	–19.5
Gravel extraction: Scenario R_1: 166.84 ha	115.1	–105.7*	–192.0
Gravel extraction: Scenario G_1: 227.73 ha	126.6	– 106.4*	–216.0

* Area of water

More detailed investigations of these local areas would entail different model systems and sets of data with a higher spatiotemporal resolution. In the vicinity of planned gravel pits and the resulting emergence of new areas of water, as expected the natural groundwater recharge rates are negative owing to the high evapotranspiration rates above areas of water. Tab. 4 shows the differences between the calculated groundwater recharge rates in the vicinity of converted areas compared to the current situation (1993) and selected scenarios, as well as among the scenarios themselves. Among the scenarios within the development frameworks, as already mentioned the changes are limited to the different gravel scenarios, whereas when comparing the scenarios of the two development frameworks REALIST and EXPANSION, the further increase in surfaced land and lower groundwater recharge are plain.

4.3 The range of fluctuation of the groundwater recharge rate

Owing to the lack of validation possibilities, using water balance models at the mesoscale level necessitates examining the range of fluctuation of the findings. Sensitivity analyses are also required. During the case study, the range of fluctuation of the results was studied using the sensitivity analysis presented by Kunze (1998). The most influential factors regarding natural groundwater recharge were taken into account (cf. Tab. 5): usable field capacity (nFK), climatic data of potential evapotranspiration (ET_P), precipitation (P) and yield class (ERT). In detail, the following assumptions were made for the sensitivity analysis:

- The usable field capacity is assigned to soil types as described by Glugla/Fürtig (1997). Since soil types are aggregated in this approach, a certain range of fluctuation is already included. A range of fluctuation of ±2 vol. % was assumed.
- The climatic data of potential evapotranspiration and precipitation have an error of ±7% (Wendland/Kunkel 1997). This results from extrapolation between the individual measuring stations into the rest of the area as well as raster assignment.
- The range of fluctuation of the yield class (ERT) was determined from the mean land yields over various years in the study area. As a result, mean yield classes and ranges of fluctuation of ±6% were derived (cf. Gluga/Fürtig 1997).

The upper and lower limit of the range of fluctuation for the individual factors can be deduced from these assumptions:

- Upper limit (MAX): nFK –2 vol. %ET_P –7% P +7% ERT –6%
- Lower limit (MIN): nFK +2 vol. % ET_P +7% P –7% ERT +6%

To calculate the range of fluctuation, the values of the individual factors in the database were adjusted accordingly. The changes to the individual factors resulted in eight new calculations. The results indicate that when the assumed maximum range of fluctuation occurs, precipitation has the greatest impact on the final result, whereas the maximum variation of the yield class has the least effect on the results (Tab. 5).

Tab. 5: Range of fluctuation of the results of natural groundwater recharge using the example of Scenario R_1.[2]

Factor	**Change**	**MIN Lower limit [mm/a]**	**Change**	**MAX Upper limit [mm/a]**
ERT	+6%	125.2	–6%	134.6
ET_P	+7%	114.5	–7%	147.3
nFK	+2 vol. %	117.2	–2 vol. %	142.1
P	–7%	105.4	+7%	156.3

ERT – Yield class

ET_P – Potential evapotranspiration

nFK – Usable field capacity

P – Precipitation

Additional calculations assuming the simultaneous occurrence of all minimum or maximum variations in all factors produced a lower limit of 82.1 mm/a and an upper limit of 198.4 mm/a. However, these are absolute maxima for a set of variations which are most unlikely to occur together. Based on their investigations, Petry et al. (2000) quote a range of tolerance of 20–25 mm/aa when using the ABIMO model. The above findings lie within this range, but are based on the consideration of individual factors. The calculations of the most likely value and the uncertainty of the findings resulted in an average tolerance range (taking into account all factors) of about 25% for natural groundwater recharge modelling (cf. Drechsler 2001). Corresponding estimates of uncertainty are carried out for all sub-projects and then taken into account within multicriteria analysis.

Compared to other groundwater recharge studies in central Germany, these investigations – taking into account the uncertainties accompanying the scale used – largely coincide. According to the N-A-U-maps, the mean groundwater recharge in the study area is about 110–130 mm/a (in heathland) and 80 mm/a (in the Elbe Valley) (cf. Institut für Wasserwirtschaft 1959). The results of our own calculations are on the whole somewhat higher. The orders of magnitude are certainly similar; however, the studies were carried out some time ago and

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[2] The mean groundwater recharge rate is 129.9 mm/a, and is included in multicriteria analysis as the most likely value.

the results were shown using a coarser raster (cf. Kunze 1998). The results tally well with investigations in areas with similar natural conditions. Modelling for the adjacent district of Dessau in Saxony-Anhalt produced similar results (cf. Volk/Bannholzer 1999; Petry et al. 2000). These studies were also carried out using the ABIMO run-off model, albeit with the use of different sets of data.

In their more detailed investigations of part of Dübener Heide, Feldhaus/Wilcynski (1997) arrived at similar orders of magnitude to our own results. Using the WHAT model, which also takes seasonal fluctuations into account, they calculated a natural groundwater recharge rate of 143–166 mm/a for the sandy areas with an annual precipitation of about 700 mm. The values we calculated are somewhat lower because of differences between the nature of the two models, as well as the different precipitation level, which at 550–610 mm/a is lower than that used by Feldhaus/Wilcynski (1997). The natural groundwater recharge rates determined by Wendland et al. (1993) for the entire Federal Republic of Germany contain values for heathland whose order of magnitude is also in line with our own calculations. Then again, in the Elbe valley they arrived at values of around 50 mm/a, which are difficult to comprehend.

All in all, the validation of mesoscale water balance studies is beset by numerous problems. This is due to the lack of blanket measurements for large areas and in this case the lack of comparable modelling with corresponding sets of data in central Germany.

To sum up, it can be stated that the methodology chosen enables the basic parameters of the water balance of large areas to be determined and sensitive areas to be highlighted. The results remain inside a range of tolerance which makes the natural groundwater recharge rates appear suitable for further usage to calculate the mean nitrate concentration in leachate.

4.4 Results of the assessment criterion ‘natural groundwater recharge minus groundwater extraction’

The principles of sustainable groundwater protection are that the amount of water extracted from a water resource should not overstrain the water balance and that no ecological damage should occur (Claussen et al. 1996, 29).

Therefore, the difference between natural groundwater recharge and groundwater extraction was chosen as a practical indicator to evaluate land-use scenarios (cf. the article by Klauer et al. in Chapter 2.1). The results of the forecasts on the development of groundwater extraction are shown in Table 6. They are based on empirical analyses in the 1990s and on assumptions for the year 2030. The assumptions regarding long-distance water extraction were based on the forecast extraction rates contained in the development frameworks (cf. Messner et al. in Chapter 2.1). Forecasting was carried out on the basis of a more realistic variant in line with the REALISTIC development framework and an optimistic variant in accordance with the EXPANSION framework. Plausible assumptions were chosen for the other extractions without distinguishing between the development frameworks.

Tab. 6: Groundwater extraction in the Torgau district (statistical surveys for 1993 and 1999 as well as assumptions for 2030)

	Groundwater extraction [million m³/a]			
	1993	**1999**	**2030**	
			REALIST	EXPANSION
Mockritz/Elsnig long-distance water extraction[a]	6.58	2.39	8.2	13.9
East Torgau long-distance water extraction[a]	8.08	6.70		
Regional waterworks[b]	0.54	0.68	0.7	0.7
Commercial water extraction	0.10	0.63	1	1.0
Non-central drinking water supply using individual wells at each building	0.13	0.09	0	0
Non-central service water extraction using individual wells at each building (e.g. for watering gardens)	0.22	0.22	0.2	0.2
Total	**15.7**	**10.71**	**10.1**	**15.8**

[a]*Only 34% of the extraction quantities was assumed to comprise groundwater extraction. The remainder is bank filtrate (cf. HGN et al. 1996, annex 8.2).*
[b]*Mehderitzsch, Neussen and Schildau waterworks*

Source: Statistical surveys for 1993 and 1999 as well as assumptions for 2030 (Volk/Geyler 2001).

Table 7 shows the differences between natural groundwater recharge and groundwater extraction for the entire study area for the scenarios of the two development frameworks REALIST and EXPANSION. The differences shown in this table underline that the differences between the alternative actions are extremely small.

All in all, the model results show that only about 10–20% of the newly formed groundwater would be used as a resource – hence complying with the principle of sustainability under which groundwater recharge exceeds extraction. Nevertheless, the differences in certain areas between groundwater recharge and extraction should not be neglected. Groundwater extraction by the long-distance waterworks mainly takes place in the Elbe floodplain, where the groundwater recharge rates are relatively low (cf. Map 10 in the appendix). The contributing areas hence tend to be found in the outer sections of the waterworks' catchment areas. Especially in the case of Mockritz waterworks, which owing to the inland position of Well Field I extracts a large proportion of groundwater, important contributing areas are located in the outer drinking water protection zone (Zone 3b west of the Elbe), or assuming its abolition outside the protection zones.

Tab. 7: Criterion values of groundwater recharge (GWR) minus groundwater extraction (GWE) for the Torgau district

	1993	**2030**					
		R_1	**R_2**	**R_3 & R_4**	**G_1**	**G_2**	**G_3 & G_4**
GWR [million m³/a]	91.2	89.1	89.4	89.1	88.9	88.9	89.0
GWE [million m³/a]	15.6	10.1	10.1	10.1	15.8	15.8	15.8
GWR-GWE [million m³/a]	**75.6**	**79.0**	**79.3**	**79.0**	**73.1**	**73.1**	**73.2**

Furthermore, when assessing the difference between natural groundwater recharge and groundwater extraction, the groundwater needed for ecological functions – especially small surface waters – must also be considered. Including such effects means that additional analyses for small individual areas are required, as well as a study of the influence of the level of the Elbe on the groundwater balance in the Elbe floodplain, the re-infiltration of surface water from rivers and streams entering the groundwater (e.g. at Weinske/Schwarzer Graben), and also the influences of agricultural drainage systems. However, no such investigations were conducted.

5. Summary and outlook

The results enable regional differentiation of the natural groundwater recharge taking into account the prevailing natural conditions and land-use types. The influence of land-use changes on groundwater recharge can be simulated for the land-use scenarios and – in connection with other information levels – both qualitative and quantitative hazard potentials can be pinpointed. The changes to the mean groundwater recharge rate for the entire area remain within a similar order of magnitude to comparable studies (cf. Tab. 1). They appear relatively low, although significant differences may occur locally. In this connection, it should be pointed out that the effects caused by simulated land-use changes (scenarios) on the natural groundwater recharge closely depend on the selection of the conversion areas and their natural conditions. In a nutshell, although the influence of the simulated land-use changes on groundwater recharge throughout the entire district can be classified as minor owing to compensation effects, pronounced differences certainly occur locally.

When evaluating the findings, it should be noted that long-term means were used which should be regarded as ‘most likely values’. Although the calculation results are expressed in absolute figures, since we are dealing with a model and given the low spatiotemporal resolution of the input data, they can only indicate orders of magnitude (Volk/Bannholzer 1999). More detailed investigations at greater scales would entail using different model systems and sets of data with a higher spatiotemporal resolution. Future investigations must increasingly concentrate on optimizing the application of water balance models at different scales (defining their predictive accuracy, comparing the calculation algorithms of different models). One step in this direction was taken in this article by determining the ranges of fluctuation of the results.

Mesoscale calculations designed to predict the effects of land-use changes on the water resources in a landscape are always hypothetical for the reasons listed above, as well as because of the long forecasting period. Nevertheless, the spatially related influences and their impact on the regional and local water balance can be roughly shown. This provides planning authorities with a decision-support tool which can be used to avoid negative consequences for the water balance. All in all, the groundwater recharge rates calculated can be regarded as suitable for further usage in calculating the mean nitrate concentrations in leachate. The assessment criterion 'groundwater recharge minus groundwater extraction' is especially significant as an indicator of the sustainability of the land-use development in question from the angle of water resources. However, given the low differences, this criterion is irrelevant for assessing action options within the framework of multicriteria analysis.

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The following publication is an English translation of the original Book Chapter.

Modelling how land-use changes affect the nitrate concentration in leachate

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1. Problem

In recent decades, the increasing intensification of agricultural production has led to more and more environmental resources being consumed. Nitrogen (N), one of the main nutrients of plants, is one of the most important factors of intensification. Since agricultural production is closely related to the weather, exactly planning nutrient usage to make sure they are completely used up by the crops is practically impossible. The surplus nitrogen can usually only be briefly stored in the ground, resulting in nitrogen entering the atmosphere and the leaching of nitrate (NO_3) on a scale which accelerates with the degree of intensification. However, nitrogen is also output by natural and semi-natural ecosystems. In a state of equilibrium, N outputs exactly match the various N inputs from the atmosphere, which total around 60 kg/ha annually (Isermann 1990; Russow/Weigel 2000). Agricultural land has a positive effect on the landscape-related nitrogen balance if the output into the atmosphere and the groundwater is considerably lower than the input from various sources.

Simulation models have increasingly been used in recent years to study and evaluate the water and nitrogen balances. These can be used as a basis to determine land usage variants which, employing the regional regulation potential, lead to nutrient outputs into neighbouring ecosystems being reduced (Franko et al. 1997; Volk/Bannholzer 1999).

The findings presented here covering the Torgau district were achieved using the CANDY simulation system[1] for N leaching beneath farmland. Land-use scenarios were worked out for various economic development frameworks (cf. Messner et al., Chapter 2.1) in order to study their impact on groundwater quality. Data concerning N leaching beneath forest and grassland were taken from the literature.

2. *Determining the nitrate concentration in leachate*

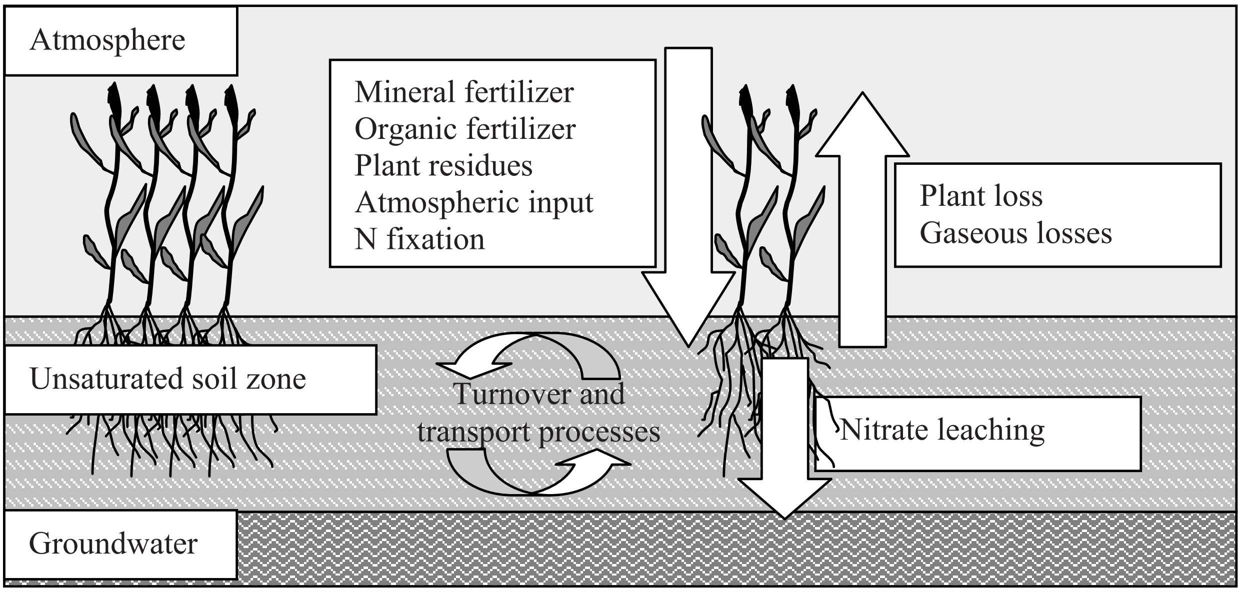
A regional assessment of leachate quality produced on the basis of nitrate leaching was required. Carrying out this assessment with the help of simulation calculations calls for complete input data (in this case farming and weather data) for the entire areas gathered over a very long period (i.e. on the scale of decades). The necessary simulation duration depends on the buffer effect of the soil and the initial values of the simulation. However, obtaining farming data featuring exact temporal and quantitative details is not feasible owing to the sheer size of the area concerned. Therefore, a way of producing model inputs on the basis of relatively imprecise input data had to be developed.

This section delivers the 'nitrate concentration in leachate' indicator for all land-use scenarios, which can be used as an input parameter for multicriteria assessment (cf. articles in Chapter 2.3). The derivation of scenarios is explained in the article by Messner et al. in Chapter 2.1. Alongside the most likely values, the uncertainties when collecting and aggregating these data also need to be determined to highlight the distribution of the possible range of values.

Fig. 1 shows the nitrogen flows in and around the unsaturated soil zone. In order to carry out a regional assessment of land use, this one-dimensional view has to be transferred to the Torgau district by each landscape unit being assessed, weighted in terms of area, and then being incorporated into the overall balance.

[1] CANDY is an acronym for Carbon and Nitrogen Dynamics. See 3.1 for more on the CANDY simulation system.

Fig. 1: Nitrogen cycle in the soil
(source: own illustration)



2.1 Spatial and temporal system limits

Since only the stationary values were relevant for the information required, all simulation objects were initialized under the standard assumption of 'average conditions'. Current (reference year 1993) and future (reference year 2030) arable farming was characterized by crop rotation over a number of years which was repeated in the simulation until the nitrogen output had become relatively constant (which in no case took more than 50 simulation years). To ensure comparability, the first 50 years were discarded in all calculations and a mean was calculated for the next 50 years in order to compensate for the influence of temporary weather conditions. This resulted in a quasi-stationary state for the reference years 1993 and 2003.

These long-term simulations were conducted using a weather generator parameterized on the basis of a 15-year measuring series recorded at Oschatz weather station. The regional rainfall distribution was reproduced by making locally specific changes to the intensity of a rainfall event based on the ratios between the total rainfall over several years in the various areas and at the location of the weather station.

The spatial disaggregation of the landscape studied was carried out by designating 'soil objects' by superimposing the soil map (Kunze 1998) with the classification into natural areas (cf. Kindler et al. and Map 6 in the appendix). Parallel to this, farming types specific to natural areas were compiled for various levels of intensity which matched the average statistical figures for cultivation conditions, livestock and yields as far as possible (cf. Schmidt et al. 2001). Mineral nitrogen fertilization was determined using the fertilizer advisory system BEFU,[2] which is commonly used in the study area.

The simulation objects themselves are produced by combining soil objects with farming types. Each simulation object is characterized by mean annual N-leaching and an average leachate formation rate for the stationary state. These individual values result in scenario-related spatial aggregation using differentiated regional weighting of the individual intensity types.

2.2 Nitrogen leaching beneath forest, grassland and arable farmland

From a spatial viewpoint, a complex landscape can be divided into a number of separate areas, each of which is homogeneous in terms of soil characteristics, weather and land use. In this case, land use was initially specified as farmland, arable grassland or forest. Urban areas and stretches of water were not taken into account. As the intensity of arable farming in particular varies, the studies were concentrated in the form of CANDY simulations on this sector, while the information for grassland and forestland is based on figures taken from the literature.

2.2.1 Forest

Nitrogen leaching from forestry areas is subject to high variation. Research into forest ecosystems reports N outputs to range between 1 and 62 N ha^{-1} a^{-1} (DVWK 1990; UBA 1994). This rate is largely determined by atmospheric N

[2] BEFU stands for BEstandesFUehrung ("Stock Management") and is a computer program provided by the Saxon Institute of Agriculture to calculate the amount of fertilizer needed.

inputs and the ecosystem's N fixation capacity resulting from immobilization caused by microorganisms and by incorporation into the biomass growth of vegetation (UBA 1994, 91). However, since the accumulation of nitrogen in the soil matrix is limited, following the completion of humus development the nitrogen largely bypasses the soil and is directly discharged into the groundwater (Köllig/Neustifter 1997).

According to measurements by Ehrhard (1999), the N outputs at two areas in heathland Dölauer Heide are currently about 3–4 kg N ha^{-1} a^{-1}. The nitrogen outputs for the base year 1993 are correspondingly assumed to be 5 kg N ha^{-1} a^{-1}; no distinction was drawn between coniferous and deciduous forest. An output rate of 30 kg N $ha^{-1}a^{-1}$ is postulated for the 2030 scenarios; this figure was deduced by assuming that by that time the buffer capacity of the forest soil will be exhausted, after which 50% of the estimated total atmospheric input (60 kg N $ha^{-1}a^{-1}$) will reach the groundwater (Wendland et al. 1993).

2.2.2 Grassland

Grassland contains a high-level of humus and compared to arable farmland is able to assimilate and utilize much more nitrogen (Rieder 1983). Walter et al. (1985) calculated average leaching losses of about 7 kg N ha^{-1} a^{-1} for various levels of fertilizer. In addition to the total amount of nitrogen applied, the degree of N leaching also depends on whether the grassland is used as meadowland or pasture. Isolated inputs dotted over pasture usage cause higher N losses (Benke 1992).

Assuming that mostly intensive indoor stock keeping is practised in the Torgau district and grassland is mainly used as hay meadow, nitrogen leaching beneath intensively used grassland was set at 10 kg N ha^{-1} a^{-1} and beneath extensively used land at 5 kg N ha^{-1} a^{-1}. No differentiation was made between heathland and Elbe floodplain areas.

Under the Saxon Protection and Compensation Directive (SächsSchAVO; SMU 1994), it can be assumed that farming on grassland in wellhead protection areas is very similar to extensive agriculture (output: 5 kg N ha^{-1} a^{-1}).

Both extensive and intensive agriculture is assumed for grassland outside wellhead protection areas. The proportion of extensive agriculture is judged on the basis of the figures cited in Part 1 of the Saxon Ministry of the Environment and Agriculture's KULAP programme for cultural landscapes. For the year 1993, it was deemed on the basis of statistical surveys that 25% of grassland outside wellhead protection areas in the Torgau area is farmed extensively. In line with the increasingly ecological direction taken by farming, the proportion of land used for extensive agriculture is estimated to rise to 60% by 2030 (cf. Herzog et al. 2001). The results are extrapolated for the whole region by calculating the mean values weighted by area of the intensity classes, similar to the procedure described for arable land in 2.2.3.

2.2.3 Arable land

Nitrogen plays an especially important role in arable land for controlling plant production. Compared to meadowland, pasture and forest, in arable land the self-regulating potential is used least of all. As a result, if poorly adapted to local conditions, farming systems cause high environmental pollution. The variety of ways of controlling nitrogen and the complex interrelations between the individual elements of an arable farming system call for a simulation system to investigate the water and solute balance. Nitrogen leaching beneath soil used for arable farming was calculated here using the CANDY simulation system. The methods and results are presented below.

3. Simulation of nitrate leaching beneath arable land

Any attempt to describe the solute balance of a landscape requires the spatial and temporal limitation of the findings. In this case it is apparent that exact dates are unimportant; what is needed are the general characteristics of the system's behaviour. They can best be assessed by consideration in the stationary state once the initial conditions no longer exert a disruptive influence.

3.1 The CANDY simulation system

The CANDY model simulates the dynamics of carbon and nitrogen in the unsaturated zone of agricultural soils (Franko 1996). Calculation is usually performed for sites down to a depth of 2 m, although greater depths can also be managed by the model. The soil profile is divided for calculation purposes into homogeneous layers each 10 cm thick. Daily steps are used by way of temporal resolution. The following sub-processes are described by the model in varying detail:

- Meteorological conditions (access to databases or generation of data records, correction of precipitation measured)
- Soil water dynamics (potential and actual evapotranspiration, seepage)
- Soil temperature dynamics
- Metabolization (mineralization and humification) of organic substance
- Nitrogen dynamics (mineralization, immobilization, uptake, leaching, gaseous losses, symbiotic N binding)
- Pesticide dynamics (not used during this study)

Starting with initial values for the parameters considered (soil temperature, soil moisture, metabolizable organic substance and mineral nitrogen), the system processes management information on soil cultivation, fertilization, etc. The system consists of a simulation model integrated into a user interface along with supplementary modules and related databases containing information on the necessary parameters, the driving variables as well as initial values and any supplementary series of measurements.

Previous trials of the model produced good results for a series of locations with different soil qualities. Assuming good model input, the soil water content can be estimated to an accuracy of about 2 vol-% and the supply of mineral nitrogen to ± 20 kg/ha (Franko et al. 1995).

Carrying out the simulation tasks requires a corresponding spectrum of data specifying quality characteristics and supplementary conditions. These fixed parameters include all the data describing the mineral and organic fertilizer used and specifying the crops to be raised.

The soil characteristics are organized in the CANDY system hierarchically. Following medium-scale site mapping, regional types can be defined as a unit spread across a number of soil profiles. Each soil profile is described as a sequence of different horizons. The soil parameters required – dry raw density, dry substance density, proportion of fines, field capacity, permanent wilting point, C_{org} level, and a leachate parameter derived from the saturated conductivity or clay content – are assigned to the individual horizons. The required parameters are available for a wide range of soil profiles.

One of the key aspects driving the model is the following meteorological data: air temperature at a height of 2 m, solar and sky radiation, and rainfall. Mean daily values for the weather data are required for every single day of the simulation period. Experience shows that temperature and radiation data can also be taken from weather stations in lowlands some way away. However, the rainfall data must definitely be recorded locally. For long-term simulations, a weather generator is parameterized from the existing climate data.

A simulation object is an area element which in terms of soil, weather and farming can be regarded as homogeneous. Various elements (usually parcels of land) can be aggregated to form a logical unit in a database. The database includes fixed data, farming data (the measures taken), measurements and status data (potential starting data).

The simulation object is defined by its fixed data, which include the following information:

1. Parcel identification.
2. Soil profile reference.
3. Reference to the weather station, including long-term mean rainfall and air temperature as well as latitude.
4. Prior situation:
   - Annual supply of carbon relevant for reproduction: this information can be calculated from details of the cultivation ratio, yield and organic fertilizer used.

  - Level of N supply: estimating the level serves to adapt the initial values for the nitrate supply in the soil profile. More precise details are possible by incorporating measured values.

5. Initial simulation details:

   - Starting date (weather and farming data must be available from this point onwards).
   - Proportion of usable field capacity used at this date.
   - Annual atmospheric nitrogen input in kg/ha.

When dealing with the fixed data, a data record containing the corresponding initial values is created in the status catalogue of the database. The current status is entered in the catalogue at the start of each new year and at the end of each simulation. This enables the simulation to be continued at a later date.

All relevant farming measures, especially the quantity of solutes added and removed, must be recorded every day. Data maintenance is analogous to managing a parcel cadastre. Relevant farming events include:

- Sowing, shooting and harvest
- Mineral N fertilization
- Organic fertilization
- Soil cultivation
- Irrigation
- Usage of pesticides

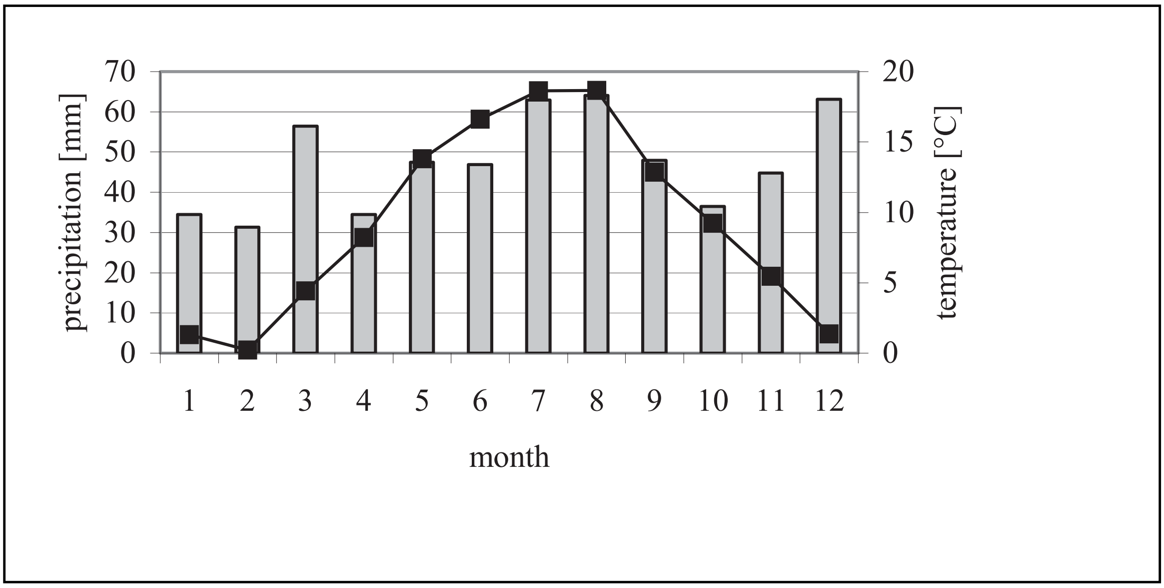
3.2 Using CANDY for N leaching beneath arable farmland

3.2.1 Data for the Torgau district

3.2.1.1 Climate

The climate data were derived from measurements taken at the climate station run by the German Weather Service in Oschatz. Values are available for a long-term series of measurements (1984–97) adapted to conditions in the Torgau district using a correction factor derived to reflect the relative precipitation ratio between Oschatz and the various land categories in the project area. The CANDY system's weather generator uses this basis to calculate random weather for the simulation period required. The course of precipitation and temperature is generated anew for each simulation year and varies in accordance with the initial data. Fig. 2 shows the mean monthly precipitation totals (533 mm) and temperature values (8.9°C) in the Elbe value.

Fig. 2: Climate diagram (columns: precipitation; line plot: air temperature)



3.2.1.2 Soil

According to the medium-scale site mapping instructions (*Mittelmaßstäbige Kartieranleitung*/MMK), the Torgau district contains 32 different types of soil. For the CANDY simulation calculations, the nine most frequent types of soil for the Elbe floodplains and heathland representing 99.5% of the area were chosen.

The soil parameters were specified on the basis of the MMK and descriptions given by Kundler (1989) as well as our own soil analyses and laboratory studies. Table 1 lists the soil types selected with their horizon structure and thickness.

Table 1: Soil types

Soil type	Depth [dm]	Horizon	Depth [dm]	Horizon	Depth [dm]	Horizon	Depth [dm]	Horizon
Pseudo-gley/para-bornw earth complex	0–3	AP	3–5	SW–AL	5–10	SD–BT	10–20	C
Gley/pseudo-gley complex	0–3	AP	3–4	SW	4–10	GO–SD		
Brown earth/para-brown earth	0–3	AP	3–6	BV	6–10	BVT	10–20	C
Pseudo-gley	0–3	AP	3–4	SW	4–20	SD		
Brown earth/podzol	0–3	AP	3–6	BSV	6–20	C		
Meadow loam/vega-gley	0–3	AP	3–6	MA	6–10	GO		
Meadow loam/vega	0–3	AP	3–8	MA	8–12	MG		
Sand/brown earth	0–3	AP	3–5	BV	5–20	C		
Loess/planosol [Staugley]	0–3	AP	3–5	ETG	5–12	BTG		

3.2.2 Characteristics of arable land use

To characterize the agricultural activities, the region was divided into four sections of fundamentally different arable usage. Classification was first carried out in the natural areas of the Elbe floodplain and heathland. The Elbe floodplain features high-yield soils, the main soil type being vega. In the heathland, light soils predominate such as sand/brown earth, which accounts for 60% of the entire arable farmland. Further fundamental differentiation was carried out with respect to wellhead protection areas, resulting in a total of four sub-regions: Elbe floodplains inside and outside the wellhead protection area, and heathland inside and outside the wellhead protection area. These units are shown in Map 6 in the appendix.

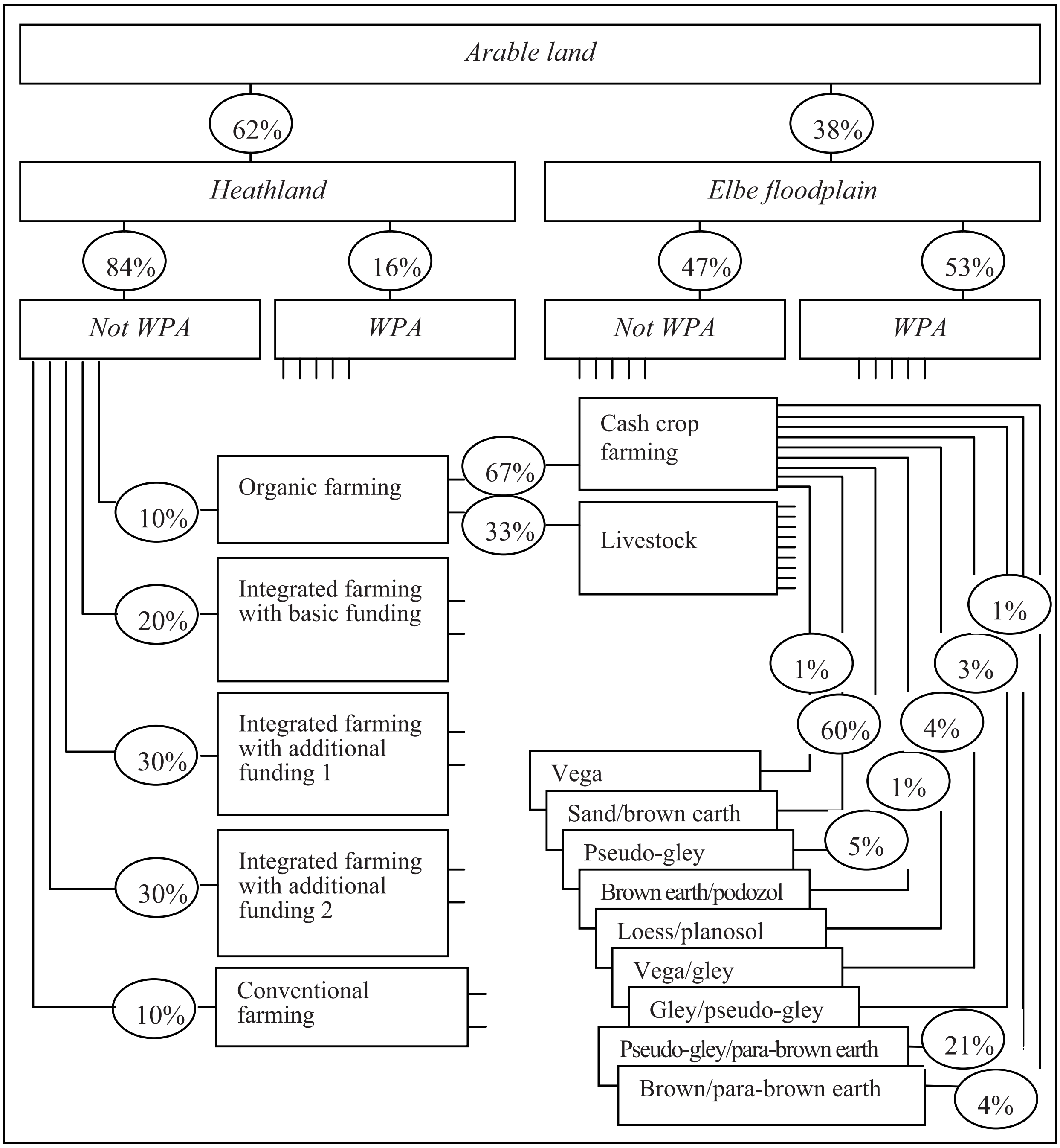
Within these units, 10 typical management forms were defined for the CANDY simulation system comprising market crop farming and livestock farming for ecological, integrated (three-stage) and conventional agriculture. Details of the crop rotation sequence and fertilizer usage are contained in Schmidt et al. (2001).

Statistics for 1997 for the administrative district of Torgau-Oschatz were used as basic data for the cultivation ratio and livestock. Yield figures were taken from the data collection maintained by the Saxon Institute of Agriculture (1999). The levels of nitrogen added correspond to those recommended by the BEFU fertilization advisory program for Saxon. Standardized input data on soil type, mineral nitrogen in the soil and development were used to calculate fertilizer usage. Market crop farms were weighted with a factor of 2 compared to livestock farms in the assessment algorithm to reflect their ratio in the region.

A total of 360 simulation objects can be created by combining all the factors of influence. Fig. 3 explains the principal structure of disaggregation using the example of an organic market crop farm located on the heathland outside the wellhead protection area.

Some 62% of the arable farmland in the Torgau district is located in the heathland, 84% of which is not part of the wellhead protection area. Ten per cent of this category is used for organic farming, of which 67% is for cash crops with no livestock (realistic scenario 2030). These farms use arable land on nine different soil types broken down by percentage. The resolution of all the objects is contained in Schmidt et al. (2001). The assumptions on the development of arable land use are described by Messner et al. in Chapter 2.1 (cf. also Herzog et al. 2001).

Fig. 3: Structure of arable activities in the Torgau district with weighting factors (WPA = wellhead protection area)

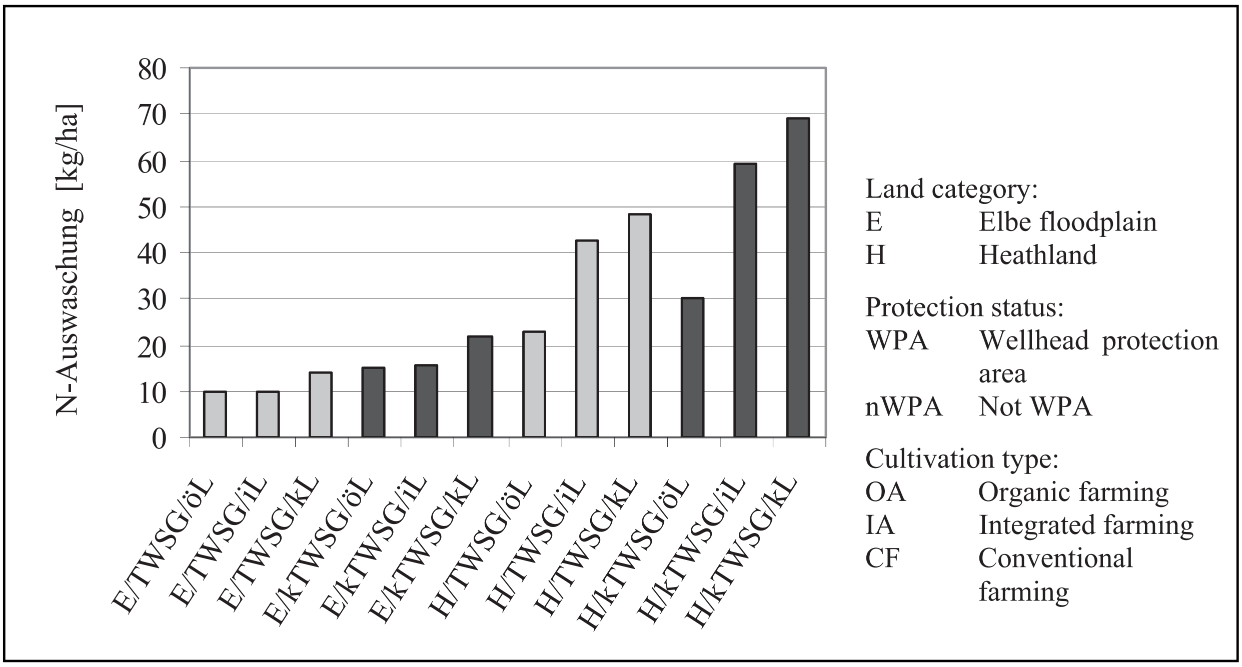


4. Simulation results

4.1 Nitrogen output and nitrate concentration

Nitrogen leaching in the groundwater largely depends on the N balances.[3] Positive N balances are potential nitrogen losses which escape into the atmosphere via microbial activities or enter the groundwater dissolved in the flow of leachate. Fig. 4 shows the N leaching rates aggregated from all the individual objects as a three-stage unit comprising land category, protection area and farming type.

Fig. 4: Nitrogen leaching beneath arable land



The leaching losses in the Elbe floodplain are between 9 and 24 kg $ha^{-1}a^{-1}$, which is relatively low compared to the heathland areas (23–69 kg ha^{-1} a^{-1}). This difference is attributable to the different soil types, with the clayey soil in the floodplains contrasting with the predominantly sandy locations in the heathland. The restrictions in the wellhead protection area have a considerable impact on

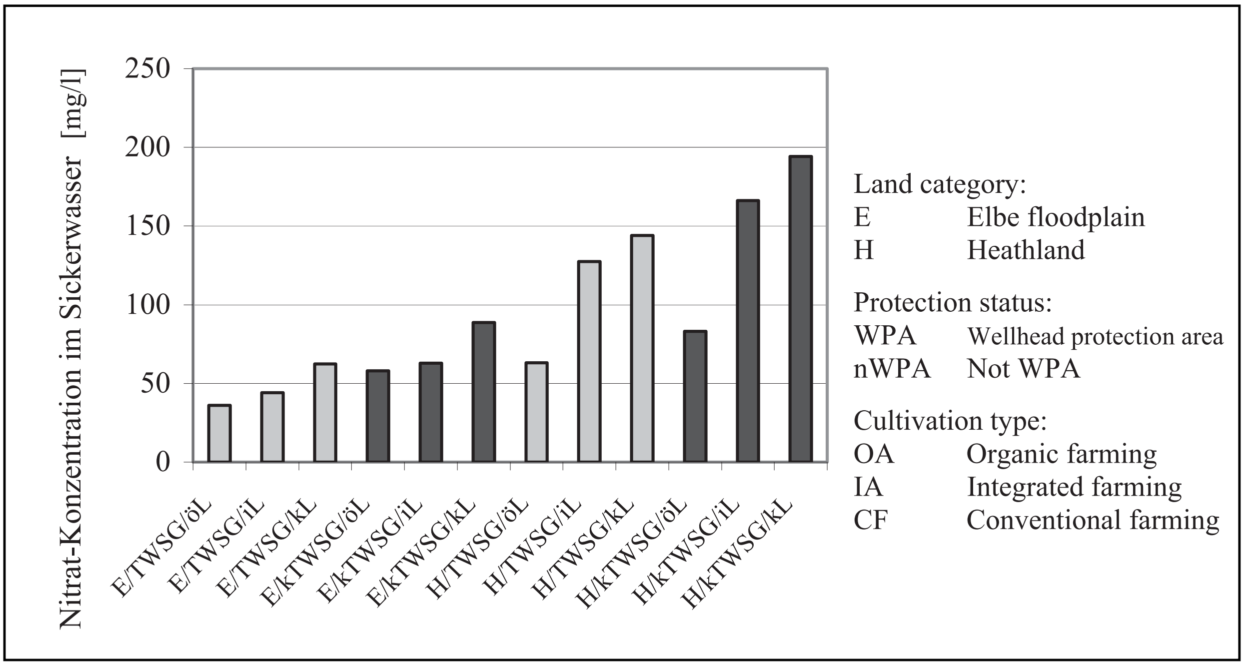
[3] The N balance is calculated from N input less N loss via crops.

both integrated and conventional agriculture with a reduction of N leaching of about 30%.

Organic farming is only affected by this difference with respect to cover crop cultivation. As a result, only low differences emerge within an area unit. The nitrate concentration in the leachate (Fig. 5) is calculated from the natural groundwater recharge rate (cf. Volk et al. in Chapter 2.2) and the N leachate using Eq. 1:

$$NO_3 \text{ concentration [mg/l]} = 443 * \text{N leaching [kg/ha]} / \text{GRR [mm]} \qquad (1)$$

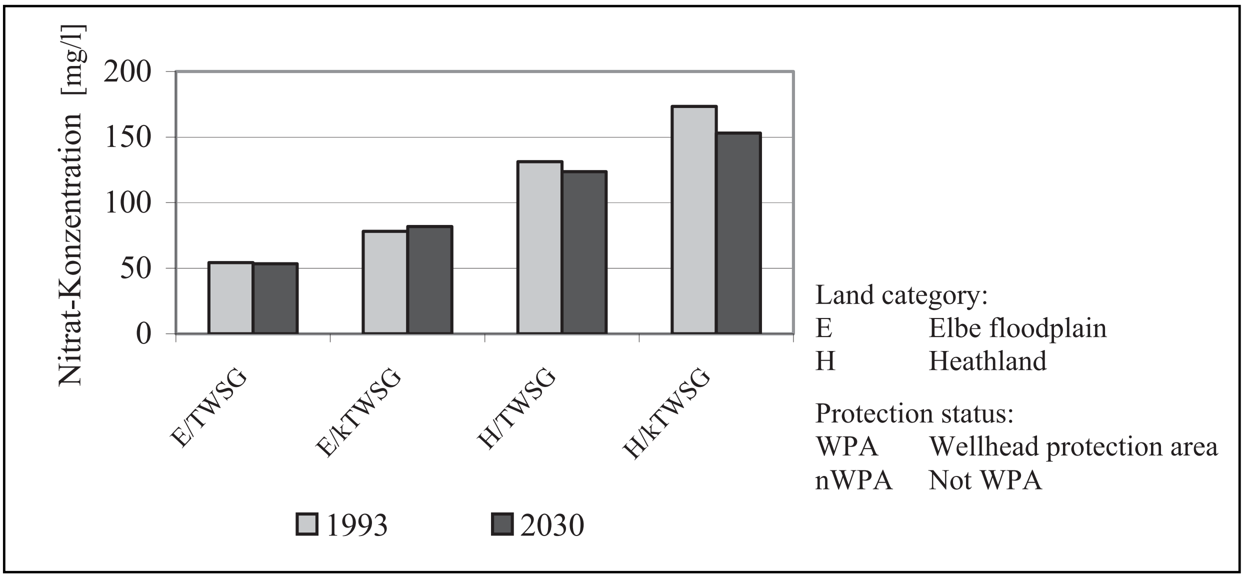
Fig. 5: Nitrate concentration in leachate beneath arable land



Of the 12 farming types defined, 10 cause the maximum nitrate limit of 50 mg/l stipulated by the Drinking Water Directive to be exceeded. Weighted concentrations for 1993 and 2030 can be derived by way of example for Scenario REALISTIC R_1 using the percentage of land accounted for by the various farming types. The aggregated data are shown in Fig 6.

In the four sub-areas, the nitrate concentrations change in the case of Scenario R_1 depending on the weighting of the individual arable farming systems. In the wellhead protection area of the Elbe floodplain, the current cultivation structure is assumed to be maintained in future years, causing the N output to also remain unchanged. The arable soil in the Elbe floodplains outside wellhead protection areas will in future be farmed more intensively owing to the high potential yield and will be subject to correspondingly higher N output rates. The extensification trend forecast is clearly apparent in the heathland areas, where the NO_3 concentration in leachate is set to improve from 6 to 22 mg/l.

Fig. 6: Nitrate concentration in leachate beneath arable land in 1993 and 2030 for Scenario REALISTIC *R_1 (by sub-areas)*



4.2 Regionalized nitrate concentration in leachate

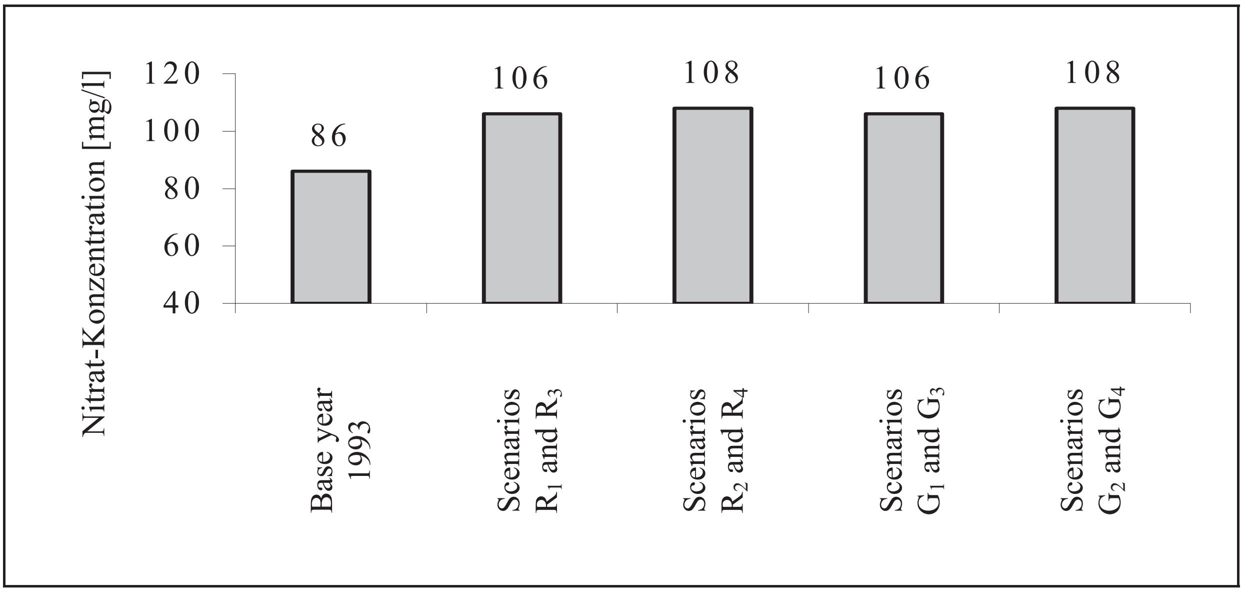
4.2.1 Aggregation and regionalization of the findings for the Torgau district

Fig. 7 shows the most likely area-weighted values for nitrate concentration in the Torgau district. The REALISTIC Scenarios R_1 to R_4 relevant for multicriteria analysis and the BOOM scenarios G_1 to G_4 are compared to the base year 1993.

The mean nitrate concentration in leachate in 1993 (*status quo*) is around 86 mg/l. The scenario values for 2030 range between 106 and 108 mg/l owing to the higher leaching rates beneath the forest. The differences in 2030 are explained by altered protection area designation in the individual scenarios.

Partial abolition of the wellhead protection area in Mockritz (Zone 3b) will slightly increase the average nitrate concentration in the leachate by 2 mg/l for the entire Torgau district. Map 12 in the appendix shows the regionalized nitrate concentration in leachate in 1993, while Map 13 shows that in 2020 by way of example for Scenario R_2. Map 14, which shows the change in the nitrate concentration in 2030 for Scenario R_2 compared to 1993 brings home the negative change that is forecast beneath forest areas and the positive development in agricultural areas.

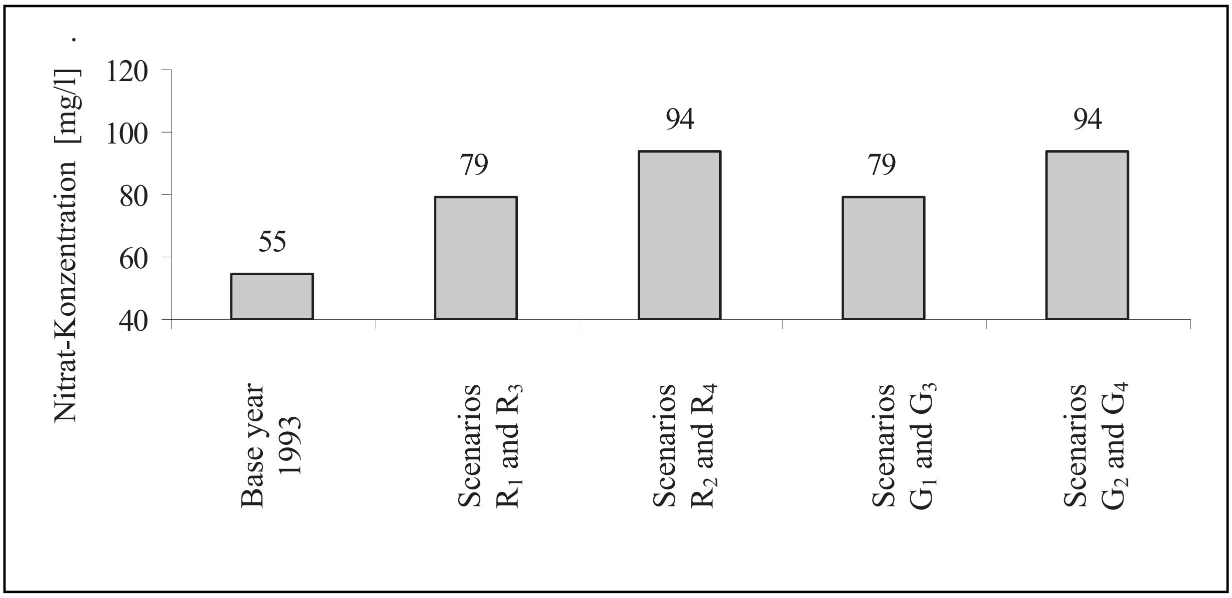
Fig. 7: Nitrate concentration in leachate



4.2.2 Aggregation and regionalization of the results for Mockritz wellhead protection area

The calculated nitrate concentration in Mockritz wellhead protection area is much higher in the various scenarios in 2030 compared to 1993 (Fig. 8). The higher nitrate concentrations in 2030 are – as for the entire Torgau district – explained by the negative development beneath woodland areas, which account for about 30% of the total area. The difference in level of 15 mg/l between complete protection area designation (Scenarios R_1, R_3, G_1 and G_3) and the abolition of Zone 3b (Scenarios R_2, R_4, G_2 and G_4) is clearly apparent.

Fig. 8: Nitrate concentration in the leachate of Mockritz wellhead protection area



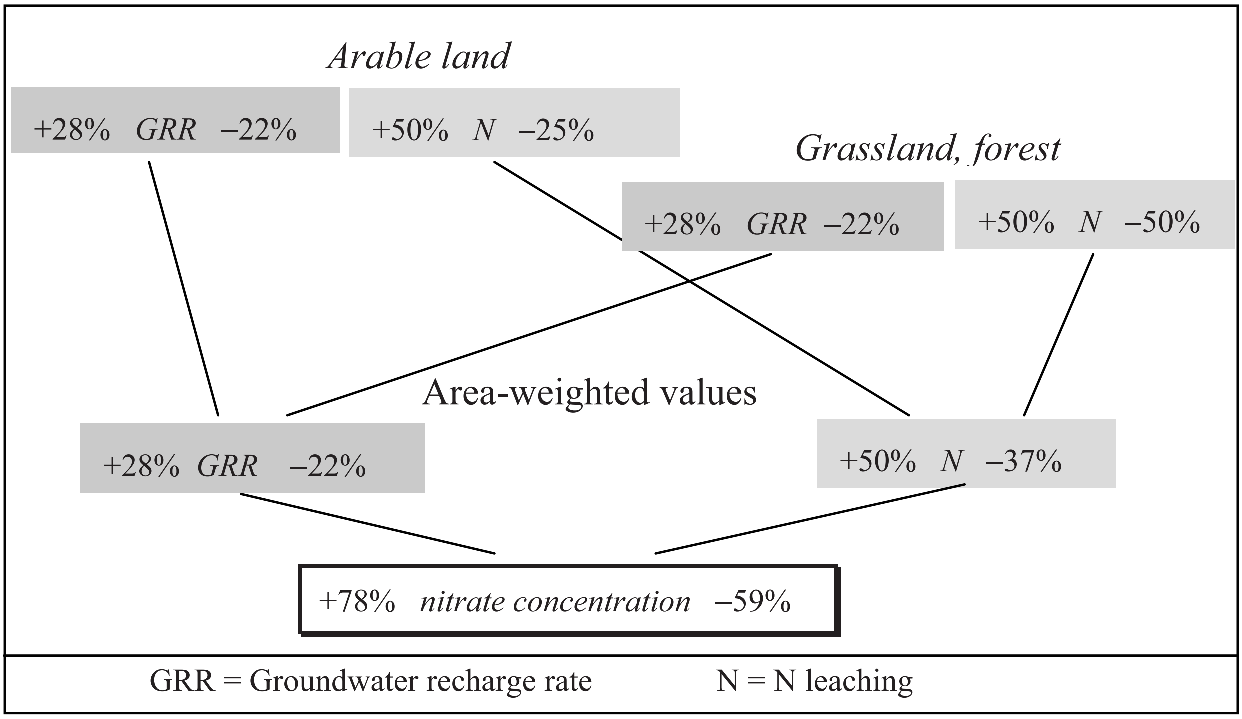
4.3 Sensitivity of the inputs and uncertainty in the results

Uncertain input data partly derived from relatively coarse statistics lead to great uncertainty in the final results. All the independent variables are added together in a single unit (N output and natural groundwater recharge) and then summarized on an area-weighted basis. The uncertainty of the nitrate concentration depends in turn on the area-weighted values of the N output and the natural groundwater recharge (Fig. 9).

The uncertainties when calculating the natural groundwater recharge were taken from the article by Volk et al. (Chapter 2.2). The changes in N leaching beneath arable farmland are influenced by the absolute range of fluctuation of the N input, which largely depends on the calculation of the fertilization requirement and the method of application used. The BEFU fertilization requirement cannot be calculated exactly owing to uncertain input data, while the practical method of its application is also subject to inaccuracy. Regional rainfall fluctuations have an impact on the N output too since the distribution of rainfall affects the speed of vertical nitrogen transport, resulting in the possibility of higher losses. Summing up these uncertainties, a maximum range of fluctuation of 50% is assumed. It can be concluded from statistical data concerning the usage of N (Wendland et al. 1993) that more rather than less fertilizer tends to be applied. This leads to the assumption that the N leaching uncertainties are between +50% and –25%.

A large fluctuation margin exists in the figures quoted in the literature for forest and grassland areas (UBA 1994), and so uncertainty must be estimated at ±50%. The assumptions result in a fluctuation margin of the actual nitrate concentration of the most likely value of +78% to –59%.

Fig. 9: Uncertainty when calculating the nitrate concentration of leachate



4.4 Assessment of the findings on nitrate concentration in leachate

The simulated extensification measures in agriculture simulated in the scenarios lead to better water quality. However, the situation beneath forestland can be expected to drastically deteriorate if the N immission rate remains the same. On balance these assumptions would lead to a slight increase in the nitrate concentration in the Torgau district.

According to the results calculated, the abolition of protection Zone 3b to the east and west of the Elbe along with Zone 3a east of the Elbe in Mockritz wellhead protection area examined in future Scenarios R_1/R_3 and G_1/G_3 will lead to the nitrate concentration rising by 16 mg/l (cf. Fig. 9) and cause the water quality to significantly deteriorate.

5. *Summary and conclusions*

The procedure for modelling the influence of land-use changes on leachate nitrate concentration largely depends on the availability of data and the simulation models which can be used on this basis. The existing stock of data for the Torgau district meant that groundwater recharge was calculated using the ABIMO run-off formation model, while the nitrogen leaching rates beneath farmland were calculated using the CANDY simulation system. The leaching rates from the grassland and forest land-use types were taken from the literature. The regionalized analysis showed that N leaching beneath forests with N-saturated soil is an especially sensitive parameter which, given the higher proportion of woodlands (28.5% of the Torgau district) will in the medium term have a highly negative impact on area-weighted leachate quality. At present, our knowledge of the behaviour of woodland soils is still too limited to provide more accurate information. The results presented here hence describe a forecast trend and are beset by large uncertainty which in future will have to be examined using comparative measurements. By contrast, the agricultural areas were easier to evaluate more accurately. The reliability of the information depends above all on the realistic disaggregation of agricultural statistics.

Comparison of the arable farming systems investigated shows that the range of measures available to promote environmentally sustainable agriculture will have a lastingly positive effect on leachate quality. In the global context, in addition to nitrate concentration in leachate, the total nitrate output is of particular importance, which in the Torgau district is about 35 kg ha^{-1} a^{-1} – far less than the German average of over 100 kg ha^{-1} a^{-1} (Wendland et al. 1993; Kolbe 2000).

One crucial problem when evaluating future land use results from the increasing buffering of anthropogenic N inputs from the atmosphere into forest ecosystems. More research needs to be carried out in order to conclude regional and global strategies to stabilize the buffering capacity of forest soils as well as to reduce N flows into the atmosphere in order to safeguard leachate quality in the long term and to bring the nitrogen cycle into an ecologically sustainable balance.

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Modellierung und Bewertung des Einflusses von Landnutzung und Bewirtschaftungsintensität auf den potenziellen Nitrataustrag in einem mesoskaligen Einzugsgebiet

Modelling and Assessment of the Impact of Land Use and Variants of Cultivation Practices on the Nitrate Leaching in a Mesoscale Watershed

M. Neubert, M. Volk und F. Herzog

Zusammenfassung: In der vorliegenden Studie wird eine Methodik zur Untersuchung des Einflusses von Landnutzung und Bewirtschaftungsintensität auf die Sickerwasserqualität am Beispiel des Torgauer Raumes (Nord-Sachsen) dargestellt. Die Untersuchungen erfolgten unter Nutzung von Modellen zur Berechnung der Grundwasserneubildung und des Stickstoffüberschusses der Landwirtschaft. Der potenzielle Nitrataustrag wurde durch die Kombination der Berechnungsergebnisse in einem Geographischen Informationssystem ermittelt. Es zeigt sich dabei eine deutliche Abhängigkeit der potenziellen Nitratbelastung von den Größen Grundwasserneubildung und Stickstoffbilanz in Beziehung zu den Bewirtschaftungsintensitäten ökologischer, integrierter und konventioneller Landbau.

Schlüsselwörter: Mesoskalige Modellierung, Nitrat, Landnutzung, Grundwasserneubildung, GIS

Summary: The authors present a method for analysing the impact of land use and variants of cultivation practices on the seeping water quality on the example of the Torgau region (North Saxony, Germany). The investigation is based on the application of both a runoff simulation model and a model simulating the nitrate surplus caused by agricultural land use. Potential fluxes of nitrate have been calculated by combining the simulation results by using a Geographic Information System. The results are indicating an explicit dependency of the nitrate balance on the parameters ground water recharge and nitrate balance in relation to the variants of cultivation practice like organic, integrated and conventional farming.

Keywords: Mesoscale modelling, nitrate, land use, ground water recharge, GIS

1 Einleitung

Die landwirtschaftliche Bodennutzung gilt als Hauptverursacher für Nitratbelastungen des Grundwassers. Dies führt aufgrund des erforderlichen Grundwasser- und Bodenschutzes zu Konflikten durch die Überlagerung von Nutzungsinteressen von Landwirtschaft, Wasserwirtschaft und anderen gesellschaftlichen Belangen. Die Lösung solcher Konflikte erfordert interdisziplinäre Bewertungsansätze, um Quantität und Qualität des Grundwassers unter Berücksichtigung der ökonomischen Entwicklung gewährleisten zu können. Die vorliegende Studie leistet einen Beitrag zu einem solchen Projekt, indem Kenngrößen des Landschaftswasser- und -stoffhaushaltes bereitgestellt werden, die in eine umfassende ökologische und sozioökonomische Bewertung einfließen (*Volk* et al., 2001).

Das Untersuchungsgebiet befindet sich im Nordosten des Regierungsbezirkes Leipzig (Nord-Sachsen) und umfasst 686 km². Die ländlich geprägte Region lässt sich in die pleistozänen Heidebereiche (nachfolgend als Heide bezeichnet) im Westen und Osten mit vorwiegend sandigen Böden und eingelagerten Geschiebelehmen sowie die holozäne Elbaue (nachfolgend als Elbtal bezeichnet) im mittleren Bereich gliedern, die überwiegend Vega- und Gley-Vega-Böden aufweist. Die mittleren Jahresniederschläge schwanken zwischen 540 mm/a im Elbtal und ca. 650 mm/a in den höher gelegenen Heidebereichen. Das Gebiet ist durch weiträumig ausgewiesene Trinkwasser- (27 % im Jahr 1999) sowie Natur- und Landschaftsschutzgebiete (50 %) geprägt. Dabei bestehen insbesondere in Trinkwasserschutzgebieten (TWSG) Restriktionen für die Landbewirtschaftung. Das im Untersuchungsgebiet geförderte Trinkwasser speist die Fernwasserversorgung für den Raum Halle–Leipzig mit einer Kapazität von täglich 600.000 m² Wasser für ca. 3,5 Mill. Menschen. Dominierende Landnutzungsform im Gebiet ist die Landwirtschaft (49 % Ackerbau, 12 % Grünland), Wälder nehmen 28 % der Fläche ein (davon 77 % Nadelwald, vgl. Abb. 1).

Das Gebiet wurde zur späteren Extrapolation der Untersuchungsergebnisse zunächst in Teilräume gegliedert. Grundlage dafür waren die Bodeneigenschaften (Heide, Elbtal) sowie der bestehende Trinkwasserschutzstatus (vgl. Abb. 4).

Die durchgeführten Untersuchungen sind der mesoskaligen Ebene zuzuordnen, für deren Anwendungsbereich sich aufgrund fehlender übertragbarer Methoden die Modellwahl schwierig gestaltet. Gerade auf dieser Ebene lassen sich aber umweltrelevante Prozesszusammenhänge in der Landschaft erkennen und ein Großteil der konkreten Raumplanungen findet hier statt. Einen Überblick über verschiedene Systeme

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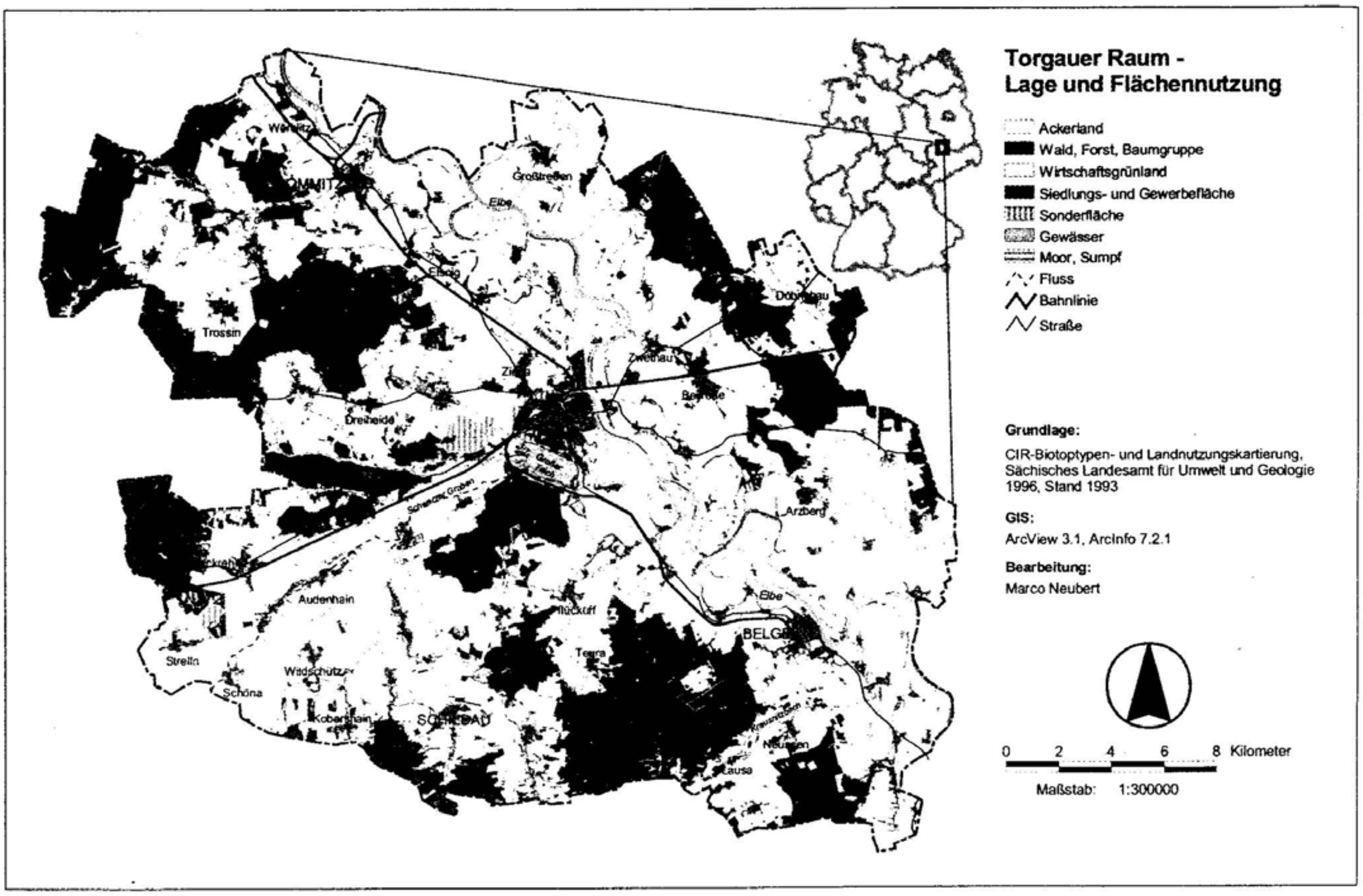


Abb. 1. Lage und Flächennutzung des Untersuchungsraumes Torgau
Fig. 1. Location and land use of the Torgau area

zur Stickstoffmodellierung gibt beispielsweise die *Bundesanstalt für Gewässerkunde* (1999). Die hier vorgestellte Methodik der Verknüpfung eines Wasserhaushaltsmodells mit einem N-Bilanzmodell und einem Geographischen Informationssystem (GIS) ist dort nicht berücksichtigt. Der Entwicklungstrend von Modellen zur N-Modellierung geht jedoch zunehmend in Richtung einer Kombination dieser drei Elemente zu integrierten Modellsystemen. Die Ergebnisse der Arbeit beziehen sich auf den derzeitigen Zustand des potenziellen Nitrataustrages (bezogen auf das Jahr 1993, Stand der Flächennutzungsdaten).

2 Methodik und Datengrundlagen

Die Methodik zur Modellierung der potenziellen Nitratkonzentration im Sickerwasser ist zentraler Bestandteil der Untersuchung. Sie lehnt sich an die Vorgehensweise von *Frede* und *Dabbert* (1998) an und ist in Abbildung 2 schematisch dargestellt.

Die verwendeten Geodaten (Landnutzungsdaten: Biotoptypen- und Landnutzungskartierung; Bodendaten: Reichsbodenschätzung, Forstliche Standorterkundung, Grundwasserdargebotsprognose; Klimadaten: Niederschlag, Verdunstung 1961–1990) lagen in digitaler Form als Vektor- oder Rasterdaten vor (vgl. *Herzog* et al., 2001). Alle Daten wurden miteinander verschnitten und die somit entstandenen kleinsten gemeinsamen Geometrien als Berechnungsgrundlage genutzt. Alle Daten sowie Berechnungsergebnisse wurden mittels GIS zur Darstellung und Auswertung aufbereitet. Die nötigen Informationen (Schlagkarteien) zur Berechnung der Stickstoffsalden wurden durch stichprobenartige Befragung verschiedenartiger Landwirtschaftsbetriebe im Untersuchungsgebiet erhoben (Tab. 1). Grundlage für die Zuordnung der Bewirtschaftungsintensität war die Teilnahme an den Sächsischen Agrarprogrammen ökologischer Landbau bzw. umweltgerechter Landbau (entspricht ökologischer bzw. integrierter Bewirtschaftung).

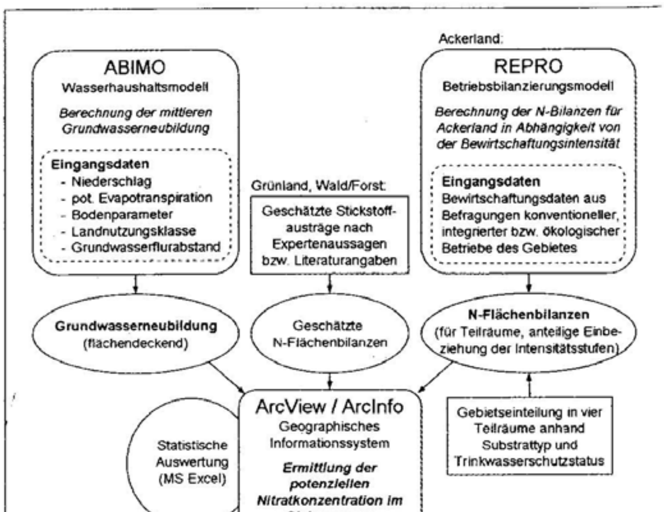
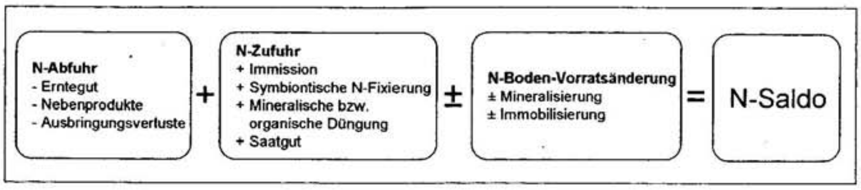


Abb. 2. Vorgehensweise und Eingangsparameter zur Ermittlung der potenziellen Nitratkonzentration im Sickerwasser
Fig. 2. Methodology and input parameters for analysing the potential fluxes of nitrate

Abb. 3. Bilanzierung der flächenbezogenen Stickstoffflüsse ackerbaulich genutzter Flächen mit dem Modell REPRO.
Fig. 3. Balancing of area-related nitrate fluxes of arable land using the model REPRO.



2.1 Verwendete Modelle und Verfahren

2.1.1 Grundwasserneubildung

Zur Ermittlung der Grundwasserneubildung wurde das Abflussbildungsmodell (ABIMO 2.1) der Bundesanstalt für Gewässerkunde angewendet (vgl. *Glugla* und *Fürtig*, 1997). ABIMO beruht auf der Berechnung des Gesamtabflusses aus der Differenz langjähriger Mittelwerte von Niederschlag und Evapotranspiration. Letztere wird mittels BAGROV-Beziehung geschätzt, welche die klimatischen Faktoren sowie die weiteren Standorteinflüsse in physikalisch und empirisch begründeter Weise verknüpft. Dabei beruht die empirische Komponente (Effektivitätsparameter n) auf einer umfangreichen Auswertung von Lysimetermessungen aus dem Lockergesteinsbereich der ehemaligen DDR. Es eignet sich somit gut für die Bedingungen des untersuchten Raumes und wurde bereits in zahlreichen Untersuchungsgebieten von Ämtern und Forschungseinrichtungen angewendet. Die Berechnungsergebnisse des Modells ermöglichen regionale Differenzierungen von Wasserhaushaltsgrößen (Evapotranspiration, Abfluss) in Abhängigkeit von den Boden- und Klimaverhältnissen sowie der Landnutzungssituation. Aufgrund der geringen Reliefenergie im Untersuchungsgebiet kann der Einfluss des lateralen Abflusses hier vernachlässigt werden. Versiegelte Flächen werden im Modell berücksichtigt (vgl. *Herzog* et al., 2001).

2.1.2 Stickstoffbilanzen

Für die Untersuchungen wurde das Modell REPRO 2000 (vgl. *Diepenbrock* et al., 1999) verwendet. Das „Black-Box"-Modell entspricht dem zur Anwendung auf größere Gebiete geeigneten Stickstoffbilanz-Verfahren gemäß dem *Deutschen Verband für Wasserwirtschaft und Kulturbau* (1996). Es herrscht ein akzeptables Verhältnis zwischen den Datenanforderungen des Modells und der Genauigkeit der Prozessabbildung. Die enthaltenen Vorgaben, Bilanz- und Umrechnungskoeffizienten basieren auf langjähriger Feldforschung sowie Literaturauswertungen. REPRO dient der betriebsbezogenen Humus- und Nährstoffbilanzierung in Landwirtschaftsbetrieben und ist in der Lage, Flächen-Nährstoffbilanzen für die Nutzungsarten Acker- und Grünland zu berechnen. Die in REPRO berücksichtigten Elemente des N-Haushaltes sind in Abbildung 3 dargestellt. Im hier beschriebenen Fall wird das Modell nicht auf real existierende Betriebe angewendet, sondern auf aggregierte Schläge einzelner Betriebe, die sich hinsichtlich ihrer Lage (Abhängigkeit des Ertrages vom Boden), ihrem Trinkwasserschutzstatus bzw. ihrer Bewirtschaftungsintensität (ökologischer, integrierter und konventioneller Landbau) unterscheiden. Für jeden Teilraum wurden quasi drei Musterbetriebe eingerichtet, die sich aus Schlägen mehrerer realer Betriebe zusammensetzen. Die Bewirtschaftungsform wurde durch die Betriebsbefragung erfasst (Tab. 1).

Die Festsetzung der atmogenen Deposition erfolgte basierend auf Forschungsergebnissen von *Mehlert* (1996), wobei ein integrales Messverfahren angewandt und ein Wert von 60 kg N ha^{-1} a^{-1} ermittelt wurde.

2.2 Die Bestimmung der mittleren potenziellen Nitratkonzentration im Sickerwasser

Die Ermittlung der mittleren potenziellen Nitrat-Konzentration im Sickerwasser erfolgte nach *Frede* und *Dabbert* (1998):

$$S_{NO3} = ((N_{BIL} - N_{DEN}) \cdot AF/GWN) \cdot 4{,}43 \cdot 100$$

Tabelle 1. *Erhobene Bewirtschaftungsdaten im Untersuchungsgebiet*
Table 1. *Collected land use data for the investigation area*

Kategorie			**Anzahl der Schläge**	**Fläche [ha]**	**Anteil an der Ackerlandfläche des Teilraumes (%)**
Naturraum	**Trinkwasser-schutz**	**Bewirtschaftung**			
Elbtal	ja	Ökologisch	3	8,4	13,9
		Integriert	11	334,9	
		Konventionell	16	538,5	
	nein	Ökologisch	–	–	6,1
		Integriert	2	85,8	
		Konventionell	6	303,2	
Heide	ja	Ökologisch	1	5,1	9,8
		Integriert	8	172,9	
		Konventionell	5	148,0	
	nein	Ökologisch	7	29,5	6,9
		Integriert	41	848,2	
		Konventionell	10	341,4	
Summe			110	2815,9	8,4

mit S_{NO3} = mittlere potenzielle Nitrat-Konzentration im Sickerwasser [mg/l]
N_{BIL} = mittlere N-Flächenbilanz [kg N ha^{-1} a^{-1}]
N_{DEN} = Denitrifikation [kg N ha^{-1} a^{-1}]
AF = Auswaschungsfaktor
GWN = mittlere Grundwasserneubildungsrate [mm]
4,43 = Umrechnungsfaktor Stickstoff zu Nitrat
100 = Umrechnungsfaktor

Das Verfahren wurde modifiziert, indem die Abschätzung der Netto-Mineralisations- und Immobilisierungsraten vernachlässigt wurde. Durch die Berechnungen mittels REPRO gehen diese Faktoren (als N-Bodenvorratsänderung) bereits in die N-Bilanz ein. Zur Bestimmung des Einflussfaktors Denitrifikation wurden ebenfalls die Angaben von *Frede* und *Dabbert* (1998) genutzt. Vereinfacht wurde dabei den Hauptbodenarten eine Denitrifikationsrate zugewiesen. Für die Sandböden der Heidegebiete wurde eine mittlere Denitrifikationsmenge von 10 kg N ha^{-1} a^{-1} verwendet. Den vorrangig in der Elbaue vorkommenden Lehm- und Tonböden wurde eine Denitrifikationsrate von 30 kg N ha^{-1} a^{-1} zugeordnet, Niedermoorböden eine Rate von 50 kg N ha^{-1} a^{-1}.

3. Ergebnisse und Diskussion

3.1 Berechnung der Grundwasserneubildung

Abbildung 4 zeigt die mittleren Grundwasserneubildungsraten berechnet auf Grundlage langjähriger Klimadaten sowie der Flächennutzung des Jahres 1993. Der flächengewichtete Mittelwert für den Gesamtraum beträgt demnach 136 mm. Dabei ist eine deutliche Differenzierung der Region zu beobachten, wobei sich die höchsten Werte in den sandigen Heidegebieten mit durchlässigen Böden und höheren Niederschlägen ergeben. Unter Waldflächen ist die Grundwasserneubildungsrate aufgrund der höheren Verdunstung weitaus geringer, als z. B. unter Ackerflächen. Flächen ohne Grundwasserneubildung werden fast ausschließlich durch größere Gewässerflächen, wie Elbe oder Großer Teich, repräsentiert. Weitere Flächen negativer bzw. geringer Grundwasserneubildung sind Folgen niedriger Grundwasserflurabstände und der damit einhergehenden erhöhten Verdunstung (vgl. *Herzog* et al., 2001; *Volk* et al., 2001).

Grundsätzlich ist zu beachten, dass es sich aufgrund der Modellstruktur bei den Ergebnissen um langjährige Mittelwerte mit relativ geringer räumlicher Auflösung handelt. Die Ergebnisse geben somit die herrschenden klimatischen Bedingungen des Jahres 1993 nicht genau wieder. Der Mittelwert der Grundwasserneubildung im untersuchten Gebiet liegt nach den Angaben des N-A-U-Kartenwerkes bei 110 bis 130 mm/a (Heidegebiete) bzw. 80 mm/a (Elbtal). Die Größenordnungen sind ähnlich, insgesamt werden die auf groben Rasterwerten basierenden und sich auf einen anderen Zeitraum beziehenden N-A-U-Werte bei der vorliegenden Berechnung überschritten. Die Berechnungen wurden Sensitivitätsanalysen unterzogen, nach denen die Abweichungen und Schwankungsbereiche der Ergebnisse in dem für mesoskalige Modellierungen tolerierbaren Bereich liegen (vgl. *Herzog* et al., 2001; *Volk* et al., 2001).

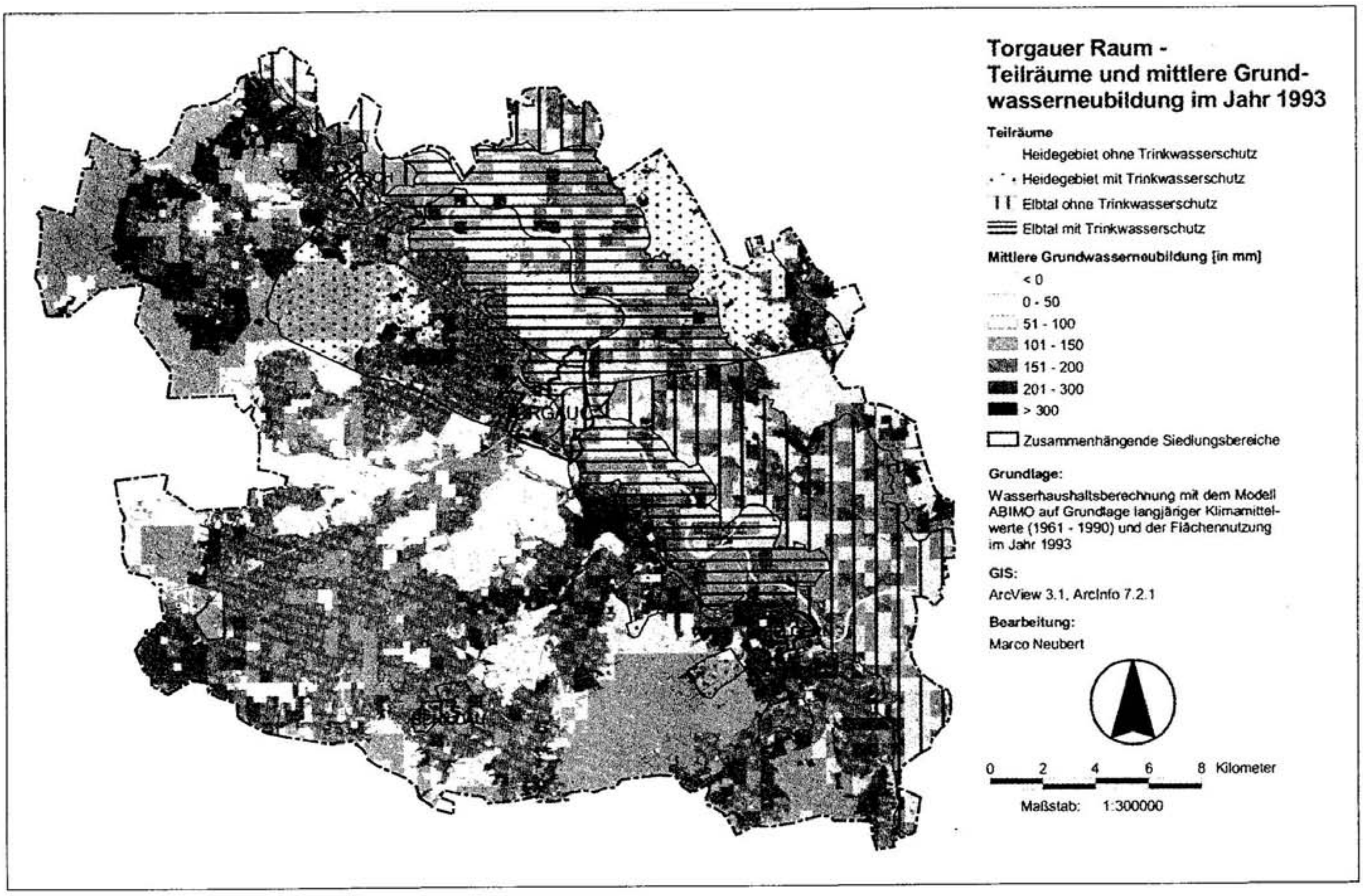


Abb. 4. *Teilräume des Gebietes und mittlere Grundwasserneubildung im Jahr 1993*
Fig. 4. *Partial regions of the area and average ground water recharge in 1993*

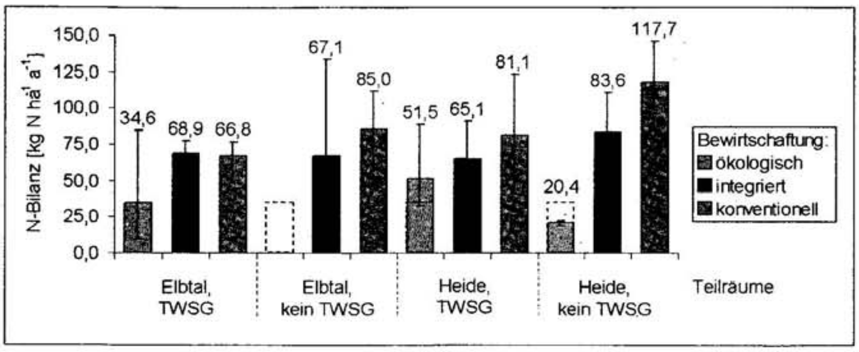


Abb. 5. *Mittlere Stickstoffsalden für Ackerland unterschiedlicher Bewirtschaftungsintensität und Lagekriterien im Zeitraum 1995 bis 1998, einschließlich der Unsicherheiten der Berechnung (Schwankungsbreite Min.–Max.)*
Fig. 5. *Mean nitrate balances of agricultural land with varying cultivation practices and location criterias for the period of 1995 to 1998, including the uncertainty of calculation (range min. to max.)*

3.2 Berechnung der mittleren Stickstoffbilanzen

Abbildung 5 zeigt die Ergebnisse der berechneten Stickstoffbilanzen für Ackerland. Die dargestellten Stickstoff-Bilanzüberschüsse charakterisieren das Nitratauswaschungspotenzial im Torgauer Raum. In der Abbildung werden die Auswirkungen der verschiedenen Bewirtschaftungsintensitäten auf die Stickstoffbilanzen ersichtlich. Die geringsten Bilanzüberschüsse weisen dabei ökologisch wirtschaftende Betriebe auf. Hohe Stickstoffsalden werden bei konventioneller Wirtschaftsform erkennbar, während die Werte bei integrierter Bewirtschaftung zumeist auf mittlerem Niveau liegen. Diese Ergebnisse werden einerseits durch unterschiedliche pflanzenbauliche Praktiken sowie geltende Bewirtschaftungsauflagen hervorgerufen. Andererseits spielen auch unterschiedliche naturräumlichen Bedingungen eine Rolle. So weisen im Elbtal gelegene Flächen geringere Bilanzüberschüsse auf, da hier das Düngungsniveau aufgrund der höheren Nährstoffspeicherkapazität niedriger ist. Zudem machen sich die die positiven Auswirkungen von Bewirtschaftungsauflagen im Bereich von TWSG durch 10 bis 30 kg N ha^{-1} a^{-1} geringere Stickstoffüberschüsse gegenüber Gebieten ohne Schutzauflagen bemerkbar.

Infolge des geringen Stichprobenumfanges für Daten ökologisch bewirtschafteter Flächen (Tab. 1) und der außerordentlich hohen Schwankungsbreiten weisen die Berechnungsergebnisse einen hohen Unsicherheitsgrad auf. Daher wurde für die weiteren Berechnungen ein mittlerer Wert von 35 kg N ha^{-1} a^{-1} für die Stickstoffbilanz aller ökologisch bewirtschafteter Flächen angenommen. Dieser Wert entspricht den Ergebnissen vergleichbarer Untersuchungen für das Wassergut Canitz (bei Wurzen), wobei 30 bis 40 kg N ha^{-1} a^{-1} Herbst-Stickstoffvorrat angegeben werden.

Zur Extrapolation auf die Flächen der vier Teilräume (Abb. 4), wurden die berechneten Stickstoffbilanzen mit realen Anteilen der Intensitätsstufen des jeweiligen Teilraumes kombiniert. Das Ergebnis sind je nach auftretender Bewirtschaftungsintensität anteilig gewichtete Stickstoffbilanzen für die einzelnen Teilräume (siehe Tab. 2). Für die gesamte ackerbauliche Fläche des Untersuchungsraumes ergibt sich demnach eine Stickstoffbilanz von 84 kg N ha^{-1} a^{-1}.

Um die Plausibilität der ermittelten Werte zu überprüfen, wurden sie mit Untersuchungen der Sächsischen Landesanstalt für Landwirtschaft verglichen. In diesen Untersuchungen werden NO_3-N-Bodengehalte Sachsens im Herbst, aufgegliedert nach unterschiedlichen Bewirtschaftungsintensitäten, dargestellt. Im Vergleich der relativen Höhe der Werte zeigen sich gute Übereinstimmungen. Beim ökologischen Landbau liegt jeweils ein geringer Stichprobenumfang vor, so dass sich hier eine Gegenüberstellung schwierig gestaltet. Im Vergleich zu den Ergebnissen von *Wendland* et al. (1993) und *Biermann* (1995) zeigt sich, dass das Niveau der Stickstoffbilanzen der Jahre 1986 bis 1989 (50 bis 150 kg N ha^{-1} a^{-1}) heute nicht mehr erzielt wird.

Vergleichende Untersuchungen ökologisch und konventionell wirtschaftender Betriebe zeigen, dass im ökologischen Landbau deutlich geringere N-Bilanz-Überschüsse auftreten (*Haas* et al., 1998). Hierin ist eine Ursache niedriger Nitratkonzentrationen in der Sickerwasserzone unter Ackerflächen des ökologischen Landbaus zu sehen.

Für nicht ackerbaulich genutzte Flächen wurden die Stickstoffausträge auf Grundlage von Literaturangaben bzw. Expertenaussagen pauschal geschätzt, wobei nur Forst- und Grünlandflächen berücksichtigt werden konnten. Die Stickstoffausträge aus Forstflächen wurden mit 5 kg N ha^{-1} a^{-1} festgesetzt. Dabei ist keine Unterscheidung in Nadel- oder Laubwald möglich. Für extensiv bewirtschaftete Grünlandflächen wurde eine Auswaschungsrate von 5 kg N ha^{-1} a^{-1} genutzt, für die intensive Variante 10 kg N ha^{-1} a^{-1} (vgl. *Franko* et al., 2001).

3.3 Mittlere potenzielle Nitratkonzentration im Sickerwasser

Grundsätzlich ist zu unterstreichen, dass es sich bei den Untersuchungsergebnissen um mittlere potenzielle NO_3-Sickerwasserkonzentrationen handelt, die aus mittleren Stickstoff-

Tabelle 2. *Mittlere Stickstoffsalden der einzelnen Teilräume für 1993 [kg N ha^{-1} a^{-1}]*
Table 2. *Average nitrate balances of the partial regions of the area for 1993 [kg N ha^{-1} a^{-1}]*

Teilraum	Elbtal, TWSG		Elbtal, kein TWSG		Heide, TWSG		Heide, kein TWSG	
Bewirtschaftung	Anteil	N-Saldo	Anteil	N-Saldo	Anteil	N-Saldo	Anteil	N-Saldo
Konventionell	55,0 %	37	60,0 %	51	25 %	20	30 %	35
Integriert	44,5 %	31	39,5 %	27	74 %	48	69 %	58
Ökologisch	0,5 %	0	0,5 %	0	1 %	0	1 %	0
Bilanz der Teilräume	68		78		69		93	

flächenbilanzen in Kombination mit mittleren Grundwasserneubildungsraten berechnet wurden.

Die Ergebnisse der Berechnung sind Abbildung 6 zu entnehmen. Sehr hohe Nitratkonzentrationen im Sickerwasser treten unter Ackerland auf, wobei aufgrund der unterschiedlichen Stickstoffbilanzen klar in Heidegebiete und Elbtal differenziert werden kann (siehe Tabelle 2). Nitratausträge, die dem Nitratgrenzwert der Trinkwasserverordnung genügen, weisen lediglich die Ackerflächen des Elbtales innerhalb von TWSG auf. Die Ackerflächen auf den gut durchlässigen Substrattypen der Heidegebiete erzielen dagegen beträchtliche NO_3-Austräge von 100 bis etwa 250 mg NO_3/l.

Die weiteren Flächennutzungen (Grünland und Forst) lassen sehr geringe Austräge erkennen. Sie liegen fast ausnahmslos unter 25 mg NO_3/l und entsprechen somit dem Nitrat-Richtwert der Europäischen Union. Bedingt durch die extensivere Nutzung des Grünlandes innerhalb von Trinkwasserschutzgebieten weisen diese Flächen geringere Austräge auf, als solche ohne Schutzauflagen. Die Austräge unter Nadelwald sind geringfügig höher als unter Laubwald, da beide Waldarten unterschiedliche Grundwasserneubildungsraten besitzen, die sich um etwa 10 mm unterscheiden.

Erwähnenswert ist, dass auf Flächen mit negativer Grundwasserneubildung (außer Gewässer) eine Stickstoffakkumulation stattfindet. Auf derartigen Flächen können die Böden durch kapillaren Aufstieg auch Nitrate aus dem Grundwasser aufnehmen. In feuchten Jahren kann diese Nitratsenke allerdings erhebliche Mengen an Nitrat zur Auswaschung freisetzen.

In Tabelle 3 sind die flächengewichteten Mittelwerte der potenziellen NO_3-Konzentration aufgeführt. Dabei zeigt sich eine Abhängigkeit der Nitratkonzentration von der Landnutzung, Bewirtschaftungsintensität sowie von den Bodenverhältnissen und dem Trinkwasserschutzstatus. Insgesamt konnte für ca. 90 % der Gebietsfläche eine Aussage über den potenziellen Nitrataustrag mit dem Sickerwasser getroffen werden. Der flächengewichtete Gebietsmittelwert der potenziellen NO_3-Konzentration beträgt 73 mg/l.

4 Schlussfolgerungen und Ausblick

Zur Modellierung der Grundwasserneubildung auf regionaler Ebene ist das Modell ABIMO gut geeignet. Die Ergebnisse erlauben eine großräumige Abschätzung bzw. Einordnung der Grundgrößen des Landschaftswasserhaushaltes. Regionale Unterschiede sowie qualitative und quantitative Gefährdungspotenziale für den Wasserhaushalt können deutlich gemacht werden und in planerische Fragestellungen einbezogen. Die Berechnungsergebnisse präsentieren sich zwar in Zahlen, geben jedoch aufgrund des Modellcharakters und der geringen räumlich-zeitlichen Auflösung der Eingangsdaten nur Größenordnungen wieder (vgl. *Volk* et al., 2001).

Mittels Stoffbilanzen, die als wichtige Agrar-Umweltindikatoren angesehen werden, können potenzielle Umweltbelastungen, insbesondere mögliche Nährstoffverluste, quantitativ ermittelt werden. Die hier berechneten Nitratausträge ermöglichen eine Abschätzung des Einflusses unterschiedlicher Landnutzungen und deren Intensitäten. Durch die Verknüpfung der Bilanzierungsergebnisse mit Standortdaten in

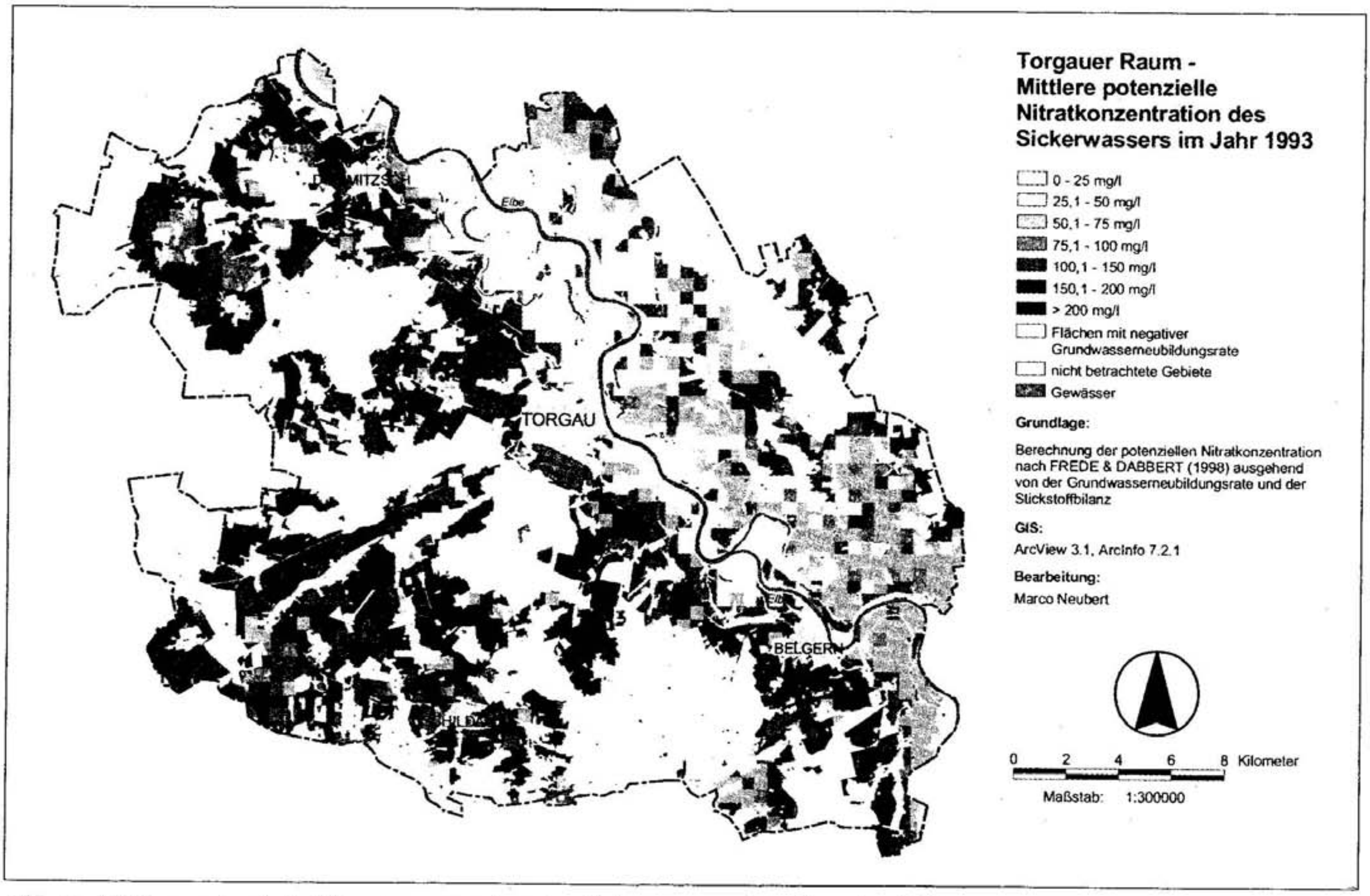


Abb. 6. *Mittlere potenzielle Nitratkonzentration des Sickerwassers berechnet für 1993*
Fig. 6. *Mean potential nitrate concentration of the seeping water calculated for 1993*

Tabelle 3. Potenzielle Nitratkonzentration im Sickerwasser verschiedener Flächennutzungen berechnet für 1993
Table 3. Potential nitrate concentration of the seeping water for different land use types calculated for 1993

Raumbezug	Fläche [km²]	Flächengewichtetes Mittel der potenziellen NO_3-Konzentration (mg/l)	Schwankungsbreite		
			Minimum [mg/l]**	Maximum [mg/l]**	Standard-abweichung
Ackerland					
Elbtal, TWSG	63,5	64,6	25,6	151,0	40,3
Elbtal, kein TWSG	63,7	81,1	40,4	312,4	49,6
Heide, TWSG	33,2	124,1	27,4	273,5	29,7
Heide, kein TWSG	176,6	160,1	57,2	385,0	44,1
Ackerland gesamt	337,0	123,6	25,6	385,0	55,8
Grünland					
Innerhalb TWSG	29,8	9,5	5,4	15,9	3,1
Außerhalb TWSG	54,7	19,2	9,5	48,5	5,0
Grünland gesamt	84,5	15,7	5,4	48,5	6,3
Wälder und Forsten					
Laubwald	34,3	12,2	4,5	165,5	3,7
Nadelwald	153,5	12,6	4,5	246,2	4,1
Wald/Forst gesamt*	198,8	12,5	4,5	246,2	5,2
Gesamtraum					
Summe	620,3	73,3	–	–	–
Unbestimmte Flächen	66,0	–	–	–	–

* Enthält geringen Anteil von Wald ohne Zuordnung zu Laub- oder Nadelwald
** Die Extremwerte werden nur kleinflächig erreicht

einem GIS ist es möglich, auf Gefährdungspotenziale im Raum hinzuweisen und der Raumplanung als Information zur Verfügung zu stellen.

Die höchste Nitratauswaschungsgefahr besteht unter ackerbaulich genutzten Flächen mit konventioneller Landbewirtschaftung. Deshalb ergibt sich für diese Flächennutzung ein dringlicher Handlungsbedarf, um schädliche Einwirkungen auf das Grundwasser zu vermeiden. Dabei ist die Agrarpolitik gefordert, zukünftig eine naturressourcenschonende Landwirtschaft zu etablieren. Insbesondere ist eine Extensivierung bzw. ökologische Bewirtschaftung sensibler Bereiche anzustreben.

Die Menge ausgewaschenen Nitrats stellt nicht die Nitratkonzentration im Grundwasser dar. Hierzu wäre eine genaue bodenhydrologische bzw. hydrogeologische Untersuchung notwendig. Die im Rahmen der Berechnungen vorgenommene grobe Denitrifikationsabschätzung kann bei Vorliegen genauerer Grundlagen verbessert werden. Zu beachten sind zudem die langen Verweilzeiten im Sicker- oder Grundwasser, so dass zwischen Stoffaustrag aus der Wurzelzone und der beobachteten Trinkwasserbelastung bis zu 20 Jahren liegen können (vgl. *Biermann*, 1995; *Länderarbeitsgemeinschaft Wasser*, 1995). Zusätzlich ist das Grundwasser im Elbtal durch die flächenhafte Auenlehmdecke gut geschützt.

Offenbar herrschen im Grundwasserleiter des untersuchten Gebietes derzeit sehr gute Denitrifikationsbedingungen, denn die hohen Stickstoffkonzentrationen im Sickerwasser von 60 mg N/l und mehr werden infolge dieser immensen Abbauleistungen bis auf Stickstoffkonzentrationen im Grundwasser von weniger als 2 mg N/l verringert (vgl. *Behrend* et al., 1999). Tatsächlich lagen die Qualitätsdaten von Wasserwerken in der Torgauer Elbaue in diesem, sehr niedrigen Bereich, jedoch mit z. T. stark steigender Tendenz (Wasserwerk Mockritz: 1998: 2,3 mg/l, 1999: 21,4 mg/l; Wasserwerk Torgau-Ost: 1998: 1,8 mg/l, 1999: 3,8 mg/l). Für eine langfristige Sicherung der Grundwasserressourcen ist eine zukünftige Verringerung der Einträge unerlässlich, da es aufgrund des irreversiblen Aufbrauchens der nitratreduzierenden Stoffe nach deren Aufzehren zu einem „Nitratdurchbruch" kommen kann (vgl. *Länderarbeitsgemeinschaft Wasser*, 1995; *Wendland* et al., 1993).

Ein weiterer Grund der geringen NO_3-Konzentrationen im Grundwasser stellen Dränagen dar. Mit dem Dränwasser werden erhebliche Nitratmengen in Fließgewässer abgeführt, die ansonsten ins Grundwasser gelangen würden. Dränagen können somit positive Auswirkungen auf die Sickerwasserqualität haben. Ein hoher Anteil des mit dem Sickerwasser ausgetragen Nitrats gelangt jedoch auf diesem Weg in die Oberflächengewässer und entfaltet dort seine negative Wirkung. Angaben zum Dränflächenanteil des Torgauer Raumes lagen während der Untersuchungen nicht vor.

Die Ergebnisse der modellierten NO_3-Konzentrationen im Sickerwasser können aufgrund der Qualität der Eingangsdaten sowie des Modellcharakters nur Tendenzen wiedergeben. Es handelt sich lediglich um eine vereinfachte Schätzmethode, welche jedoch für Bewertungen auf mittlerer Maßstabsebene als akzeptabel beurteilt werden kann. Durch die Kombination mit einem Geographischen Informationssystem können flächendifferenzierte Rückschlüsse auf die Auswirkungen durch unterschiedliche Landnutzungsarten und -intensitäten gezogen werden. Der Vergleich mit Ergebnissen von *Franko* et al. (2001) mit anderer Methodik zeigt überwiegend gute Übereinstimmungen. Weiterer Forschungsbedarf besteht hinsichtlich der genaueren Prozessaufklärung der Nitratauswaschung, der atmogenen Deposition sowie der Entwicklung des Nitrathaushaltes der Wald- und Grünlandgebiete.

Nutzungsmöglichkeiten der Ergebnisse bieten sich neben der wissenschaftlichen Anwendung vor allem in der Raumplanung als Entscheidungshilfe für raumwirksame Akteure. Eine Einbindung in laufende kommunale oder regionale Planungsprozesse ist denkbar. So ließe sich mit Hilfe der vorliegenden Ergebnisse gegen eine angestrebte Auflösung von Trinkwasserschutzgebieten im Untersuchungsraum argumentieren. Generell wäre eine stärkere Kooperation zwischen Wasser- und Landwirtschaft in derartigen Konfliktgebieten wünschenswert. Mit politischer und planerischer Unterstützung ließe sich so eine ressourcenschonende Landwirtschaft umsetzen. Denn der Landwirtschaft kommt mit einem genutzten Ackerlandanteil von 50 % des Untersuchungsgebietes eine entscheidende Rolle beim Boden- und Wasserschutz zu.

Auf Basis verschiedener Szenarien können die Auswirkungen unterschiedlicher Bewirtschaftungsintensitäten auf die Sickerwasserqualität deutlich gemacht und im Sinne des Naturressourcenschutzes Empfehlungen für standortgerechte Bewirtschaftungsformen gegeben werden (vgl. *Volk* et al., 2001; *Franko* et al., 2001).

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Towards the implementation of the European Water Framework Directive? Lessons learned from water quality simulations in an agricultural watershed

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ABSTRACT

The main objective of the European Water Framework Directive (WFD) is the achievement of a good ecological and chemical status of the water environment (water bodies). This status corresponds to the limit value of Germany's Working Group of the Federal States on Water Problems Issues (LAWA) for water quality class II (3 mg/l total nitrogen). The rivers in the intensively cropped Upper Ems River basin (northwestern Germany) show total nitrogen concentrations in excess of 5–10 mg/l. Hence, the objective of our study was to find a land use and land management scenario that would reduce the total nitrogen concentration to meet the WFD requirements for good ecological and chemical status. We developed consecutive land use and management scenarios on the basis of policy instruments such as the support of agro-environmental measures by Common Agricultural Policy and regional landscape development programs. The model simulations were done by using the Soil and Water Assessment Tool (SWAT). Results of SWAT scenario calculations showed that drastic measures, which are unrealistic from a socio-economic point of view, would be needed to achieve the water quality target in the basin (reduction of arable land from 77.2% to 46% [13% organic farming], increase of pasture from 4% to 15%, afforestation from 10% to 21%, increase of protected wetlands from 0% to 9%, etc.). The example shows additionally that the achievement of the WFD targets is only possible with a consideration of regional landscape and land use distinctions. A related problem yet to be addressed is the general lack of measured water quality data with which to calibrate and validate water quality models such as SWAT. This adds considerable uncertainty to already complicated and uncertainty situations. Thus, improved strategies for water quality monitoring, and data accessibility must be established.

Introduction

The application of large amounts of mineral and organic fertilizers in intense agricultural regions of Europe contributes to excessive nutrient loads in soils, ground, and surface water bodies. Nitrogen leaching from agricultural land is a common problem in many European countries with intensive agricultural production. The contribution of agriculture to nonpoint source pollution of surface waters is estimated to be 55% for the European Union (Isermann and Isermann, 1995; Kersebaum et al., 2003) and 48% for Germany (Isermann, 1990). In contrast, other regions experienced decreasing agricultural intensity in recent years (EC, 1998; Zebisch, 2002; Westhoek et al., 2006). Such land use trends are not only influenced by general driving forces like macroeconomic developments and demographic changes but also by policy instruments such as Common Agricultural Policy (CAP) with the support of agro-environmental measures (Rossing et al., 2007), national and regional landscape development plans, or by the implementation of environmental programs such as the European Water Framework Directive (WFD) (EC, 2000).

An example of an agricultural region with intensive use and high numbers of livestock is the Upper Ems River Basin in northwestern Germany (Gömann et al., 2005). About 77.2% of the watershed is covered by agricultural land. As a consequence, very high total nitrogen (total-N) loads and frequent concentrations greater than 5–10 mg/l of total-N substantially impair the water quality of the Ems River (Jarvie et al., 1997; Volk et al., 2008). Germany's Working Group of the Federal States on Water Problems Issues (LAWA) requires for instance 3 mg/l of total nitrogen as limit value for surface waters (water quality class II) (LAWA, 1998). The LAWA water quality classification corresponds to the classification used by the WFD. Thus the current situation in the Ems River Basin is far from the postulated environmental targets of the WFD. According to recent scientific findings, a further decline of annual nitrogen

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surpluses down to 50 kg N per ha agricultural area and an increase of the denitrification potential (e.g. extensification of land use, backwater or plugging of drainage systems, restoration of wetlands and improvements of morphological water structure) would be necessary to achieve this goal (Gömann et al., 2005). Regional implementation of such measures would have far-reaching impacts on agriculture, which represents also a major challenge for the implementation of the WFD, in addition to the rather tight time frame for such changes: The first deadline for achieving the environmental objectives of the WFD is 2015. Thus, research attempts to answer the question: How realistic is the achievement of the WFD water quality targets in such river basins dominated by agriculture?

Modelling tools, which take into account possible land use and management scenarios, can be helpful in determining measures to achieve a target ecological status (Kersebaum et al., 2003; Chaplot et al., 2004; Krause et al., 2008). Examples of such models include HSPF (Bicknell et al., 2001), AGNPS (Young et al., 1987), MIKE-SHE (Refsgard, 1997), Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998; Neitsch et al., 2002; Gassman et al., 2007), and SWIM (Krysanova et al., 1998). Overviews of different eco-hydrological models are given in Volk and Steinhardt (2001), Krysanova and Haberlandt (2002), Horn et al. (2004), and Arnold and Fohrer (2005). In connection with the question above, the main objective of the study was thus to develop a final land use and land management scenario that would result in the required reduction of total nitrogen in the rivers to achieve the LAWA's water quality class II. The final scenario was comprised of consecutive land use and land management scenarios formulated from relevant policies.

Experiences of different European and national projects dealing with the model-supported implementation of the WFD revealed that the available models – and here especially integrated model systems – are still far from being suitable for operational applications. This is especially the case for water quality (Euroharp-Project, 2007). For optimum working efficiency of the models in the management processes it is required that they contribute information of a wide range of abiotic and biotic aspects of hydrology and demanded by the decision makers, which cannot be achieved by single groundwater, water quality or erosion models. Thus, we checked the suitability of the publicly available river basin model Soil and Water Assessment Tool (Arnold et al., 1998) to represent adequately general trends of water quality changes resulting from various measures based on land use and management change. SWAT was found to be one suitable integrated model that is able to simulate water quality – but further testing is needed (Horn et al., 2004). The results of the EUROHARP-project (EUROHARP, 2007) found SWAT "highly capable to simulate the effect of nutrient management, land use changes and water measures on N-losses."

The studies are part of the modelling component of the FLUMAGIS[1] project which supports the assessment and three-dimensional visualization of hydrological ecological and socio-economic conditions and management (Volk et al., 2008).

Methods

Study area

The analysis was carried out for the Upper Ems River basin in northwestern Germany, which covers an area of 3740 km^2. The hydrological processes in the basin are influenced by increasing precipitation amounts from the Northwest and Central basin (700 mm/year) and the Southeast (1200 mm/year). The basin is a predominantly flat landscape with widespread permeable sandy soils. The River Ems has its sources at the foothills of the Teutoburger Wald mountains – with maximum altitudes of about 360 m above sea level – and flows through the North German Lowlands to the North Sea. Fig. 1 shows the location of the study area and the associated land use pattern.

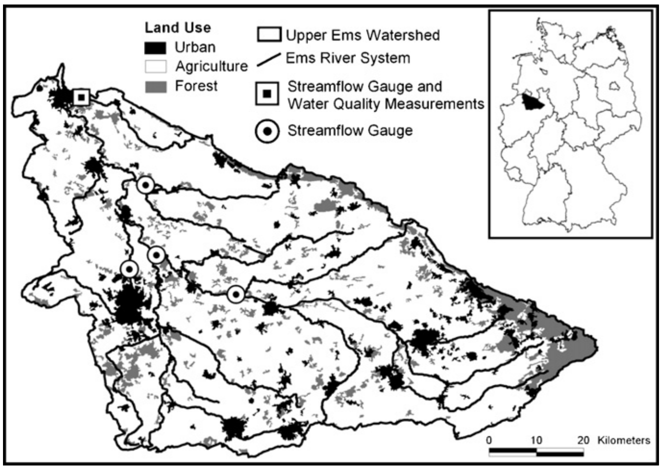

Fig. 1. Location of the study area in Germany and main land use types in the region.

The Ems basin is situated in one of the most intensive agricultural regions in Europe. Arable land covers approximately 77.2% of the area (the average in Germany is 50%; BMELV, 2008), which has led to a dramatic loss of landscape diversity. The proportions of the other land use types are 9.9% for forest, 8.9% for urban areas, 3.9% for pasture and 0.1% for other areas. Intensive livestock production has contributed to severe environmental problems, as evidenced by the exceedance of the nitrogen value of the water quality class II by a factor three to four for some Ems River gauges. Jarvie et al. (1997) showed that the Ems River had the highest load of total-N per unit catchment area of 12 investigated European catchments and concluded that this problem is "probably derived from agricultural sources such as artificial fertilizers and slurry, as it drains the north west corner of Germany, an area of intensive agricultural production (mixed farming, dairying and pig farming)."

Model description

SWAT was developed to quantify the impact of land management practices in large, complex watersheds with varying soils, land use, and management conditions over a long period of time (Arnold and Fohrer, 2005). It is an operational or conceptual model that operates on a daily time step. Many studies world-wide have used SWAT for evaluating the impact of land use scenarios and management practices on water quality (Saleh et al., 2000; Santi et al., 2001; Vaché et al., 2002; Chaplot et al., 2004; Pandey et al., 2005; Tripathi et al., 2005; Behera and Panda, 2006; Santhi et al., 2006; Gassman et al., 2007). SWAT uses readily available inputs, has the capability of routing runoff and chemicals through streams and reservoirs, and allows for the addition of flows and the inclusion of measured data from point sources. Moreover, SWAT has the capability to evaluate the relative effects of different management scenarios on water quality, sediment, and agricultural chemical yield in large, ungauged basins. A command structure is used for routing runoff and chemicals through a watershed similar to the structure included for routing flows through streams

[1] FLUMAGIS is an acronym for "Interdisziplinäre Entwicklung von Methoden und Werkzeugen für das Flusseinzugsgebietsmanagement mit Geoinformationssystemen" (Interdisciplinary development of methods and tools for the planning process and measurement control for river basin management with geo-information systems) (see http://www.flumagis.de/english/e_index.htm).

and reservoirs, adding flows and inputting measured data on point sources. Using the routing command language, the model can simulate a basin subdivided into grid cells or subwatersheds. Additional commands have been developed to allow measured in-stream and point source data to be input and routed with simulated flows.

Model sub-basin components can be divided into the following: hydrology, weather, sedimentation, soil temperature, crop growth, nutrients, pesticides, and agricultural management. Hydrology processes simulated include surface runoff estimated using the SCS curve number or Green–Ampt infiltration equation; percolation modelled with a layered storage routing technique combined with a crack flow model; lateral subsurface flow; groundwater flow to streams from shallow aquifers; potential evapotranspiration by the Hargreaves, Priestley–Taylor and Penman–Monteith methods; snowmelt; transmission losses from streams; and water storage and losses from ponds (Arnold et al., 1998). The model has been widely used but also further developed in Europe (e.g. Krysanova et al., 1998; Eckhardt et al., 2002; Van Griensven and Bauwens, 2003). SWAT was chosen for this research for three main reasons: its ability to simulate river nitrogen concentration on the catchment scale, its European wide use (Arnold and Fohrer, 2005; Bärlundt et al., 2007; Gassman et al., 2007), and its potential to simulate agricultural management practices (Turpin et al., 2005; Arabi et al., 2008). In previous studies SWAT was evaluated against the diffuse pollution benchmark criteria developed by the BMW project, and it was found to have potential with respect to the Water Framework Directive requirements (Dilks et al., 2003).

Model inputs

The Arc View-Geographic Information System interface of the SWAT2000 version (Di Luzio et al., 2004) was used to develop the SWAT input files. Recently available GIS maps for topography, land use, and soils were used. Table 1 gives an overview on the used model input data. Typical management practices such as crops grown, fertilizer application, and tillage operations for different land uses were gathered from state agricultural statistics and from the statistical yearbook of the State of North Rhine Westphalia. Atmospheric nitrogen deposition, which can be considered by the model, was assumed to be 2.5 mg N/l (between 35 and 45 kg/ha/year depending on rainfall conditions) (Gauger et al., 2002; StUA, 2005). To simulate the loading of water and pollutants from sources not associated with a land area (e.g. sewage treatment plants), we included the values of 100 sewage treatment plants (provided by State Environmental Agency) as point sources along channel networks into the model.

Model setup and calibration

In river basin models such as SWAT, land cover properties, which are relevant for the considered processes, have to be characterised by plant-specific parameters. Technically, simulation of a land cover change signifies a modification of the values of these parameters in certain parts of a catchment. Thus, reliable results can only be obtained if the parameter values for the land covers involved are known with some accuracy (Eckardt et al., 2003).

Thus, before developing the scenarios and implementing them into the model, a comprehensive sensitivity analysis of selected model parameters and different management practices was carried out using virtual catchments (Volk and Schmidt, 2004; Volk et al., 2008).

The simulation period for the current conditions and the scenarios was between 1980 and 2000. The simulation of the current conditions was based on the recent land use distribution (cp. Section "Study area" and Fig. 1) and a land management with a crop rotation of fodder corn, barley, and wheat which is applied on 90% of the arable land (LDS, 2001). The first simulation (scenario 0) was a status quo scenario with the current land use situation which was used to calibrate the model. The simulated discharge was calibrated at six gauges using two objective functions, the Nash–Sutcliffe coefficient of efficiency (NSE) (Nash and Sutcliffe, 1970), and the PBIAS (percent bias). The NSE was selected because it is dimensionless and is easily interpreted. When the measured variable is simulated exactly by the model, NSE equals 1. If NSE < 0, the predictive precision of the model is lower than when the mean of the values measured is used. PBIAS measures the average tendency of simulated flows to be larger or smaller than their observed counterparts (Gupta et al., 1999).

In the calibration period (period between 1986 and 2000; daily values) an average NSE of 0.75 and a PBIAS of 2% were achieved (daily values). In the validation period (period between 1970 and 1985; daily values) the NSE was 0.65 and the average PBIAS 2.5%. Taking into account that the discharge regime of the Ems River is heavily modified, these results can be considered as satisfactory.

In contrast to the availability of long-term daily discharge time series in the study area, there is a lack of water quality data. Unfortunately, it was almost impossible to strictly calibrate the model

Table 1
Model input data sources for the Upper Ems River basin

Data type	Scale	Source	Data description/properties
Topography	1: 50,000	State Survey Office	Elevation, overland and channel slopes, lengths
Soils	1:1,000,000	German Soil Map BUEK1000	Soil physical properties such as bulk density, texture, saturated conductivity, etc.
Land use (CORINE land cover)	1:100,000	Federal Statistical Office	Land use classifications
Land management information	–	State agricultural statistics	Fertilizer application rates and timing, planting and harvesting information
		Statistical yearbook of North Rhine Westphalia	
Weather	24 Stations (daily rainfall) 5 Stations (daily temperature) 5 Stations (daily wind speed) 6 Stations (relative humidity)	German Weather Service (DWD)	Daily precipitation and temperature
Sewage treatment plants	100 treatment plants (area-weighted to 4 point sources)	State Environmental Agency	l/sec sewage
Livestock	–	Landscape development program (MUNLV, 2001)	Livestock units per ha
Atmospheric deposition	–	State Environmental Agency; Gauger et al. (2002)	Concentration of nitrogen in rainfall (2.5 mg N/l)

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Table 2
Limit values for nitrogen, water quality class II (LAWA, 1998)

	Total N	Nitrate-N	Nitrite-N	Ammonium-N
Maximum concentration (mg/l)	≤3.0	≤2.5	≤0.1	≤0.3

for water quality. Instead, we were forced to calibrate the model on the basis of five average annual nitrogen concentration values for the period between 1990 and 1995. Only for the year 2001 three data samples per month were available. Thus, it is dispensable to present the calibration results of an objective function in this case. Anyway, it was possible to fit the model to the average annual values adequately.

Land Use and Management Options

The high infiltration rates of the sandy soils favour nitrogen leaching to the groundwater and the nutrient transport to the river. Hence, we investigated how far the change of land use and management practices could improve the water quality and meet the standards of the WFD.

The objective was to determine a land use scenario with the corresponding land management practices to achieve the limit value for total nitrogen concentration of the water quality class II (good ecological status after WFD). The limit values for selected nutrients are listed in Table 2.

We included the main targets of relevant programs such as the support of agro-environmental measures by the Common Agricultural Policy, which is considered also in development programs for rural areas, and the landscape program of the State of North Rhine Westphalia (KULAP) to represent realistic trends of land use changes and management practices (EC, 1998; MUNLV, 2001; Zebisch, 2002; BMVEL, 2004; StUA, 2006) with our scenarios. The landscape programs in Germany are instruments of environment and agriculture policy to implement the demands of the European agricultural reform (EWG 2078/92) and in the Agenda 2000 (EG 1257/1999). Table 3 shows general trends for land use types with increasing and decreasing land demand. These trends were published in the landscape program of North Rhine Westphalia (2002) and in development programs for rural areas (2003). They assumed that these trends would continue at least until 2006. But the last developments with the increased biofuel crop production (Busch, 2006) show that such predictions of land use trends can be highly uncertain even for short times. We considered some of the trends listed in Table 3 because at the time of our studies no reliable information about the biofuel crop production in the study area was available.

Various options are possible for developing land use related measures to reduce nutrient pollution. In a first step, we made a distinction between the following three types of measures: (a) reduction of arable land, (b) extensification of land management and (c) renaturation measures. The measures and changes implemented in one scenario were consecutively integrated in the following scenarios to finally simulate the necessary reduction of nutrient inputs into the ecosystem.

(a) Reduction of arable land
We assumed that a reduction of arable land in the basin (currently 77.2% of the basin) could be considered as an important measure to reduce the nutrient inputs into the ecosystem. In the scenarios we decreased the proportion of arable land stepwise to the benefit of pasture and forest.

(b) Extensification of land management
This type of measure was focused on alternative land management practices as suggested by the landscape program of the State of North Rhine Westphalia and the support of agro-environmental measures by CAP. The measures include the regulation of land use and land management practices that takes so-called “good agricultural practices” (GAP) as the reference, as defined in the “Bundesnaturschutzgesetz” (Nature Conservation Law), the “Bundesbodenschutzgesetz” (Soil Protection Law) and regulations at federal and state level referring to, e.g. fertilizer application rates and crop protection (Rossing et al., 2007). The land use configuration was not changed in these scenarios. Extensification measures included:
- reduction of livestock units per hectare on pasture land (one livestock unit corresponds to 500 kg live weight per hectare),
- reduction of the applied amounts of mineral and organic fertilizer applications,
- application of conservational and eco-farming practices.

(c) Renaturation measures
In the past several meanders have been artificially cut-off and the river has been heavily regulated for flood protection and to use the floodplains for agricultural production. These measures are suggested by the floodplain program (floodplain protection program) for the Ems River (StUA, 2006). Several weirs have been built to control the discharge. Currently, some river sections are under reconstruction. We implemented the following renaturation measures in the floodplains in the last scenarios:
- reduction of livestock units per hectare on pasture land in floodplains or no land management in floodplains, respectively,
- implementation of filter strips/buffer zones,
- reconnection of oxbow lakes to the channel network.

Implementation and preparation of the scenarios

Starting from the simulation of the current conditions, eight further land use scenarios were calculated with SWAT. Table 4 summarizes the agro-environmental measures that we considered in the model. They were developed successively in direction of a target scenario that finally would come close to the water quality objective of the WFD. The areas for the necessary land use changes were mainly chosen by catchment characteristics (permeability of the soils, groundwater table) and the degree of human impairment (river channel regulation, nutrient leaching).

In order to implement scenarios with reduced arable land, it was necessary to create new land use maps by using GIS operations. For scenario 1, we reduced arable land from 77.2% to 64.4% to the benefit of pasture (3.9–16.5%). Therefore, floodplains with typical alluvial soils currently used by agriculture were selected and the use was converted to pasture.

For scenario 2 an extensification measure of pasture was simulated by reducing the livestock units from 2.6 to 1.4 per hectare as suggested by the landscape development program (MUNLV, 2001) with the corresponding reduction in the amount of manure. Moreover, and in contrast to the conventional pasture management, no additional mineral fertilizer was applied.

Table 3
Land use types with increasing and decreasing land demand (due to the landscape program of North Rhine Westphalia 2002 and specific support programs for rural areas of 2003)

Land use types with increasing land demand	Land use types with decreasing land demand
Settlement and traffic areas Recreation areas Lakes and ponds Forest Organic farming	Agricultural land (mainly arable land)

Table 4
Implementation of agro-environmental measures in the model

Scenario	Measure	Implementation in the model
1	• Increase of pasture/decrease of arable land	Modification of the land use file (Shape-files). Allocation of the land use type pasture on former arable land.
2	• Extensification of pasture	Reduction of livestock density to 1.4 livestock units per hectare by modification of the management scenarios. Reduction of the amount of fertilizers.
3	• Afforestation of arable land	Modification of the land use file (Shape-files). Allocation of the land use type forest on former arable land.
4/5	• Implementations of conservation tillage practices	No ploughing, only cultivator and harrow (reduction of tillage depth), reduction of mixing efficiency.
	• Extensification of arable land	Reduction of the amount of applied mineral fertilizers accordant to 0.7 livestock units.
	• Modification of crop rotation schemes	Implementation of complex management scenarios over several years with short fallow periods.
6	• Oxbow reconnection (improvement of river morphology)	Increase of the river length on the HRU level. Modification of Manning's roughness coefficient "*n*" (CH_N(2)) for main channel flow.
	• Riparian buffer strips	Modification of the parameter FILTERW.
7	• Abandonment of the floodplain use	Conversion of arable land and pasture in riparian zones to areas without management. Change of floodplain pasture into wetland (according to soil and groundwater conditions).
8	• Increase of pasture/decrease of arable land	Conversion of arable land to pasture (randomly chosen).

Table 5
Developed land use scenarios to finally receive the required water quality situation of the WFD

Number	Scenario		Effect on						
			Water balance				Nitrogen components		
			$Runoff_{tot}$ (mm/year)	$Runoff_{surf}$ (mm/year)	Base flow (mm/year)	ET_a (mm/year)	N_{tot} (mg/l)	Nitrate (mg/l)	Ammonium (mg/l)
0	Current conditions Arable (conv.) Pasture Forest Urban Others	 77.2% 3.9% 9.9% 8.9% 0.1%	387.8	121.6	266.2	400.7	**5.98**	5.41	0.21
1	Land use change I Arable (conv.) Pasture	 77.0% → 64.4% 3.9% → 16.5%	372.5 *−3.9%*	104.8 *−13.8%*	267.7 *+0.6%*	413.4 *+3.2%*	**5.89** ***−1.5%***	5.4 *−0.7%*	0.19 *−10.5%*
2	Land management change I Reduction of live stock units pasture	 2.6 → 1.4	372.5 *0.0%*	104.8 *0.0%*	267.7 *0.0%*	413.4 *0.0%*	**5.72** ***−2.9%***	5.2 *−3.1%*	0.19 *0.0%*
3	Land use change II Arable (conv.) Forest	 64.4% → 53.2% 9.9% → 21.0%	362.2 *−2.8%*	96.5 *−7.9%*	265.7 *−0.8%*	423.8 *+2.5%*	**5.56** ***−2.7%***	5.06 *−2.7%*	0.19 *0.0%*
4	Land management change II Arable (conv.) Arable (cons.)	 53.2% → 47.0% 0% → 6.2%	365.1 *+0.8%*	89.1 *−7.7%*	276.0 *+3.8%*	420.8 *−0.7*	**5.11** ***−8.1%***	4.63 *−8.7%*	0.18 *−4.3%*
5	Land management change III Arable (conv.) Arable (cons.)	 47.0% → 40.2% 6.2% → 13.0%	370.3 *+1.4%*	84.7 *−4.9%*	285.6 *+3.6%*	414.9 *−1.4%*	**4.51** ***−11.8%***	4.06 *−12.2%*	0.16 *−7.9%*
6	River channel changes Extension of river length 10 km Riparian buffers 10 m Pasture Floodplains (not used)	 16.5% → 15.4% 0% → 1.2%	370.7 *+0.1%*	85.3 *+0.7%*	285.4 *+0.1%*	414.6 *−0.1%*	**4.48** ***−0.6%***	4.04 *−0.5%*	0.16 *0.0%*
7	Land management change IV Pasture Floodplains (not used)	 15.4% → 8.1% 1.2% → 8.5%	371.3 *+0.2%*	85.2 *−0.1%*	286.1 *+0.2%*	413.9 *−0.2%*	**4.33** ***−3.4%***	3.91 *−3.3%*	0.15 *−0.6%*
8	Land use change III Arable Pasture	 40.2% → 33.2% 8.1% → 15.2%	367.0 *−1.2%*	74.6 *−12.5%*	292.4 *+2.2%*	417.6 *+0.9*	**3.83** ***−11.5%***	3.42 *−12.5%*	0.15 *0.0%*

The effects of the scenarios on hydrological and nitrogen components (first row) are presented. Italic numbers in the second row indicate their change to the previous scenario in percent.
$Runoff_{tot}$ = total runoff; $Runoff_{surf}$ = surface runoff; ET_a = actual evapotranspiration; N_{tot} = total nitrogen.

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In scenario 3, 11.2% of the arable land was converted into forest. Areas with potential low production capacity were identified by using the soil map. Where applicable, these soils were converted from agriculture to forest evergreen.

For scenarios 4 and 5, the management practices were changed from conventional farming to eco-farming practices on selected arable land. This was done for 6.2% of the arable land in the first step (scenario 4) and another 6.8% in the second step (scenario 5). The widespread distribution of poor sandy soils requires the application of huge amounts of fertilizers to achieve a reasonable agricultural production. Because this situation would not meet the requirements of organic farming, it was necessary to choose areas in the watershed with more fertile soils and comparatively low sand contents. In these scenarios we: (i) reduced the amount of applied mineral fertilizers accordant to 0.7 livestock units, (ii) implemented soil conservational tillage practices (no ploughing, only cultivator and harrow (reduction of tillage depth), reduction of mixing efficiency, and (iii) changed the management with a focus on reduced time of bare soil (3-year crop rotation and intercropping) (Frede and Dabbert, 1999).

In scenario 6, renaturation measures such as the reconnection of oxbow lakes were implemented, and filter strips and buffer zones were created. In order to simulate the reconnection of oxbow lakes to the channel network, the river length was increased about 10 km. Due to the modified river morphology, the roughness of the river bed was increased by changing the Manning's roughness coefficient "*n*" for the main channel (parameter CH_N(2) in SWAT; Neitsch et al., 2002) from 0.044 to 0.06. Buffer zones around the river were simulated by increasing the width of edge-of-field filter strip (parameter FILTERW in SWAT; Neitsch et al., 2002) from 0 to 10 m in the corresponding HRU files. The floodplain areas in scenario 6 remained in pasture use. In scenario 7, the floodplain areas (over alluvial soil types) at the border of the river network were taken out of management. To simulate this, the pasture land use type was converted to wetland with its corresponding default plant parameters in the SWAT database.

The area of pasture land was crucially reduced in the former scenario. In the final scenario arable land was converted to pasture (sites were chosen randomly).

Results and discussion

Table 5 shows the simulation results of all scenarios for hydrological and nitrogen components as well as their change to the previous scenario in percent. Total nitrogen values are highlighted in the table because they were used as main indicator for water quality. The simulation results have been used for cost assessment of selected management measures (Volk et al., 2008). It should be emphasised here that the final scenario can give only an impression on how intensively land management must be changed in order to achieve the water quality targets of the WFD. It cannot be considered as recommendation for spatially explicit implementation of measures in reality.

At first sight, the most effective measures to reduce the total nitrogen concentrations are changes in management practices such as in scenarios 4 and 5 (conventional farming to eco-farming practices). But also scenario 8, representing a land use change measure, has a large effect on nutrient reductions. Notice also that similar

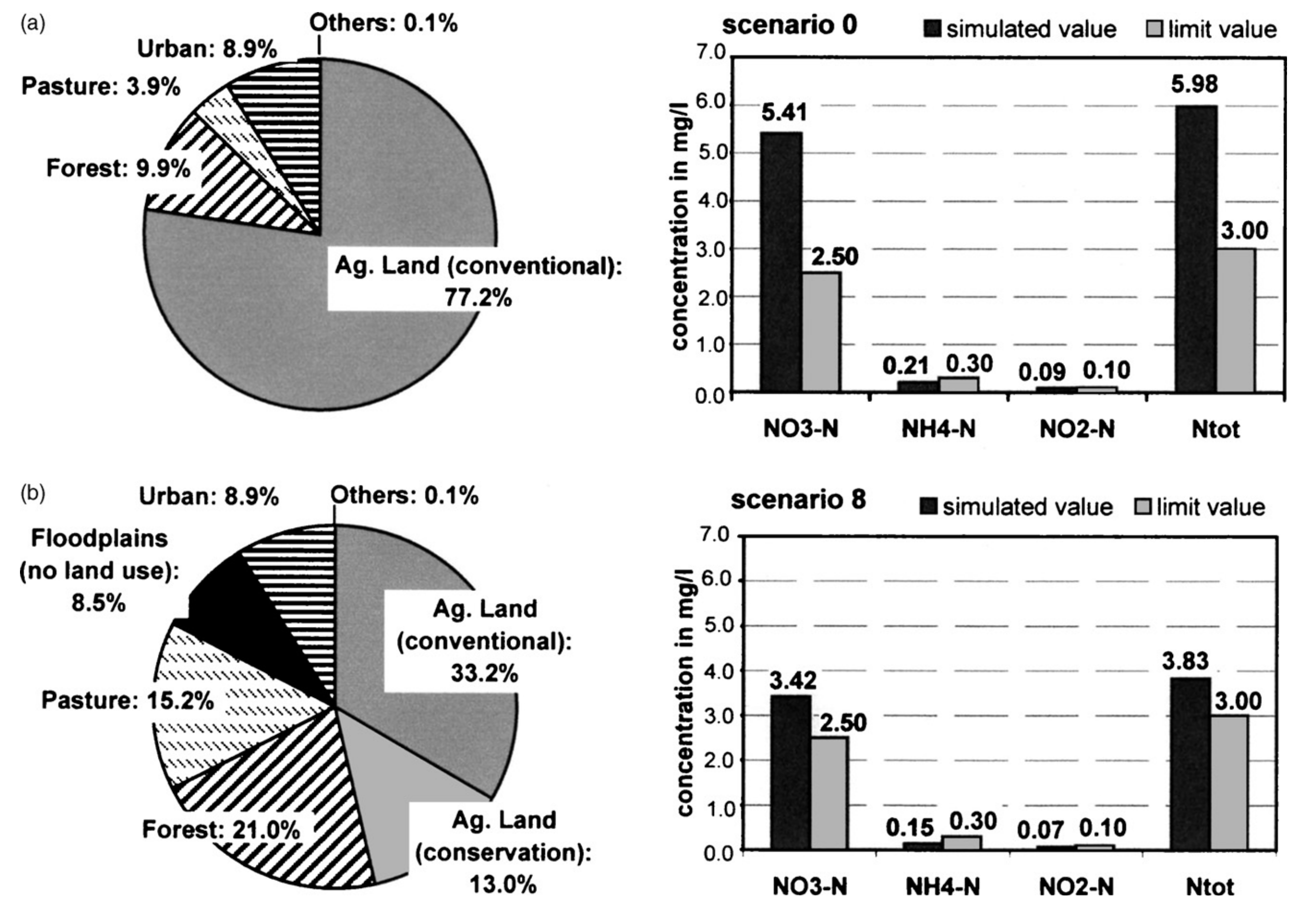

Fig. 2. Current land use distribution and nitrogen concentrations (a) and scenario 8 (b) with the land use distribution and simulated nitrogen concentrations to achieve the good ecological status for the Upper Ems River. N_{tot} values include 0.26 mg organic nitrogen for (a) and 0.2 mg/l organic nitrogen for (b).

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measures (arable land to pasture) represented by scenario 1 and 8 resulted in widely varying effectiveness. In scenario 1 the total area of pasture was increased by 13% compared to 7.1% in scenario 8, but the reduction effect for total-N and nitrate was much higher in scenario 8. An explanation for these discrepancies could be that in scenario 1 the nutrient inputs in the entire catchment are still very high and exceed critical thresholds, which limits the effect of reduced nutrient inputs. In scenario 8, however, nutrient inputs probably fall below a critical catchment threshold and thus increase the effect of the measure. This would also explain the relative inefficiency of the measures implemented in scenarios 1–3.

The impact of renaturation measures on the simulated water quality, represented by scenario 6, can almost be neglected, although one might expect a larger influence from implementing riparian zone and river channel renaturation measures. Generally, complex processes of accumulation and decomposition of nutrients in riparian zones influence the nutrient outputs out of these buffer zones, but the simulation of riparian zone impacts on river water quality is represented by a simple function in SWAT2000.

Another topic to be stressed here is the cause and effect delay between catchment response to implemented measures in the model and in reality. In the model the impact of measures takes effect immediately because the scenarios always start in the same year with same initial conditions. On the one hand this is necessary to compare the results of different scenarios, but it is far from reality. Depending on catchment characteristics, such as permeability of soils and initial nutrient loads, the impact of actual land use and management changes will usually be delayed. This delay may be many years – which represents another problem for the implementation of the WFD.

In order to achieve the good ecological status for nitrogen at the Upper Ems River, the nitrogen concentration has to be reduced by 50% of the mean annual average. This would require substantial, expensive changes of land use and land management intensity as well as of the river morphology. Fig. 2a and b shows the current conditions and the target scenario (scenario 8) that comes closest to the requirements of the water quality class II for nitrogen.

In addition to river channel changes, this scenario includes a general reduction of the arable land and the fields with conventional management, an implementation of conservation management on 13% of the agricultural land, afforestation, an increase of pasture, and conversion of floodplain land uses to buffer zones. However, this is not realistic from an economic point of view, since the drastic cuts for the farmers would be so strong that most of them would have to give up their farms. Agro-economic calculations have shown, for instance, that the mentioned changes in the floodplains alone would cost around between 500 and 800 € per hectare per year (31.6 million euro per year) (Volk et al., 2008) depending on regional soil qualities and management intensities. This measure is expected to result in intense conflicts with affected farmers (Volk et al., 2008).

Finally, we cannot predict if any measures will be implemented in the future because of increased biofuel crop production (Busch, 2006). This increase is expected to substantially affect land use patterns in Europe and thus also control the implementation of land use related measures to improve water quality. "The European Commission esteems that the measures provided for by the (biomass) Action Plan (CEC, 2005) shall lead to an increase in the use of biomass (solid biomass, biogas, biofuels, renewable municipal waste) that should reach approximately 150 million tons of oil equivalent (MTOE) in 2010 (55 MTOE intended for electricity production, 75 MTOE intended for production of heat and 19 MTOE intended for transport)" (EC, 2008).

Conclusions

The results have shown that SWAT is able to adequately represent general trends of water quality changes resulting from measures based on land use and management scenarios. Especially area-related measures, such as changed tillage operations, fertilizer applications, etc., can be described reasonably; however, more sensitivity analysis is required to answer the question: how detailed we have to parameterize management operations in SWAT for large area applications? In addition, measures based on linear structures (such as riparian zones) or spatially explicit measures are not represented satisfactorily and need to be improved.

Nevertheless, such modelling experiments help to better understand river system behaviour, especially identifying areas of highest diffuse pollution. Knowing these sources and hotspot areas, it is easier to identify useful measures for reducing actual nutrient loads in the river network and for achieving the "good ecological status" by the WFD. A dynamic catchment model taking into account water and nutrient processes as a function of vegetation, land use and human impacts, driven by climate conditions, can provide a very functional tool for creating a river basin management plan taking into account possible changes, which the basin could be confronted in future.

In general, the lack of long time series of water quality data with daily time step and higher spatial resolution has limited our capacity to evaluate the simulations – which represents a general problem and results in uncertainty. In addition, the existing monitoring programs for water quality in Europe are not suitable yet to deliver a sound database for the simulations (Jarvie et al., 1997; EEB and WWF, 2005; Allan et al., 2006). Main reasons for that are: (1) the high costs of the needed procedures which result in sparse water sampling (every two to five weeks), and (2) by sometimes insufficient cooperation between the relevant authorities, NGO's and research institutes. In the future, remote sensing has the potential to become a useful tool to provide information about water quality distribution in water bodies in order to overcome the lack of water quality data. Several authors have studied how space-borne remote sensing can be used for mapping of water quality in lakes; although little attention has been paid to rivers yet, Onderka and Pekárová (2008) described already a methodology how a Landsat ETM image was used to map the spatial patterns of suspended particulate matter in the Slovak portion of the Danube River.

The results of our investigations show that there is an urgent need to reduce the nonpoint nutrient inputs from agriculture within the study area. In addition, more efforts are needed to reduce emissions and, subsequently reduce atmospheric nitrogen deposition. The German Government aims therefore at a reduction of the emissions to 30% of the value of 1990 (Presse- und Informationsamt der Bundesregierung, 2004).

What we learned from the scenario simulations is that taking economical aspects into account, it will be almost impossible to achieve the environmental objectives of the WFD in our agricultural intensively used study area up to the year 2015. The results suggest that the achievement of the WFD environmental targets is only possible with a consideration of regional landscape and land use distinctions (different natural conditions, intensively used areas, areas with decreasing land use intensity, etc.), which would be more realistic. A "balanced" approach could be also taken into account where we could ask if it is possible to balance out areas of pollution with areas without any or only less pollution. However, the success of land use related measures to improve the water quality will also depend on the future increase of biofuel crop production, which generates specialized land management patterns to maximum biomass production. This could lead to conflicts between water protection and energy needs.

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Integrated nutrient transport modelling with respect to the implementation of the European WFD: The Weiße Elster Case Study, Germany#

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Abstract

The goal of the European Water Framework Directive (WFD) is to protect and enhance the status of aquatic and terrestrial ecosystems. To reach this objective an integrated methodology for the implementation of the WFD is essential. The methodology presented was developed within an interdisciplinary research project on the highly polluted 4th order Weiße Elster River basin, a large subcatchment of the Saale basin (Germany), which is part of the UNESCO-IHP HELP program. The project focuses on nutrient management in order to achieve a good ecological status of surface waters. The paper focuses on an integrated modelling of nitrogen transport and comprises combined terrestrial and in-stream transport processes. The mitigation of diffuse and point sources pollution is thereby essential to meet the environmental objectives. Land-use scenarios on both organic farming systems and best management practices were analysed and compared with different strategies to reduce point source. The results show that the possible reduction of nitrogen inputs from point sources is much lower compared to the reduction of diffuse inputs from agricultural land use. The results on in-stream nitrogen transformation show that different morphological factors influence the nitrogen retention considerably. The potential of management measures to reduce nitrogen loads by river restoration measures seems to be limited. This is caused by infrastructural facilities that restrict attaining a natural state of river morphology.

Keywords: river basin management, nutrient transport, river restoration, SWAT, WASP

Introduction

The European Water Framework Directive (WFD) states the goal that all waters in the European Union should reach a good status by 2015 (European Parliament, 2000). In order to achieve this goal the member states need to set up river basin districts, each one having a management plan that includes a program of measures which will achieve good status in the most cost-effective manner. This involves an evaluation of different policy measures, both with respect to the effects of nutrient reducing measures as well as its economic consequences, upon which policy makers can base their decisions. The overall objective of the case study is to develop a decision support methodology for the implementation of the Programmes of Measures (PoM) according to the WFD with special focus on the impact analysis including nutrient reduction (contribution to the environmental objectives) and economic analysis (costs of the measure). The methodological approach for decision support structures the implementation of the PoM into six phases (see Fig. 1).

The paper concentrates on the impact analysis comprising the evaluation of management scenarios to reduce nitrogen loadings using the Weiße Elster catchment as a case study. The catchment is part of the Elbe basin in Germany. The scenario analysis is carried out using appropriate hydrological nutrient transport and river water quality models. Specific objectives are to:

- Assess the impact of different agricultural management practices on the reduction of nitrogen yield for different baseline and management scenarios using the Soil Water Assessment Tool (SWAT)
- Evaluate the importance of different river restoration measures with respect to nutrient transport and assess model uncertainties using the WASP5 river water quality model.

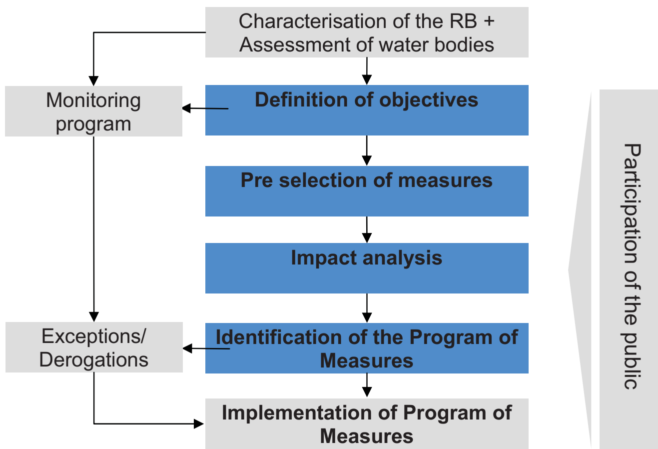

Figure 1
Structure of the Program of Measures (PoM) with steps in the planning process (RB = river basin)

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Methods

Case study

Catchment characteristics

The Weiße Elster River basin is a subcatchment of the Saale River which is the second largest tributary of the Elbe River. The catchment area is about 5 300 km² and is mainly situated in the German States of Sachsen (Saxony), Thüringen (Thuringia) and Sachsen-Anhalt (Saxony-Anhalt). The river originates from the Erzgebirge (Ore Mountains) in the Czech Republic, is 250 km long and has a mean discharge of 26 m³/s (gauging station Oberthau). The river channel structure is very diverse with near-natural stretches as well as concrete-lined segments.

The land use (Fig. 2) in the basin is dominated by agricultural activities (43% cropland, 16% pasture) especially in the lower part, and forest (21%), mainly in the upper part. The upper part of the basin is mountainous characterised by steep slopes and narrow valleys with hardly any floodplains and scarce groundwater resources. The geology is characterised by igneous and metamorphic rocks and consolidated tertiary rocks (sandstone). The lower part of the basin is situated in the lowlands and mainly consists of Pleistocene coverage. Precipitation varies between 500 mm in the northern part of the basin (lowlands) to 1 000 mm in the southern part (mountains). The annual runoff varies approximately between 50 and 600 mm.

Main field crops are grains and root crops in the northern part of the catchment and forage crops in the southern more hilly parts of the basin. Livestock in the area consists mainly of cattle. Settlements, industrial areas, and infrastructure account for 16% of the land use with Leipzig and Halle being major cities located in the catchment. The area south-west of Leipzig is characterised by active and reclaimed open pit mines. The implementation of the WFD for the Weiße Elster River is coordinated by the Saale Basin Co-ordination Group which is formed by representatives of the State ministries of environment (Thuringia, Saxony, and Saxony – Anhalt, Bavaria and Lower Saxony).

Meteorological data were made available from the German national meteorological service. There are about 60 precipitation and 11 climate stations in and around the Weiße Elster basin. Daily data were made available for most of the precipitation gauging stations, while six-hourly or hourly data were available for the climate stations. Time series data were collected from 1990 to 2003. Daily water level measurements were made available for about 20 gauging stations in the Weiße Elster catchment. These stations are managed by the environmental agencies of the Federal States. Time series of water level and discharge data were used from 1990 to 2002. There are about 20 water quality monitoring stations in the Weiße Elster catchment. But measurements were taken only 1 to 2 times per month. An extensive number of physico-chemical properties were measured. Data of water extraction and discharge to the river were mostly taken as permitted values.

A digital elevation model was made available at 50 m resolution. Land use information was derived from Landsat imagery at a spatial resolution of 30 m for 1989 and 1999. Furthermore a detailed biotope map derived from aerial photography was used. Several soil maps were made available in digital format with spatial resolutions of 1:1 000 000 and 1: 200 000.

Description of problems

Water quantity and quality are closely related to the various economic activities in the river basin. Agriculture, urbanisation as well as open pit mining have contributed to the chemical and

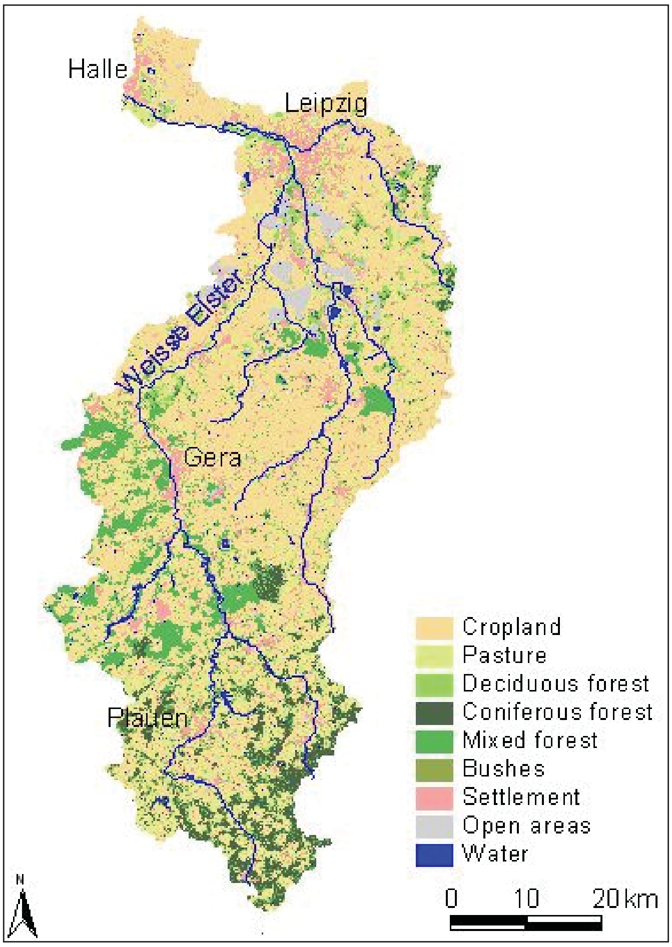

Figure 2
Land use in the Weiße Elster Catchment

biological pollution of the Weiße Elster River. Most of the Weiße Elster River has been classified as moderately polluted with some segments as critically polluted (classification according to German water quality standards). Main problems are nutrients (N & P) with high ammonium concentrations in some river reaches. Also, salt concentrations (esp. sulphate) are quite high due to open-pit mining and other industrial activities. However, the sulphate loads are not considered to be ecologically relevant. It is not yet clear whether the discontinuation of mining activities and flooding of open-pit mines will alter salt concentrations of the Weiße Elster. Although water quality has substantially improved in the last few years, a comprehensive remediation programme is necessary to meet the WFD targets.

Nutrient concentrations in the Weiße Elster River and its major tributaries given as 90-percentile of concentrations (2001) range between 6.1 and 13.0 mgN/ℓ and 0.14 and 0.74 mgP/ℓ. Diffuse sources have been estimated to have contributed to the overall nutrient load by 84% (nitrogen) and 65% (phosphorus). High NH4 and PO4 concentrations at the lowland river reaches are caused by high sewage inputs from urban areas. Main water uses conflicting with water quality targets according to WFD are:

- Agriculture vs. water quality: agriculture contributes to a great extent to diffuse pollution load. This load conflicts with ecosystem restoration and impairs drinking water quality.
- Industrial and municipal discharge vs. water quality: other important reasons for the poor water quality of the Weiße Elster are point pollution from industrial and municipal effluents, considerable water abstraction for industrial use and flooding of mining lakes.

Model set-up

The evaluation of management measures is carried out with regard to improvement of water quality. Two models were applied: the integrated Soil and Water Assessment Tool (SWAT) which simulates the water balance and nitrogen transport, and the Water Quality Analysis Simulation Program (WASP5) which simulates the hydrodynamic and in-stream transformation processes. Water and nitrogen transport of the entire Weiße Elster basin is simulated with the Soil and Water Assessment Tool (SWAT). SWAT is a deterministic continuous process-based model coupling hydrological, biogeochemical, and ecological processes at the river basin-scale (Arnold et al., 1994; Krysanova and Haberlandt, 2002). According to the stepwise approach the simulation first focuses on the hydrological cycle and secondly on the transport of nitrogen.

The Weiße Elster catchment was divided into 108 subcatchments based on the hydrological characteristics. SWAT model calibration and validation was carried out using time series data from 1991 to 2000. After parameter sensitivity evaluation, the calibration was carried out manually using the most important parameters. A detailed sensitivity analysis of the SWAT model can be found in Van Griensven et al. (2002). The calibration runs of the hydrological model were assessed by visual comparison of the simulated and the observed hydrographs and objective criteria (e.g. Pearson's Product-Moment Correlation Coefficient, the coefficient of determination, the Nash & Sutcliffe coefficient of efficiency, yearly absolute volumetric error measures). They quantify the degree of agreement between the observed and simulated values.

The water quality model WASP5 (Water Quality Analysis Simulation Program) is a one- to three-dimensional numerical model and includes a deterministic approach to describe the hydrodynamics and the turnover of nutrients and chemicals in water column and sediments. It was developed at the U.S. Environmental Protection Agency (Ambrose et al., 1993). The WASP5 modelling system consists of three stand-alone computer programs, that can be run in conjunction or separately: DYNHYD is a hydrodynamic model, which is based on the Saint Venant equations; EUTRO can be used to model oxygen depletion, eutrophication, and nutrient enrichment in the river; and TOXI simulates the sediment transport and the fate of toxic inorganic and organic chemicals. In this study a modified version of DYNHYD (Warwick, 1999) was used which allows the consideration of weirs. Also an extended version of EUTRO was applied (Shanahan and Alam, 2001), which consists of nine model variables: biomass of phytoplankton (PHYT), biomass of periphyton (PERI), dissolved oxygen (DO), biochemical oxygen demand (BOD), ammonia nitrogen (NH_4), nitrate nitrogen (NO_3), organic nitrogen (ON), phosphate (PO_4) and organic phosphorus (OP). The complex system of these variables is described by several processes, such as growth and decay of the autographs, settling, re-aeration, sediment oxygen demand, nitrification, denitrification and mineralisation. In total up to 59 temperature coefficients and kinetic parameters are used. Additional information on latest model modification can be found in Wagenschein and Rode (2008). The main advantage of WASP5 compared to other water quality models is its flexibility as it offers a possibility to build one-, two- or three-dimensional networks. Complex aquatic systems can be subdivided into lateral, vertical and longitudinal segments. Another advantage is the freely available source code of WASP5, which makes it possible to implement additional processes and components in the modelling system.

The Weiße Elster River water quality model set-up consists of 872 river cross-sections. Uncertainty analysis based on the Monte Carlo approach was carried out for the calibrated model. Discharge and nutrient load input data were obtained from the water authorities and additional measurement campaigns. Point source data from sewage systems were directly used as inputs into the WASP5 model for the Weiße Elster River. Frequentist and Bayesian techniques are the most common methods for model or parameter identification (Omlin and Reichert, 1999). In this study frequentist analysis was used as much less time was needed. It comprises two steps:

- Assessment of parameter identifiability by sensitivity analysis and calculation of compensation measures (Reichert and Vanrolleghem, 2001)
- Calibration of the WASP5 model with the automatic parameter estimation tool PEST (Doherty, 2004) using the 8 most important parameters (see also Wagenschein and Rode, 2008).

All other parameters were defined using literature values. A detailed sensitivity analysis can be found in Wagenschein (2006). During the calibration process, PEST allows quantification of 95% parameter confidence limits based on the solution of the covariance-matrix.

As criteria to measure the model performance the Index of Agreement *d* (Willmott, 1982) and the Nash-Suttcliffe-criteria *E* (Nash and Suttcliffe, 1970) were used. The *d* includes values of between 0 and 1, with values close to 1 indicating a good agreement of the model results to the measured data. The coefficient of efficiency (E) was selected because it is dimensionless and is easily interpreted. When the measured variable is simulated exactly by the model, E equals 1. If $E < 0$, the predictive precision of the model is lower than when the mean of the values measured is used.

Scenario analysis

The validated SWAT model provided the basis for the analysis of the status quo and land-use management scenarios for the reduction of nitrogen inputs in the Weiße Elster catchment. The following land-use management measures were analysed:

- Different shares of organic farming on total arable land, scenarios amount to the shares of 5%, 10%, 20%, and 30% of organic farming
- Land-use distributions in 2010 according to the present agricultural policy expectations (RAUMIS 2010; Gömann et al., 2003)
- Land-use distribution in 2010 according to a liberalisation of the agricultural market (RAUMIS-TLB; Gömann et al., 2003).

Additionally the effect of three river restoration scenarios on the nutrient concentration is investigated:

- River restoration Scenario 1 analyses measures of the river maintenance program of the water authorities on a 57.3 km river reach in the lower part of the Weiße Elster River. It consists of local extensions of river width, local increase of river bottom roughness, additional shadowing by riparian vegetation and the removal of one weir.
- River restoration Scenario 2 assumes natural morphological conditions in all reaches of the river, which are not restricted by roads, railways or urban areas. These unrestricted reaches comprise 37.5% of the study river section. Natural conditions were defined using morphological parameters of

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undisturbed reference sites. Natural sites were defined by a sinuosity value of 1.3 for river section upstream of the City of Zeitz and 1.7 for river sections downstream of Zeitz. Reaches with modified river structure show lower values due to straightening of river meanders. Hence, this scenario leads to higher sinuosity in the study reaches and an increase of flow length of 16.4%. The river cross-sections in natural conditions are defined by a mean width of 32 m and a mean depth of 1.1 m for the upper part of the river section upstream of Zeitz. Downstream of Zeitz the mean width of 24 m and the mean depth of 1.5 m was used. Mannings coefficient was taken as 0.033 for the upper part and 0.035 for the lower part of the river reach according to Pottgiesser and Halle (2004).

- River restoration Scenario 3 is a hypothetical scenario, which assumes morphological reference conditions for the whole river section. Natural mean values of sinuosity, w/d-ratio and Mannings coefficient were defined according to Scenario 2. Additionally, all weirs were removed from the hydrodynamic model and constant channel slopes were assumed for these river reaches.

The land use RAUMIS 2010 and RAUMIS-TLB scenarios reflect baseline conditions under different agricultural policy conditions. The RAUMIS model is an agricultural market model which is able to consider the impact of the world market as well as the European agricultural market on the agricultural sector in Germany. RAUMIS allows calculating crop rotations, the share of arable land and pasture and associated crop yields as well as livestock in the county level (Gömann et al., 2003). Main differences between these scenarios compared to status quo are a moderate reduction of 9% (RAUMIS 2010) to an extreme reduction of 43% of agricultural land use (RAUMIS-TLB) in the Weiße Elster catchment. The county level land use data have been disaggregated on the 50 meter raster level of the original land use map according to potential crop yields of the arable land (Marks et al., 1992). Potential crop yield was calculated according to the site characteristics, such as soil, relief, water balance, climate and erosion risk. Agricultural areas with low potential crop yield are assumed to convert to fallow. Only areas with a minimum size of 1 ha were considered. The new land use maps of the two baseline scenarios have been used as input for the SWAT scenarios simulation.

Simulation of different shares of conventional and organic farming on total agricultural land use in the catchment is based on randomly distributed changes in land use. Organic farming is represented in the SWAT model by modifying crop rotations and fertiliser application. SWAT simulates the changes in nitrogen loads according to the baseline and management scenarios for every of the 108 subcatchments in the Weiße Elster basin. The scenario analysis is carried out using time series data from 1976-2000 for the calculation of long-term mean yearly nitrogen loads.

The effect of river morphology on the nitrogen retention was analysed in a first step by sensitivity analyses with the WASP5 model. The Mannings coefficient (as a measure for river bottom roughness), the width/depth-ratio (w/d-ratio) and the sinuosity were sequentially varied by ±10%. The effect on the model output variables is expressed by the elasticity ε, which can be approximated by:

$$\varepsilon \approx \frac{\Delta x}{\Delta\Theta}\frac{\Theta}{x} = \frac{x_i - x_0}{((1+p/100)\Theta - \Theta}\frac{\Theta}{x_0} = \frac{x_i - x_0}{p/100 x_0}$$

where:

x_0 and x_i are the output variables before and after variation of parameter Θ by the value p (10%)

The influence of the morphological parameters "Mannings n", width/depth ratio and sinuosity on nitrogen concentrations were compared with the influence on other model variables like organic phosphorus (OP), ortho phosphorus (PO_4-P), oxygen (O_2), phytoplancton (PHYT), periphyton (PERI), and biological oxygen demand (BOD). Elasticity ε of the selected morphological parameters was determined for each model variable.

Results

Model calibration

Calibration of the SWAT model shows good results for most of the discharge gauge stations with Nash and Sutcliffe values between 0.5 and 0.8. Lowest values of efficiency for model calibration are achieved in subcatchments with considerable impact of open pit mining. Due to these limitations the visual comparison of model predictions and observations is important. An example of the results of the hydrological model calibration is given in Fig. 3 for gauging station Zeitz with a catchment size of 2 476 km² and a Nash and Sutcliff coefficient of 0.7. Also the visual inspection shows a good agreement of measured and simulated discharge.

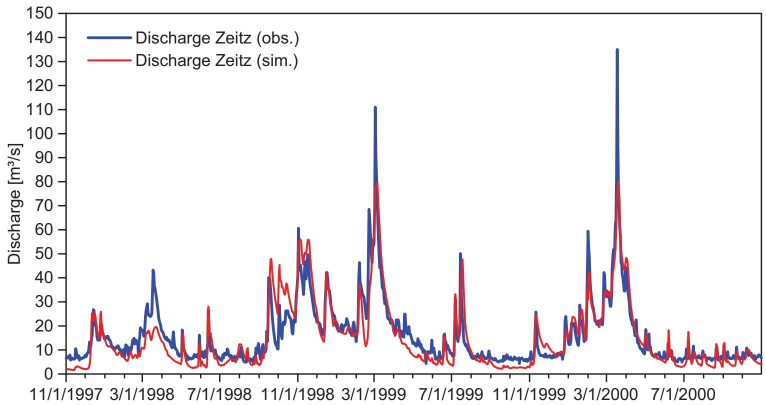

Figure 3

Measured and simulated daily discharge of the Weiße Elster River at gauging station Zeitz (calibration mode)

TABLE 1
Index of Agreement (*d*) and Nash-Suttcliffe-Criteria (*E*) for the calibration runs (after Wagenschein and Rode, 2008)

Criteria	Number of values n	PHYT	PERI	DO	CBOD	NH_4	NO_3	ON	PO_4	OP
d	14 (*8)	0.98	0.72*	0.99	0.99	0.87	0.99	0.97	0.98	0.95
E	14 (*8)	0.86	-1.43*	0.43	0.86	0.30	0.89	0.17	0.61	0.60

The calibration of the hydrological model serves as basis for the calibration of the nitrogen model. The calibration results for nitrogen loads of a 10-year period (1989-1998) are reasonable with Nash and Sutcliffe values of 0.47 for gauging station Läwitz (100 km²) and 0.48 for gauging station Gera-Langenberg (2 167 km²). River water quality simulations carried out with the WASP5 model are shown in Table 1 for selected model variables. Nitrate nitrogen is simulated in the calibration run with a d-value of 0.99 and an E-value of 0.89. In contrast, the fit of ammonia nitrogen is unsatisfactory. This is caused by low concentrations during the observed time periods, which amounts less than 0.03 mg/ℓ. While simulated phytoplankton values show a good agreement with measured values, periphyton biomass shows larger errors. Due to the large variability of periphyton biomass the small number of periphyton measurements does not represent the mean value of a model segment. Figure 4 shows the calibrated chlorophyll-a concentrations for the selected river reach of the Weiße Elster for the 2nd September 2003 and 3rd August 2004. Measured and simulated chlorophyll-a concentrations show a good agreement.

The calibrated model was validated for a half-year period in 2001. The results were compared with the data of the monitoring programme conducted by the official water authorities. For nitrate nitrogen the modelling results are reasonable with NS value of 0.56, although measured values are slightly underestimated. For ammonia there are larger deviations with a NS value of 0.14, which are mainly caused by temporal variability of the ammonia inputs by sewage plants. Measured phytoplankton concentrations are well represented by the measured values. Inorganic phosphorus is also slightly underestimated by the model.

Analyses of land use scenarios and agricultural management measures

The yearly nitrogen loads from the scenario analyses for the time series from 1991-2000 at the discharge gauging station Gera-Langenberg in the middle part of the Weiße Elster catchment is shown in Fig. 5. The figure shows only small differences between the selected shares of organic farming, the baseline scenario business as usual (RAUMIS 2010) and the status quo. All scenarios lead only to a small reduction in the nitrogen load with an increased share of organic farming at total agricultural land. However, even a share of 30% organic farming does not reduce the nitrogen load substantially. Reduced nitrogen inputs due to organic farming do not always lower nitrogen leaching from soil zone to the same extend compared with conventional farming. This can be explained by considerable lower crop yield and plant uptake of nitrogen. The baseline scenario regarding the liberalisation of the agricultural market (RAUMIS-TLB) leads to a considerable reduction of the nitrogen load in the Weiße Elster catchment. This is caused by a large reduction of arable land of 34% and pasture of 8.6% compared to the business as usual RAUMIS 2010 scenario. For both scenarios, the area weighted (kg/ha•a) reduction of the long-term mean nitrogen load is expressed as deviations from the status-quo scenario. There are large differences between both scenarios and a large spatial variation of nitrogen load reduction in each scenario with a maximum of 21 kg/ha•a in two subcatchments after liberalisation of the agricultural market (RAUMIS-TLB). This is due to a large shift of arable land to pasture or fallow in connection with high nitrogen loads of the former arable land. The SWAT scenario for different shares of organic farming on the total agricultural land use shows a large variation of nitrogen load reduction within the Weiße Elster catchment.

Highest reduction of nitrogen loads can be observed in the upper part of the catchment. This reduction increases with an increase in organic farming. In contrast, in most lowland subcatchments, organic farming leads to a slight increase of nitrogen inputs in the river system (Fig. 6). It can be concluded that an overall increase of organic farming does not ensure a reduc-

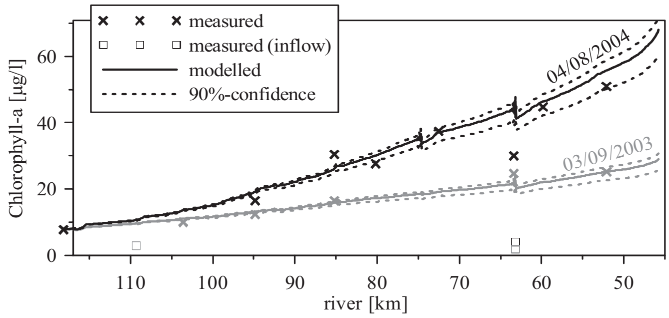

Figure 4
Measured and simulated concentrations of nitrate-N in the selected river reach of the Weiße Elster (calibration mode)

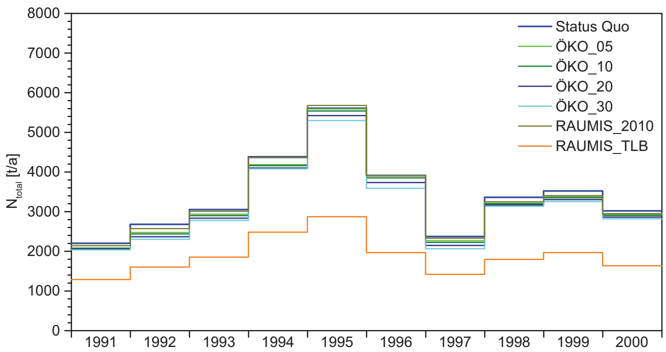

Figure 5
Nitrogen loads of SWAT scenarios at the discharge gauging station Gera-Langenberg

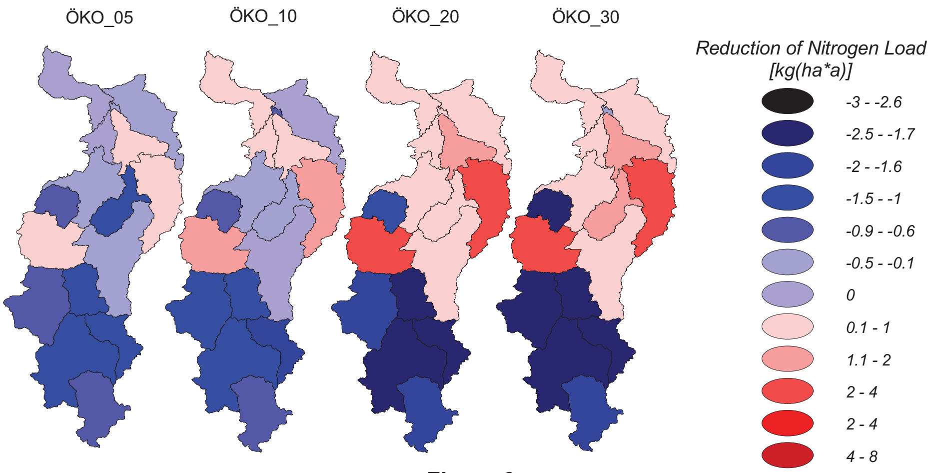

Figure 6
Nitrogen load changes of SWAT scenarios of organic farming compared with the status quo scenario

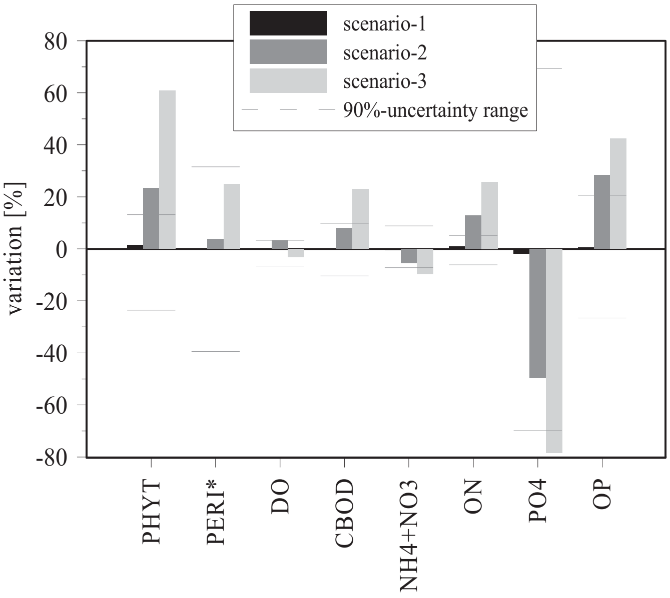

Figure 7
Percentage variation of WASP water quality variables at the outlet of the river section for three restoration scenarios (after Wagenschein and Rode, 2008)

tion of nitrogen leaching form agricultural land. Site specific characteristics like local to regional climate conditions and soil type distribution have to be taken into account. A more promising strategy seems to be an appropriate adjustment of the crop rotation at site specific locations. Of course this is also true for organic farming land-use practices.

Impact of river restoration and river morphology on nitrogen retention

The effect of the shape of the river cross section characterised by the sinuosity, the w/d-ratio and the Mannings coefficient on nitrogen concentrations was analysed by sensitivity analyses. The simulation study shows, that sinuosity has the strongest effect on all model variables. The influence of the w/d-ratio is lower and the effect of the Mannings coefficient is negligible. Furthermore, the elasticity of the model variables is varying: the effect of sinuosity is very strong for the components of autotrophic biomass. This accounts for a rise in the residence time and an increase of settling surface for periphyton. For the inorganic nitrogen the elasticity is lower because it is affected by several divergent processes. The most significant processes are that:

- The assimilation of nitrogen increases with increase in autotrophic biomass
- Nitrogen retention by benthic denitrification increases with increase in sediment surface. The elasticity of the inorganic phosphorus is comparably higher. The main reason is that its concentrations are quite low. This results in a higher sensitivity in contrast to autotrophic growth.

Three morphological scenarios were examined. They are characterised by deviating spatial intensities and combinations of restoration measures. River restoration Scenario 1 (municipal river maintenance program) has nearly no effect on the model variables (Fig. 7). River restoration Scenario 2, which assumes river channel restoration for unrestricted reaches, shows larger changes of inorganic nitrogen concentration (NH4+, NO3) compared to Scenario 1. Nitrogen concentrations decrease by 5.4% at the end of the 70.6 km river section. This can be explained by an increase of the autotrophic biomass (PHYT and PERI) and enhanced retention due to benthic denitrification.

River restoration Scenario 3 (hypothetical scenario) results in a decrease of inorganic nitrogen of 9.9%, which is caused by an increase of the water-sediment interface. The removal of the weirs results in an increase of flow velocity and a corresponding decrease of residence time. This leads to a decrease of nitrogen turnover by about 3% compared to the present state.

The reduction of inorganic phosphorus (PO_4-P) is more significant compared to nitrogen. This can mainly be attributed to the growth of phytoplankton and periphyton. As a result the organic phosphorus (OP) increases. The larger change of inorganic phosphorus in comparison to inorganic nitrogen can be explained by quite low concentrations (< 0.03 mg PO_4/ℓ) in the present state. Furthermore 90% confidence limits were often larger than the predicted effect of the measure; this was especially true for the Scenarios 1 and 2. The assessment of these results has to take into account that the baseline scenario covers a time period of two weeks of summer conditions. For winter periods the nutrient concentration changes would be smaller because of lower biomass of algae and lower denitrification rates. Hence, the mean annual changes probably would be smaller too. The effect of the most feasible measures on the concentration of inorganic nitrogen, which are realised in river restoration Scenario 2, is quite low. Generally, the positive effects of rehabilitation measures on nutrient concentrations might be larger in small rivers compared to large rivers because of the larger w/d-ratio and a more intensive exchange between the water body and the hyporheic zone.

Conclusions

Within the HELP project a methodological approach for the implementation of the program of measures of the European WFD was developed and the use of water quality models for the impact assessment was demonstrated for the 4th order Weiße Elster catchment in central Germany. From the modelling study using SWAT it can be concluded that the investigated organic farming scenarios do not ensure a considerable reduction of high nitrogen loading from agricultural land of the studied catchment. Only the scenario on liberalisation of the agricultural market leads to a considerable reduction of nitrogen loads due to large reduction of agricultural land use of 42.6%. The scenario analysis shows that sufficient reductions of nitrogen loads with respect to the ambitious goals of the European WFD can only be achieved with a considerable change of agricultural land use.

With regard to the river water quality modelling study it can be concluded that the impact of the most feasible measures on the concentration of inorganic nitrogen, which are realised in the river restoration Scenario 2, is quite low. Little effect on the yearly mean of inorganic phosphorus is also expected. The reason for that is that the autotrophic assimilation is low and the substance regimes between sediments and the water column are on average balanced, since the seasonal dependence of fixation through sorption or mobilisation by desorption or erosion is preponderant.

The parameter uncertainties are high and sometimes larger than the effect of the investigated river restoration management scenarios. The case study shows that easily applicable measures for the reduction of diffuse nutrient (especially nitrogen) loads may not be sufficient to reach the goal of a good water quality status requested by European WFD.

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Integrated ecological-economic modelling of water pollution abatement management options in the Upper Ems River Basin

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A B S T R A C T

This paper presents the results of the FLUMAGIS project, in which we developed a spatial decision support system (SDSS) to support the implementation of the European Water Framework Directive (WFD). The modelling approach is based on the integration of ecological and socio-economic assessment methods, scale-specific and GIS-based data and knowledge modelling and visualization techniques. The project study area is the intensively cropped Upper Ems River Basin in north-western Germany. A method was developed that enables the transfer of scale-specific data and information. Analyses were performed for baseline conditions and specific management and planning scenarios to improve water quantity and quality at micro-, meso- and macro-scale. The results of the study indicate that substantial, expensive water and land management changes at different scales would be necessary to achieve the WFD water quality targets in this basin. Ecological-economic analysis of cost-effectiveness reveals that the costs of achieving certain goals of the WFD can vary more than tenfold depending on which measure is chosen out of the pool of management alternatives. Moreover, the study shows that the differentiation between landscapes and other regional characteristics although considered essential to the successful implementation of WFD measures is very data intensive.

1. Introduction

The European Water Framework Directive (WFD) is designed to solve water management problems as a function of natural river basin boundaries instead of administrative borders. The focus on river basins and surface water bodies as reference units, the consideration of natural scientific as well as socio-economic aspects and an emphasis on public participation represents a paradigm shift in integrated European water management and policy (Hirschfeld et al., 2005; Jessel and Jacobs, 2005). Thus, the implementation of the WFD poses significant new challenges to water managers, planning authorities, researchers and stakeholders, increasing the demand for new Geographical Information Systems that incorporate spatial decision support systems (SDSS), simulation models, and other tools to analyze, interpret, and display spatial information for river basin planning.

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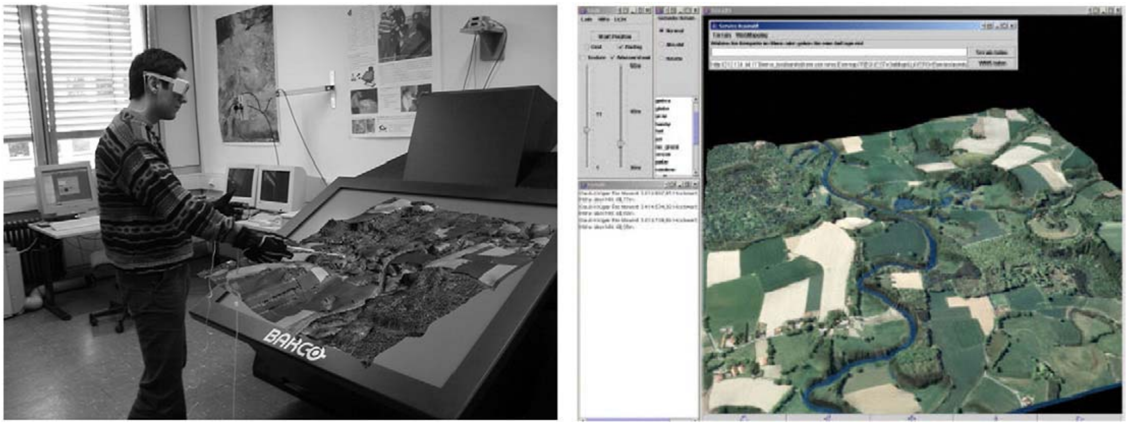

Fig. 1 – (left). Stereoscopic 3D visualisation environment (work bench). The other picture (right) shows the desktop PC environment.

To meet this increasing demand, a new interactive tool was developed in the FLUMAGIS[1] project, which supports the assessment and 3-dimensional visualization of hydrological, ecological and socio-economic conditions and management effects in river basins. Simulating virtual environments within FLUMAGIS aims to elaborate future planning and management scenarios on the basis of an interdisciplinary data and knowledge platform in accordance with the WFD. Management alternatives and effects of planning scenarios can be discussed and evaluated by scientists, landscape planners and decision makers, and potentially citizens in a participatory planning process (Fig. 1). Thus, the FLUMAGIS tool can aid the decision-making process within the WFD and similar water quality assessments.

In this paper, we present the FLUMAGIS ecological-economic modelling component, which is a key part of the overall system framework. The modelling component consists of the:

a) definition of relevant scale levels and scale transition;
b) development of land use scenarios;
c) floodplain assessment and simulation of the river basin management strategy impacts on hydrology, nutrient transport and land use; and
d) socio-economic assessment of measures to improve water quantity and quality.

Ecological and economic simulations were carried out using the models ABIMO (Glugla and Fürtig, 1997), SWAT (Arnold et al., 1998; Neitsch et al., 2002; Gassman et al., 2007), NASIM (Hydrotec, 2001) and BEMO (Kleinhanss et al., 1999). The integration of data and simulation models in FLUMAGIS is facilitated using a combination of GIS and knowledge base (KB). Knowledge-based SDSS represent the knowledge and the experience of experts for a special area of interest and combine "the ability to simulate the heuristic reasoning of experts with an explanation facility for justifying their reasoning and conclusion" (Zhu et al., 1998). Alternative management strategies are translated via defined rules and relations to scale-specific input data files (land use patterns, management scenarios) in order to be able to carry out the economic and ecological simulations. Analyses were carried out for baseline conditions and specific planning scenarios at the micro-, meso- and macro-scale within the Upper Ems river basin located in north-western Germany. The land use scenarios that were developed for the study are based on existing spatial planning, nature protection and water management programs as well as European Union Common Agricultural Policy (CAP). The scenarios were designed to be scale-specific, but are at the same time also consistent with their adjacent scale levels. In addition, an actor analysis was carried out within selected case studies to evaluate different strategies of local stakeholders.

Hence, the specific objectives of the study presented here are: a) present a description of the FLUMAGIS framework and the approach used to address different spatial scale requirements (Section 2); b) provide a description of the hydrological, ecological and economic models and assessment approaches used within the FLUMAGIS framework (Section 3); and c) describe the development of scenarios and the simulation results at different scale levels (Section 4). The conclusions of the study are presented in Section 5.

2. The FLUMAGIS approach

2.1. *Model linkage and knowledge base*

In the WFD, the assessment of the actual state of water quality in river systems is mainly based on biogeochemical and hydro-morphological indicators. The living conditions for organisms in a river system depend on the a-biotic environment that is heavily influenced by land use activities. Thus, the development of management plans requires an integration of hydrological, biological, chemical and socio-economic aspects. The FLUMAGIS approach integrates simulation models from different disciplines to evaluate river basin management options and to forecast their effects on water quality, habitat conditions and socio-economic sectors (Fig. 2).

The starting point of the FLUMAGIS approach is an assessment of the current ecological status of the river system —

[1] FLUMAGIS is an acronym for "Interdisziplinäre Entwicklung von Methoden und Werkzeugen für das Flusseinzugsgebietsmanagement mit Geoinformationssystemen" (Interdisciplinary development of methods and tools for the planning process and measurement control for river basin management with geo-information systems) (see http://www.flumagis.de/english/e_index.htm).

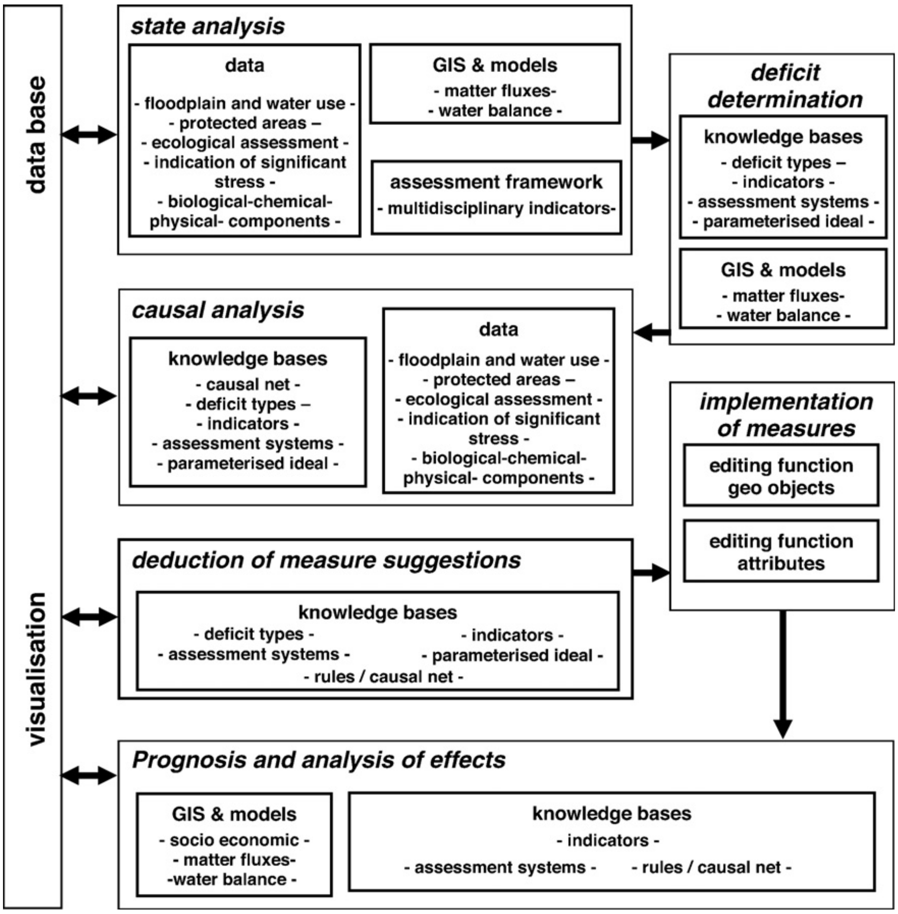

Fig. 2 – FLUMAGIS-functionality in relation to system components.

drawing on biological, chemical and physical indicators. A deficit analysis identifies the degradation of the river system relative to a "good ecological status" reference scenario defined by the WFD. A causal analysis further provides relevant information about the reasons and main determinants underlying the actual deviation from the WFD target conditions. In the next step, the FLUMAGIS system proposes a range of potential management options to improve the ecological status, which can then be chosen and "implemented". Finally, the ecological and socio-economic effects of the chosen measures are simulated by drawing on the interlinked models. The ecological impacts of these simulations can then be re-evaluated. If the selected management strategy does not yet meet the "good ecological status", the steps in the analysis can be repeated until the desired outcome is achieved.

The ecological and economic models are linked in FLUMAGIS via a GIS and a knowledge base (KB), integrating expert knowledge to allow the investigation of different scenarios before decision-making.

Data transfer between the ecological and economic models is organised by spatial units. These units are mainly determined by land use classes (CORINE land cover and data from the Authoritative Topographic-Cartographic Information System ATKIS) intersected with spatial planning units (municipalities, counties, subbasins, etc.). Economic impacts resulting from changes in agricultural land management (e.g., alternative cropping systems, tillage operations, fertilizer application practices, and livestock densities) act as new input data for the ecological modelling. On the other hand, ecological objectives and land use scenarios deduced from the ecological deficit analysis serve as input information for the economic model.

The knowledge-processing module provides both visualisation and a framework that encompasses all the other methods and models within the system. The KB includes all data, methods, rules and frame conditions that are relevant to river basin management. Software developers are not able to fill the KB with the expert knowledge from the different disciplines, hence the KB-framework has to be understood and filled by different experts. The KB is editable and stored in a formalized and machine-readable way. We have set up a causal network of different components that serves as the core of the integrated KB. We developed a concept that combines approved methods such as Coloured Petri Nets (CPN) under the ontologically structured knowledge base on the basis of a Bayesian Belief Network.

Simple inference methods can be applied with the use of causal networks to analyse the existing data and draw conclusions from it. Principally, every node of the network can serve as a user-selected starting point for analysis and prognosis. In an iterative process the linked nodes are traced either by the causes-relation (property causingNodes), or the effect-relation (property effectedNodes) — until no relations to other knots are found anymore. Depending on the type of knot further inference questions are possible (Table 1).

Ontologies represent taxonomies and categorizations of specific domains. They are organized hierarchically by the type-subtype relation and show concepts of different levels of abstraction (Sowa, 2000). Concepts are specifications of

Table 1 – Potential inference questions of the causal network

Knot type	Inference question	Property
Action knot	What are the consequences of a measure?	effectedNodes
	What is the aim of the measure?	aimedNodes
Index knot	What are the causes of an environmental condition or problem?	causingNodes
	What are the effects of the environmental condition or problem?	effectedNodes
– Additionally		
Purpose knot	By which measures can we achieve the aim?	aimingNodes
Deficit knot	By which measures can we remove the deficit?	requiredActionNode
Potential knot	By which measures can we implement the potential?	requiredActionNode

entities including attributes and relationships. An important advantage of ontology-based systems can be found in an increased interoperability and reusability by using a semantic reference system (Kuhn and Raubal, 2003).

We have chosen Protégé as a KB-platform because it is an open extendable system. Basically Protégé serves as a knowledge editor to create domain ontologies using the frame logic (Noy et al., 2000). In frame logic, rules, axioms and constraints can be seen as predicates of the concepts they aim at, so they follow an object-oriented approach and can be accessed more efficiently compared to rule-based systems. After having developed some extensions we are able to access databases that contain data for evaluations and analysis provided by other extensions and models (Borchert, 2003).

2.2. A scale-specific method for integrated ecological-economic modelling

The WFD defines only one scale level — the so-called report scale (1:500,000). For the implementation of water protection measures and their efficiency control this is not sufficient. Successful achievement of the WFD objectives requires the use of scale-specific tools for the investigation and visualisation of the ecological situation in river basins and the effects of water protection measures. Scale-appropriate simulation of nutrient fluxes and balances is necessary, because structures, functions and processes change with scale (Blöschl and Sivapalan, 1995; Steinhardt and Volk, 2003; Quinn, 2004; Jessel and Jacobs, 2005; Hein et al., 2006). On the basis of these experiences and the existing scale recommendations of spatial planning, water management, landscape ecology, and nature protection regulations, we propose three spatial scales (micro-, meso- and macro-scale) for adequately describing water and matter balances as well as conducting economic assessments (see Section 3.3). Furthermore, we developed a transferable method that supports scale-specific analyses. The procedure is able to determine the scale-specific applicability of different models and assessment systems. Detailed information describing the underlying theory of this approach is given in Steinhardt and Volk (2003) and Volk and Schmidt (2004).

The scale levels incorporated in the FLUMAGIS system (Fig. 3) include: a) the WFD report scale (macro-scale, 1:500,000) for calculating magnitudes of water and nutrient balance and the identification of risk zones (such as for nitrate leaching), b) a meso-scale (1:10,000 to 1:25,000) for detailed modelling of water, material and energy fluxes (qualitative and quantitative information within the risk zones), and c) a detailed level for measurement planning and efficiency control (micro-scale, 1:1,000 to 1:5,000). The study areas discussed here represent these three scale levels.

The relevant effects of water quality measures are represented by these different scale levels. The selection of the models and the assessment methods used to describe the ecological-economic situation were based on these scale considerations. A main objective of the model application is that they deliver valid results for each of these scale levels (Dalgaard et al., 2003). The transfer of information to the next higher or lower scale level is accomplished using hydrological indicators (see Section 3.1).

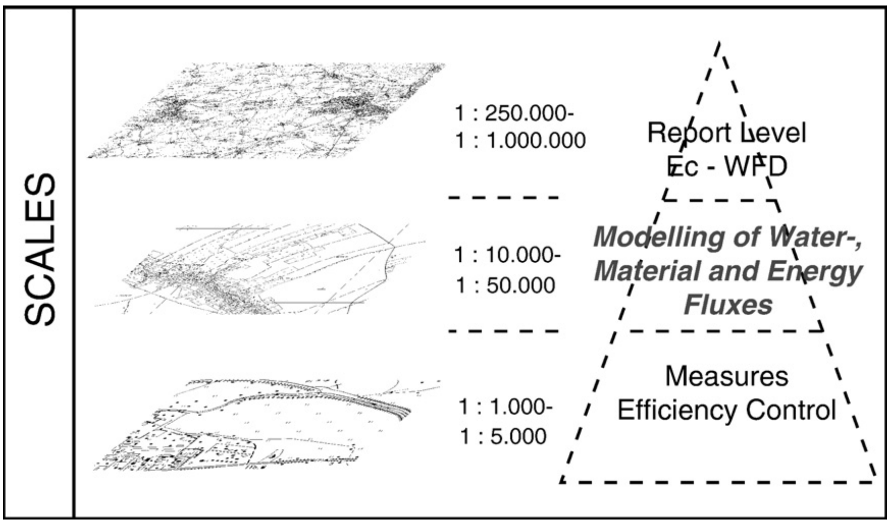

Fig. 3 – Scale-levels for the description of the water and nutrient balance within the project FLUMAGIS.

2.3. Study areas

The analysis was carried out for the Upper Ems River basin in northwestern Germany, which covers an area of 3740 km^2 (see Fig. 4). The river basin represents the macro-scale level in this FLUMAGIS application. The hydrological processes in the basin are influenced by increasing precipitation amounts from the Northwest and Central basin (700 mm/year) and the Southeast (1200 mm/year). The basin is a predominantly flat landscape with widespread permeable sandy soils. The River Ems has its sources at the foothills of the Teutoburger Wald mountains - with maximum altitudes of about 360 m above sea level —and flows through the North German Lowlands to the North Sea.

Detailed studies were carried out in three sub-basins. The selected sub-basins (section “Münstersche Aa”) cover areas between 160 and 350 km^2 and corresponds to the meso-scale. The Ems floodplain between Telgte and Greven (13.5 km^2) represents the study area for investigations with a high spatio-temporal resolution on the micro-scale level. Fig. 4 shows the location of the study areas and associated land use patterns.

The River Basin is situated in one of the most intensively used agricultural regions in Europe. Arable land covers approximately 77% of the area (the average in Germany is 50%), which has led to a dramatic loss of landscape diversity. The proportions of the other land use types are 9.9% for forest, 8.9% for urban areas, 3.9% for pasture and 0.2% for other areas. Intensive livestock production has caused severe environmental problems, as evidenced by the exceedance of the suggested WFD 3 mg/l nitrogen concentration limit in the river by a factor three to four for some Ems River gauges (see also Section 4.1).

3. Models and assessment methods

3.1. Hydrology and water quality

Computer-based modelling systems are being increasingly used for the investigation of land use changes on water quantity and quality in river basins of different sizes. Examples of such models include HSPF (Bicknell et al., 2001), AGNPS (Young et al., 1987), MIKE-SHE (Refsgard, 1997), SWAT (Arnold et al., 1998; Neitsch et al., 2002; Gassman et al., 2007), and SWIM (Krysanova et al., 1998). Overviews of different eco-hydrological models are given in Volk and Steinhardt (2001), Krysanova and Haberlandt (2002), Horn et al. (2004), and Arnold and Fohrer (2005). In spite of various existing models, questions still persist about a) their scale-specific applicability due to different systems dynamics, and b) their efficient application in environmental planning, which are both especially true for larger scales (Blöschl and Sivapalan, 1995; Krysanova et al., 1996; Wooldridge and Kalma, 2001; Steinhardt and Volk, 2003; Fohrer et al., 2005). To overcome these issues, Rekolainen et al. (2003) developed a conceptual framework for identifying the need and role of models in the implementation of the WFD. They state that the framework provides a basis to assure that proper tools will be available and selected for different purposes within the implementation process. The need for such work is confirmed by projects such as “Integrated Catchment Water Modelling (CatchMod)” that has been established by the European Commission (EC, 2005). They advocate the development of harmonised modelling tools and methodologies for the integrated management of water at river basin and sub-basin scales. For

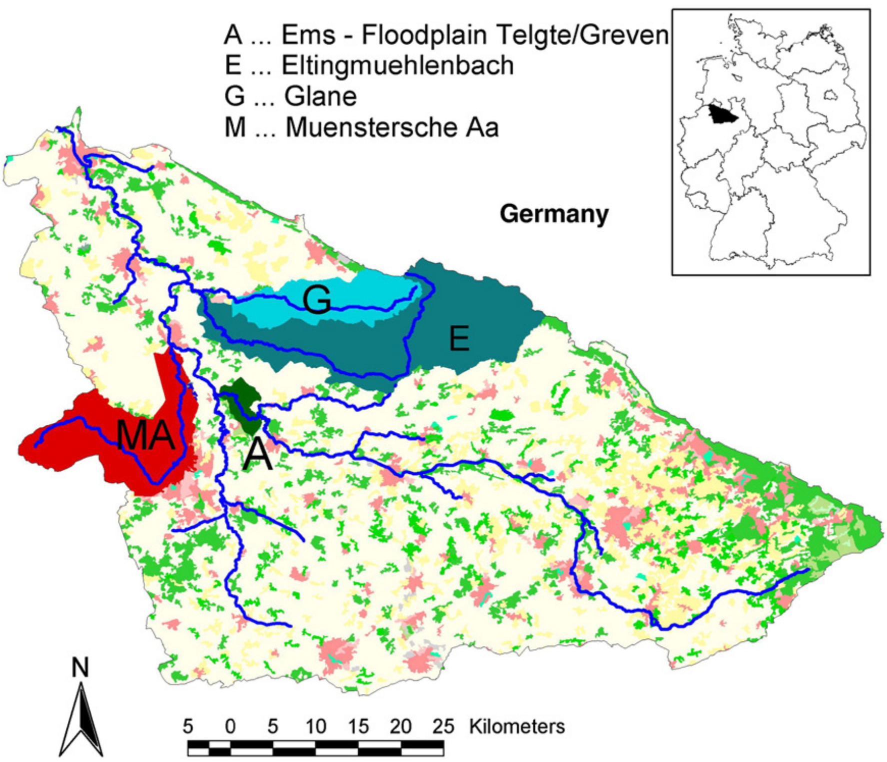

Fig. 4 – Location of the study area in Germany and land use pattern of the region. Agriculture is the dominating land use type.

Table 2 – Matrices for indicator use in different water balance models depending on time-steps of rainfall input data (example)

Indicator	NASIM (6 min) (micro- to meso-scale)	SWAT (daily data) (meso- to macro-scale)	ABIMO (annual data) (macro-scale)
Mean discharge (MQ)	x	x	x
Monthly mean discharge (MoMQ)	x	x	–
Mean discharge for summer and winter period (MQ summer/winter)	x	x	–
High water discharge value in period × years (HQx [x = 1; 2; 10; 50; 100])	x	–	–
High water discharge (HQ)	x	–	–
Mean high water discharge (MHQ)	x	–	–
Mean low water discharge (MNQ)	x	–	–

The table shows the results of an analysis of how well the models represent the different scale-relevant indicators. The crosses indicate that the model is suitable to represent the indicator.

medium to large-sized river basins various studies have shown that SWAT is an appropriate tool for simulating the impact of land use on water quantity and quality (Van Griensven, 2002; Arnold and Fohrer, 2005; Fohrer et al., 2005; Gassman et al., 2007). This is because SWAT covers all procedures from data pre-processing to hydrological and nutrient modelling in one package with several water use and crop management simulation options. Keeping in mind these methodological developments, we selected the following simulation models as especially suitable for our project: NASIM 3.10 (Hydrotec, 2001), SWAT 2000 (Neitsch et al., 2002) and ABIMO 2.1 (Glugla and Fürtig, 1997). The selection was determined by the scale-specific requirements of the project, the scale-specific applicability of the models as proven by intense examinations using artificial catchments (Volk and Schmidt, 2004), data availability, and the experiences of the project team with these models. The scale-specific suitability (Table 2) and transition between the scale levels (Fig. 5) was tested using a selection of hydrological indicators calculated by the models (e.g. Indicators of Hydrological Alteration (IHA) as proposed by Richter et al. (1996)) at the gauged basin outlets while taking variability and uncertainty into consideration. The hydrological gauges - where the discharge is measured and the indicators are simulated —serve as "calculation knots" (Fig. 5). Mean discharge ("MQ", Table 2), for instance, represents an indicator "Type A" as shown in Fig. 5. It can be derived from low resolution data and is thus suitable for all scales. Twenty-five scenarios were simulated over a period of 24 years. The simulation with the highest temporal resolution (NASIM) was used as the reference simulation. The calculated indicators then underwent a semi-quantitative evaluation and according to the deviations were subdivided into four classes: ++ = very good (0–10% deviation), + = good (>10-20% deviation), o = suitable (>20–40% deviation), and – = unusable (>40% deviation). These deviations form the basis of the evaluations.

3.2. Ecological asessment

The WFD obliges the EC member states to monitor the ecological status of surface waters using biological communities. The directive focuses on the ability of biological communities to evaluate human disturbance effects over time in correspondence with disturbance types. Ecological river quality is measured against an almost natural condition. For the FLUMAGIS river assessment modules we employ approaches based on macro-zoobenthic community, macrophytes, and a typological classification of watercourses. A floodplain assessment module was added, since the ecological

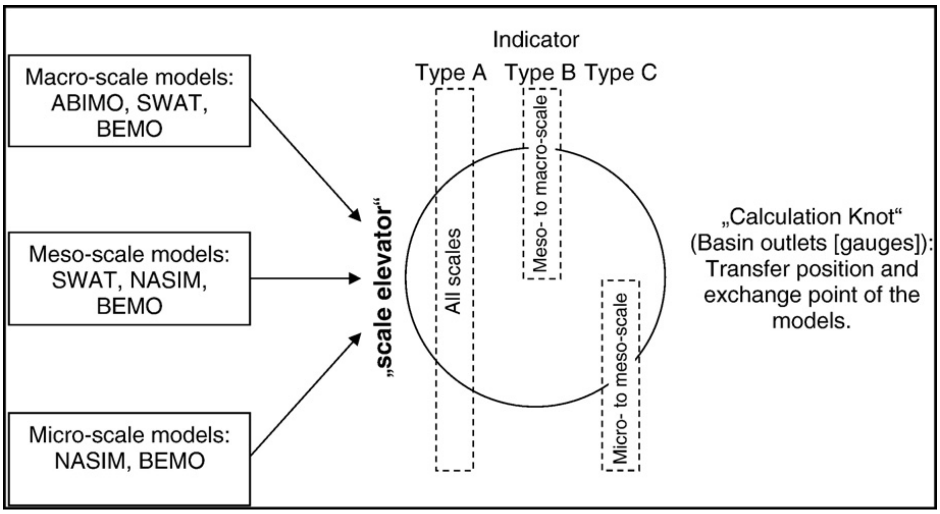

Fig. 5 – Scale-specific application of economic and hydrological models.

status of floodplains has a major impact on the ecological river quality (Lutosch et al., 2002; Bohn, 2004; Korn et al., 2005).

We focus here on the assessment of the current ecological status of the floodplain, which can be measured by a floristic-phytosociologic evaluation approach on the biotope level (micro-scale). It is based on the guide for the evaluation of Flora–Fauna–Habitat types (FFH) and the so-called Section 62 biotope types in the State of North Rhine Westphalia (Verbücheln et al., in press; MUNLV, 2004), which was aligned to the methodological recommendations in Annex V of the WFD. The evaluation approach takes into account especially the identification, assessment and scaling of nutrient-based, hydrologic and morphological pressures in the floodplain. These pressures may have a significant negative impact on the achievement of the good (ecological) status of the a) surface and groundwater bodies, b) groundwater-dependent terrestrial and directly water-dependent biotope types, and c) habitat (biotope) types according to the Flora-Fauna-Habitat-Directive (Council Directive 92/43/EEC). For the identification of the biotope types the analysis of the scenic character was used (Heidt et al., 1997). Based on this inference of regional site-specific a-biotic and biotic characteristics and parameters the selection of the relevant biotope types was carried out using the "overall pool" of biotopes existing in the Ems River floodplain. Subsequently, five ecological state classes were established for the biotope types as required by the WFD using the above mentioned evaluation criteria given by Verbücheln et al. (in press). The reference conditions for the best ecological state were inferred by a combined application of the concepts of potential natural vegetation (Kowarik, 1987) and orientation by nature (cf. Kowarik, 1999) and the examination of historic data. As a result, standardised assessment schemes for the biotope types are available.

Starting from the assessment schemes, a detailed deficiency analysis can be conducted by comparing the current state with the good ecological status. Based on this analysis, spatially explicit environmental conservation and development targets as well as suitable measures for their realization can be defined. The simulation of potential ecologic and economic consequences resulting from the implementation of these measures can be accomplished through scenario analysis. With these scenarios, the expected ecological impacts on floodplains can be evaluated in the form of biotope type changes taking into account that these biotope types are based on a-biotic and biotic characteristics and human use (Ssymank et al., 1993). For each biotope a comprehensive description was given and using the mentioned evaluation criteria based on literature survey and expert knowledge. Subsequently, this information was used to devise rules and algorithms for the prognosis of measure-induced potential biotope type (land use) changes. An example of such a scenario was elaborated in the FLUMAGIS micro-scale case study "floodplain development" (see Section 4.3).

3.3. *Socio-economic analysis*

Different management options for achieving certain environmental goals are associated with different costs. In addition to the degree of effectiveness, the severity of potential conflicts among stakeholders may vary. Therefore the socio-economic analysis assesses not only costs — as a part of a cost-efficiency analysis. It also carries out an actor network analysis to obtain information about the potential social consequences of the WFD implementation.

The agricultural-economic model BEMO is a representative farm model, which has been applied to estimate the economic effects of alternative agricultural management options. Originally developed to assess the potential economic consequences of changes in national and EU agricultural policies (Kleinhanss et al., 1999), it was modified and extended to environmental policy measures (Hirschfeld, 2006). BEMO is a partial, supply-oriented linear programming model based on representative individual farm data. For FLUMAGIS the model was calibrated using regional management data (yields, shares of different cultures, livestock numbers, prices, capacities) to represent the status quo in the year 2001. Based on the developed management scenarios, BEMO simulations characterize the potential economic effects on agriculture. The simulations of the WFD management scenarios reveal reductions in gross margins in comparison with the status quo situation. The differences are the economic costs of the management options. These costs give an indication of how severely farmers might be affected by policies intended to reduce diffuse emissions.

Sealed surfaces cause higher soil and nutrient runoff and have adverse hydrological effects on the river system. Therefore different measures to reduce the share of surface runoff contributed by sealed areas were proposed based on similar existing projects.

The most radical measure to improve the hydrological regime was the simulated afforestation of agricultural areas. The economic effects were calculated using average regional land prices (assuming landownership of the affected areas would go to the regional water authorities) and estimating the costs for planting small trees.

In addition to the costs the degree of potential conflicts and the acceptance of the proposed measures were assessed through a stakeholder analysis in the study area. Important stakeholders affected by the management options were identified as well as their goals, interests and perceptions. The analysis was based on the conceptional framework of a dynamic actor network analysis (DANA) for the identification of relevant actors, problem definition and description of conflicts (Bots et al., 1999).

The conflict potential connected with the proposed measures evaluated was examined using the following criteria: the nature of the conflict, the acceptance of measures, the regional relevance of the affected water uses, and the distribution of the costs. The nature of the conflicts was differentiated into 'soft' and 'hard' conflicts — with 'soft' characterising conflicts concerning differing perceptions regarding the decision-process or the chosen means and 'hard' conflicts meaning tangible monetary losses for certain actor groups. The relative relevance of the affected uses as well as the regional economic power were assessed using indicator ratios of regional employment, contribution to gross domestic product (GDP) and land use shares. Concerning the distribution of costs, a distinction was made whether the costs would be borne by individual farmers or public agencies. These four indicators were aggregated to obtain a conflict potential index on a six-level ordinal scale. This approach integrates potential social effects into a model-based decision support system (see also Hirschfeld et al., 2005; Dehnhardt et al., 2006). Hence, conflict potential and calculated

Table 3 – Examples of scale-specific environmental measures in the project

Scale level	Area	Measure
Macro-scale	Upper Ems River Basin	– Change arable land to pasture in areas with hydromorphic soils in floodplains – Limiting nitrogen surplus from fertilizers on arable land and pasture to a maximum of 50 kg/ha/year
Meso-scale	Subbasin Münstersche Aa	– Reduction of surface sealing – Afforestation
Micro-scale	Floodplain between Telgte and Greven	– Extensification, renaturation of floodplain areas (change arable land to rough grazing, floodplain forests, pasture to rough grazing, riparian buffer strips)

costs were integrated into the knowledge base as socio-economic attributes of the proposed management options.

4. Simulation results at different scale levels

The simulations in FLUMAGIS are based on case studies focusing on water quality, quantity and floodplain ecology problems in the Upper Ems basin within the context of the WFD. The objective of the case studies was to develop river basin management options, the simulation of their ecological and economic effects (cost-efficiency-ratios) and an assessment of potential conflicts with relevant stakeholder interests. A scale specific status quo land use scenario was furthermore developed for each case study. For five case studies approximately 50 management scenarios were developed. Examples of these management measures at different scale levels are presented in Table 3. The simulation results will be presented in the next sections.

4.1. *Macro-scale*

At the macro-scale, we examined the effects of current land use patterns on runoff dynamics and surface water quality. The deficit analysis on this scale identified nutrient input in the Ems River system as the central problem. The annual average of Total Inorganic Nitrogen (TIN) concentration at the basin outlet was around 6 mg/l, i.e. two times higher than the defined environmental target for the WFD, with some values exceeding the target value by up to four times.

To improve the ecological situation, two out of 40 possible measures were chosen for the simulation (see Table 3). The first management option (conversion of a considerable share of arable land into pasture land in areas with hydro-morphic soils in floodplains) resulted in a remarkable change in the land use situation. The share of arable land dropped from 77% to 64% of the total area. Moreover, it caused a reduction in fertilizer application and led to higher evapotranspiration rates. As a result, the simulated nitrogen concentrations in the Upper Ems decreased by 10% — accompanied by high economic costs of 31.6 million euro per year, i.e. between 500 and 800 euro per hectare per year, depending on regional soil qualities and management intensities. This measure is expected to result in hard conflicts with affected farmers.

The second management option investigated was fixing an upper limit of 50 kg per hectare per year for nitrogen surplus to be realized mainly by the reduction of mineral fertilizers in the SWAT model and an upper bound to TIN-surplus in the BEMO model. Nitrogen surplus is calculated by deducting nitrogen removal with harvested plants from the total amount of nitrogen applied to the agricultural area (differentiated into arable and pasture land) for average farms “created” with average regional data (district level). This second management option proved to be less costly (1.7 million euro per year), and is expected to be accompanied by less conflicts and yields a more pronounced reduction of nitrogen concentrations in the Ems river (reduction of about 17%).

A change from arable to pasture land is usually associated with income losses and leads to conflicts with affected farmers. Full compensation of the income losses (from public budgets) is expected to reduce the conflict potential in the region. Since land use changes may be associated with restrictions in farmer’s long-term development, voluntary acceptance of such management options nevertheless remains low.

4.2. *Meso-scale*

Focusing on the sub-catchment of the Münstersche Aa, which currently has a poor hydro-ecological status (rank “4” in the classification system with 5 ecological classes as described in Section 3.2; see Table 4), regional and local land use management options were examined again to improve the runoff conditions.

Increasing urbanisation and surface sealing cause severe environmental problems such as decreasing infiltration and groundwater recharge rates. The temporary input of high water amounts from sealed areas in river systems affects living conditions in small urban rivers. Several measures exist in

Table 4 – Dynamics of high water discharge (HQ)

	Status quo [m^3/s]	Situation after measure simulation [m^3/s]	Change [%]	Assessment status quo [rank]	Assessment after measure simulation [rank]
HQ1	1449	1235	−15	4	2
HQ2	1515	1210	−20	4	2
HQ5	1792	1423	−21	4	2
HQ10	1996	1580	−21	4	2
HQ50	2462	1937	−21	4	2
HQ100	2661	2091	−21	4	2

HQ1, etc. represents the flood recurrence interval (high water discharge value in period × years (HQx [x = 1; 2; 10; 50; 100]). Status quo and change of the high water discharge dynamics after the measure simulation is presented.

Table 5 – Costs of different sealing disconnection measures (Shaft infiltration = surface runoff from a defined area is collected to percolate to the subsurface; through infiltration = surface sealing with material that allows percolation and infiltration of water to the subsurface; removal of sealing = reduction of the total amount of sealed area to reach natural infiltration conditions)

Measure	Costs [€/m^2]	Area [ha]	Costs [€]
Shaft infiltration	8.99	190.2	17,101,302
Through infiltration	21.26	190.2	40,440,623
Infiltration total		380.4	57,541,925
Removal of sealing	51.13	380.4	194,481,664

urban water management to achieve better infiltration and higher groundwater recharge. Three of these measures to disconnect sealed surfaces and to remove sealing were simulated and assessed concerning their costs and hydro-ecological effects. The costs of the alternative measures are presented in Table 5.

The hydraulic modelling results show that the implementation of these measures will have nearly identical hydro-ecological effects on the river system. Shaft infiltration is thereby the most cost-effective alternative.

4.3. *Micro-scale*

In the case study "floodplain development", an evaluation of the current ecological state was accomplished for a representative floodplain area (267 ha) of the river basin. The assessment approach is described in Section 3.2. Due to intensive agricultural land use and the high degree of hydro-engineering extensions of the river and the adjacent riparian zone, most biotope types (predominantly intensively used arable land and grassland) of this floodplain area were assigned the ecologic status "dissatisfying" and "poor".

Based on the deficiency analysis suitable measures for the improvement of the current ecological situation were suggested and analysed concerning their socio-economic and ecologic impacts. As a result it was suggested to convert:

- 16.4 ha arable land (14.7% of the case study area) into extensive used grassland;
- 2.7 ha arable land (2.4% of the case study area) into floodplain forest;
- 2.5 ha grassland (5.4% of the case study area) from intensive to extensive utilisation; and
- establish along 1477 m of selected areas along the river bank of the Ems River riparian buffer strips (with an average width of 20 m), covering an area of 2.95 ha.

The cost assessment at micro-scale is based on an assessment of the land use structure using detailed GIS-maps. Seven representative 'average' farms were generated reflecting the observed land use structure defined by average size, livestock numbers and mineral fertilizer use calculated from municipal and regional statistics. These data sets were used to optimise the respective production programmes under the different management scenarios. Table 6 shows the results of the model runs. The simulated costs lie between 20,000 and 25,000 euro per year.

5. Conclusions

A SDSS approach was presented in this study that was based on the integration of GIS based visualization techniques, scale-specific modelling, and knowledge processing methods for ecological and socio-economic assessment of water management planning measures. Management options were determined based on a deficit analysis, referring to the environmental goals of the WFD. The visualization tool proved to be an especially valuable approach to explain both system interrelationships within the basin and the impacts of management options to the various stakeholders involved. This is considered an important contribution to the required improvement of stakeholder participation in the WFD (Van der Helm, 2003).

The focus on multiple scales is also considered of paramount importance to ensure a sound assessment for the selection and implementation of the necessary measures to achieve the WFD objectives. In the case studies presented in this paper, potential measures have been defined at three different scale levels and ranked as a function of their cost-efficiency:

At the macro-scale, two alternatives were evaluated: a) a reduction of arable land, and b) setting the tolerated nitrogen surplus on an average farm to an upper limit of 50 kg per ha per year. The latter appeared to be by far the more cost-efficient alternative. While the conversion of arable land into grassland would cost around 32 million euro per year for the entire study area (3740 km^2), limiting nitrogen surpluses could be implemented

Table 6 – Costs of the "floodplain development measures" scenario (micro-scale)

Measure	Total gross margins	Per hectare of affected area	Total costs in the case study region (267 ha)
	[%]	[€/year]	[€/year]
Conversion of intensive arable into extensive grassland (including a reduction of livestock numbers)	−5.0	823	13,497
Land abandonment (without reducing livestock numbers)	−2.4	772	6521
Extensification of grassland	−0.04	41	103
Combined strategy	−7.6	627	20,603
Plus reduction of livestock numbers			
Abandonment + livestock reduction	−4.0	1288	10,871
All measures incl. livestock reduction	−9.2	759	24,851

for less than 2 million euro per year — leading to similar or even stronger reductions of nutrient loads in the river Ems.

At the meso-scale (area sizes between 160 and 350 km^2), we focused on potential measures to improve the current runoff conditions that result in frequent high flood events. We examined the effects of a measure that disconnects 30% of the sealed surfaces in the sub-basin study area from the river system. Runoff from the sealed areas is instead redirected into an infiltration system. An alternative measure would have been to remove the sealing for 30% of the sealed area. Shaft infiltration proved to be by far the more cost-effective of these measures, resulting in similar positive hydrological effects such as mitigation of peak flow and increasing infiltration (see Table 4), which positively affects also the living conditions (habitats) in small urban rivers.

The micro-scale case study “floodplain development” was carried out using seven ‘average’ farms. These farms were generated as a function of observed land use structures defined by average size, livestock numbers and mineral fertilizer use. The agricultural-economic model simulations showed costs ranging between 20,000 and 24,000 euro for the different land use change management options simulated for the case study area of 264 ha.

The complexity underlying the FLUMAGIS approach represented a challenge for data management. Because FLUMAGIS is developed as a SDSS for WFD river basin management, it needs to be able to integrate publicly available (government) data. Therefore, we examined the availability of geo-referenced (spatial) data (e.g. climate, soil, land use, water supply and waste water treatment, agricultural management) for river basin management at different scales. The entire data set was integrated within a GIS ‘space’ with spatial reference to the entire basin and the investigated sub-basins. The data sets for the defined spatial units (municipality, sub-basin, river basin) served as input information for the economic and ecological simulation models.

We encountered a number of problems in connection to the scale definition and the simulation requirements:

The development of homogeneous data sets at different scale levels is difficult and costly. The existing data sets of the respective federal states are compiled using different data management methods, which results in possible incompatibilities and hence errors when the data are put together and processed for large river catchment applications.

Most of the economic farm data are available only at aggregated levels (municipalities, counties, federal states) due to confidentiality laws. It is therefore nearly impossible to assess management strategy effects on micro-scale economic and ecological conditions.

There is a lack of long-term water quality time series data on a daily basis and of high spatial resolution, which complicates simulation evaluations. Approximately 600 water quality gauges exist in our investigated river basin, but daily sampling data are not available for any of them.

The inclusion of the nutrient balance model into the model system appeared to be crucial. Two additional challenges were faced here. First, complex water and nutrient simulation models are expert systems. Their integration into a planning tool is subject to certain restrictions such as calculation time and user knowledge. Second, there is often a mismatch between current water quality monitoring data and the required quantification of matter cycle processes in river basins and corresponding integrated river basin modelling. These difficulties need to be overcome for future WFD implementations.

One crucial aspect to secure a successful implementation of WFD management options is to convince the stakeholders in the river basin that a good ecological status is a goal worth achieving. The FLUMAGIS decision support system supplies the user with information on current deficits, possible solutions and costs, conflict potential and ecological effects of potential measures. The visualisation of model results and land use changes provides the ability to literally see and “feel” what a good ecological status means in a specific spatial context. It therefore provides a valuable tool for transparent and conflict-reducing participation and planning processes.

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A3. STATUTORY DECLARATION

Eidesstattliche Erklärung / Statutory Declaration

Hiermit erkläre ich an Eides statt, dass ich die vorliegende Habilitationsschrift mit dem Titel „Scale appropriate analysis, assessment and management of landscape water and matter dynamics“ selbständig und ohne fremde Hilfe verfasst und keine anderen als die angegebenen Hilfsmittel und Quellen benutzt habe. Die den benutzten Werken wörtlich oder inhaltlich entnommenen Stellen wurden als solche kenntlich gemacht.

I herewith formally declare that I have written the submitted habilitation thesis with the title „Scale appropriate analysis, assessment and management of landscape water and matter dynamics“ independently. I did not use any outside support except for the quoted literature and other sources mentioned in the thesis.
I clearly marked and separately listed all of the literature and all of the other sources which I employed when producing this academic work, either literally or in content.

Dr. Martin Volk, 29. Oktober 2008.

A4. Curriculum vitae

Curriculum Vitae

Name:	Dr. rer. nat. Martin Volk
Born at:	July 11, 1964, in Hausen, Kreis Gießen (Hessen)
Family status:	Married, 1 Child
Nationality:	German

Education

1984
- General Certificate of Education at Ricarda-Huch-Schule in Gießen, Germany

1985 – 1990
- Studied Physical Geography at the Justus-Liebig-University in Gießen (received diploma degree 1990).
- Student placements at the University of Calgary, Canada (1988), and ETH Zurich, Switzerland (1989).
- Participation at the Geoscientific Expedition to Spitsbergen (Svalbard, Norway), SPE ´90 (1990) (supported by the German Research Foundation DFG).

1990 – 1994
- Received doctoral degree (Dr. rer. nat.) at the Justus-Liebig-University of Gießen.
- Participation at scientific projects in the Swiss Alps (Wallis, 1989 – 1992)
- Participation at the Geoscientific Expedition to Spitsbergen (Svalbard, Norway) SPE ´91 (1991) (supported by the DFG).
- Worked for an Engineering Company (Hydraulics, environmental geology, soil mechanics) in Limburg (State of Hessen, Germany) (1992)
- Stayed in South East China to evaluate German Environmental Projects (Climate and Erosion Research) of the Volkswagen Foundation and the Max-Planck-Society (1993)
- Research Scientist with Prof. Dr. L. King (1993)
- Received an Award for an outstanding study from the Association for the Promotion of Science and Humanities in Germany

Carreer

1993-1995
- Employee with an Engineering office (Hydraulics, environmental geology, soil mechanics) in Bad Dürrenberg (State of Saxony-Anhalt, Germany)

1995
- Worked freelance (geotechnical, hydrological and survey consulting)

1995 – 1999
- Research scientist at the Department of Applied Landscape Ecology of the Centre for Environmental Research (UFZ) (terminable position).

2000
- Senior scientist at the UFZ (permanent position). Leader and participant of several projects dealing with river basin and natural resources management on national and European level (BMBF, EU, INTERREG) (2000 to 2008).

2001
- Research placement with the USDA-ARS in Temple, Texas, USA (supported by the German Research Foundation DFG).

2003-2004
- Sabbatical at the Texas A & M University and the USDA-ARS in Temple, Texas, USA (research visitor stipend of the USDA).

2004
- Deputy Director of the Department Computational Landscape Ecology of the UFZ
- Deputy head of the research cluster “River Basin Management”
- Head of the working group “Abiotics”

2005 - 2008
- Head of the Working group “Integrated modelling, remote sensing and data assimilation”
- Leader of the project “Quantification of Ecosystem Services and Trade Offs”

Dr. Martin Volk, 29. October 2008